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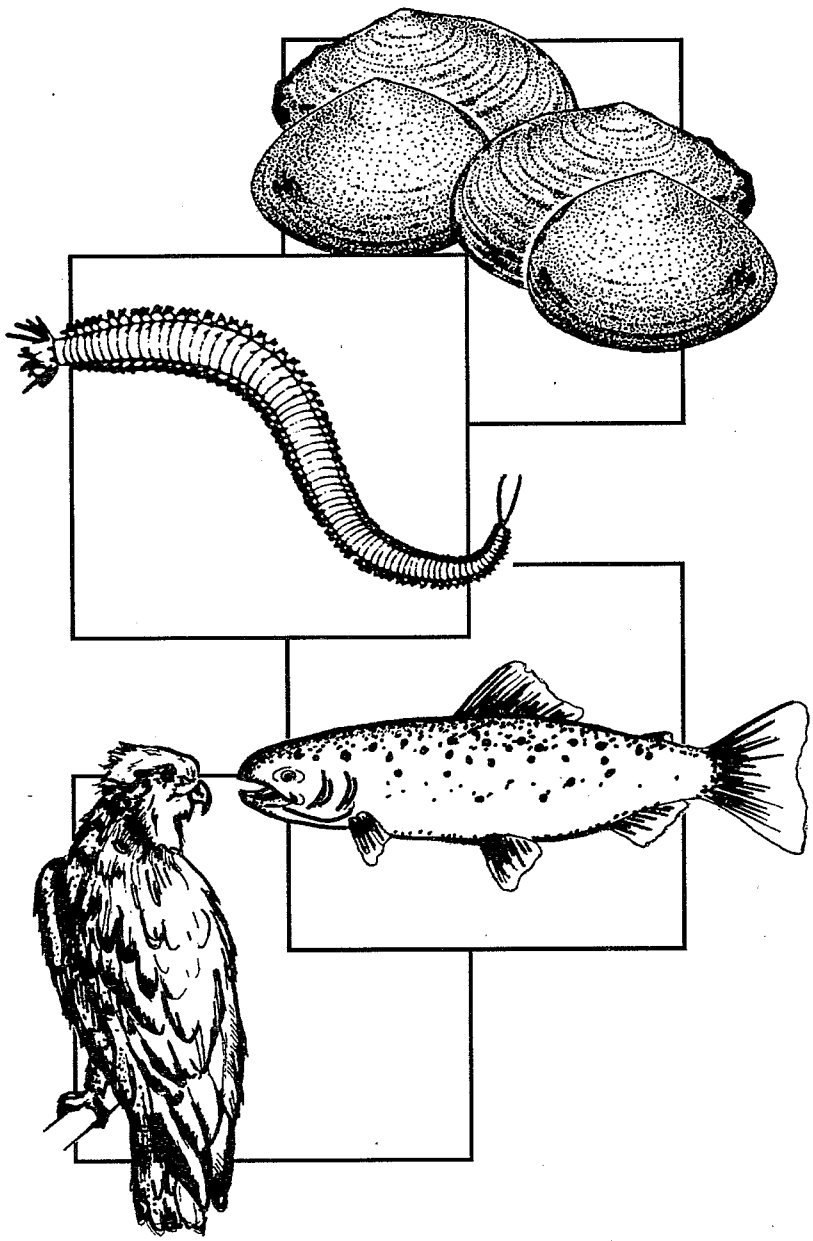
National Sediment Bioaccumulation Conference

Proceedings



Proceedings

National Sediment Bioaccumulation Conference

September 11-13, 1996
Bethesda, Maryland

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Abstract

During September 11-13, 1996, the U.S. Environmental Protection Agency (EPA) sponsored the National Sediment Bioaccumulation Conference in Bethesda, Maryland. The assessment of bioaccumulative sediment contaminants has become an important issue for EPA and other agencies. Data compiled to date for EPA's National Sediment Quality Survey and the National Listing of Fish and Wildlife Consumption Advisories indicate that the presence of bioaccumulative substances in sediments is a potentially serious national problem. EPA uses bioaccumulation data to make regulatory decisions in a number of its programs. The Superfund Program uses bioaccumulation data to assess contaminated sites for cleanup. Results of bioaccumulation tests are used in the assessment of new and existing industrial chemicals under the Toxic Substances Control Act (TSCA) and in the review process for National Pollutant Discharge Elimination System (NPDES) permits and dredged material discharge permits. Bioaccumulation studies are also required to support registration of pesticides under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA). Advances in the science of bioaccumulation testing and assessment and interpretation of test results will benefit all of these programs.

The purpose of this conference was to present the current state of our knowledge of assessment of bioaccumulative sediment contaminants and to discuss how bioaccumulation data are integrated into EPA's decision-making processes. The conference was organized into the following seven sessions:

Session One	Field and Laboratory Methods for Measuring Bioaccumulation
Session Two	Interpretation and Applications of Bioaccumulation Results
Session Three	Modeling Bioavailability of Sediment Contaminants
Session Four	Food Chain Models and Bioenergetics
Session Five	Human Health-Based Risk Assessment
Session Six	Ecological-Based Risk Assessment
Session Seven	Integrating Bioaccumulation Results into EPA's Decision-Making Process

Each session consisted of individual presentations and a discussion period with questions and comments from the audience and responses by the speakers. The Proceedings document contains a paper and graphics summarizing each speaker's presentation and a summary of audience questions and panel responses during the discussion period. It also contains the conference agenda and additional information developed for the conference.

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The National Sediment Bioaccumulation Conference was funded jointly by two offices in the U.S. Environmental Protection Agency, the Office of Science and Technology (in the Office of Water) and the Office of Research and Development (ORD). The Standards and Applied Science Division in the Office of Science and Technology (OST) organized the conference. Tetra Tech, Inc. provided logistical support for the conference and production support for the proceedings under EPA contract number 68-C3-0374.

A planning workgroup consisting of representatives from OST, ORD, Great Lakes National Program Office (GLNPO), and EPA Region 5 participated in developing the agenda and identifying speakers for the conference. The workgroup members were: Tom Armitage, EPA OST; Ed Barth, EPA ORD; Mike Kravitz, EPA OST; Amy Pelka, EPA Region 5; Rich Pruell, EPA ORD; Lara Pullen, EPA Region 5; Dave Redford, EPA OWOW; Leanne Stahl, EPA OST; Marc Tuchman, GLNPO; Howard Zar, EPA Region 5; and Chris Zarba, EPA ORD. The contributions of these individuals in planning the conference are greatly appreciated.

Finally, the contributions of the invited speakers and moderators are gratefully acknowledged. Their efforts were critical to the success of the conference and production of the proceedings.

Welcome and Introduction

Dr. Elizabeth Southerland
Acting Director, Standards and Applied Science Division
Office of Science and Technology
U.S. Environmental Protection Agency

Dr. Thomas Armitage
Acting Chief, Risk Assessment and Management Branch
Office of Science and Technology
U.S. Environmental Protection Agency

Dr. Southerland

Dr. Southerland welcomed the participants and turned the podium over to Dr. Thomas Armitage to outline the agenda for the conference.

Dr. Armitage

I would like to take this opportunity to welcome you to the National Sediment Bioaccumulation Conference sponsored jointly by the Office of Water and the Office of Research and Development. I would also like to acknowledge the assistance we received in planning the conference from the Great Lakes National Program Office and EPA Region 5.

We are very pleased to finally be holding this conference. Planning for this conference began over a year ago. We persevered through the federal budget crisis and government shutdowns, and are delighted to finally have all of you here to participate. The response to our announcement of this bioaccumulation conference was overwhelming. When we started planning this conference, we had no idea that so many people would be interested in attending. Over 400 people are here today, which clearly indicates that assessment of bioaccumulative sediment contaminants is an issue of major interest and importance in the scientific and regulatory community.

For the next two and one-half days, we will be hearing from a number of distinguished experts in the field of assessment of bioaccumulative sediment contaminants. We have organized the conference into several sessions. Each session will be chaired by a moderator who will lead a discussion and question and answer session following the presentations. We encourage audience participation during the discussion period and have provided microphones in the aisles for your comments and questions. During the first two days of the conference, there will be sessions covering field and laboratory methods for measuring bioaccumulation, interpreting and applying bioaccumulation results, modeling bioavailability of sediment contaminants, and assessing human health and ecological risks associated with bioaccumulative contaminants. On the final day of the conference, an EPA program panel will address how results of bioaccumulation assessments are being integrated into EPA's decision-making process.

The assessment of bioaccumulative contaminants in sediment has become an important issue for EPA and other Federal agencies. From our work to date on the National Sediment Quality Survey and the Listing of Fish and Wildlife Advisories, it is becoming increasingly apparent that the presence of bioaccumulative substances in sediments is a potentially serious widespread national problem. EPA evaluated more than 21,000 sampling stations nationwide to develop the National Sediment Quality Survey. Of the sampling stations evaluated, 26 percent were classified as Tier I and 49 percent were classified as Tier 2. Tier I is a category indicating that associated adverse effects are probable, and Tier II is a category indicating that associated adverse effects are possible, but expected infrequently. The classes of bioaccumulative contaminants most frequently associated with contaminated sediment sites include metals, polychlorinated biphenyls (PCBs), organochlorine pesticides, and polynuclear aromatic hydrocarbons (PAHs). The 1996 update of the Listing of Fish and Wildlife Advisories includes all available information describing state-, tribal-, and federally-issued fish consumption advisories for the 50 states, the District of Columbia, four U.S. territories, and several Native American tribes. It has been expanded to also include fish advisories issued for the 12 Canadian provinces and territories. The total number of advisories in the United States increased by 26 percent from 1995 to 1996. Advisories increased for four major contaminants, including mercury, PCBs, chlordane, and DDT. More monitoring accounts for part of the increase in the number of advisories, but it also indicates a greater awareness of the problem and underscores the

need for increased monitoring and reliable techniques for assessing the risks of exposure to these chemicals.

EPA uses information and data on bioaccumulation to support program responsibilities in a number of its regulatory programs. The Office of Pesticide Programs requires bioaccumulation studies for the registration of pesticides under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA). The Office of Pollution Prevention and Toxics uses bioaccumulation data to assess new and existing industrial chemicals under the Toxic Substances Control Act (TSCA). The Superfund Program in the Office of Solid Waste and Emergency Response uses bioaccumulation data to assess contaminated sites for cleanup. The Office of Water incorporates results of bioaccumulation tests in the review process for dredged material disposal permits and National Pollution Discharge Elimination System (NPDES) permits. Advancing the state of our knowledge in bioaccumulation testing and assessment will benefit all of these programs.

A number of important unresolved issues remain for bioaccumulation assessment, including uncertainties in test procedures and limited understanding of the relationships between contaminant body burdens and adverse ecological effects. I would like to again acknowledge the Office of Research and Development (ORD) cosponsorship of this conference and stress that we are looking to ORD to maintain leadership in answering some of these questions.

We organized this conference to provide a forum to discuss the current state of our knowledge of assessment of bioaccumulative sediment contaminants and to examine how bioaccumulation data are integrated into EPA's decision-making processes. We will be publishing a conference proceedings and are currently working on a report summarizing the status of bioaccumulation testing and interpretation for the purpose of sediment quality assessment. This conference and our status report will help us to better understand the state of the science, to identify the data and knowledge gaps, to focus research efforts on providing answers to the most important questions, and to eventually develop consistent cross program guidance on interpretation of bioaccumulation data for use in EPA's regulatory programs.

Bioaccumulation Overview and Approaches

Gilman D. Veith
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Research Triangle Park, North Carolina

Contaminated Sediments: State of the Science and Future Research Directions

Gilman D. Veith
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It is a real pleasure to be here today and speak at the first National Sediment Bioaccumulation Conference. As many of you know, when Bob Huggett came into the Office of Research and Development (ORD) as the Assistant Administrator, he brought with him a strong belief that sediment contamination is one of the compelling research issues that we face as a nation. He still holds that belief. Bob is at an international meeting for the Agency, so I am presenting for him today on behalf of ORD. However, my comments reflect my own perspective.

This conference is exciting because it represents the culmination of many years of work related to sediments. It also demonstrates, better than many other examples in environmental research, how science can be applied in a partnership of researchers, risk managers, regulators, private interest groups, and agencies at all levels of government to accomplish risk reduction.

About ten years ago, Jim Falco, Director of Environmental Processes Research in EPA, asked that I meet with him and several members of the Office of Water. He wanted to discuss building the scientific framework necessary to establish the concept of sediment quality criteria. The existing methods then were highly empirical and often unreproducible, and they did not allow us to predict effects. We needed criteria to make sense of the field measurements we were making. He asked if the ORD laboratories could provide support for development of sediment quality criteria, and we agreed to get the laboratories involved.

Soon after we organized scientists familiar with sediment issues and began research, a Senator involved in revising the Clean Water Act visited our laboratory and wanted to know about sediment contamination and why it needed attention. After giving a technical presentation that included an explanation of bioaccumulation factors under steady-state conditions, he surprised us by asking "what is a sediment?" There is a communication challenge, in addition to the challenge of developing a workable regulatory strategy for contaminated sediments. In the last ten years, research has taught us how to begin making sense out of the data on sediment effects.

In addition to developing sediment toxicity tests, we have established methods to measure the bioaccumulation potential of sediment contaminants. When we published our first work on bioaccumulation, we chose the term "bioaccumulation potential" rather than "bioconcentration factor" because kinetics and thermodynamics really intersect in bioaccumulation testing. It is very difficult to predict the kinetics of uptake because it is largely experimentally controlled and most of the bioaccumulation methods are operationally defined. "Bioaccumulation potential" is primarily a thermodynamic term that separates chemicals that bioaccumulate from those that do not. If organic chemicals are unlikely to bioaccumulate, they would probably not even be in the sediments. The "bioaccumulation potential" represents the probability that a chemical is going to penetrate an ecosystem by dispersing from a source and moving up food chains to cause effects at levels we might not have anticipated.

Sediments were recognized as the final repository for contaminants as early as the 1970s. During that decade we were still struggling to define toxicity and to establish permitting and control programs. Today, the issue of reducing risks from sediments that have already been contaminated is an important problem that merits attention. Right now we are making good progress on work in dredged material management, food chain modeling, and development of tissue-based residue methodologies to address this problem. The academic community and agencies at all levels of government have efforts underway to describe the dynamics of food chains and to define how to use this information in a risk assessment. The Office of Water is spearheading one such effort by developing the bioaccumulation paper entitled *Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment: Status and Needs.* About 40 people from several EPA offices are collaborating to draft that report, which will include information on chemicals of concern and methods for assessing bioaccumulation, an Agency summary on bioaccumulation data collection and interpretation, and recommendations for further research.

Another sediment-related effort being conducted at an ORD laboratory is the development of toxicity identification evaluation (TIE) procedures to analyze complex chemical mixtures.

I want to summarize briefly how we formulate priorities for budgeting in ORD and how conferences like this can help. Surprisingly enough, when we went through the budget planning process, some participants expressed the belief that contaminated sediments are not a serious national problem, that sediments do not present a significant risk to the environment, and that sediments do not merit further research based on their negligible risks. At first I laughed because I did not think they were serious, but then I realized that they were serious. They made their case using some of the same data we were using to try to show that sediment contamination is a significant problem. Data from a spatially subsampled survey conducted under the Environmental Monitoring and Assessment Program (EMAP) in ORD covered estuaries in the Virginian, Louisianan, and Carolinian provinces. Looking at the benthic condition with some of the EMAP indicators, one could see that about 29 percent of the estuaries had degraded benthic condition.

That information alone, however, does not convince skeptics that sediment contamination is a significant problem. The skeptics viewed 29 percent of our estuaries as impacted to mean 71 percent were not.

I realized that we have not clearly answered the "So what?" questions with respect to ecological consequences. How much impact is too much? When are fisheries and the productivity of coastal waters affected? How much degradation could exist in the benthic systems to streams and lakes before there are ecological consequences and loss of integrity? We can say that there are 17,000 square kilometers in our estuaries where you can measure an effect in the benthos, compared to reference conditions. That is a large area. So what? Similarly in the Great Lakes, many have overlooked the fact that reproduction in major species of the food chain has been shut off since the 1940s due to chemical residues. EMAP combines measures of effect with stressors and has associated the degraded communities with sediment contaminants. For example, when you link some of the areas of degraded benthic condition with measures of observed toxicity, you will find that these are the sites where survival of benthic organisms in bioassays was below 80 percent. This evidence establishes an association strong enough to say that parts of the benthic community degradation are due to contaminated sediments. The question that remains is how we can establish a cause-effect relationship between the loss of integrity and the sediment residues over large areas.

The importance of these field studies in strengthening the risk assessment process for sediment certainly plays into the risk-based priority setting in ORD. The new strategic plan for ORD is based on the risk paradigm. We now evaluate a problem based on whether or not we think the science can reduce the uncertainty associated with estimating effects, exposure, the assessment capability, and risk management. There are some issues where ORD has concluded that we do not need to do more effects research, because further effects research would not contribute significantly to our understanding of the problem. Contaminated sediments is one issue where major uncertainties still exist in effects and exposure assessment. In the process of planning the budget according to risk-based priorities, we have consolidated the intramural research in ORD to focus on contaminated sediments by designating this area as a budget subcomponent. ORD has created a work force of about 45 full-time equivalents (FTEs), which represents an increase in the level of effort for sediment issues in the laboratories. We are revising the research strategy for contaminated sediments, and research recommendations that come out of this conference will be considered in developing the plan, particularly for projects directed at the Office of Water's needs.

I would also like to direct your attention to the extramural research program, known as the Science to Achieve Results (STAR) Program, that ORD initiated in 1995. STAR is a competitive research grants program being run by ORD's National Center for Environmental Research and Quality Assurance in Washington, D.C. This year the grants program included only about $2 million specifically for issues involving contaminated sediments. There were more than 40 proposals and 8 of these proposals received high rankings by the peer review panel. The small STAR budget for the sediment program could only fully fund four or five grants this year, but it is a start. We anticipate a modest increase in funding for contaminated sediment research grants in FY 1997.

We may have temporarily won the battle to convince planners that contaminated sediments are impacting a significant part of our resources. Through the ORD planning process, ORD has agreed to focus more resources on research related to contaminated sediments. In FY 1998 there may even be an increase of about $3 million of extramural money to support the laboratories in the ORD intramural research program. We need to maintain this focus and increase our effectiveness through partnerships with other interested EPA offices and federal agencies, including Office of Water, the Superfund Program, the Great Lakes National Program Office, the Chesapeake Bay Program, the National Oceanic and Atmospheric Administration (NOAA), the U.S. Army Corps of Engineers, and the U.S. Geological Survey. We also need to be refining approaches that are implementable at the state level and work with the states to achieve risk reduction. Today is a good time to rededicate ourselves to working as partners and to make the contaminated sediment issue the best example of how we can work together to solve the environmental problems.

The ORD research program will continue support for pollution prevention efforts. We will be addressing the issues remaining for sediment quality criteria, including data to support the guidelines and response to public comment on the technical basis for the criteria. Another area ORD will support is applying what we know about the sediment effects of chemicals to a cleanup strategy for a particular Superfund site. There are some excellent presentations coming up in this conference on how to apply the concept of criteria to restoration.

The management of risks associated with the disposal of dredged material will require a strong EPA

collaboration with the U.S. Army Corps of Engineers and the States in order to develop a workable process. ORD will support that effort. We have identified research needs such as tissue-based effects models to understand the bioavailability and pharmacokinetics that explain why residues bring about long-term damage to ecosystems. Food chain models are becoming ever more sophisticated, but they need further development. There is a critical need to develop population level assessment methods to link up the "So what?" questions of the declining biological diversity and loss of certain species. Assessment of the impact of complex chemical mixtures will be necessary to unravel the causative agents in the chemical mixtures that are often found in sediments.

Finally, I would like to emphasize the importance of the development and application of remediation methods. There has been much progress on remediation of contaminated sediments, with mitigation success at Superfund sites. However, the mitigation options are still too limited for larger areas. In terms of communication challenges, we need a breakthrough to instill hope in our political leaders that we can actually remediate contaminated sediments. We have to overcome what seems to be an overwhelming pessimism that we will ever reduce the risks posed by sediments except by waiting decades. The work that our Region 2 Office is doing on treating harbor sediments in the New York area is resulting in some of the most exciting new developments in the last five years. They appear to be setting the stage for the future in sediment remediation.

As we participate in this conference and prepare for the next decade of work to reduce the impacts of sediments in the environment, I encourage you to avoid letting our energies become fragmented by using data to support a personal or organizational interest. The danger I see in the sediment literature is the number of papers designed to support hypotheses rather than test hypotheses. There is a need for debate in risk management, certainty among uncertainties in our scientific understanding of sediment interactions which merit debate, but the objective application of the scientific method in gathering data to support these debates is crucial.

Session One: Field and Laboratory Methods for Measuring Bioaccumulation

Peter Chapman, Panel Moderator
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Methods for Assessing Sediment Bioaccumulation in Marine/Estuarine Benthic Organisms

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Methods for Assessing Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates

Peter Landrum
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Kinetic Models for Assessing Bioaccumulation

Methods for Assessing Sediment Bioaccumulation in Marine/Estuarine Benthic Organisms

Henry Lee II
U.S. Environmental Protection Agency, Office of Research and Development, Coastal Ecology Branch, Western Ecology Division, Newport, Oregon

Introduction

Organic pollutants can be divided into two general classes (see page 1-15). The first consists of compounds with high water solubility and low K_{ow} values, such as acetone, that do not tend to adsorb to particulates. Their major reservoir is the water column, where they are accumulated through bioconcentration. The second suite of organic pollutants are the low water solubility, high K_{ow} compounds, such as DDT, dieldrin, and polychlorinated biphenyls (PCBs), that readily partition to particulates. They accumulate in the sediment where they can be bioaccumulated by benthic organisms. From the benthos, these high K_{ow} compounds can be introduced into higher trophic levels through trophic transfer. Other presentations at this conference are addressing the trophic transport and the effects of these pollutants. This presentation will focus on methods to measure/predict the bioavailability of sediment-associated contaminants to marine/estuarine benthos, and the use of sediment bioaccumulation tests in particular.

When designing any test method, it is critical to assess how are the data are going to be used. Tissue residue data can be used in risk assessments in a number of ways, which are summarized in the figure on page 1-16. The column labeled "Steady-State?" refers to whether estimates of steady-state tissue residues are required to adequately address the particular component of a risk assessment. For the sediment bioaccumulation tests, the need for steady-state data defines how long the test needs to be conducted to assure that steady-state tissue residues have been approached. The column labeled "Max Residue?" refers to whether an upper limit estimate of the tissue residue is required to address the particular risk assessment component. That is, is it necessary to use a duration, species, or test method that tends to maximize uptake?

As suggested in the figure on page 1-16, residues approaching steady-state are required for quantitative ecological or human health risk assessments other than hazard identification and identifying specific uptake routes. Tissue residues substantially less than steady-state residues will underestimate both exposure and effects, and this error will propagate through the risk assessment (e.g., estimate of trophic transport). The need for the more stringent requirement of an upper-limit estimate of tissue residue depends upon the specific question. For example, amphipods metabolize PAHs to a greater extent than most bivalves. If the goal is to assess PAH exposure to amphipods, an amphipod would be a suitable test species. However, if the goal is to extrapolate PAH bioavailability to other species, an upper limit estimate would be more appropriate, and a bivalve species should be used.

Bioaccumulation Methods

There are a suite of methods available to assess or to predict bioaccumulation that will be discussed in various presentations during the conference (see listing on page 1-16). Two approaches provide direct measures of existing conditions: the field approach and the bioaccumulation test. Both approaches involve measuring tissue residues in either field-collected or laboratory-exposed organisms. These direct approaches have high ecological relevance but can be costly, and they have limited ability to predict tissue residues resulting from changes in sediment contamination (e.g., after a clean-up).

Sediment bioaccumulation models can serve as cost-effective "screens" to determine when direct measurements are required and as a method to predict tissue residues when direct measurements are not practical. The two general types of sediment bioaccumulation models are equilibrium-based and kinetic approaches (see Landrum et al., 1992). The equilibrium-based approaches assume steady-state conditions between the organism and the environment, whereas the kinetic approaches describe bioaccumulation as the net effect of rate processes. The two equilibrium models are bioaccumulation factors (BAF = tissue residue/sediment concentration) and the equilibrium partitioning model. The two basic types of

kinetic approaches are kinetic process models and bioenergetically based toxicokinetic models. Kinetic models can be more accurate than the steady-state models, but they require extensive data. A decision tree has been developed to guide the risk assessor in choosing which tests or models should be used to assess or predict bioaccumulation given the project goals (Boese and Lee, 1992) (see figure on page 1-17).

Equilibrium Partitioning Bioaccumulation Model

The equilibrium partitioning model is based on the theory that neutral organic pollutants partition between the lipid phase in the organism, sediment carbon, and interstitial water until equilibrium is obtained (see page 1-18). Assuming that organic carbon is the only sink for neutral organics in the sediment and that lipids are the only sink in the organism, the model becomes:

$$Ctss/L = (Cs/TOC) * BSAF$$

where:

Ctss = tissue concentration at steady-state (µg/g)
L = lipid content (g/g)
Cs = sediment concentration (µg/g)
TOC = total organic carbon in sediment (g/g)
BSAF = biota-sediment accumulation factor (g carbon/g lipid)

The biota-sediment accumulation factor (BSAF) has also been referred to as the "accumulation factor" (AF). BSAFs can be determined empirically for each pollutant from laboratory exposures or field surveys. If partitioning is not a function of lipid or carbon type, the value of the BSAF for a compound should not vary among sediments or species. However, data from bioaccumulation tests indicate that differences in the sediment can affect the BSAF values. Tests were run with spiked sediments using the deposit-feeding clam *Macoma nasuta,* one of the bioassay animals for marine and estuarine systems. Two sediment types were spiked with 13 PCB congeners and hexachlorobenzene (HCB) at concentrations of about 50 parts per billion for each congener or compound. Test results show differences in the BSAFs for the two sediment types (see page 1-18), so the equilibrium model did not completely account for sediment differences. The results also show dramatic differences in BSAF values among congeners. A possible kinetic explanation for these results are that PCB congeners with lower K_{ow} values undergo rapid degradation whereas PCB congeners with higher K_{ow} values have limitations moving across membrane surfaces which can result in low uptake. Nor does the equilibrium partitioning bioaccumulation model totally account for species differences. In a study of a DDT-contaminated site, we found that field-collected *Macoma nasuta* had tissue residues of total-DDT and dieldrin 7 to 9 times higher than filter-feeding bivalves (Lee et al., 1994, see Figure 15, page 1-22). It seems, then, that there is about a 2- to 10-fold uncertainty in BSAF values. Even with these uncertainties, BSAFs have utility as a screening tool and in extrapolating among species or sediments. Additionally, this uncertainty can be reduced by extrapolating among similar feeding and sediment types.

Sediment Bioaccumulation Test

A laboratory test is often the preferred method to evaluate a specific sediment and to generate BSAFs under controlled conditions. Although these tests had been conducted for over a decade, there was no standardized methodology. Scientists at the U.S. Environmental Protection Agency (EPA) in Newport, Oregon developed a sediment bioaccumulation test for marine and estuarine systems, published it in a guidance manual in 1989, and revised it in 1993 (Lee et al., 1993, see page 1-19). Since then, the EPA scientists have worked with Peter Landrum of NOAA to develop a guide for sediment bioaccumulation tests for marine and freshwater benthic invertebrates (ASTM, 1995). The bioaccumulation test includes six key procedures (see page 1-19): (1) 28-day exposure duration, (2) use of sediment-ingesting organisms, (3) no supplemental food added, (4) independent exposure of species, (5) recommended accuracy of 80 percent of steady-state tissue residues, and (6) use of long-term tests or toxicokinetic approaches for greater than 80 percent accuracy or for slowly accumulated compounds. Information used to support the recommended procedures is discussed below.

The recommendation for testing 28 days was based on a literature review of percentage of steady-state residue levels achieved in 10 days (the period then used for testing dredged materials) and 28 days. Results were available for a variety of compounds such as PCBs, dioxins, furans, PAHs (or PNAs), and metals (see page 1-20). These compounds generally achieved 80 percent of steady-state tissue residues within 28 days. We evaluated the adequacy of the 28-day duration in the experiment exposing *Macoma nasuta* to the sediments spiked with 13 PCBs and HCB (see discussion under Equilibrium Partitioning Bioaccumulation Model). The experiment was run for 120 days. The figure on page 1-20 shows the results for PCB congeners 153 and 209. Though bioaccumulated to different amounts, both PCB congeners approached or exceeded 80 percent of steady-state residue after 28 days, as did the other congeners.

Data from other studies indicate that a period of 28 days can be insufficient for bioaccumulation testing. The United Heckathorn Superfund site in San Francisco Bay is highly contaminated with DDT and dieldrin. A bioaccumulation test was conducted with *Macoma nasuta* for 90 days using sediment from the most contaminated site. Test results for total DDT (DDT, DDE, and DDD) are displayed on page 1-21. At 28 days, the tissue residues only reached about one-third of the steady-state residues. The graphs of the tissue residues for the three most abundant compounds show even worse results (see page 1-21).

These results raise the question of whether DDT and its metabolites are different than PCBs or whether the difference is due to field versus spiked sediments. It also raises the question about the adequacy of the 28-day test. There are practical limitations to consider in setting the length of any laboratory test. A proposal to resolve this problem is to maintain 28 days as the standard duration, but to multiply the 28-day residues by an "adjustment factor" which is the ratio of the steady-state residue to the 28-day residue. Adjustment factors would be developed through long-term lab studies. For the DDT compounds and dieldrin, these adjustment factors ranged from 1.7 to 10.8, with a value of 2.9 for total DDT. For compounds which accumulate rapidly, such as the low molecular weight polycyclic aromatic hydrocarbons (PAHs), the adjustment factor should approach 1.

Several criteria for organism selection for use in marine/estuarine bioaccumulation tests are listed on page 1-22. Of these, the requirement for using sediment-ingesting organisms to maximize uptake of sediment-associated contaminants is critical. The figure on page 1-15 (modified from Landrum, 1989) helps illustrate why. An organism can accumulate sediment-associated contaminants from the interstitial water or from ingested particles. Compounds with higher K_{ow} values will be associated mainly with the particulate phase, so particle ingestion will be the primary uptake route for these chemicals. A series of experiments conducted with hexachlorobenzene, a low solubility compound, showed that at least 70 percent of the uptake in *Macoma nasuta* was from ingested particles. Although it can be argued that if all the phases are in equilibrium the uptake phase does not matter, the 7- to 9-fold higher residues in the sediment-ingesting *Macoma nasuta* compared to filter-feeding bivalves at the United Heckathorn site (see page 1-22) clearly demonstrates the importance of sediment ingestion on bioaccumulation. Based on similar reasoning, supplemental feeding is not recommended as the addition of uncontaminated food could "short-circuit" the solid-phase uptake route and result in an erroneously low evaluation of bioavailability. The marine/estuarine environment contains a number of deposit-feeding animals (various bivalves and polychaetes) that meet the criteria, particularly with respect to providing sufficient biomass for chemical analysis at the end of the test. The 1993 guidance manual and the ASTM guide identify animals suitable for bioaccumulation testing in marine and estuarine systems.

With any laboratory test there is the question of whether it accurately predicts tissue residues in field organisms. At the United Heckathorn Superfund study site, we were able to compare tissue residues in laboratory-exposed *Macoma nasuta* and *Macoma* collected at several of the field sites. As mentioned above, the 28-day exposure underestimated steady-state of DDT and dieldrin, so it was necessary to use the adjustment factors from the 90-day test at station 1 to correct the 28-day residues from the other five sites. After adjustment, ratios of laboratory to field tissue residues for total DDT and dieldrin at each station (see graph on page 1-23) ranged from about 0.5 to 3. At least for this suite of high K_{ow} neutral organics, the standard test appears to predict field residues within 2- to 3-fold.

Research Needs

Areas requiring further study to advance the science related to sediment bioaccumulation assessment are summarized on page 1-23. They include: (1) interlaboratory round-robin testing; (2) field validation of the bioaccumulation test, particularly for PAHs and metals; (3) identification of local test species, especially for subtropical, subarctic, and oligohaline habitats; (4) "standardization" of lipid methods for derivation of BSAFs; (5) evaluation of effects of sediment storage and spiking on bioavailability; (6) refinement of the experimental design, including criteria for controls and references; and (7) evaluation of kinetic and physiological-based alternatives to the 28-day bioaccumulation test. Of these, perhaps the most troubling question is whether there is slower uptake or a lower bioavailability with field-contaminated sediments compared to spiked sediments, as suggested by the slow uptake rates for DDT compared to the rates from the spiked PCBs. Enhanced bioavailability of spiked sediment could potentially result in erroneously high BSAFs and toxicity.

The above research needs address how to conduct tests and their accuracy and precision. The overriding question, however, is: "What is the ecological significance of tissue residues?" This needs to be addressed at several scales. At the scale of individual benthos, the question is, "What are the effects of the accumulated toxicants on growth, fecundity, and survival?" Use of critical body residues, as are being developed for neutral narcotics, is a promising approach. However, in nearly all cases, the goal of marine/estuarine ecological risk management is to protect a resource or higher levels of biological organization (e.g., "ecological integrity"). To achieve this goal, we will need to develop the insights and methods to translate toxic effects on individuals into effects on populations and communities. Addressing these higher levels of biological organization will require assessments at larger spatial scales than the classical "end-of-the-pipe" evaluations, and will often require evaluation of contaminated sediment effects in the context of multiple stressors. This will be a critical challenge for the future.

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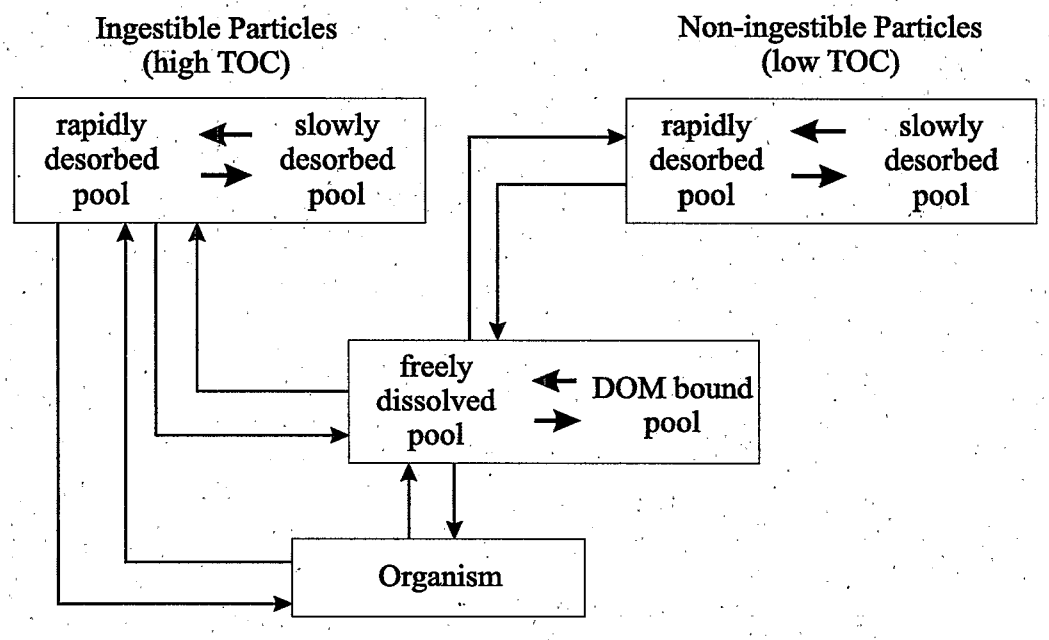
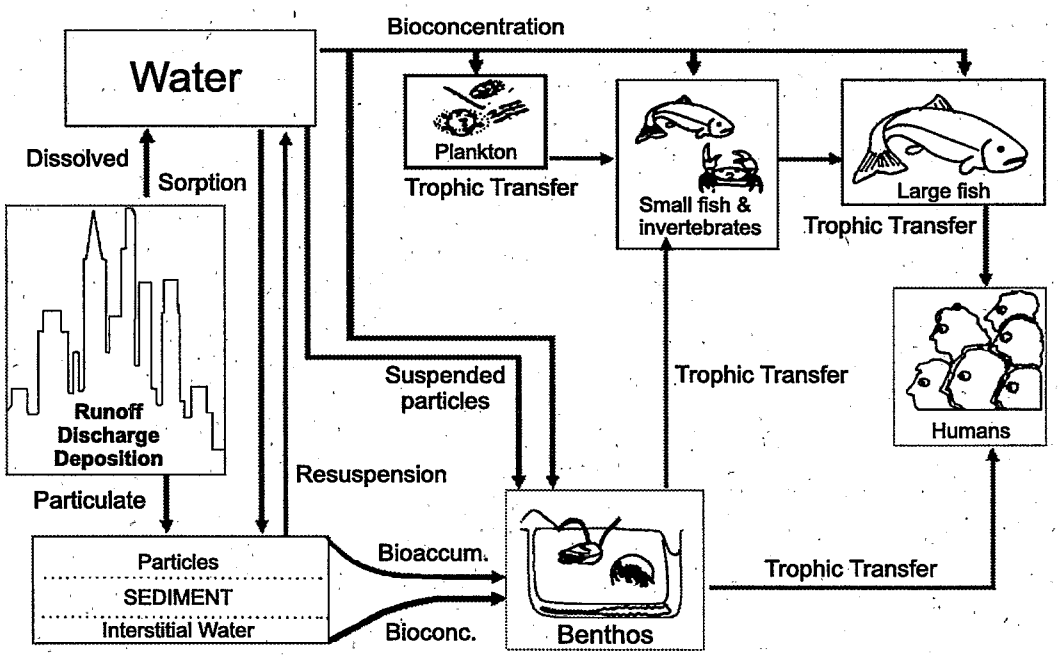
BIOAVAILABILITY OF SEDIMENT-ASSOCIATED POLLUTANTS
AQUEOUS AND PARTICULATE POOLS



Idealized Pollutant Pathways in Marine Ecosystems



– = High water solubility, low Kow, rapidly metabolized
– = Low water solubility, high Kow, slowly metabolized

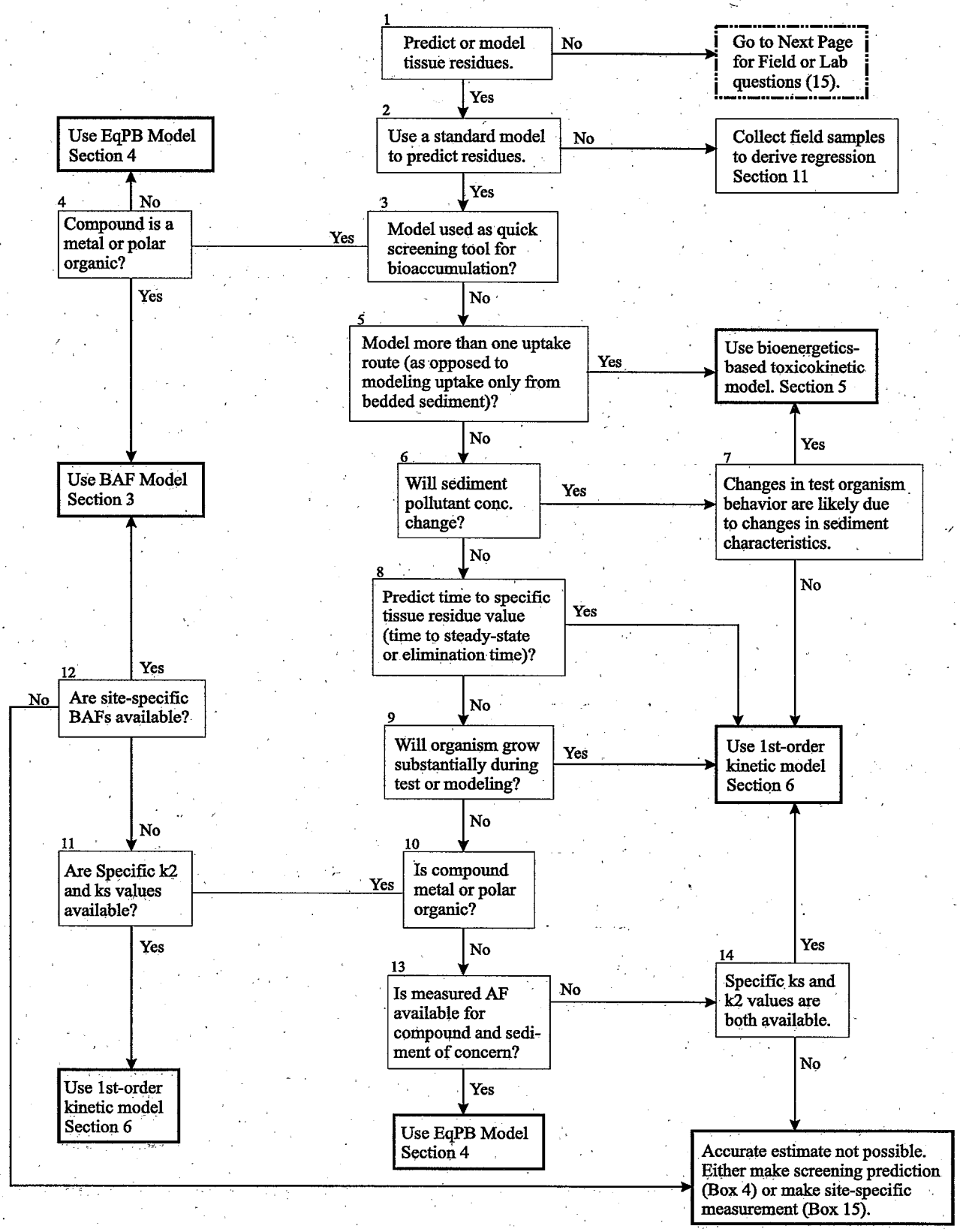
USE OF TISSUE RESIDUE DATA IN RISK ASSESSMENTS

	Steady-State?	"Max" Residue?
I. HAZARD IDENTIFICATION		
• Identify bioavailable compounds	No	No
• Quantitative measure of bioavailability	No	No
II. EXPOSURE ASSESSMENT		
• Quantify exposure to assessment endpoint species (e.g., edible clam)	Yes	No
• Test species is an indicator (measurement endpoint) for exposure to other species	Yes	Yes
• Test species is prey for higher trophic levels	Yes	Yes
III. ECOLOGICAL EFFECTS ASSESSMENT		
• Tissue residue effects on benthos	Yes	Yes/No
• Derive "Tissue Residue Criteria"	Yes	Yes
IV. HUMAN HEALTH EFFECTS ASSESSMENT	Yes	No
V. RESEARCH		
• Evaluate Sediment Quality Criteria and bioaccumulation models	Yes	Yes
• Determine importance of uptake routes	No	Yes/No

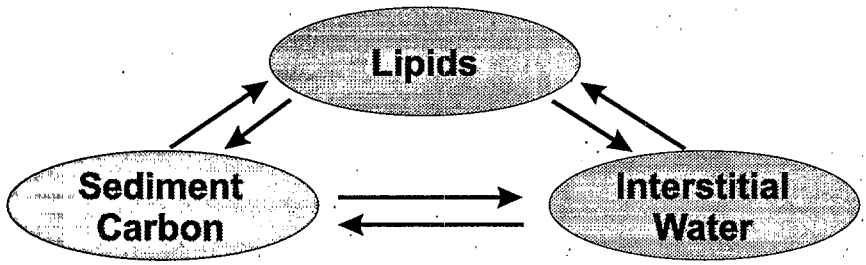
PREDICTING BIOACCUMULATION OF SEDIMENT POLLUTANTS BY BENTHIC ORGANISMS

- FIELD APPROACH
- BIOACCUMULATION TEST
- STEADY-STATE MODELS
  - BIOACCUMULATION FACTORS (BAFs)
  - EQUILIBRIUM PARTITIONING (BSAFs)
- KINETIC MODELS
  - COMPARTMENT-BASED MODELS (1ST-ORDER KINETIC MODEL)
  - PHYSIOLOGICAL- & ENERGETIC-BASED MODELS

SUMMARY OF QUESTIONNAIRE CHOICES. NUMBERS ABOVE DECISION BOXES REFER TO QUESTIONNAIRE CHOICES.



EQUILIBRIUM PARTITIONING BIOACCUMULATION MODEL



$$Ctss / L = (Cs / TOC) * AF$$

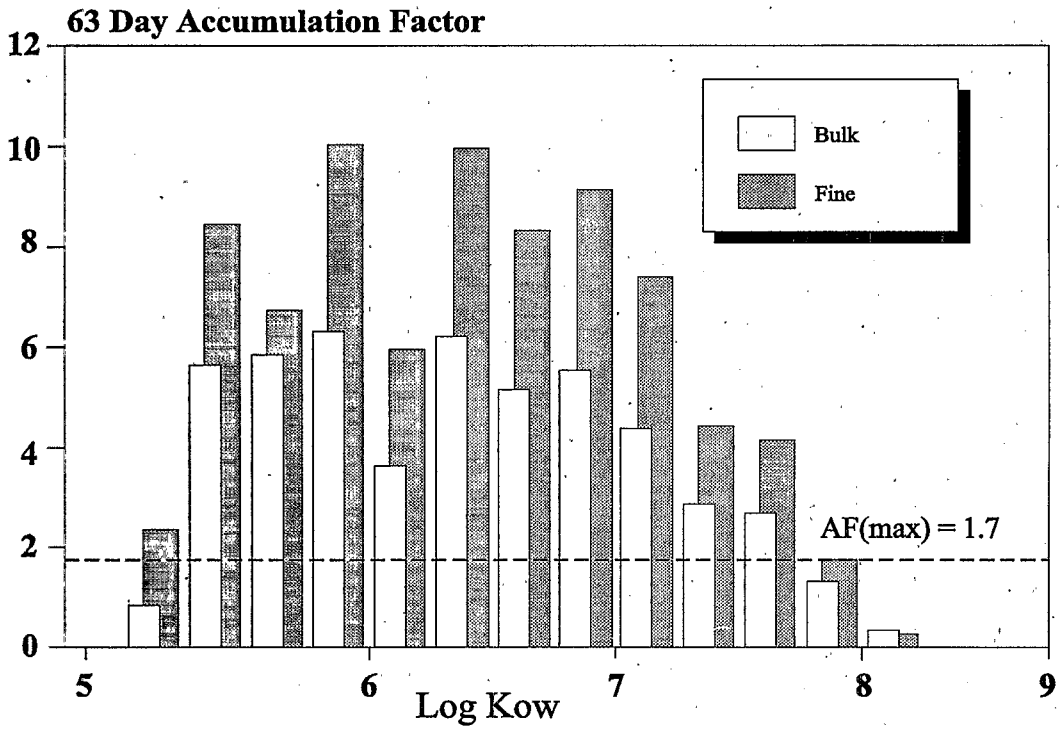
$$AF = (Ctss / L) / (CS / TOC)$$

Where:

Ctss = Tissue conc. at steady-state (ug/g)
L = Lipid content (g/g)
TOC = Total organic carbon in sediment (g/g)
Cs = Sediment conc. (ug/g)
AF = Accumulation Factor (g carbon/g lipid)

1) Tissue residues cannot exceed the concentration set by partitioning (AF < = 2)
2) AFs do not vary among species, sediments, or compounds

PCB Congener AFs in Fine and Bulk Sediments



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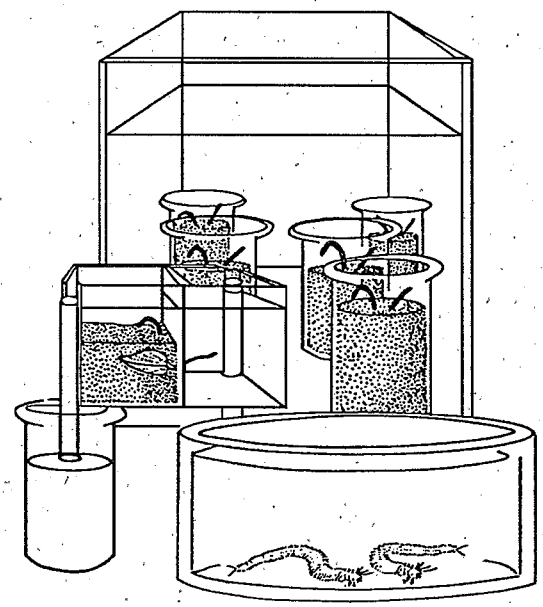
EPA

Guidance Manual

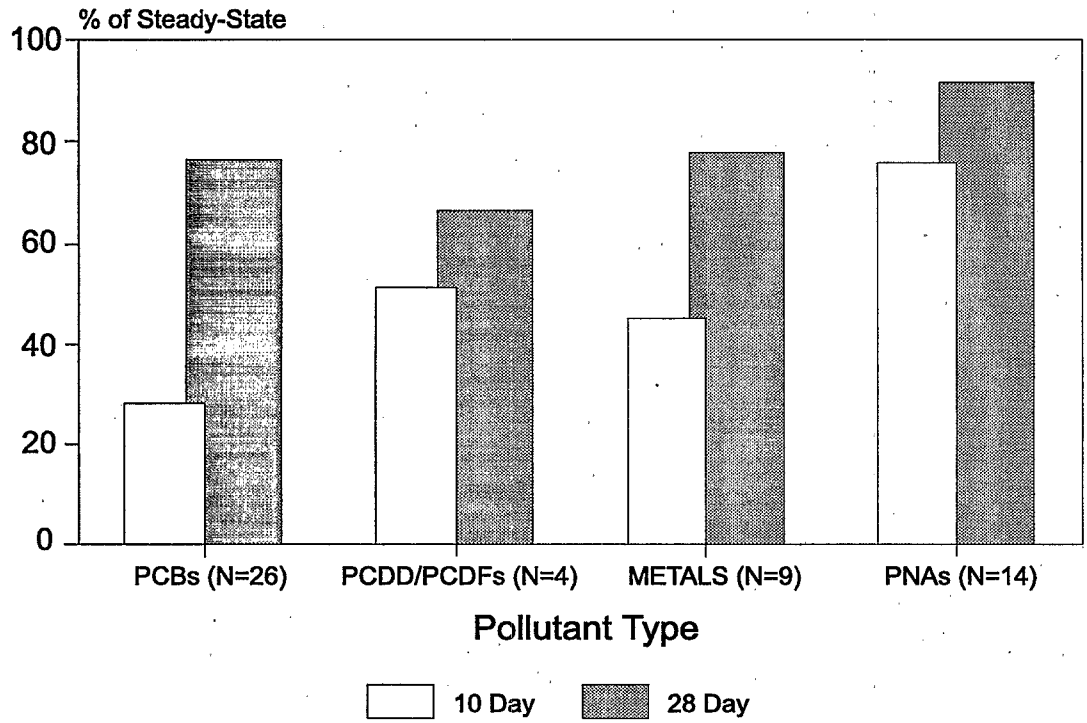
Bedded Sediment Bioaccumulation Tests



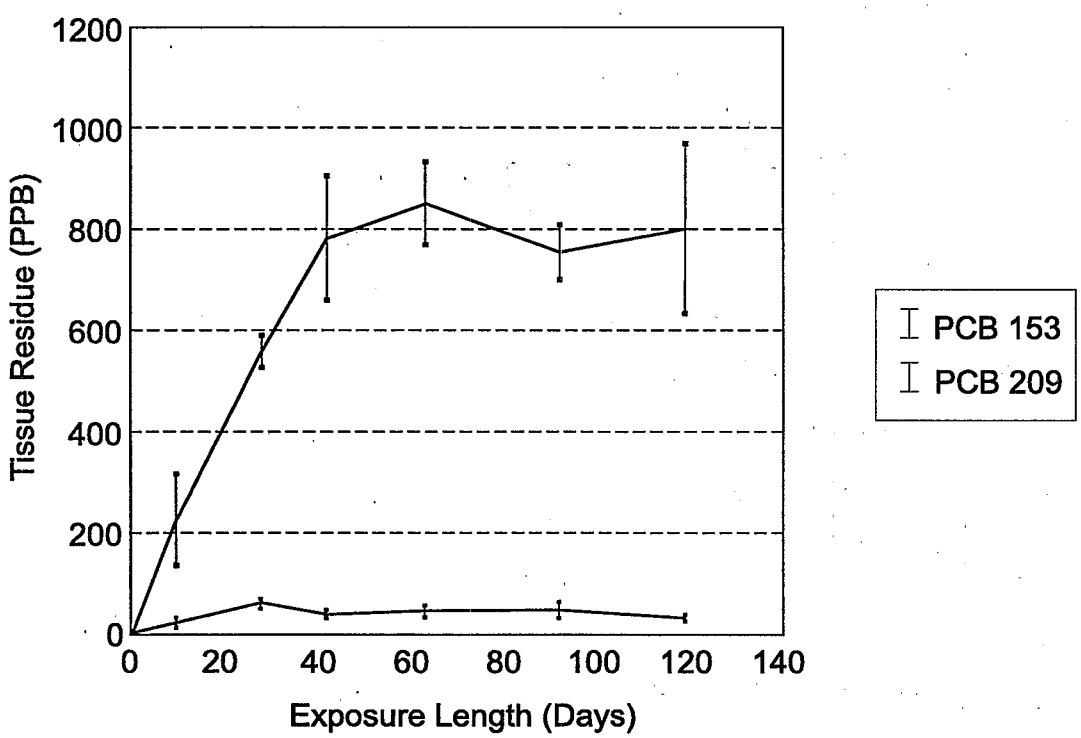
SEDIMENT BIOACCUMULATION TEST
KEY PROCEDURES

1. 28-DAY EXPOSURE DURATION.
2. SEDIMENT-INGESTING ORGANISM REQUIRED.
3. NO SUPPLEMENTAL FOOD USED.
4. SPECIES EXPOSED INDEPENDENTLY.
5. 80% OF STEADY-STATE TISSUE RESIDUES RECOMMENDED ACCURACY.
6. LONG-TERM TESTS OR TOXICOKINETIC APPROACHES USED FOR >80% ACCURACY OR SLOWLY ACCUMULATED COMPOUNDS.

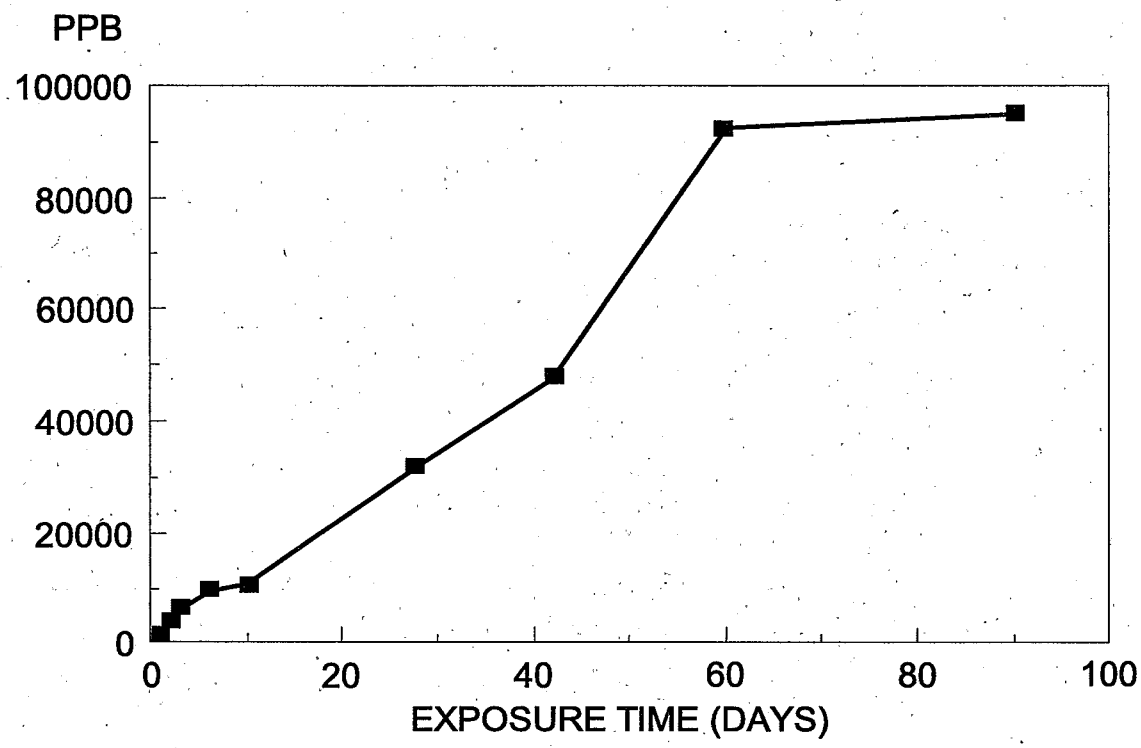
Percent of Steady-State



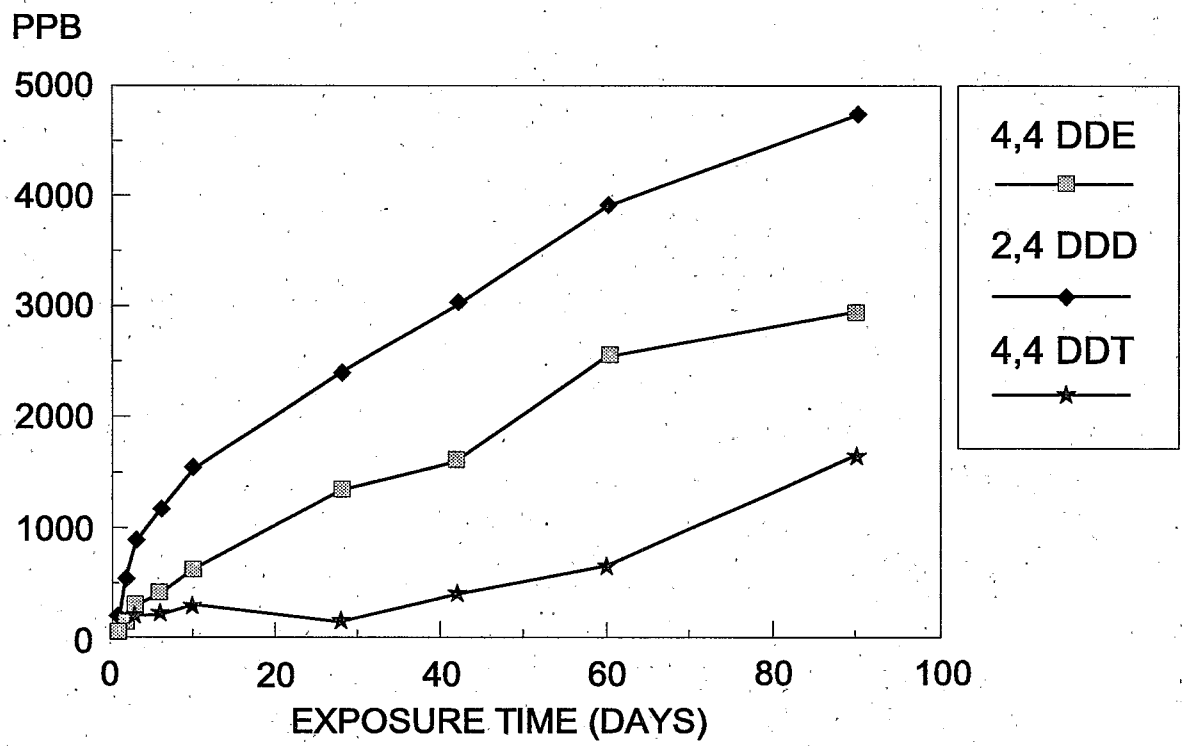
UPTAKE OF PCB CONGENERS 153 VS 209 BY *MACOMA NASUTA*



SUM DDT UPTAKE



TISSUE UPTAKE



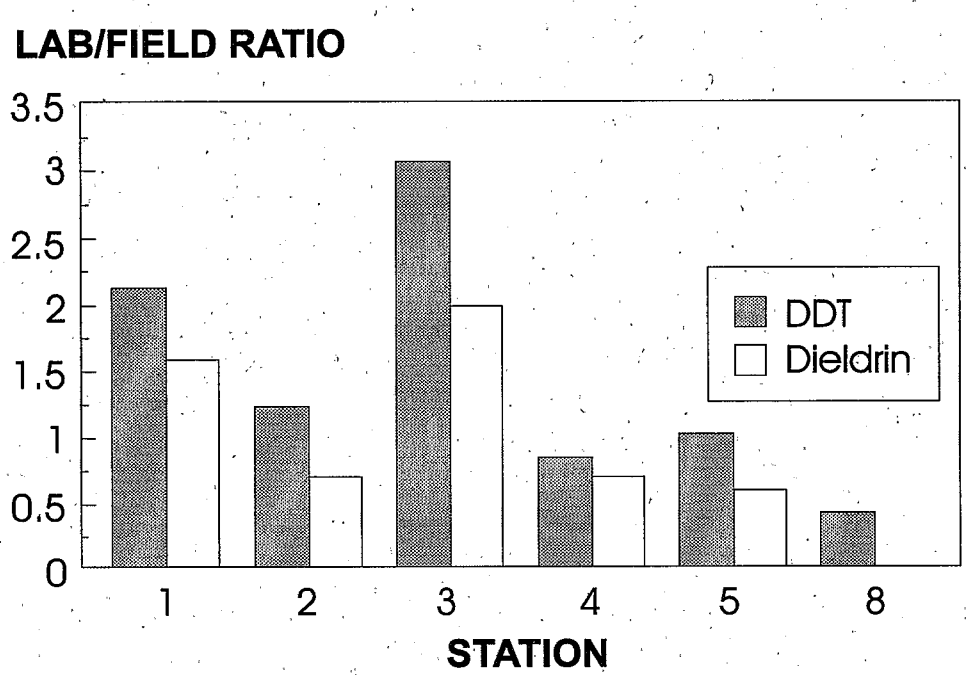
CRITERIA FOR ORGANISM SELECTION

1. Sediment ingester
2. Infaunal (preferably non-tubicolous)
3. Hardy
4. Easily collected or cultured
5. Sufficient biomass for chemical analysis
6. High bioaccumulation potential
7. Feeding behavior understood
8. Suitable for mechanistic/kinetic studies

BSAFs IN FIELD-COLLECTED *MACOMA NASUTA* VERSUS FILTER-FEEDING BIVALVES IN RICHMOND HARBOR

	Macoma	Filter-Feeding Bivalves	Difference
ΣDDT	0.75	0.10	7.5X
DIELDRIN	1.13	0.13	8.7X

LAB/FIELD TISSUE RESIDUE RATIOS



SEDIMENT BIOACCUMULATION
RESEARCH NEEDS

- INTERLABORATORY ROUND-ROBIN
- FIELD VALIDATION (PAHs, METALS)
- LOCAL TEST SPECIES, ESP. FOR SUBTROPICAL, SUBARCTIC, AND OLIGOHALINE HABITATS
- "STANDARDIZATION" OF LIPID METHODS FOR BSAFs
- EFFECTS OF SEDIMENT STORAGE AND SPIKING ON BIOAVAILABILTY
- REFINEMENT OF EXPERIMENT DESIGN, INCLUDING CRITERIA FOR CONTROLS AND REFERENCES
- EVALUATION OF KINETIC & PHYSIOLOGICAL-BASED ALTERNATIVES TO 28-DAY TEST

WHAT IS ECOLOGICAL SIGNIFICANCE?

- PREDICT TISSUE RESIDUE EFFECTS
- INTEGRATE TISSUE RESIDUE EFFECTS INTO ECOLOGICAL RISK ASSESSMENTS
  - ECOLOGICALLY RELEVANT SPATIAL SCALES
  - MULTIPLE STRESSORS - MULTIPLE ENDPOINTS
  - COMPARATIVE RISK TO OTHER TRADITIONAL & NON-TRADITIONAL STRESSORS

Methods for Assessing Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates

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Over the past 10 years, a variety of methods have been described for evaluating the toxicity of sediment-associated contaminants with freshwater invertebrates (i.e., USEPA, 1994; ASTM, 1997a). However, only a limited number of standard methods are currently available for assessing bioaccumulation of contaminants from field-collected or laboratory-spiked sediments (see page 1-31). Standard guides have recently been published for conducting 28-day bioaccumulation tests with the oligochaete *Lumbriculus variegatus* including determination of bioaccumulation kinetics for different compound classes (USEPA, 1994; ASTM, 1997b). These methods have been applied to a variety of sediments to address issues ranging from site assessments to bioavailability of organic and inorganic contaminants using field-collected and laboratory-spiked samples (Schuytema et al., 1988; Nebeker et al., 1989; Ankley et al., 1991; Call et al., 1991; Carlson et al., 1991; Ankley et al., 1993; Kukkonen and Landrum, 1994; Brunson et al., 1998; see ASTM, 1997b for a listing of these citations). Results of laboratory bioaccumulation studies with *L. variegatus* have been confirmed with comparisons to residues (polychlorinated biphenyls, PCBs; polycyclic aromatic hydrocarbons, PAHs) present from field populations of oligochaetes collected from the same sites as sediments used in the laboratory exposures (Ankley et al., 1992; Brunson et al., 1998). Additional method development is under way to evaluate bioaccumulation kinetics and to provide additional data confirming responses observed in laboratory sediment tests with benthic communities in the field.

Selection of Test Organisms

The choice of a test organism has a major influence on the relevance, success, and interpretation of a test. Various organisms have been suggested for use in studies of chemical bioaccumulation from freshwater sediments (Table 1). The following criteria outlined in Table 1 were used to select *L. variegatus* for bioaccumulation method development by USEPA (1994) and ASTM (1997b): (1) ease of culture and handling, (2) known chemical exposure history, (3) adequate tissue mass for chemical analyses, (4) tolerance to a wide range of sediment physicochemical characteristics, (5) low sensitivity to contaminants associated with sediment, (6) amenability to long-term exposures without feeding, (7) ability to accurately reflect concentrations of contaminants in field-exposed organisms (i.e., exposure is realistic), and (8) data confirming the response of laboratory test organisms with natural benthic populations. Thus far, extensive interlaboratory testing has not been conducted with *L. variegatus*. Other organisms did not meet many of the selection criteria outlined in Table 1, including mollusks (valve closure), midges (short life cycle), mayflies (difficult to culture), amphipods (i.e., *Hyalella azteca*: small tissue mass, too sensitive), cladocerans and fish (not in contact with sediment).

Testing Procedures for *Lumbriculus variegatus*

The 28-day bioaccumulation test with *L. variegatus* described in USEPA (1994) and ASTM (1997b) is conducted with adult oligochaetes at 23°C with a 16L:8D photoperiod at an illuminance of about 500 to 1000 lux. Test chamber size ranges from 4 to 6 L, and the chamber contains 1 to 2 L of sediment and 1 to 4 L of overlying water with five replicates recommended for routine testing. To minimize depletion of sediment contaminants, a ratio of 50:1 total organic carbon in sediment to dry weight of organisms is recommended. A minimum of 1 g (wet weight)/replicate, with up to 5 g/replicate should be tested. Organisms are not fed during a bioaccumulation test (see page 1-36).

Table 1. Selection criteria for sediment bioaccumulation test organisms (EPA, 1994; ASTM, 1997b; Ingersoll et al., 1995). A "+" or "-" rating indicates a positive or negative attribute; "NA" is not applicable; and "?" is unknown.

Criterion	*Lumbriculus variegatus*	Mollusks	Midges	Mayflies	Amphipods	Cladocerans	Fish
Laboratory culture	+	-	+	-	+	+	+
Known chemical exposure	+	-	+	+/-	+	+	+
Adequate tissue mass	+/-	+	-	+	-	-	+
Low sensitivity to contaminants	+	+	-	-	-	-	+/-
Feeding not required during testing	+	+	-	+	-	-	+
Realistic exposure	+	+/-	+	+	+	-	-
Sediment physico-chemical tolerance	+	?	+/-	-	+	NA	NA
Response confirmed with benthic populations	+	?	?	?	+	?	-

If sediments could be toxic to *L. variegatus*, a 4-day toxicity screening test should be conducted before starting a bioaccumulation test (ASTM, 1997b). Endpoints monitored in the toxicity test are survival and behavior. Test organisms should burrow into test sediment because avoidance of test sediment by *L. variegatus* may reduce bioaccumulation. Survival of *L. variegatus* in the toxicity screening test should not be significantly reduced in the test sediment relative to a control sediment. Additional requirements for test acceptability are outlined in USEPA (1994) and ASTM (1997b).

At the end of the bioaccumulation test, live oligochaetes are transferred to a 1-L beaker containing overlying water without sediment for 24 hours to eliminate gut contents (oligochaetes clear more than 90 percent of the gut contents in 24 hours). A correction for the extent of elimination from the body burden may need to be made for compounds with log K_{ow} less than 5. Oligochaetes are not placed in clean sediment to eliminate gut contents because clean sediment can contribute 15 to 20 percent to the dry weight of the oligochaetes, resulting in a dilution of contaminant concentrations on a dry weight basis. Minimum tissue mass required for various analyses at selected lower limits of detection are listed in USEPA (1994) and ASTM (1997b). Depending on study objectives, total lipids can be measured on a subsample of the total tissue mass of each replicate sample. Dry weight of oligochaetes can be determined on a separate subsample from each replicate.

Because bioaccumulation tests are often used in ecological or human health risk assessments, the procedures are designed to generate estimates of steady-state tissue residues. Eighty percent of steady state is used as the general goal for a test (ASTM, 1997b). An option when conducting a bioaccumulation test is to perform a kinetic study to estimate steady-state concentrations instead of conducting a 28-day bioaccumulation test (e.g., sample on Days 1, 3, 7, 14, 28). A kinetic test can be used when 80 percent of steady state will not be obtained within 28 days or when more precise estimates of steady-state tissue residues are required (see page 1-37).

Case Studies

Methods for conducting bioaccumulation tests with *L. variegatus* have varied slightly over the years; however, test conditions (e.g., test length, exposure systems) have been consistent enough for evaluation of the robustness of the guidance outlined in USEPA (1994) and ASTM (1997b). In a study with sediments from the lower Fox River in Green Bay, Wisconsin, Ankley et al. (1992) compared the bioaccumulation of PCBs by *L. variegatus* exposed in the laboratory to PCB residues in collections of oligochaetes from the field. Good agreement was observed between PCB concentrations in the laboratory and field organisms, particularly for those congeners with K_{ow} values <7 (see Figure 1). This indicates that for super-hydrophobic chemicals, laboratory exposures longer than 28 days may be required to reach equilibrium.

Good agreement was also observed in bioaccumulation between *L. variegatus* exposed in the laboratory for 28 days and field-collected oligochaetes from sediments collected from the upper Mississippi River (Brunson et al., 1998). About 90 percent of the corresponding concentrations of PAHs were within a factor of 3 between the laboratory-exposed and field-collected oligochaetes (see Figure 1). Concentrations that differed by more than a factor of 3 included

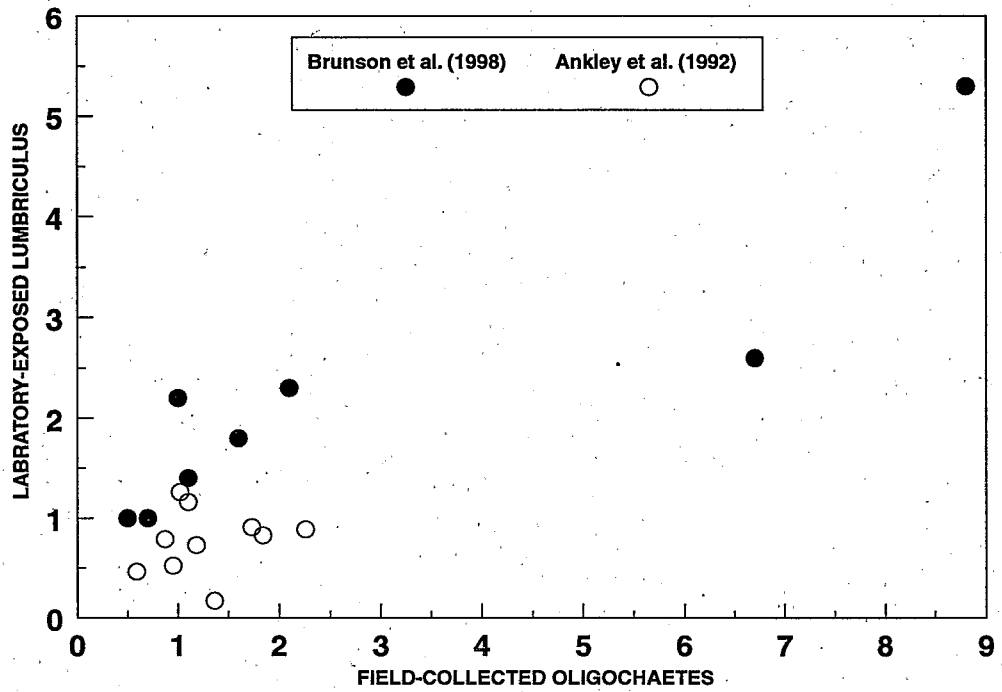

Figure 1. Biota-sediment accumulation factors (BSAFs) for laboratory-exposed *Lumbriculus variegatus* and field-collected oligochaetes for PAHs (Brunson et al., 1998) and PCB homologs (Ankley et al., 1992).

naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, 2,6-dimethylnaphthalene, 1,6,7-trimethylnaphthalene, phenanthrene, 1-methylphenanthrene, and benz(a)anthracene. Tissue concentrations of naphthalenes were generally higher in field-collected oligochaetes relative to laboratory-exposed oligochaetes (naphthalenes are low molecular weight (LMW) PAHs with log K_{ow} values less than 4.5). Compounds with similar concentrations in both the laboratory-exposed and field-collected oligochaetes included a similar number of high molecular weight (HMW) and LMW PAHs. These compounds included biphenyl, fluorene, 1-methylphenanthrene, pyrene, fluoranthene, chrysene, and benzo(e)pyrene. Most of these compounds are intermediate in molecular weight and log K_{ow} (except for benzo(e)pyrene, which has the highest molecular weight and log K_{ow} compared to these other compounds). Compounds with concentrations typically higher in the laboratory-exposed oligochaetes compared to field-collected oligochaetes were primarily HMW PAHs. These compounds included phenanthrene, benz(a)anthracene, benzo(b,k)fluoranthene, and perylene (with log K_{ow} greater than 4.5).

Differences between tissue concentrations in the laboratory-exposed and field-collected oligochaetes may be the result of differential exposure, including the following factors: (1) LMW PAHs may be lost during the sampling of sediments from the field; (2) spatial heterogeneity of contaminants in the field may have resulted in differential accumulation; (3) the route of exposure for oligochaetes in the field is through sediment, food, and overlying water, while the primary route of exposure to oligochaetes in the laboratory is sediment; and (4) species-specific differences in exposure exist between *L. variegatus* and the native oligochaetes.

Concentrations of DDT reached 90 percent of steady state by Day 14 of a 56-day test with *L. variegatus* exposed to field-collected sediments (unpublished data). However, LMW PAHs (i.e., acenaphthylene, fluorene, phenanthrene) generally peaked by Day 3 and tended to decline to Day 56. Concentrations of HMW PAHs (i.e., benzo(b)fluoranthene, benzo(e)pyrene, indeno(1,2,3-c,d)pyrene) typically either peaked by Day 28 or continued to increase during the 56-day exposure. Bioaccumulation of contaminants by indigenous oligochaetes that were recovered on Day 28 from the same chamber with introduced *L. variegatus* were also evaluated. Peak concentrations of select PAHs and DDT were similar in the indigenous oligochaetes and in *L. variegatus* exposed in the same chamber (unpublished data). Bioaccumulation of metals from sediments has also been evaluated using *L. variegatus*. Ankley et al. (1991) reported elevated concentrations of Cd and Ni in worms after 10-day exposures to field-collected sediments where the metal (Cd + Ni):acid-volatile sulfide ratio exceeded 1, but not in samples where the ratio was <1. Ankley et al. (1994) also found that worms did not bioaccumulate metals from three sediments containing elevated concentrations of Cd, Ni, Zn, Cu and Pb, when there was sufficient acid-volatile sulfide to complex metals.

Biota-Sediment Accumulation Factors

Biota-sediment accumulation factors (BSAFs) were calculated for *L. variegatus* by dividing the lipid-normalized tissue concentrations by the organic carbon-normalized sediment concentrations (Table 2; Brunson et al., 1998). For laboratory-exposed oligochaetes, mean BSAFs ranged from 1.1 for benz(a)anthracene to 5.3 for naphthalene. For field-collected oligochaetes, mean BSAFs ranged from 0.5 for benz(a)anthracene to 8.8 for naphthalene. For individual samples, BSAFs for naphthalene ranged from 1.6 to 10.1 in laboratory-exposed oligochaetes and

Table 2. Mean biota-sediment accumulation factors (range in parentheses) reported by Lee (1992) and by Brunson et al. (1998). NR is not reported.

Compound	Lee (1992)	Brunson et al. (1998) Lab-exposed oligochaetes	Brunson et al. (1998) Field-collected oligochaetes
Naphthalene	NR	5.3 (1.6-10.1)	8.8 (2.5-26.6)
2-methyl naphthalene	NR	2.6 (0.9-5.1)	6.7 (2.2-12.2)
Pyrene	0.4 (0.18-0.5)	2.3 (0.8-3.9)	2.2 (0.7-5.6)
Fluoranthene	NR	1.8 (0.9-3.9)	1.6 (0.6-4.9)
Chrysene	NR	1.5 (0.7-2.4)	1.1 (0.3-2.0)
Benz(a)anthracene	0.4 (0.2-0.6)	1.1 (0.4-2.5)	0.5 (0.4-0.7)
Benzo(b,k)fluoranthene	0.4 (0.2-1.0)	NR	NR
Perylene	NR	2.24 (0.5-4.7)	1.02 (0.3-1.9)

2.5 to 26.6 in field-collected oligochaetes. The BSAFs for pyrene, benz(a)anthracene, and benzo(b,k)fluoranthene were typically greater that BSAFs reported for marine organisms in Lee (1992) for these compounds (Table 2). BSAFs were also calculated using PCB homolog data reported in Ankley et al. (1992) for laboratory-exposed *L. variegatus* and field-collected oligochaetes (Figure 1). BSAFs were similar between laboratory-exposed and field-collected oligochaetes in both Ankley et al. (1992) and Brunson et al. (1998); however, BSAFs reported in Brunson et al. (1998) were typically greater (0.5 to 8.8) than BSAFs from Ankley et al. (1992; 0.17 to 2.26; Figure 1).

A theoretical value of 1.7 for BSAFs has been estimated based on partitioning of nonionic organic compounds between sediment carbon and tissue lipids (ASTM, 1997b). A BSAF of less than 1.7 indicates less partitioning into lipids than predicted, and a value greater than 1.7 indicates more uptake than can be explained by partitioning theory alone (Lee, 1992). The majority of the BSAFs in Figure 1 and Table 2 were within a range of about 0.5 to 2.6, suggesting the theoretical BSAF value of 1.7 could be used to predict these mean BSAFs with a fair amount of certainty. However, mean BSAFs for naphthalene (8.8) and 2-methyl naphthalene (6.7) in the field-collected oligochaetes were elevated relative to a theoretical BSAF of 1.7 (Table 2), with BSAFs for individual samples as high as 10.1 for laboratory-exposed oligochaetes and 26.6 for field-collected oligochaetes. The higher BSAFs in the field-collected oligochaetes may be the result of (1) exposure to contaminants in the overlying water; (2) spatial differences in sediment contamination (i.e., sediments were not sampled from a depth representative of the habitat of the oligochaetes); or (3) taxon-specific differences in exposure. BSAFs substantially different from the theoretical value of 1.7 may also result from the system not being at equilibrium (i.e., depletion or release of contaminants in pore water).

In summary, procedures for evaluating the bioaccumulation of contaminants associated with freshwater sediment using the oligochaete *L. variegatus* have been well described. Results of laboratory studies using these procedures are generally similar to the bioaccumulation of contaminants exhibited by oligochaetes in the field. Ongoing research includes further evaluations of bioaccumulation kinetics and field validation of laboratory bioaccumulation methods, use of formulated sediments and sediment spiking, and standardization of micro-lipid analytical methods.

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Methods for Assessing Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates

USEPA National Sediment
Bioaccumulation Conference
Wednesday September 11, 1996
Hyatt Regency, Bethesda, MD

Chris Ingersoll, Eric Brunson, Jim Dwyer
U.S. Geological Survey
Columbia, MO

Objectives:

- Standard methods
- Approaches
- Laboratory exposures
  - Methods
  - Lab to Field comparisons
  - Kinetic studies
- Differences among standard methods
- Future directions

Standard sediment methods

	ASTM	EPA	EC
Toxicity: Fresh water	E1706	1994a	1996a,b
Toxicity: Estuarine & Marine	E1367, E1611	1994b	1992a, 1997a?
Toxicity: Soil	E1676	1986	1994a, 1999?
Bioaccumulation	E1688	1989, 1994a	none
Collection	E1391	1995, 1996?	1994b
Manipulation	E1391	1995, 1996?	1995
Guidance	E1525	1994a,b	1996b, 1997b
Quality Assurance	E1525*	1995	1992b, ISO 9000

Approaches:

- Laboratory-exposed organisms
- Field-collected organisms
- Bioaccumulation factors (BAF)
- Equilibrium partitioning models (BSAF)
- Kinetic models
- Bioenergetic models

Approaches (cont.):

- Bioaccumulation factor:

  BAF = [tissue]/[sediment]

- Equilibrium partitioning models:

  Biota-sediment accumulation factor

  BSAF = [tissue/lipid]/[sediment/TOC]
  ~ 1.7 (4.0 USEPA-USCOE; 1991)

Approaches (cont.):

- Assumptions associated with BSAFs:
  - sediment only source
  - equilibrium & not kinetically limited
  - no metabolic degradation
  - lipid = lipid, TOC = TOC

Selection criteria: Toxicity testing organisms

	HA	DS	CT	CR	LV	TT	HS	MO	CL
Sensitivity	+	-	+	-	+	-	-	-	-
Round robin	+	-	+	-	-	-	-	-	-
Contact sed.	+	+	+	+	+	+	+	+	-
Culture	+	-	+	+	+	+	-	-	+
Taxonomy	+	+/-	+/-	+/-	+	+	+	+	+
Ecological	+	+	+	+	+	+	+	+	+
Geographical	+	+/-	+	+	+	+	+	+	+/-
Physico-chem.	+	+	+/-	+	+	+	-	+	NA
Field validation	+	+	+	+	+	+	+	-	+
Peer review	+	+	+	+	+	+	+	-	+/-
Endpoints	SGM	SBA	SGE	BS	BR	SR	SG	B	SGR

Selection Criteria: Freshwater bioaccumulation testing organisms

	LV	Mol	Mdg	May	Amp	Cla	Fish
Culture	+	-	+	-	+	+	+
Tissue mass	+/-	+	-	+	-	-	+
Sensitivity	+	+	-	-	-	-	+/-
Feeding	+	+	-	+	-	-	+
Realistic exposure	+	+/-	+	+	+	-	-
Physico-chem.	+	?	+/-	-	+	NA	NA
Field validation	+	?	?	?	+	?	-

LV: *Lumbriculus variegatus*, Mol: Mollusks, Mdg: Midges, May: Mayflies, Amp: Amphipods, Cla: Cladocerans

Selection Criteria: Recommended freshwater bioaccumulation testing organisms

Species	*Feeding*	*Biomass*	*Sensitive*	*Culture*	*Data*
Chironomus tentans	*FF/ SDF*	+	-	++	++
Chironomus riparius	*FF/ SDF*	+	-	++	++
*Diporeia spp.**	*SSDF*	-	+	-	++
Hexagenia spp.	*SDF*	+	-	-	++
Hyalella azteca	*SSDF*	-	-	++	++
*Lumbriculus variegatus**	*SSDF*	-	++	++	++
Earthworms	*SSDF*	++	?	++	-

FF = filter feeder; SDF = surface deposit feeder
SSDF = subsurface deposit feeder; Adapted from ASTM E1688

Relative sensitivity: water 10-d LC50s (ug/L; ASTM E1706)

Chemical	*Hyalella azteca*	*Chironomus tentans*	*Lumbriculus variegatus*
Copper	35	54	35
Zinc	73	1125	2984
Nickel	2.8	NT	158
Cadmium	780	NT	12160
Lead	<16	NT	794
p,p'-DDT	0.07	1.23	NT
p,p'-DDD	0.17	0.18	NT
p,p'-DDE	1.39	3.0	>3.3
Dieldrin	7.6	1.1	NT
Chlorpyrifos	0.086	0.07	NT

Lumbriculus variegatus (oligochaeta):

- Location: North America and Europe
- Habitat: tunnels aerobic sediments lakes, rivers, ponds
- Behavior:
  - buries anterior portion in sediment and undulates posterior end in overlying water for respiration
  - processes >12 x weight/day

Lumbriculus variegatus (cont.):

- Adults:
  - 40 to 90 mm length
  - 1.0 to 1.5 mm diameter
  - 5 to 12 mg wet weight
  - about 1% lipid
- Reproduction: asexual (i.e., architomy)
- Culture:
  - adults of various size
  - population doubles in about 8 to 12 days at 23C

Laboratory exposures:

- Single sampling time:
  - steady state (i.e., Day 28?)
  - ANOVA and BSAFs
- Kinetic study:
  - time course (i.e., Day 1, 3, 7, 14, 28, 56)
  - regression models (i.e., K_S and K_2)
- Depuration:
  - experimental (i.e., 24-h gut purge)
  - regression models (i.e., K_S and K_2)

BIOACCUMULATION	*Lumbriculus*: EPA & ASTM	*Macoma*: EPA & ASTM	Polychaetes: EPA & ASTM
Temperature (C)	23	NS	10-25
Luminance (lux)	750	750	750
Photoperiod	16:8	16:8 to 12:12	16:8 to 12:12
Chamber (L)	4-6	NS	NS
Sediment (L)	≥1.0	NS	NS
Water (L)	≥1.0	NS	NS
Water renewal	R	S/R	S/R
Age	adult	adult	juvenile
Loading	>1 g/rep.	1 g/50 g sed.	1 g/200 g sed.
Feeding	No	No	No
Replicates	5	8	8
Duration (days)	28/kinetics	28/kinetics	28/kinetics
Endpoints	B	B	B
Acceptability	T,A,B,R	T,A,B,R	T,A,B,R

Percent of steady state (ASTM E1688)

Compound	Day 10	Day 28	Organism
Phenanthrene	67	95	amphipod
Benzo(a)pyrene	96	100	mayfly
Benzo(a)pyrene	43	75	clam
Benzo(a)pyrene	32	66	amphipod
Chrysene	43	87	clam
Hexachlorobenzene	35	70	clam
Hexachlorobiphenyl	88	100	mayfly
Aroclor 1242	18	87	polychaete
Total PCBs	23	87	clam
Cadmium	17	50	shrimp

Percent loss during gut purging (ASTM E1688)

Compound	24 h	72 h	Organism
PCB	3	8	shrimp
Hexachlorobenzene	4	12	clam
Benzo(a)pyrene	4	12	amphipod
Phenanthrene	11	33	amphipod
Benzo(a)pyrene	14-26	43-99	mayfly
Phenanthrene	77-100	-	mayfly
HCBP	14-26	43-99	mayfly

Errors associated with gut purging:

- Gut sediment error greatest:
  - selective ingest high TOC
  - large gut
  - early in exposure (low uptake)
  - cmpds. not bioaccumulated
- Purging error greatest:
  - rapidly depurated/metabolized cmpds.
  - dilution by uncontaminated sediment

Performance-based criteria:

- Survival (should; 4-d screening test for LV)
- Avoidance (should)
- Food (should; measure chemicals of concern)
- Water quality (should)
- Culture conditions (should)
- Reference toxicants (must: monthly/start of test)
- Physico-chemical characteristics (should)
- Temperature (must; i.e., consistent life stage)
- Storage sediment (2-8 weeks; no consensus)
- Spiked sediment (1 month holding before testing)

Mean BSAFs for PCBs (Lee 1992)

ORGANISM	BSAF
Yoldia limatula	10.6
Nereis virens	10.0
Macoma nasuta	5.9
Yoldia limatula	5.7
Nereis virens	5.2
Macoma nasuta	3.4
Nereis virens	3.2
Nereis virens	1.9
Macoma nasuta	1.8
Nereis virens	0.5
Macoma nasuta	0.4
Oligochaetes*	0.8/0.9

Mean BSAFs for other compounds

COMPOUND	BSAF1	RANGE	BSAF2	RANGE
Chlordane	4.7	4.0-5.9	-	-
Hexachlorobenzene	3.1	2.1-4.1	-	-
DDD	2.1	0.4-4.8	-	-
DDE	1.3	0.7-2.8	-	-
2,3,7,8-TCDD	0.7	0.5-0.8	-	-
Pyrene	0.4	0.2-0.5	1.1-2.3	0.7-5.6
Benzo(b,k)fluoranthene	0.4	0.2-1.0	0.6-0.8	0.3-1.5
Chrysene	0.4	0.2-0.6	1.0-1.4	0.3-2.4
Benz(a)anthracene	0.4	0.2-0.6	0.5-1.0	0.4-2.5
Benzo(a)pyrene	0.2	0.05-0.9	-	-

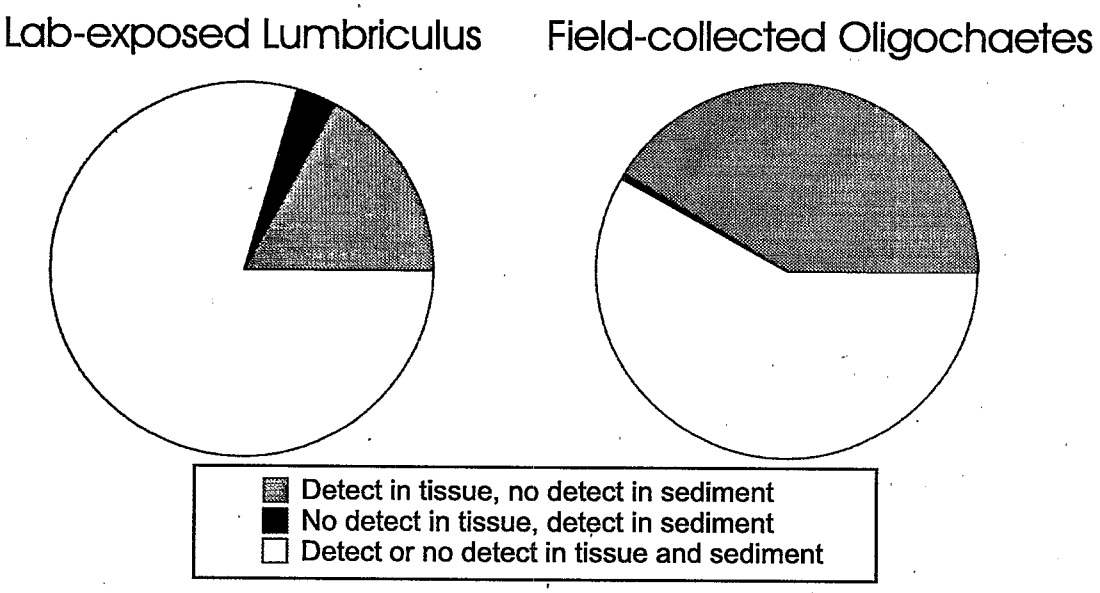
BSAF1 (Lee 1992) and BSAF2 (Brunson et al. 1998)

Mean BSAFs for oligochaetes

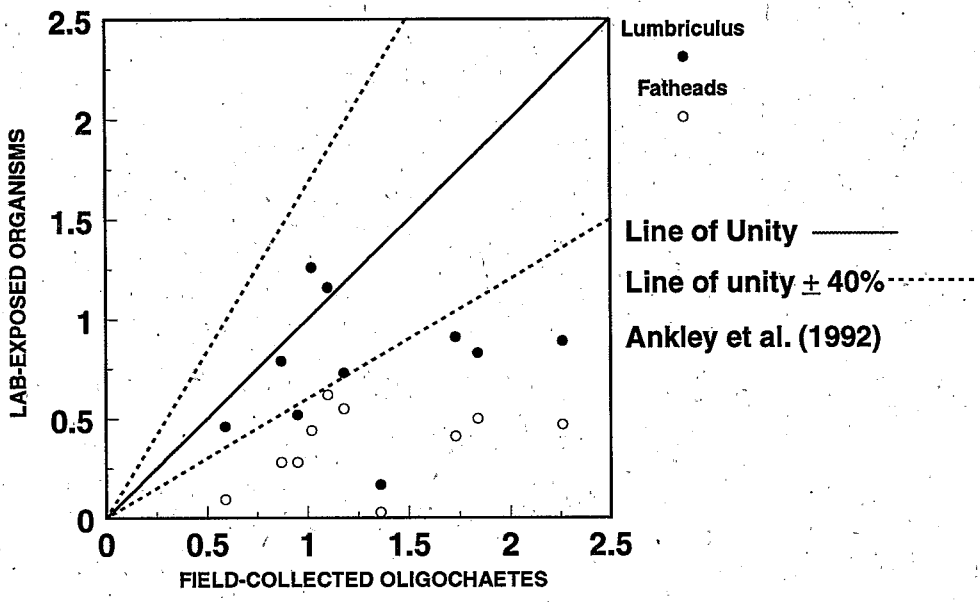
COMPOUND	LAB	RANGE	FIELD	RANGE
2-methylnaphthalene	2.6	0.9-5.1	6.7	2.2-12.2
Benz(a)anthracene	1.0	0.4-2.5	0.5	0.4-0.7
Benzo(b,k)fluoranthene	1.0	0.6-1.5	0.7	0.3-1.5
Chrysene	1.4	0.7-2.4	1.1	0.3-2.0
Fluoranthene	1.8	0.85-3.9	1.6	0.6-4.9
Naphthalene	5.3	1.6-10.1	8.8	2.5-26.6
Perylene	2.2	0.5-4.7	1.0	0.3-1.9
Pyrene	2.3	0.8-3.9	2.1	0.7-5.6

Brunson et al. 1998

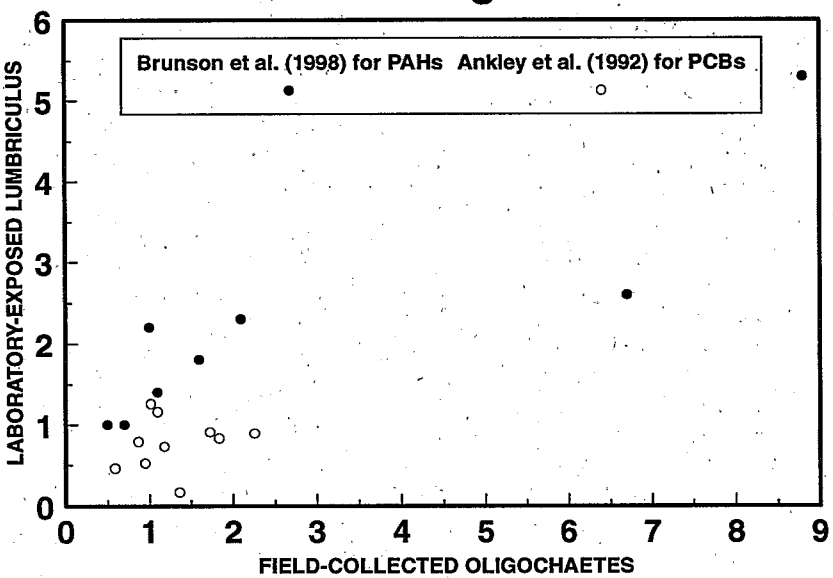
Frequency of Detected Concentrations: Tissue vs sediment (Brunson et al., 1998)

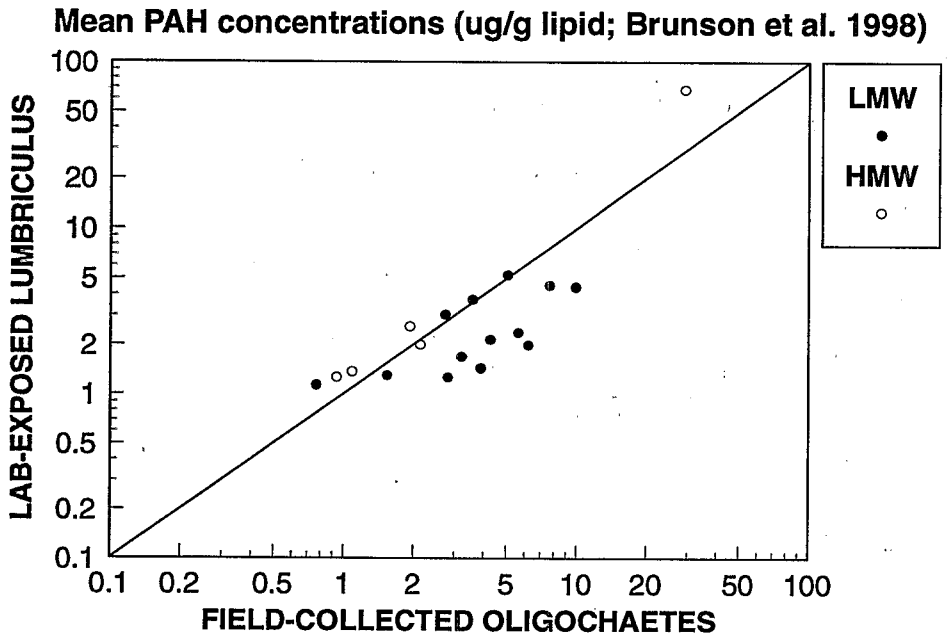


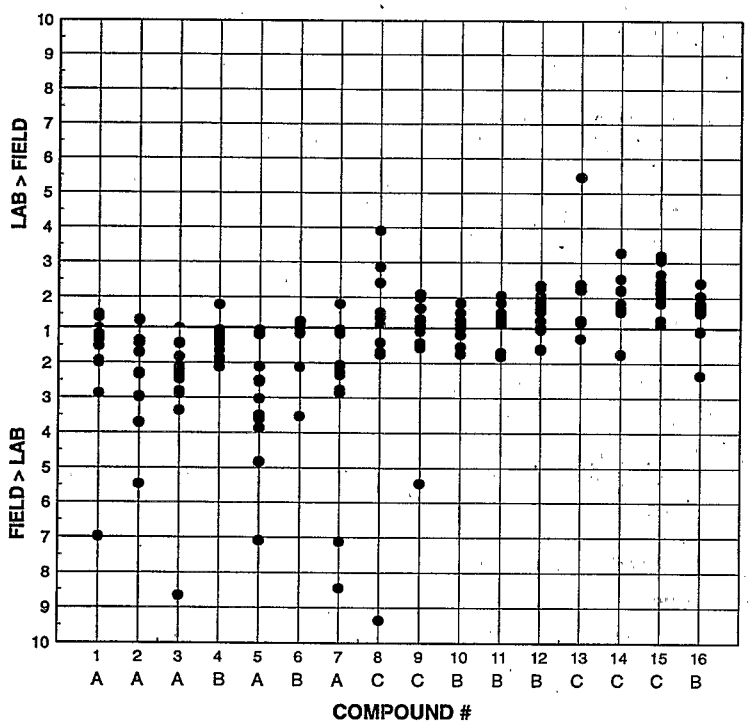
BSAFs for PCBs: Oligochaetes and fish



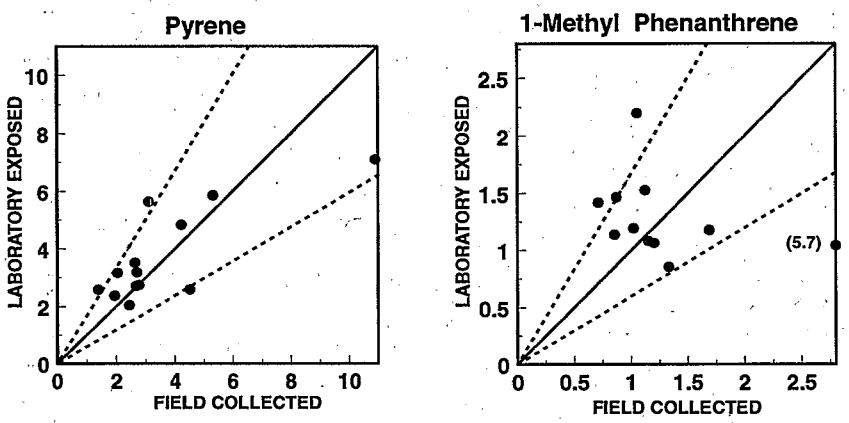
BSAFs for oligochaetes




Mean PAH concentrations (ug/g lipid; Brunson et al. 1998)
LAB-EXPOSED LUMBRICULUS
100
50
20
10
5
2
1
0.5
0.2
0.1
0.1 0.2 0.5 1 2 5 10 20 50 100
FIELD-COLLECTED OLIGOCHAETES
LMW
HMW


LAB > FIELD
FIELD > LAB
10 9 8 7 6 5 4 3 2 1 2 3 4 5 6 7 8 9 10
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16
A A A B A B A C C B B B C C C B
COMPOUND #

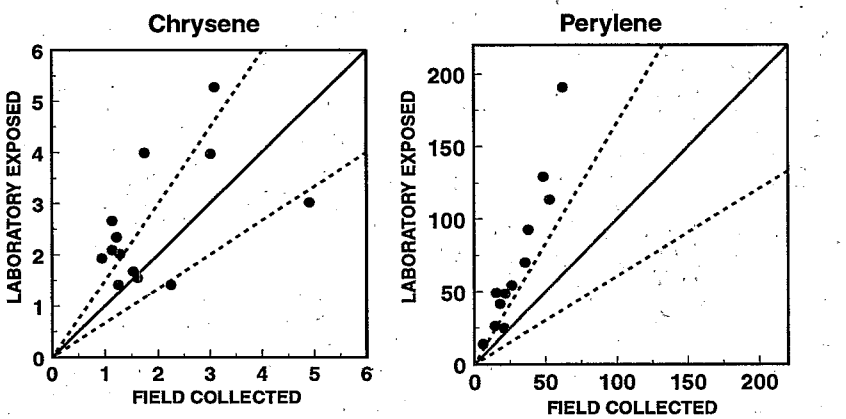
Tissue Concentrations (ug/g of lipid)



Line of Unity ---------- Line of unity ± 40%

Other compounds displaying a similar pattern (i.e. field ≈ lab):
Fluoranthene, Fluorene, Benzo(e)pyrene

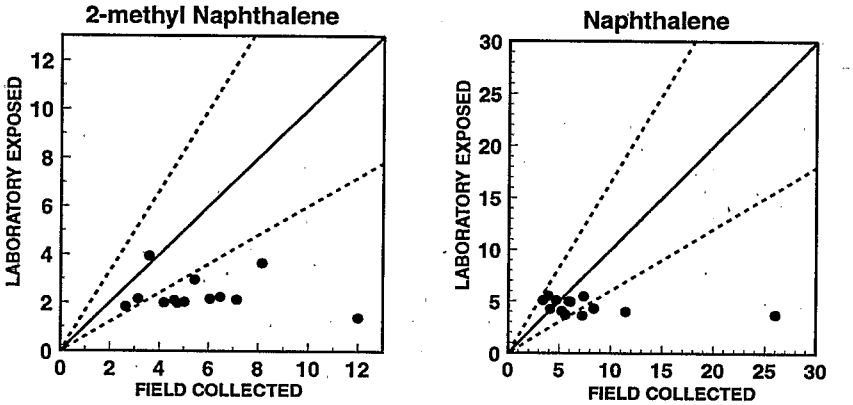
Tissue Concentrations (ug/g of lipid)



Line of Unity ---------- Line of unity ± 40%

Other compounds displaying a similar pattern (i.e. lab>field):
Benzo(a)anthracene, Benzo(b,k)fluoranthene

Tissue Concentrations (ug/g of lipid)



Line of Unity ---------- Line of unity ± 40%

Other compounds displaying a similar pattern (i.e. field>lab):
other Naphthalenes

Differences among standard sediment methods:

- Static vs. flow-through (fresh vs. marine)
- Type & quantity of food (toxicity vs. bioaccum.)
- Age (*Hyalella* and *Chironomus:* EC vs. EPA)
- Duration & endpoints (*Hyalella*: EC vs EPA)
- Sieving sediment (EC vs. EPA and ASTM)
- Sediment storage (2 to >8 weeks; consensus?)

Future directions:

- Spiking & formulated sediments
- Quality assurance:
  - Lab certification (EC)
  - Reference toxicants
- Standardization of micro-lipid methods
- Kinetics and bioenergetics models
- Field validation

Kinetic Models for Assessing Bioaccumulation

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Kinetic models for exposure of aquatic organisms were developed for water-only exposures to develop a method for shorter-term studies to estimate steady state (Branson et al., 1975; Neely, 1979). The toxicokinetics were assumed to be driven primarily by the thermodynamic differences in the chemical activities in the storage compartment (organism) and the source compartment (water). The driving force for the ultimate storage may be limited through a number of kinetically limiting steps representing a number of potential mechanisms. Some of these limits are external to the organism, some are external but driven by the interaction of the organism with its environment, and some represent internal physiological processes of the organism.

What are some of these physiological and environmental limitations? The rate of presentation of the contaminant to the uptake membrane may be limited by diffusion within the source compartment. The extreme example of a diffusion-restricted environment is sediment, where the diffusion path can be very tortuous. Such diffusion limitations can be reduced by organism behavior including increases in respiration, resulting in the active pumping of water across the gills and increased ingestion rates, which exposes the organism to a larger volume or mass of the source compartment and thus the contaminant. Environmental factors such as temperature may alter physiological processes and result in changes in physiological features such as respiration and metabolism. These in turn may alter the volume of the source compartment encountered and the resultant exposure. The balance may result in greater or lower concentrations over time. Limitations at the physiological level also include limitations of compound transport from the site of accumulation to the ultimate storage site within the organism. Such processes can limit or enhance the transport from the site of uptake. If the transport from the site of uptake is limited, then the apparent difference in chemical activity forcing the transport into the organism may be reduced and the rate process slowed. Likewise, if the internal distribution is rapid, a large chemical activity gradient can be maintained and the rate will be rapid. Changes in the metabolic rate within the organism can also alter the rate of biotransformation, and thus the ultimate rate and potential for the accumulation of the parent compound. However, in these cases the flux into the organism remains high, and if the metabolite is the toxic form, its flux and accumulation will be enhanced.

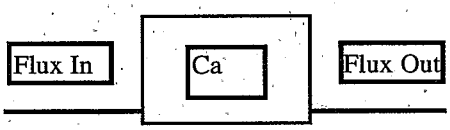
Using the simplest model of accumulation and loss, some of these various factors can be demonstrated, e.g., the effect of respiration rate on the uptake process. Assuming no biotransformation:

$$\frac{\Delta C_a}{\Delta t} = Flux_{in} - Flux_{out}$$

In this form, it is difficult to quantitate the changes in concentration in the organism or predict them, but by making additional restrictions, quantitation and prediction become possible. The formalisms available for this simple model can be represented in compartment, clearance volume, or fugacity forms. In this simple model, the various formalisms can be interconverted (Landrum et al., 1992a). However, each of the approaches has slightly different assumptions to yield mathematically equivalent results.

In addition to compartment-based kinetic models, both physiologically and bioenergetics-based models have been employed to describe the accumulation and distribution of contaminants in aquatic organisms (Landrum et al., 1992a). For instance, a bioenergetics model for the clam, *Macoma nasuta*, was studied with the contaminant hexachlorobiphenyl (Boese et al., 1990). In this case, the routes and rates of accumulation and loss could be well defined. The difficulty with these approaches is the need for a large amount of data to parameterize the model. These models are particularly useful for relating the exposure of organisms to fundamental processes such as respiration and feeding rates.

In the compartment-based formalism and the absence of biotransformation:



$$\frac{dC_a}{dt} = k_u C_s - k_e C_a$$

Where:

k_u = uptake clearance (ml g^{-1} h^{-1})
C_s = concentration in the source (ng ml^{-1})
k_e = elimination constant (h^{-1})
C_a = concentration in the organism (mg g^{-1})

In this form, some restrictions that are not usually recognized come into play. First, the concentration in the source is the bioavailable concentration. If the total concentration is included as the source concentration, it will affect the estimate of k_u. This is useful for assessing differences in bioavailability assuming that the conditions of the experiment do not vary significantly. In water with little complexing capacity, it is generally the concentration in the water that is the source concentration. However, there are examples where water concentrations are modified by the presence of dissolved organic carbon (DOC) (Landrum et al., 1992a). In sediment, the fraction that is bioavailable is less clear and often the total sediment concentration or a concentration on some normalized basis, i.e., carbon, is employed. It is generally understood that C_s is not limiting and that the system is homogeneous. If this is not true, then the flux into the organism will vary considerably over time. Such variation would preclude exact integration of the differential equation and make it necessary to perform numerical integration, for which substantially more information is required.

In the compartment formalism, k_u is the clearance of the source compartment by the organism, usually expressed on a weight-specific basis for the organism and a volume- or mass-specific basis for the source compartment. It is assumed to remain constant over the course of a study or prediction. Whether it actually remains constant over longer predictions or measurements is questionable. This term includes the interaction between the source compartment and the organism and distribution rates for both the internal and external distribution of the compound. K_u is also conditional based on factors that affect the physiology of the organism, the chemistry of the source compartment, and the interaction between these two. Thus if k_u is to be used for comparison of differences in bioavailability, as it often is in many sediment contaminant evaluations, the conditions of the experiment need to be constant across a range of conditions, such as across a range of sediments, so that the comparisons will be valid. In addition, k_e is also assumed to remain constant and is subject to many of the physiological changes that occur in the organisms.

With aqueous exposures, there is an exact conversion between the compartment model and both the clearance volume model and a fugacity model. There are, however, some subtle differences in assumptions and definitions in these models. For the fugacity model, the system tracks the differences in chemical activity within the system. The concentrations are given in terms of moles per volume, and the relative capacities of the systems for the compound are expressed in terms of both partition coefficients and fugacity capacities. The model could work as well in sediment exposures as in aqueous exposures for it is really a compartment model in which the terms are redefined. In the case of the clearance volume model, there are some assumptions that are usually not readily recognized. In the aqueous case, the fluxes into and out of the organism are assumed to primarily occur across the same membrane or at least the membrane resistance is assumed to be equal. This leads to the description of the steady-state condition as the relative capacity of the organism compared to the source compartment. For sediments, it would be necessary to add additional routes of exposure to be accurate with this model. The volume of distribution would reflect the relative capacity of the aqueous phase of the system. Thus, it is not as easy to directly apply the clearance volume model to the sediment environment without modification of the mathematical formalism.

The simple compartment model approach to toxicokinetics in sediments has been employed to demonstrate differences in bioavailability among classes of contaminants, sediments, the effects of concentration, and the impact of normalization procedures. The toxicokinetics approach has shown that two major classes of contaminants, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) seem to have differential bioavailability when the log K_{ow} values are essentially the same (Landrum and Faust, 1991). This also seems to be the case for the relative bioavailability of other chlorinated hydrocarbons and PAHs as well (Harkey et al., 1994). This approach was also useful in attempting to evaluate the relative importance of exposure to interstitial water versus the exposure to whole sediment, suggesting that multiple routes can be important (Harkey et al., 1994). In this case, the relative k_u normalized for the organic carbon content in the media was much greater for exposure to whole sediment than for exposures to interstitial water, suggesting that the exposure in sediment employed additional sources (routes) of exposures compared to simple exposure to interstitial water. Exposures to varying concentrations of sediment-associated contaminants can cause accumulation of sufficient doses that the uptake is affected. When *Diporeia* spp. were exposed to pyrene, k_u increased to a maximum and then tended to decline at doses that produced mortality (Landrum et al., 1994). Finally, the relative bioavailability among sediments of individual contaminants has been estimated through exposures under essentially identical conditions but with differing amounts of DOC in water and among sediments. In one study, the variation in the bioavailability as measured using uptake clearance demonstrated that for sediments collected in Lake Michigan normalization to organic carbon removed essentially all the variability (Landrum and Faust, 1994). However, in the comparison to materials from another source, in this case a soil from Florissant, Missouri, the carbon normalization was not adequate to describe the difference in bioavailability and the differences increased with log K_{ow}. Additional work has shown that the range of variability among

sediments after carbon normalization is somewhat greater than a factor of 10 for selected organic contaminants.

The compartment approach has demonstrated some limitations to our ability to understand and measure the accumulation of contaminants from sediments. The first appearance of complications with this approach was demonstrated in the accumulation of a series of PAHs from a single sediment. The shapes of the kinetics curves varied with log K_{ow} of the compound. It was originally thought that this was an equilibrium problem between the sediment particles and the interstitial water (Landrum, 1989) similar to the observed chemical equilibrium and extraction problem with chemical analyses (Karickhoff, 1980). Experimental designs seemed to indicate that this disequilibrium was a valid issue (Landrum, 1989; Landrum et al., 1992b). However, exposures of organisms to field-collected sediments also seemed to show some of the same kinetic complications where organisms would in some cases rapidly accumulate a compound only to lose concentration over time. Since the sediments were field-collected, it was thought that they were less out of equilibrium than those dosed in the laboratory. Experimentally, it appears that the concentration of biologically available material is changing over time. If the desorption of a compound is inadequate to maintain the interstitial water concentration from the surface easily desorbed concentration, then the bioavailable component of the total contaminant load in the sediment will decline because diffusion within the sediment is limited. When the interstitial water is a greater source than sediment ingestion, as is probably the case early in the exposures, then the potential for depletion of the bioavailable fraction would seem more likely. This issue seems to be more problematic for compounds with log K_{ow} values less than about 5. When pyrene was studied with a laboratory-dosed sediment, the shape of the kinetic curve appeared to be a classic first-order curve with steady state. However, when the elimination estimated from such a curve was compared to directly measured elimination values, it appeared to be inordinately fast. This suggests that the depletion process is important from these sediments (Landrum, 1989; Landrum et al., 1992b). The issue becomes more pronounced with phenanthrene, a compound with a smaller log K_{ow} value. Compounds with larger log K_{ow} values do not generally show such depletion. However, in some work with oligochaetes, which have a faster initial uptake rate than *Diporeia*, an initial uptake with a subsequent plateau or decline was evident. Overall, it would appear that the rate of accumulation from the aqueous phase dictates some of this process and likely it is the relative rate of uptake versus the rate of desorption that is important since diffusion in sediment should be slow. In most of the work that has been performed to date, it is not possible to separate out the mechanisms that may be contributing to such observed variation in the kinetics. Some of the data that would be necessary to evaluate this include good estimates of the desorption rate constants from sediment and estimates of the feeding rate on sediment particles and the associated assimilation efficiencies.

Another feature of the kinetics with *Lumbriculus variegatus* that produced an apparent increase with a subsequent decline in contaminant concentration was found to result from loss of weight by the organisms. *Lumbriculus* from the culture are healthy and fat. When exposed in sediments, they sometimes lose weight and lipids with subsequent losses of contaminant, while in one study, the kinetics on a lipid basis formed a standard first-order uptake that approached steady state (Kukkonen and Landrum, 1994). In addition to the impact on the weight, when *Lumbriculus* were exposed in different ratios relative to the organic carbon content of the sediment, but at the same organic carbon-normalized concentrations, the kinetics changed, suggesting an interaction (Kukkonen and Landrum, 1994). At least by performing a kinetic study, the relationship to steady state for these organisms can be observed instead of assumed and variations in the conditions that impact the steady state determined.

Because of the limitations—in particular, the potential absence of homogeneity of sediment systems—moving from a concentration-based to a mass balance-based model that incorporates more of the physical-chemical processes can help demonstrate which processes are important in the accumulation process. A mass balance model was established to examine the accumulation of sediment-associated contaminants. The model attempted to parameterize the partitioning phenomena as well as the accumulation by several routes for *Diporeia* (Landrum and Robbins, 1990). This first pass at a mass balance model did demonstrate the importance of the role of desorption of contaminants from sediment, perhaps coupled with diffusion limits, on the accumulation process. Further, the desorption rate from particles seemed to be very slow compared to the uptake processes and may well dictate the bioavailability of sediment-associated contaminants along with the ingestion rate and assimilation efficiency. In the model as originally formulated, there was a general absence of data on desorption rates, assimilation efficiencies, and feeding rates.

Today, the ability to estimate the assimilation efficiency for ingested sediment remains extremely difficult. The difficulties are essentially twofold. First, it is nearly impossible to determine the concentration of contaminant in the ingested fraction of sediment for many invertebrates. For oligochaete worms that are general feeders, e.g., they do not strongly select particles, estimating the ingested contaminant concentration is easier. The second issue is to estimate the fraction of material that is retained by the organism. This is generally performed using a non-assimilated tracer. Polydimethylsiloxane and ^{51}Cr have been used, but in both cases the tracers do not sorb to the same particles or in the same proportion as the contaminants (Lydy and Landrum, 1993; Kukkonen and Landrum, 1995). Another approach has been to estimate the relative loss of carbon and subsequently estimate contaminant loss (Lee et al., 1990; Lydy and Landrum, 1993). However, there are not even good assimilation values for the carbon from sediments, so this approach is limited. Overall, development of good assimilation efficiencies is required to improve the estimation of contaminant

accumulation from sediments through kinetic models except with compounds with large log K_{ow} values, where desorption and uptake from interstitial water is of lesser importance.

Summary

Kinetic studies can demonstrate factors that affect the accumulation process and the relationship between the length of exposure and steady state.

Kinetic studies do not seem to be able to determine steady-state potential in all cases due to complications with changes in bioavailability or the length of time required to achieve steady state, which is coupled to changes in physiology.

Mass balance models or models incorporating more of the processes suggest that the desorption rate from sediment particles is an important kinetically limiting process.

Ingestion as the primary route of exposure compared with interstitial water is a less dominant route for chlorinated hydrocarbons compared to PAHs and a less dominant route for more hydrophilic compounds.

To improve kinetic models, data on ingestion rates, assimilation efficiency, desorption rates, feeding selectivity, and measure of contaminant concentrations on ingested particles need improved description and improved chemical measurement techniques. It will be important to move from bulk sediment measures to measures that reflect the exposure environment.

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Day One: September 11, 1996

Session One: Questions and Answers

After each session, there was an opportunity for questions and answers and group discussions pertaining to the speakers' presentations.

Q (Maurice Zeeman, U.S. EPA, Office of Pollution Prevention and Toxics): I heard Gil Veith talk this morning and I would like to comment briefly on a point he made. At a gut level, I believe issues related to bioaccumulation and sediment assessment are important issues. I would like to seek reactions from this panel or others to the following comments. Most of us are scientists. We deal with a Cartesian system where we have divided things up into little discrete parts, because they are too complex to try to handle as a group. The risk assessment paradigm is a perfect example. The key issue is how and when to start putting the parts back together to make some sense out of it. Throughout the conference, I would like to ask that you focus again and again on what Gil Veith said in his presentation where he got that "So what?" reaction from a fairly knowledgeable group of people to the statement that 29 percent of the sediments in major estuaries were being impacted. As we go through our methods and our models for bioaccumulation and sediment assessment, try to think about what additional research will provide essential data to help the public, lawmakers, and regulatory agencies make sound decisions and take sensible actions.

Betsy Southerland:

We are looking for ecological significance. What we are going to try to do, as we go through the next couple days of this conference, is look at the tissue residues from bioaccumulative pollutants and see what impact they have on aquatic life, what impact they have on wildlife consumers of that aquatic life, and their potential for human health impacts. We already have some field studies conducted by Rick Swartz that show a significant impact on populations of other benthos if there is acute toxicity in the amphipod test species. That certainly gives us an indication that at least the acute toxicity tests have ecological significance. What I want to hear today, which was one of the prime drivers for organizing this conference, is what information is available on bioaccumulation tests and tissue residue measurements to make equally important interpretations of ecological significance. Interpretation is the key. We can all do laboratory tests, but then we have to determine what the tests mean. Would anyone else care to comment?

Peter Landrum:

I will add a few comments. It has been my observation that whenever a particular group studies some aspect of the lower end of the food web, there is generally less interest in the work than if it involves fish or fish-consuming wildlife and birds. There seems to be a general lack of recognition of the supporting ecosystem required to maintain the reproduction and the productivity of the higher levels of the food chain. I do not know if it is within the purview of this group to be able to drive that point home. This point applies to studying phytoplankton productivity or contaminants at the lower end of the food web. The issue remains that the connections between the lower food web and the upper food web get lost, particularly when you move into the regulatory realm of lawmakers responsible for environmental legislation.

Q (Peter Chapman, EVS Environment Consultants): I would like to address a question to any of the panel members. We have talked about bioaccumulation in terms of relating it to toxicity effects we already see occurring. One of the attractions bioaccumulation has to me is the possibility of anticipating things before they occur. At this point, do we have any examples where bioaccumulation data have really enabled us to predict impacts, and, if not, how much further do you think we have to go before we can do that?

Henry Lee:

One example would be the work Rick Swartz did on the sum PAH model. Since these are neutral narcotics, we discussed bypassing sediment concentration and just going directly to the tissue residues. That would eliminate the issue of bioavailability. I think that is possible, if we can get good relationships between tissue residues and some

ecologically significant effects. I would also like to add to some of Peter Landrum's earlier comments. Peter, I think the onus is on us to show that these changes in the benthos are important enough that they have affected fisheries populations or wildlife. The public is not primarily interested in amphipods, oligochaetes, or clams, but in fisheries and wildlife. If we cannot protect these populations, then we have failed.

Q (Norm Rubinstein, U.S. EPA, Office of Research and Development): The session this morning really focused on the issues of measuring bioaccumulation and predicting bioavailability. I imagine as the meeting progresses, we will get into the other side of the issue, which is the corresponding effects. But I am curious to see how you gentlemen feel about our current ability to predict bioavailability and bioaccumulation, either thermodynamically or kinetically. Do you have a sense of confidence in our ability to identify bioavailable fractions and go on from there to identify the corresponding ecological effects?

Peter Landrum:

My feeling about that, Norm, is that if we are talking about neutral organic compounds from sediments, we can probably predict bioaccumulation within a factor of ten to twenty, if that is adequate. I think trying to get any better predictions than that right now is not possible because, as I pointed out earlier, we do not have a complete understanding of how contaminants partition among sediment size fractions and the degree of feeding selectivity by the organisms. Without this information, we really do not know how much contaminant an organism is exposed to.

Chris Ingersoll:

We have been focusing quite a bit this morning on the nonpolar organics, but what you have not seen today is some of the work that has been done relative to metals and acid volatile sulfides (AVS). Some of the metals are able to be predictive of bioaccumulation that we are seeing. A series of papers on AVS and metal bioavailability will be published in the December 1996 Society of Environmental Toxicology and Chemistry (SETAC) journal. That issue will also include a good review article by Gary Ankley and others for a variety of studies to address the question of whether or not we can predict bioaccumulation relative to AVS and SEM (simultaneously extracted metal). He found that the SEM/AVS approach offers a more reliable and predictable tool than what is currently available.

Q (Steve Bay, Southern California Coastal Water Research Project): Peter Chapman listed coupling tissue residues with toxicity responses as one of the key issues for this session. But I was wondering about how the recommendations to often use insensitive or hardy test animals will impact this issue. Are we going to end up with a really nice data set, but no ability to couple residue levels with toxicity responses because we do not understand how to predict the responses from uptake residues in insensitive organisms? Are we in danger of that, or will we be able to figure that out once we get a good data set together?

Peter Landrum:

I think we could be somewhat in danger because these insensitive organisms are going to tell us that they can accumulate more than some of the sensitive organisms will. The sensitive animals will pass the toxicity threshold and produce a response. Insensitive organisms should give you some idea of what the maximum amount that could accumulate in an organism would be. This would allow you to at least define the level that you would have to drive down to protect against responses in other organisms.

Chris Ingersoll:

We, as toxicologists, need to develop adequate designs for studies involving water or sediment to measure toxic effects and bioaccumulation in the same exposures. I am really looking forward to hearing from some of the panelists later this afternoon about their databases. Some data are available, but you really have to search the literature to find those kinds of data sets.

Peter Chapman:

I have found it useful in toxicity tests, where the chemicals and organisms are appropriate, to measure bioaccumulation and toxic effects. Among other things, it can help me sort out what may be causing any effects I see. We are trying to move in that direction, but it is a very valid concern.

Q (Joe Greenblott, U.S. EPA, Office of Research and Development): I would like to ask a question about how models fit into the experimental work within the decision-making and risk assessment framework. What level of attention is being given to developing and researching these models and developing laboratory data to the conservative level required for decision-making, in light of the variability in laboratory data and the large uncertainty associated with the predictive models?

Henry Lee:

I will give you a different perspective. We are working now to determine ecosystem responses and cumulative effects. The variability is even greater for this work than for the data you saw here. I think you have to go to a risk aversion philosophy or a more environmentally protective approach. We cannot accurately predict a dose response, a stressor response, but we know it is bad to lose prey in a system or to lose a wetland. The direction we are going to have to go is to a risk aversion strategy. That is where comparative risk factors in, so we can determine how important one risk is versus another. Even if we cannot be quantitative, we can at least rank the risks.

Q (Eric Rifkin, Rifkin & Associates): A number of the panelists today referenced BSAFs, biota-sediment accumulation factors, and how they varied based on whether or not you were using lipid-normalized organisms and carbon-normalized sediment. Could the panelists comment on whether BSAFs can be used in a generic context or whether they need to be used in a site-specific context? I would also be interested in hearing how they can be used in developing sediment quality criteria in general.

Henry Lee:

It comes down to how well you need to know the answer. I agree with Peter that we can predict bioaccumulation within an order of magnitude. If that is good enough for the neutral organics, then you are home free. If you need a better answer, then you have a problem. The better the answer you need, the more expensive and more site-specific it gets. But I am comfortable working within the order of magnitude range.

Q (Lynn McCarty, L.S. McCarty Scientific Research and Consulting): I would like to raise a question related to an earlier question. The question is about sensitivity. I am always concerned that we do not really define what sensitivity is. Are we referring to sensitivity defined according to exposure-based tests or, since we have been discussing tissues residues, are we referring to sensitivity on a residue basis or received dose effect? In fact, as I will show this afternoon, you can consider these things for the same set of data and come to quite different conclusions. Most of the differences in sensitivity that I have seen in the literature can be readily expected and predicted from differences in modifying factors such as the size of the organism, the metabolic activity, and temperature. Until we clearly define what sensitivity means, we should not be making comparisons about which animals are more sensitive than others. In sediment testing, as we have seen from some of the discussions this morning, the variability that results from the differences in media and conditions of those media dramatically affects the accumulation rates and amounts of accumulation. Therefore, sensitivity is a confusing factor that needs to be clearly identified before we make some final pronouncements about it.

Peter Landrum:

I think if you go back and look at the toxicological literature, sensitivity had to do with differences between species. Sensitivities were usually determined, at least in the mammalian literature, with a defined dose approach. A known dose of something was given to two different species by the same route. There is no doubt that the route of exposure is going to alter the response that you expect, particularly for sediment where there are a lot of confounding factors that can influence the dose received. You can take one animal and move it from sediment to sediment with the same compound and get a change in response, because factors change that influence the bioavailability and, therefore, the received dose. If we are going to talk about sensitivity between organisms, we need to talk about the sensitivity based on the received dose.

Henry Lee:

I would like to add a comment on sensitivity. When Chris and I talk about sensitivity, we are referring to a value that is empirically derived. That was the basis for the sensitivity Chris showed in his diagram during his talk. In determining the sensitivity of potential test animals, we need to find animals that will survive for 28 days or however long it takes to reach steady state.

Q (Tom O'Connor, NOAA, National Ocean Service): Two of you have agreed with each other that the predictions based on equilibrium partitioning are good to within one order of magnitude. How would you assess the imprecision of extending the equilibrium partitioning methodology to body burdens in fishes?

Henry Lee:

In general, the imprecision is greater. We derived BSAFs for two fish that have limited home ranges in our DDT Superfund study. These BSAFs turned out to be relatively close to the values we derived for *Macoma* and other benthic organisms. However, these values are more variable for fish like flatfish that have extensive home ranges. But I think we can at least determine a maximum value for demersal fishes.

Q (Tom O'Connor): So the imprecision is in the range of the fish, not in the equilibrium between a given sediment and a fish?

Peter Landrum:

I think you have to consider the routes of exposure as well. If you have a pelagic fish and it is not feeding on things that are well connected with the sediment, then it is inappropriate to try to make a connection between the sediment and that fish. Whereas if you have an organism like a flatfish that is feeding on benthic organisms, it might be easier to make the connection. But you still have additional routes of exposure that you need to consider.

Q (Tom O'Connor): Yes, but equilibrium does not matter for this.

Peter Landrum:

But that is assuming the process is passive. All the models I talked about imply a passive process. When we look at fish in particular, we may no longer be considering a passive process as the sole driving force. So, the thermodynamics that you are trying to consider in terms of

sediment to the fish are no longer applicable. If feeding is taking place and the benthic organism is in equilibrium with the sediment, we could make the assumption that there is a connection between fish and sediment. But if feeding and digestion are taking place, they are active processes that may change the thermodynamics that apply to a particular fish.

Q (Tom O'Connor): So, the answer to my original question is that you cannot apply equilibrium at all to extrapolate the fish.

Peter Landrum:

Not unless they are eating the sediment.

Session Two: Interpretation and Applications of Bioaccumulation Results

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Reference Sediment Approach for Determining Sediment Contamination

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Development of Tissue Residue Threshold Values

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Use of Tissue Residue Data in Exposure and Effects Assessments for Aquatic Organisms

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Comments on the Significance and Use of Tissue Residues in Sediment Toxicology and Risk Assessment

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Quantification of Ecological Risks to Aquatic Biota from Bioaccumulated Chemicals

Reference Sediment Approach for Determining Sediment Contamination

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One of the more contentious and highly publicized regulatory programs capturing headlines today is the dredging and dredged material disposal program, which is jointly administered by the U.S. Army Corps of Engineers (USACE), the U.S. Environmental Protection Agency (EPA) and, in inland waters, the respective state and local authorities. Within this regulatory program, one of the more controversial issues is the identification and use of reference sediments to facilitate the permitting process for dredged material disposal.

The application of a reference sediment approach to identify sediment contamination in dredged materials was developed more than twenty years ago for the ocean dumping program. In this discussion, which reflects my personal views and should not be construed as Agency policy, I will review the statutory basis for the dredged material program, the rationale used to develop this reference-based regulatory approach, and the operational definition of reference sediment as described in the current testing manual. I will also identify how the reference approach is applied today, how it can be improved, and what future research is needed to support the program.

Program Statutes

Section 103 of the Marine Protection, Research, and Sanctuaries Act (MPRSA) stipulates that all operations involving the transportation and dumping of dredged material into ocean waters must be evaluated to determine the potential environmental impacts of these activities. Similar legislation in Section 404 of the Clean Water Act (CWA) addresses dredging and disposal operations in inland waters. There is a pending rule change that will make program operations and use of reference sediments consistent for both ocean and inland waters. This discussion, however, will focus on the existing requirements in the ocean dumping regulations.

The Green Book

The testing manual for Section 103 of the MPRSA is commonly referred to as the Green Book (USEPA/USACE, 1991). It was first developed in 1977 and later revised in 1991 by a joint USEPA/USACE technical committee. The Green Book provides specific guidance for conducting the evaluative protocols used to assess the suitability of dredged materials for open water disposal. It also defines test materials, reference sediments, and control treatments, and specifies details on test methodologies and provides guidance for the interpretation of data generated by these tests. Much of the remaining discussion will be based on material from this manual.

Reference Sediment Approach

In 1976 the ocean disposal regulatory program adopted the reference sediment comparison approach because at that time it was considered the most direct and pragmatic way to satisfy the requirements of the MPRSA. The statutory language of the MPRSA prohibits dumping of dredged materials containing certain constituents as other than trace contaminants unless they are "rapidly rendered harmless." This regulatory language comes directly from the London Convention which represents international agreements on ocean disposal activities. The reference comparison approach provides a mechanism that allows regulators to make decisions that take into account existing background or baseline conditions in the vicinity of the disposal site as defined by the specific test endpoints selected. The rationale as originally applied viewed dredging and disposal activities not as remediation activities aimed specifically at improving existing condition, but as activities that at a minimum must be conducted in a manner that would not further degrade environmental conditions at the disposal site.

Evaluation of benthic impacts for the dredged material program employs sediment toxicity tests and bioaccumulation tests (Green Book). The benthic sediment toxicity tests consist of 10-day whole sediment exposures conducted with appropriately sensitive species and focuses on mortality as the test endpoint. The standard bioaccumulation test involves a 28-day exposure with tissue residue contaminant levels in selected biota as the test endpoint.

The reference sediment is the key to making regulatory decisions concerning dredged material disposal (Figure 1). Results from benthic toxicity and bioaccumulation tests using representative biota and reference sediments are compared to results from tests using dredged sediments (test material). These comparisons serve as the basis for determining the potential for adverse ecological impacts resulting from dredged material disposal at the environs of the disposal site.

Initiating the Dredging Process

The dredging process begins with identifying the need for dredging a navigation channel. The decision on whether or not to dredge is based primarily on economic and societal issues. Navigation channels must be maintained to support marine transportation, which is the primary mode for global trade, currently estimated at about three trillion dollars annually. The management of dredged material, however, is an environmental issue that is further defined by applying the federal standard which stipulates that disposal occur according to the least cost option that is environmentally appropriate. Once the need to dredge is determined, a disposal site is identified (USEPA, 1990).

The next step in the process is to identify and collect appropriate reference sediment. There are two approaches for identifying the reference material. One approach is to physically locate a reference site, define its location and boundaries, and collect sediments from the site. The other approach, referred to as the reference area approach, involves collecting sediment samples using a grid overlay which provides a composite of sediments representative of the disposal site. The reference material represents the existing benthic environmental conditions in the vicinity of the disposal site. The reference material must satisfy the definition in the Green Book, particularly with respect to three critical requirements (Figure 2). Reference sediment should be: (1) substantially free of contaminants; (2) as similar as possible to the grain size of the sediment in the dredged materials and at the disposal site; and (3) reflective of conditions that would exist at the disposal site if no dredged material disposal had taken place and that represent all other influences on sediments in the area. The test material (dredged material) is then collected in a manner representative of the sediments from the reach and depth of the project area to be dredged. Finally a control sediment is identified which is free of contaminants and known to satisfy the physical requirements (i.e. grain size) of the test organism. This material serves as the laboratory control treatment.

Program Testing

As stated above, testing consists of conducting benthic toxicity and bioaccumulation tests on the reference material and the test dredged material with appropriate laboratory controls and comparing the results of each test. Interpretation of the benthic toxicity test data is fairly straightforward. Dredged material is predicted to be acutely toxic to benthic organisms when mean test organism mortality is statistically greater than the reference sediment and exceeds mortality in the reference sediment by at least 10 percent. Statistical differences between test and reference treatments can be greater than 10 percent depending on the inherent variability measured in specific test species (e.g., a 20 percent value for mortality can be used for amphipods).

Interpretation of the benthic bioaccumulation test results is more complicated. Comparisons between the test material and the reference treatments do not provide pass/fail criteria, but rather serve as a screen for a more comprehensive analysis. Results are compared initially to determine whether there are statistically significant differences in contaminant residue levels between the test treatments and the reference treatment. This comparison indicates that contaminants are present in greater than trace quantities with the potential for bioaccumulation and possible trophic transfer. A more comprehensive analysis involves comparisons of tissue residue levels to FDA action levels and consideration of the toxicological properties of the contaminants of concern. If tissue concentrations of one or more contaminants of concern are statistically greater than the FDA action levels, then the test dredged material is predicted to result in benthic bioaccumulation of contaminants. If tissue concentrations are either statistically less than FDA action levels or there are no FDA action levels for the contaminants of concern, the information is considered insufficient to reach a conclusion about the potential for bioaccumulation. In these cases further evaluation, including assessment of additional factors such as the magnitude of bioaccumulation relative to biota indigenous to the disposal site, may be necessary to address the potential for biomagnification and trophic transfer and associated health risks.

The test endpoints (percent mortality and level of contaminant bioaccumulation), as measured in laboratory tests with reference sediment, serve as de facto standards in making regulatory decisions regarding the disposal of dredged material. Obviously, the selection of appropriate reference sediment is of paramount importance if the goals of environmental protection are to be realized.

Reference Site Selection

Current practice in selecting a reference site inevitably requires some degree of compromise to meet the somewhat ambiguous requirements of a reference material (i.e., "substantially free" of contaminants, yet "as similar as possible" to the dredged material and disposal site sediments, and reflective of conditions "had no disposal occurred"). In some areas of the country, groups responsible for dredging and dredged material management are exploring ways to insure consistency and suitability of reference materials selected for dredged material evaluation. This is a function, in part, of the open process employed by the USACE and USEPA to solicit and promote involvement of the states, the local communities,

and stakeholders in establishing what they believe represents environmentally appropriate and acceptable reference material for their localities.

A review of data generated by benthic toxicity and bioaccumulation tests in reference sediments from major dredging projects in New York, Boston, San Francisco, and Puget Sound (personal communication with regional ocean disposal coordinators) indicate that the reference sediments employed in these areas reasonably reflect baseline conditions characteristic of minimally impacted environments Amphipod survival in the 10-day acute toxicity tests averaged around 90 percent in the first three areas (Figure 3). Survival was slightly lower in Puget Sound where a more regionally integrated approach is applied. Tissue residue data from bioaccumulation tests conducted on reference material in all four areas using the polychaete *Nereis virens* and the clam *Macoma nasuta* for polychlorinated biphenyls (PCBs), cadmium, and mercury are shown in Figure 4. PCB tissue concentrations measured 20 to 30 ppb, and mercury and cadmium tissue concentrations measured less than 0.05 ppm on a wet weight basis. These concentrations also approximate current background levels typically measured in minimally impacted aquatic systems (USEPA, 1997). These data support the contention that selection of reference sediments in these major U.S. ports are serving their intended purpose of providing appropriate levels of environmental protection for the dredged material disposal program. However, given the subjectivity inherent in the reference sediment selection process, I believe establishment of national criteria for identification of appropriate reference material would further promote greater consistency in regulatory decision making, and perhaps enhance the margin of environmental protection provided by the current regulatory protocol.

Future Needs

Criteria which define the minimum requirements for reference material selection that will maintain environmental integrity and sustainability at the environs of the disposal site should be developed as additional guidance for the dredged material evaluation process. An example of where this is already happening is Puget Sound where regulators have developed an extensive database that accurately represents background conditions. This database provides the basis for selecting reference materials and establishing statistically based criteria used for dredged material evaluation. Although development of such databases is expensive and time consuming, this approach could serve as a model for insuring appropriate reference standards are employed for dredged material evaluation.

Additional research needs include improving our ability to predict community, population and ecosystem level impacts from existing effects test endpoints (i.e., acute toxicity), and development of tissue residue-effects linkages for contaminants of concern (Figure 5). What does a whole-body residue concentration mean in terms of the health of the organism, its ability to reproduce, and the propensity for a contaminant to biomagnify? Sufficient data to answer these questions currently do not exist. Predictive models that elucidate residue effects linkages along with the data to validate these modeling efforts should be developed. Efforts to develop tissue residue databases must also continue to ensure effective interpretation of bioaccumulation data.

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Figure 1.

How is the reference sediment approach used to make decisions?

- Potential adverse impacts are determined on the basis of comparison of test endpoints (benthic toxicity and bioaccumulation) between test and reference treatments

---

Figure 2.

Reference Sediment: Definition

- *Substantially free* of contaminants
- *As similar as possible* to the grain size of the dredged material and sediment at the disposal site
- *Reflects conditions at the disposal site that would exist had no dredged material disposal taken place; reflects all other influence on sediments in the area*

Figure 3.

Amphipod Survival in Reference Sediments from Major U.S. Ports

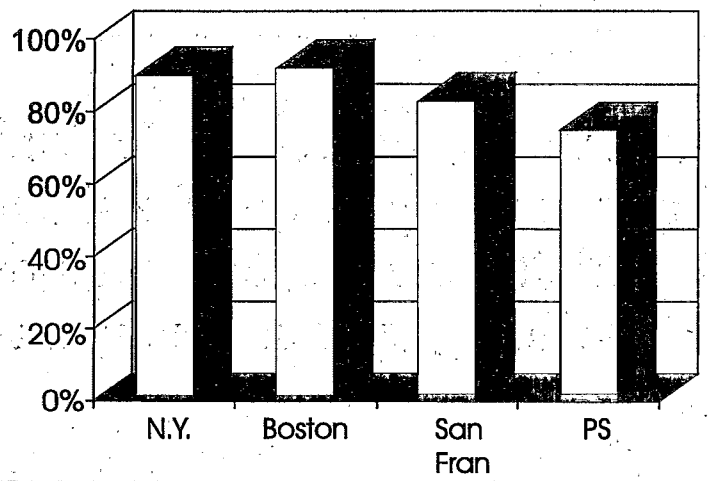

Figure 4.

Bioaccumulation in Reference Sediment (mg/kg wet wt.)

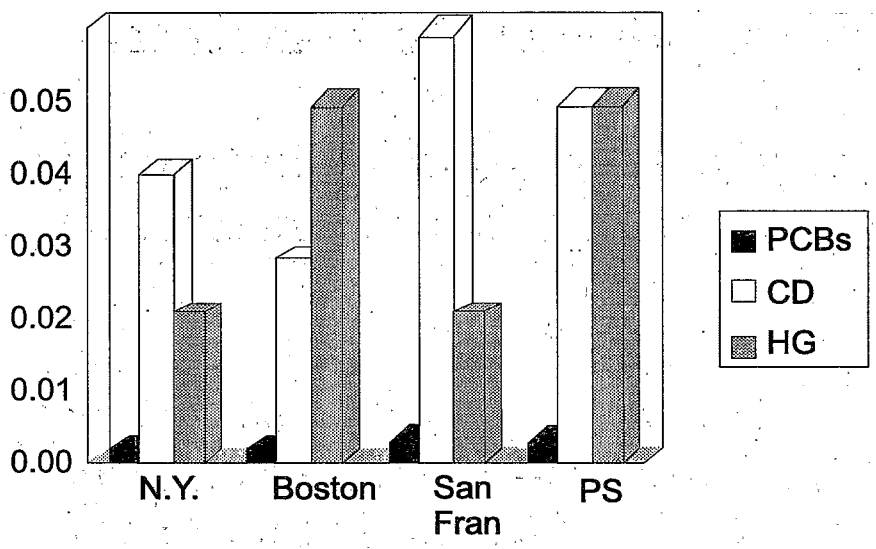

Figure 5.

Reference Approach Suggested Improvements

- Establish minimum criteria for reference site selection that insures environmental integrity and sustainability are preserved (i.e., background database approach used in Puget Sound)

- Develop extrapolation models for single species toxicity endpoints that provide meaningful ecological interpretation at the community and population levels

- Elucidate residue-effects relationships to provide meaningful interpretation of bioaccumulation data.

Development of Tissue Residue Threshold Values

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Background

A chemical residue-based approach for evaluating dose, also called critical body residue (CBR) or lethal body burden (LBB), has been advocated as an improvement for prediction of toxicity to organisms in the environment (Friant and Henry, 1985; McCarty, 1986; Cook et al., 1987; Van Hoogen and Opperhuizen, 1988; Cook et al., 1991; McCarty, 1991; McCarty et al., 1991; Tas et al., 1991; Landrum et al., 1992; and McCarty and MacKay, 1993). The correlation of body residues to toxic effects (residue-based dose) has a number of advantages over using an exposure-based approach (i.e., water or food concentrations that cause toxic effects). As outlined by McCarty and MacKay (1993), these advantages include the following: (1) bioavailability is explicitly considered; (2) accumulation kinetics are considered, which reduces the confounding effect of exposure duration when interpreting results; (3) uptake from food (as distinct from water) is explicitly considered; (4) toxic potencies are expressed in a less ambiguous manner, facilitating identification and investigation of different modes of toxic action; (5) effects of metabolism on accumulation are considered; (6) mixture toxicity can be more readily assessed; and (7) experimental verification can be more readily determined between laboratory and field.

Bioaccumulation testing is being used increasingly in various environmental monitoring and regulatory programs involving sediments. In these tests, sediment organisms are exposed to sediment samples for a prescribed time period (e.g., 28 days). Following this uptake period, exposed organisms are analyzed for chemicals of interest. While these tests are one mechanism for assessing the bioavailability and accumulation of sediment contaminants, they do not intrinsically predict the toxicological effects of bioaccumulative toxicants. For this prediction, some association between tissue residues and toxicological effects must be developed. Thus, bioaccumulation tests are a natural application for residue-based effects assessment.

To help evaluate the basis for, and applicability of, residue-based effects assessment, we have undertaken the development of a comprehensive database containing literature data on tissue concentrations of toxicants and associated biological effects for aquatic animals. The purpose of this presentation is to describe the database and provide some examples of analyses that can be conducted from these data.

Database Content and Development

Pertinent literature was identified through several search mechanisms, including electronic databases (e.g., POLTOX I®; Cambridge Scientific Abstracts), in-house literature files, Current Contents®, and other assorted sources. For all literature, hard copies of the primary literature were obtained and are maintained in the project files.

From this literature, residue/effect information was manually extracted. General inclusion criteria were:

- Organism was a marine or freshwater fish, invertebrate, or aquatic lifestage of amphibian (terrestrial animals, birds, and plants were not included);
- There was a *measured* chemical concentration in the whole body or in a specific tissue; and
- There was some observation of biological effect in the form of survival, growth, or reproduction (physiological and biochemical endpoints were not considered).

In general, only data from exposures using a single chemical were used; information from mixture studies was not used unless the mixture contained only related chemicals (same mode of action). Control treatments were required as a basis for comparison of biological effect, except in studies where survival was ≥90 percent (thus survival was not reduced). All chemical types (e.g., organic and inorganic, ionic and nonionic) were included.

For references meeting these criteria, specific information was extracted for inclusion in the database. Database fields are as follows:

- **Study Type**: acute or chronic
- **Chemical Name**: exact chemical form (e.g., metal salt) is included parenthetically
- **CAS Number**

- **Log K_{ow}**
- **Molecular Weight**
- **Species**
- **Life Stage:** life stage exposed, or range if multiple life stages were exposed
- **Lab/Field:** whether exposures were conducted in the laboratory or in the field
- **Test Conditions:** static, static-renewal, flow-through, microcosm, mesocosm, etc.
- **Exposure Route:** water, sediment, diet, injection, maternal
- **Exposure Concentration:** measured if given, otherwise nominal
- **Test Duration:** in days
- **Tissue Analyzed:** whole body, soft parts, blood, carcass, organ(s)
- **Tissue Residue:** in µg/g wwt and µmol/wwt
- **Biological Response:** survival, growth, or reproduction; no effect observations included
- **Reference:** literature source
- **Comments:** flags on unusual conditions (e.g., control performance problems, discrepancies in the data, etc.)

As of September 1996, the database contained residue data from more than 480 literature sources, spanning 237 chemicals, and resulting in approximately 3,000 individual residue/effect pairs. Chemicals with the greatest number of residue/effect pairs at this time are:

- **More than 400 residue/effect pairs**
  cadmium
- **100 to 250 residue/effect pairs**
  DDT, TCDD, hexachlorobenzene, mercury, PCB(s), selenium
- **40 to 100 residue/effect pairs**
  aminocarb, arsenic, copper, 2,4 dinitrophenol, endosulfan, endrin, fenvalerate, kepone, lead, lindane, nickel, 4-nitrophenol, pentachlorophenol, terbufos, toxaphene, tributyltin, zinc

Example Data Sets

Compilation and analysis of the data are ongoing at this time. However, as an example of data analyses than can be performed using the database, we extracted data for chlorpyrifos (Jarvinen et al., 1983; Macek et al., 1972; Serrano et al., 1995; Hansen et al., 1986; Montanes et al., 1995) and kepone (Buckler et al., 1981; Hansen et al., 1977a, 1977b; Fisher and Clark, 1990; Sanders et al., 1981; Stehlik and Merriner, 1983; Fisher et al., 1983; Fisher et al., 1986; Goodman et al., 1982).

Figures 1 and 2 display the residue/effect pairs for chlorpyrifos and kepone, segregated by biological endpoint (survival, growth, or reproduction). Although all residues reported for these chemicals were for whole-body analyses, the exposure regimes varied widely with regard to species, lab versus field, and route of exposure, among other variables. Regardless of these differences, these values do suggest a range of chemical residues associated with biological effects, with the threshold for reported effects in the vicinity of 1 µg/g wwt for both chemicals.

Once data entry, accuracy checking, and initial analysis are complete, it is our intention to make this database available to the scientific community for further analysis.

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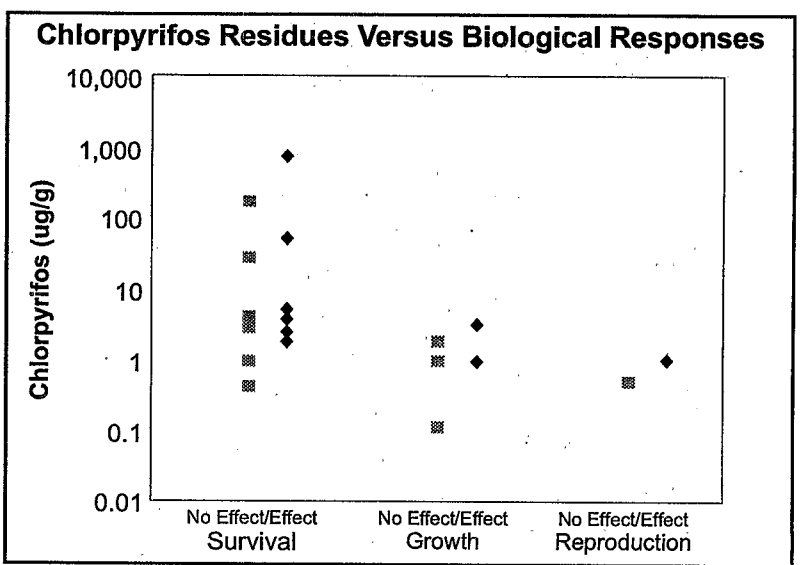

Figure 1. No effect (squares) and effect (diamonds) concentrations for chlorpyrifos in tissues.

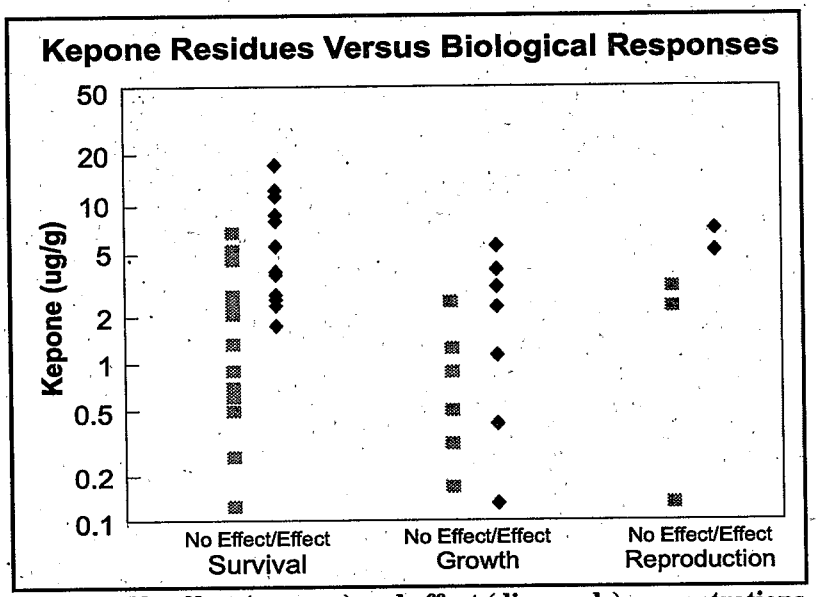

Figure 2. No effect (squares) and effect (diamonds) concentrations for kepone in tissues.

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Development of Tissue Residue Threshold Values

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Impetus for Research

- Need for interpretive guidance for bioaccumulation testing
- Desire for decision criteria that are based on biological effects

Impetus for Research

- Proposal that risk assessment be based on tissue residues rather than concentrations in environmental matrices
- Assemble data necessary to evaluate a tissue residue approach
- Evaluate tissue residue approach relative to mode of action or other characteristics

Tissue Residue/Toxicity Database

- Exhaustive search of literature for residue data linked to biological effect observations
- Selection criteria designed to maximize quality, comparability, and interpretability of resulting data

Scope of Data Collection

- marine and freshwater, fish and invertebrates
  - does not include amphibians, terrestrial vertebrates or birds
- endpoints focused on survival, growth, and reproduction
  - histological/biochemical/physiological endpoints not included
- virtually all chemicals included, regardless of mode of action

Database Fields

- Acute/chronic
- Chemical name
- CAS number
- Log Kow
- Molecular weight
- Species
- Life stage
- Lab/field
- Test conditions
- Exposure route
- Exposure concentration
- Test duration
- Tissue analyzed
- Residue
- Effect
- Reference
- Comments

Criteria for Data Inclusion

- Measured tissue residue (whole body or specific tissue)
- Effect data or statement concerning the health of the test organisms
- Mixture papers used only if no effect was observed
- Control not necessary if no mortality was observed

Database Content

- Currently, the database contains approximately:
  - 485 references
  - 200 chemicals
  - 2,552 residue/effect pairs

Largest Datasets

- More than 400
  - cadmium
- 100-250
  - DDT
  - TCDD
  - hexachlorobenzene
  - mercury
  - PCB(s)
  - selenium
- 40 to 100
  - aminocarb
  - arsenic
  - copper
  - 2,4 dinitrophenol
  - endosulfan
  - endrin
  - fenvalerate
  - kepone
  - lead
  - lindane
  - nickel
  - 4-nitrophenol
  - pentachlorophenol
  - terbofos
  - toxaphene
  - tributyltin
  - zinc

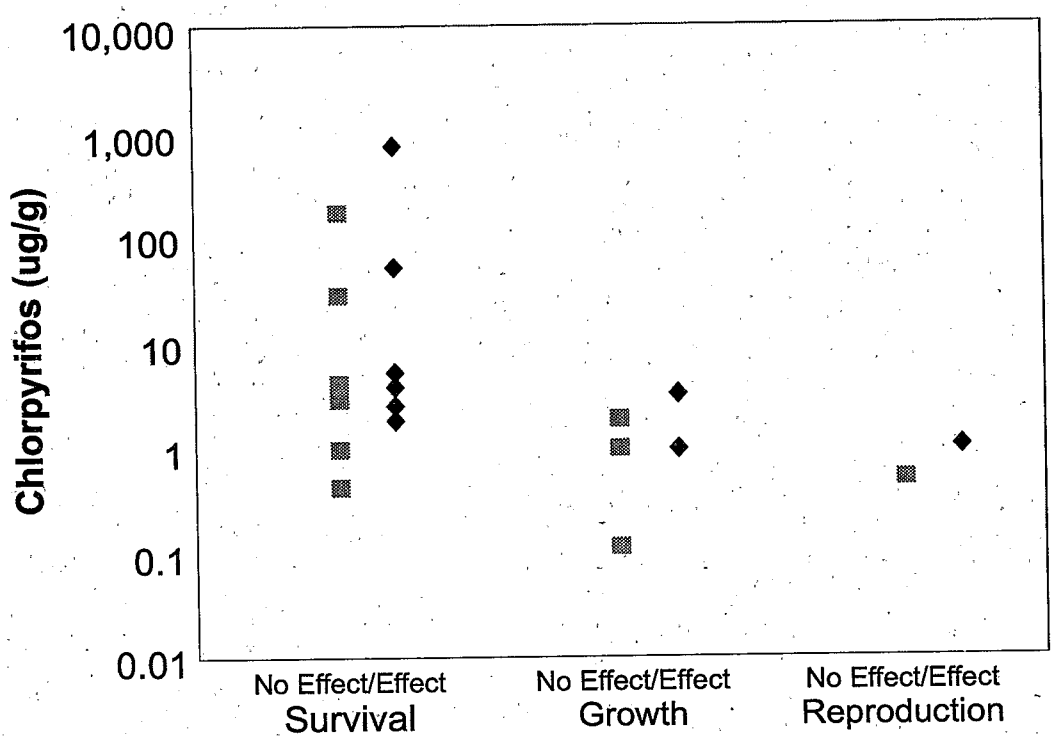
Kepone Data

- 32 residue/effect pairs
  - 8 references
  - 6 species (3 fish, 3 invertebrate)
  - lab exposures only
  - water, diet, parental exposures
  - exposures 4 to 141 days

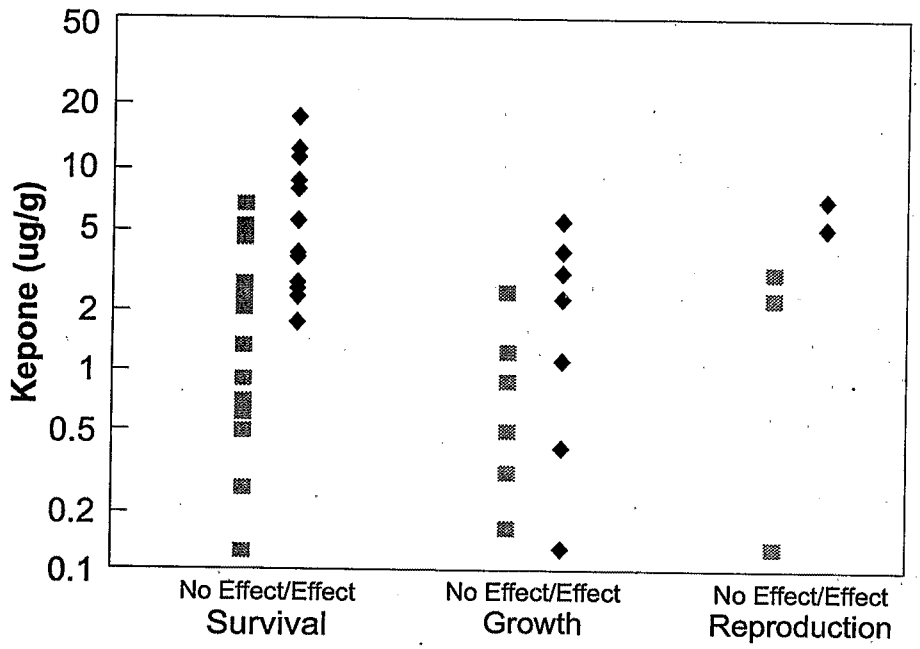
Chlorpyrifos Data

Species (n)	Age	Days	S/G/R	Source
Fathead minnow (6)	Larva	Lab 200	S,G,R	Jarvinen et al. 1983
Bluegill (2)	Juvenile	Field 63	S	Macek et al. 1972
Largemouth Bass (2)	Juvenile	Field 63	S	Macek et al. 1972
Mussel (*Mytilus*) (2)	Adult	Lab 4	S	Serrano et al. 1995
Mussel (*Venus*) (1)	Adult	Lab 4	S	Serrano et al. 1995
Gulf toadfish (4)	ELS	Lab 49	S,G	Hansen et al. 1986
Isopod (*Asellus*) (3)	Adult	Field 23	S,G	Montanes & Hattum 1995

Chlorpyrifos Residues Versus Biological Responses

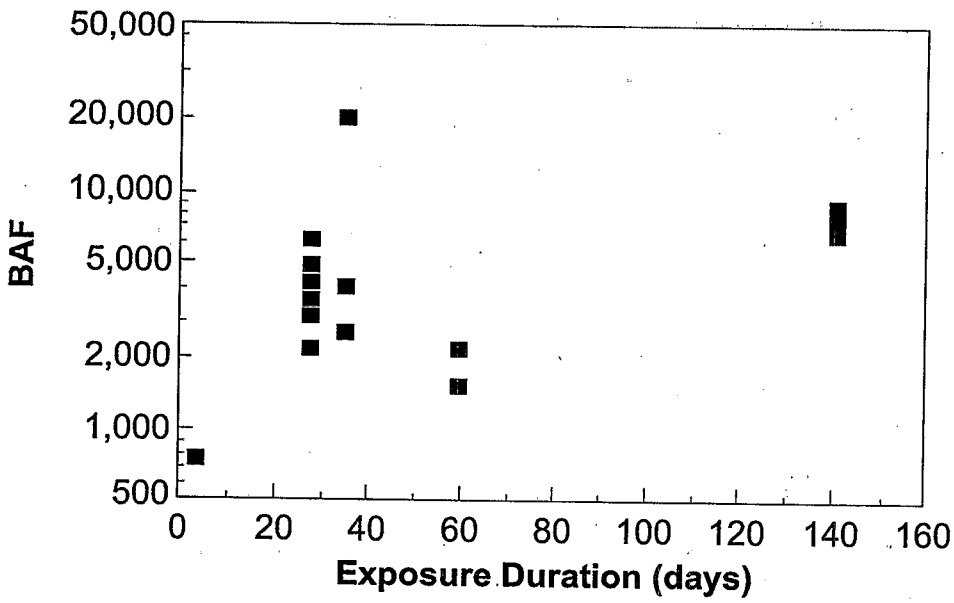


Kepone Residues Versus Biological Responses



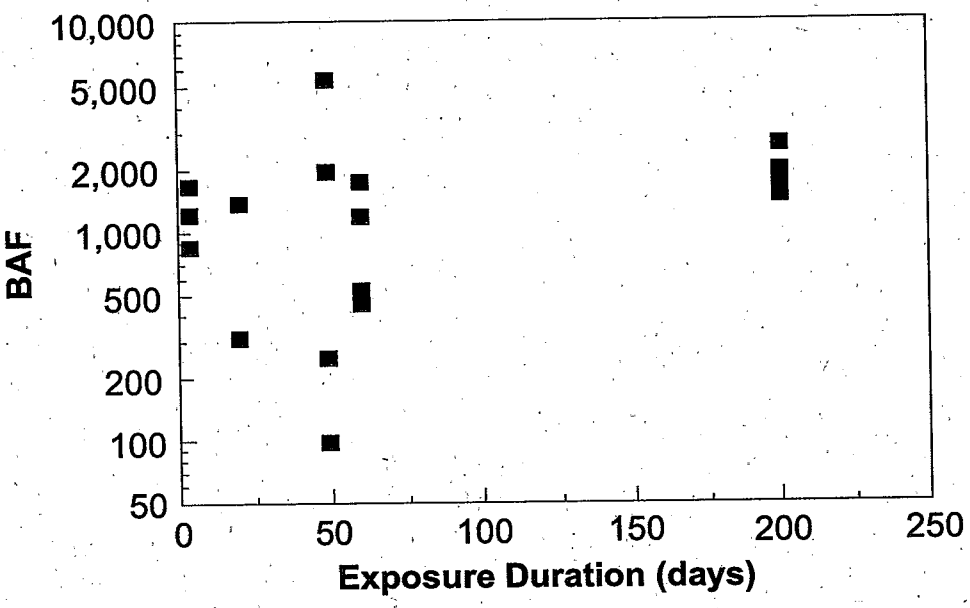
BAF versus Exposure Duration

Kepone (water exposures)



BAF versus Exposure Duration

Chlorpyrifos (water exposures)



Paired Effect/No Effect Data

Chlorpyrifos

	Effect	No Effect	Geo. Mean
Survival			
Gulf toadfish	770	175	367
Mussel (*Mytilus*)	53	4	14.6
Fathead minnow	5.11	3.03	3.93
Isopod (*Asellus*)	1.79	0.97	1.32
Bluegill	3.82	0.42	1.27
Largemouth Bass	2.55	0.47	1.09
Growth			
Fathead minnow	3.03	0.95	1.70
Gulf toadfish	0.95	0.14	0.36
Reproduction			
Fathead minnow	0.95	0.47	0.67

Paired Effect/No Effect Data

Kepone

	Effect	No Effect	Geo. Mean
Survival			
Sheepshead minnow	11	4.7	7.19
Fathead minnow	3.8	2.6	3.14
Sheepshead minnow	2.3	1.3	1.73
Sheepshead minnow	2.5	0.9	1.50
Spot	2.7	0.7	1.37
Fathead minnow	1.7	0.26	0.66
Growth			
Fathead minnow	3.8	2.6	3.14
Sheepshead minnow	2.2	1.2	1.62
Sheepshead minnow	0.41	0.30	0.35

Interpretation Issues

- Quantity and type of data varies greatly between chemicals
- Target tissue data not available consistently
- Relatively few data for individual PAH
- PCB mixtures vs. single congeners

Use of Tissue Residue Data in Exposure and Effects Assessments for Aquatic Organisms

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Under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, or Superfund), the National Oceanic and Atmospheric Administration (NOAA) is responsible for protecting and restoring natural resources that may be injured by releases from Superfund sites in coastal areas. As part of NOAA's role of providing EPA with technical support in the evaluation of natural resource concerns at hazardous waste sites, NOAA's Coastal Resources Coordination Branch often recommends collecting tissue residue data.

Tissue residue data can play an important role at different stages in the process of evaluating hazardous waste sites, including ecological risk assessment, modeling conducted to evaluate different remedial alternatives, and the monitoring necessary to determine the effectiveness of remediation. An ecological risk assessment is a major component of hazardous waste site evaluations. For an ecological risk assessment to provide useful information for remedy selection, exposure should be related both to the source of contamination and to an important toxicity endpoint. This means that it is necessary to link any estimated or observed adverse effects to site-related contamination, which emphasizes the importance of identifying and quantifying major exposure pathways. Because bioaccumulation is often more of a concern for higher-trophic level organisms, which may receive much of their contaminant exposure via the food web, any effort to develop bioaccumulation-based criteria should identify and address significant, sensitive toxicity endpoints in fish and other higher-trophic level organisms. For example, reproductive or early life stage toxicity in fish is both a sensitive toxicity endpoint and an endpoint with potentially significant implications for the health of a fish population.

The appropriate application of tissue residue data depends upon the type of contaminant under consideration. For some contaminants, tissue residue concentrations may be directly linked to toxicity endpoints. Also, the concentrations of contaminants in the tissues of aquatic organisms often provide a direct measure of bioavailability that integrates exposure over time and integrates exposure from water, sediment, and food web pathways. In this presentation, I will be focusing primarily on the use of field-collected tissue residue data in exposure and effects assessments for aquatic ecological risk assessment. Because of the importance of bioaccumulation for higher-trophic level organisms, I will focus my discussion on the use of tissue residue data in assessing toxicity and exposure in fish. Addressing these issues is particularly important since currently available criteria for sediment explicitly exclude any consideration of contaminant exposure through the food web. "These [sediment] criteria do not address the question of possible contamination of upper trophic level organisms..." (USEPA, 1993a). And, "...a site-specific investigation appears to be the only available method for performing an evaluation of the effect of contaminated sediments on the body burdens of upper-trophic level organisms" (USEPA, 1993b).

Aquatic risk assessment requires information on the bioavailable fraction of the contaminants in the system. Accurate and meaningful assessments of exposure from water and sediment are difficult due to a potentially high degree of temporal and/or spatial variability. Because of high bioconcentration factors, particularly for hydrophobic organic contaminants, very low analytical detection limits are required to ensure that concentrations below analytical detection limit values could not contribute to significant exposure concentrations. Determining the bioavailable contaminant fraction is also complicated by partitioning suspended particulates and dissolved organic carbon. Modeling food web exposure in fish requires making many assumptions for which there is frequently little site-specific or reliable information available. Collection of site-specific tissue residue concentrations can be used to reduce some of this uncertainty in the application of the exposure models used in the assessment. In

many cases, tissue residue data can be used to provide time-integrated and location-specific information on bioavailable contaminant concentrations.

Metals

The bioaccumulation of metals in benthic macroinvertebrates can provide a useful measure of the extent and magnitude of contamination that temporally integrates exposure via the water column and sediment. Because benthic invertebrates represent an important source of food for fish, the accumulation of metals in invertebrates, which may have some tolerance to chronic exposure to metals, may also serve as a significant source of exposure to fish. Exposure via the food web may be particularly important for fish early life stages, which are also very sensitive to waterborne exposure to metals.

The combined effects of reduced survival and growth in fish early life history stages, which may have important implications for adverse effects on the health of a local fish population, is one of the most sensitive toxicity endpoints for waterborne exposure to a variety of metals (such as cadmium, copper, lead, mercury, and zinc). Laboratory studies on the effects of metals contamination on young of the year fish in the Clark Fork River in Montana demonstrated that feeding metals-contaminated invertebrates (at concentrations comparable to those found in the river) resulted in reduced growth and survival in rainbow trout (Woodward et al., 1994) and reduced growth and physiological abnormalities in rainbow trout and brown trout (Woodward et al., 1995). These studies also indicated that the exposure via dietary intake could be at least as important as aqueous exposure in age-0 fish. Another study, however, found no effects on survival or growth in age-0 rainbow trout that were fed metals-contaminated brine shrimp at similar concentrations (Mount et al., 1994). Additional studies are needed to more clearly resolve the relative importance of additional dietary exposure and to develop a method for determining the total effective dose that includes all exposure pathways.

In conducting a risk assessment for metals-contaminated sites, tissue residue concentrations in invertebrate prey organisms may reduce the uncertainty in estimating exposure via the water column and sediment (by integrating exposure over time at specific locations) as well as also providing information on exposure via the food web. This may be particularly important in assessments involving vulnerable receptor populations (for example, the early life history stages of threatened and endangered salmonid populations).

Polycyclic Aromatic Hydrocarbons (PAHs)

Because fish rapidly metabolize and excrete PAHs, fish tissue residue concentrations of parent PAH compounds do not provide a useful measure of exposure to fish (Varanasi et al., 1989). However, exposure to PAHs has been linked to reproductive impairment, immune dysfunction, increased incidence of liver lesions, and other histopathological endpoints in fish (Malins et al., 1987; Johnson et al., 1988; Varanasi et al., 1992). High concentrations of PAHs in stomach contents of fish from a PAH-contaminated harbor indicate that the consumption of contaminated benthic invertebrates can be an important exposure pathway (Malins et al., 1985).

Measuring tissue concentrations in invertebrates that have minimal capacity for metabolism of PAHs, such as bivalve shellfish, can provide location-specific and temporally integrated information on bioavailable PAH concentrations, which can be used in food web models of exposure to fish and other higher trophic level organisms. Using tissue residue concentrations from caged bivalves allows for more control of specific location and other variables such as organism size or age. Other methods for determining the relative differences in PAH exposure include measuring concentrations of PAH metabolites in fish bile (Krahn et al., 1984, 1986), which can provide useful information on recent exposure to PAHs, and semipermeable membrane devices (SPMDs), which provide temporally integrated measurements of dissolved and presumably bioavailable PAHs in the water column at specific locations (Lebo et al., 1992; Huckins et al., 1996). The patterns of PAH concentrations in field-collected or caged bivalves and SPMDs can also play an important role in linking exposure to the source of PAHs using various fingerprinting methods.

Polychlorinated Biphenyls (PCBs)

Fish tissue residue PCB concentrations, unlike PAHs, can provide useful information on both exposure and toxicity. Reproductive and early life stage effects (e.g., reduced larval growth or survival), where the primary exposure to the developing larvae results from the maternal transfer of accumulated PCBs to the eggs, is one of the most sensitive toxicity endpoints for PCB effects in fish. Consequently, PCB concentrations in the ovaries of mature female fish immediately prior to spawning may be the most useful measurement for estimating potential reproductive effects in species of concern. Higher-trophic level species, which tend to accumulate the highest concentrations of PCBs, may be at the greatest risk.

Field-collected tissue residue PCB concentrations in fish can play an integral role in exposure assessments, since they provide a measure of exposure via all exposure pathways. This is particularly important since fish generally receive much of their PCB exposure via the food web. The composition of the PCBs found in environmental samples often differs substantially from the original PCB source (e.g., commercial Aroclor mixtures). This difference results from physicochemical and biological processes that differentially affect individual PCB congeners (Safe et al., 1987). For example, the solubility and volatility of individual PCB congeners range over several orders of magnitude and greatly affect partitioning between sediment-water and water-air interfaces. Thus, less chlorinated congeners tend to be more soluble in water and are more quickly removed from sediment into water.

Biological processes also alter the pattern of congener distributions in tissue residue samples. Differences in uptake, depuration, metabolism, and efficiency of assimilation among individual congeners may result in large differences between the PCB mixtures in samples and in commercial Aroclor mixtures, and these differences would be expected to increase with increasing trophic level.

Because of the major differences between the PCB congener distributions in environmental samples and in commercial mixtures, analysis methods based on Aroclor standards can provide only a semi-quantitative estimate of total PCBs (Safe et al., 1987). Modeling PCB behavior in the environment, monitoring trends in PCB concentrations, and relating fish tissue PCB residues to their sources can be considerably enhanced by congener-specific analysis. Samples of several species of fish collected from multiple locations over 170 miles of the Hudson River were analyzed for PCB congeners as part of the ecological risk assessment for a Superfund Remedial Investigation. Samples of the same species at a given location had very similar congener compositions, although their total PCB concentrations sometimes varied by a factor of 2 or more on either a wet weight or lipid-normalized basis. The PCB congener compositions and concentrations in fish tissue are being used in the ecological risk assessment to help clarify the relationship between the PCB sources in the river and the fish body burdens.

Two general screening approaches are available to evaluate the potential toxicity of PCBs to fish based on PCB body burdens: comparison to total PCB concentrations associated with adverse effects, or estimation of dioxin toxic equivalent concentrations and comparison to No Observable Adverse Effect Levels (NOAELs) for 2,3,7,8-tetrachloro-dibenzo-p-dioxin (TCDD). A variety of field and laboratory studies provide evidence of reproductive and other adverse effects in different fish species at low total PCB tissue residue concentrations. The total PCB concentrations in fish tissue residue samples associated with adverse effects range over several orders of magnitude, which may reflect the differences in toxicity among different PCB mixtures and/or a high degree of inter-species difference in sensitivity to the effects of PCBs. The combination of these factors results in considerable uncertainty in estimating the risk of reproductive toxicity to selected fish species based on total PCB tissue residue concentrations.

The coplanar PCBs—a group of PCB congeners that is similar in structure and biological activity to the highly toxic polychlorinated dioxins and dibenzofurans—are considered to be the most toxic PCBs (Safe, 1984). Although most of these coplanar PCBs are found at relatively low concentrations in the commercial PCB mixtures, several have been identified as important components of PCB tissue residues in aquatic biota and may be preferentially accumulated, particularly by higher-trophic level organisms (Safe, 1984; Hansen, 1987; Smith et al., 1990; Schwartz et al., 1993). Recent studies suggest that only the non-ortho substituted PCB congeners exhibit dioxin-like activity in fish (Walker and Peterson, 1991; Newsted et al., 1995; Zabel et al., 1995). The early life stage toxicity of these individual PCB congeners has been investigated in trout and compared to the toxicity of dioxin to develop fish-specific toxic equivalent factors, or TEFs (Walker and Peterson, 1991; Zabel et al., 1995). The TEFs for the non-ortho congeners calculated for trout are 1 to 2 orders of magnitude less than the TEFs determined for mammalian systems. Thus, using mammalian-based TEFs for PCBs may overestimate the potential for dioxin-like reproductive toxicity in fish.

Conclusions

Tissue residue data can play an important role in aquatic ecological risk assessments at hazardous waste sites, in part by linking exposure to the source of contamination and to important toxicity endpoints.

The development of bioaccumulation-based criteria should identify and address toxicity endpoints in fish that are both sensitive and ecologically significant. This is particularly important for higher-trophic level species, which may receive much of their contaminant exposure via the food web.

For some contaminants, such as PAHs and possibly metals, dose-response models that include all exposure pathways need to be developed. Tissue residue concentrations in macroinvertebrates may provide key input into exposure models for these types of contaminants.

Tissue residue PCB concentrations in fish have important applications to both exposure and toxicity assessments. Because the PCB congener distributions in environmental samples often differ considerably from the original commercial PCB mixtures, congener-specific PCB analysis of fish tissue samples provides more useful information on exposure than a standard Aroclor analysis. PCB concentrations in the ovaries of mature female fish immediately prior to spawning may be the most useful measurement for estimating potential reproductive toxicity in fish species of concern.

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Comments on the Significance and Use of Tissue Residues in Sediment Toxicology and Risk Assessment

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Introduction

The explicit use of body residues in ecotoxicology and risk assessment is relatively new and is currently hampered by the limited availability of residue-effect data. However, the body-residue-based approach is theoretically sound and ultimately lies at the heart of existing environmental-media-based methodologies. It is expected that residue-effect methodologies will soon supplant toxicity-only and bioaccumulation-only based procedures. In the interim, methods to combine and exploit existing toxicity and bioaccumulation knowledge are being developed and refined. However, there is still an opportunity to examine and direct how residue-based approaches may be best employed. Five topics are reviewed and recommendations offered.

What are the assumptions, applications, and limitations for each bioaccumulation or risk assessment methodology being described?

All bioaccumulation-related approaches make the assumption that bottom-up extrapolation is valid and has been validated. That is to say that results and information obtained in laboratory or controlled testing can be readily generalized and extrapolated to population/community/ecosystem levels of organization. This is essentially a matter of faith or at best a policy decision. Such approaches may be useful and/or effective in protecting the environment, but they are not firmly based in science since most such ecological organization theories have not been validated.

For example, the conventional levels of a biological organization model are presented by Odum (1971). However, the practical ability to exploit this model for environmental protection and environmental risk assessment is not universally accepted. Consider Fry's Paradigm, which addresses it: "You take the properties of a level of organization and use those observations to analyze the next level of organization below it. If you take the properties too many steps down, you're being stupid; and you cannot go the other direction." (Kerr, 1976)

Fry's Paradigm clearly indicates that the very essence of much of the current risk assessment process—extrapolation from laboratory testing at the level of the individual organism (and below) to effects in the field at the population, community, and ecosystem levels of organization—is unwise if not impossible. Furthermore, it should be pointed out that the model presented by Odum is not the only model for looking at how ecology is organized. In fact, there is no single, generally accepted ecosystem paradigm (Botkin, 1990 as cited by Suter, 1993). Less than half of about two dozen ecosystem concepts proposed by ecologists have been employed by risk managers/assessors and only about one-third are used with any regularity (Vigerstad, manuscript).

Munkittrick and McCarty (1995) examined the uneasy relationship between ecology and toxicology known as ecotoxicology. They point out that such conceptual models of environmental impacts, although considering direct and indirect factors, do not usually consider what they term nondirect factors, or induced factors as they are called in other disciplines. Nondirect factors are those which do not originate with a response to a chemical stressor and cannot be expressed in terms of a toxicological dose. For example, what is the dose metric of loss of half of the habitat of a species and how is it quantitatively combined with the direct and indirect effects of toxicant stress?

Basing a comprehensive and extensive regulatory framework on using a risk assessment process that depends heavily on the discipline of ecology, which itself is in turmoil and undecided on a generally accepted paradigm, and toxicology, which is struggling to expand beyond the boundaries of the laboratory, is clearly policy rather than science. The biblical warning about building an edifice on a foundation of sand appears appropriate. It may be good policy, but scientists

and risk assessors should be wary of confusing science-policy with science.

How can bioaccumulation assessment be effectively applied to human health and ecological risk assessments?

Bioaccumulation in and of itself is not an adverse environmental effect. Only accumulation that is associated with an adverse effect in organisms which have accumulated the material, either directly or via the food chain, is of importance. Furthermore, many materials produce adverse effects without any significant accumulation. For environmental protection and associated regulations bioaccumulation is not the issue, adverse effects are!

More work has been done on examining dose-response relationships for changes and adverse effects in the lab, effects that are possible, than for changes and adverse effects in the field, effects that are probable. As too little is known about ecosystems and effects at higher levels of organization (Calow, 1994), it is not surprising that relationships between lab effects at the individual/population levels and field effects at the community/ecosystem levels are poorly understood.

Furthermore, science policy must provide technical definitions of significant adverse effects in the field that are quantifiable by ecologists and risk assessors and consistent with current ecological theories. It is safe to say, with the multiplicity of changes and putative adverse effects being studied and reported in the literature, that currently there is no general agreement on what constitutes significant adverse environmental effects. Risk assessment is useful only in a comprehensive risk management framework where risk management goals are specified in the technical terms that scientists practicing risk assessment can quantitatively address (McCarty and Power, in press).

What are the requirements for selecting species for bioaccumulation testing? Are indigenous species necessary?

Using residue information and appropriate bioassay interpretation, real differences in species sensitivity can be separated from differences associated with bioavailability and toxicokinetics. Indigenous species may be useful and may provide important information but only when the nature of "sensitivity" differences are clear and sensitivity itself is clearly defined. For example, the influence of bioavailability, exposure medium, and various modifying factors (e.g., body size) is often not considered in toxicity test interpretation. There is an almost mythical belief that a most sensitive species can be determined and that this determination will be useful. However, most sensitivity examinations use exposure-medium-based LC50s or similar estimates and do not fully interpret toxicity test results to remove the effects of modifying factors. Such efforts are largely futile (Power and McCarty, manuscript).

Some headway can be made with a residue-based approach. Lanno and McCarty (in press) discuss a case comparing pentachlorophenol toxicity to a freshwater fish, a freshwater benthic invertebrate, and the common earthworm. Based on exposure-based LC50 bioassay results, it appears that the fish is more sensitive (threshold LC50 of 0.00039 mmol/L) than the benthic invertebrate (threshold LC50 of 0.0019 mmol/L). The threshold LC50 of 0.14 mmol/kg dry soil for the earthworm is not comparable due to the differences in exposure media. However, when the lethal body residues (LR) are examined, a sensitivity comparison is possible between the three organisms. The LR50 range is 0.08-0.17, 0.33-0.79, and 0.3-101 mmol/kg wet weight for the fish, earthworm, and benthic invertebrate, respectively. The fish appears to be slightly more sensitive, but the effect of differing body lipid contents has yet to be determined and may alter the relationship.

How can tissue-specific residue levels be coupled with chronic toxicity response data to develop dose-response relationships for bioaccumulative contaminants?

This issue is examined in detail in McCarty and Mackay (1993) and in Rand et al. (1995). The basic approaches are estimation using existing toxicity data and bioaccumulation relationships (i.e., exposure-based toxicity estimate* bioconcentration factor = whole-body residue-based toxicity estimate) and generation of residue-effect data from new experimental testing.

Work by Mayer (Mayer et al., 1986, 1992) on the relationship between acute and chronic toxicity endpoints is particularly valuable in both approaches. He has established that, in many cases, the lower tail (specifically, LC0.01) of the distribution of acutely lethal toxicity is equivalent to the maximum acceptable toxicant concentration (MATC)/lowest observed effect concentration (LOEC) obtained in chronic toxicity testing for growth/survival but not reproduction endpoints. Figure 1 (modified from Figure 5, McCarty and Mackay, 1993) illustrates the point. When the toxicity data from a test are transformed to a linear relationship using the log-probit transformation, any proportional response can be interpolated or extrapolated. When measured exposure-based (LCx) or residue-based (LRx) toxicity data are available, values can be directly estimated. Similarly, LRx data can be estimated from LCx data where bioconcentration factor information is available.

This approach provides many additional insights due to a more complete exploitation of toxicity test information. It was the basis for the suggestion that the current separate acute toxicity, chronic toxicity, and bioconcentration tests be combined into a single aquatic toxicity test protocol (McCarty, 1991). Such a combined approach would require an alteration in direction from

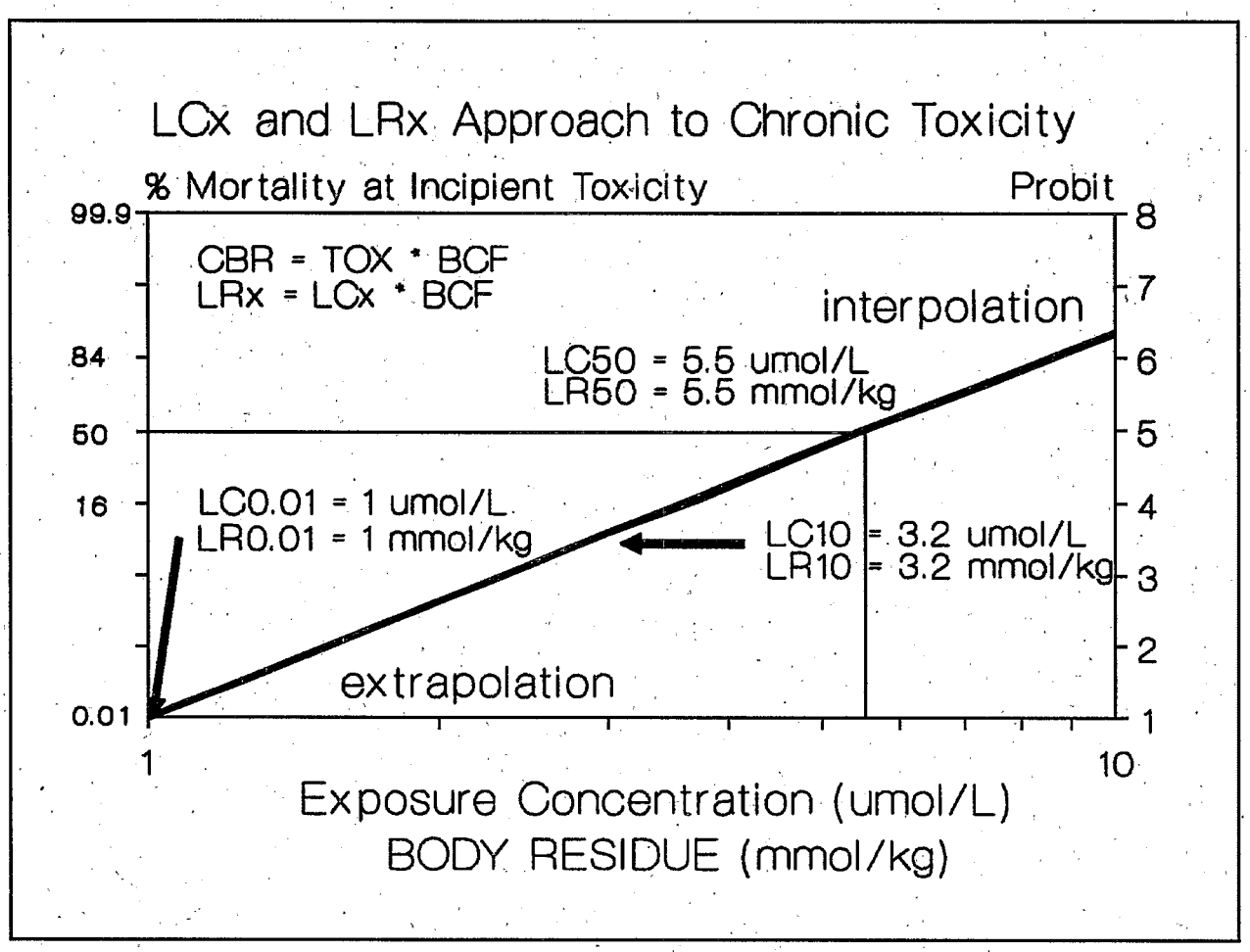

Figure 1. LCx and LRx approach to chronic toxicity.

the current minimalistic trend in testing protocols. Despite the additional observations and sampling that would be required, it is still likely to be less costly than the current trio of tests. As well, when conducted as scientific research experiments, such an approach will ultimately be more informative than abbreviated "regulatory" testing.

Although there is an increasing demand for more chronic toxicity data, both exposure-based and residue-based chronic information have a number of limitations. These include the following: chronic data are more expensive to collect, chronic results usually have greater uncertainty, some chronic endpoints are not readily interpretable, relationships between whole body levels versus those found in selected tissues have not been worked out, and differences in body lipid content confound precise residue-based interpretation.

It should be possible to develop protocols and procedures to address these limitations. The current limited availability of chronic residue-effect data can be overcome by estimation and new experimentation. However, the desirability of shifting the bulk of ecotoxicological testing to chronic effects is based on the largely mythical belief that chronic data are somehow "better" as the basis of environmental regulations. This appears to be largely based on the assumption that chronic effects in the laboratory are equivalent to and readily comparable to chronic effects in actual field situations. This issue is addressed in the next section.

How can bioaccumulation assessment methods, including testing and models, be used to address population-level effects?

At the present time only acutely toxic effects and major growth or reproductive effects can be effectively modeled by population modelers. The major difficulty is density-dependent responses, which are poorly known. The objective of environmental protection is protection of communities and local ecosystems. However, density-dependent and other interspecies interactions, which are largely unknown, represent a poorly quantified level of complexity that effectively renders extrapolation from bioassay to field largely an exercise in professional judgment, not quantitative analysis and modeling (Power and McCarty, manuscript).

Laboratory toxicity testing is focused primarily on addressing bioavailability, kinetics, and the resistance/

tolerance of organisms exposed under highly controlled conditions. The density-dependent factors that influence toxic effects in the field are rarely studied in the lab. An exception is work by Arthur and Dixon (1994). Juvenile fathead minnows were placed in 1-L screen cages at a density of 1, 5, and 10 individuals per cage and then placed in a flow-through system exposure apparatus. The chronic growth effects of pentachlorophenol and 2,4,5-trichlorophenol were examined for 28 days. Up to about twofold differences in chronic toxicity were found, e.g., low-density LOEC = 71 µg/L PCP; high-density LOEC = 121 µg/L PCP.

There is a significant effect of density on the outcome of a toxicity test in tightly controlled experimentation in the lab with a single species over a relatively short time period. It is clear that, with the myriad of opportunities for confounding influences in field situations of multiple populations under varying conditions, density-dependent factors remain a serious obstacle to extrapolating laboratory testing data to the field. The severity of the problem increases as consideration moves from short-term acute effects to long-term chronic effects as there is more opportunity for density-dependent factors to operate the longer organisms are stressed but still living. Thus, more chronic residue-based toxicity information alone will not improve the success of extrapolating such information to the field. A much better understanding of the influence of density-dependent factors is also required before any substantial improvement can be expected.

Conclusions

1. Basic ecological theories need further development and clear separation from science policy.
2. Bioaccumulation is not intrinsically an adverse effect endpoint.
3. Body residues can help identify true differences in species sensitivity by improving the understanding of modifying factors.
4. Chronic body residue data can assist in interpretation of toxicity testing results if improved methods and analyses are adopted.
5. Improvement in lab-to-field extrapolation requires both greater residue-based toxicity knowledge and a better understanding of the density-dependent modifying factors acting within and between species in the field.

Recommendations

Policy. Separate science from science-based policy by use of a clear risk management framework.

Toxicity 1. Develop a single generic bioassay protocol that integrates acute and chronic toxicity as well as bioconcentration.

Toxicity 2. Do not use a local species unless at least one standard species, selected from a very restricted list, is also tested. Clarify that an indigenous species is actually more sensitive than the standard species using tissue residue-effect relationships to determine the influence of modifying factors such as body size, temperature, behavior, and nutritional characteristics.

Ecology. Further develop a basic ecological paradigm and enhance population/community knowledge, especially for density-dependent interactions.

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Quantification of Ecological Risks to Aquatic Biota from Bioaccumulated Chemicals

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Abstract

A simple sounding yet difficult to answer question is "What concentration of a chemical in the tissues of aquatic biota is harmful to the biota itself?" This question is of particular importance in ecological risk assessment, where measurements of chemical residues in aquatic biota are often available, but the interpretation of their effect on biota is difficult. The objective of this work was to define tissue residues for a number of chemicals which, if not exceeded, pose little threat of risk to aquatic biota. These tissue screening concentrations (TSCs) were designed to be nonsite- or species-specific indicators of low risk residue levels. TSCs have been derived for 152 chemicals, both metals and organics, using a one-compartment first-order kinetic model. These TSC values are currently being used in ecological risk assessments to identify chemicals of potential concern, thus narrowing the focus of the risk assessment. To confirm the validity of the TSCs, a literature review of whole body tissue residues associated with adverse toxicological or ecological effects was performed. The review currently contains over 1400 records of tissue residues associated with adverse effects of 120 of the TSC chemicals. For chemicals where the TSC values are applicable, 94 percent of the literature reviewed indicates that adverse effects occur only at tissue residues higher than the TSC values. This is comparable to the U.S. Environmental Protection Agency's (USEPA) ambient water quality criteria, which are designed to be protective of 95 percent of aquatic genera. Analysis of this literature indicates that as groups (1) marine and freshwater biota are equally sensitive to chemical residues in their tissues, and (2) benthic and pelagic biota are equally sensitive to chemical residues in their tissues. Sufficient literature is available for a number of chemicals to permit direct estimates of the likelihood residues in aquatic biota from a site pose adverse risks. The existence of a tissue residue literature database relating adverse effects to residue levels eliminates one of the primary shortcomings preventing the use of the tissue residue approach in sediment quality criteria development: lack of documentation of tissue residues related to adverse toxicological effects. The database also provides indirect confirmation of a primary assumption of the equilibrium partitioning approach to sediment quality criteria development: benthic biota have a similar range of sensitivity to chemicals as do pelagic biota.

Introduction

Historically, the primary use of aquatic biota tissue residue data in ecological risk assessments has been to provide an indication that the biota have been exposed to chemicals at a site. Seldom have efforts been made to directly quantify ecological risks from bioaccumulated chemicals.

The most common approach to quantifying ecological risks to aquatic biota is to divide the concentration of a chemical in water or sediment by a toxicity reference value (TRV), a concentration which if exceeded is expected to result in adverse ecological effects. The resulting hazard quotient is used as a measure of risk to biota, with the likelihood of adverse effects increasing with increasing magnitude of the hazard quotient. TRVs available for use in ecological risk assessment include ambient water quality criteria (AWQC) and sediment quality values from several sources. Unfortunately, there is not a comparable set of TRVs for use in assessing ecological risks from tissue residues in the biota themselves, despite a sizable amount of research relating body burdens to toxic effects and the availability of several literature reviews of this information (McCarty and Mackay, 1993; McKim and Schmieder, 1991; Dillon, 1984).

Chemical concentrations in water and sediment are surrogates for the actual dose of chemical at the site of toxic action in biota. The use of these surrogates for the actual

dose has many limitations and introduces uncertainties in assessing adverse effects of chemicals on aquatic biota, some of which are outlined below.

- The bioavailable fraction of the total chemical concentration in exposure media may not be known.
- Surrogates do not consider multiple uptake routes of chemicals by biota.
- Intermittent, pulsed, or varied exposures cannot be readily assessed.
- Short exposure times can result in nonsteady-state tissue residues and variable toxicity.
- Metabolic transformations of toxicants which enhance or reduce toxicity are not considered.
- Animal behavior (i.e., seasonal migration or toxicant avoidance) is not accounted for.
- Analytical chemistry limitations (e.g., nondetectable concentrations in exposure media) mean the dose is often unknown.

By quantifying risks based on tissue residues associated with adverse toxicological or ecological effects, the above complicating factors are largely eliminated and a more accurate risk assessment can be performed.

The objectives of this work were as follows:

1. To derive risk-based screening concentrations (RBSCs) for assessing ecological risks from chemical residues in aquatic biota tissues to the aquatic biota themselves.
2. To confirm the utility of the derived RBSCs in ecological risk assessment.

Methodology

USEPA's ambient water quality criteria are designed to be protective of 95 percent of all aquatic genera (Stephan et al., 1985). By extension of this principle, tissue residues bioconcentrated from criteria concentrations should also, if not exceeded, be protective of 95 percent of all aquatic genera. This is the fundamental assumption behind the approach used to derive toxicity reference values for bioaccumulated chemicals in aquatic biota.

Tissue residues in aquatic biota which, if exceeded, may describe residues associated with adverse toxicological or ecological effects are termed tissue screening concentrations (TSCs) in this paper. This is because the primary use of the TSC values is as a screening tool to select chemicals of potential concern (COPCs) in ecological risk assessments. COPC identification shortens the list of chemicals carried through the entire ecological risk assessment process. TSCs are intended to be nonsite- or species-specific indicators of tissue residues which, if not exceeded, pose little threat of adverse risk to aquatic biota.

TSCs were derived from the one-compartment first-order kinetic (1CFOK) toxicological model given in Equation 1.

$$\frac{dC_b}{dt} = (k_u \times C_w) - (k_e \times C_b) \qquad (1)$$

where:

C_b = chemical concentration in biota (mg/kg)
t = time (hours)
k_u = chemical uptake rate constant (L/kg/hr)
C_w = chemical concentration in water (mg/L)
k_e = chemical elimination rate constant ($hour^{-1}$)

If the chemical concentration in water is assumed to be constant, Equation 1 may be exactly integrated to yield Equation 2.

$$C_b = C_w \times \frac{k_u}{k_e} \times (1 - e^{-k_e t}) + (C_{b(t=0)} e^{-k_e t}) \qquad (2)$$

If it is further assumed that the animal starts with no tissue residue of the chemical of interest and the tissue residue is at steady state with respect to the water concentration, Equation 2 reduces to Equation 3.

$$C_b = C_w \times \frac{k_u}{k_e} \qquad (3)$$

By redefining the terms in Equation 3, C_b becomes the tissue screening concentration, C_w becomes an ambient water quality criterion (AWQC), k_u/k_e is a bioconcentration factor (BCF), and the redefined Equation 3 can be used to derive tissue screening concentrations, as shown in Equation 4.

$$TSC = AWQC \times BCF \qquad (4)$$

Although the derivation of TSCs is based on sound toxicological concepts, in practice they are derived simply by multiplying a water quality criterion by a bioconcentration factor.

To provide a more conservative screening value, water quality criteria used in the calculations are the lower of USEPA's freshwater or chronic ambient water quality criteria. Since some of USEPA's water criteria documents are for classes of chemicals (e.g. PAHs, chlorinated benzenes, chlorinated phenols), a single criterion value may be used in the derivation of multiple TSC values. For metals with hardness-dependent criteria, a hardness of 50 mg/L $CaCO_3$ was assumed. If only an acute criterion was available for a given chemical, the acute criterion was divided by 8 to estimate a chronic criterion. The bioconcentration factors used were taken from the human health portion of the USEPA water quality criteria documents. For metals, the BCFs are geometric means of measured BCF values, whereas for organics, the criteria BCFs were calculated using a regression equation relating octanol-water partition coefficient to BCF. Most of the BCF values used are found in the *Superfund Public Health Evaluation Manual* (USEPA, 1986).

To confirm the validity of the TSC values, a literature review was performed of papers relating measured whole body, wet weight tissue residues to adverse toxicological or ecological effects. For papers where dry weight tissue residues were reported, a conversion to wet weight was made assuming 80 percent water content if the actual water content was not given in the paper. Effects considered in the review were population and community effects, mortality, reproduction, growth,

behavioral, cellular, biochemical, or physiological changes.

The literature database currently contains the following information for each citation: chemical name, tissue screening concentration, tissue residue associated with effect (or the no effect residue), common and scientific names of the species studied, toxicological or ecological effect, a safety factor (defined in Equation 5), a footnote field that contains information on the exposure conditions of the study, and the full literature citation.

To describe the difference between TSC values and the tissue residues associated with adverse effects, a safety factor (Equation 5) was calculated.

$$TSC\ safety\ factor = \frac{Tissue\ concentration\ associated\ with\ effect}{TSC} \quad (5)$$

The safety factor provides qualitative evidence of the level of protection TSCs provide aquatic biota from adverse effects. Computationally, the safety factor can also be considered a hazard quotient for a measured tissue residue associated with a specified effect as given in the literature.

It must be noted that the TSC values are intended to identify tissue residues which, if not exceeded, pose little or no risk to aquatic biota. They are not intended to define tissue residues that are protective of avian, mammalian, or other wildlife species that prey upon aquatic biota.

Results

Table 1 provides a representative example of the 152 currently available tissue screening concentrations. All tissue residues given in Table 1 have units of mg/kg whole body, wet weight. As shown in Table 1, the TSC values span a wide range of tissue residues predicted to have little or no effect on aquatic biota.

The literature review currently contains nearly 500 citations and 1400 records associating tissue residues to adverse effects. Of these, approximately 10 percent of the individual records describe no observed adverse effect tissue residues. A range of tissue residues have been associated with adverse ecological or toxicological effects. Figure 1 shows the distribution of tissue residues associated with adverse cadmium effects. The distribution in Figure 1 is typical of that for most chemicals, where no or only a few citations indicate that effects occur below the tissue screening concentration, but most adverse effects occur at tissue residues above the tissue screening concentration.

At least one literature citation is available for 120 of the 152 chemicals for which TSCs exist. Ten or more residue-effect records are available for about 40 chemicals. Cadmium has the most literature information available of any chemical in the database, while PCBs have the most information available for any organic chemical. Other chemicals that have a substantial amount of literature available include mercury, copper, zinc, dioxin, pentachlorophenol, and several chlorinated insecticides.

Once the no observed adverse effect residues are removed from the database, 83 percent of the tissue residues associated with adverse effects are concentrations higher than the tissue screening concentrations. Over half of the tissue residues associated with adverse effects at concentrations lower than the TSCs are for chemicals that are rapidly (within a few hours or days) metabolized to more toxic compounds. The rapidly metabolized chemicals are mostly PAH compounds, although the chlorinated insecticide aldrin is also rapidly converted to a more toxic metabolite, dieldrin. Figure 2 shows tissue residues of benzo(a)pyrene associated with adverse effects. For benzo(a)pyrene, every citation available shows an adverse effect at a whole body concentration below its TSC value.

Once chemicals rapidly metabolized to more toxic forms are removed from the literature database, the predictive ability of the TSC values to identify residue levels below which adverse effects are unlikely improves. By performing the TSC to adverse effects literature comparison without chemicals rapidly metabolized to more toxic forms, 94 percent of the tissue residues associated with adverse effects are higher than the tissue screening concentrations.

The calculated safety factors (Equation 5) provide a qualitative indication of the conservative nature of the TSC values. For all residue effect citations in the literature database, the geometric mean safety factor is 15, while the arithmetic mean safety factor is 41. The safety factors for individual literature citations are generally largest for measures of mortality and smallest for biochemical endpoints.

Table 1. Derivation of selected tissue screening concentration values.

Chemical	AWQC µg/L	AWQC Source	BCF L/kg	TSC mg/kg
Aldrin	1.3	Marine acute	4,670	0.71
Benzo(a)pyrene	300	PAH marine acute	11,100	416
Cadmium	0.66	Freshwater chronic	64	0.042
Copper	2.9	Marine acute	200	0.17
4,4'-DDT	0.001	Freshwater chronic	53,600	0.054
Dieldrin	0.0019	Freshwater chronic	4,760	0.0090
Dioxin (2,3,7,8-TCDD)	0.00001	Freshwater chronic	5,000	0.000050
Hexachlorobenzene	3.68	Freshwater chronic	8,690	32
Mercury	0.012	Freshwater chronic	4,994	0.060
PCB	0.014	Freshwater chronic	31,200	0.44
Tributyltin	0.01	Marine chronic	693	0.0069
1,2,4-Trichlorobenzene	50	Freshwater chronic	2,800	140
Zinc	59	Freshwater chronic	47	2.8

AWQC - USEPA Ambient Water Quality Criterion
BCF - Bioconcentration Factor
TSC - Tissue Screening Concentration

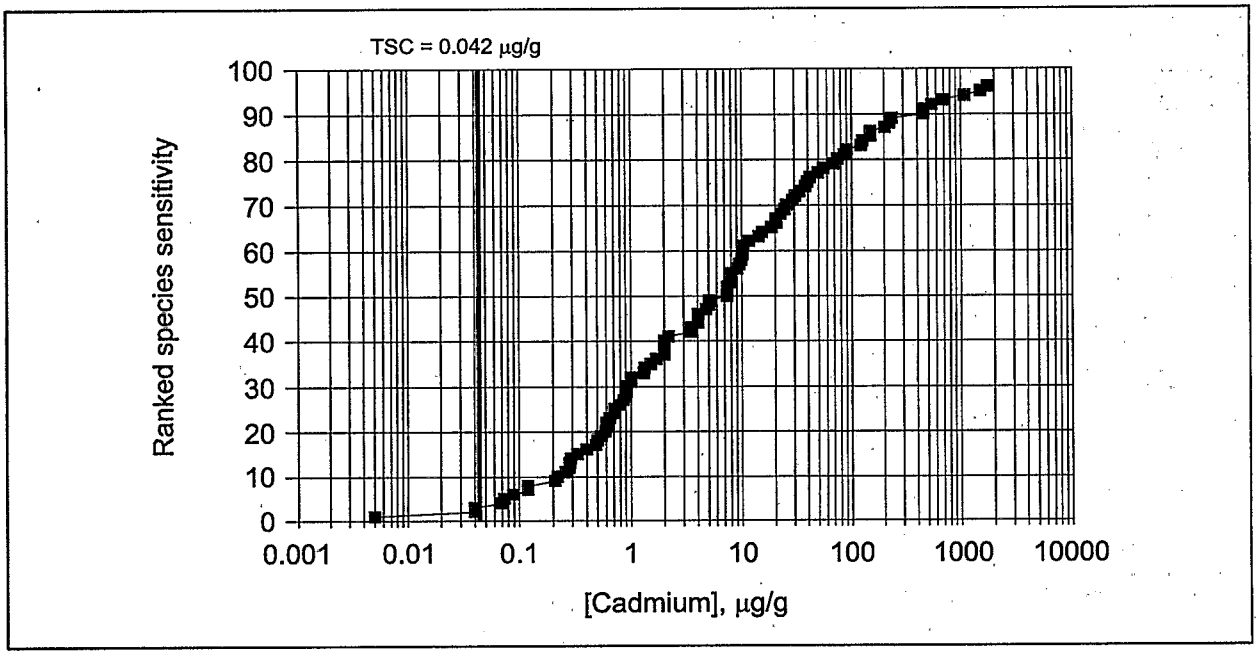

Figure 1. Tissue residues associated with adverse effects: Cadmium.

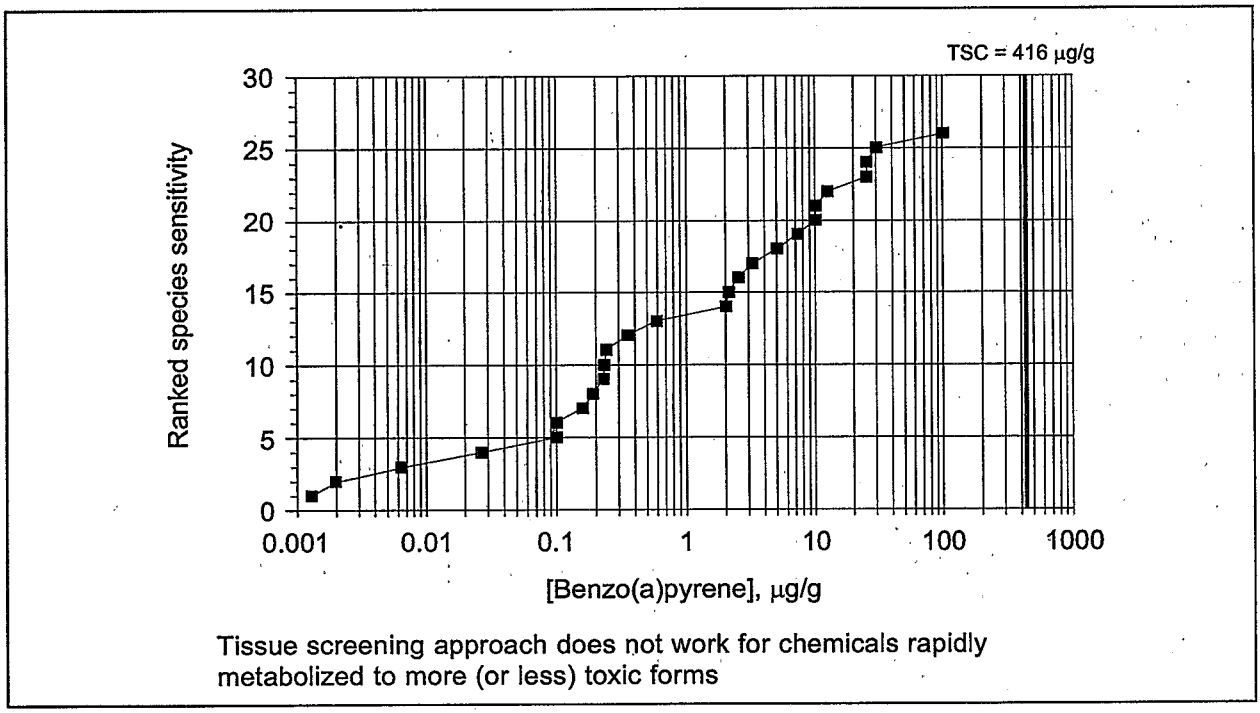

Figure 2. Tissue residues associated with adverse effects: Benzo(a)pyrene.

The safety factors have been used to make several statistical comparisons among various subsets of the literature data. These comparisons have found that, as groups:

1. There are no significant differences in residue levels that cause adverse effects in freshwater or marine biota.
2. There are no significant differences in residue levels that cause adverse effects in benthic compared to pelagic biota.
3. There are no significant differences in the sensitivity of biota to chemicals under field or laboratory exposure conditions.

Although the results given above appear broadly applicable based on the literature review, there are exceptions in some instances. For example, arsenic residues associated with adverse effects in freshwater biota are much lower than arsenic residues that adversely affect marine biota.

Table 2. Human health exposure scenario used to derive toxicity reference values (TRVs) for comparison of TSC values to TRVs for consumers of fish and shellfish.

Exposure Scenario Parameter	Value	Exposure Scenario Parameter	Value
Exposure frequency	350 days/year	Fish and shellfish ingestion rate	6.5 grams/day
Exposure duration	30 years	Target noncancer risk	1
Body weight	70 kilograms	Target cancer risk	1×10^{-6}

Tissue screening concentrations have also been compared to human health toxicity reference values (TRVs) for chemicals in fish and shellfish consumed by humans. The exposure scenario and risk assumptions used to derive the human health TRVs are presented in Table 2. For noncarcinogenic chemicals where comparisons could be made, the ecological TSC values were lower than the human health TRVs in 61 of 67 comparisons (91 percent). For carcinogenic chemicals, the human health TRVs were lower than the ecological TSCs in 36 of 42 comparisons (86 percent).

Discussion

To date, TSCs have been used as a screening tool in ecological risk assessments at several Superfund sites. In all cases, they have identified chemicals of potential concern (COPCs) that have also been identified as COPCs in at least one other human health or ecological risk assessment scenario at the same site. This type of agreement with other risk assessment procedures provides corroborative evidence that the TSCs are successfully identifying chemicals worthy of detailed investigation in ecological risk assessments.

The largest shortcoming of the procedure appears to be its lack of applicability to chemicals that are rapidly metabolized to more toxic forms. Many PAH compounds elicit adverse effects at tissue residues several orders of magnitude below the PAH tissue screening concentrations (Figure 2). Although this may be due in part to the use of an old water quality criterion (USEPA, 1980a) for PAHs, a mechanistic reason for this observation can also be given.

Many PAHs are known to be rapidly metabolized to more toxic compounds (USEPA, 1980a). A tissue screening concentration based on residues of a less toxic parent compound will not represent a safe concentration of a more toxic metabolite. For rapidly metabolized chemicals, a TSC for the more toxic metabolite should be used to assess risks, rather than the TSC for the parent compound. An example of this approach is shown in Figure 3. Aldrin is rapidly metabolized by many species to dieldrin (USEPA, 1980b). The limited amount of

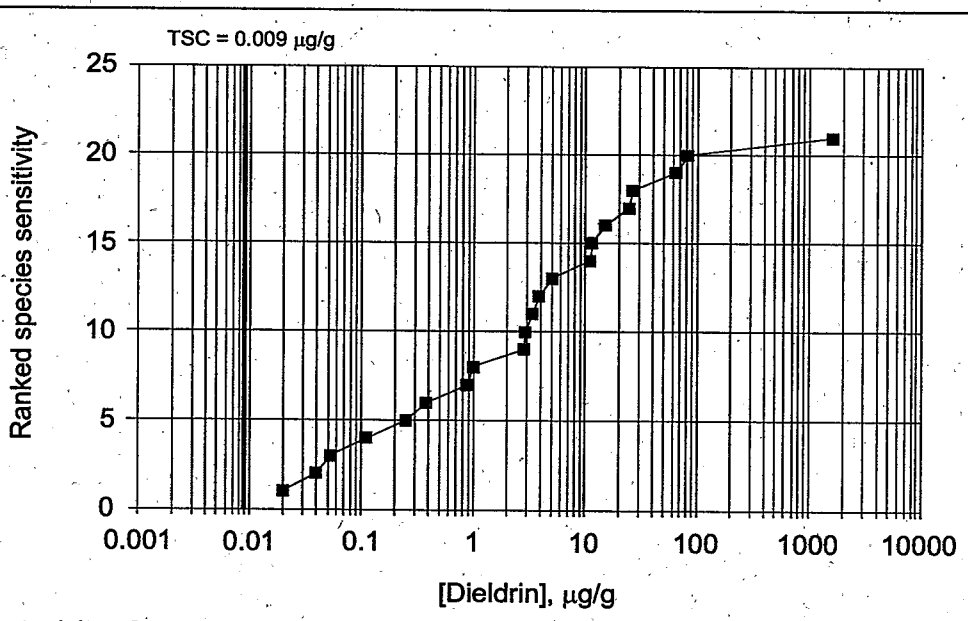

Figure 3. Tissue residues associated with adverse effects: Dieldrin.

aldrin residue data associated with adverse effects brackets the aldrin TSC value. A substantial amount of literature is available for dieldrin residue toxicity (Figure 3). In all cases, dieldrin residues associated with adverse effects are higher than the dieldrin TSC value. Unfortunately, TSC values do not currently exist for PAH metabolites, limiting the utility of the TSC approach to chemically and metabolically stable toxicants.

They are also not applicable for chemicals whose toxicity does not result from an internally absorbed dose. Examples of chemicals in this second category may be aluminum and iron, whose toxicity largely comes from formation, under certain water quality conditions, of a flocculent material that suffocates aquatic biota.

Several of the TSC values appear to be overly conservative based on the literature review. In particular, no adverse effects have been associated with copper residues below 3 mg/kg or zinc residues below 20 mg/kg, considerably higher than the respective calculated TSC values of 0.17 and 2.8 mg/kg. In practice, we are now using the 3 mg/kg copper and 20 mg/kg zinc values as screening concentrations in ecological risk assessments. Many aquatic species can regulate their body burdens of copper and zinc. The 3 and 20 mg/kg screening values for copper and zinc are much closer to the known or estimated physiological requirements of these two elements in aquatic biota (van Tilborg and van Assche, 1996; White and Rainbow, 1985) than are the TSC values in Table 1.

At least one TSC is not sufficiently conservative for use as a screening tool in ecological risk assessment. The hexachlorobenzene TSC of 32 mg/kg is higher than 9 of the 10 literature citations associating hexachlorobenzene residues with adverse effects. If the hexachlorobenzene TSC is recalculated using the bioconcentration factor from Table 1 and the Canadian Water Quality Guideline of 0.0065 µg/L instead of the draft USEPA ambient water quality criterion of 3.68 µg/L, the resulting TSC is 0.056 mg/kg. This recalculated TSC is lower than all 10 hexachlorobenzene adverse effect residue levels reported in the literature. Use of the Canadian Water Quality Guidelines is currently under investigation for use in calculation of additional TSC values for chemicals where USEPA currently has no ambient water quality criteria.

The availability of a literature database of tissue residues associated with adverse effects permits the use or derivation of several other ecological risk estimation methods. The hazard quotient approach has already been discussed. For aquatic species where a substantial amount of literature is available, the best approach may be the direct identification of tissue residues associated with adverse effects. Rainbow trout (*Oncorhynchus mykiss*) and blue mussels (*Mytilus edulis*) are the freshwater and marine species with the most tissue residue information available in the literature. By comparing the distribution of tissue residues associated with adverse effects (Figures 1-3) to the residue distribution in animals from a site of interest, probabilistic risk assessments could be performed. Other endpoints analogous to sediment or water quality criteria or guidelines, such as apparent effects thresholds (AETs), lowest observed adverse effect levels (LOAELs), or tissue residues above which effects on a defined percentile of species occur could all be calculated from the literature database. The defined percentile approach could be the tissue residue equivalents of the Long and Morgan (1991) effects range-low (ER-L) and effects range-median (ER-M) sediment quality guidelines.

Although the primary focus of this paper has been on the use of tissue residue information to define ecological risks to aquatic biota, the tissue residue approach also has applicability to the derivation of sediment quality criteria. USEPA (1993) has identified the tissue residue approach as a technically valid approach for the derivation of sediment quality criteria. One of the major identified shortcomings of the tissue residue approach to sediment criteria development is the absence of a database of residue levels associated with toxicity (USEPA, 1993). The database developed to confirm the utility of the TSC values could also serve to eliminate this identified shortcoming of the tissue residue approach to sediment criteria development. The database also provides indirect evidence that benthic biota, as a group, are equivalent in their response to toxicants to pelagic biota, a fundamental assumption of the equilibrium partitioning approach to deriving sediment quality criteria.

Summary and Conclusions

Although the literature database compiled during this study allows a number of hypotheses to be tested and conclusions to be drawn, the three primary conclusions that have been drawn to date are:

1. Tissue residues of chemicals in aquatic biota can, for many chemicals, be used to directly assess ecological risks to aquatic biota.
2. Chemicals for which tissue residues cannot be used to quantify risks can be identified from mechanistic considerations.
3. The level of protection from adverse risk provided by the tissue screening concentration approach is comparable to that provided by USEPA's ambient water quality criteria.

The TSC method appears to provide a conservative initial screen capable of eliminating from an ecological risk assessment chemicals that do not pose significant risks to aquatic biota. Exceedance of a tissue screening concentration does not automatically imply that an observed tissue residue poses an adverse risk to biota. It does, however, identify those chemicals which require more detailed investigation in an ecological risk assessment.

The existence of an interpretive tool for assessing risks or hazards to aquatic biota from bioaccumulated chemicals has many potential applications in addition to ecological risk assessment. Environmental assessments, dredging bioassessments, and criterion and standard development are three of the many possible uses. Interpretation of tissue residues has the potential to provide substantially more information than its current primary use, which is as an indicator of exposure to chemicals.

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Question

Can contaminant tissue residues in aquatic biota be used to define ecological risks to aquatic biota?

Objectives

1. **To derive risk-based screening concentrations (RBSCs) for assessing ecological risks of chemical residues in aquatic biota tissues**
2. **To confirm the utility of the RBSCs in ecological risk assessment**

A Newer Objective

❸ To directly quantify ecological risks from chemical residues in aquatic biota tissues

Available Toxicity Reference Values in Aquatic Toxicology

- Ambient water quality criteria (AWQC)
- Sediment quality criteria

The Fundamental Principle of Toxicology

The magnitude of the toxic response is proportional to the toxicant concentration at the site of toxic action

Tissue Screening Concentrations (TSCs)

Whole body, wet weight tissue residues of chemicals which, if not exceeded, pose little chance of causing adverse toxicological or ecological harm to aquatic biota. TSCs are intended to be non-site or species specific.

Uses of Tissue Screening Concentrations

- Primary use is as a screening tool to select chemicals of potential concern in ecological risk assessment
- Can also be used as the denominator in hazard quotient calculations

Toxicological Basis for Tissue Screening Concentrations

1CFOK model
(integrating form assuming constant water concentration)

$$C_b = C_w \times (k_u/k_e) \times (1 - e^{-k_e t}) + (C_{b(t=0)} e^{-k_e t})$$

Where:

C_b = chemical conc. in biota (mg/kg)
t = time (hours)
C_w = chemical conc. in water (mg/L)
k_u = chemical uptake rate constant (L/kg/hr)
k_e = chemical elimination rate constant ($hour^{-1}$)

Toxicological Basis for Tissue Screening Concentrations

If it is further assumed that:

❶ The initial tissue residue is zero, and

❷ The animal is at steady state

The 1CFOK model reduces to . . .

Calculation of Tissue Screening Concentrations

$$TSC = WQC \times BCF$$

Where:

TSC = tissue screening concentration
WQC = water quality criterion (mg/L)
BCF = 3% lipid normalized bioconcentration factor (L/kg)

Example TSC Values

(All values mg/kg whole body, wet weight)

Chemical	TSC
Arsenic	1.6
Cadmium	0.042
4,4'-DDT	0.054
Dioxin (2,3,7,8-TCDD)	0.00005
Mercury	0.12
PCB	0.44
Tributyltin	0.006
1,2,4-Trichlorobenzene	140

TSCs currently available for over 150 chemicals

"The direct prediction of chronic toxic effects from measured or predicted tissue residues requires validation before it can be widely endorsed."

p.7-7, USEPA Sediment Classification Methods Compendium (1993)

Performed literature review of measured tissue residues associated with toxicological effects. It currently contains about 1400 records, 490 literature citations, and information on 118 of the 152 chemicals for which TSCs exist

Database Structure

1. Chemical name
2. TSC value
3. Residue concentration associated with effect
4. Species
5. Toxicological or ecological effect
6. Safety factor
7. Footnote
8. Literature citation

Safety Factor

Effect tissue concentration
TSC

Provides qualitative evidence of the level of protection TSCs provide aquatic biota from a specific effect. Also could be considered a hazard quotient for the specified effect at a given tissue residue.

Criteria for Inclusion in Database

- Had to report measured tissue residues
- Only papers reporting whole body residues
- No limitation on toxicological endpoint
- No limitation on route of exposure
- Included both laboratory and field studies
- Minor limitations on aquatic species included

Distribution of Tissue Residue Literature

Range of citations	No. of chemicals
1 - 9	82
10 - 19	20
20 - 29	8
30 - 39	3
40 - 49	1
50 - 59	2
60+	2

Assumptions in Validating Utility of TSCs

- No difference in sensitivity of freshwater, estuarine or marine biota
- No differences in sensitivity of benthic, epibenthic or pelagic biota
- No differences in response of laboratory and field exposed biota
- Interested only in identifying risks to aquatic biota—TSCs not designed to be protective of piscivorous birds and wildlife

IT WORKS!

(At least for most chemicals)

Results of Literature Review

- About 10% of results describe no observed effect at a given tissue residue
- For the entire database, 83% of adverse effects occur at concentrations above TSC values
- When chemicals which are rapidly metabolized to more toxic compounds are removed, 94% of reported effects occur at concentrations above TSC values

Results of Literature Review

- Geometric mean TSC safety factor for entire database is 15
- Arithmetic mean TSC safety factor for entire database is 41

Possible Approach for Chemicals Rapidly Metabolized to More Toxic Compounds

Aldrin

TSC = 0.71 mg/kg

2 of 2 records show adverse effects below TSC (4 no effect records)

Dieldrin

TSC = 0.009 mg/kg

0 of 19 records show adverse effects below TSC (1 no effect record)

Care Must be Taken When Using TSCs in Risk Assessment

Hexachlorobenzene

- TSC = 32 mg/kg
- Derived from USEPA AWQC of 3.68 ug/L
- 9 of 10 records show adverse effects below TSC (11 no effect records)
- Deriving a TSC from the Canadian Water Quality Guideline of 0.0065 ug/L
- TSC = 0.056 mg/kg
- 0 of 10 records show adverse effects below TSC

Care Must be Taken When Using TSCs in Risk Assessment

Copper and zinc are two examples where TSCs may be overly conservative

Copper

TSC = 0.17 mg/kg, toxicity threshold at 3 mg/kg

Zinc

TSC = 2.8 mg/kg, toxicity threshold at 20 mg/kg

Many aquatic species can regulate their Cu and Zn burdens

Methods for Quantifying Ecological Risks of Tissue Residues

1. Hazard quotients
2. Apparent Effects Threshold (AET)
3. Effects on defined percentile
4. Direct assessment
5. Probabilistic risk assessment

Results from Risk Assessments Performed to Date

Tissue residue approach and TSCs have identified only chemicals of concern (COCs) which have also been identified as COCs by one or more other human health or ecological risk scenarios

Results Derived from Literature Review and Analysis of Safety Factors

- No significant difference in residue levels causing adverse effects in freshwater and marine biota (arsenic an exception)
- No significant difference in residue levels causing adverse effects in benthic and pelagic biota
- No significant difference in response of biota in field and laboratory exposures

Comparison of Human Health Toxicity Reference Values to Ecological TSCs

Compared TSCs to TRVs for a defined human health exposure scenario for seafood consumers

Exposure frequency	350 days/year
Exposure duration	30 years
Body weight	70 kg
Target noncancer risk	1
Target cancer risk	1×10^{-6}
Seafood ingestion rate	6.5 grams/day

Results of Comparing Human Health TRVs and Ecological TSCs

1. Ecological TSCs were lower than human health TRVs for 61 to 67 (91%) of chemicals where a comparison could be made
2. Human health TRVs were lower than ecological TSCs for 36 of 42 (86%) of chemicals where a comparison could be made

Conclusions

1. Tissue residues of chemicals in aquatic biota can, for many chemicals, be used to directly assess ecological risks to aquatic biota
2. Chemicals for which tissue residues cannot be used to quantify risks can be identified from mechanistic considerations
3. The level of protection from adverse risk provided by the tissue residue approach is comparable to that provided by USEPA's ambient water quality criteria

Day One: September 11, 1996

Session Two: Questions and Answers

After each session, there was an opportunity for questions and answers and group discussions pertaining to the speakers' presentations.

Q (Dave Michaud, Wisconsin Electric Power Company): Dr. Rubinstein, on your last slide you referenced the need to perhaps establish a minimum criteria for reference site selection. What are your thoughts in terms of groups of chemicals, or on a chemical-specific basis?

Norm Rubinstein:

Without having the residue effects data, we are limited in regard to interpreting chemicals and concentrations. I am talking more in terms of determining what represents a healthy ecosystem. What are the things we are measuring that satisfy our need to insure we are maintaining environmentally consistent conditions? This involves developing much broader databases, much like is done in the Puget Sound Dredged Disposal Analysis (PSDDA) Program. From information developed by this program, we know that the benthic communities are functioning and the animals that are supposed to be there are there.

Q (Dave Michaud): So, the concept might be, for example, going to an area where you have a healthy benthic community, taking sediment samples, and analyzing them for a suite of possible contaminants.

Norm Rubinstein:

Right, and then using that as your point of comparison. This is known as a reference comparison.

Q (John Zambrano, NYS Department of Environmental Conservation): Dr. Rubinstein, in your definition of reference sediment you have three components. The first and the third could be in conflict. The first component is one that is substantially free of contaminants, and the third component reflects the site if no material had been disposed of there. What do you do when they are in conflict, and could you explain how those two components relate to the purpose of using a reference sediment?

Norm Rubinstein:

I will be the first to admit that I think the definition does need to be revisited. The intent, when we started thinking about this issue, was to recognize the fact that many of the areas that require maintenance dredging are in highly industrialized and urbanized areas where the benthic habitat has been degraded over long periods of time. We have never looked at this program as a remediation program. The intent was to insure that we do not cause further degradation. So, as impractical as the language sounds, it was in fact a realistic way of getting a handle on what was there. We do have to go back, look at this definition, and establish what we are now considering to be environmentally acceptable material.

Q (Gayle Garman, NOAA): Dr. Rubinstein, I am familiar with the PSDDA program. I think it is interesting that you hold that up as an example for us, and yet the data you showed indicated that there was a lower survival rate for the amphipods at the control or reference sites in Puget Sound than for the other harbors. So, the Puget Sound approach did not seem to be the most protective approach for the data that you showed us. I would also like you to address the fact that you are talking about a healthy benthic community, and what we are focusing on here is bioaccumulation. A healthy benthic community does not necessarily indicate whether or not there is a potential for bioaccumulation.

Norm Rubinstein:

Yes, amphipod mortality was a little higher in Puget Sound and that is exactly the point of the utility of a toxicity endpoint in a given species. Mortality at 20 or 23 percent in a test species may not be indicative of a significant impact at a population level. When you look at the sediments in Puget Sound, it is my understanding that for factors other than chemical constituents like grain

size or ammonia, this is about the typical response in terms of the toxicity exposure for these animals. So, it is consistent with what I am saying, which is that percent mortality alone may not be a useful tool. Mortality has to be put into a more ecologically relevant context. And I agree with your statement that a healthy benthic community does not necessarily indicate whether or not there is a potential for bioaccumulation.

Q (Edward Zillioux, Florida Risk-Based Priority Council): Dr. Mount, I noticed that you said you limited your data collection to fish and invertebrates. I realize that this is a tremendous undertaking and there may be logistical reasons that you did not go further. But I would recommend that you consider including wading birds, because not only do they provide useful residue effect relationships that are in the literature, but they could also be a good link to higher trophic levels. We looked at this for mercury and found quite useful relationships that showed up in work conducted by Don Porcella and Jani Benoit. And we also found that the residue effect relationships derived from the field samples correlated fairly well with residue effects derived from laboratory studies with a typical white rat and mallards.

David Mount:

You raise a good point. I should emphasize that our decision to limit our database to aquatic species was not any sort of biological judgment, but purely a logistical decision. And you are absolutely right that there are lots of issues regarding bioaccumulation that extend well beyond the aquatic community.

Q (Arthur Asaki, U.S. Army Center for Health Promotion and Preventive Medicine): Dr. Mount, I congratulate your work in this area. It is something that has been needed for a long time, and I am glad somebody has done it. I looked at your data, your effect data and no-effect data, graphically represented. You had yellow squares for no-effect data and red diamonds for effect levels. There was quite a bit of overlap in those data points, which is to be expected. Later in your presentation, you showed a table for chlorpyrifos and kepone where the lowest effect level and the highest no-effect level did not overlap. Could you explain that?

David Mount:

I will try to explain what I think you are asking. The figure I presented included all data that were reported for that chemical, regardless of whether it was just one data point or several data points. The tables showed results of individual studies. If you look at the sheepshead minnow data for kepone, you will see several entries for sheepshead minnow in the table with different values for different studies. Each line of the table corresponded to a single study. The comparisons were reduced to just effect/no-effect pairs, in a sense kind of culling the data set, which probably reduced some of the variability. But all sources of variability, such as intra-species and intra-experiment, were represented in the figure.

Q (Maurice Zeeman): This question is for Dr. McCarty. There is a lot of talk going on today about dose-response relationships and tissue residue-based paradigms for toxicity assessment. I was wondering if we are going to have to start looking at some of these old chemicals in new ways, because of some new ways of looking at endpoints. Endocrine disrupters research is getting to be very interesting and it is suggesting, in essence, that dose-response relationships may not be all that important for these kinds of chemicals. When you are exposed to this trivial level of chemical may be more important than giving it to an adult later on at a much higher level or at different levels. What effects do you think that will have, if any, in terms of looking at dose-response relationships, tissue residue concentrations, and bioconcentration?

Lynn McCarty:

I do not think that the endocrine modulator people really believe they are going to modify the basic assumptions of toxicology. The description you presented is that their perception of dose response is giving a strong dose and getting a strong response. Everything is clear and understood. I think it is even more important in the sorts of things they are talking about for low level responses. We are still talking about a dose response. It is just down at low doses and at different endpoints than what we have previously looked at. I do not think it is any different at all. The standard toxicological paradigm applies. We do not have to throw out the paradigm because we do not understand the specifics of this case. I think the paradigm applies; we just have to get a greater understanding of what is going on. I think we see the echoes of this problem with PAHs. The residue for PAHs is no longer a marker of exposure for the organism, because it is so readily metabolized that what you measure today is not what the organism got a year ago. It may not be the dose that was reflective of what is causing effects to the organism today. That is the very same problem that the endocrine people are talking about, and it is simply the next level of effort. We have been very lucky in that we have many organic chemicals which are very recalcitrant to degradation, and so they can serve as their own markers of exposure. For these chemicals, the tissue residue that we see today is fairly reflective of the exposure that the organism received in the past and, therefore, is recently attributable to the effect that we see today. But we know there are situations where that does not occur. However, it still does not negate the need to know what the dose was at the time that the effect was initiated, and we have to develop procedures for estimating that. But I think there are people suggesting to simply bypass the whole scientific process and assessment of this. I think there is good science to be done and, if you throw out dose

response, then you throw out all science that is applicable to things. That certainly can be done, but I am not suggesting that.

Q (Peter Chapman): All chemicals are not the same for a variety of reasons. Do you think it is possible that we will be able to develop body burden-to-effect relationships for all chemicals within a reasonable time frame with reasonable effort? Or should we dedicate our efforts to those chemicals we think we will be able to do that for? Some of them may require so much effort and time it may not be worthwhile.

Lynn McCarty:

We are not going to look at every chemical. There is no question about that. I used the example I took from the EPA laboratory in Duluth on the modes of action. I think it is a brilliant piece of conceptual work in saying that we, as toxicologists, have been oppressed by the chemists for so long, because they are the people doing all of the chemical work and they always tell us the chemical relationships in terms of chemical descriptors. Well, I am a toxicologist and I want to have groupings according to toxicology groupings. I do not care what the structure of the chemical is. I want to know about the chemical based on effects. That is the first step in going in that direction. I think as we apply this tissue residue approach, it is going to allow us to get better estimates of those things and begin to categorize things on the basis of the effects that they have. We will be able to classify those effects into mechanistically related groups. It will also allow us the ability to look at mixtures, and hopefully that will allow us to address larger groups of things, conserve the limited resources available, and still improve our ability to do the tasks that have been set for us.

Q (Phillip Rury, Arthur D. Little, Inc.): Burt, since the tissue screening concentration (TSC) method seems to have validated the pertinence of aquatic water quality criteria to protecting aquatic biota from residue effects, how would you respond to Lynn McCarty's assertion that the superiority of chronic tests as a basis for regulatory criteria is a "myth?"

Burt Shephard:

I do not know if I would totally call it a myth. We are really measuring different sides of the same coin. If we expose the animal to the same concentration of chemical for a longer time, you begin to see chronic effects first. And if you keep exposing that hypothetical animal to that same concentration for a longer and longer time, you keep bioaccumulating more and more chemical. Eventually you will begin to run into acute toxicity, where you will reach a lethal body burden and the animal will expire. So, I think what we are really looking at is a temporal difference involving how long organisms are exposed to a given concentration. This is especially the case for chemicals that just keep on bioaccumulating the longer we expose them. You can start to see chronic effects at low tissue residues. As you gain more and more residue, you begin to get mortality.

Q (Phillip Rury): Lynn had not really elaborated on that comment, that zinger up there about it being a myth, and perhaps he would care to take this time to do so now?

Lynn McCarty:

What I was trying to caution against was the feeling that all we need is more chronic toxicity data and we will be able to solve all our problems. I definitely think that is not the case. And I think that there are better ways of obtaining that information than doing chronic toxicity testing in the way that we are doing it now. The point I am trying to make is that I think there are better ways of achieving the same end more cost-effectively using our knowledge, rather than having to create specific data points for every chemical and every situation that we want to look at. Chronic toxicity testing will not be our salvation and it is not the holy grail. It has to be taken into context.

Q (Peter Chapman): When I look at chemicals, including organics that come in via lipids and metals that are taken up via evolutionary mechanisms for uptake, I view them for our purposes here in two ways. Some chemicals that accumulate in organisms can be measured and this information may tell us something we can relate to effects. An example would be PCBs. Other chemicals, such as PAHs, accumulate in some organisms, but not in others because they are metabolized. Either we can look at the metabolites for organisms that metabolize the parent compounds or, as Jay pointed out, we can measure these chemicals in an organism that does not metabolize them. But in addition to that, within the group of chemicals that accumulate in organisms without forming metabolites, there are also chemicals that are regulated and those that are not. For instance, consider the essential metals. I think Burt made my point very well in his talk when he mentioned that copper and zinc proved to be problems for him. They proved to be problems because he was using bioconcentration factors that will not work for essential metals. These organisms must take up the essential metals to survive, and they will fight against the concentration gradient to retain them. I am wondering if we really should not look at the way the chemical acts. Maybe certain chemicals work better than others and we should focus our attention on these. Some chemicals may be a lost cause, and we should not put our effort into them for a variety of reasons. They have got complications that we should leave until later to address. Is that a reasonable way to look at this situation, or do you think, as a panel, that we should just go for it as a whole lot? What are your feelings?

Lynn McCarty:

I think that the sort of thing you want to do is what Burt has done. I only wish that I had done what he has done. I would have at least liked to have had the

opportunity, because I think it is an excellent example of how you both improve the understanding of the situation and point out the limitations. You have the opportunity to see where things work and, more importantly, to understand why they work. It also allows you to focus on the exceptions, yet they happen to be particularly problematic based on other information. So, I think the general danger in doing any of this residue-based approach is simply that we are looking at the basic paradigm of toxicology dose response and are trying to get a better understanding of the dose so we understand where our response comes from. There are situations where trying to make the methodology apply to all chemicals will make it so incredibly complicated and expensive that it is almost impossible to do. We basically do as is done in the sediment program by applying a tiered-testing approach. Essentially you focus on chemicals that can be addressed by simple assumptions with simple approaches. You only use more complicated evaluations and approaches to address the chemicals that do not fit into a simplified scheme. Understanding that they all are basically surrogates, it is simply a matter of not picking the right surrogate. This is related to Maurice's point earlier about hormone modulators. It does not negate the whole concept of the dose response in toxicology. It simply means that the dose surrogate you were using is not good enough for this particular situation.

Q (Peter Chapman): I know, but I think you are simply adding to my point that we need to be very careful and not delude people. If we do go to a tissue residue versus effects relationship, this is not going to work for everything. There are going to be some exceptions, and I think some very important exceptions. People get deluded in their thinking when they look at some of the data, because we are not always clear that we are talking about organics, lipids, and relationships that may be a little easier, than say, for the essential metals. And we have to be very clear about this. I agree with you whole-heartedly about the tiering approach, but I think my working hypothesis at this point is that we are going to reach the end of the rainbow. Eventually, we will develop a relationship for some chemicals under some circumstances, between effects and tissue body burdens. But we will not successfully do it for a number of others for a variety of reasons.

Lynn McCarty:

I appreciate that, but I just wanted to point out that I recognize that problem. I have been very careful in writing about this to try not to make it a be all and end all. The appropriate cautions or caveats are in there. Whether people actually see them, when they read it, is another story. But at least I think it is very, very important to do exactly what you said. The worst thing that could happen is to present this as the solution to everything, because it is not.

Burt Shephard:

I just might add to that a little. There is certainly no holy grail in this business. Clearly, the tissue residue approach is not going to work for everything. If we want to take the time and effort and money to quantify residues of metabolites of PAHs that are related to adverse effects, we can certainly do that. But if, on the other hand, we already have an approach in sediment quality criteria that seems to be pretty protective of our biological resources from the effects of PAHs, why do we need to look at metabolites at all? We have a method that works, so we should use it. If we need to use multiple methods for the laundry list of chemicals that we have to look at in this business, I certainly do not have a problem with using multiple methods. We should use sediment criteria where they are appropriate. If tissue residues work better for some chemicals or some situations, we should use them.

Q (John Connolly, HydroQual, Inc.): I have a comment and then a question that I think is related to the comment. The comment is that we have been using the term "ecological risk assessment" a lot, and yet everything I have heard is really referring to some sort of screening to evaluate chemicals of potential concern. I do not know if that is ecological risk assessment as much as it is just deciding whether or not there is a potential problem at a site. I think we need to make that distinction. The question is directed to Dave Mount. When we look at body burden relationships to toxicity, there have been some studies that have looked at relationships across the population, and they have shown that there is a range of body burdens. So there is a sensitive organism that responds at a low body burden, and then there is a very hardy organism that does not respond until you get to a very high body burden. That distribution of body burdens gives us information about population response that presumably would allow us to take the step beyond the screening tool to evaluate whether or not body burdens are potentially going to have a population effect. Given the way are you structuring the database, are you going to incorporate some of that kind of information that may allow us to take that step?

David Mount:

The answer, of course, is yes and no. There are several issues that you bring up. One is where there were ranges of concentrations for individual organisms within the population that were evaluated. You can consult the original citation to get more information on the ranges given in the database. There is also some variation in the literature. Studies either analyze the organisms that died, those that survived, or some combination thereof. Those notations are made in the database, so we may be able to use this in our analysis. I really believe that one of the critical uses of the database will be more as a pointer to answer specific questions and less as the endpoint in itself.

I might diverge a little bit and address the previous question. Certainly we are looking very actively at tissue residue-based approaches, but I think you have to bear in mind a couple things. One is they are most effective when you already have the tissue residue, which indicates they are directly applicable. An example would be a

bioaccumulation test that is done as part of a dredged material management monitoring program. A lot of the decisions that get made are not related, or are not dealt with, at the level of the tissue residue. You still have to bridge back to environmental concentrations that relate to those residues. And that reopens the whole bag of worms that we were trying to avoid by jumping to tissue residues. So, we cannot fool ourselves that there are no problems just because some data that seemed disparate collapse when we look at it on the basis of tissue residue. There is still the fact that those chemicals that collapse on the basis of tissue residue did not necessarily collapse on the basis of environmental exposure.

So, the tissue residue approach is not so much a direct interpretive tool, but it may teach us about groups of chemicals or making estimates of acute or chronic effect thresholds for chemicals that we have relatively little data for. It does straighten out some QSAR relationships that were formerly based on water concentrations, but were muddled by differences in uptake or something else. And to me that is the real scientific importance of the concept. I think the direct regulatory significance is almost secondary. We need to make use of all the information we have. But there are a relatively small number of instances where that information is necessarily directly relevant. For example, in a risk assessment, if you have extremely high residues, you have some information about existing risk. Almost always what is of interest in a risk assessment is future risk or risk under various management alternatives. And unless you can link those up, it will not do all the good you want it to.

Q (Hector Laguette, Brown and Root Environmental): A considerable amount of the discussion so far on tissue residues has been based on lipid-normalized values, and I wonder if any consideration has been given to the possible effect of the contaminants themselves on the lipid metabolism of the organisms prior to the moment when we do this normalization of concentrations. How may this artifact be affecting some of the approaches that we are talking about?

Burt Shephard:

On the database that we compiled, less than 25 percent of the papers that we compiled reported the lipid content of the species. So, it is really hard to make a judgement, at least on what I have looked at. I do not know how Dave feels about that.

David Mount:

Very true. Somebody mentioned this morning that if they report lipid data, there are some issues of how it was measured and how relevant that measure may be. To support your point, I think Peter Landrum presented data this morning to show exactly how lipid metabolism affected interpretation of residue-based data. In that case, lipid normalization tended to explain the variation rather than confound it. But it is a relevant point.

Lynn McCarty:

One of the things I have a great deal of concern about is actual lipid normalization of toxicity test results. It is perfectly reasonable to do it for bioaccumulation purposes, but when you normalize whole body residue levels to a standard lipid content, you are making the assumption that the whole body lipid content is reflective of the lipid content at the site of toxic action. This is not an assumption I would care to make. And we have very little information about that sort of thing. So, I think you have to be very cautious in normalizing the data when you are talking about toxicity. However, Burt has done this and I think it has worked out. But we are doing it out of ignorance, not necessarily out of knowledge. The fact that it worked is not a reflection of whether it is right or not. Maybe we were just lucky that time. Until we understand that, we will have to be very careful about normalizing toxicity data to lipid content.

Burt Shephard:

I am not sure I would agree that it was luck. There is good reason to suspect that it would work out, but it is an assumption. I will grant you that.

Q (Hector Laguette): I guess just from the point of view of ecological risk assessment, it is one more of those things that ends up being in the uncertainty analysis. It is something that should be considered at the end.

Burt Shephard:

Another problem with lipid normalization is that the lipid content of many species varies seasonally or annually, so how do you take that into account as well? The types of lipids also vary. There are a lot of assumptions. I do not know if I was lucky or if the EPA data that I based my data on was good. It might be a little bit of both. In this case, it seemed to work out, but there is certainly some concern about lipid normalization.

David Mount:

I think one of the issues that really comes to the floor when we start talking about all these other variables is that some of these principles work very well, in general, and they make good predictions of mean responses across groups of chemicals. But there are subtleties in organismal factors, physical and chemical factors, or all sorts of other things that cause individual chemicals to deviate from that behavior. In a lot of regulatory programs, that deviation is not considered acceptable. Making your best estimate and constructing a worst reasonable case are two very different tasks. There are exceptions that people consider to be quite relevant. Some of these exceptions are not accounted for in some of the very generalized models that we use in this sort of analysis. We all use log log plots, and the noise around a log log plot is important to the decision that gets made.

Burt Shephard:

The decision is also based in part on what you are going to use your data for. Dave and I both did literature reviews, but, in some cases, we had very different criteria as to how we decided a paper could or could not be used in our literature, just because we had different uses for the data.

Q (Tom O'Connor, NOAA): I did have a question for Burt Shephard that addresses this issue of how extensive the problem of coastal contamination is. Your toxic threshold concentration for cadmium, as I recall, was about 0.04 parts per million. If I convert that to a dry weight number, that is something like 0.2 parts per million. I think that number is exceeded by most of the mussels and oysters in the United States. Are we to conclude that the contamination has put all these animals at risk?

Burt Shephard:

We have run into the same problem. I will use cadmium as an example. I have spent part of the summer up in the Aleutian Islands where very few point sources occur. We have some blue mussel data from up and down the Aleutian Island chain that we have been collecting for background information for use in a risk assessment at a military base closure site in the Aleutians. As fate would have it, the typical cadmium concentrations are about half a part per million. We have a number of mussels from various sites with no known point sources over one part per million. That may be just the natural background. For some reason, the mussels seem to be doing fine there. Something I did not talk about at all, especially for metals, is naturally occurring compounds. It is very important in the risk assessment to do a proper background comparison with your site data. Background comparisons can be done several ways. You can do the mean of your sample population versus your background population. You can also do, for lack of a better term, hot spot comparison, comparing a high end mussel versus some part of your distribution. You asked if I thought the mussels are contaminated nationwide and showing effects. No, I do not. But very clearly, some other species is going to show an effect at half a part per million. I mentioned earlier that if you have species specific information, that is obviously the best way to do a risk assessment. If you have a range of data for blue mussels, and you know that half a part per million cadmium causes no adverse toxicological or ecological effect on blue mussels, then you would certainly use that in preference to a tissue screening number.

David Mount:

I think that addresses a real hazard and what I consider a real abuse of a lot of assessment tools. We had a discussion last week about one-tailed and two-tailed criteria. If you are below a one-tailed criteria, for example, you are confident that there is no effect. But there is no implication of effect if you exceed that. If you look at the derivation of the tissue screening number, they are entirely one-tailed from the way that they were developed. There is no reason to infer effect from an exceedance. In fact, if you look at the way water quality criteria were developed and the way they were written, they are really one-tailed criteria. But people consistently infer effect from an exceedance of a criterion, which is not completely wrong. You should recognize, though, that when you make that inference, you are buying into a set of assumptions that may or may not apply. So, the exceedance of one of the screening levels in a healthy organism should not come as a surprise to any of us. The question is whether or not you consider the generalizations that went into the derivation of that number.

Q (Tom O'Connor): In that light at the other extreme, Jay Field had a lot of data for PCBs in the fishes of the Hudson River. Jay, what did you have for effects of these PCBs?

Jay Field:

We were comparing tissue concentrations of PCBs to literature-derived effect concentrations for total PCBs and dioxin, using dioxin equivalent values for coplanar PCBs. We did not measure effects in Hudson River fish directly.

Q (John Haggard, General Electric Company): Jay, one of the things we are planning on the Hudson River is to investigate and remediate active water column sources of PCBs. We believe they are influencing the top surface sediments. The subject of a lot of the debate over the years on remediation has been buried sediments, which have different PCB congener signatures. In your work, Jay, with the congeners and the fish, have you been able to sort out the sources of the PCBs based on the congener distributions, or are you still working on that?

Jay Field:

No, we did not attempt to distinguish among water column, surface sediment or subsurface sediment sources. I think you need other information to do that. You have in-place sediment, recent releases of material through ground water or non-aqueous phase layers as you have at Bakers Falls. You also have sediment that is resuspended and/or transported down river in every spring flood. So separating out what is coming to the fish via the water column (either suspended or dissolved) from recent releases or sediment transport from past years is difficult to determine based on congener pattern alone. But the congener patterns in fish show a clear signal of what they are exposed to at different locations along the river gradient.

Session Three: Modeling Bioavailability of Sediment Contaminants

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Equilibrium Partitioning and Organic Carbon Normalization

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Estimating Bioaccumulation Potential in Dredged Sediment Regulation

Philip M. Cook
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Development of Bioaccumulation Factors for Protection of Fish and Wildlife in the Great Lakes

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From Modeling to Criteria: Integrated Approach to Criteria Development

Equilibrium Partitioning and Organic Carbon Normalization

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Establishing sediment quality criteria (SQC) requires a determination of the extent of the bioavailability of sediment-associated chemicals. A full discussion of the Equilibrium Partitioning (EqP) approach to developing SQC is presented elsewhere (Di Toro et al., 1991; USEPA, 1993). The main focus of this presentation is field observations of partitioning in sediments. Consider a sediment sample that is segregated into various size classes after collection. The particles in each class were in contact with the pore water. If sorption equilibrium has been attained for each class, then letting $C_s(j)$ be the particle chemical concentration of the jth size class, it is true that

$$C_s(j) = f_{oc}(j)K_{oc}C_d \qquad (1)$$

where $f_{oc}(j)$ is the organic carbon fraction for each size class j, K_{oc} is the partition coefficient for sediment organic carbon, and C_d is the pore water concentration. On an organic carbon-normalized basis, this equation becomes

$$C_{s,oc}(j) = K_{oc}C_d \qquad (2)$$

where $C_{s,oc}(j) = C_s(j)/f_{oc}(j)$. This result indicates that the organic carbon-normalized sediment concentration of a chemical should be equal in each size class because K_{oc} and C_d are the same for each size class. Thus a direct test of the validity of both organic carbon normalization and EqP would be to examine whether $C_{s,oc}(j)$ is constant across size classes in a sediment sample.

Data from three field studies, Prahl (1982), Evans et al. (1990), and Delbeke et al. (1990), can be used to test this prediction. In Prahl's study, sediment cores were collected at three stations near the Washington State coast (Stations 4, 5, and 7). These were sieved into a silt-and-clay-sized fraction (<64 μm) and a sand-sized fraction (>64 μm). This latter fraction was further separated into a low-density fraction (<1.9 g/cm^3) and the remaining higher-density sand-sized particles. The concentrations of 13 individual polycyclic aromatic hydrocarbons (PAHs) were measured in each size fraction.

It is important to realize that these size fractions are not pure clay, silt, or sand, but are natural particles in the size classes denoted by clay, silt, and sand. The organic carbon fractions, shown on Figure 1, range from 0.2 percent for the high-density sand-sized fraction to greater than 30 percent for the low-density fraction. This exceeds two orders of magnitude and essentially spans the range usually found in practice. For example, 90 percent of the estuarine and coastal sediments sampled for the National Status and Trends program exceed 0.2 percent organic carbon (NOAA, 1991).

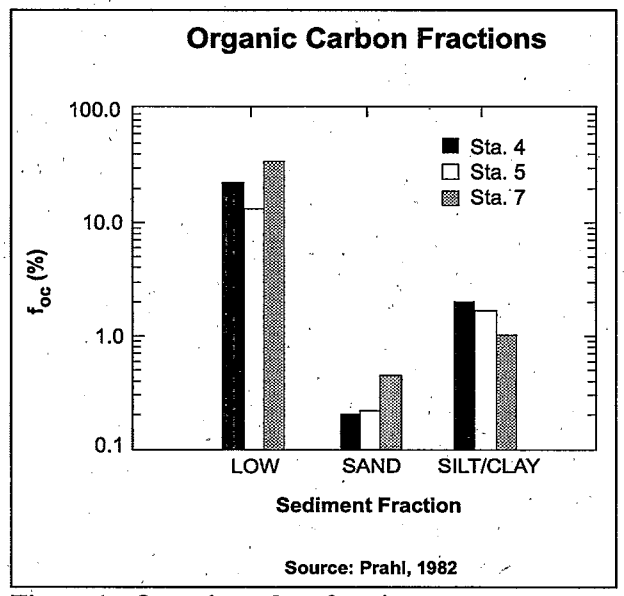

Figure 1. Organic carbon fractions: Sediment fraction.

Figure 2 (top) compares the dry weight-normalized clay/silt-sized fraction sediment PAH concentrations, $C_s(j)$, to the sand-sized high- and low-density PAH concentrations on a dry weight basis. The dry weight-normalized data have distinctly different concentrations—the low-density high-organic carbon fraction is highly enriched, whereas the sand-sized fraction is substantially below the clay/silt fraction concentrations. Figure 2 (bottom) presents the same data but on an organic carbon-normalized basis, $C_{s,oc}(j)$. In contrast to dry weight normalization, the PAH concentrations are essentially

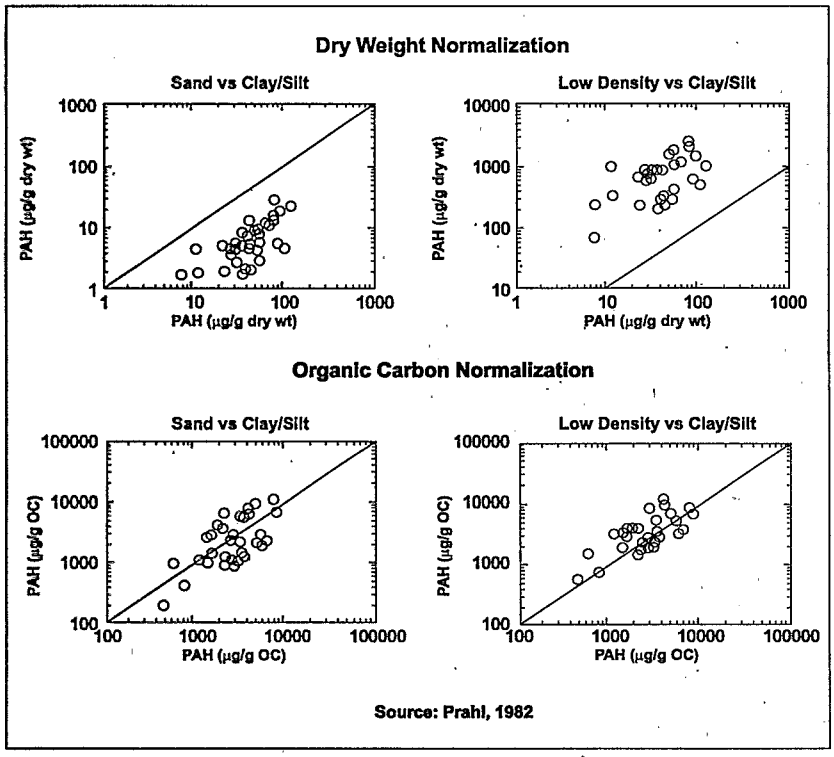

Figure 2. Dry weight normalization and organic carbon normalization.

the same in each size class, as predicted by Equation 2. The lines in Figure 2 represent equality and have been added as a visual aid.

In the field data from Evans et al. (1990), sediments were collected at five sites along the River Derwent, Derbyshire, United Kingdom, and separated into six sediment size classes. The size classes were representative of clay and silt (<63 µm) to coarse sand (1.0 to 2.0 mm). Organic carbon content and total PAH were measured in each sediment size class. Figure 3 presents the size classes and associated organic carbon content. Evans et al. attribute the bimodal distribution of f_{oc} to two types of organic matter. Organic matter in the 1.0 to 2.0 mm size class may be from fragmentary plant material, while the size classes less than 500 µm have organic carbon content that is the result of aging humic material. The organic content in this study ranges from 2.0 to 40 percent.

Figure 4 presents a comparison of PAH concentration for different sediment classes for dry weight normalization and organic carbon normalization. The top left panel compares PAH concentrations on sand (63-500 µm) and clay/silt (<63 µm) on a dry weight basis. The top right panel compares PAH concentrations on coarse sand (0.5-2.0 mm) and clay/silt (<63 µm) on a dry weight basis. The data indicate that the PAH concentration is higher in the coarse sand fraction of sediment. Recall from Figure 3 that the clay/silt and coarse sand fractions contain higher fraction organic carbon content. The bottom panels of Figure 4 present the organic carbon-normalized comparison of PAH concentrations by sediment class. For both panels, the organic carbon-normalized PAH concentrations are similar regardless of the sediment size class as predicted by Equation 2. The lines in Figure 4 represent equality and have been added as a visual aid.

Lastly, Delbeke et al. (1990) collected sediments from seven sites in the Belgian continental shelf and the Scheldt estuary. These sites were analyzed for eight polychlorinated biphenyl (PCB) congeners and organic carbon in the bulk sediment and clay/silt (<63 µm) sediment fraction. In addition, analyses of the samples were done to determine the percent of size fractions ranging from 500 µm to 3 µm that made up the sample. The PCB congeners tested for in this study were IUPAC numbers 28, 52, 101, 118, 153, 138, 170, and 180.

Using concentrations reported for bulk samples, concentrations for clay/silt samples, and percent size fractions of each sample, calculations were done to estimate concentrations on the >63 µm portion of the sample. Similar calculations were done to determine

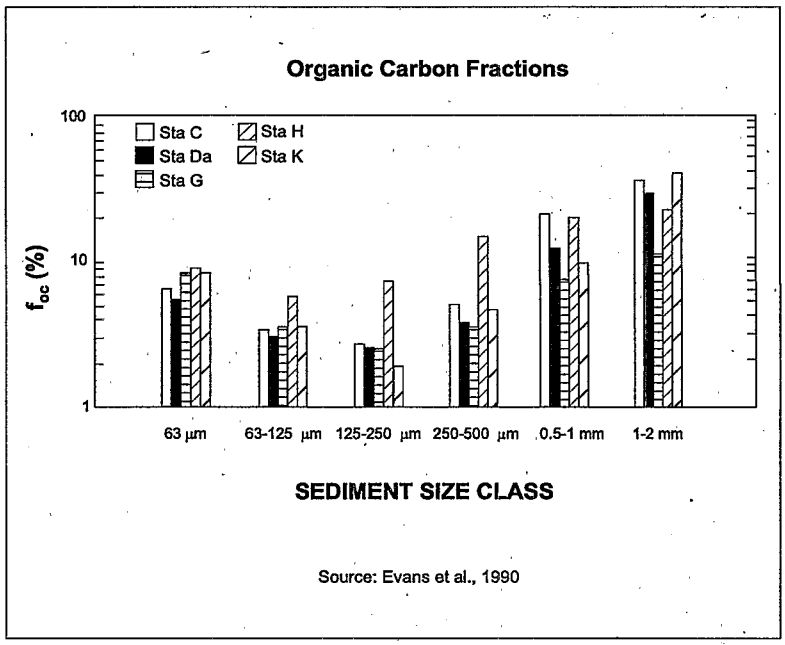

Figure 3. Organ carbon fractions: Sediment size class.

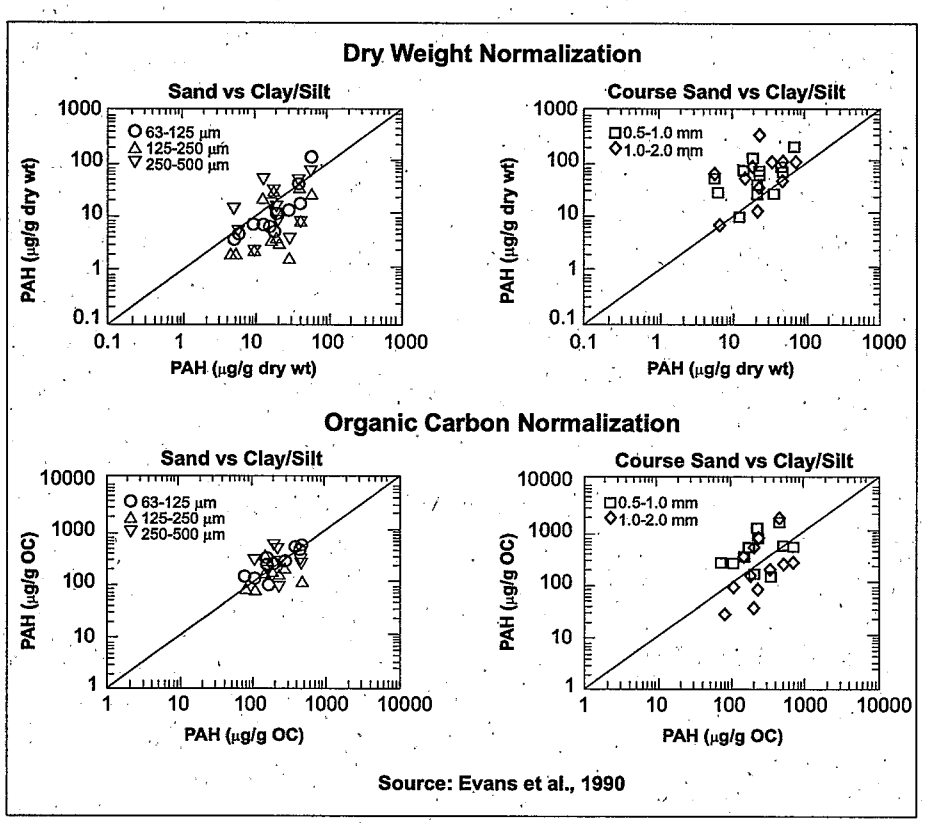

Figure 4. Dry weight normalization and organic carbon normalization.

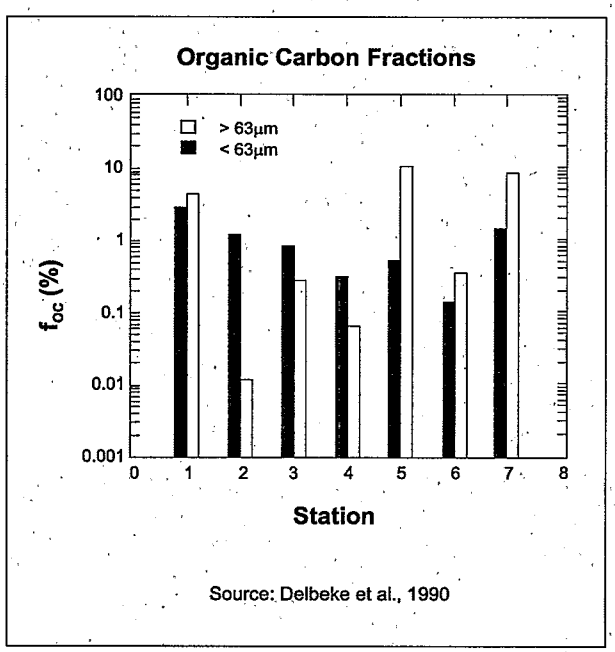

Figure 5. Organic carbon fractions: Station.

organic carbon content on the >63 µm portion of the sample. Organic content varied from 0.01 percent to 10 percent inclusive of both <63 µm and >63 µm portions of the sediment.

Figure 5 presents the percent organic carbon on the <63 µm portion of the sample (black bar) and on the >63 µm portion of the sample (white bar). Comparisons of the PCB congener concentrations on a dry weight basis (top) and on an organic carbon basis (bottom) are shown in Figure 6. Organic carbon content in the >63 µm class size at stations 2 and 4 is 0.01 percent and 0.06 percent, respectively, as indicated in Figure 5. The data for these stations are shown on Figure 6 using filled symbols. Though an f_{oc} >0.2 percent has been presented as the value for which organic carbon normalization applies, normalization at these f_{oc} values seems appropriate for this data set.

The top panel of Figure 6 indicates no evident relationship between PCBs in the <63 µm sample and PCBs in the >63 µm sample on a dry weight basis. When concentrations in either class size are normalized to organic carbon content, the concentrations are similar for both class sizes as shown in the bottom panel. This indicates that PCB concentrations are similar across sediment class sizes, which supports organic carbon normalization. The lines in Figure 6 represent equality and have been added as a visual aid.

The data from Figures 2, 4, and 6 are analyzed to quantify the reduction in variability when the data are carbon-normalized. The variation of the y-axis values from the line of unity presented in Figures 2, 4, and 5 represents the residual difference of the y values from the x values. Coefficients of variation for these residuals are presented in Figure 7 for dry weight- (black bar) and organic carbon-normalized (gray bar) data. The coefficients of variation are reduced significantly. It can be concluded from the data of Prahl (1982), Evans et al. (1990), and Delbeke et al. (1990) that the organic carbon-normalized PAH and PCB concentrations are relatively independent of particle size class and that organic carbon is the predominant controlling factor in determining the partition coefficient of the different sediment size particles in a sediment sample. The organic carbon concentration of the high-density sand-sized fraction in Prahl's data (0.2 to 0.3 percent) suggests that organic carbon normalization is appropriate at these low levels. The data from Evans et al. suggest that EqP can be applied to organic carbon originating from more than one source, that is, fragmentary plant matter and aging humic material.

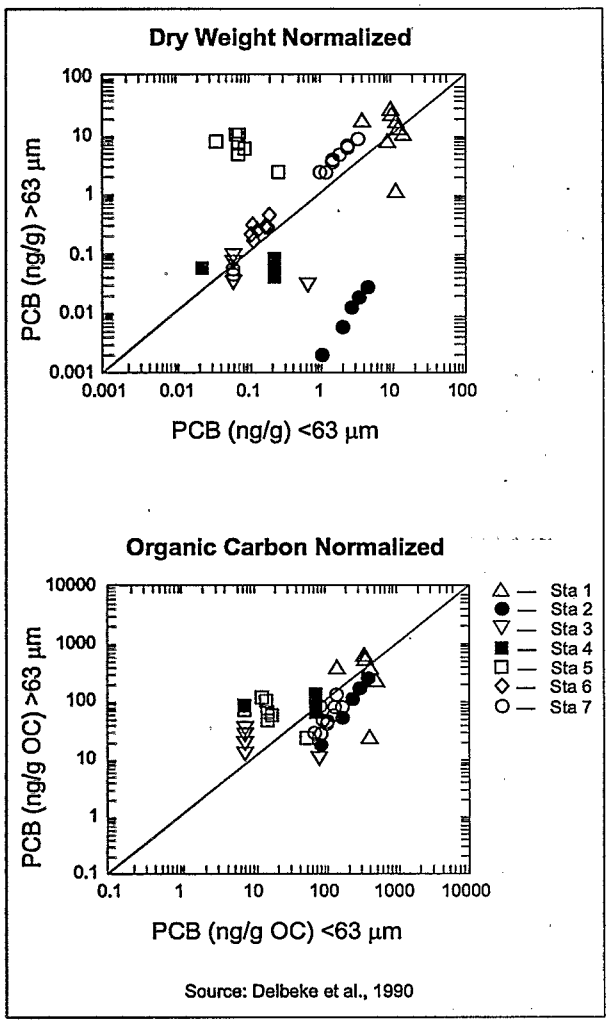

Figure 6. Dry weight normalized and organic carbon normalized.

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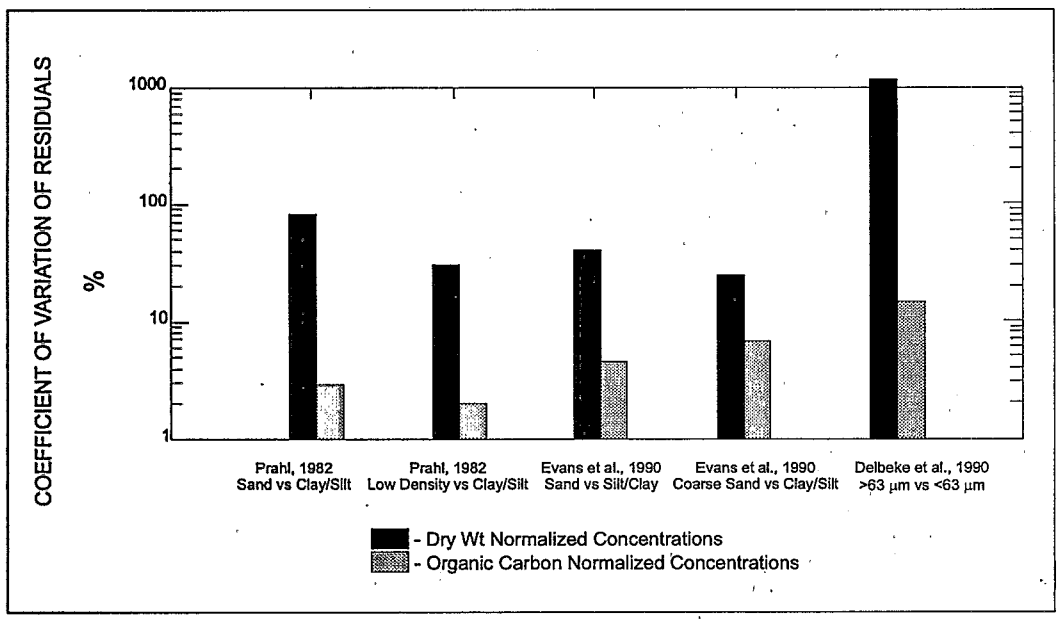

Figure 7. Coefficients of variation.

Estimating Bioaccumulation Potential in Dredged Sediment Regulation

Victor A. McFarland
U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, Mississippi

The bioaccumulation potential of neutral organic chemicals in dredged sediments is estimated from sediment chemistry. A simple model is used in which concentration data are normalized on sediment organic carbon and organism lipid and a biota/sediment accumulation factor (BSAF) is applied. This presentation describes an experiment comparing predicted with actual measured tissue concentrations of polynuclear aromatic hydrocarbons (PAH) bioaccumulated from sediments by the clam *Macoma nasuta*.

Testing Requirements

Open water disposal of sediments dredged for navigation purposes in the United States is regulated under provisions of section 103 of the Ocean Dumping Act (Marine Protection, Research, and Sanctuaries Act, MPRSA, of 1972) and section 404 of the Clean Water Act (CWA of 1972). Both laws require that the potential for bioaccumulation of toxic chemicals associated with dredged sediments be assessed as part of the regulatory process when such chemicals are suspected or known to be present. The guidance manuals for implementation of the laws use a tiered approach (USEPA/USACE, 1991, 1994). In the first tier, historical data, if it exists, is used to judge whether there is a reason to believe that bioaccumulating chemicals may be present and may be bioavailable to exposed organisms. In the second tier, bioaccumulation potential is estimated from sediment chemistry data. If the potential for bioaccumulation appears to exist, then 28-day laboratory exposures of benthic organisms and analysis of their tissues may follow in the third tier. A fourth tier exists for special situations in which a decision could not be made in any earlier tier.

Theoretical Bioaccumulation Potential

The Tier II sediment screening test is referred to as "Theoretical Bioaccumulation Potential" (TBP) and is based on equilibrium partitioning. TBP is used only for neutral organic chemicals and is simply the application of a BSAF to sediment concentration data in order to estimate the concentration that would result in an organism exposed to the sediment as its only source of contamination. In aquatic systems, neutral organic chemical contaminants such as the PCBs, PAHs, dioxins, etc. primarily partition to the organic carbon of sediments and the lipids of biota. Recognition of this fact led to the convention of normalizing concentrations of neutral chemicals in sediments on the basis of their total organic carbon (TOC) content and similarly normalizing concentrations in biota on the basis of their lipid content. BSAFs are the ratio of the normalized concentrations. Multiplying the TOC-normalized concentration of a neutral chemical in sediment by the BSAF provides an estimation of the steady-state body burden of that chemical in a sediment-exposed organism:

$$TBP = 4(C_s/\%TOC)\%L \qquad (1)$$

where C_s = concentration in whole sediment, L = total extractable lipids in an organism of interest, and "4" is the value assigned representing a generalized BSAF for all neutral chemicals and biota.

In theory, the BSAFs for individual neutral chemicals should not differ greatly from one another and can be approximated by a single value. However, the model is a simplification that does not account for differences in bioavailability, the metabolism of compounds, disequilibrium and non-constancy of exposure, organism feeding behavior, or any of the other kinetic or measurement processes that influence bioaccumulation. As such, TBP is protective but not necessarily predictive.

Empirical BSAFs

BSAF data are compiled in a database accessible by modem through an electronic Bulletin Board System at the U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, MS (The Contaminants BBS, 601-634-4380). Entries include data on organism lipid content, lipid extraction method, organic carbon content of sediment, length of exposure, etc., and the source of the data is given. The data are severely skewed, with most

BSAF values reported as less than 1.0. The frequency distribution of all BSAF entries presently in the database is shown in Figure 1. These data contrast sharply with the universal AF used in the Tier II TBP calculations. The universal BSAF (4) is at the 94th percentile of the database and is about 12-fold greater than the median. Table 1 shows the medians and the interquartile ranges of the BSAFs in the database by group. It can be seen that the medians for the chlorinated groups (PCBs, dioxins/furans, other chemicals) did not differ statistically. However, the median PAH BSAF ranged 16- to 22-fold lower than any of these and is lower than the universal BSAF by a factor of 125. The BSAF data were compiled with the expectation that their use in Tier II dredged sediment bioaccumulation potential calculations would provide more predictive estimations than does the current practice. This assumption was tested using the priority pollutant PAH data of two studies conducted at the Waterways Experiment Station (McFarland et al., 1994a, 1994b; McFarland, 1995).

Field Study

Sediments and their resident infaunal and benthic polychaetes and bivalve mollusks were collected at a location on the continental shelf of the New York Bight Apex. The location had low, but in most cases detectable, concentrations of PAH compounds. Sediment TOC content averaged 0.5 percent in the four samples taken. The biota were pooled by taxonomic group and their tissues analyzed for PAHs and for lipids. Three to five replicates of the seven taxonomic groups were possible, resulting in a total of 24 samples. The sediment and organism PAH data were used to calculate BSAFs for the 15 individual PAHs and for the total of the 15 using "Bootstrap" estimation methods (Manly, 1991).

Laboratory Study

Two sediments were collected in the San Francisco Bay system. One was surficial material from the central Bay (designated "Reference") and the other was from an inner harbor turning basin known to be contaminated (designated "Contaminated"). The total PAH concentration of Reference was about 1.6 $\mu g\ g^{-1}$ and that of Contaminated was about 50 $\mu g\ g^{-1}$. TOC contents were similar (0.926 and 1.10 percent). Bentnose clams, *Macoma nasuta*, were collected from a clean area (Tomales Bay, California) and exposed to the sediments in a flow-through facility for 28 days. Exposures were to either bedded sediment or to sediment continuously suspended at about 50 mgL^{-1}. At the end of the exposures the tissues of the clams were analyzed for the 15 priority pollutant PAHs, as in the field study sediments and organisms. Lipids were also measured.

Bioaccumulation potential of the PAH compounds in the San Francisco Bay sediments was estimated using the TBP model and the PAH BSAFs generated by the New York Bight field study. The lipid input data were the mean concentrations of lipids in the Tomales Bay clams at the end of the 28-day exposures in each treatment. The predicted (TBP) PAH concentrations are plotted against the concentrations measured at the end of the 28-day exposures in Figure 2 for the Reference sediment and Figure 3 for the Contaminated sediment. Vertical and horizontal bars and caps are standard errors. The diagonal line represents perfect agreement between predicted and observed concentrations. Observations falling below the line are cases in which measured values exceed predicted. Observations above the line are the reverse. Overestimates and underestimates are nearly equal in the Reference sediment exposures (15 and 17, respectively) and are equal in the Contaminated sediment exposures (16 and 16).

Figure 4 identifies the PAH compounds and shows the distribution of the ratioed means. In the Reference sediment exposures, 56 percent of the predicted concentrations were within a factor of 2 of the measured concentrations. In the Contaminated sediment exposures, 63 percent were within a factor of 2. For the Reference, 72 percent were within a factor of 3 and 75 percent of the Contaminated were within a factor of 3. PAH compounds for which the predicted/measured concentration exceeded or was less than a factor of 3 were acenaphthylene, naphthalene, fluorene, dibenz[a,h]anthracene, and pyrene in the Reference sediment exposures. In Contaminated sediment exposures similar results were observed for acenaphthylene, anthracene, fluorene, and indeno[1,2,3-cd]pyrene. Only for anthracene, fluorene, and naphthalene were statistically significant ($P<0.05$) differences found when t-tests were performed comparing the bootstrap mean TBPs with the mean measured 28-day concentrations.

Although there were individual cases showing large differences between predicted and measured concentrations, Figures 2 through 4 reveal numerous close correspondences. In the case of the total (of the 15) PAH, the effect of averaging over- and underestimations results in very good correspondence. For the totals, the poorest correspondence was 0.76 for the suspended Contaminated sediment exposures, and the closest correspondence was a ratio of 0.97 for total PAH in the suspended Reference sediment exposures.

Acknowledgments

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TABLE 1. DESCRIPTIVE STATISTICS AND COMPARISONS OF GROUPED DATA

Group	n	Median BSAF	25%ile	75%ile	t-test on rankits
All BSAF	689	0.520	0.139	1.453	A*
Invertebrates	608	0.440	0.101	1.120	A
Fish	81	1.600	0.735	2.600	B
Field Studies	492	0.337	0.065	1.525	A
Laboratory Studies	197	0.670	0.428	1.393	C
PCBs	404	0.718	0.230	1.820	C
PAHs	110	0.032	0.006	0.299	D
Dioxins/Furans	129	0.514	0.269	1.275	C
Other Chemicals	46	0.598	0.065	1.800	AC

*Groups with same letter are not significantly different, P < 0.05

FIGURE 1. FREQUENCY DISTRIBUTION OF BIOTA/SEDIMENT ACCUMULATION FACTORS (BSAF) IN CONTAMINANTS BBS DATABASE

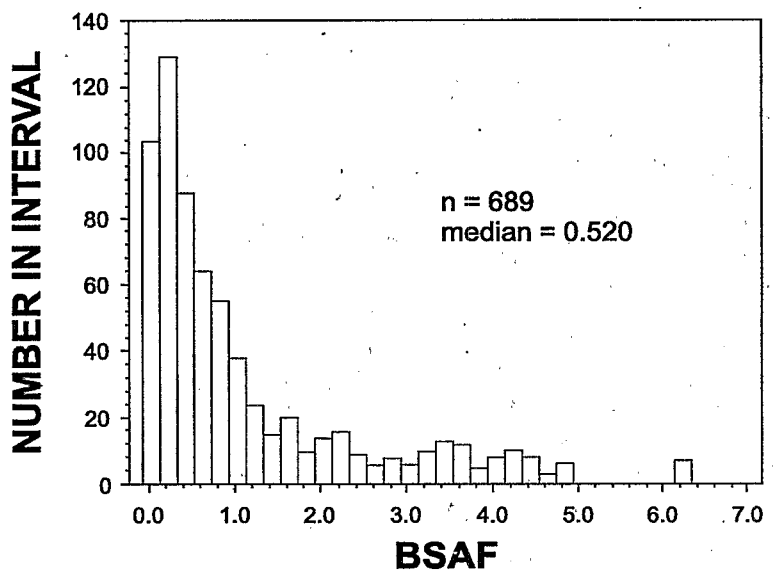


FIGURE 2. TBP-PREDICTED vs MEASURED PAH BIOACCUMULATION IN CLAMS EXPOSED TO REFERENCE SEDIMENT

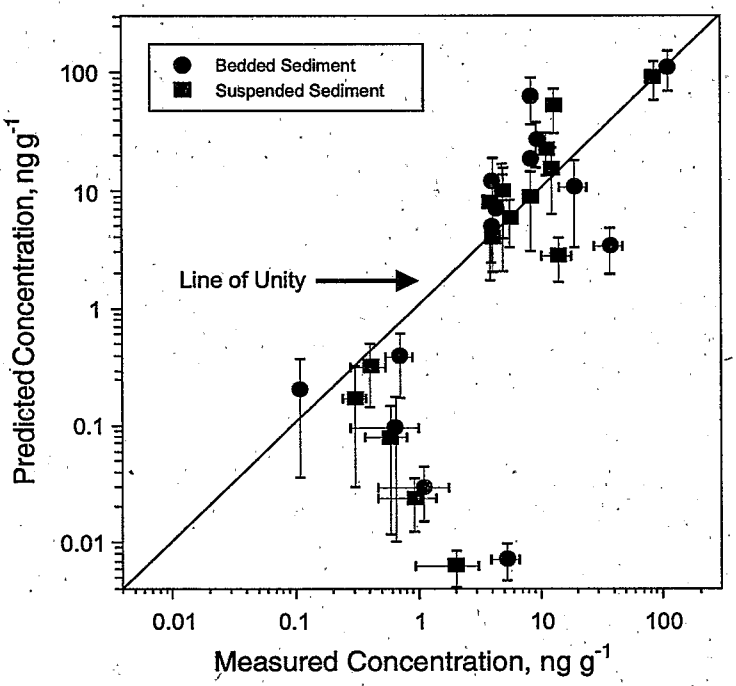


FIGURE 3. TBP-PREDICTED vs MEASURED PAH BIOACCUMULATION IN CLAMS EXPOSED TO CONTAMINATED SEDIMENT

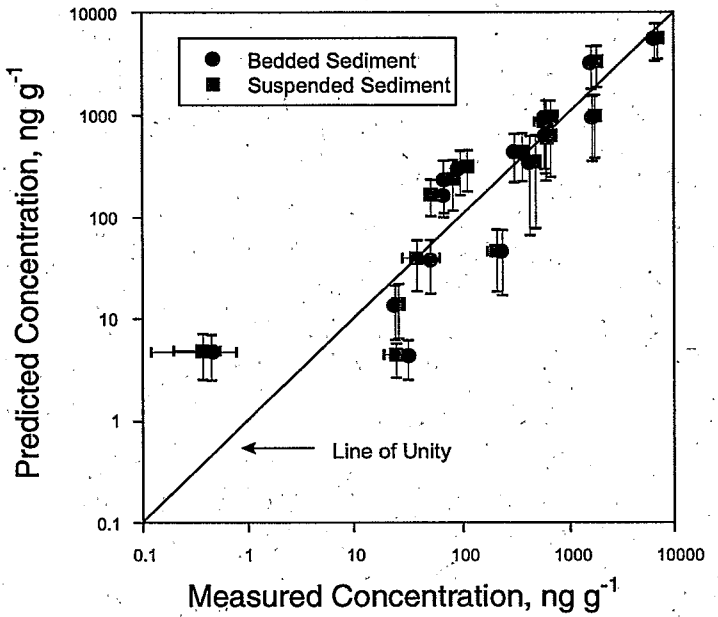
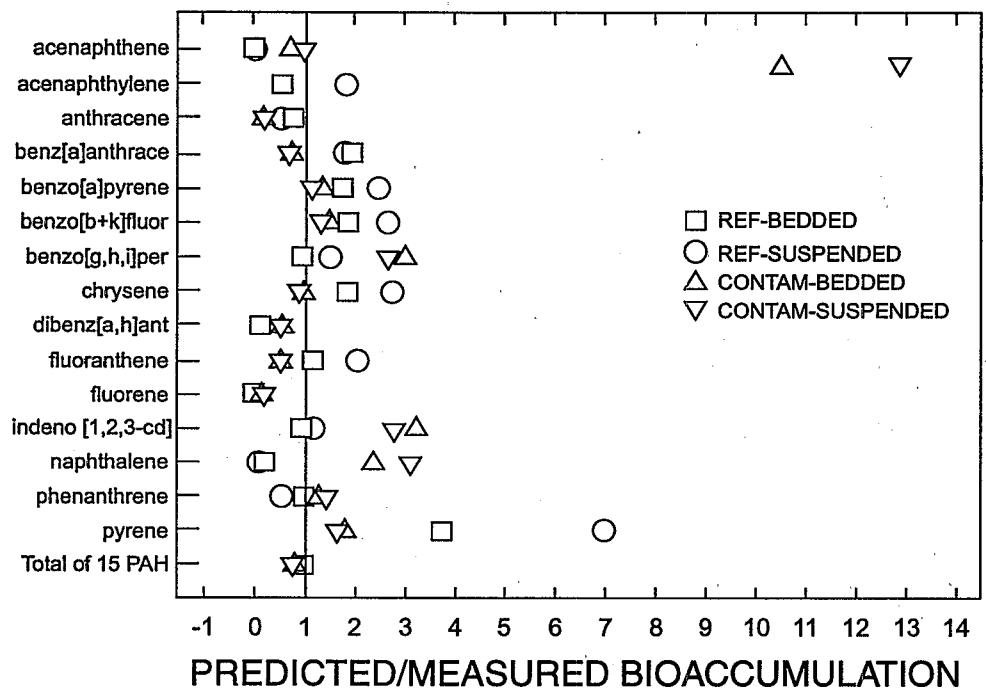


FIGURE 4. AGREEMENT BETWEEN PREDICTED AND MEASURED BIOACCUMULATION: TBP/TISSUE CONCENTRATIONS



ESTIMATING BIOACCUMULATION POTENTIAL IN DREDGED SEDIMENT REGULATION

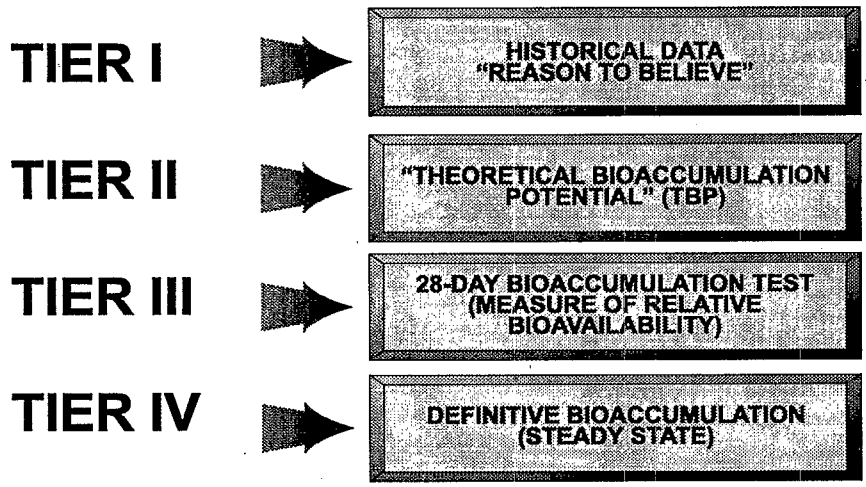
Victor A. McFarland

USAE Waterways Experiment Station
Vicksburg, Mississippi

OPEN-WATER DREDGED MATERIAL DISPOSAL IS REGULATED BY:

- **Section 103 of PL-532 of 1972, the Marine Protection, Research and Sanctuaries Act (coastal waters)**
- **Section 404 of PL-530 of 1972, the Clean Water Act (inland and coastal waters)**

THE IMPLEMENTATION MANUALS USE A TIERED-TESTING APPROACH



APPROPRIATE USE OF TBP:

"The TBP calculation in Tier II is applied as a coarse screen to predict the magnitude of bioaccumulation likely to be associated with nonpolar organic contaminants in the dredged material."*

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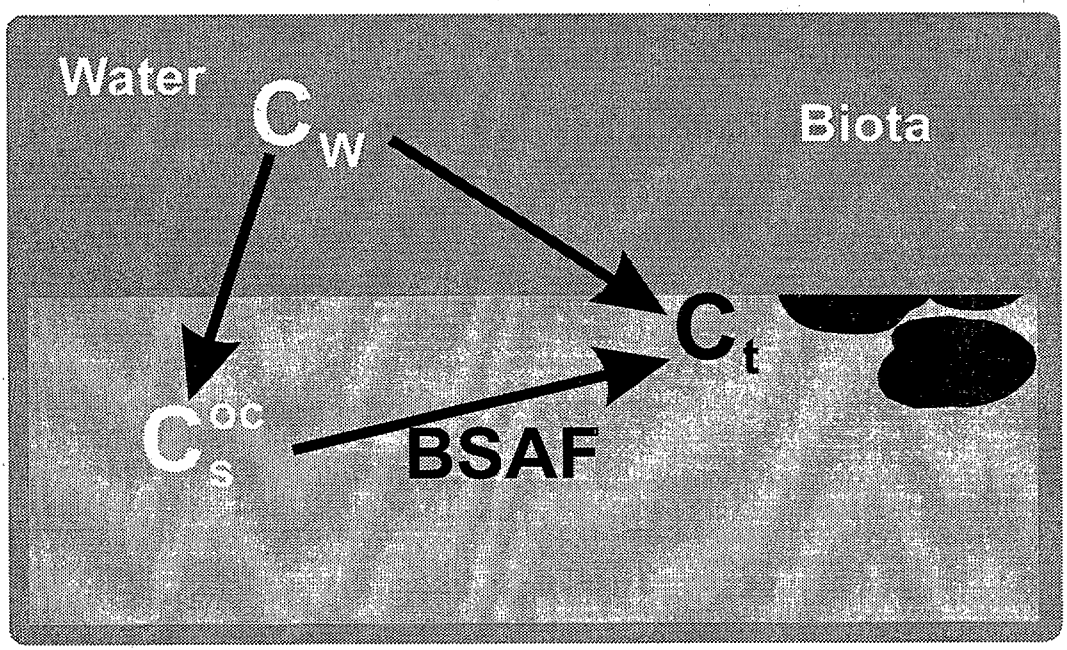
*Draft Inland Testing Manual, June 1994, p. 120.

TBP:

- Is based on equilibrium partitioning
- Applies only to neutral chemicals
- Uses sediment chemistry data
- Normalizes concentration data on:
  - Organic carbon in sediments
  - Lipid in exposed organisms

EQUILIBRIUM PARTITIONING



SOLUBILITIES OF SOME ORGANIC SOLUTES IN ORGANIC SOLVENTS OR IN WATER, g/mL

	TOLUENE OR BENZENE	OLIVE OIL OR PEANUT OIL	CHLORO-FORM	CARBON TETRA-CHLORIDE	WATER
NAPHTHALENE	0.29	0.13	0.50	0.50	8.80E-7
PHENANTHRENE	0.42	NA	NA	0.42	1.06E-9
LINDANE	0.29	NA	0.24	NA	1.70E-9
DDT	0.78	0.11	NA	0.45	3.10E-12

TBP = 4(Cs/%TOC)%L

TBP = Estimated tissue concentration
4 = An "appropriate" bioaccumulation factor
Cs = Chemical concentration in sediment
TOC = Sediment total organic carbon
L = Organism lipid

☆ **The factor, 4, is a universal biota/sediment accumulation factor (BSAF)**

DESCRIPTIVE STATISTICS AND COMPARISONS OF GROUPED DATA

Group	n	Median BSAF	25%ile	75%ile	t-test on rankits
All BSAF	689	0.520	0.139	1.453	A*
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Dioxins/Furans	129	0.514	0.269	1.275	C
Other Chemicals	46	0.598	0.065	1.800	AC

*Groups with same letter are not significantly different, $P < 0.05$

TWO STUDIES:

- **East Coast**
- **Field study**
- **Collected benthic infauna and sediments**
- **Analyzed both and calculated BSAFs**

→

- **West Coast**
- **Lab study**
- **Collected high and low contamination sediments and analyzed**
- **Collected mussels and clams from clean site**
- **Used field study BSAFs to calculate TBPs**
- **Exposed bivalves to the sediments**
- **Compared TBP estimations with actual bioaccumulation**

MEANS COMPARISONS

- Performed t-tests on bootstrap mean TBP and measured mean 28-day tissue concentration for each compound in each treatment
- 5-6 replicates of each treatment
- 1,024 bootstrap iterations for each comparison
- only anthracene, fluorene, and naphthalene were significantly different at $P < 0.05$

CONCLUSION

- For PAH compounds and probably for other neutral organics empirical BSAFs can provide reasonably good estimates of bioaccumulation potential
- Estimates for total PAHs are very good

Development of Bioaccumulation Factors for Protection of Fish and Wildlife in the Great Lakes

Philip M. Cook and Dr. Lawrence P. Burkhard
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Duluth, Minnesota

Bioaccumulation factor (BAF) development for application to the Great Lakes, and in particular for the recent Great Lakes Water Quality Initiative (GLWQI) effort of U.S. EPA and the respective Great Lakes states, illustrates the importance of the linkage between sediments and the water column and its influence on exposure of all aquatic biota. This presentation included a discussion of the development and application of bioaccumulation factors for fish, both water-based BAFs and biota-sediment accumulation factors (BSAFs), with emphasis on the role of sediments in bioaccumulation of persistent, hydrophobic non-polar organic chemicals by both benthic and pelagic organisms. Choices of bioaccumulation factors are important because they will strongly influence predictions of toxic effects in aquatic organisms, especially when chemical residue-based dose-response relationships are used.

There were two principal bioaccumulation factor expressions used by the GLWQI. The BAF_{ℓ}^{fd} is based on lipid-normalized concentration of the chemical in the organism with respect to the concentration of freely dissolved (bioavailable) chemical in the water. The BSAF is the lipid-normalized concentration in the organism with respect to organic carbon-normalized concentration in the sediments. BSAFs were used to determine BAF_{ℓ}^{fd}s for chemicals with concentrations which have not been measured in Great Lakes water but are detectable in sediments and fish. Equilibrium partitioning of persistent non-polar organic chemicals generally occurs between sediments and benthic invertebrates and a thermodynamic equilibrium or fugacity approach is useful for describing the degree of equilibrium (or disequilibrium) associated with bioaccumulation in fish. However, measured BAF_{ℓ}^{fd}s and BSAFs generally indicate a state of non-equilibrium partitioning from sediments to fish. BAF_{ℓ}^{fd}s and BSAFs may be determined and applied as steady-state relationships which incorporate degrees of disequilibrium, such as normally present between water and sediment or between fish and water due to the biomagnification phenomenon.

The bioaccumulation cube (Figure 1) is a conceptual model which includes the most important generic factors that must be considered when predicting bioaccumulation from measured or predicted concentrations of chemicals in the water and sediments of the ecosystem. The bioavailability considerations that remain, after incorporating the influence of organism lipid, organic carbon in water and sediments, and trophic level into BAF_{ℓ}^{fd}s and BSAFs to reduce uncertainty for site-specific bioavailability conditions, are shown on the z-axis. Basically, this residual bioavailability factor is the chemical distribution between water and sediment which can vary between ecosystems or vary temporally and spatially within an ecosystem. Chemical properties which influence bioaccumulation are shown on the x-axis. The octanol-water partition coefficient (K_{ow}) is the primary indicator of chemical hydrophobicity and bioaccumulation potential. A second chemical factor is metabolism by organisms in the food chain. Metabolism is strongly related to chemical structure as well as the presence or absence of specific metabolizing enzymes in different organisms in the food chain. The rates of metabolism of a bioaccumulative chemical by the different organisms in a food chain will determine the extent to which rates of elimination of the chemical will be faster than that predicted on the basis of K_{ow} in the absence of metabolism. Conceptually, the cumulative effect of metabolism in the food chain can be equated to a factor $K_{metabolism}$ which could be subtracted from K_{ow} to correct for the degree of reduced bioaccumulation associated with metabolism.

Ecosystem conditions, such as riverine/lacustrine character, temperature, and trophic condition, as illustrated on the y-axis of the bioaccumulation cube, may be an important consideration when trying to extrapolate from one ecosystem to another. The degree to which quantitative differences in bioaccumulation may be attributable to particular ecosystem conditions, apart from the influence of organic carbon which is handled as a bioavailability variable, is not well known. Bioaccumulation data of the quality needed for quantitative measurement of these relationships for different ecosystems are very limited. Finally, recognition that the food chain has to be defined when modeling bioaccumulation and biomagnification creates a fourth dimension, the trophic

level, within the bioaccumulation cube; i.e. the bioaccumulation cube exists for each trophic level or more specific definition of food chain.

The procedures used in the GLWQI for predicting bioaccumulatation are reported in the form of a technical support document (U.S. EPA, 1995) which is available from NTIS. Although bioaccumulation factors can be specific for ecosystem conditions and for the structure of the food chain, they may be less site-specific in regard to bioavailability conditions. However, in the GLWQI, BAF_ℓ^{fd}s were developed based upon the concentration of freely dissolved chemical in water because this greatly reduces bioavailability conditions as a source of variability between sites. The fraction of the chemical in water which is not partitioned to particulate organic carbon and dissolved organic carbon (fraction freely dissolved) is considered to be the fraction which is bioavailable. The fraction freely dissolved (f^{fd}) chemical can be estimated as:

$$f^{fd} = \frac{1}{1 + POC \cdot K_{poc} + DOC \cdot K_{doc}} = \frac{1}{1 + POC \cdot K_{ow} + DOC \cdot K_{ow}/10} \quad (1)$$

where POC is the concentration of particulate organic carbon, DOC is the concentration of dissolved organic carbon, and K_{poc} and K_{doc} are the respective organic carbon-water partition coefficients. For the GLWQI, K_{poc} was set equal to K_{ow} for each chemical, and K_{doc}, based on Great Lakes water data, was set equal to one-tenth of K_{ow}, reflecting the lesser partitioning power of dissolved organic carbon.

The distribution of chemicals between the sediment and water column can be characterized with a sediment-water concentration quotient (Π_{socw}) which is the ratio of the organic carbon-normalized concentration in surface sediments to the freely dissolved chemical concentration in water. With Lake Ontario data (Oliver and Niimi, 1988) and the POC and DOC partitioning model (equation 1), Π_{socw} can be related to the degree of chemical equilibrium or disequilibrium (Figure 2). The values of $\log \Pi_{socw}$ for chemicals with varying $\log K_{ow}$ are above the line which represents a fugacity ratio of one, or Π_{socw} equal to $\log K_{ow}$. This reflects a degree of chemical disequilibrium in Lake Ontario which is probably common to most of the Great Lakes since the early 1970s. Based on linear regression, the Π_{socw} value for these data would be about 25 times K_{ow}, or about 25-fold theoretical disequilibrium between the sediment and the water column under recent conditions in Lake Ontario. A significant portion of this disequilibrium may be an ecosystem characteristic as a consequence of the difference between the fraction of organic carbon in sediments and the fraction of organic carbon associated with particulate material in the overlying water.

BAF_ℓ^{fd} predictions for chemicals with $\log K_{ow}$s greater than 5 are quite sensitive to variations in Π_{socw} when $\Pi_{socw}/K_{ow} = 25$ (Figure 3). Using the Gobas bioaccumulation model (Gobas, 1993), Burkhard (1997) demonstrated that a 10 percent increase in the Lake Ontario Π_{socw}, which is the same as increasing the sediment concentration by 10 percent while holding the water column concentration steady, results in nearly a 10 percent change in BAF_ℓ^{fd}s. Predictions of bioaccumulation in the form of the BAF_ℓ^{fd}s for organisms throughout the food chain are strongly influenced by change in the Π_{socw} value when water is at disequilibrium with surface sediments (Burkhard, 1997). In other words, BAF_ℓ^{fd}s for chemicals with $\log K_{ow}$s > 5 are strongly benthically linked under this condition of disequilibrium, even when the benthic food chain connection may be small. If the ratio of Π_{socw} to K_{ow} is close to one (equilibrium between the water and sediment which theoretically could occur in other locations), the sensitivity of the Π_{socw} is quite different (Figure 4). The relationship for sculpin is more sensitive, reflecting its stronger benthic connection to the sediments. The BSAF sensitivity to change in Π_{socw} is the opposite of the BAF_ℓ^{fd} sensitivity; i.e., when the BAF_ℓ^{fd} sensitivity is large, the BSAF sensitivity is small. This is not surprising since Π_{socw} equals the ratio of the BAF_ℓ^{fd} to the BSAF. In summary (Figure 5), the sensitivity of BAF_ℓ^{fd}s and BSAFs to Π_{socw} depends on the K_{ow} of the chemical, the benthic/pelagic relative contribution to the food chain, and the degree of disequilibrium between the sediment and the overlying water. The relative sensitivity of BAF_ℓ^{fd}s and BSAFs to Π_{socw} could be an important determinant, under particular site-specific conditions and chemical K_{ow}s, of whether a water-based or a sediment-based bioaccumulation factor should be used for prediction of bioaccumulation.

Selection of BAF_ℓ^{fd}s for the Great Lakes Water Quality Initiative involved a tiered approach. Preference was given to high-quality, field-measured values. Unfortunately, for many chemicals these values do not exist. Second preference was given to BAF_ℓ^{fd}s predicted using a BSAF methodology, which is described below. The third and fourth tier procedures involved calculation of food chain multipliers to account for biomagnification. In the third tier, the food chain multiplier is multiplied by a measured bioconcentration factor (BCF) and, in the fourth tier when BCFs are not available, the octanol-water partition coefficient is used as a surrogate for the lipid-normalized bioconcentration factor based on freely dissolved chemical in water (BCF_ℓ^{fd}).

The second tier method, which uses BSAFs to calculate BAF_ℓ^{fd}s, was derived using the following relationship between BAF_ℓ^{fd}, BSAF, and Π_{socw}:

$$\Pi_{socw} = \frac{BAF_\ell^{fd}}{BSAF} \quad (2)$$

For many chemicals in the Great Lakes, under present-day conditions, ratios of Π_{socw} to K_{ow} (fugacity ratios between sediment and water) are similar:

$$\frac{(\Pi_{socw})_i}{(K_{ow})_i} \approx \frac{(\Pi_{socw})_r}{(K_{ow})_r} \quad (3)$$

With this condition, equation 2 can be substituted into equation 3 and, following rearrangement, the resulting equation 4 can be used to calculate $(BAF_\ell^{fd}s)_i$ for chemicals (I) such as TCDD which cannot be routinely measured in water at this time:

$$(BAF_\ell^{fd})_i = \frac{(BAF_\ell^{fd})_r\,(BSAF)_i\,(K_{ow})_i}{(BSAF)_r\,(K_{ow})_r} \qquad (4)$$

The BSAF method uses reference chemicals (r), such as PCB congeners, for which $(BAF_\ell^{fd})_r$s can often be accurately measured. If a reference chemical that has the same K_{ow} as the unknown chemical is chosen, the K_{ow} ratio in the equation becomes one and the calculation is simplified further. Figure 6 illustrates BAF_ℓ^{fd}s calculated from Lake Ontario data with the BSAF method. The distribution of the PCB congener BAF_ℓ^{fd}s (circles) provides the expected increase in bioaccumulation and biomagnification with respect to magnitude of K_{ow}. In contrast, the dioxins and furans are predicted to have quite smaller bioaccumulation factors (on the basis of K_{ow}). This is primarily because of the effect of metabolism in fish which is effectively measured by the BSAFs and incorporated into the BAF_ℓ^{fd}s. Even though metabolism rates are slow, the increase in elimination rate of the parent chemical is significant and dramatically reduces the bioaccumulation potential with respect to PCBs with the same K_{ow}.

To evaluate the BSAF method, BAF_ℓ^{fd}s measured in Lake Ontario were compared to predicted BAF_ℓ^{fd}s calculated from Lake Ontario BSAFs (Figure 7) and were in good agreement. A further evaluation was performed by comparing BAF_ℓ^{fd}s calculated from BSAFs for PCBs in Lake Ontario for trout to BAF_ℓ^{fd}s independently calculated from BSAFs in Green Bay for brown trout (Figure 8). These values were also in agreement despite the disparate ecosystems and measurements involved.

The determination of food chain multipliers (ratio of BAF_ℓ^{fd} to K_{ow}) for prediction of BAF_ℓ^{fd}s from K_{ow}s was accomplished with a food chain model (Gobas, 1993) using Lake Ontario data and conditions. The objective was to calculate food chain multipliers for bioaccumulative organic chemicals and use them with K_{ow} to estimate BAF_ℓ^{fd}s. Figure 9 illustrates how the predictions with food chain multipliers compared to measured BAF_ℓ^{fd}s with the data reported by Oliver and Niimi (1988) for Lake Ontario. The agreement is very good, particularly for the higher K_{ow} chemicals which are not metabolized, which in this case are primarily PCBs. The food chain multipliers that were calculated using the Gobas model reflect a biomagnification potential that is a function of K_{ow} for the two top trophic levels of predator fish and forage fish. Under the conditions of the model, the zooplankton are considered to be at equilibrium with respect to the water and benthic invertebrates at equilibrium with respect to the sediment.

Bioaccumulation factors can facilitate bioaccumulation predictions in ecological risk assessments involving impacts of complex mixtures of chemicals. The joint toxicity of chemicals that share similar structure and common mode of toxic action with TCDD (2,3,7,8-tetracholorodibenzo-p-dioxin) can be predicted with an additivity model using the TCDD toxicity equivalence approach. TCDD toxicity equivalence factors (TEFs) are essentially toxicity potency estimates relative to TCDD such that a chemical with a TEF of 1 has a potency equal to that of TCDD. The toxicity equivalence concentration (TEC) is calculated as the sum of the products of the TEF times the concentration for each chemical in a mixture. Often a key question involves what media the concentrations should be based upon. TECs have been calculated based on chemical concentrations in effluents, sediments, or water. TECs based on concentrations of the chemicals in tissue are preferable because of the direct connection between concentration in the tissue and the dose-response relationship for a toxic effect. If a TEC associated with a contaminant mixture in sediments is desired, the TEC can be expressed in terms of the concentrations expected in a fish as a result of bioaccumulation (TEC_{fish}). The TEC_{fish} can be directly related to the potential for toxic effects in the fish (Cook et al., 1997a), or to wildlife or humans that eat the fish, and is calculated as the sum of products of the organic carbon-normalized concentration in sediment (C_{sac}) times the BSAF (equation 5) and the TEF for each of n chemicals. The fraction lipid (f) in the fish is included when the TEC_{fish} is calculated on the basis of whole fish, rather than lipid-normalized concentration. A TEC_{fish} can similarly be calculated for chemical concentrations in water with BAF_ℓ^{fd}s.

$$TEC_{fish} = \sum_{i=1}^{n} (C_{soc})_i\,(BSAF_{fish})_i\,(f_l)\,(TEF_{fish})_i \qquad (5)$$

Equation 5, as adapted for lake trout eggs, was used retrospectively to determine the influence of TCDD and related chemicals on lake trout reproduction and population dynamics of lake trout in Lake Ontario since 1920 (Cook et al., 1994, 1997). Radionuclide-dated sediment sections from sediment cores were essential for construction of a historical exposure record which was then linked to effects on lake trout through the lake trout egg TCDD dose-early life stage mortality response relationship (Walker et al., 1994). The use of the lake trout egg TCDD dose-response relationship requires use of TEFs and BSAFs based on concentrations of chemicals in trout eggs to calculate TCDD toxicity equivalence concentrations which were based on lake trout eggs (TEC_{egg}s). Figure 10 summarizes the information that was generated by this analysis. The y-axis represents the chronology in years for 1 cm increments from the key reference sediment core from eastern Lake Ontario which were analyzed by high resolution gas chromatography/high resolution mass spectrometry. This core was selected to represent trends in Lake Ontario exposure conditions in this century. The toxicity equivalents of the more important congeners present in the 1 cm increments of sediment are plotted cumulatively on the x-axis such that each horizontal bar represents the TEC_{egg} predicted for lake trout on the basis of equation 5. The $BSAF_{egg}$s were adjusted to accommodate the effect of differences in $\prod_{socw}$ attributable to greater chemical loadings to the lake prior to 1970.

The relationship between predicted TEC_{egg}s and predicted degree of lake trout early life stage mortality is also indicated in Figure 10. For example, during the period from the 1940s to the 1970s, 100 percent early life stage mortality is predicted for this very sensitive fish species. The epidemiological records from Lake Ontario are consistent with this prediction, despite complications associated with the presence of non-chemical stressors such as sea lamprey predation and over-fishing during the period in which the lake trout population declined to virtual extinction by 1960. During the period of recovery of Lake Ontario after 1970, when lake trout were stocked and large populations of adult trout were restored, natural reproduction has not been achieved. However, eggs were collected from the stocked fish, fertilized, and early life stage development and survival monitored under laboratory conditions. The incidence of mortality due to blue-sac syndrome associated with TCDD was observed in the laboratory and is in good agreement with the percent mortality predicted with the TEC_{egg} model. After 1985, measured concentrations of TCDD and related chemicals in lake trout eggs were below the threshold for mortality associated with the blue-sac syndrome, and field observations of lake trout sac fry presence on spawning reefs increased with the first observations of 1- to 2- year old lake trout from natural reproduction occurring in 1994.

BAF_l^{fd}s and BSAFs can be critical components of water and sediment criteria development. For example, to create a sediment criterion for the protection of future lake trout populations in the Great Lakes, the following factors should be considered: (1) the concentration of TCDD in the eggs that would be associated with no adverse effects to early life stages of lake trout (this includes effects at sublethal exposures that reduce survival of sac fry and alvins) and (2) the $BSAF_{egg}$s for lake trout under a Π_{socw} condition expected during the time period of interest. The concentration of TCDD in sediment that would be associated with a lack of adverse effects (sediment criterion) can be calculated from the concentration of TCDD in the egg associated with the threshold for toxic effects divided by the $BSAF_{egg}$. Similarly, a water criterion can be calculated using the BAF_l^{fd} for TCDD.

To relate these TCDD criteria, whether they be sediment criteria or water criteria, to organisms exposed to complex mixtures of TCDD and related chemicals, differences in bioaccumulation must be considered along with the differences in toxic potency as expressed with TEFs. Bioaccumulation potentials of the different chemicals contributing to dioxin toxicity risks can be referenced to TCDD as with TEFs. In the GLWQI these were named bioaccumulation equivalency factors. The sum of TCDD toxicity equivalents is directly comparable to the water quality criterion or sediment quality criterion for TCDD under the TCDD toxicity equivalence model.

In conclusion, BAF_l^{fd}s and BSAFs are essential for application of chemical-residue-based criteria in risk assessments and are interrelated through the sediment-water concentration quotient (Π_{socw}), particularly for the more hydrophobic organic chemicals. They can incorporate elements of site-specific bioavailability. They are a specific interface between exposure assessments and effects assessments. They are a primary output in food chain bioaccumulation models used for validation of the models and comparisons to field data. Finally, measured bioaccumulation factors are effective for prediction of bioaccumulation in association with water or sediment quality criteria when used appropriately with time-averaged exposure estimates for persistent organic chemicals, especially those with a higher K_{ow}.

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Figure 1.

Framework for Predicting Bioaccumulation Factors Applicable to Different Aquatic Ecosystems

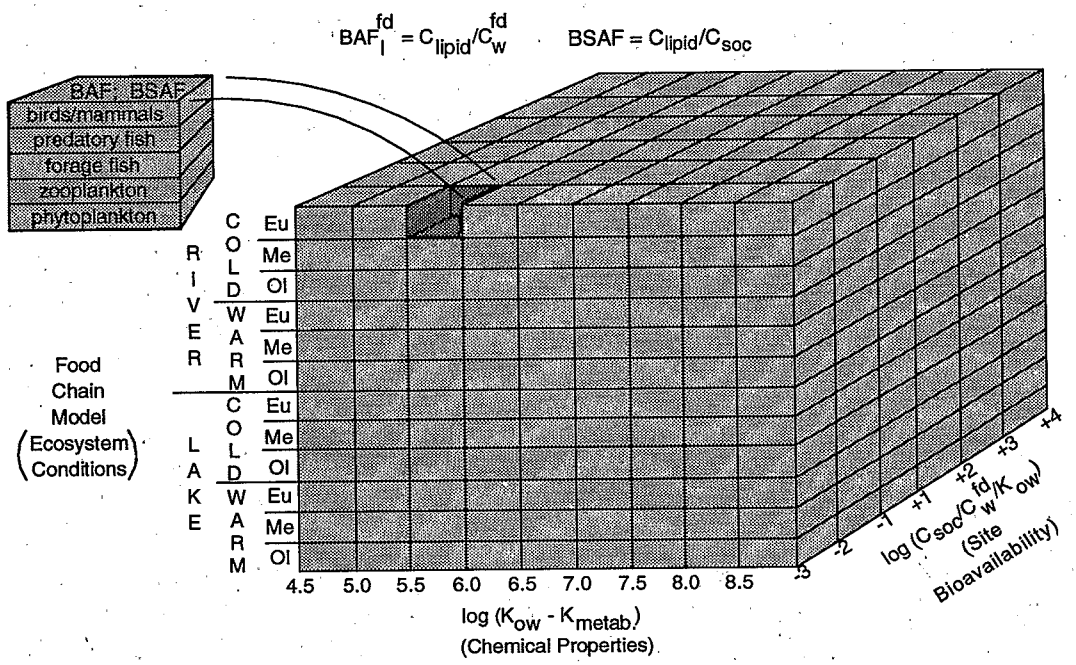


Figure 2.

Sediment-water column chemical concentration quotient for Lake Ontario data of Oliver and Niimi (1988)

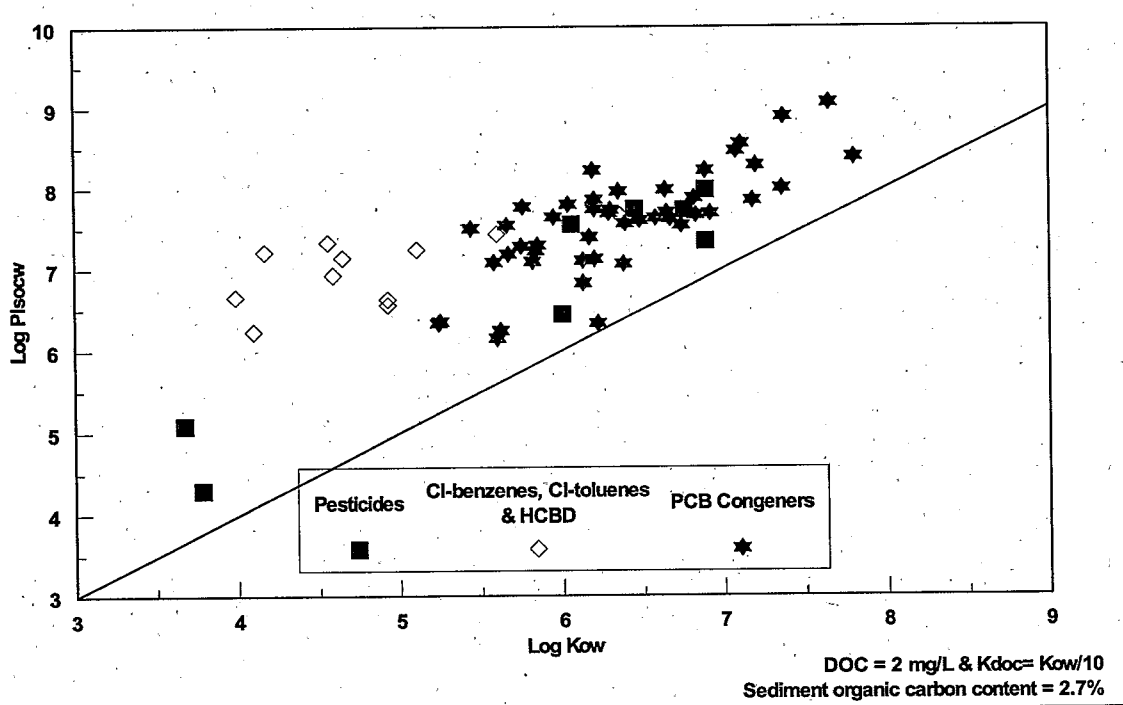


Figure 3.

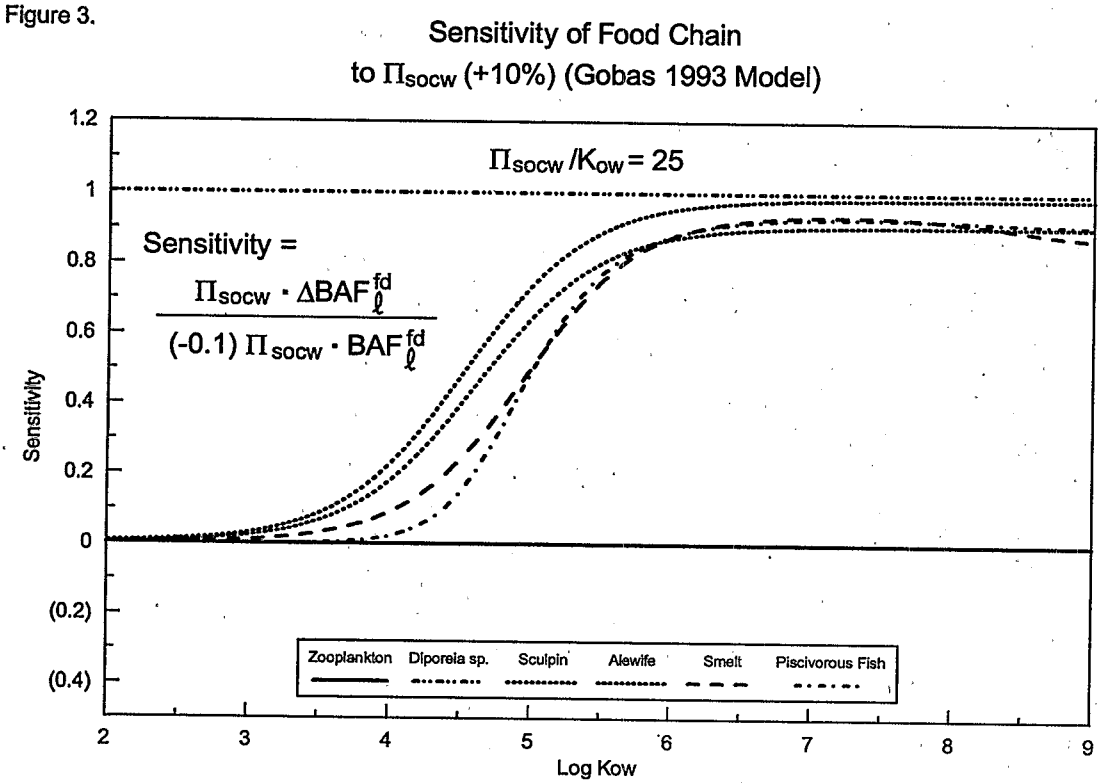


Figure 4.

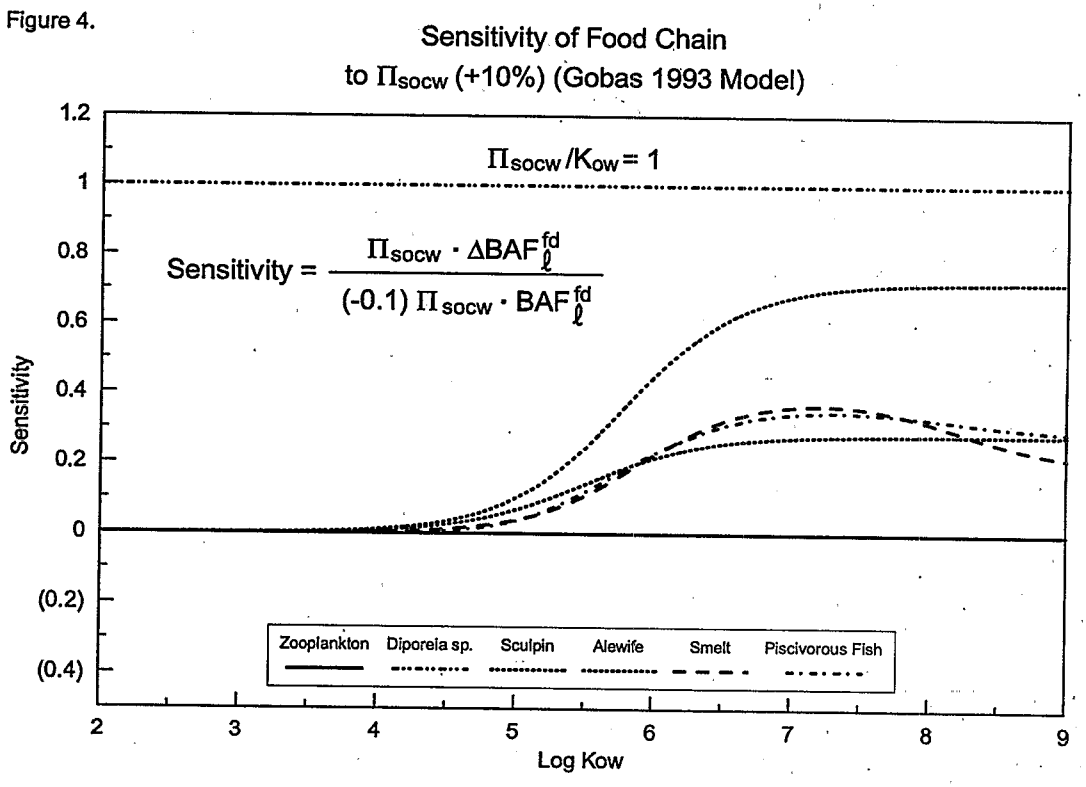
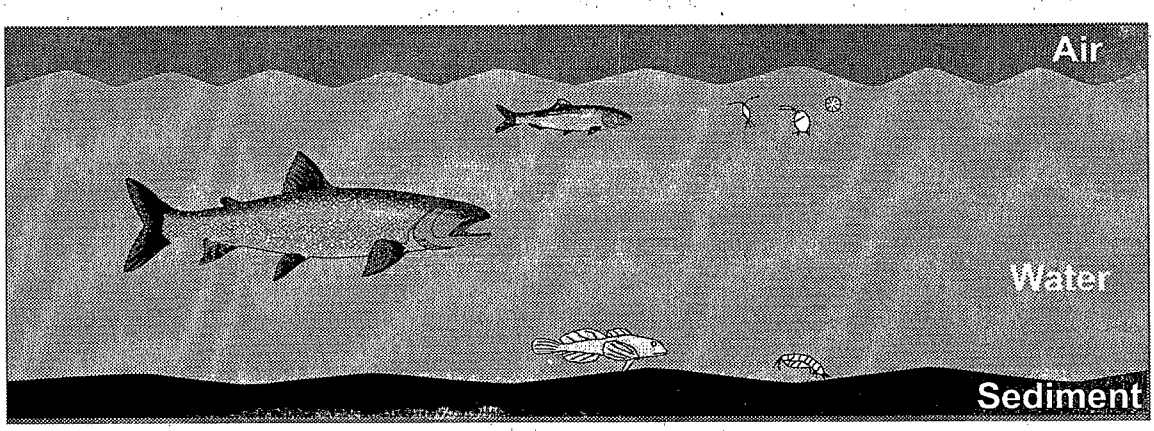


Figure 5.



$$\Pi_{socw} = \frac{C_{soc}}{C_w^{fd}} = \frac{BAF_l^{fd}}{BSAF}$$

Sensitivities of BAF_l^{fd} s and BSAFs to changes in Π_{socw} depend on:

1. K_{ow} of chemical
2. Benthic/pelagic contribution to food chain
3. Disequilibrium between sediment and overlying water (SOC - POC - C_w^{fd})

Figure 6.

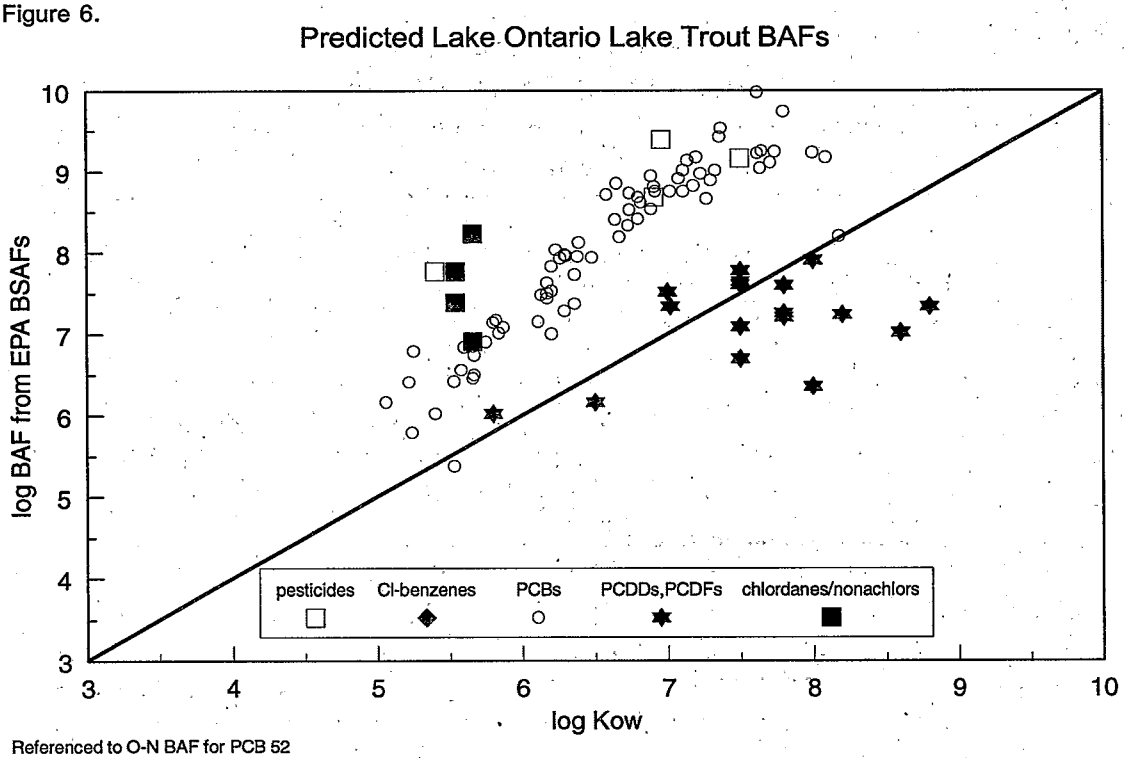


Figure 7.

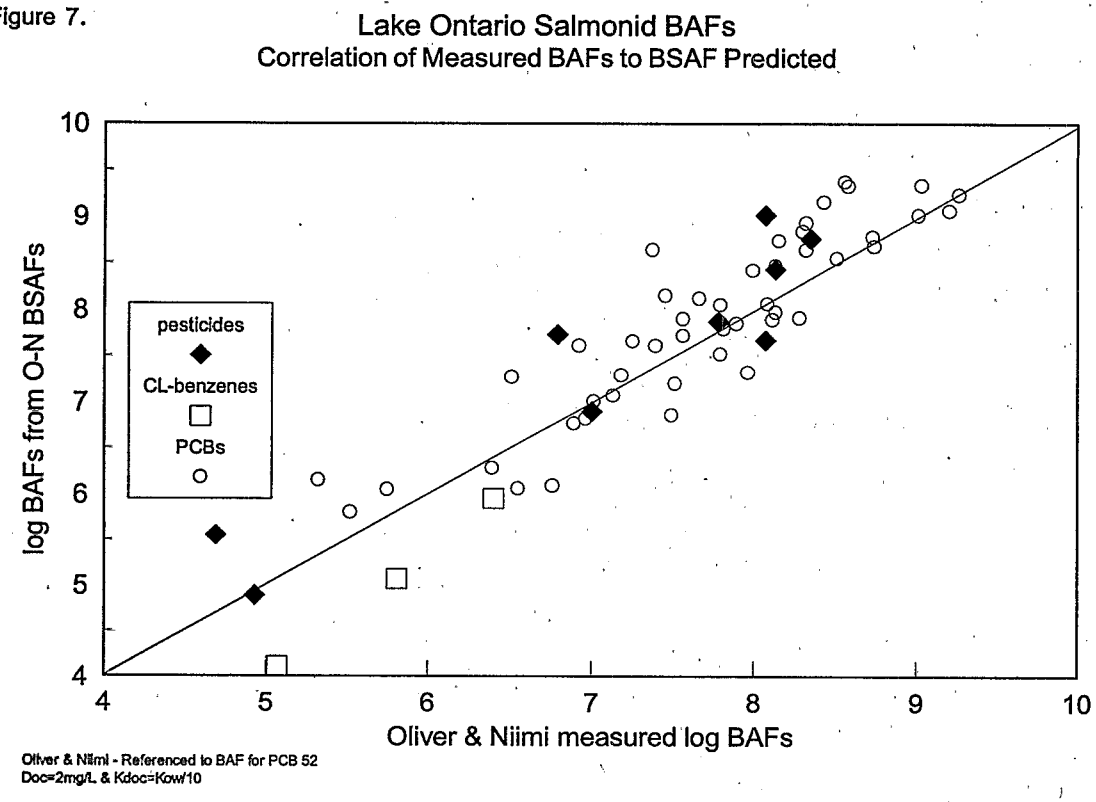


Figure 8.

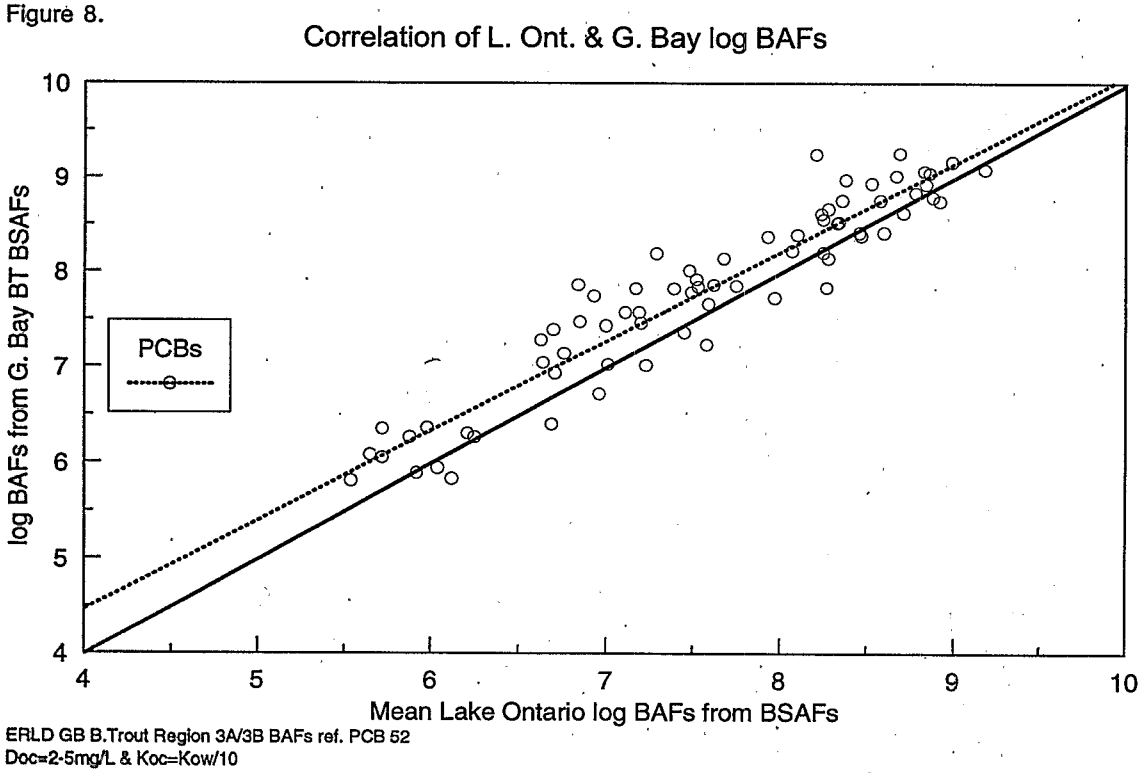


Figure 9.

Measured and Predicted BAFs for Piscivorous Fishes
Measured BAFs from Oliver and Niimi (1988)

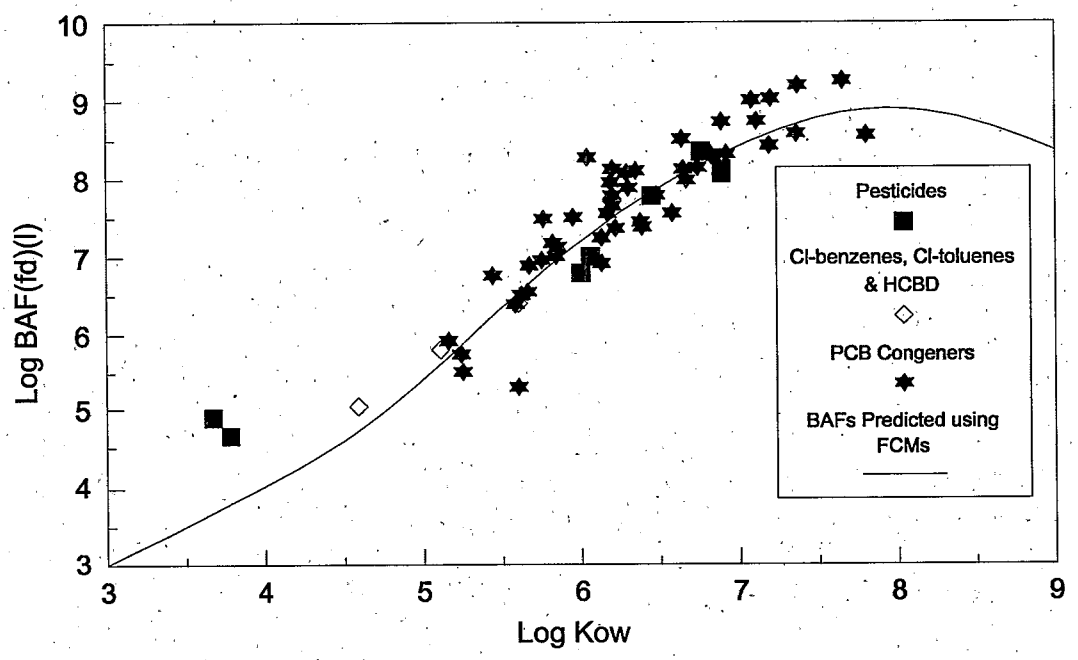
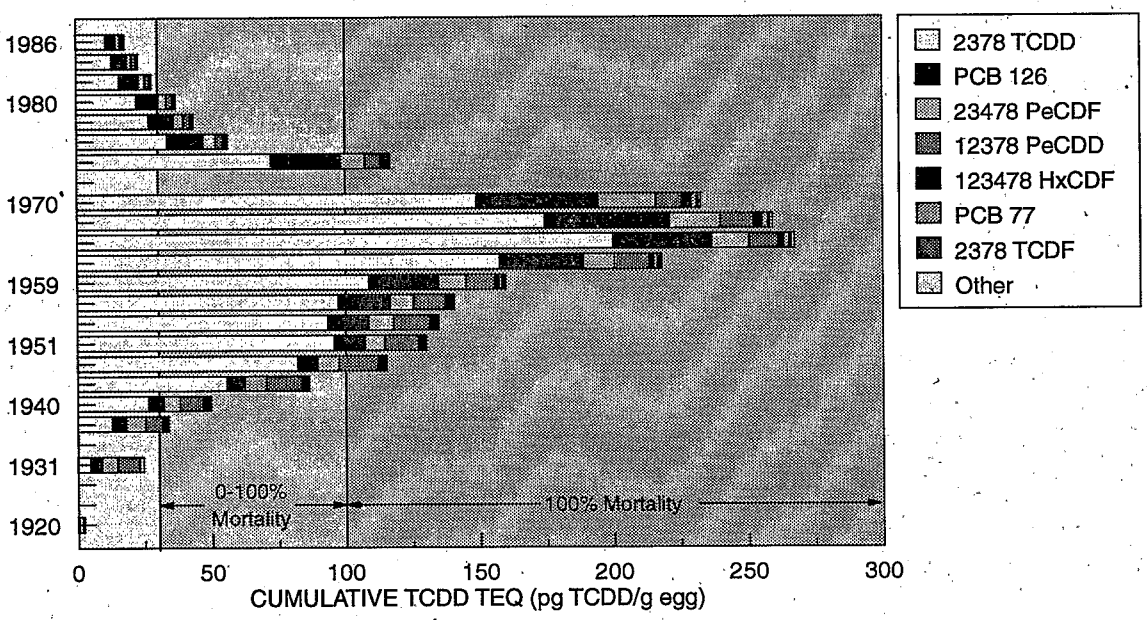


Figure 10.

Lake Ontario Lake Trout Egg RTEC Retrospective
Contribution of Individual Chemicals



From Modeling to Criteria: Integrated Approach to Criteria Development

Mary C. Reiley
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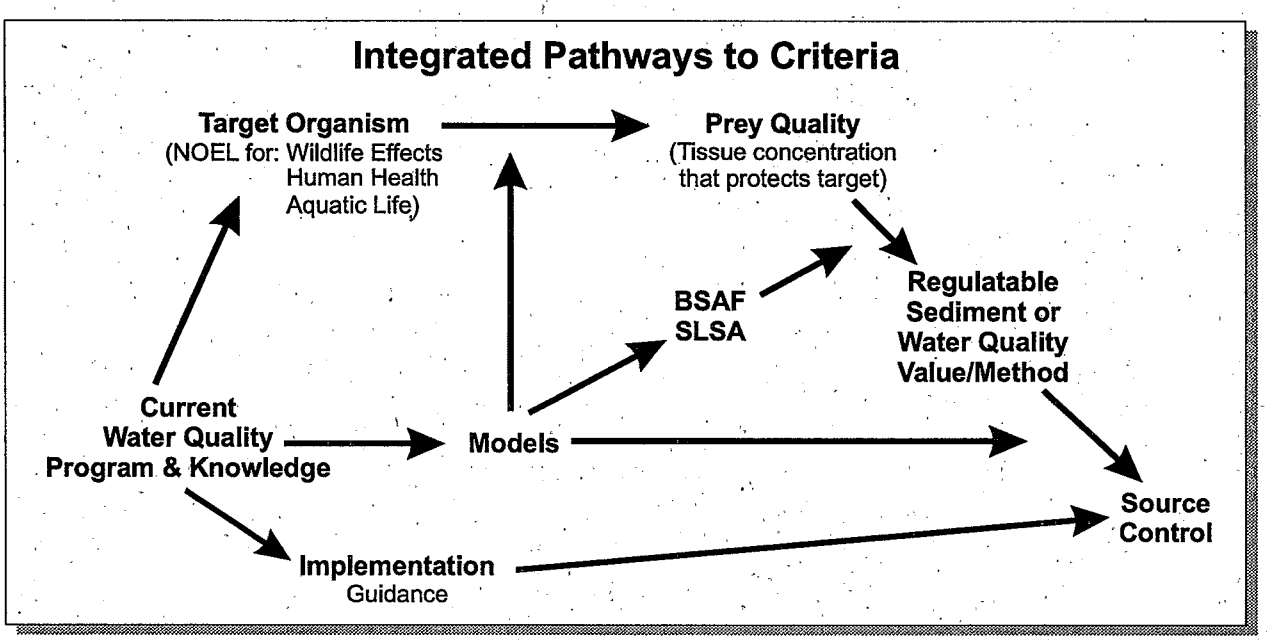
The Office of Water's Water Quality Criteria Program (CWA Section 304(a)) is at a major crossroads in its evolution. The national program currently develops chemical criteria from the priority pollutant list (CWA Section 307(a)) based on toxicity testing with little or no consideration of mixtures, bioaccumulative potential, or exposure pathway. Individual criteria (aquatic life, human health, sediment, and wildlife) were developed for a host of chemicals, often with the result of several criteria types for a given chemical. The most stringent of these criteria was then used to derive the water quality goals of an aquatic system or a permit limit. To develop assessment methods and criteria that will align more closely with the stated objective of the Clean Water Act, "... to restore and maintain the chemical, physical, and biological integrity of the Nation's waters," the program would like to integrate criteria development using the risk assessment paradigm. The goal is to develop criteria and methods more efficiently and with more insight into the effects of chemicals on aquatic systems (including those terrestrial organisms which are aquatic dependent, e.g., fish eaters).

The Office of Science and Technology (OST) proposes to develop water quality criteria by evaluating the different possible pathways of exposure and determining the pathway that is most critical for a particular chemical. To accomplish this task, OST will develop a strategy to merge the technical and programmatic aspects of water quality criteria programs (human health, drinking water, aquatic life, wildlife, sediments, biocriteria) to address ecosystem threats for a specific chemical (or class of chemicals). The product developed under this strategy will be a multi-pathway model(s) for establishing human health and ecological criteria, with particular attention paid to the most sensitive pathway. The program is working with the Society for Environmental Toxicology and Chemistry (SETAC) to design a Pellston Workshop that will address the state of the science and research needed to be able to construct integrated assessment tools. The figure below is a conceptual diagram of how this may be achieved.



From Modeling to Criteria:

Integrated Approach to Criteria Development for Bioaccumulative Chemicals

5 Types of Water Quality Criteria

Aquatic Life
Human Health
Sediment Quality
Wildlife
Biocriteria

Integrating Criteria

Chemicals
Designated Use/Target Organism
Fish Tissue
Measurement Media
Regulatory Implementation

Research and Implementation Issues

Incorporate metabolism into the model(s).
Change residue values into acceptable loadings.
Account for multiple sources.
What level of protection do we strive for?
Package model(s) and guidance for practicality.

Day One: September 11, 1996

Session Three: Questions and Answers

After each session, there was an opportunity for questions and answers and group discussions pertaining to the speakers' presentations.

Q (Bob Barrick, PTI Environmental Services): Dom, I was going to just sit here and listen, but you brought up Fred Praul's work and it is such a central part of your talk that I really need to speak. As you know I work with Fred. His conclusions from his research are 100 percent opposite of your conclusions.

Dominic Di Toro:

Exactly right. You can conclude what you like.

Q (Bob Barrick): I want to make three points here. One is that when you are going ahead and using bulk organic carbon to normalize sediments, it misses the fact that organic carbon is disproportionately found in the different fractions. It is not a weighted average. Unless you quantify the organic carbon in each of those size and density fractions, you are going to come up with the wrong answer. Bulk organic carbon is not necessarily appropriate, even if organic carbon works. That is also the case with what Fred found, which was a disproportionate amount of PAH versus organic carbon in those fractions. Now, that is obscured a bit because Fred did present arithmetic plots in his thesis and you have converted those to log plots, which has dampened down the variability. If you take that out, it is a pretty big scatter in there.

Dominic Di Toro:

Bob, we have, as you know, organic carbon measurements. He made measurements in each of the size classes, so we know what the organic carbon concentration in each size class is. So, you divide by the organic carbon concentration in each size class.

Q (Bob Barrick): Dom, my point was that Fred did it properly. What I am saying is that when you are taking the organic carbon fractions and applying them out in the real world, where everybody has a TOC number, that number is not according to a particular compartment within the sediment. It is a bulk sediment number, and to use that to measure against some kind of chemical that may be found only in one compartment is not what Fred found. If Fred found that, that may be an erroneous way to proceed.

Dominic Di Toro:

Let me just comment on the comment. If you actually take the bulk organic carbon concentration and take the chemistry per unit carbon, it also works for all the size fractions. That is because it is the same in each fraction. So, if you add up the same numbers, you get the same number.

Q (Bob Barrick): It works better if you put it on a log plot. The third thing I wanted to say was that in discussing all these things, it also ignores the difference in something else that Fred found in his research. This was the difference between labile and refractive particles, even though you may be able to extract organic compounds out of both of those using organic solvents in a laboratory. In fact, in nature where exposure is actually happening, there may be no exchange from the refracted particles and considerable exchange from labile particles. And to use the data to say that it supports equilibrium in the environment, when it is looking at an organic carbon-extracted system, is also making a little simplification. The only reason I really wanted to bring this up is because it is something that is now a part of your talk, and it is really counter to the conclusions of his dissertation work.

Dominic Di Toro:

I think the data analysis stands on its own. And you are right. Fred concluded exactly the opposite, which struck us as a little peculiar since the data seemed to indicate what I presented to you. There is a problem, by the way, with bioavailability. All organic carbon is not the same and all phases of sediments and chemicals are not the same. The issue is whether the difference is so large as to make organic carbon normalization useless, or are we arguing about the extent to which we can collapse responses

and understand what we are seeing? If the variability was so large as to be useless, then I would say fine, use dry weight normalization. In fact, why not use wet weight normalization in sediments? People have used it for biota. Why is it not a good idea in sediments? The point is you have to have some kind of database and theoretical understanding of what you are doing. The data speak for themselves.

Q (Dennis Leonard, Detroit Edison): Dr. Cook, could you comment briefly on the development of the BAF for mercury in the Great Lakes? The BAF model that you discussed required the steady-state assumptions to be present. The mercury concentrations that we have in the Great Lakes vary by about a factor of 100. Tributary littoral concentrations are influenced by high concentrations of mercury in rainfall. Open water concentrations are affected by transfer of mercury back to the atmosphere; therefore, we have concentrations ranging from about 15 ng/m³ to 0.5 ng/m³ at any point in time and space. Is the steady-state assumption really valid, and how do you develop a mercury criterion when you have this hundred-fold variation?

Philip Cook:

I would love to answer your question, but I cannot. I do not know that much about the mercury bioaccumulation. If there is someone else here who feels they have that knowledge, I would welcome their participation.

Bob Barrick:

We have done quite a bit of work on mercury and mercury uptake. Mercury is a totally different animal because how it is associated in cells is quite different from other metals. It is pretty complicated. There is literature available, and I think there is a whole conference on mercury that focuses on its uptake in the environment. Mercury does not fit neatly into these models. Consult the literature because there is information on how that should be done differently.

Philip Cook:

I can agree with that.

Q (Gayle Garman, NOAA): I am pleased to hear someone talk about the fact that the environment is not really in equilibrium. Yet you are still making an assumption of steady state and this is for the Great Lakes. Generally, I work in estuaries and they are very dynamic systems. I am wondering if you could comment on our assumptions of steady state or equilibrium for estuaries, and whether you think the modeling approach that you have taken for the Great Lakes is applicable in an estuarine situation?

Philip Cook:

I certainly would not advocate using a steady-state model where we know it would not produce a valid result. There is obviously a need for nonsteady-state models. To some extent, these empirical tools that I have discussed here today may be usable, when necessary, in nonsteady-state conditions. I think you have to get a good prediction of an average exposure relationship, whether it be sediment or water, and then take into consideration the properties of the chemical that you are trying to model. If it is a lower K_{ow} chemical, I think the fluctuating exposure problem becomes more severe.

Q (Charles Kovatch, University of South Carolina): Could you further explain your food chain multiplier model parameter?

Philip Cook:

Essentially, the food chain multiplier is an expression for the organism that you are trying to model, which represents the increase in the BAF over the octanol-water partition coefficient. So, it is an expression of the total effect (in the food chain) of biomagnification on the bioaccumulation factor. It is applied when you do not have the bioaccumulation factor specific for that chemical, so you use a bioconcentration factor or an octanol-water partition coefficient as the surrogate for the equilibrium accumulation. The food chain multiplier, when applied to either the BCF or K_{OW}, predicts what the biomagnification effect would be on the bioaccumulation factor.

Q (Doug Hotchkiss, Port of Seattle): The comment was made in this last talk and in others that we have a lot of data out there. When we make these lists of data, like the compilations of data by URS and others, it looks like we have a lot of data. I have been involved recently in trying to use some of that data on a very site-specific regulatory basis. This is just a word of caution and a note to everybody out there. These lists may be very good for screening purposes to identify what we should really worry about. But when you apply them to regulating in a site-specific situation, you need to take a really close look at each individual paper. If you look at the specifics of how that number was generated, you will find that some of those numbers can be very soft for regulatory purposes. For example, in a study using a spiked sediment, you should check if they rinsed out the interstitial spike before they ran the tests. Some of the numbers can also be listed incorrectly from a paper. These lists can be a great help, but you ought to reexamine the values that are going to be critical in the decision-making process for both the regulated and the regulatory community. You should sit down and take a critical look at what those numbers really mean before just automatically moving ahead with them.

Q (Weldon Bosworth, Dames and Moore): I would like to direct questions to Dr. McFarland and Dr. Di Toro. For those of us that work in wetlands, we occasionally see organic carbon levels that are substantially higher than the data you presented on your slides. There are also substantial amounts of dissolved organic matter. When we apply this on a screening level or for a Tier I risk

assessment, we need to determine what the potential is for bioavailability. Referring just to neutral chemicals, we might want to go to carbon normalization. How comfortable would you feel about applying your theoretical bioaccumulation potential estimate to something that would range up to 30 or 40 percent organic carbon? What empirical or theoretical data do you have to support your answer?

Victor McFarland:

At 30 or 40 percent organic carbon, you might have quite a lot of something that is not natural organic carbon. But it could also be natural material like peat. What we ordinarily encounter is what you see in ship channels. I have not tried to measure anything like you have described. What we have done is to see how low you can go, rather than how high you can go. In the types of situations that we generally see, we have material containing up to 3 percent and occasionally up to 5 percent TOC. But, in fact, I have had to search hard to find anything that I could use in doing those kinds of studies that was much above 3 percent. We have some idea of how low you can go, which kind of agrees with Dom, but I could not answer about higher TOC ranges.

Dominic Di Toro:

We know a little bit about that. The carbon fractionation data that I showed you went as high as 10 or 20 percent. The theoretical limit is 40 percent, meaning that 40 percent on a dry weight basis of organic matter, sludge, or other material is carbon. So, that is the upper limit. Quite a bit of work has been done on partitioning of hydrophobic chemicals to digested sewage sludge. We used a fair amount of that data in the mid-1980s to establish the partitioning relationships that we use. For partitioning experiments and suspensions, I know that carbon normalization appears to work up to 20 or 30 percent.

Q (Weldon Bosworth): I do not mean to suggest that you could probably get away with a Tier 1 assessment on a situation like that. The result of carbon normalizing and then adjusting bioavailability for that would show that there would not be much that is bioavailable. It becomes a critical issue when you consider the relative risks of leaving material there that may not be bioavailable compared to tearing up the whole wetland and trying to restore it. There is not much work being done on bioavailability in conditions when the organic carbon is that high.

Q (Paul Jacobson, Langhei Ecology, Inc.): I have to say I was a bit bemused by Dom's comments regarding the role of ecology in this whole process. I feel compelled to make a short comment on that. I think it is true that toxicology was developed as a science and applied to pollution control decades ago, because ecology was really not up to the task. However, I think that hardly validates the assertion that ecology process and content really is not relevant today and into the future. And I think that for the past 20 years or so, pollution control has really been defined in toxicological terms. The upshot of the Edgewater consensus was that the approach of the last 25 years is not going to get us where we need to go. There is a need for a more ecologically oriented perspective.

Dominic Di Toro:

I did not mean to say that I think ecology is irrelevant. What I meant to say was I did not see how the type of ecological work that I think is being thought about would get us there any quicker. I think that simple observations of disturbed ecosystems, with no idea of what the causality is about and without the sort of toxicological information that we have and need to develop, is a waste of time. That was the caution, not that we should not do ecology, but rather if you look in the risk assessment paradigm and open the tool box to find what methods actually exist to do this problem, you find a remarkable lack of quantitative methods.

Paul Jacobson:

Well, then, perhaps we are in complete agreement. I think that the need is for good ecology.

Dominic Di Toro:

Yes, exactly right. I am railing against what I see as a return to an approach that I would call Victorian naturalism. That approach involves observing, making measurements, and trying to make some elaborate arguments linking observed changes to causation.

Q (Peter Landrum, NOAA, Great Lakes Environmental Research Laboratory): I was interested in your talk, Dom, since we have done some of that particle size separation and carbon normalization. I would say that the variance in our laboratory-dosed sediments is somewhat larger than what you showed on your log log plots. We see at least an order of magnitude. And, in some cases, we see the condition where the big organic particles do not seem to adsorb as much material.

Dominic Di Toro:

I think what that says, Peter, is that your lab simulations are not at equilibrium. The data I showed you are field data sets.

Q (Peter Landrum): Right, but I still think where you have bigger particles, they are not necessarily going to be at equilibrium.

Dominic Di Toro:

Well, I ask you to look at three data sets that strongly suggest that large particles with high organic carbon concentrations look just like small particles, and that kind of equilibrium must have been there for that relationship

to work. That relationship tests two very interesting assertions: that the system is at equilibrium and that carbon normalization gets rid of the effect of other sedimentological properties.

Q (Peter Landrum): If you put that on a linear plot, you will see that there is still quite a lot of variance there.

Dominic Di Toro:

There is no doubt about that, but put the raw data on a linear plot.

Q (Peter Landrum): Well, that is what I do.

Dominic Di Toro:

Well, then it looks like hash. If you do not carbon normalize, then it is all over the place.

Q (Peter Landrum): Carbon normalization actually does, in most cases, reduce the variance. I will not say in all cases. The other thing, though, that we have seen is even though you look at that carbon normalization, what we see for the bioavailability in the selection of particles by the organism is that they are selecting particles that we cannot even distinguish on a carbon-normalized basis. We see this based on the measurement of concentrations in the fecal pellets, where they are actually selecting particles at a much higher concentration than we can measure relative to our bulk chemical measurements. That is another issue that needs to be considered relative to what we get out of carbon normalization relative to bioavailability.

Dominic Di Toro:

It occurs to me, Peter, that maybe the problem is that you have to incubate your lab sediments long enough to get equilibrium across the particle spectrum.

Q (Bob Barrick): I am glad that Peter Landrum spoke up because the major conclusions from Fred's thesis directly support a particle-selective model. It says the particles are important, and if there is feeding on a particle-selective basis, then you will have dramatically different results. His thesis directly supports that rather than supporting an equilibrium perspective.

Dominic Di Toro:

My only comment is Fred's data are what they are.

Q (Susan Kane Driscoll, Virginia Institute of Marine Science): Dr. McFarland, I was interested in the BSAF database that you showed where the median value for the PAHs was lower than for the chlorinated compounds. Do you have any impression about whether it is more important that there is metabolism of the PAHs, or if you are seeing some sort of kinetic limitation to accumulation, or if it is a bioavailability question like for soot particles?

Victor McFarland:

Well, fish certainly tear PAHs up, and I do not think that the bivalves are totally devoid of some metabolic capability there, too. But I do not think that is the whole answer. Anthropogenic PAHs are associated primarily with soot. The chemical is going to be distributed throughout the soot particle. In order to desorb from the interior of a soot particle, I think you would have to imagine something like a chromatography column, where you have many steps of desorption, sorption, desorption, and so on, which would make the process quite slow. I think there is some evidence for this in the literature. It is a complex process. There is certainly much more to it than just one or two things that determine bioavailability and the BSAFs that we have measured for organisms and chemicals.

Q (G. Fred Lee, G. Fred Lee & Associates): I would like to ask Mary, what, if any, timetable EPA has for putting where we are now into a regulatory framework?

Mary Reiley:

A long one. If we look at the past history, it is a long process from the time we have data and come up with the theory, to moving that into something that can be practically implemented into a regulatory program, and then to actually getting it implemented. There is an introductory period to get the research accomplished, about 5 years to make it practical, and then another 5 to 10 years to get it implemented. I think a lot of people believe that if we publish something today, it is implemented tomorrow. That is not the way it works. It takes a lot of time to get the bugs out. Five years would be nice, but I would say to actually see it being implemented in a regulatory program on a routine basis would take longer than that.

Session Four: Food Chain Models and Bioenergetics

Lawrence Burkhard, Panel Moderator
U.S. EPA, Office of Research and Development,
Duluth, Minnesota

Frank Gobas
Simon Fraser University,
Burnaby, British Columbia, Canada
Food Chain Models for Predicting Bioaccumulation

John P. Connolly
HydroQual, Inc.,
Mahwah, New Jersey
Use of Food Web Models to Evaluate Bioaccumulation Data

Robert V. Thomann
Manhattan College,
Riverdale, New York
Bioaccumulation Modeling of PCBs in the Hudson Estuary: A Review and Update

Food Chain Models for Predicting Bioaccumulation

Frank Gobas
Simon Fraser University, Burnaby, British Columbia, Canada

Please contact the speaker for information on this presentation.

Use of Food Web Models to Evaluate Bioaccumulation Data

John P. Connolly and David Glaser
HydroQual, Inc., Mahwah, New Jersey

Introduction

A typical use of bioaccumulation factors is to derive clean up levels or criteria/standards for water and sediment. The target sediment or water concentration (TSC or TWC) is established by dividing a maximum allowable tissue level (MATL) by either a biota-to-sediment accumulation factor (BSAF) or a bioaccumulation factor (BAF). Where possible, the BSAF and the BAF are defined by measurement (USEPA, 1995). The implicit assumption is that the BSAF and the BAF are estimates of an invariant steady-state relationship between tissue concentration and sediment or water concentration. By that we mean the following: the BSAF and BAF are applicable to conditions that differ from the measurement conditions such that achieving a sediment concentration equal to the TSC or a water concentration equal to the TWC will result in a tissue concentration equal to the MATL.

Applicability of Measured BSAFs and BAFs

Measured BSAFs and BAFs can be applied to the clean up level or criteria/standards problem only if they describe the invariant steady-state relationship that is implied by their use. Numerous examples exist where this is evidently not the case.

BSAFs and BAFs vary with location

In the Hudson River, the polychlorinated biphenyl (PCB) BAFs measured in 1990 for largemouth bass decline from ca. $40x10^6$ to $10x10^6$ L/kg lipid with distance from the PCB source. Conversely, the BSAFs increase from about 1 to 2 g OC/g lipid (see page 4-9). In Green Bay, the PCB BSAFs for carp increase with distance from the source by factors of 5 to 10 (see page 4-9). The ratio of ppDDE (p,p'-DDE) in kelp bass to ppDDE in white croaker from the Southern California Bight increases with distance from the source, changing by more than a factor of 7 (see page 4-10).

BSAFs and BAFs vary with time

At Stillwater in the upper Hudson River, largemouth bass BAFs vary among years from about $3x10^6$ to $30x10^6$ L/kg lipid. Similarly, the BSAFs vary from about 2 to 4 g OC/g lipid (see page 4-10).

BSAFs and BAFs vary among species at similar trophic levels

Green Bay carp and alewife are at similar trophic levels, as indicated by similar levels of PCB accumulation relative to particles at the base of the food web (see page 4-11). Yet, alewife have much lower PCB concentrations. Southern California Bight dover sole and white croaker both consume benthic invertebrates, yet the PCB and ppDDE BSAFs of the sole are 2 to 3 times lower than those of the croaker (see page 4-11).

To explore the potential causes of the variability in BSAFs and BAFs, consider that they are presumed to reflect a steady-state relationship among contaminant concentrations in biota (v), dissolved water (c_{dl}), and/or sediment (r_2). In general, both sediments and water column may contribute to fish contaminant loads, so

$$v = ac_{dl} + br_2$$

Because the use of a BSAF or a BAF implies an invariant relationship between the biota and media considered (i.e., sediment or dissolved water), a constant relationship must exist between dissolved water and sediment:

$$r_2 = K_{ws} c_{dl}$$

where K_{ws} is the water-sediment partition coefficient.

These equations yield the following expressions for BSAF and BAF:

$$BSAF = \frac{a}{K_{ws}} + b$$

$$BAF = a + bK_{ws}$$

The coefficients a and b are functions of food web structure, bioenergetics, and toxicokinetics. The partition coefficient K_{ws} is a function of contaminant loading and all

of the fate and transport processes. Lack of steady-state conditions is an obvious cause for BSAF and BAF variability. BSAF and BAF variation with location could also be attributable to variations in K_{ws} or food web structure. BSAF and BAF variation with time may result from variations in K_{ws} or in bioenergetics that result from physiological changes (e.g., lipid content). BSAF and BAF variation among species may be due to food web structure, bioenergetics, or toxicokinetic differences.

Interpretation of BSAF and BAF data must involve an assessment of whether the implicit assumption of an invariant steady-state condition is accurate. Is a measured BSAF or BAF value appropriate for regulatory application or should it be adjusted or discarded? Such interpretation is hampered because of the probable site-specificity of BSAFs and BAFs and the many factors that can invalidate the invariant steady-state assumption.

Use of Food Web Models to Validate Bioaccumulation Data

Food web models provide a means for validation because they mechanistically describe the bioaccumulation process and can ascribe causality to observed relationships between biota and sediment or water. The utility of models as validation tools for data is predicated on the accuracy of the models. Two issues are important: (1) is the bioaccumulation process sufficiently well characterized to permit the use of models as diagnostic tools? and (2) is the uncertainty of model calculations small enough to allow discrimination among measured BSAF or BAF values? A model of PCBs in Green Bay and DDE and PCBs in Southern California Bight illustrate the robustness of the models. In Green Bay, a single model structure with one set of bioenergetic and toxicokinetic parameters accurately reproduced congener and ΣPCB concentrations in five fish species in five locations that extend over an order of magnitude gradient in exposure concentrations (see page 4-14).

Monte Carlo analysis of a model of ppDDE in dover sole indicates that model uncertainty is not greater than the uncertainty of field BAF data, about a factor of 2 (see page 4-16).

Example

Green Bay carp PCB data were used as an example of spatial variability in estimated BSAF values. To interpret these data a bioaccumulation model was developed using the same toxicokinetic parameters as used for the other Green Bay species. The computed BSAFs are consistent with those observed in Zone 3B. In contrast, the model was not capable of reproducing the Fox River and Zone 4 BSAF data while maintaining parameter values within experimental limits (see page 4-17). Thus, the modeling suggests that the Zone 3B data represent a steady-state relationship between carp and sediment. Further, it appears that in the other zones the carp were not at steady state with the sediments, possibly because the sampled sediments do not properly describe the exposure concentrations seen by the carp.

Southern California Bight ppDDE and PCB data in three species of birds were used to assess from what location the birds received their contaminants. To interpret these data, a bioaccumulation model was developed using the same toxicokinetic framework for all three species. Measured prey and predator concentrations were consistent for the peregrine falcon and bald eagle, suggesting that our view of the feeding behaviors and feeding locations of these species is reasonable. However, prey and predator levels were not consistent for the double-crested cormorant on Santa Barbara Island, suggesting that the cormorants feed in less contaminated areas than previously assumed.

Conclusions

- Food web models provide a means to interpret bioaccumulation data.
- Models are necessary to test assumptions implicit in data-based BSAF and BAF values used for regulatory purposes.
- Two potential applications of models are:
  (1) to refine the database of BAF and BSAF values used for regulatory purposes and
  (2) to increase confidence in regulatory decisions having substantial economic implications.

References

USEPA. 1995. *Great Lakes Water Quality Initiative technical support document for the procedure to determine bioaccumulation factors.* EPA-820-B-95-005. U. S. Environmental Protection Agency, Washington, DC.

- A typical use of bioaccumulation factors is in the derivation of clean up levels or criteria/standards for water and sediment:

Sediment Target Level (STL in mg/kg)

$$STL = v_c / BSAF$$

Water Column Target Level (WTL in µg/l)

$$WTL = v_c / BAF$$

- Where possible, the BSAF and the BAF are defined by measurement.

- The implicit assumption is that the BASF and BAF values are estimates of an invariant steady-state relationship between tissue concentration and sediment or water concentration

  i.e., achieving a sediment concentration equal to the STL or a water column concentration equal to the WTL will result in a tissue concentration equal to v_c

The steady-state relationship among contaminant concentrations in biota (v), dissolved water (c_{d1}) and sediment (r_2) is:

$$v = ac_{d1} + br_2 \quad (1)$$

If *a* and *b* are both non-zero, an invariant relationship requires that:

$$c_{d1} = K_{ws} r_2 \quad (2)$$

which yields:

$$BSAF = aK_{ws} + b \quad (3)$$

$$BAF = a + \frac{b}{K_{ws}} \quad (4)$$

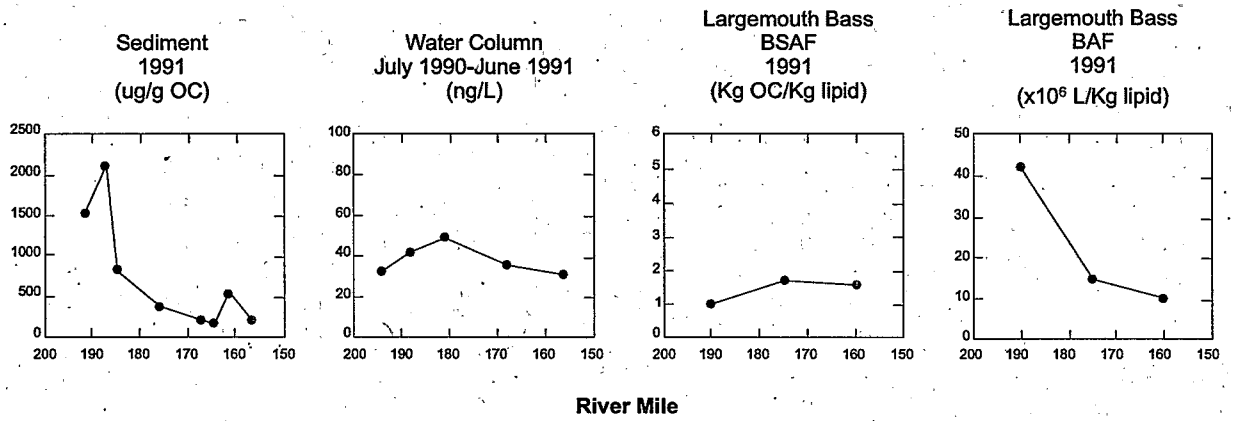
and, thus, requires that a, b and K_{ws} are invariant

The constraints are restrictive, because:

- Equation (1) is only valid at steady-state
- *a* and *b* are functions of food web structure, bioenergetics and toxicokinetics
- K_{ws} is a function of contaminant loading and all of the fate and transport processes

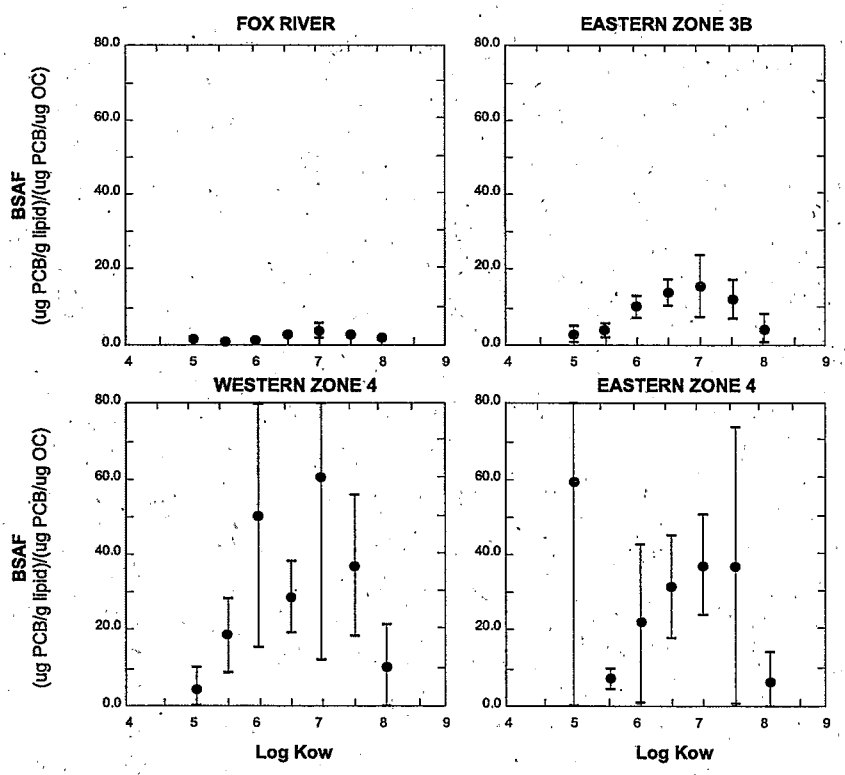
Thus, it is probable that many BSAF and BAF values calculated from measurements do not reflect an invariant, steady-state condition.

How can we interpret BSAF and BAF data?

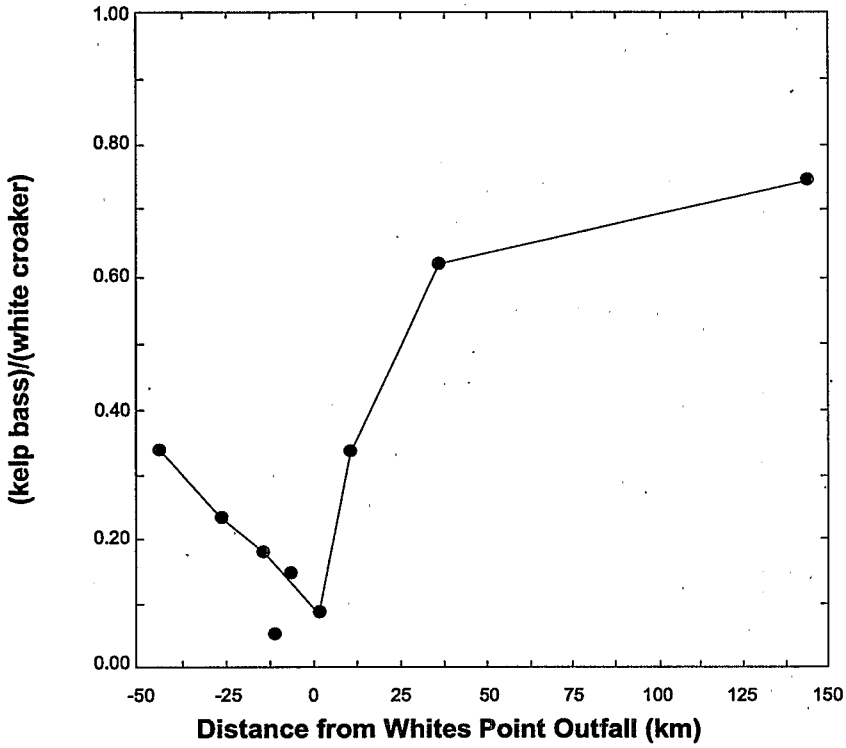


Total PCBs in the upper Hudson River: Spatial Patterns
Sediment, water column and largemouth bass in 1991

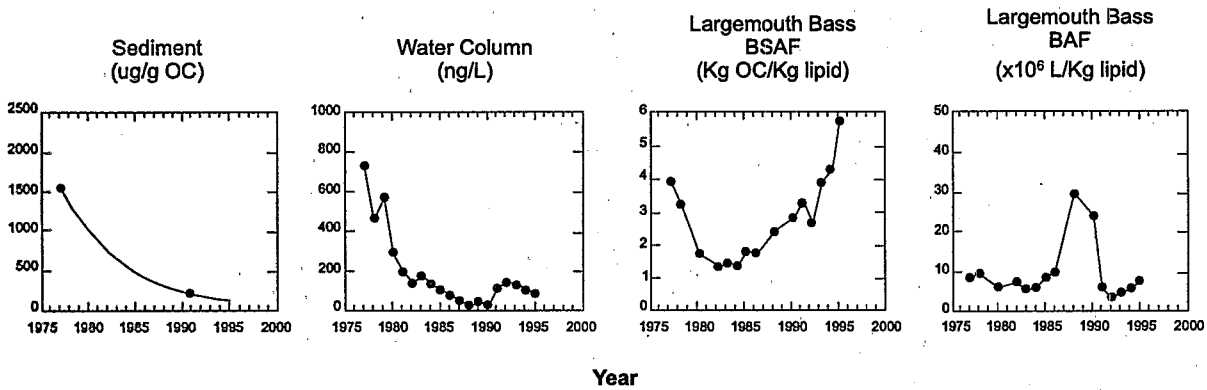
HydroQual, Inc.



Bioaccumulation Factors for PCBs in Green Bay Carp
PCB Congeners Grouped into 0.5 Log Kow Bins
Data are Arithmetic Means and 95% Confidence Intervals

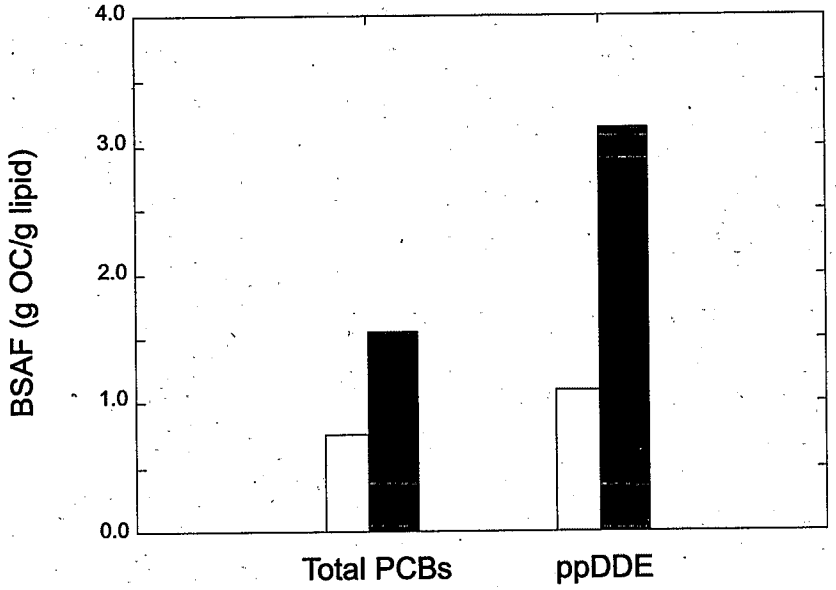


Ratio of (kelp bass)/(white croaker) lipid-based ppDDE concentrations in the Southern California Bight.



Total PCBs in the upper Hudson River: Temporal Patterns
Sediment, water column and largemouth bass from Stillwater

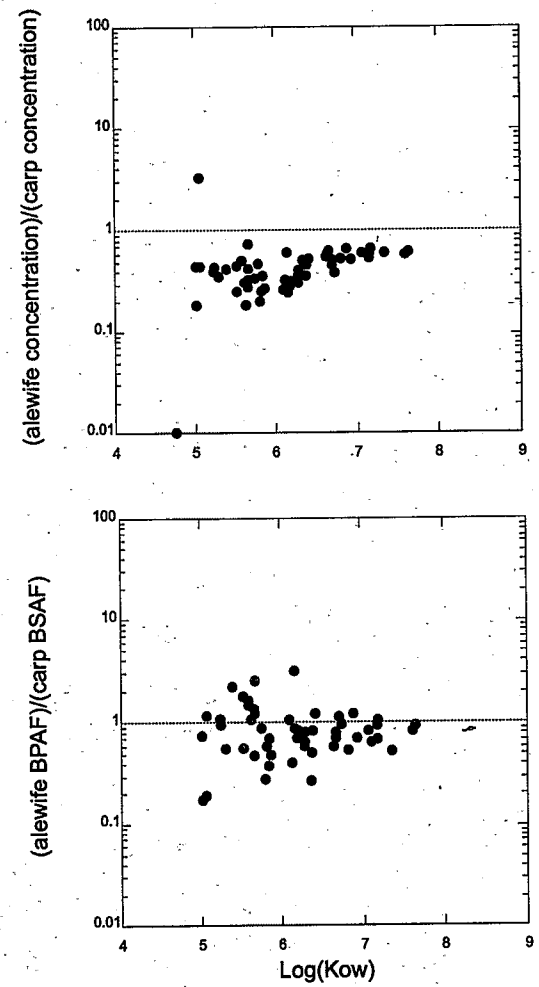
HydroQual, Inc.



open bars: Dover Sole

filled bars: White Croaker

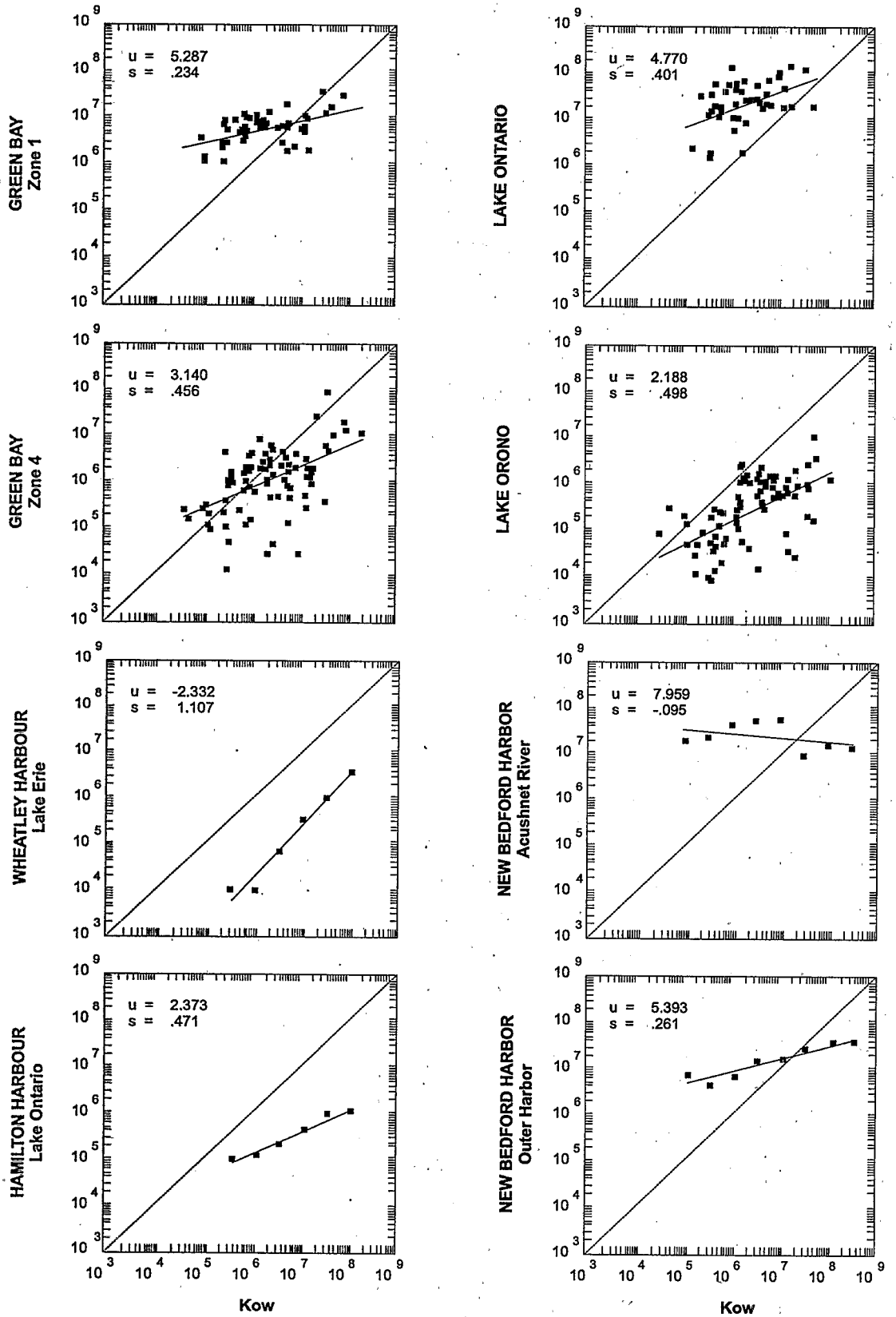
Total PCBs and ppDDE BSAFs in fish from the Southern California Bight



PCB levels in a pelagic forage fish (alewife) and a benthic fish (carp) in Green Bay zone 3B. Top panel: Ratio of lipid-based concentrations. Bottom panel: Ratio of alewife BPAF/carp BSAF

$$\text{BPAF} = \frac{\mu\text{g PCB/g lipid alewife}}{\mu\text{g PCB/g OC phytoplankton}}$$

$$\text{BSAF} = \frac{\mu\text{g PCB/g lipid carp}}{\mu\text{g PCB/g OC sediment}}$$



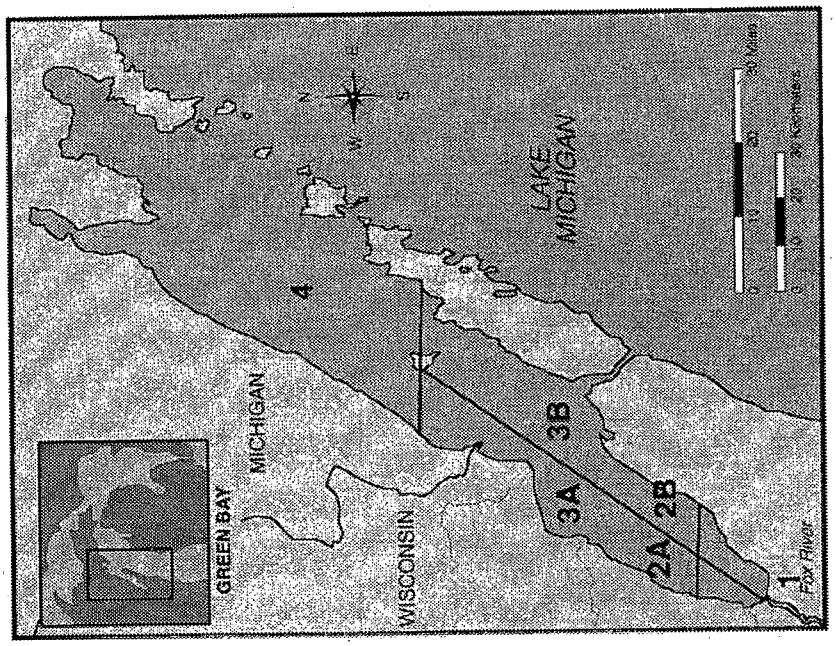
Water-Sediment partitioning for PCB homologs and congeners (L/kg OC)

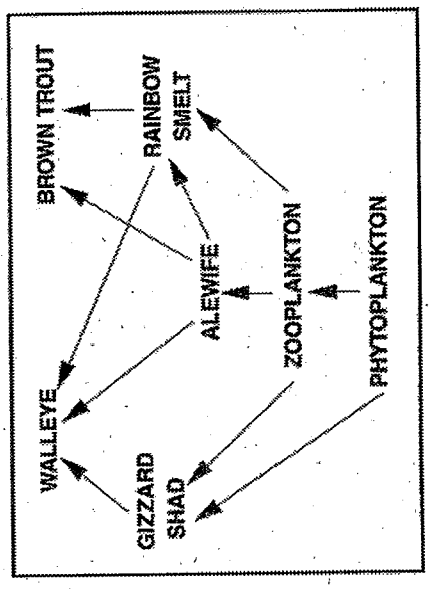
How can we interpret BSAF and BAF data?

By using food web bioaccumulation models.

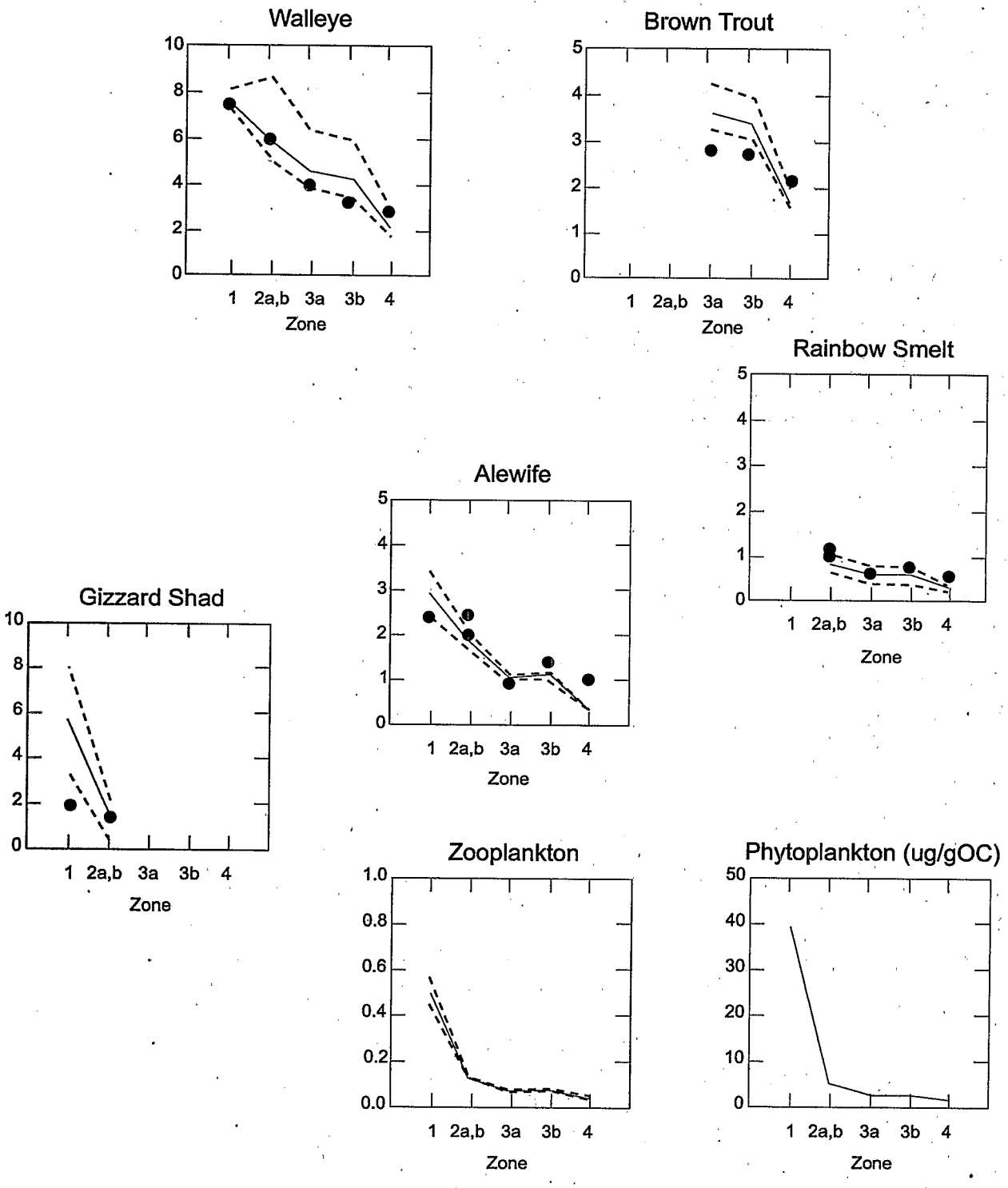
Models provide this capability because:

- They describe the process of bioaccumulation and can account for water & sediment exposure, time variability and food web structure
- They are credible: capable of reproducing field data using parameterizations that are supported by experimental data and are consistent within and across food webs
- Their uncertainty is not overwhelming



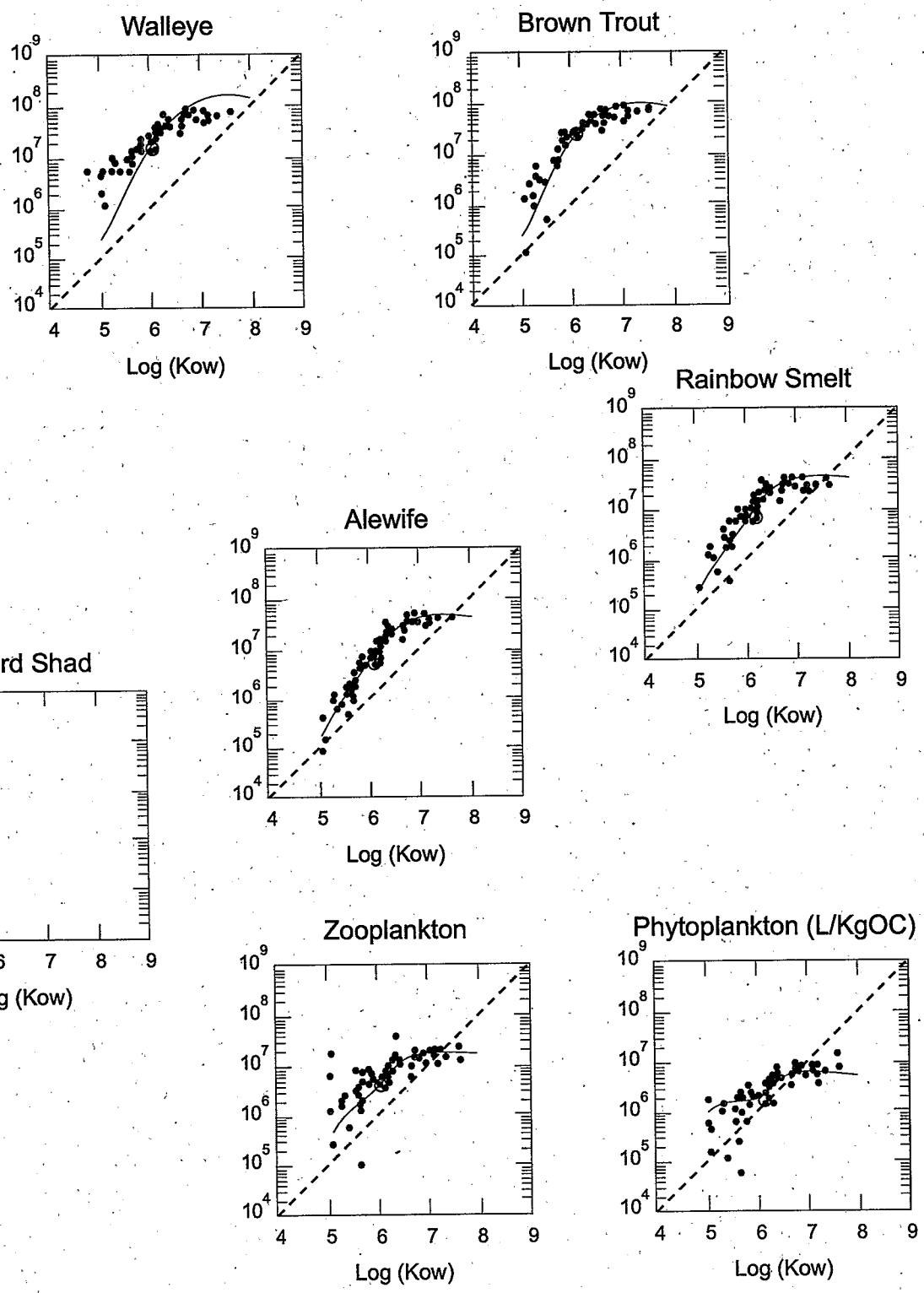


GREEN BAY WET WEIGHT-BASED CONCENTRATION TOTAL PCBs

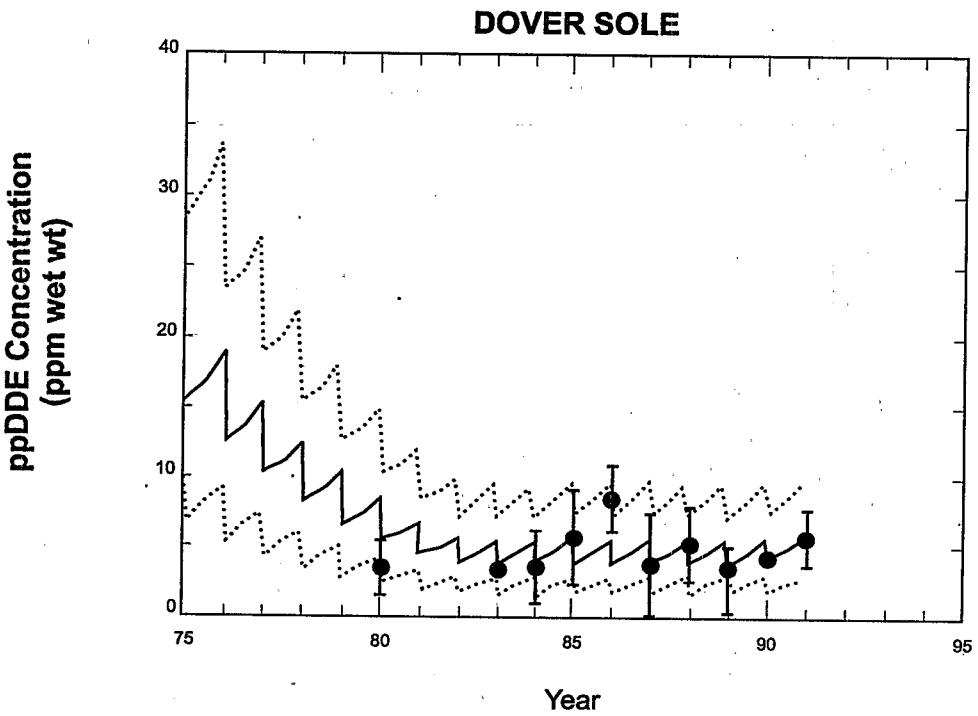


Food Web Model Calibration. Computed and observed total PCB concentrations, µg/g whole body wet weight, for all spatial zones. Lines: model calculations. Filled circles: data.

GREEN BAY LIPID-BASED BAFs



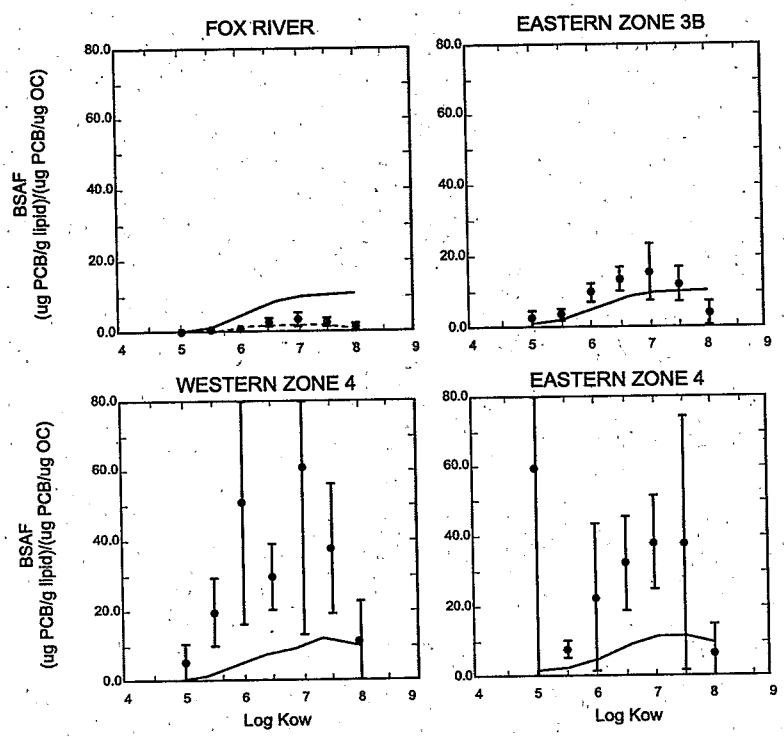
Food Web Model Calibration. Computed and observed lipid-based bioaccumulation factors for zone 3A. L/Kg lipid.



Comparison of Predicted and Observed ppDDE Concentrations in Dover Sole
Solid Lines are Model Results
Dashed Lines are 95% Confidence Intervals around the Mean (Monte Carlo)
Data are Arithmetic Means and 95% Confidence Intervals around the Mean

Using a model to examine the Green Bay carp data we conclude the BSAF values in the Fox River and zone 4 represent non-steady state between the fish and the sampled sediment

- Differences in toxicokinetics and bioenergetics do not account for the spatially variable BSAFs
- Differences in food web structure do not account for the spatially variable BSAFs



Bioaccumulation Factors for PCBs in Green Bay Carp
PCB Congeners Grouped into 0.5 Log Kow Bins
Data are Arithmetic Means and 95% Confidence Intervals

Conclusions

- Models provide a means to interpret bioaccumulation data
- Models are necessary to test assumptions implicit in data-based BAF and BSAF values
  - to refine the database of BAF and BSAF values used for regulatory purposes
  - to increase confidence in regulatory decisions having substantial economic implications

Bioaccumulation Modeling of PCBs in the Hudson Estuary: A Review and Update

Robert V. Thomann and Kevin J. Farley
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Introduction

The issue of polychlorinated biphenyl (PCB) accumulation in the striped bass of the Hudson estuary has been a matter of concern for several decades. Early discharges of PCBs to the estuary resulted in total PCB concentrations in the striped bass during 1978 to 1982 of 5 to 10 times the U.S. Food and Drug Administration (FDA) action level of 2 µg/g(wet). The fishery has been closed to commercial harvesting since the mid-1970s.

Bioaccumulation Model

A linked homolog-specific, physicochemical bioaccumulation model was constructed (Thomann et al., 1989, 1991) to help address questions of the effectiveness of upstream controls on PCBs in reducing the PCB concentration in the striped bass or, if no action is taken, the time it would take to reach acceptable levels in the fish. Model calibration was accomplished with striped bass data from 1978 through 1987 and indicated that more than 90 percent of the observed PCB concentration was due to food web transfer. Projections were made of the expected decrease in PCB concentration in the striped bass from 1987 to 2010. The projections were based on a PCB homolog-specific modeling framework that included a time-variable, age-dependent striped bass bioaccumulation model. Those results indicated that under a no-action scenario, the average PCB concentration in the striped bass in the mid to lower estuary would decline below the FDA action level of 2 µg/g(wet) by about 1994 to 1995.

Model Post-Audit

Data obtained by the New York State Department of Environmental Conservation (as stored in TAMS, 1996) are now available to check the projections up through 1994. Observed mean total PCB concentrations in 2- to 5-year-old striped bass in the mid-lower estuary (designated SB2 group) declined from about 3.5 µg/g(wet) in 1988 to about 1.6 µg/g(wet) in 1994. A correction of the original projection was necessary due to a numerical error in exposure concentration in Food Web Region #4, the Long Island Sound and New York Bight area. This correction resulted in a reduction of less than about 0.5 µg/g(wet) from the original projection. Using the original input load projections, the comparison to the observed data for SB2 is shown in Figure 1 and indicates that the model underestimated the mean in 1989 and 1990 but adequately reproduced the means in 1992 and 1994. The variability in the data is large with 5 to 95 percentile

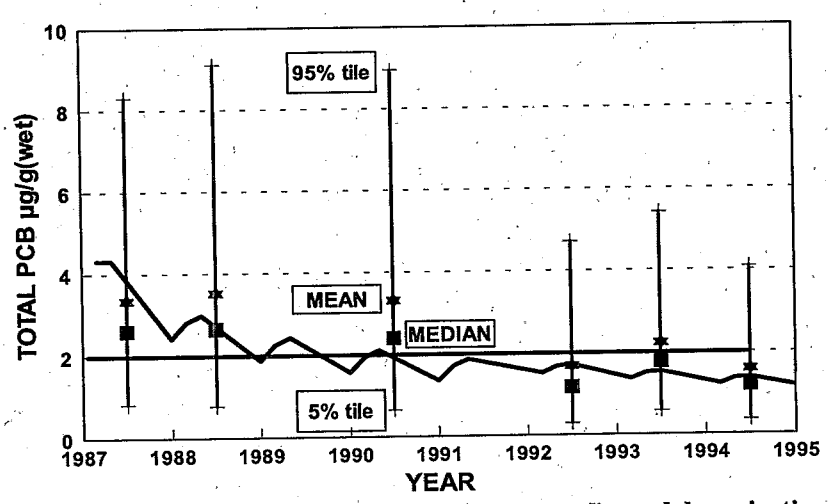

Figure 1. Comparison of original (as corrected) model projection (solid line) made in 1987 to observed data of striped bass (2-5 yr old), Food Web Region #2.

range in individual samples of about an order of magnitude distributed approximately lognormally. Projections of the percent frequency of samples below a target level mean of 2 µg/g(wet) were also made, and comparison to observed data (Figure 2) indicated that the original model overestimated the frequency =< 2 µg/g(wet) in 1990 but tracked the general trend in the data of an increasing frequency=<2 µg/g(wet) in the years 1992 to 1994.

The model has been updated in two areas: assessment of the effect on striped bass from a temporary increase in upriver loading and an evaluation of a varying migration pattern of the striped bass. To assess the effect of the increase in loading, a fivefold increase over the originally projected load from the upper river was added for the one year of 1991. Preliminary results indicate a calculated increase of total PCB concentration of about 0.4 to 0.8 µg/g(wet) for 1992 to 1994 in the SB2 group. The effects of the increased load are dissipated in about 5 years in the mid-lower estuary.

For the effect of varying migration, recent research by Secor and Piccoli (1996), who estimated the exposed salinity regime in striped bass from measurements of the strontium/calcium (Sr/Ca) in the otolith, has indicated that some male striped bass remain in the mid to upper estuary and do not migrate to the ocean. Figure 3 shows a comparison of female and male SB2 total PCB concentrations from the data base. The higher median concentration in the males is observed, as well as considerably higher extreme concentrations in the male SB2 over the female SB2 group. The model results for nonmigratory behavior, shown in Figure 4, indicate an approximate doubling of the concentration in the striped bass and help explain the elevated concentrations that have been observed.

Projections to Year 2010

Revised long-term projections based on varying bass migration patterns and the original load projections are shown in Figure 5 for the mean and Figure 6 for the estimated frequency of occurrence =< 2 µg/g(wet). The revised projections indicate that by the year 2000, the mean total PCB concentration is estimated to be between 0.8 and 1.8 µg/g(wet) with an estimated 75 to 95 percent of the total PCB concentrations in the SB2 group = < 2 µg/g(wet). However, for the mid-upper estuary, calculated mean total PCB concentration is about 2.1 µg/g(wet) for 2- to 5- year-old bass, indicating that this region can be expected to have elevated concentrations over a longer a period of time than the mid estuary region.

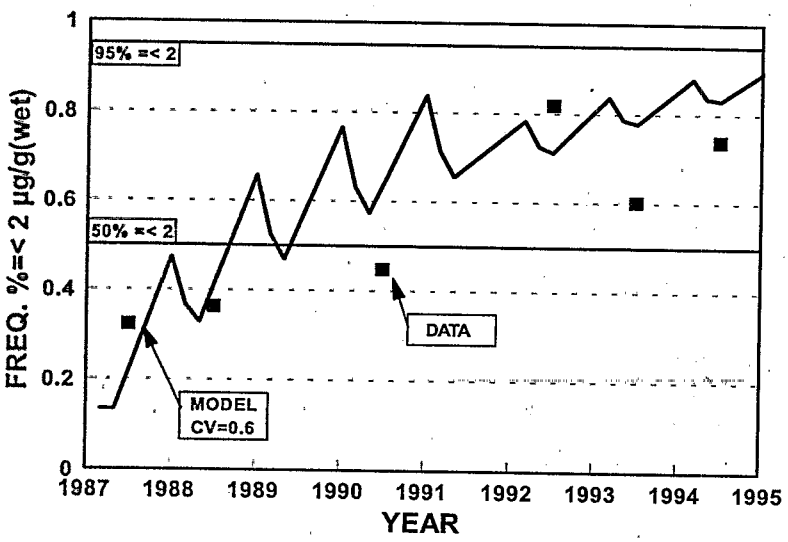

Figure 2. Comparison of observed vs. model exceedance frequencies (%=< 2 µg/g). Striped bass (2-5 yr old), Food Web Region #2.

Conclusions

For this work, which is still in progress, it is concluded that the original model generally tracked the observed decline in mean total PCB concentrations in the striped bass for the mid to lower estuary although there are year-to-year variations in the mean that are not captured by the model. A simulation of a 1-year "pulse" load of five times the estimated load in 1991 indicated that such an input persists in the SB2 group for about a 5-year period. Migration behavior is particularly significant and contributes to the observed variability in the SB2 data. The revised projections estimate that SB2 total PCB concentrations will continue to decline to levels below the FDA action level. It should be noted, however, that such projections are strictly related to the projections of the load and, of course, the model itself. Reevaluation of the loading using contemporary estimates and model recalibration are currently under way. There is a continual need to monitor the input PCB loads and the concentration in the striped bass and couple such monitoring with periodic model post-audits and updates.

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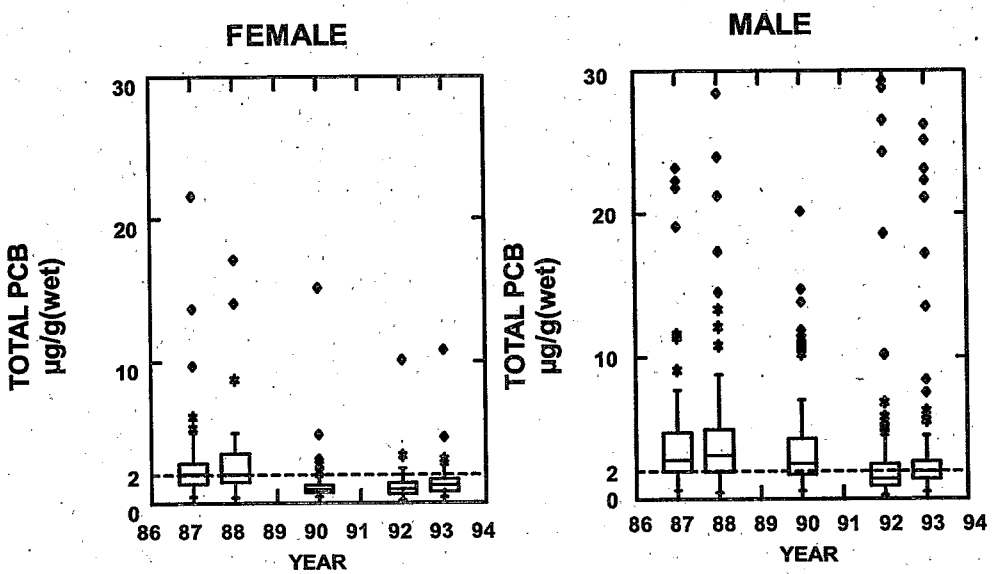

Figure 3. Variation in total PCB concentration in striped bass (2-5 yr old), Food Web Region #2. Line in box = 50th %tile, box boundaries = 25th and 75th %tile, symbols = outliers.

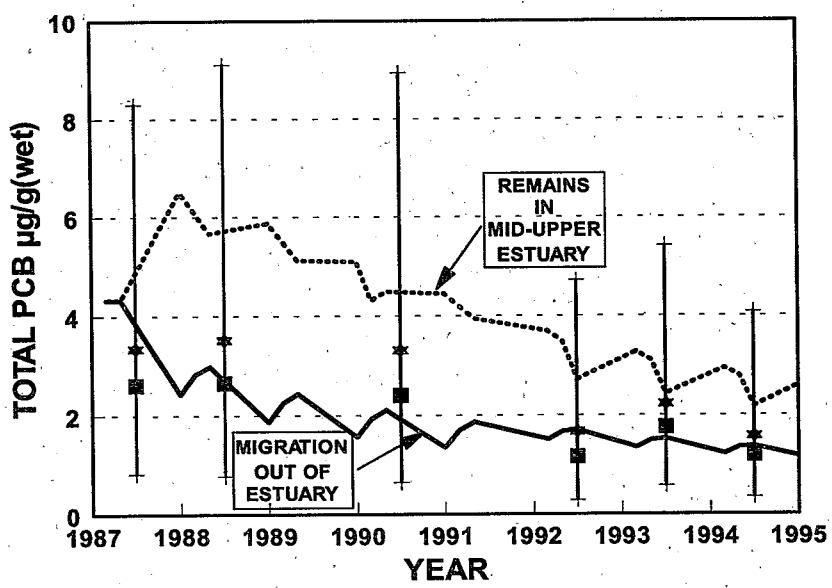

Figure 4. Calculated effect of varying migration of striped bass (2-5 yr old), Food Web Region #2.

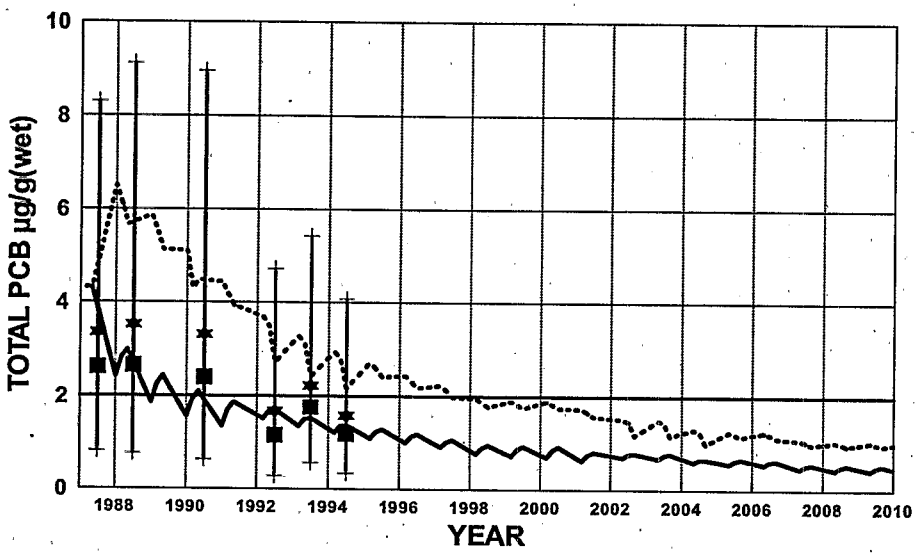

Figure 5. Long term model projection using original (1987) load projections with varying migration by striped bass. Solid line: striped bass migrates out of estuary; dashed line: fish remains in estuary. Striped bass (2-5 yr old), Food Web Region #2.

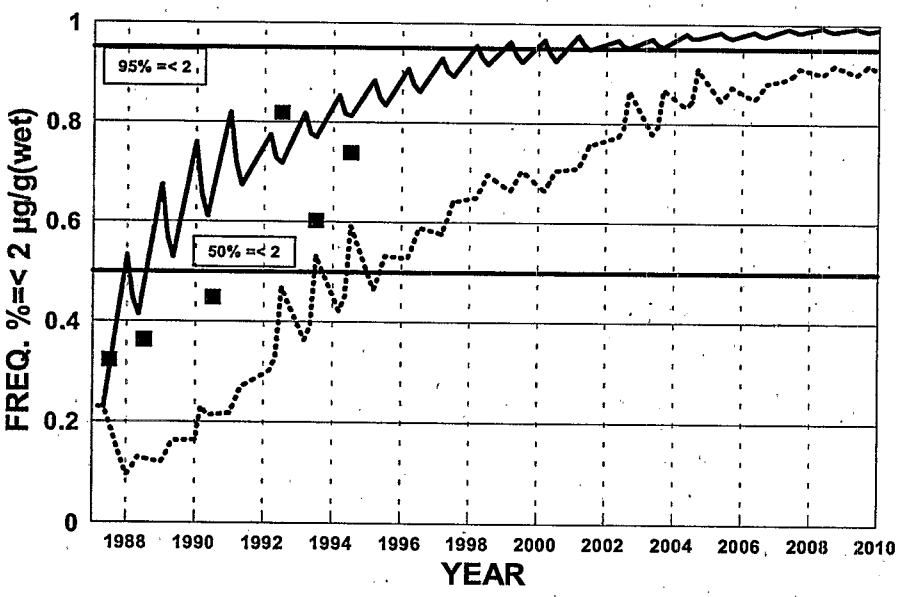

Figure 6. Long term model projection of frequency of striped bass =< 2µg/g with varying migration by striped bass. Solid line: striped bass migrates out of estuary; dashed line: fish remains in estuary. Striped bass (2-5 yr old), Food Web Region #2.

Day Two: September 12, 1996

Session Four: Questions and Answers

After each session, there was an opportunity for questions and answers and group discussions pertaining to the speakers' presentations.

Q (Ron Sloan, NYS Department of Environmental Conservation): One of the things that Bob brought up, which is very interesting from our standpoint as researchers in a regulatory agency, is the possibility of regulating on the basis of sex in fish. This would be very difficult to do, but it raises a very valid point in terms of male and female differences, not only for accumulation and behavior in the estuary, but also in other systems and for other species of fish. For a number of years we have noted tremendous sex differences, with the males always being higher in concentration than the females for striped bass, American shad, white perch, the Pacific salmon in Lake Ontario, and a number of other species. Bob, is there as much work going on with you and other researchers as it relates to sex differences?

Robert Thomann:

I am not doing anything myself, but I have been following the literature rather closely. There is a lot more information now. I think John has done some work on seals. Recently, the Dutch did some work on exposing males and females and looking at bioaccumulation in the laboratory, which showed significant differences. But I have a question back to you. Did I understand your opening comment to mean that you really are thinking about regulating by sex?

Q (Ron Sloan): No, no. There is no real way to be able to do that, but it is very bothersome that we do not have any good explanation for why the males are so much higher in concentration, even controlling for gamete production and the lipid consequences associated with voiding of egg material from the females. The sex differences are still there and they are very dramatic.

Robert Thomann:

Well, I think it is because they are staying in the estuary.

Q (Ron Sloan): Maybe for striped bass, but even if you look at the males in the lower part of the estuary that are freshly migrating in from the ocean, the males will tend to be higher in concentration. That still does not explain the sex differences that we see very dramatically in Pacific salmon and white perch. One of the reasons I raised the point is because we have been looking the past few days now at the uncertainties associated with BAFs and BSAFs, and I think a lot of the uncertainty and the variability is associated with sex differences when we start looking at fish.

Robert Thomann:

Well, I think that is true. The sex differences, at least for this case, do contribute significantly to the variation. If you calculated BAFs here, they would be all over the lot. As John pointed out, part of it is the disequilibrium between the fish and the water. This would be the same for BSAFs. They would be very time variable and very spatially variable.

Q (Ron Sloan): We have looked at that to some extent, and we cannot really account for it because the lipid concentrations are so low. Gamete loss cannot explain the differences.

John Connolly:

We have looked at a lot of data in a more generic sense and it seems to be a mixed bag. We do see a difference for some species, but we do not for other species. You particularly see it in aquatic mammals. Milk production and the loss due to that explains the differences for the aquatic mammals. But I do not think we understand the reason for those differences in fish.

Q (Ron Sloan): My primary reason for bringing this up as a question or an issue is that, if we are interested in explaining a lot of the variability that we are observing, we need to take into account the sex differences.

Q (Nelson Thomas, U.S. EPA, Office of Research and Development): This is a question for John and Bob. In Frank's presentation, he alluded to an uncertainty factor

of 3 at the 95 percent confidence level. Did you look at your 95 percent confidence level in your calculations and figure out how far off you would be like Frank did?

John Connolly:

The one that I showed, which was for Dover sole, was about a factor of 2. In a couple of other calculations, where we have done uncertainty factors, the factor of 2 to 3 is pretty robust. That is what we are generally seeing in these calculations.

Robert Thomann:

That is true. If you look closely at those percent exceedance frequencies, you will see about the same kind of variability.

Larry Burkhard:

John, I have a slide of the probability distribution for the 2-year old flounders from your New Bedford harbor model. This figure is from a publication in the SETAC Journal. It illustrates the uncertainty that they found using a Monte Carlo simulation for his model. You can see here that a factor of 2 to 3 is right on line. I just wanted to point out the kind of uncertainties that you have with these models.

Robert Thomann:

I interpreted Nelson's question as asking what our uncertainty was in the 95th percentile estimate, or in other words, in the tail of the distribution. We have the ability to project or analyze for the 95th percentile to a factor of about 2 or 3.

Q (Jay Field, NOAA): In situations where we have vertical gradients in terms of sediment concentrations, has anybody looked into the implications of biological activity, which may vary seasonally or by location, and the impact it might have on some of the models? How would any effects that we see affect the way we collect information?

John Connolly:

With some of the work I have done, I have seen more location differences. I have not paid a lot of attention to the seasonal differences that might exist. So, I cannot comment on seasonal differences within a system. But clearly, we see differences from system to system, depending to a large extent on the type of organisms comprising the benthic infauna. In determining whether or not we are worried about the top centimeter or the top 5 centimeters as being biologically available material, that is more of an issue for higher level organisms. This is something that again introduces considerable uncertainty, both in models and into the BAF and BSAF. That was one of the points I was trying to make with the Fox River. The model results suggest that the carp are seeing material recently deposited in the river, as opposed to material that is buried a few centimeters down. If carp were exposed to material a few centimeters down, they should have been at much higher concentrations than they were.

Robert Thomann:

The only calculation I have done in that area was associated with the cadmium and the blue crab in Foundry Cove, where there was a considerable amount of sediment mixing and sediment bioturbation. In that particular case, bringing up deep sediment to the surface and then letting that get into the food chain, actually retarded or slowed down the recovery of the cove. Recovery was delayed because you are just accessing higher concentrations over a longer period of time. It is very interesting. It increases the flux to the water column, so you are actually depurating out the sediment with a higher flux. But, because you are also reaching down into higher concentrations, you also retard the response.

Q (Ken Finkelstein, NOAA): Dr. Thomann, you presented a figure early in your talk that showed the concentrations in the striped bass with a no-action or as is. You also showed on the same figure one that would indicate that there was no change at all, even with some kind of decrease in the load. Actually, I think you decreased it down to zero, or in other words, it was remediation. I was wondering if you could shed some light on that because it puts some question in remediation plans, both there and elsewhere. I have seen this at other sites with models where the models do not show any kind of significant change with drastic cleanup operations.

Robert Thomann:

What you are focusing on is the calculation that was done in 1987 and 1988. There actually is a difference between remediating the upstream sediment and the no-action. The difference is only in the sense that it takes you a little bit longer to get to a certain percent frequency below 2, if you do not do anything. But eventually they come together. The reason for that is we have projected in the calculation, which I showed in the load projection, that the upstream load will continue to decline whether you do something or not in a no-action case. So, that was that load projection going down. That is what we did in 1987, and that is what we are still doing today. That is being revised, but we are still in the process of doing that. That load is declining, but there are also downstream loads, including loads in the metropolitan area, nonpoint source loads, and atmospheric loads. That calculation essentially said that, after a period of time, the relative contribution from the downstream sources became more dominant, and the upstream source became less dominant. So, its impact became less and less as time went on, and remediating it had less and less of an impact.

Q (Ken Finkelstein): Maybe I can also relate a little bit of a different experience that we have had in doing similar types of calculations. Consider the case where the

remediation is remote to where you are looking. For example, you are remediating in the upper Hudson River and looking at benefits in the lower Hudson River. In many cases, you do not see a big bang because that load is not controlling the concentrations and the exposure in the remote area. In the case of the lower Hudson, it may be because there is another load. In other cases, it may be that past historical discharges are controlling the system, rather than the current discharges coming from an upstream area. In these cases, eliminating the upstream source will not show an impact in the downstream area since that source was not contributing to contaminant levels in the fish downstream.

Robert Thomann:

That says nothing about the advantage of remediating for the upriver area. There is a local benefit from that.

Q (Arnold Kuzmack, U.S. EPA, Office of Science and Technology): I would like to make a couple of comments of a somewhat philosophical nature. First, in terms of the discussions of the uncertainty being within a factor of 2 or 3, I think we have to be careful of how we use the terminology here, particularly when distinguishing between variability and uncertainty. Uncertainties tend to deal more with things that we do not know. For example, I am sure when you were doing your original models in the 1980s, you assumed that the fish were homogeneous in a migration pattern. You probably did not realize you were making that assumption. If you had listed your assumptions, that probably would not have appeared there, and it turned out to be wrong. That is an example of uncertainty. There are other uncertainties in just how well can we measure, what all we are including in the models, and so forth. The second comment I would like to make relates to John's discussion of the data versus modeling. I absolutely agree with your comment that whenever you use data you have some sort of implicit model. I would indicate caution on the part where you said that the models incorporate everything we know about the science. I think almost by definition they do not include everything. They include some sort of schematization and rationalization of our knowledge. And we sort of discard the things we do not understand that are in the data. I think because of that we have to be careful to do an appropriate degree of ground truthing of the models, which you all did in your research. I think one could criticize regulatory agencies in perhaps putting too much reliance on models that do not have that kind of ground truthing, particularly for situations like some permit process applications or decision-making based on model results that are not clearly understood. I think in these sorts of situations, you have to be pretty careful about having the models run away with things. I think that is what probably the data advocates are trying say.

Robert Thomann:

I think the other point, too, is deciding when a model is useful for regulatory or management purposes. Who makes the decision that models are ready for use in some kind of decision-making process? I have always felt that regulators and lawyers do not make that decision. They are not in a position to evaluate it. The people who ultimately make the determination of whether a model is suitable for management purposes or decision-making are members of this community. It is the scientific community that determines when the model can account for all the important factors and produce results that make sense relative to the data. This is the group that throws holy water on it.

Q (Dave Michaud, Wisconsin Electric Power Company): John, in looking at the carp data from the Fox River and Green Bay system that you presented, I am not surprised about what you saw. The carp population in the Fox River is a restricted population that lives in a highly regulated environment with a number of pools controlled by dams. You see greater variability in Green Bay because there is an intense depositional gradient along the east coast, and the carp can and do move freely between Green Bay and Lake Michigan. I would guess that you were probably sampling fish that were 10 to 15 years old, so they have a unique uptake history relative to the alewife. That leads me to my second comment. At one point in your presentation you grouped alewife and carp together in terms of trophic levels. I am not sure I understand the logic behind that. Except for an occasional amphipod or mysid, alewife feed primarily on plankton. The food habits work that we have done on carp in Lake Michigan suggests that it feeds almost exclusively on benthic organisms. We have found very little plankton in examining the gut contents. I would not call these the same trophic levels. How would you also factor in the obvious age differences in terms of exposure? Typically, alewife live only 3 years, while carp can survive for more than 25 years.

John Connolly:

The data that we had was for carp over a rather wide range of ages, so we were not looking specifically at carp that were 14 or 15 years old. When I was talking about the Fox River, I was referring to the lower Fox River below DePere Dam, where the carp are not confined in pools. But you just raised a lot of questions that I think are all valid. The point I was trying to make is that models allow you to explore those questions, as opposed to just using them to generate a number you can run around and extrapolate with.

Q (John Zambrano): This is a question for the panel from a regulator. Yesterday Philip Cook described the work that EPA did for the Great Lakes Initiative (GLI) for the purposes of establishing BAFs for setting water quality standards. What does the panel think of that work, particularly its applicability to the rest of the nation and the use of a disequilibrium constant of 25?

Frank Gobas:

My understanding of the process is that they used a bioaccumulation model for a particular system to derive

bioaccumulation factors, food chain multipliers, and BSAFs. Then they applied these values to different systems, assuming that they are applicable for these other systems. What we try to do in our modeling is exactly the opposite. We are providing people with models that allow them to put in site-specific factors so that they can derive these values, the food chain multipliers, BSAFs, and BAFs, for various systems. We believe that not all systems are the same and that you cannot extrapolate from one system to another. So, my feeling is that the models can be used much more effectively when they are applied on a site-specific basis to work out these important bioaccumulation properties.

John Connolly:

I think you have to put the GLI work into context. What the GLI work was attempting to do was to develop bioaccumulation factors that could be applied to the Great Lakes. I do not think they had it in mind to use these factors for national criteria. So, I do not think that there is any intent to say that this approach is automatically applicable nationwide. I think that the GLI documents are very careful to point out a lot of the things that came out in these talks, in terms of the cautions in the use of this data. They are very clear in talking about site-specific application. They provided a generic number because it was their responsibility to do that. But they also provided all of the cautions associated with the application of those generic numbers.

Larry Burkhard:

I might add that in the GLI itself there is a tiering, starting with field data from the Great Lakes and ending with something that is totally model-derived from very generic kind of parameters from Lake Ontario. That is reflective of the knowledge that we had at the time that that was constructed and advanced.

Q (Mike Kravitz, U.S. EPA, Office of Science and Technology): This is sort of along the lines of the last question. What happens in terms of marine waters? What is the value of using things like BSAFs in systems that are so dynamic? Will we be able to do that?

Robert Thomann:

BAFs and the BSAFs have served us well and will continue to serve us well as kind of a first-order screening. There will always be a utility to BAFs and BSAFs combined with the kinds of calculations we have been showing. So, I do not think it is totally a question of discarding entirely the BAF and BSAF concept. That is not the point. The point is to be aware of the kinds of variability that can exist and the causes of that variability in those two numbers. In marine systems, I think ultimately we would get to the point of looking much more specifically at fully time-variable calculations, which, up to a point, are entirely doable today by just about anybody. Again, we must recognize all of the caveats and all of the assumptions that have been made. But I think we may be heading in the direction of working on those kinds of systems, supplemented by the BAF and BSAF concepts.

Q (Gayle Garman, NOAA): I have spent most of my career dealing with issues of whether a site should be remediated and to what degree it should be remediated, particularly sites with PCB contamination. So, I was particularly interested in Dr. Thomann's presentation and his projections for the Hudson River. I found it a little disturbing since it seems to imply that the PCBs in the Hudson River are going to essentially be assimilated. Am I going to end up doing calculations of assimilative capacity for PCBs? I think about that and also about some data we have for Puget Sound, which show that the salmon that return to Puget Sound actually have higher levels of PCBs in their tissue than the salmon that leave Puget Sound. So, I wonder again, if your models are very accurate for the little part of our system that you are modeling, but that they do not show us what happens outside of that system. I would like you to comment on that and what you think your models imply for decisions about remediation.

Robert Thomann:

Again, I think remediation has an impact locally. There is no doubt about that, since that has been shown any number of times. This was a question of remediating an upriver site and its impact on a migratory fish population. So that is the focus. One should not conclude from the results that remediation does not affect a population or the concentration of PCBs in a population. It does. But it depends on its relationship to the target organism. I do not mean to imply the question, why remediate? Why do the PCBs seem to be declining? Fundamentally, what happens in this calculation is the PCBs ultimately are being flushed out into the ocean and the exchange with the ocean is diluting it. There is also a continual burial of PCBs. If you look at the rate of decline of the striped bass over time and make an estimate of average burial rates over that period of time (and this is the crudest calculation), you find out that PCBs are continually being buried. Also, the assumption here is that the upstream load is declining. Why is that declining? It is declining for similar kinds of reasons. But remember, this is a long-term projection. There are two issues you need to be aware of. One is this calculation assumes that the loads to the system from point sources and nonpoint sources will continue to decline. That may not be the case. They may level off at some point. The second is that this calculation assumes when the fish goes out to the ocean, it sees zero PCBs. That is what we did in the old days. We are revisiting that part of the calculation, now that much better data are available for what is going out into the ocean. So we send these larger fish out 8 or 9 months of the year, and they see zero PCBs. If the concentrations out there are now higher than zero, which in fact they are, then those fish are going to come back at a different concentration than we had originally calculated.

Risk Assessment Overview

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Please contact the speaker for information on this presentation.

Session Five: Human Health-Based Risk Assessment

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Methodology for Assessing Human Health-Based Risks

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U.S. EPA Region 5,
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Bioaccumulation Models and Applications: Setting Sediment Cleanup Goals in the Great Lakes

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Use of Human Health- and Ecological-Based Goals in Developing a Whole River Sediment Strategy: Fox River, Wisconsin

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Olympia, Washington
Development of Health-Based Sediment Criteria for Puget Sound

Alex Lechich
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Development of Bioaccumulation Guidance for Dredged Material Evaluations in EPA Region 2

Methodology for Assessing Human Health-Based Risks

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The purpose of this presentation is to discuss how and why human health risk assessments are conducted at contaminated sediment sites. The main components of risk assessments (i.e., site characterization, toxicity assessment, exposure assessment, and risk characterization) are described. In addition, the advantages and limitations of using a risk assessment approach, as it applies to bioaccumulative contaminants, are discussed. The variety of methods by which human health risk assessments can be conducted are described, and these methods are illustrated with examples from baseline and comparative risk assessments conducted as part of the Great Lakes National Program Office's (GLNPO) Assessment and Remediation of Contaminated Sediments (ARCS) program.

Why Use a Risk Assessment Approach?

Risk assessment provides a scientific basis by which estimates of noncarcinogenic and carcinogenic risks can be estimated at a contaminated site. These risk estimates can be used to address management questions about a site; for example, what are the baseline (i.e., current) risks to residents living in the Sheboygan River, Wisconsin, Area of Concern (AOC)? or how would risks change at the Buffalo River, New York, AOC if various remediation alternatives were implemented? A risk assessment approach can also be used for regulation and enforcement purposes, as well as for the selection of clean-up criteria.

In the United States, the triggers that initiate a risk assessment include regulatory action leading to placement on the National Priorities List or a state-equivalent list of contaminated sites. Thus, human health risk assessments are conducted at all Superfund sites. In contrast, Canada has additional trigger mechanisms for risk assessment, including real estate transactions, rezoning/redevelopment, regulatory placement on a priority list, new legislation, or regulatory orders (Golder Associates, 1993). Canadians also have the option of using a criteria-based approach instead of a risk-based approach at contaminated sites.

The primary advantages of using a risk assessment approach include the following:

- Provides a quantitative basis for comparing and prioritizing risks.
- Improves the understanding of risk.
- Acknowledges the inherent uncertainty in estimating risk.
- Estimates clear, consistent endpoints (e.g., cancer).
- Separates risk assessment from risk management.

The limitations and uncertainties associated with human health risk assessments are discussed in the following sections.

Components of Human Health Risk Assessments

The specific components of human health risk assessments are described in this section. The U.S. Environmental Protection Agency (EPA) has developed exposure and risk assessment guidance for use at Superfund sites (USEPA, 1988, 1989a, 1989b, 1991) which can be applied to other contaminated sediment sites. Refer to this guidance for detailed information about how to perform a site characterization, toxicity assessment, exposure assessment, and risk characterization for a contaminated site. A brief review of these steps is given below.

Site Characterization

Available site information is reviewed, and relevant site samples are gathered, analyzed, and evaluated for appropriate quality assurance measures in this step. Potential site-related contaminants are compared with background values. In addition, potential contaminants of concern are identified, and a set of data is developed for use in the risk assessment.

Toxicity Assessment

Verified toxicity values are obtained from EPA's Integrated Risk Information System (IRIS) database. For chemicals lacking a "verified value," interim toxicity values can be obtained from EPA's Health Effects Assessment Summary Tables (HEAST) and from the literature. The reference dose (RfD) is the toxicity value used to evaluate noncarcinogenic effects (e.g., neurotoxicity, reproductive and developmental toxicity, immunotoxicty, organ-specific toxicity). The slope factor is the toxicity value used in evaluating carcinogenic effects. It quantitatively defines the relationship between dose and response. The EPA weight-of-evidence classification scheme indicates the strength of evidence that the contaminant is a human carcinogen.

Exposure Assessment

This component involves the greatest use of assumptions and professional judgment when site-specific information is lacking. The major parts of an exposure assessment include the following: characterize the physical setting, identify potentially exposed populations, identify potential exposure pathways, estimate exposure concentrations, and estimate chemical intakes. The most important exposure pathways identified at five Great Lakes AOCs included the consumption of contaminated fish and/or waterfowl (Crane, 1996). In addition, most of the carcinogenic risk was due to elevated concentrations of polychlorinated biphenyls (PCBs) in fish tissue. In the exposure assessment, it is important to identify sensitive subpopulations such as sport anglers, ethnic groups that consume a greater proportion of fish in their diet, and pregnant women who consume contaminated fish. During the past few years, the Agency for Toxic Substances and Disease Registry (ATSDR) has funded a Great Lakes Human Health Effects Research Program that will provide more exposure information about sensitive subpopulations in the Great Lakes area, as well as how fish preparation and cooking practices can reduce bioaccumulative contaminants in fish tissue.

Risk Characterization

In this step, the exposure and toxicity estimates are combined into an integrated expression of human health risk. Noncarcinogenic effects are evaluated by comparing an exposure level over a specified time period with an RfD derived from a similar exposure period. Carcinogenic effects are estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to the potential carcinogen. This risk is computed using average lifetime exposure values that are multiplied by the oral slope factor for a particular chemical. Noncarcinogenic and carcinogenic risk estimates are summed separately for all chemicals in an exposure pathway (e.g., fish consumption). The risk estimates are then summed for multiple exposure pathways. This summation does not account for any synergistic or antagonistic effects that may occur among chemicals. The risk characterization also includes an evaluation of the uncertainties associated with the risk estimate. Uncertainty is reduced by using as much site-specific information as possible. Refer to USEPA (1989a), Golder Associates (1993), and Crane (1996) for additional information on evaluating uncertainties.

Risk Assessment Methodologies

Given the risk assessment framework described above, human health risk assessments can vary in their complexity and scope. Risk assessments can be conducted in a qualitative, semi-quantitative, or quantitative way. The level of detail and cost associated with conducting a risk assessment increase as more quantitative results are needed. The human health risk assessments conducted for the ARCS program (Crane, 1996) were done in a semi-quantitative way because uncertainty was expressed in a qualitative way and quantitative risk estimates were estimated. Semi-quantitative risk assessments compose the greatest proportion of risk assessments conducted in the United States.

Semi-quantitative risk assessments are conducted using a deterministic approach (see Table 1). In this approach, a single number is used from each parameter set to calculate a single value of risk. Quantitative risk assessments are conducted using a stochastic approach (see Table 1), whereby a distribution of values for each parameter set is used in the risk calculation and a distribution of risk is produced. Some jurisdictions, such as British Columbia, are starting to require quantitative risk assessments at some contaminated sediment/soil sites.

Human health risk assessments conducted for the Superfund Program include baseline conditions and future land use exposure scenarios. Future land uses do not need to be considered if the management questions for the non-Superfund site are concerned only with current risk conditions to the public. Comparative risk assessments can be conducted to assess the risk, relative to the baseline risk, that would result from the implementation of various sediment remedial alternatives. The ARCS Risk Assessment and Modeling Workgroup developed a comparative risk assessment framework to integrate the results from baseline risk assessments, field data, and mass balance modeling to provide estimates of the potential impact of remedial actions on human health, aquatic life, and wildlife (USEPA, 1993). A demonstration of this approach was conducted for the Buffalo River, New York, AOC (Crane, 1995).

Problems with Current Risk Assessment Practice

Risk assessment is an evolving science and is not without its limitations. Some of the major problems associated with human health risk assessments include the following:

- Use of arbitrary exposure scenarios.
- Excessive credence to the carcinogen classification.

Table 1. Components of deterministic and stochastic risk assessments (Golder Associates, 1993).

Deterministic	Stochastic
Low degree of public interest	High degree of public interest
Relatively low degree of site-specific uncertainty	Relatively high degree of site-specific uncertainty
Deterministic result is far from action level	Deterministic result is close to action level
Small-scale project (scope, budget, schedule)	Large-scale project (scope, budget, schedule)
Routine application	Non-routine application
Large number of potential contaminants and/or pathways (rapid screening or triggering tool)	Small number of potential contaminants and/or pathways
Initial model development	Model refinement
	Quantification of uncertainty
	Detailed value-of-information analysis (project planning)

- Excessive reliance on findings from animal cancer tests.
- Lack of toxicity data for many components/metabolites of bioaccumulative contaminants (e.g., PCB congeners, some PCB Aroclors, polynuclear aromatic hydrocarbons).
- Lack of a quantitative level of risk that is universally acceptable or unacceptable.
- Poor assessment of noncarcinogenic health effects.
- Uncertainty of risk estimate may be poorly characterized.
- Risks may not be communicated well to the public.
- Environmental inequity not always considered for low-income and minority populations.

Despite the above limitations, risk assessment has been demonstrated as an effective way to prioritize how scarce funding sources should be spent to provide the most benefit to human and ecological health. EPA has adopted a risk assessment approach, as has the Minnesota Pollution Control Agency, for determining priorities and shaping regulatory planning/policy making. As the scientific and policy issues associated with conducting and communicating human health risk assessments are strengthened, the public will benefit from this increased level of confidence in risk estimates.

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Bioaccumulation Models and Applications: Setting Sediment Cleanup Goals in the Great Lakes

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Background

The Great Lakes, much of which is within Region 5 of the U.S. Environmental Protection Agency (USEPA), has significant contamination of hydrophobic organic compounds such as PCBs and dioxins. Because much of this contamination is found in sediments, consideration of bioaccumulation is required when making environmental decisions. Regulatory actions that deal with bioaccumulative hydrophobic organic compounds are generally either waterway management (Clean Water Act section 404) decisions or remedial decisions under statutes such as Superfund, and the Resource Conservation and Recovery Act (RCRA). The focus of this talk will be on the latter category of remedial decision-making.

The presentation will discuss the following:

- The need and precedent for the use of bioaccumulation methods in a regulatory setting. Region 5 evaluation of bioaccumulation methodologies using Great Lakes data (sediments to fish workgroup, or S2F).
- S2F-recommended process of using bioaccumulation methods in combination with risk assessment.
- Case studies and lessons learned.

Summary

Environmental decisions often require that sediment cleanup goals be developed. Issues such as what level is safe to leave behind, which sediments need to be removed for it to be safe, and which remedial option offers the best risk reduction, need to be addressed at the many sites with bioaccumulative hydrophobic organic compounds. In general, there are few sediment regulatory levels that directly address human health. PCBs, a frequent contaminant in the Great Lakes, are regulated under the Toxic Substances Control Act at 50 ppm or greater, which is not, however, considered to be health-based. In some cases there are mandates to return to background levels, and there are also the ecological guidelines developed by the Province of Ontario and the National Oceanic and Atmospheric Administration (NOAA). While these are all useful and relevant, there was a perceived gap in having actions that use these as cleanup goals to be protective of human health.

A Regional workgroup was formed that sought to address the issue of how to address bioaccumulative compounds such as PCBs and dioxins in order to be protective of human health. To promote consistency and clarity on the topic, the workgroup developed Regional guidance based on their deliberations. Because of the concern about PCB bioaccumulation, the scope of the workgroup's efforts was always on hydrophobic organic compounds. Because in general it is not appropriate to extrapolate methods or recommendations to other contaminants that behave differently in the environment (such as metals), the workgroup and document only addressed methods appropriate to hydrophobic organic compounds. The workgroup, named Sediments to Fish, or S2F, contained technical staff from several disciplines and areas of the agency. The goal of the workgroup was to: (1) review relevant literature on bioaccumulation; (2) evaluate various methodologies using Great Lakes data and include a discussion of data issues; and (3) recommend how best to develop cleanup goals or, more accurately, how to best use bioaccumulation methodologies in regulatory actions.

A draft guidance document was completed in 1994. The basic structure of the document is:

- Overview.
- Review and discussion of methods.
- Evaluation and validation of methods (includes discussion of data issues).
- Recommended process for how to use the biota-to-sediment accumulation factor (BSAF) method with risk assessment procedures for specific use in a regulatory setting.

The document was reviewed extensively, including Region 5 states, the State of Washington, USEPA

Headquarters, and NOAA. Overall, the reviews were supportive and recommended refining the document further and adding more information. The document has been revised based on this review, and changes include a new appendix on BSAFs found in the literature (values were checked in the primary reference) and additional case studies. Currently, one of the case studies is being revised to incorporate additional data made available recently. When this revision is made, the document will go through academic peer review.

It is clear that the best way to accurately assess bioaccumulation is to use more complex models utilizing pharmacokinetic parameters. However, for purely practical reasons, they were not considered to be useful at present in a regulatory setting where the data needs of these models would not be met and time would also be insufficient. Three methods, then, were analyzed in detail—BCF, BAF, and BSAF. It is useful to note that in general the BCF and BAF methods relate fish tissue levels to the water column, whereas the BSAF method relates fish contamination to sediment (often the medium where levels need to be set and where contamination more likely lies).

Methods were tested by comparing predicted fish levels (using the various methods) with actual fish data. This was done at four locations—Saginaw, Michigan; Buffalo, New York; Ontario, Canada; and Manistique, Michigan. Data needs were considerable for this process; data were needed to input into all methods (water column, sediment, etc.) and a separate data set was needed to determine site-specific BSAFs. Conclusions from this process were as follows:

- When considering all case studies as a whole, BSAF consistently gave the most reliable estimates of fish tissue concentrations relative to other methods. However, for specific case studies, some of the other methods were slightly more or about as accurate (e.g., BAF—using a measured BAF and dissolved values in the Buffalo case; and BCF—using a measured BCF and total concentration values in the Saginaw case).
- In all of the studies, the modified BAF was the least accurate method.
- How data were used was found to have a significant impact. Accuracy in predicting fish levels depended on whether all data were available, how one decided to handle heterogeneously distributed data, and which bioaccumulation factors were used and how they were developed. From this limited review, it appears that site-specific or field-derived bioaccumulation factors improve the accuracy of the results.
- In general, BCF tended to underestimate fish levels and BAF tended to overestimate fish levels.

Another important conclusion from this method validation chapter was how to appropriately use data. Major recommendations are shown in Table 1. Specific results of the data validation using Great Lakes data are shown in Tables 2 through 4.

The document also discusses how to incorporate human health issues, applying the BSAF method, in conjunction with risk assessment. In this way, one can develop options in a regulatory setting such as setting sediment cleanup goals. The general process for setting cleanup goals is shown in Figure 1. Essentially, using the basic exposure and risk equations from the *Risk Assessment Guidance for Superfund* (RAGS), a target level in fish is set. It is important to note that the process shown assumes that the contaminant of concern (PCBs, or other hydrophobic organic compound) is appropriate, and no other compound that would not be assessed in this way is important in making a cleanup decision. It also assumes that the only pathway of concern is ingestion of contaminated fish. (No other pathway is discussed here, such as ingestion of sediment or surface water.) Note, in addition, that setting the target level in fish using risk assessment requires decisions be made regarding acceptable levels of risk (e.g., 10^{-6}). The target fish level can also be a regulatory level such as a threshold for *setting* a fish advisory (e.g., 0.05 ppm).

The target fish level is put into the BSAF equation and solved for concentration in sediment. Important issues to consider when doing this analysis are: (1) choosing a BSAF, which can be site specific or can be chosen from the literature (variations will arise and can have great impact on results); (2) quality of total organic carbon (TOC) data, which can have a great impact on results as well; (3) using appropriate species of fish and keeping species-specific considerations throughout (lipid values, range, etc.); and (4) applying the risk assessment process carefully and considering many options, such as different exposure scenarios for different fishing behaviors.

This process has been applied in several cases in Region 5. Two that will be discussed are Saginaw and Manistique. Results shown were used in decision-making in Saginaw, as part of a Natural Resource Damage Assessment (NRDA). USEPA looked at current health risks, developed cleanup goals, and assessed how a possible action could address human health risk. The lower Saginaw River is characterized as having widespread lower level (less than 5 ppm on average) sediment contamination at the surface and somewhat more contamination at depth in sediment. Site-specific BSAFs were calculated for two species, walleye and carp. Although a range of BSAFs could be calculated depending on data considerations (averaging of sediment data, etc.), they can be summarized as being 0.3 for walleye and 0.6 for carp. This information is contrasted to Manistique, which is characterized as having localized sediment "hot spots" of PCB contamination, the majority of which is not at the surface. Site-specific BSAFs were also calculated although sediment and fish data were somewhat more limited. Again a range of site-specific BSAFs were considered due to possible data handling choices, but they can be summarized as 0.4 for carp and .07 for walleye. Sediment cleanup goals for these two sites obviously follow from the BSAF differences. Saginaw cleanup goals are much lower than those calculated from Manistique, Michigan. A summary is provided in Figure 2.

Table 1. Data Issues and Recommendations for Sediment to Fish Predictive Methods

Data Issue	Recommendation
Averaging of fish or sediment data (geometric vs. arithmetic mean)	Geometric Mean
Use of sediment organic carbon data	Point normalize
Handling of non-detect data	Use one-half the detection limit
Uneven clustering of sediment samples	Surface area weighting
Calculating site specific BSAF value using sediment and fish data	Temporarily matched sediment and fish data
Type of fish sample	Filet (human health endpoint) Whole fish (ecological endpoint)
Bioaccumulation method	BSAF

Table 2. Comparison of Predicted Fish Tissue Concentrations (from draft S2F document)

All values shown are ppm PCBs	Saginaw (walleye)	Buffalo (carp)
BCF (meas./total)	1.4	0.35
BAF (meas./diss.)	38	0.42
BAF (meas./total)	38	3.5
BSAF	1.4	-0.871
Actual	1.5	2.8

(Using geometric means of relevant data)

Table 3. (from draft S2F document)

Manistique, MI	PREDICTED FISH CONCENTRATIONS IN CARP (in mg/kg)				
WATER-TO-FISH OR SEDIMENT-TO-FISH PREDICTIVE MODEL	SAW SEDIMENT; ARITHMETIC MEANS	SAW SEDIMENT; GEOMETRIC MEANS	ARITHMETIC POINT-NORMALIZED SEDIMENT; ARITHMETIC MEANS	ARITHMETIC MEANS FOR ALL PARAMETERS	GEOMETRIC MEANS FOR ALL PARAMETERS
BAF-modified	390 - 550	380 - 550	520 - 740	230 - 330	79 - 110
TBP (pf=4)	88	86	120	52	18
BSAF (value in parentheses is site-specific BSAF used)	9.0 (0.41)	8.6 (0.40)	13 (0.45)	4.6 (0.35)	2.9 (0.64)
Actual Fish Tissue Concentration	6.5	6.2	6.5	6.5	6.2

[1]SAW = surface area weighted

[2]Values in this column are the same regardless if sediment data are normalized before or after calculating geometric means.

Table 4. (from draft S2F document)

Manistique, MI	PREDICTED FISH CONCENTRATIONS IN WALLEYE (in mg/kg)				
WATER-TO-FISH OR SEDIMENT-TO-FISH PREDICTIVE MODEL	SAW SEDIMENT; ARITHMETIC MEANS	SAW SEDIMENT; GEOMETRIC MEANS	ARITHMETIC POINT-NORMALIZED SEDIMENT; ARITHMETIC MEANS	ARITHMETIC MEANS FOR ALL PARAMETERS	GEOMETRIC MEANS FOR ALL PARAMETERS
BAF-modified	200 - 340	180 - 310	270 - 460	120 - 204	37 - 64
TBP (pf=4)	24	22	33	15	4.6
BSAF (value in parentheses is site-specific BSAF used)	0.99 (0.16)	0.87 (0.16)	1.5 (0.18)	0.51 (0.14)	0.29 (0.25)
Actual Fish Tissue Concentration	0.34	0.25	0.34	0.34	0.25

[1]SAW = surface area weighted

[2]Values in this column are the same regardless if sediment data are normalized before or after calculating geometric means.

1) Set acceptable contaminant level in fish
set risk target level (i.e. 10^{-6} cancer, HI=1)
determine appropriate exposure parameters
for fish ingestion pathway
2) Calculate total organic carbon (TOC) in the area
determine fish exposure area
use most recent sediment data
use surface samples
3) Determine fish which are consumed
Consider:
local fish consumption patterns
at least two species to target
species-specific lipid levels
4) Calculate/Select BSAF
use site-specific data to calculate BSAF
and/or
choose literature value from Appendix E
(match species and site)
5) Calculate sediment cleanup goal

$$Cs = \frac{Cf\ (1) \times TOC\ (2)}{BSAF\ (4) \times Lipid\ (3)}$$

Figure 1. Recommended Methodology for Determining Sediment Cleanup Goals.

Cancer Risk Level	Exposure Assumptions	Site	CUGs for PCBs (ppm)
10 E-6	**sport** fisher eating average of **20 g walleye**/day, **25%** from site	Saginaw	0.06
		Manistique	1
10 E-5	**sport** fisher eating average of **15 g walleye**/day, **25%** from site	Saginaw	0.8
		Manistique	13
10 E-4	**subsistence** fisher eating average of **75 g walleye**/day, **50%** from site	Saginaw	0.8
		Manistique	13

Figure 2. Comparison of Saginaw and Manistique Cleanup Goals (CUGs).

Bioaccumulation Models and Applications:
Setting Sediment Clean Up Goals in the Great Lakes

Overview

- need and precedent for use of bioaccumulation methods in regulatory setting (linking sediment levels with health endpoint)
- Region 5 evaluation of bioaccumulation methodologies
- Region 5 recommended process in use of methodology and in linking with risk assessment
- Case Studies/lessons learned

National Bioaccumulation Conf.
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Bioaccumulation Models and Applications

Need and precedent for bioaccumulation methods for regulatory purposes

- assessing bioaccumulation potential
- setting cleanup goals
- evaluating remedial options
- estimating risks when limited data are available

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Bioaccumulation Models and Applications

Alternative Clean Up Goals/Remedial Targets for Contaminated Sediments

- TSCA (50 ppm for PCBs)
- Ontario Sediment Quality guidelines
- NOAA Sediment Thresholds
- background

☆ gap in having actions protective of human health

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Bioaccumulation Models and Applications

Region 5 Workgroup - Sediments to Fish (S2F)

- multi-media and -division
- focused on hydrophobic organic compounds
  - e.g., PCBs, PCB-like compounds, dioxins
- Guidance document
  -reviewed literature and several methodologies
  -tested methods using Great Lakes data
  -recommended process using BSAF
- draft completed in 1994, reviewed by States and Federal agencies; significant revisions; interim final by end 1996; academic peer review.

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Bioaccumulation Models and Applications

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Bioaccumulation Models and Applications

Key Questions/Issues that Emerged from S2F

- ◇ Which methodology is most accurate in predicting fish tissue concentrations?
- ◇ Appropriately handling data
- ◇ Science Policy Issues

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Bioaccumulation Models and Applications

Methodologies Available to Assess Bioaccumulation of Contaminated Sediments

- BAF
- BCF

> relates fish tissue concentration to water column

- BSAF - relates fish tissue concentration to sediment
- biokinetic models - requires pharmacokinetic parameters

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Bioaccumulation Models and Applications

Testing methodologies

The ability of the various models to accurately predict fish tissue concentrations were tested using actual fish tissue, water column, and sediment data from the following sites:

- Saginaw
- Buffalo
- Ontario
- Manistique

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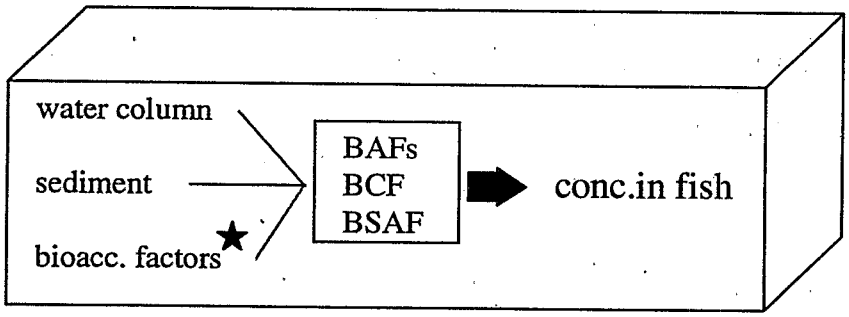
Bioaccumulation Models and Applications

S2F: Testing Methodologies

Data needs were considerable
- -need data to input into all methodologies (BAF, BCF, BSAF) and solve for conc. in fish
- -compare estimated level in fish to actual fish data



★ a separate data set was needed to develop site-specific bioaccumulation factors

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Bioaccumulation Models and Applications

Equations used for Evaluating Bioaccumulation Methods:

$$BCF = \frac{C_f}{C_w}$$

$$BAF = \frac{C_f}{C_w} \qquad \text{Modified BAF} \Rightarrow C_f = \frac{(BCF * FC) * C_s}{f_{oc} * K_{oc}}$$

$$BSAF = \frac{C_f}{C_s}$$

where: C_f = concentration in fish; C_w = concentration in the water column;
C_s = concentration in the sediment; FC=food chain multiplier;
f_{oc} = fraction organic carbon;
K_{oc} = partitioning coefficient (between water and organic carbon)

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Bioaccumulation Models and Applications

Comparison of Predicted Fish Tissue Concentrations

from draft S2F document

all values shown are ppm PCBs	**Saginaw** (walleye)	**Buffalo** (carp)
BCF (meas./total)	**1.4**	.35
BAF (meas./diss.)	38	.42
BAF(meas./total)	38	**3.5**
BSAF	**1.4**	**.87-1.7**
Actual	**1.5**	**2.8.**

(Using geometric means of relevant data)

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Bioaccumulation Models and Applications

Predicted PCB carp tissue concentrations for the Manistique River

(from draft S2F document)

WATER-TO-FISH OR SEDIMENT-TO-FISH PREDICTIVE MODEL	PREDICTED FISH CONCENTRATIONS (in mg/kg)				
	SAW[1] SEDIMENT: ARITHMETIC MEANS	SAW SEDIMENT: GEOMETRIC MEANS	ARITHMETIC POINT-NORMALIZED SEDIMENT: ARITHMETIC MEANS	ARITHMETIC MEANS FOR ALL PARAMETERS	GEOMETRIC[2] MEANS FOR ALL PARAMETERS
BAF-modified	390 - 550	380 - 550	520 - 740	230 - 330	79 - 110
TBP (pf=4)	88	86	120	52	18
BSAF (value in parentheses is site-specific BSAF used)	9.0 (0.41)	8.6 (0.40)	13 (0.45)	4.6 (0.35)	2.9 (0.64)
Actual Fish Tissue Concentration	6.5	6.2	6.5	6.5	6.2

[1] SAW = surface area weighted

[2] Values in this column are the same regardless if sediment data are normalized before or after calculating geometric means.

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Bioaccumulation Models and Applications

Predicted PCB walleye tissue concentrations for the Manistique River

(from draft S2F document)

WATER-TO-FISH OR SEDIMENT-TO-FISH PREDICTIVE MODEL	PREDICTED FISH CONCENTRATIONS (in mg/kg)				
	SAW[1] SEDIMENT: ARITHMETIC MEANS	SAW SEDIMENT: GEOMETRIC MEANS	ARITHMETIC POINT-NORMALIZED SEDIMENT: ARITHMETIC MEANS	ARITHMETIC MEANS FOR ALL PARAMETERS	GEOMETRIC[2] MEANS FOR ALL PARAMETERS
BAF-modified	200 - 340	180 - 310	270 - 460	120 - 204	37 - 64
TBP (pf=4)	24	22	33	15	4.6
BSAF (value in parentheses is site-specific BSAF used)	0.99 (0.16)	0.87 (0.16)	1.5 (0.18)	0.51 (0.14)	0.29 (0.25)
Actual Fish Tissue Concentration	0.34	0.25	0.34	0.34	0.25

[1] SAW = surface area weighted

[2] Values in this column are the same regardless if sediment data are normalized before or after calculating geometric means.

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Bioaccumulation Models and Applications

Data Issues & Recommendations

Data Issue	Recommendation
Averaging data (geometric vs. arithmetic mean)	Geometric Mean
Use of sediment organic carbon data	Point normalize
Non-detect data	Use one-half the detection limit
Uneven clustering of sediment samples	Surface area weighting
Calculating site specific BSAF	Temporally matched sediment and fish data
Type of fish sample	Fillet (human health endpoint) Whole fish (ecological endpoint)
Bioaccumulation method	BSAF

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Bioaccumulation Models and Applications

Bioaccumulation Method Evaluation Conclusions:

- when evaluating all the case studies as a whole, BSAF consistently gave reliable estimates; although other methods in some cases were slightly more or as accurate as BSAF
- estimates using measured values for bioaccumulation factors were more accurate than those using calculated factors
- accuracy in predicting fish levels strongly depended on data quality, availability, and handling of data (e.g., heterogeneously distributed data)
- in most cases: BAF-modified was least accurate, BCF tended to underestimate, and BAF tended to overestimate

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Bioaccumulation Models and Applications

Recommended Methodology for Determining Sediment Cleanup Goals

1) Set acceptable contaminant level in fish

2) Calculate total organic carbon (TOC) in the area

3) Determine fish which are consumed

4) Calculate/Select BSAF

5) Calculate sediment cleanup goal

$$Cs = \frac{Cf\ (1) \times TOC\ (2)}{BSAF\ (4) \times Lipid\ (3)}$$

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Bioaccumulation Models and Applications

Evaluating Remedial Options

- **Consider several options, for example:**

  -dredge sediments greater than 50 ppm, 10 ppm

  -leave sediment in place

  -other options

- **Calculate surface area weighted sediment concentration for each scenario under evaluation**

  [C_s]

- **Determine BSAF for representative fish species (literature, site specific)**

  [BSAF]

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Bioaccumulation Models and Applications

Evaluating Remedial Options, cont'd

- **Use BSAF to model contaminant concentration in fish under each scenario**

$$C_f = \frac{(BSAF) \times (C_s) \times (Lipid)}{(Organic\ Carbon)}$$

- **Use exposure equation to calculate human intake of contaminant**

- **Use toxicological risk information (IRIS) to calculate human health risk under each scenario**

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Bioaccumulation Models and Applications

Determining an Appropriate BSAF Value Species Specific Issues

Issue	Consideration
Foraging Range	Consult with fish biologist to help confirm that the average contaminant concentration in the sediment is representative of the fish's range.
Pisciverous vs. Bottom Feeding	Bottom feeding fish will likely have a higher BSAF and thus should be included to represent a more conservative scenario.
Lipid Content	Fatty fish tend to accumulate more contaminants and thus inclusion will add a more conservative scenario.
Presence of Species at Site	Choose a species endpoint that is actually found at the site.
Consumption of Species	Choose a species which is being consumed by the local population.

Bioaccumulation Models and Applications

Critical Science Policy Decisions

- Choice of bioaccumulation model
- Choice of fish species (e.g. pelagic, bottom-feeding)
- Choice of BSAF value
- Determining appropriate sediment concentration term
- Defining fish exposure area/site boundary

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Bioaccumulation Models and Applications

Critical Science Policy Decisions, cont'd

- Handling of "non-detect" data
- Choice of fish consumption rate
- Determination of endpoints of concern
- Accurately estimating TOC

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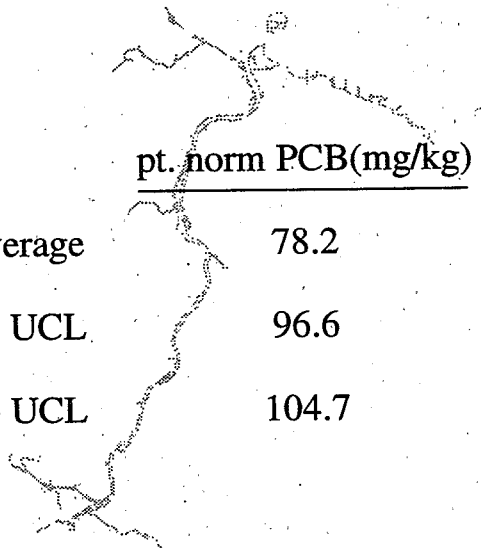
USEPA Region 5

Current Potential Health Risks (MDEQ 1986-1993)

Species	Concentration		Cancer Risk Range		HQ Range	
walleye	0.43		3.04E-05	2.64E-03	0.20	17.11
carp	3.1		2.19E-04	1.90E-02	1.42	123.37
bass	0.21		1.46E-05	1.27E-03	0.10	8.24
whitefish	0.04		2.94E-06	2.55E-04	0.02	1.65
perch	0.13		9.16E-06	7.94E-04	0.06	5.16
pike	0.15		1.07E-05	9.25E-04	0.07	6.01
trout	0.08		5.64E-06	4.89E-04	0.04	3.18

USEPA Region 5

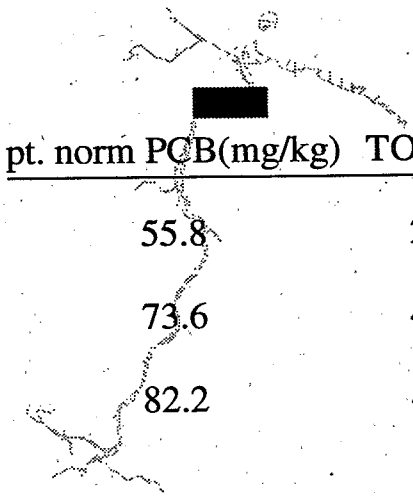
22-Mile Surface- Present



	pt. norm PCB(mg/kg)
Average	78.2
90% UCL	96.6
95% UCL	104.7

USEPA Region 5

22-Mile Surface - Post Dredge



	pt. norm PCB(mg/kg)	TOC(%)
Average	55.8	2.3
90% UCL	73.6	4.1
95% UCL	82.2	4.6

USEPA Region 5

Calculating Risk-Based Cleanup Goals

- Set Target Fish Tissue Level

	Acceptable Risk Level	1.00E-06	1.00E-06		Haz. Index	1	1
	cancer slope factor	7.7	7.7		Ref. Dose	2.00E-05	2.00E-05
Body weight (kg)		70	70			70	70
Averaging time (days)		25550	25550			10950	10950
Ingestion rate (kg/day)		0.02	0.13			0.02	0.13
Fraction ingestion (%)		0.25	1			0.25	1
Absorption (%)		1	1			1	1
Exp. frequency (days/year)		365	365			365	365
Exposure duration (years)		9	30			30	30
Concentration in Fish		0.0141	0.0002			0.28	0.0108

USEPA Region 5

Calculating Risk-Based Cleanup Goals

Input Target Fish Levels into BSAF=>CUGs

Sediment Cleanup Goal ->			CS=(toc x CF) / (BSAF x lipid)		
Cancer (E-6)					
sport	CF	toc	BSAF	lipid	CS
walleye	0.01414141	2.3	0.2813299	1.98	0.0583901
carp	0.01414141	2.3	0.5319693	12.2	0.0050116
subsis.	CF	toc	BSAF	lipid	CS
walleye	0.0001632	2.3	0.2813299	1.98	0.0006737
carp	0.0001632	2.3	0.5319693	12.2	5.783E-05

Noncancer (HI=1)					
sport	CF	toc	BSAF	lipid	CS
walleye	0.28	2.3	0.2813299	1.96	1.1679221
carp	0.28	2.3	0.5319693	1.98	0.6114122
subsis.	CF	toc	BSAF	lipid	CS
walleye	0.0107692	2.3	0.2813299	1.98	0.0444663
carp	0.0107692	2.3	0.5319693	1.98	0.0235159

USEPA Region 5

Post-Remediation Health Risks

- Input Sediment Levels into BSAF=>Resulting Levels in Fish

range walleye	CS	toc	lipid	BSAF	CF	
	0.9	2.3	1.96	0.28	0.2157678	
	1.3	2.3	1.96	0.28	0.1558323	
	55.8	1	0.0196	0.28	0.3076849	(normal.)
	82.2	1	0.0196	0.28	0.4532563	(normal.)
range carp	CS	toc	lipid	BSAF	CF	
	0.9	2.3	12.2	0.53	2.5395752	
	1.3	2.3	12.2	0.53	1.8341377	
	55.8	1	0.122	0.53	3.6214343	(normal.)
	82.2	1	0.122	0.53	5.334801	(normal.)

USEPA Region 5

Post-Remediation Health Risks

Define Exposure Scenarios and Calculate Risk							
walleye	Cancer	low	high		N/C	low	high
	CF	0.307685	0.453256		CF	0.307685	0.453256
	BW	70	70		BW	70	70
	AT	25550	25550		AT	25550	25550
	IR	0.02	0.13		IR	0.02	0.13
	FI	0.25	1		FI	0.25	1
	AB	1	1		AB	1	1
	EF	365	365		EF	365	365
	ED	9	30		ED	9	30
	slope	7.7	7.7		RfD	2.00E-05	2.00E-05
	RISK	2.18E-05	2.78E-03		HQ	0.141284	18.03775
carp	Cancer	low	high		N/C	low	high
	CF	3.621434	5.334801		CF	3.621434	5.334801
	BW	70	70		BW	70	70
	AT	25550	25550		AT	25550	25550
	IR	0.02	0.13		IR	0.02	0.13
	FI	0.25	1		FI	0.25	1
	AB	1	1		AB	1	1
	EF	365	365		EF	365	365
	ED	9	30		ED	9	30
	slope	7.7	7.7		RfD	0.00002	0.00002
	RISK	0.000256	0.032695		HQ	1.662903	212.3033

Bioaccumulation Models and Applications

Exposure Assumptions used at Manistique

Variable	Value	
-ingestion rate (kg/day)	.015 - .054 .075 - .130	-recreation "average" - "high end" -subsistence "average" - "high end"
-fraction ingested from area	25 % 50% 100%	-recreation "average" -recr. "high end" & subsis. "avg." -subsistence "high end"
-exposure frequency (days/year)	365	(IR's are daily rates)
-exposure duration (years)	9 30	"average" scenarios "high end" scenarios
-body weight (kg)	70	
-averaging time (days)	3285 10950 25550	"average" noncancer "high-end" noncancer cancer

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Bioaccumulation Models and Applications

Manistique Sediment Cleanup Levels Summary (PCBs)

Cancer	"Average" Scenario risk	"Average" Scenario target sediment level (ppm)	"High End" Scenario risk	"High End" Scenario target sediment level (ppm)
Recreational fishing	10^{-6} 10^{-4}	1.3 130	10^{-6} 10^{-4}	0.0096 0.96
Subsistence fishing	10^{-6} 10^{-4}	0.13 13	10^{-6} 10^{-4}	0.002 0.2

Noncancer (Immuno-toxicity and Reproductive Effects) *Haz. Index = 1*	**"Average" Scenario** target sediment level (ppm)	**"High End" Scenario** target sediment level (ppm)
Recreational fishing	25	0.63
Subsistence fishing	2.5	0.13

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USEPA Region 5

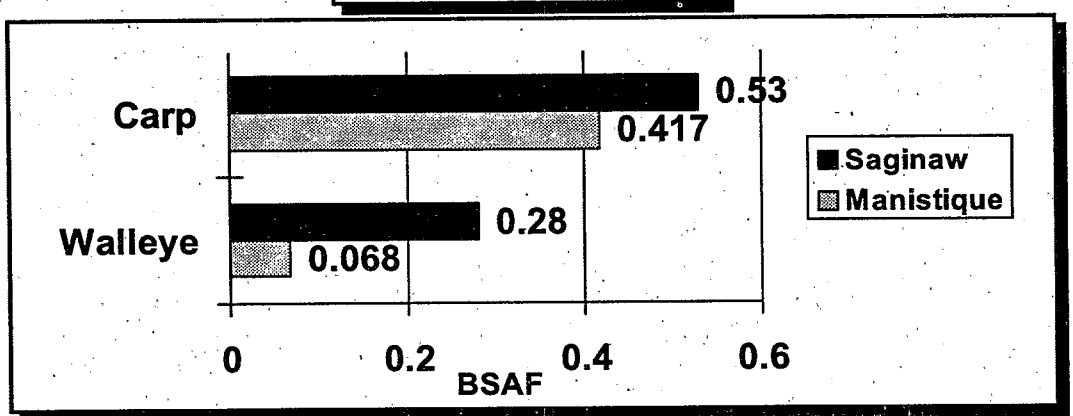
Comparison of Saginaw and Manistique Cleanup Goals (CUGs)

Cancer Risk Level	Exposure Assumptions	Site	CUGs for PCBs (ppm)
10 E-6	**sport** fisher eating average of **20 g walleye**/day, **25%** from site	**Saginaw**	**0.06**
		Manistique	**1**
10 E-5	**sport** fisher eating average of **15 g walleye**/day, **25%** from site	**Saginaw**	**0.8**
		Manistique	**13**
10 E-4	**subsistence** fisher eating average of **75 g walleye**/day, **50%** from site	**Saginaw**	**0.8**
		Manistique	**13**

USEPA Region 5

Biota To Sediment Accumulation Factors (BSAFs) in Saginaw and Manistique

Carp	Saginaw	41.6/78.2
	Manistique	102.2/245
Walleye	Saginaw	22/78.2
	Manistique	16.7/245



Bioaccumulation Models and Applications

Conclusions:

-bioaccumulation models applied in a regulatory setting, with careful consideration, appear to provide reasonably accurate results and allow actions to consider human health
[note: models evaluated for hydrophobic organic compounds only]

-data quality and quantity clearly are critical to usability and accuracy

-important considerations include:

- -accurately estimating: TOC, fish species differences (% lipid, range, etc.), concentrations of contaminant at the surface that fish are/were exposed to, and bioaccumulation factors;
- -fish ingestion rates,
- -setting acceptable risk levels (cancer vs noncancer)
- -critically and clearly communicating limitations, assumptions, and uncertainties.

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Use of Human Health- and Ecological-Based Goals in Developing a Whole River Sediment Strategy: Fox River, Wisconsin

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The Wisconsin Department of Natural Resources (WDNR) has been working with the Fox River Coalition (FRC), a planning and implementation group composed of local, state, and federal partners, to develop a whole river sediment strategy. It was evident early in the process that numeric criteria alone would not suffice in developing a cost-effective whole river sediment strategy. Numeric criteria alone cannot address issues regarding local benefits of remediation in numerous river reaches or evaluate benefits of reducing PCB transport to Green Bay in lieu of local fish tissue reductions. Other techniques that could factor in the size of the existing problem and unique features of the Fox River Basin were needed.

This presentation reviews one alternative approach. The approach relies on the tools and data generated during the Green Bay Mass Balance Study (GBMBS) to help identify and prioritize remediation areas and attempts to quantify the benefits of remediation in terms of reduced PCB transport to Green Bay and reductions in fish tissue concentrations. To date, efforts have focused on developing a basic set of remediation scenarios that illustrate the environmental benefit and implementation considerations of varying levels of sediment remediation. The next step will be to develop preliminary costs for these scenarios. With "ballpark" costs linked to the associated environmental benefits and implementation issues, these scenarios will help the FRC and WDNR develop a whole river sediment strategy. The strategy will strive for cost-effective progress toward achieving established water quality goals and elimination of fish consumption advisories.

The Fox River and Green Bay were studied as part of the 1989 Green Bay Mass Balance Study (GBMBS) (Beltran, 1992). The study focused on the transport and fate of PCBs from the outlet of Lake Winnebago through the entire Lower Fox River and into Green Bay, as well as the accumulation of PCBs in the aquatic food web of Green Bay and the Fox River downstream of DePere. One significant conclusion of the GBMBS is that the source of essentially all (>99 percent) of the PCBs transported by the river originates from the river sediments. Further, a 1989 inventory of sediment PCBs estimates the Fox River sediments contain 4,000 kg (8,800 lbs) and 26,000 kg (57,200 lbs) of PCBs upstream (32 miles) and downstream of the DePere dam (7 miles), respectively.

Sediment remedial action scenarios for the Fox River were simulated by using the WASP5 model (Water Quality Analysis Program) for the Fox River upstream of DePere (WDNR, 1995) and the IPX (In-Place Pollutant eXport) model for downstream of DePere (Velleux et al., 1996). These water quality models developed during the GBMBS were useful tools for evaluating how PCBs moved through the Fox River/Green Bay System during 1989. As demonstrated through a post-auditing procedure, the Fox River water quality models can predict PCB concentrations to within 20 percent to 30 percent of observed PCB concentrations. The post-audit results indicate that the Fox River models are excellent tools for evaluating the impact and effectiveness of proposed sediment remediation efforts.

With the intent of making model simulations realistically represent implementation issues, two assumptions were made. The post-remediation residual PCB concentration used for the simulations was set to 2.5 ppm for the uppermost sediment layer. Also, to simplify the simulation of multiple remediation scenarios, all remediation was assumed to occur at one point in time—July 1, 2000. July 1, 2000, was chosen to represent the midpoint of a 5-year period from early 1998 through the end of 2002.

The Fox River PCB transport models were used to predict PCB concentrations in fish for different reaches of the river as well as PCB mass transported over the

DePere dam and to Green Bay from January 1, 1996, to December 31, 2020. Both models predict surface sediment PCB concentrations and PCB mass transport over time. Predicted surface sediment PCB concentrations were used to calculate PCB concentrations in fish tissue using a simple model, biota-sediment accumulation factor (BSAF) (Di Toro et al., 1991). The BSAF is a measure of site-specific bioaccumulation potential of fish from exposure to contaminated sediments. The model is relatively simple and describes bioaccumulation based on the lipid (fat) content in fish and the amount of contamination and organic carbon content of the sediment. A site-specific BSAF can be calculated as:

$$BSAF = (C_f/f_l) / (C_s/f_{oc}) \qquad (1)$$

where C_f is the pollutant concentration in fish, f_l is the fraction of lipid content in fish, C_s is the pollutant concentration in sediment, and f_{oc} is fraction of organic carbon in sediment. For ease of evaluating the remedial scenarios, the BSAF was applied instantaneously to simulated surface sediment PCB concentrations to simulate fish tissue concentrations over the 25-year model simulation. The fraction of lipid in fish and the fraction of organic carbon in sediment were assumed to remain equal to preremediation conditions.

Site Selections

A nonparametric statistical model, based on fuzzy set theory, was employed to prioritize contaminated sediment sites both upstream of DePere and in the 7 river miles downstream of DePere. This fuzzy set analysis provides a systematic technique for comparing a set of alternatives and identifying more preferable ones based on multiple decision criteria or factors (Table 1).

Table 1. Variables used for fuzzy set ranking of contaminated sediment sites.

PCB mass	Bioavailability Index (OC-normalized PCB in top layer)
PCB mass/area	PCB mass remaining—25-yr "no action" scenario
PCB mass/volume	PCB mass delivery during Mass Balance Study year
Mercury concern	PCB mass delivery under a modeled 100-yr storm event
River position	PCB mass delivery—25-yr "no action" scenario

In general, PCB mass delivery under a simulated 25-yr "no action" scenario and bioavailability were the most important variables while river position and mercury were the variables of least importance.

The prioritizing of sediment sites was conducted separately for upstream and downstream of DePere based on the individual model's configuration. Upstream of DePere, the sites were prioritized by river segment. The IPX model used downstream of DePere artificially divides the last 7 miles of river into 96 Sediment Management Units (SMU). Site prioritization downstream of DePere was by SMU. Upstream of DePere, the five top-ranked river segments contain Deposits A, POG, C, D/E, N, and EE/GG/HH. Deposits A, C, D/E, and POG are all within the first river reach from the outlet of Lake Winnebago, which is locally known as Little Lake Butte des Morts (LLBdM).

The site rankings were used to assemble four sediment remediation scenarios that attempt to accommodate remediation upstream of DePere while balancing the importance of reducing PCB bioaccumulation into fish tissue downstream of DePere with reductions of PCB transport to Green Bay. The four scenarios are:

- No Action [Figure 1 (1)].
- Deposits A, C, POG upstream of DePere and the 3 top SMUs downstream of DePere [Figure 1 (2)].
- Deposits A, C, POG upstream of DePere and the 17 top SMUs downstream of DePere [Figure 1 (3)].
- Deposits A, D/E, POG, N upstream of DePere and the 50 top SMUs downstream of DePere [Figure 1 (4)].

Endpoints

To provide a base level of communication that should be easily recognized and understood by the majority of the public, changes in risk associated with the three scenarios were expressed in terms of allowable fish consumption rates. The proposed Uniform Great Lakes Sport Fish Consumption Advisory Protocol (GLSFATF, 1993) was selected because it recognized the limitations and inadequacy of the FDA tolerances for marketplace fish and set forth a protocol based on a weight-of-evidence health protection value. The protocol provides a range of consumption advice expressed in consumption terms, which provides adequate protection but also allows people to selectively eat sport fish as often as they wish. The advisory categories for PCBs are listed in Table 2.

Table 2. Proposed uniform Great Lakes sport fish consumption advisory levels.

Consumption Rate (227g/meal)	Tissue PCB concentration
Unrestricted consumption (225 meals/year)	<0.05 ppm
One meal per week	0.05 to 0.22 ppm
One meal per month	0.22 to 0.95 ppm
Six meals per year	0.95 to 1.89 ppm
Do Not Eat	> 1.89 ppm

To incorporate ecological concerns into the evaluation of the three remediation scenarios, fish tissue concentrations that would be protective of fish-consuming birds and mammals were also included. Using methodologies from the Great Lakes Water Quality Initiative (USEPA, 1995), an ecologically protective fish tissue concentration of 0.023 ppm derived from mink data was included in the evaluations of the scenarios.

Typical Model Simulation Results

Figure 1 illustrates the type of results generated by this modeling approach. Results have been limited to single fish species in only two distinct river segments (LLBdM and downstream of DePere) and PCB transport to Green Bay. Similar results can be generated for any parameter listed in Table 1 (except river position) in any particular river reach. Other fish species or tissue types (i.e., whole fish) can be easily substituted for the species analyzed provided there are enough data to estimate the average PCB concentration around 1990 for locations above DePere and around 1995 for below DePere.

Conclusions

General conclusions of the modeling done to date include:

- The problem is large and environmental benefit of sediment remediation is directly proportional to the effort expended.
- Remediation of select "hot spots" upstream of the DePere dam has little influence on transport to Green Bay.
- Remediation upstream of the DePere dam has little influence on fish tissue downstream of DePere.
- Remediation of large areas with low PCB concentrations achieves greater reductions in transport with similar reductions in fish tissues.
- Remediation of areas downstream of DePere has greatest influence on fish tissue concentrations downstream of DePere and PCB transport to Green Bay.
- Remediation decreases the time necessary to achieve a specific fish tissue endpoint.

Next Steps

With the goal of a cost-effective whole river sediment strategy in mind and the environmental benefits of these scenarios in hand, attention will shift toward developing total costs of these scenarios. Critical to this step will be developing unit costs for the remedial techniques most likely to be used. These costs could be applied across the board to each scenario, resulting in "ballpark" costs. Alternatively, each scenario could be reviewed in detail and specific techniques and costs applied to each sediment area or group of areas. Most likely, an iterative combination of these approaches will be attempted. Ultimately, with discussions focused on the cost of achieving specific environmental benefits, a cost-effective whole river strategy can be refined from this basic set of scenarios.

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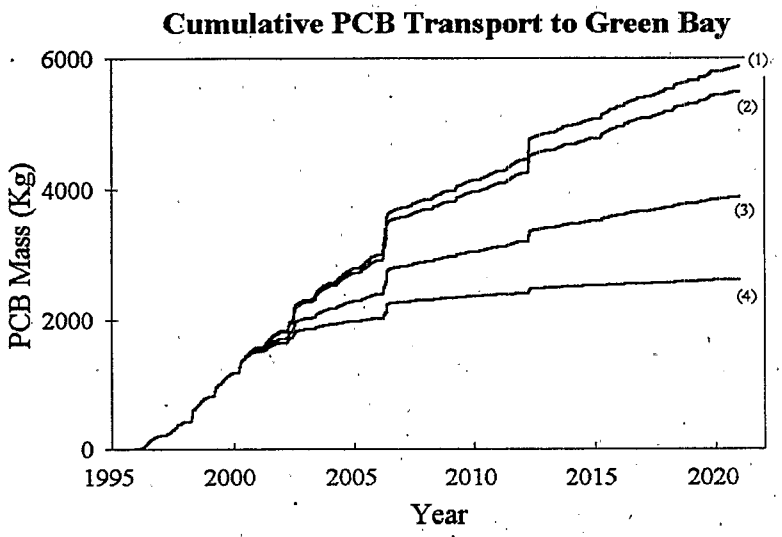

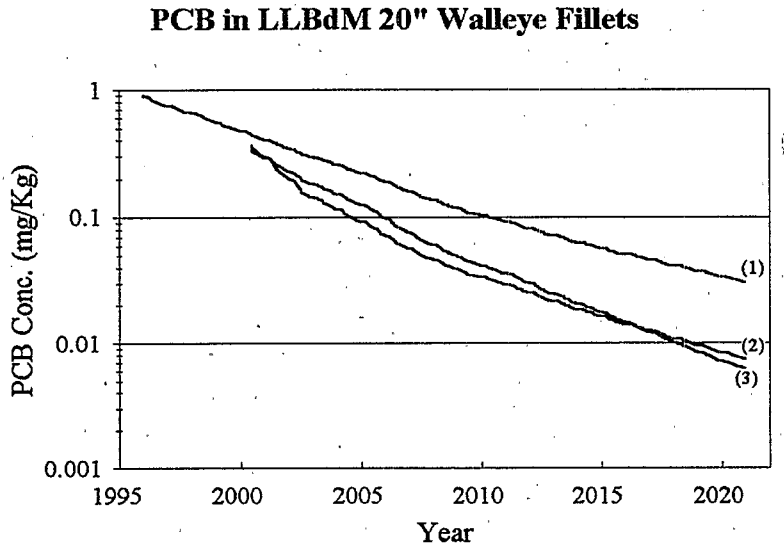

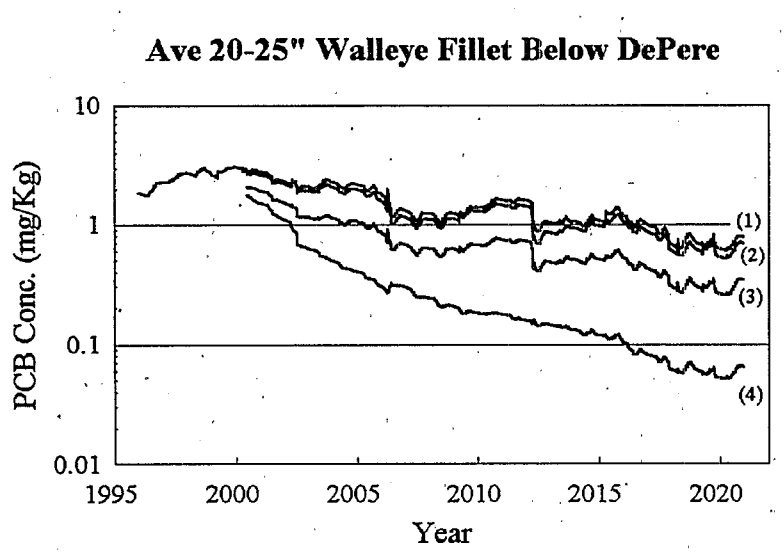

Figure 1. Typical remediation simulation results.

Development of Health-Based Sediment Criteria for Puget Sound

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Problem Statement

Discharge of many industrial, agricultural, and domestic chemicals into Puget Sound has resulted in sediments with elevated levels of pollutants, especially in urban bays and estuaries. Many of these pollutants accumulate in fish and shellfish. As a result, there is concern that contaminated sediments pose a health threat to humans through the consumption of seafood that has bioaccumulated chemicals directly or indirectly from sediments.

To address the problem of contaminated sediments, the Washington State Department of Ecology (Ecology) adopted the Sediment Management Standards (SMS) (Chapter 173-204 WAC) in 1991. The purpose of these standards is to "reduce and ultimately eliminate adverse effects on biological resources and significant human health threats."

Sediment Management Standards (SMS) Background

The SMS currently establish 47 chemical-specific sediment quality standards (SQS), which are designed to protect benthic organisms. No human health-based criteria were available for inclusion at the time of SMS adoption. As a result, standards for the protection of human health from contaminated sediments are being developed on a case-by-case basis. To increase consistency and allow for more timely decision-making, Ecology is developing human health-based sediment quality criteria (HHSQC) that will be incorporated into the current SMS.

Once adopted into the SMS, human health criteria will be used in conjunction with existing ecological criteria to make cleanup, source control, and dredging decisions. The rule presently includes two levels of criteria: (1) the Sediment Quality Standards (SQS), which set a goal of "no adverse impacts," and (2) a higher Regulatory Limit (RL), which allows for "minor adverse impacts." Regulatory decisions are made on a site-specific basis in the range between these two levels. The HHSQC will also include two criteria levels, based on different levels of risk.

Implementation Framework

Ecology's proposed construct for human health criteria relies on a tiered approach. "Tier I" is intended to allow for an initial evaluation to determine if sediment chemical concentrations pose a significant human health risk. If so, additional site-specific analysis would be available under "Tier II" to verify the results of the Tier I analysis and to take into account any uncertainties associated with Tier I values. In addition, Ecology is proposing the use of tissue data as a confirmatory step under Tier II to validate assumptions regarding bioaccumulation potential.

Criteria Calculation/Methodology

The primary human route of exposure to contaminated sediments is via the consumption of contaminated fish and/or shellfish. Human health criteria are developed by applying U. S. EPA's risk assessment methodology to calculate risk from consumption of potentially contaminated fish/shellfish. To quantitatively establish the link between sediment and biota, a biota-sediment accumulation factor (BSAF) is used.

The following formula and input parameters are being proposed to develop HHSQC for carcinogenic compounds:

$$HHSQC = \frac{R * BW * AT * 1000}{CPF * ED * IR * BSAF * FL}$$

where:

R = risk level of 10^{-6} for SQS (a 10^{-5} risk level is proposed for the RL)
BW = adult body weight of 70 kg
AT = averaging time of 75 years
CPF = chemical-specific cancer potency factor as defined by EPA (IRIS)
ED = exposure duration of 30 years
IR = fish ingestion rate of 52 grams/day (based on results of consumption studies of Native American populations in the Puget Sound region (Toy, 1995))
BSAF = chemical-specific biota-sediment accumulation factor
FL = fish lipid of 3%

Technical Development Work

Several technical reports have been completed to support the development of HHSQC. One of these, *Tier I Report: Development of Sediment Quality Criteria for the Protection of Human Health*, completed by the Washington State Department of Health (DOH), investigated technical issues related to the development of health-based sediment criteria (DOH, 1995). The report describes four areas of research: (1) determination of background concentrations; (2) a prioritization of chemicals of concern; (3) development of recommended BSAFs; and (4) fish consumption rates and recommendations.

DOH identified over 200 potential chemicals of concern based on data in Ecology's SEDQUAL database. These chemicals were prioritized into one of three groups (Groups 1, 2, and 3) based on whether they had EPA toxicity values (CPFs or RfDs), the frequency with which they were detected in urban areas, and their ability to bioaccumulate in aquatic biota (as measured by their K_{ow}).

Ecology is focusing its efforts on developing HHSQC for six organic chemicals or chemical groups from the Group 1 chemicals of concern list which are of primary human health concern in Puget Sound and for which our confidence in the toxicity and the BSAF is the highest. These chemicals are known to bioaccumulate, have been found in Puget Sound fish or shellfish, and are likely to present a risk to humans at levels that are lower than current ecologically based sediment criteria. These chemicals are:

- DDT and metabolites
- Hexachlorobenzene
- Hexachlorobutadiene
- HPAHs (high molecular weight polycyclic aromatic hydrocarbons)
- PCBs
- Polychlorinated dibenzodioxins and furans

There are other chemicals of concern for which a scientifically defensible BSAF could not be developed at this time, including several inorganic compounds such as mercury. For these chemicals, Ecology is proposing to develop target tissue levels that can be used to make source control and cleanup decisions in the absence of an HHSQC.

BSAF Development

Initial efforts by DOH focused on the use of a bioenergetics-based equilibrium-partitioning model (Thomann et al., 1992). However, an empirically based approach, relying on information from both the published and gray literature, was developed and recommended. This was due to the availability of empirical data as well as a lack of appropriate input parameters for the model. This approach is described in the final DOH Tier I Report (DOH, 1995).

DOH compiled empirically derived BSAF values from a variety of sources for a range of aquatic species and chemicals. Over 1,200 BSAFs for upper trophic level fish were identified for a set of organic chemicals of concern. These BSAFs were primarily field-derived and represent both marine and freshwater species.

DOH derived BSAFs from these literature values using descriptive statistics based on grouping chemicals by chemical class and log K_{ow}. DOH recommended the use of the 75th percentile BSAF values for criteria development.

To provide additional support for BSAF development, Ecology hired an outside contractor (PTI Environmental Services) to analyze the data set compiled by DOH. PTI used linear and nonlinear multiple regression analysis to investigate the effects of chemical-specific and species-specific characteristics on BSAF values and to estimate BSAFs (PTI Environmental Services, 1995).

Based on the results of the multiple linear regression analysis, separate BSAF calculation equations were developed for different chemical classes, feeding types, and taxonomic groups. A variety of upper confidence limits were calculated for these prediction equations. Regressions were found to be statistically significant for PCBs and dioxins in finfish and for PAHs and PCBs in shellfish. However, the R^2 ranged from 0.70 for dioxins in finfish to 0.058 for PAHs in shellfish.

Using the results of the DOH and PTI analyses, Ecology has developed preliminary HHSQC. BSAFs developed by regression analysis for PCBs and dioxins in finfish, as well as those for PAHs in shellfish, are being applied. BSAFs for other compounds, such as DDT and metabolites, are based on the results of a descriptive statistical analysis (using the 90th upper confidence limit on the mean).

Technical/Policy Implications

Several technical and policy issues have been raised in the process of developing HHSQC. Debate over criteria development has focused primarily on three input parameters: (1) level of risk for cancer-causing chemicals, (2) fish consumption rates, and (3) BSAFs. Each of these issues involves both technical and policy decisions.

Because of the controversial nature of these criteria, Ecology has been working closely with an advisory committee made up of representatives from industry, the environmental community, tribes, ports, and other government agencies, in the development of the criteria. The committee has been providing input on Ecology's technical development work and the related implementation issues.

In addition to questions about the technical/scientific methods being applied, concerns about the cost and liability implications of the HHSQC have been expressed by some potentially affected groups. In an attempt to respond to these concerns, Ecology is currently conducting "case studies." These case studies are a review of existing sediment cleanup site decisions to assess the potential impact of the HHSQC on these (and other) sites. The case studies will also allow the agency to improve the implementation strategy based on lessons learned in the field.

After completion of the case studies, Ecology expects to continue with the rule development process, which includes proposal of a draft rule in mid-1997 and

completion of a cost/benefit analysis (as required by state law). Ecology is looking toward adoption of HHSQC by the end of 1997.

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Development of Health-Based Criteria for Sediments in Puget Sound

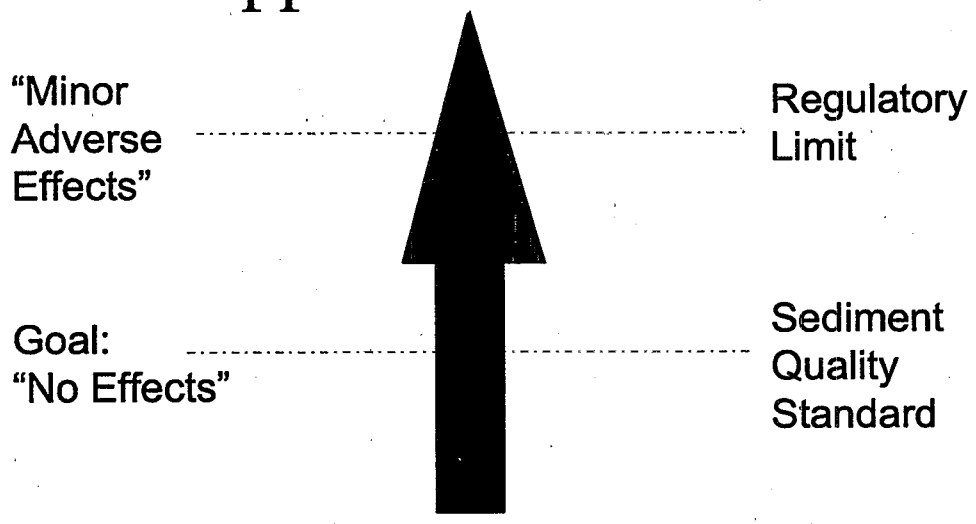
WASHINGTON STATE DEPARTMENT OF ECOLOGY

Laura B. Weiss, M.P.H.

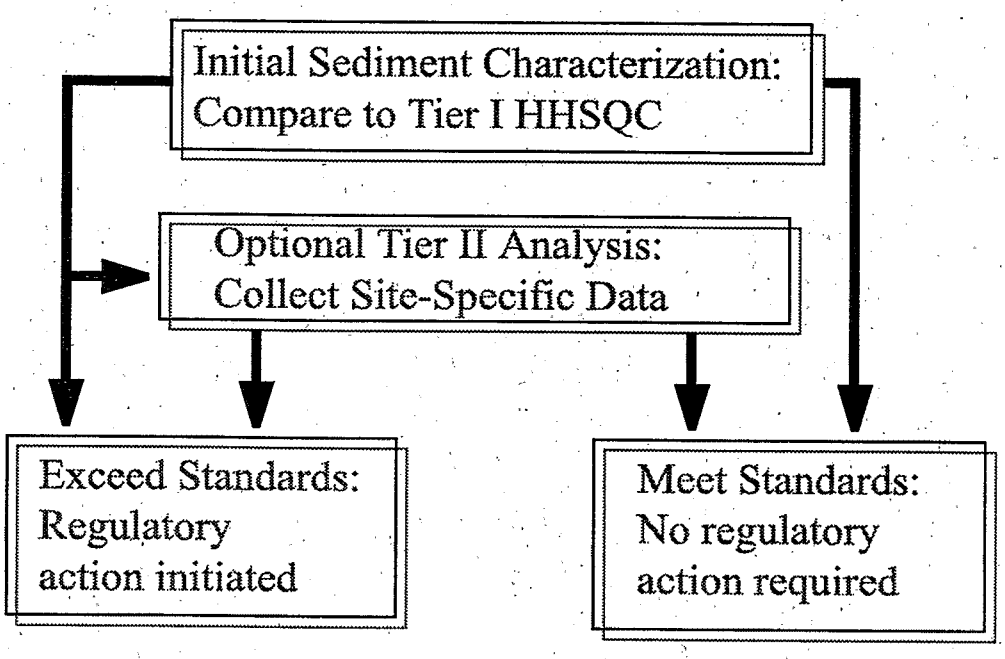
Washington State Department of Ecology

Environmental Review and Sediments Section

Sediment Management Standards Application Model



HHSQC Implementation Framework



Derivation of Health-Based SQC for Carcinogens

$$\text{Sediment Concentration} = \frac{\text{Risk x BW x AT}}{\text{CPF x BSAF x IR x FL x ED}}$$

where:

Risk = 1 in a million (10^{-6}) / 1 in 100,000 (10^{-5})
BW = body weight of 70 kg
AT = lifetime of 75 years
CPF = chemical-specific cancer potency factor (IRIS)
BSAF = chemical-specific biota-sediment accumulation factor
IR = fish ingestion rate of 52 grams/day
FL = fish lipid of 3%
ED = exposure duration of 30 years

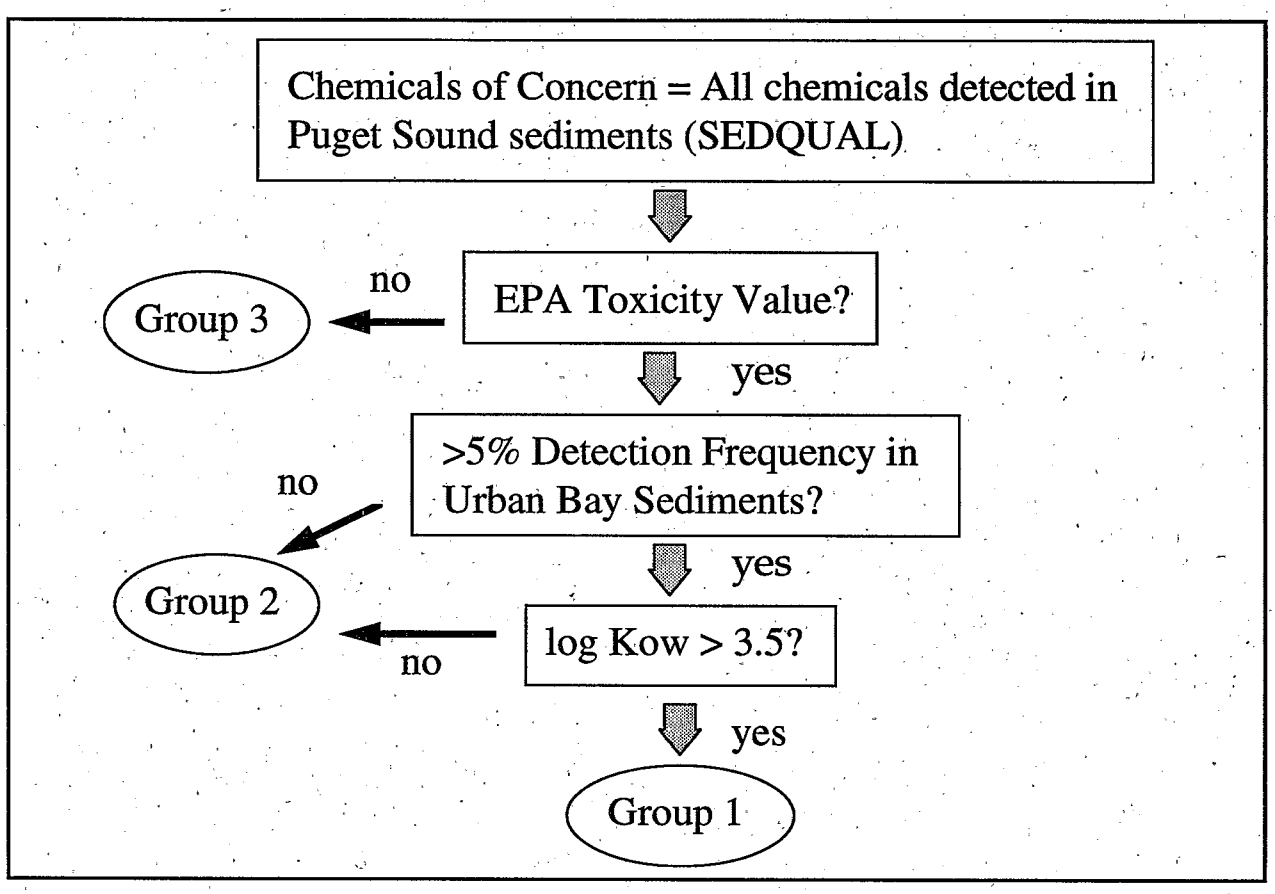
Technical Development Work

- Tier I Report (DOH, 6/95)
- Use of Distributional Analysis (Male, 6/94)
- Fish Tissue Regulatory Options Paper (Male, 9/94)
- Analysis of BSAFs for Metals (PTI, 10/95)
- Analysis of BSAFs for Organics (PTI, 11/95)
- Chemicals of Concern Analysis/Display (PTI, 10/95)
- DOH Tier II Report (DOH, 5/96)
- BSAF Validation for Puget Sound (PTI, 9/96)

Tier I Report (DOH, 1995)

Patrick, McBride, Hardy and LaFlamme

- Areas of Research:
  - Determination of background concentrations
  - Chemicals of concern (3 groups)
  - BSAF development
  - Fish consumption review and recommendations
- SQC calculated based on DOH recommendations



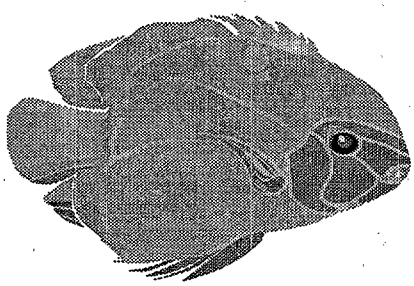
Human Health Chemicals of Concern

- Aldrin
- DDD, DDE, and DDT
- Hexachlorobenzene
- Hexachlorobutadiene
- HPAHs (TEQ)
- PCBs
- Pentachlorophenol
- Polychlorinated dibenzodioxins and furans (TEQ)

*(**Note:** This list includes only Group 1 organic chemicals with the highest bioaccumulation potential. Confidence in calculated BSAF values for these 8 chemicals or chemical groups ranges from high to low.)*

Approaches for BSAF Development

- Thomann Food-Web Model
  - Requires further data collection
  - Validation recommended
- Empirical, Literature-Based Values
  - Regression analysis
  - Descriptive statistics
- Use Fish Tissue Data to Validate



BSAF Analysis for Nonpolar Organics -- DOH

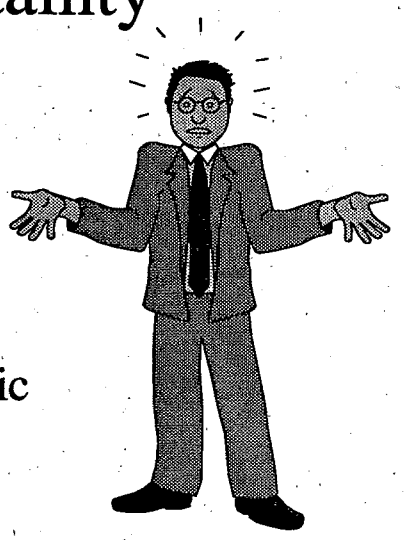
- Compiled data from:
  - Parkerton (1991)
  - COBIAA (1992)
  - EPA GLWQI (1994)
- Mostly field data, from marine and freshwater
- 1,200 BSAF values for upper trophic level fish
  - Grouped by chemical class
    - Found too much variability among chemicals
  - Grouped by Kow and chemical class
    - Accounts for important chemical characteristics

An Alternative BSAF Analysis for Nonpolar Organics

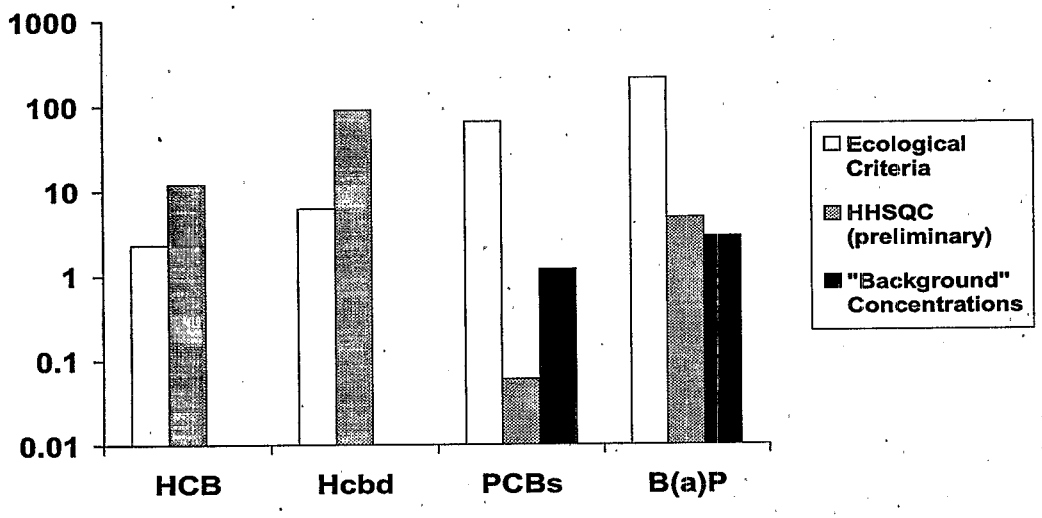
- Alternative statistical analysis of data compiled by DOH (linear and non-linear regression analysis)
- Included a total of 1,591 data points (finfish and shellfish)
- Found significant regressions for PCBs and dioxins in finfish and for PCBs and PAHs in shellfish (variable R^2)
- Results for PCBs in finfish consistent with DOH
- Less consistency for other chemicals of concern

Managing Uncertainty

- Criteria Development
  - Monte Carlo Analysis
- Implementation
  - Two Tiers allow for site-specific data collection and analysis
  - Tissue collection and analysis



HHSQC, Ecological Criteria and Background Concentrations



Issues/Concerns Raised

- Technical/scientific methods
- Consistency between Ecology programs
- Liability and cost associated with cleanup
- Statutory authority
- Implications for source control
- Environmental equity
- Speed of rule development
- Protection of resource and future generations

What Next?

- Case Studies:
  To answer specific questions about implementation and rule impacts
- Draft rule proposal in mid-1997
- Complete cost/benefit analysis
- Rule adoption in late 1997

Development of Bioaccumulation Guidance for Dredged Material Evaluations in EPA Region 2

Alex Lechich
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Introduction

The work described here was done under the auspices of the New York/New Jersey Harbor Dredging Forum, which was convened in the Summer of 1993 by the U.S. Environmental Protection Agency Region 2 (Region 2), the New York District Corps of Engineers (NYDCOE), the New York State Department of Environmental Conservation (NYSDEC) and the State of New Jersey Department of Environmental Protection (NJDEP). The Forum was convened to try to solve the serious problems facing dredging and disposal of dredged material from the Harbor. The problems stem from the implementation of revised dredged material testing procedures, which resulted in many more proposed projects being found not suitable for ocean disposal, and the lack of regional alternatives to ocean disposal. The Forum drew together representatives of all of the local interested parties, including government, industry, shipping, labor, and public environmental interests. In early Forum meetings, eight general areas to be addressed were identified and defined by the participants, and workgroups were assembled which were tasked to address these specific areas.

The Disposal Criteria Workgroup, chaired by Mario Del Vicario, Chief of the Place-Based Protection Branch in Region 2, was tasked to address the perceived "gaps" in criteria for evaluating tests for ocean disposal of dredged material. The workgroup initially identified eleven issues to address. The issue of evaluation of bioaccumulation test results was number one on the list. In prioritizing all eleven issues, however, the workgroup realized that some of the other eleven issues could be resolved more quickly and would address outstanding issues relating to the regional testing manual. After some of these issues were addressed, the Bioaccumulation Subgroup of the Disposal Criteria Workgroup was formed in the Spring of 1994, and began to seriously address bioaccumulation after evaluating the status of other ongoing efforts, including national efforts. The subgroup included a representative each from Region 2, NYDCOE, NYSDEC, NJDEP, Exxon Biomedical Services (as an industry representative), and from a consortium of environmental groups that included the Environmental Defense Fund, the American Littoral Society and Clean Ocean Action.

This discussion will describe the methods that were developed by the subgroup, the current status of the work, including other related work on development of BSAFs from project test results, and the actions that are anticipated in the future.

General Approach

The workgroup decided to develop information in two general areas, risk-based evaluation methods and field background data on benthic invertebrate tissue residues from areas around the ocean disposal site. This approach follows the general guidance on evaluation of bioaccumulation test results in the Green Book (1991), using eight guidance factors for evaluating bioaccumulation results (see *1991 Green Book Bioaccumulation Factors* on page 5-53). Of these eight factors, the workgroup felt there were important information needs for factors four and eight, the toxicological importance of contaminants which exceed reference material and exceedance of concentrations found in organisms living in the vicinity of the disposal site.

The various toxicological measures of a contaminant (i.e., human health, ecological endpoints) are the most important indicators of its potential for adverse effects, while information on background concentrations in benthic organisms is necessary to evaluate the potential for significantly degrading the existing conditions in the area around the disposal site. The toxicological importance of a contaminant needs to be related to the potential risk to an end receptor, which, for dredged material evaluations, is measured at the benthic invertebrate level of the food chain. The workgroup decided to use a method for estimating human health and wildlife risk proposed and developed by John Zambrano of NYSDEC with workgroup involvement. The method is a "standards"-based assessment, as opposed to a site-specific or

case-specific risk assessment. Since the workgroup had been tasked to develop an approach on its own resources, a phased approach was recommended that initially used available information and established methods where possible. Information was readily available in the EPA IRIS (1994, 1995) database and elsewhere to derive risk-based guidance values with the principles that EPA used to derive water quality criteria. This would be compared to other risk-based information and field background tissue data, and combined in an overall strategy for evaluation of test results. As new information or national guidance values became available, the approach would be modified.

Depending on the level of confidence that could be placed on guidance values synthesized from the risk-based and background information, an overall evaluation strategy would probably still be necessary. This is because the level of confidence in guidance values that would be appropriate for passing or failing a proposed ocean disposal project based on exceedance of, for instance, one contaminant (essentially setting "bright line" standards) would be difficult to achieve, given the uncertainties and assumptions in the available methods. A weight-of-evidence type strategy would be necessary if the resulting guidance values do not have this level of confidence. The overall approach, then, would include a human health and ecological (aquatic and wildlife) risk-based component, a field background tissue component and a combined evaluation strategy.

Human Health Approach

The risk-based approach for carcinogens starts with effects data and, after consideration of consumption and other exposure factors, ends with concentration-related levels of risk. In a site-specific approach, actual "project" concentrations are used through hazard and exposure assessments to result in a description of levels of risk for the project. For noncarcinogens, reference doses (RfDs) are used with the same exposure factors as in the cancer risk method. The standards approach used broad assumptions regarding exposure, although the method does include site-specific information that was developed for a representative food chain and wildlife species at potential risk.

The human health model first calculates an acceptable toxicological dose for carcinogens by dividing a selected cancer risk by the cancer potency factor and multiplying by a 70 kg body weight and a 10^3 unit conversion factor (see Human Health Method on page 5-53). For noncarcinogens, an RfD is multiplied by body weight. Toxicological data was obtained from IRIS, and information from the National Toxics Rule (1992) was used for a total PCB reference dose (RfD). An acceptable concentration in seafood is calculated by dividing an acceptable toxicological dose by a seafood consumption factor. This is then converted to an acceptable concentration in benthic invertebrates using a food chain factor, a whole fish-to-filet ratio, and a lipid adjustment (to 2 percent lipid).

EPA has recommended 6.5 g/day as a national average fish consumption rate in its guidance to the states and has utilized that value in the National Toxics Rule (1992). It is recognized that other values may be appropriate to reflect any regional differences; however, there was no effort made to determine a different regional rate. This issue involves extensive consideration of either regional survey data or comparable data, and of average versus higher percentile consumers. The workgroup felt that, for this initial effort, the procedures in the method includes other cancer and noncancer risk components that can account for geographic variables and population extremes.

Trophic Transfer

Initially, the workgroup evaluated food chain multipliers (FCMs) used in the EPA Water Quality Standards Handbook, which used a food chain model developed by Thomann (1989). Scientists from EPA's Office of Research and Development (ORD) recommended that the Gobas model (1993) be used, which was used in the EPA Great Lakes Initiative (GLI). The ORD recommendations came in a meeting held in Region 2 that was chaired by Bob Huggett, EPA Assistant Administrator for ORD, and included researchers from various ORD research laboratories as well as representatives from several EPA Headquarters offices and Regions to support Region 2 in this effort. The Gobas model assumes equilibrium partitioning in benthic organisms, and it was determined that this model reasonably represented field study data for the GLI. To use the Gobas model in our effort, a regional food chain with representative lipid values and organism masses for all trophic levels needed to be developed. A representative coastal ocean demersal food chain for the region was developed in consultation with National Marine Fisheries Service and other experts. It includes information on lipid and mass for each trophic level organism, as well as average values by trophic level for input into the model.

Larry Burkhard of EPA's Mid-Continent Ecology Division (Duluth, MN) performed the model runs. The resulting multipliers are applied on the basis of log K_{ow}, which were obtained from EPA guidance for the GLI (1993). Humans were assumed to eat primarily from the fourth trophic level, so the multipliers from level 2 to 4 were used to calculate the increase in concentration from benthos to human seafood (see Tables 5 and 6 on page 5-55). Since the Gobas model is considered applicable for lipophilic organics that do not metabolize, it is not applicable for metals and PAHs. For cadmium, it was assumed that bioaccumulation occurs through water, and biomagnification does not occur. The trophic transfer factor for methyl mercury was obtained from the GLI (1995). PAHs are metabolized in many higher trophic organisms, but not in some benthic organisms. A trophic transfer factor of one was initially assumed; however, since there is little evidence for the potential of a significant pathway to humans, the workgroup has not as yet decided whether to include PAHs in the human health method. Information from several studies on PAHs in lobsters, including hepatopancreas concentrations, is being evaluated with respect to modeling a potential human pathway. An ingestion rate adjustment, as well as

adjustments according to EPA PAH Toxic Equivalences, will probably be used if the workgroup decides to include the carcinogenic PAHs in the human health method. These adjustments would result in higher human health guidance values than were initially calculated (see Table 7 on page 5-56).

Whole Body/Filet Ratio

A whole body-to-filet ratio was used in the human health method. New York State data indicated a range of 1.2 to 1.5 as being applicable to lipophilic substances. The mid-range of 1.35 was used in the model for all lipophilic substances. A value of one was used for cadmium and methyl mercury. A lipid adjustment to 2 percent lipid was used to reflect the average lipid content of trophic level 2 in the representative demersal food chain.

Wildlife Approach

Initially, the workgroup considered developing wildlife protection values similarly to the human health model, but including adjustments for less stringent effects protection and reflecting the need to protect populations instead of individuals. There was no consensus reached by the workgroup on appropriate adjustments. Also, recommendations by ORD GLI reviewers to address their concerns related to this technique could not, in the opinion of the workgroup, be implemented within the scope of this effort. Therefore, the workgroup decided to calculate wildlife reference doses (WRfD) from the GLI information for three compounds common to the GLI and regional contaminants being addressed, DDT, PCB and mercury. The WRfDs were calculated using the test doses and uncertainty factors from the GLI Tier 1 wildlife criteria equation and criteria documents for protection of wildlife. The WRfDs are conceptually equivalent to the human RfDs. An acceptable toxicological dose is calculated by multiplying the WRfDs by the body weights of species to be protected (see Wildlife Method and related information on pages 5-56 through 5-57).

Representative wildlife species information was developed that included body weight and fish consumption rate at each trophic level. A variety of sources were used to develop the wildlife species information, including discussions with marine observers and scientists and the EPA Wildlife Exposure Factors Handbook.

Based on observations and review of studies in the area, it was determined that the harbor seal would be a good representative mammal potentially at risk, since it has been observed in the dump site vicinity. Other marine mammals and sea turtles were not determined to be at much risk in this scenario, based on information from sightings regarding their regional residence times and feeding behavior. Studies of food consumption and weights of harbor seals in the wild were not available, so information was obtained from the NY Aquarium, which has held several specimens at different times.

Discussions were held with seabird observers, including the Mahomet Observatory and Cape May Observatory in New Jersey, and available information on avian species potentially at risk was compiled. A critical factor for evaluating species at risk was the ability to feed from the demersal food chain at the depths found at the Mud Dump Site (50-80 feet). Cormorants, old squaw duck and red-throated loons were observed in the area and can dive to these depths. Since there was no information available on body weights or food consumption for these species, input parameters used in the GLI for the belted kingfisher were used in the model. Although it is acknowledged that these birds have different body weights and probably different ingestion rates than the belted kingfisher, the ingestion rate to body weight ratios, which are used in the model, are probably within reasonable limits. The herring gull data as used in the GLI was used directly for New York Bight apex gulls to compare with the other wildlife results. A full description and assessment of the above information is available upon request.

The food chain for wildlife was assumed to be similar to the one constructed for humans. Wildlife values for individual species are calculated using the acceptable doses, aquatic food consumption rates and trophic transfer factors. The trophic transfer factors are weighted according to the relative food consumption at each trophic level. As was done in the GLI, the final wildlife value is considered the most stringent of the mammalian and avian results.

Aquatic Risk-Based Approach

An aquatic risk-based approach was also proposed that would estimate risk levels for finfish, which were considered the upper trophic level predators in the aquatic system. In this approach, toxicity information in the EPA AQUIRE (Aquatic Toxicity Information Retrieval) and other databases would be evaluated and the appropriate data compiled and manipulated to yield benthic organism guidance levels (see Aquatic Risk-Based Approach on page 5-58). An order of preference for acceptability of toxicity information was developed, with the key preference being tissue residue-based toxicological data. Uncertainty factors would be obtained from Calabrese and Baldwin (1993). Bioconcentration factors, if necessary, would be obtained from the GLI or other sources, as available. The food chain multipliers developed for the other risk-based approaches would be used here also.

Additional databases are currently being reviewed for tissue residue-based toxicity information, since the amount of information that has been identified thus far is meager for many compounds. It is hoped that information on aquatic risk will be completed in the near future and can be incorporated with any additional information or modifications that are identified for the overall evaluation strategy.

Background Tissue Residue Database

There were benthic tissue contaminant data available from various studies that had been conducted in the vicinity of the Mud Dump Site. This information was preliminarily evaluated and summarized, but it was

determined by the workgroup that a dedicated sampling survey should be conducted to get a better representation of overall field background benthos concentrations. Sampling stations for this effort were located farther from the disposal site than the existing studies, but included areas of potential contamination from other sources. Surveys were conducted in the Spring and Summer of 1995. The data has been evaluated by the workgroup, and the appropriate statistical representation and use of the data will be determined. An interesting finding in this effort is that, although the preliminary information was carefully selected to exclude data from potentially contaminated sediments, the more recent data is generally lower in concentrations for most contaminants. This seems to indicate a general improvement in the benthic conditions of the New York Bight apex since cessation of sewage sludge disposal in June 1992. This is, in fact, the determination of NOAA's National Marine Fisheries Service laboratory in Sandy Hook, New Jersey, from studies conducted in the area since cessation of sludge disposal (NOAA,1995).

Future Actions

An example of the issues that will need to be resolved in completing an overall evaluation strategy is illustrated by the results for PCB and DDT. For DDT, the lower bound (10^{-6}) human health risk level at 20 ppb is within reasonable proximity to the wildlife value and the background range. For PCB, however, an approximation to these other values is only seen at the upper bound risk range of 70 ppb. For many of the contaminants, the risk levels are in the range of the more recent (lower) background levels. For PAHs, as was noted above, if it is decided that PAHs should be included in the human health risk method, application of other factors would raise the preliminary human health risk levels. The upper bound risk levels would then range considerably greater than background values. These varying factors will have to be weighed by the workgroup in completing an evaluation strategy.

Bioaccumulation tests are expensive and time-consuming. Biota-sediment accumulation factors (BSAFs) are being developed from some of the regional testing results. This could enable the estimation of bioaccumulation in benthos from sediment chemical analyses using the theoretical bioaccumulation potential (TBP) relationship, expressed as

$$TBP = AF\ (C_s/\%TOC)\ \%L\ ,$$

where TBP is expressed on a whole-body wet weight basis in the same units of concentration as C_s, and

C_s = concentration of nonpolar organic contaminant in sediment

%TOC = total organic carbon content of sediment expressed as a decimal fraction

%L = organism lipid content expressed as a decimal fraction

A suite of tests that were conducted for federal navigation projects in New York/New Jersey Harbor has produced a large and comparable data set. Preliminary results are promising; calculated BSAFs for 2,3,7,8-TCDD in sandworms were in the range of 0.11 to 0.19, with a standard deviation of 0.03. Pruell et al. (1993) conducted a long-term bioaccumulation study using sediments with high dioxin content (~600 pptr 2,3,7,8-TCDD) and several benthic species, and determined the time to steady state and accumulation factors for PCBs, dioxins and furans. Applying an empirically derived 28-day to steady-state exposure ratio factor for sandworms (~4.0) from this study, the calculated 28-day BSAFs compare closely to the steady-state accumulation factor for sandworms (0.46) derived from the 1993 study.

As additional data for other projects are incorporated for dioxin and other contaminants, the level of confidence in the ability to accurately estimate tissue accumulations from sediment concentrations should be such as to greatly reduce the need to conduct individual bioaccumulation tests.

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NEW YORK/NEW JERSEY HARBOR
DREDGING FORUM

EPA REGION 2

NEW YORK DISTRICT CORPS OF ENGINEERS

NEW YORK STATE DEPARTMENT OF
ENVIRONMENTAL CONSERVATION

STATE OF NEW JERSEY DEPARTMENT OF
ENVIRONMENTAL PROTECTION

REPRESENTATIVES FROM GOVERNMENT, INDUSTRY, PUBLIC
ENVIRONMENTAL INTERESTS, SHIPPING, LABOR

Disposal Criteria Workgroup
(Chair, Mario Del Vicario, EPA Region 2)

Bioaccumulation Subgroup

EPA REGION 2 - Alex Lechich

NYD CORPS OF ENGINEERS - Lisa Rosman/Oksana Yaremko
NEW YORK STATE DEPT. OF ENVIR. CONSERVATION -
John Zambrano

STATE OF NEW JERSEY DEPT. OF ENVIR. PROTECTION -
Ruth Prince, Ph.D.

EXXON BIOMEDICAL SERVICES - Tom Parkerton, Ph.D.
(Industry representative)

RAMAPO COLLEGE - Angela Christini, Ph.D.
(Representing public environmental interests including Environmental
Defense Fund, American Littoral Society and Clean Ocean Action)

CRITERIA WORKGROUP
BIOACCUMULATION STRATEGY DEVELOPMENT

THREE COMPONENTS:

1. Human health & ecological (aquatic and wildlife) - risk-based

2. Field benthic (background) tissue level

3. Combined evaluation strategy

NY DREDGED MATERIAL FORUM CRITERIA WORKGROUP
(BIOACCUMULATION SUBGROUP)

- GENERALLY USED GREEN BOOK PROCEDURES:

  IF STATISTICALLY ABOVE REFERENCE, GO TO 8 FACTORS

  FOCUSED ON COMBINATION OF FACTORS ASSOCIATED WITH TOXICOLOGICAL IMPORTANCE AND TISSUE LEVELS IN VICINITY OF DISPOSAL SITE (BACKGROUND) COMPARISON

- WORKGROUP DECIDED ON "STANDARD" AS COMPARED TO "SITE SPECIFIC" RISK-BASED APPROACH BECAUSE OF TIME AND RESOURCE CONSTRAINTS

  INFORMATION FROM THIS EFFORT CAN BE USED TO LATER "FEED INTO" A SITE SPECIFIC RISK APPROACH

- TOXICOLOGICAL INFORMATION USED IN RISK-BASED APPROACHES FOR

  HUMAN HEALTH (CANCER AND NON-CANCER)

  AQUATIC RESOURCES

  WILDLIFE RESOURCES

- ALL APPROACHES BEGIN WITH CALCULATION OF ACCEPTABLE CONCENTRATION IN FISH, THEN APPLICATION OF ADJUSTMENTS FOR TROPHIC TRANSFER, LIPID AND OTHERS, RESULTING IN BENTHIC INVERTEBRATE TISSUE LEVEL

1991 GREEN BOOK BIOACCUMULATION FACTORS

(1) number of species in which bioaccumulation from the dredged material is statistically greater than bioaccumulation from the reference material

(2) number of contaminants for which bioaccumulation from the dredged material is statistically greater than bioaccumulation from the reference material

(3) magnitude by which bioaccumulation from the dredged material exceeds bioaccumulation from the reference material

(4) toxicological importance of the contaminants whose bioaccumulation from the dredged material exceeds that from the reference material

(5) phylogenetic diversity of the species in which bioaccumulation from the dredged material statistically exceeds bioaccumulation from the reference material

(6) propensity for the contaminants with statistically significant bioaccumulation to biomagnify within aquatic food webs

(7) magnitude of toxicity and number and phylogenetic diversity of species exhibiting greater mortality in the dredged material than in the reference material

(8) magnitude by which contaminants whose bioaccumulation from the dredged material exceeds that from the reference material also exceed the concentrations found in comparable species living in the vicinity of the proposed disposal site.

HUMAN HEALTH METHOD

ACCEPTABLE TOXICOLOGICAL DOSE

1. CANCER - $\frac{\text{CANCER RISK LEVEL}}{\text{CANCER SLOPE FACTOR}} \times \text{BODY WEIGHT}$

2. NONCANCER - (RfD) (BODY WEIGHT)

$$\frac{\text{ACCEPTABLE TOXICOLOGICAL DOSE}}{\text{FISH CONSUMPTION}} = \text{ACCEPTABLE CONCENTRATION IN FISH}$$

$$\frac{\text{(ACCEPT. CONC. IN FISH) (WHOLE BODY/FILET FACTOR)}}{\text{(TROPHIC FACTOR)}} = \text{ACCEPTABLE CONCENTRATION IN BENTHIC INVERTEBRATES}$$

BODY WEIGHT - 70 Kg
FISH CONSUMPTION - 6.5 g/day
WHOLE BODY/FILET RATIO - 1.35
LIPID ADJUSTMENT - to 2%

FOOD CHAIN MODEL

- INITIALLY USED FCMs IN EPA WATER QUALITY STANDARDS (1993) HANDBOOK - THESE USED THOMANN 1989 FOOD CHAIN MODEL
  - DOES NOT INCLUDE BENTHIC COMPARTMENT
  - USED 10% LIPID FOR ALL TROPHIC LEVELS
- BOB HUGGETT (AA, ORD) MEETING HELD ON MARCH 22-23, 1995 WITH ORD SCIENTISTS, HQ OFFICES AND REGIONS IN SUPPORT OF REGION 2 EFFORT
  - RECOMMENDED USE OF GOBAS 1991 MODEL
- GOBAS MODEL
  - ASSUMES EQUILIBRIUM BETWEEN SEDIMENT AND BENTHOS
  - EPA FOUND GOOD CORRELATION FOR EQUILIBRIUM AND USED MODEL IN THE GREAT LAKES INITIATIVE (GLI)
  - CONSIDERED APPLICABLE FOR LIPOPHILIC ORGANICS THAT AREN'T METABOLIZED - WASN'T ASSUMED APPLICABLE FOR METALS OR PAHs
    - FOR CADMIUM, BIOAC THRU WATER, NO BIOAMAG TTF = 1
    - FOR MERCURY, TTF FROM GLI TSD, APPENDIX E
    - FOR PAHs, METABOLIZED IN MANY HIGHER ORGS TTF = 1
- Adjusted results to reflect two percent lipid - as was found for trophic level two (benthos) in the representative food chain

TROPHIC LEVEL (food preferences)	% LIPID (wet wt.)	MASS (gms) (wet wt.)
TL IV		
● fluke (summer flounder) (sand lance, juvenile fish, mysid shrimp)	1.2	500
● hake (mix red, silver, others) (small crabs, mysids, bivalves, polychaetes)	1.1	200
● bluefish (fish, small crabs, mysids, polychaetes, squid)	11	1000
● tautog (mussels, barnicles, mysids, crabs)	3.8	800
● lobster (hepato + muscle) (crustaceans, polychaetes, live and dead fish, mollusks)	6	500
approx. average	4.5	600
● juvenile finfish/sandlance (copepods, mysids, cladocerans, fish eggs, mollusca larvae)	4.2	10
● crustaceans (crabs, shrimp) (small polychaetes & mollusks, deposited and suspended organic matter, smaller crustaceans)	4.5	10
approx. average	4.4	10
TL II		
● polychaetes (Nephtys) (deposit and sediment ingesting or organic matter, carnivores of minute organisms)	2.0	1
● bivalve molluska (Nucula) (deposit and filter feeders-phytoplankton, organic detritus)	3.0	0.1
● crustaceans (Crangon, mysids) (deposit and filter feeders-phytoplankton, organic detritus, carnivores of minute organisms)	1.0	0.1
approx. average	2	0.4

Table 5

TROPHIC TRANSFER FACTORS
VS.
LOG KOW

Log Kow	Level 3 to 2	Level 4 to 2
4.0	1.0	1.0
4.1	1.0	1.1
4.2	1.0	1.1
4.3	1.0	1.1
4.4	1.0	1.1
4.5	1.0	1.2
4.6	1.0	1.2
4.7	1.0	1.3
4.8	1.0	1.3
4.9	1.0	1.4
5.0	1.0	1.5
5.1	1.0	1.5
5.2	1.0	1.6
5.3	1.0	1.8
5.4	1.1	1.9
5.5	1.1	2.0
5.6	1.1	2.1
5.7	1.1	2.3
5.8	1.1	2.4
5.9	1.1	2.6
6.0	1.1	2.7
6.1	1.1	2.8
6.2	1.1	2.9
6.3	1.2	3.0
6.4	1.2	3.0
6.5		

Table 6
Trophic Transfer Factors

Substance	Log Kow	Trophic Transfer Factor	
		Level 4 to 2	Level 3 to 2
aldrin	6.0	2.6	1.1
anthracene-	1	1	
benzo(a)anthracene	-	1	1
benzo(k)fluoranthene	-	1	1
benzofluoranthene, 3,4-	-	1	1
benzo(a)pyrene	-	1	1
cadmium -	1	1	
chlordane5.8	2.3	1.1	
chrysene -	1	1	
DDD	6.1	2.7	1.1
DDE	6.4	2.9	1.2
DDT	6.2	2.8	1.1
dibenz(a,h)anthracene	-	1	1
dieldrin	5.0	1.4	1.0
fluoranthene	-	1	1
fluorene -	1	1	
heptachlor5.0	1.4	1.0	
heptachlor epoxide	4.1	1.1	1.0
inden(1,2,3-cd)pyrene	-	1	1
methyl mercury	-	6.3	1.3
PCBs	6.1	2.7	1.1
pyrene	-	1	1

[1]Factors for trophic level 4 to 2 are used to derive the criteria for human health.

Table 7

PRELIMINARY BENTHIC ORGANISM GUIDANCE VALUES, ug/kg
(Human Health)

Substance	Cancer (10-6)	Noncancer
aldrin	0.3	200
benzo(a)anthracene*	2*	--
benzo(k)fluoranthene*	2*	--
benzofluoranthene,3,4*	2*	--
benzo(a)pyrene*	2*	--
cadmium	--	10,000
chlordane	5	300
chrysene*	2*	--
DDD	20	--
DDE	10	--
DDT	20	1,000
dibenz(a,h)anthracene*	2*	--
dieldrin	0.7	500
heptachlor	2	5,000
heptachlor epoxide	1	200
indeno(1,2,3-cd)pyrene*	2*	--
methyl mercury	--	200
PCBs, Total	0.7	100

***NOTE:** Values for 7 carcinogenic PAHs will be either revised upward (by incorporation of lower ingestion rate - based on Quincy Bay study of lobster hepatopancreas consumption - and application of TEFs), or will not be included in human health assessment.

WILDLIFE METHOD

INITIALLY, CONSIDERED USING ADJUSTMENTS TO HUMAN HEALTH METHOD, FOR LESS STRINGENT OVER-ALL EFFECTS PROTECTION AND REFLECTING NEED TO PROTECT POPULATIONS INSTEAD OF INDIVIDUALS - NO CONCENSUS; IMPRACTICAL

USED EPA GREAT LAKES INITIATIVE (GLI) INFORMATION
FOR 3 COMPOUNDS (DDT, PCB, MERCURY) IN THIS EFFORT

CALCULATED WRfDs WITH TEST DOSES AND UFs FROM GLI
TIER 1 WILDLIFE CRITERIA EQUATION AND CRITERIA DOCUMENTS FOR PROTECTION OF WILDLIFE

Calculated WRfDs (ug/kg-d)

Species	DDT	PCB	Mercury
mammalian	80	30	16
avian	9	200	13

(WrfD)(body weight) = Wildlife Acceptable Dose

Wildlife Acceptable Dose (ug/d)

harbor seal	9600	3600	1920
belted kingfisher	1.4	30	2
herring gull	9.9	220	14.3

WILDLIFE PARAMETERS

Selected Species	Adult Body Weight (Kg)	Aquatic Food Ingestion Rate (F), (kg/day) Troph.Lev.3	Troph.Lev.4
Harbor Seal	120	0	9.6
Belted King Fisher	0.15	0.0672	0
Herring Gull	1.1	0.192	0.0480

Example Computation

Acceptable Doses (AD), divided by the aquatic food consumption rates (F) and the trophic transfer (TT) factors. The trophic transfer factors are weighted according to the relative food consumption at each trophic level. An example computation is presented below for the Herring Gull and DDTr.

$$\text{Species Criterion (Gull, DDTr)} = \frac{AD}{[(F_{TL3})(TT_{3-2})+(F_{TL4})(TT_{4-2})]}$$

$$\text{Species Criterion (Gull, DDTr)} = \frac{9.9}{[(0.192)(1.1)+(0.0480)2.8]}$$

Species Criterion (Gull, DDTr) = 29.2 ug/kg

WILDLIFE GUIDANCE VALUES (ug/kg)

Individual Species

Species	DDTr	Mercury	PCBs
Harbor Seal	357	31.7	134
Belted King Fisher	18	22.2	405
Herring Gull	29.2	26	637

PRELIMINARY
FINAL WILDLIFE VALUES (ug/kg)

DDTr	20
Mercury	20
PCBs, Total	100

DDT and PCB values are based on two percent lipid. Concentrations are based on wet weight.

AQUATIC RISK-BASED APPROACH

1. Using EPA AQUIRE (Aquatic Toxicity Information Retrieval) database, searched for information on BCCs encompassing all finfish species - (Class Osteichthyes - bony fishes)

-also used review chapter in CRC Critical reviews in Aquatic Sciences, entitled "Effects of Environmetnal Pollutants on Early Fish Development"

2. AQUIRE Data: Order of Preference

Chemical Reporting:

1. Tissue Concentrations
2. Water Concentrations

- UFs from Calabrese and Baldwin (1993). Performing Ecological Risk Assessments
- BCFs from GLI Derivation of Proposed HH and Wldlf BCFs for GLI

Effects Reporting:

1. Reproductive Endpoints
   a) Embryonic or Larval Malformation
   b) Egg Production
   c) Hatchability/Survivorship
   d) Embryonic or Larval Mortality
2. Adult Chronic
3. Adult Acute

Endpoints Reporting:

1. NOAEC
2. MATC
3. LOAEC
4. Effects Concentrations
5. EC50, LC50

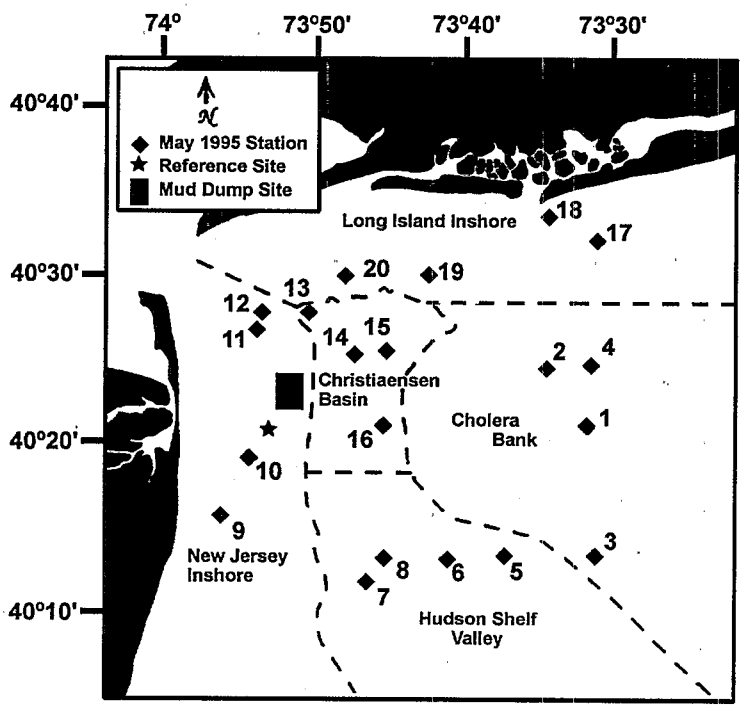

Figure 1. Map showing the five New York Bight Apex Regions and sampling locations for May 1995 Body Burden Survey.

ISSUES TO BE ADDRESSED

- try to reach concensus on appropriate statistical use of background data
- try to reach concensus on contaminant-specific guidance values that include human health, ecological and background values
  - how do they compare?
  - in some cases, roughly comparable @10^{-6} HH carc. risk level
  - for PCBs and PAHs, HH carc. values are comparable only @ 10^{-5} - 10^{-4} risk levels
- will evaluate comments from peer/public review on acceptability of approach and on derivations and conclusions

Biota-Sediment Accumulation Factors From Regional Dredging Projects

- (draft) results of bioaccumulation tests of nine NY/NJ Harbor dredging sites (federal projects) plus a reference site

$$BSAF = \frac{(Tc/L)}{Sc/TOC}$$

- Dioxin/Furan 28-day BSAFs

  TCDD - 0.11 - 0.19 (mean 0.14, SD 0.03)

  TCDF - 0.14 - 0.28 (mean 0.20, SD 0.04)

- Comparable to Pruell et al. 1993 AFs for dioxins/furans 28-day and steady state results
- Validation with other project results

COMPOUND	HUMAN HEALTH		WILDLIFE		FIELD BKGRD
	10-4	10-6	1	2	(rough approx.)
PCB	70	0.7	300	110	50 - 150
DDT	2000	20	20	260	10 - 20

Wildlife:

1. Sub-group effort
2. New York State Dept. Of Envir. Conser. - Niagara River Biota Contamination Project: Fish Flesh Criteria for Piscivorous Wildlife, Newell, et al., July 1987

Day Two: September 12, 1996

Session Five: Questions and Answers

After each session, there was an opportunity for questions and answers and group discussions pertaining to the speakers' presentations.

Q (John Connolly, HydroQual, Inc.): Amy, in your work with BSAFs, are you assuming that if you cut the sediment concentration in half, the fish concentration will drop by a factor of 2? How do you know that is true?

Amy Pelka, U.S. EPA Region 5:

It probably is not true. It depends on where the site is. The BSAFs are not linear at one site that has 20,000 parts per million PCBs. Here, as the concentration increases, it does not go up with the BSAFs. Similarly, as the concentration decreases, it does not go down by half. Looking at this further, we decided that it is very important to consider when and where you are taking your sediment sample. This sample represents what you think presently and in the future or what you are trying to create theoretically. I am not sure whether we can do that, because we do not always have the data. We are seeing that there is some kind of curve where the concentration tops out and the BSAF is not going to increase or decrease. The way to deal with this issue presently is by deciding whether the samples represent what we think are steady-state levels. Sometimes we have looked at caged fish data from upstream to see if that represents what the bioaccumulation might look like at these low levels versus the high levels. You do not want to do BASFs on a hotspot, for example. You would want to look at something lower.

Q (John Connolly): My big concern is that it is not a conservative assumption. For example, suppose there are sources you do not know about, as in Puget Sound. We have contaminated fish, and we say we are going to go in and remove sediment and expect contaminant levels in the fish to decrease. But what we do not realize is that some residual sources are contributing to the problem. We remove sediment and it does not get better, because we did not address those sources. If we are talking about particular issues that may have significant economic costs (for example, dredging that costs $50 million or $100 million), is it worth spending $500,000 or $1 million to find out that cutting the sediment concentration is going to be protective of the ecosystem and human health, before you make the decision based on the sort of routine procedure that has been outlined?

Amy Pelka:

I agree that you should be clear about where your sources are before you go about trying to attack a problem and setting cleanup goals. And you are right that it is not a conservative assumption. I do not know if it was clear, but for Saginaw there were very low levels of PCBs on the surface. I showed you only normalized numbers, but, on average, the Saginaw levels are only 2 parts per million. The fish are a lot more contaminated there than they were in Manistique, which was much more contaminated. The curve for this example actually flattens down at the bottom as well as at the top. I am trying to show that, if you get a lot of contamination, it does not mean there will be more bioaccumulation at the bottom. It never really goes away. You need to see where you are and look at the system to decide what you are going to do. Sometimes that is a question of whether just modeling should be done. There are a lot of Superfund sites in Region 5 where this is a problem, and you may not have the option of modeling to come up with an answer due to the high cost. It may not reduce enough of the uncertainty. So you could spend nothing, if you consider me free, or you could spend $12 million like they did to model Green Bay. In some cases that make sense and, in other cases, it does not. It really depends on the circumstances.

Q (John Connolly): I think you do have to approach it on a case-by-case basis. I have one last question. You mentioned offhandedly that mass removal was a good thing. I am not sure why that is true.

Amy Pelka:

I wanted to make it clear that risk assessment is one way to look at whether or not a site has bioaccumulation at levels of concern, but it is not the only way to assess

whether or not you have a problem. In some cases, you will not see the differences in a risk assessment. For example, with Saginaw, the levels are not going to decrease in the surface concentrations in sediment and the levels in fish populations probably will not decrease significantly either. From a risk assessment perspective, this is because the uncertainty is too large when estimating the different ingestion rates associated with risk assessments. Therefore, it is appropriate to look at mass as well. Maybe it is appropriate for EPA or a state agency to remove PCBs because they do not want loadings to the Great Lakes. I do not want people to think risk assessment is the only approach, because sometimes mass can be a reasonable approach.

Q (John Connolly): For your case in Saginaw, consider an example where you may have 2 parts per million on the surface and 100 parts per million at depth. If you go in and dredge, you may end up with a residual concentration of 5 parts per million. In this example, you may have made the problem worse.

Amy Pelka:

The goal of the removal project is to remove the surface material. But there are cases where dredging probably will increase the surface concentrations, and how do we mitigate that? The problem with Saginaw was that it had 22 miles of contamination at 2 parts per million. There would not be enough money to pay EPA Region 5 to bring it down to 0.5 parts per million. That was the point. I agree with you. We worry about that a lot. Sometimes with the dredging, what you actually think you are going to get as a result will often be higher than what you started with. A big issue is whether that is good.

Robert Paulson, Wisconsin Department of Natural Resources:

It can also certainly affect transport into the future with whatever residual you leave behind, if the residual is at a level that will work itself up into the system a little bit more. You can reduce significantly the transport into the future by dealing with just mass. This is just one variable.

John Connolly:

I agree, but I think that is really on a case-by-case basis. If your contaminants are high 2 feet down, they may really be locked away forever. But you have to evaluate that on a site-by-site basis.

Q (Robert Paulson): Are they really all locked away forever? This is the point we are questioning.

Amy Pelka:

There is another set of modeling you can do. With the risk assessment, you can try to determine whether or not certain storm events are going to reveal that and what the bioaccumulation will be after that. So, you can torture yourself with that consideration, too, if you want to.

Q (Ed Pfau, Ohio EPA): My question is directed to Laura Weiss. You had mentioned that, as part of your uncertainty analysis, you had gone back and revisited the numbers using uncertainty analysis with the stochastic model. You alluded to the fact that it was surprising that your point values came out at about the 80th percentile from the final stochastic analysis. What particular inputs in the equation in the algorithm did you use distributions for, and were the distributions from the same original database that the point values were derived from?

Laura Weiss, Washington Department of Ecology:

We distributed as many input parameters as we could that were appropriate to be distributed. Some parameters were fairly simple and obvious like body weight and exposure duration. We also distributed fish consumption rate, fish lipid, and BSAFs. We kept risk level constant and evaluated one of the more controversial factors, the cancer potency factors (CPFs). A consultant was hired to evaluate CPFs. This is something EPA has been grappling with over time as Monte Carlo analysis has become more popular. The results of this analysis showed that there was too much uncertainty and none of the approaches that were evaluated were really defensible. So, ultimately the toxicity factor was held constant as well.

Q (Ed Pfau): Was the database for the point values basically the same from which the distributions were drawn?

Laura Weiss:

Yes, it basically was. We relied on local data as much as possible, especially for parameters like fish consumption rate.

Q (Ed Pfau): You talked about using various methodologies to determine an effective concentration both in fish tissue and for sediments. You also talked about the surface area weighted average and use of geometric means. Is it appropriate at some point in the future, if the stochastic approach becomes more viable, that use of the distribution with its appropriate shape of distribution would be a reasonable substitute for either one of those methods of trying to average out concentrations in fish tissue or in sediment concentrations?

Laura Weiss:

On a site-specific basis, it is a potential option we might want to look at. There might be a place for it in the Tier 2 analysis. However, I think it affects the other programs in our agency as far as how they deal with Monte Carlo analysis, particularly how the results are analyzed and whether the input parameters are appropriate. We will need to develop guidance for its use.

Amy Pelka:

You want to make sure the set of data that you have is representative of what you want to look at, which may be the sediment concentration at the surface. If you have varying spacial intensities where there are several samples around the rot spots, your distribution still is not representative of the true distribution, even if you have run a Monte Carlo analysis. It all depends on how the distribution was done. A distribution, or Monte Carlo analysis, is not inherently evil. It just depends on how it is used. It can be useful.

Q (Ed Pfau): So the surface area weighting is to remove the bias in sampling, and that would not be something that would be addressed in a Monte Carlo distribution? Is that correct?

Amy Pelka:

I am not sure that just because you have a surface area weighting means that you cannot use a Monte Carlo distribution. I do not think Monte Carlo takes away that bias in sampling. If it shows you the mean and if you are still using a distribution that is based on a small subset of the data, you are still going to have the same problem.

Q (Malcolm Watts, Zeneca, Inc.): I am pleased that one person, Mr. Paulson, did at least mention cost in passing, and I am very upset that the cost issues have largely been ignored. I am part of the handful of people here from the regulated community, and we see these models being generated with inordinate costs possibly associated with the results. This is very disturbing. The methodology of the models seems to be quite good, except when conservative factors are introduced which build upon one another to give inordinately cautious results. In particular, with those sites that I have been involved with, I have found the science base is not worth anything. The data that Ms. Pelka referred to as being very difficult to find and evaluate, I believe, is characteristic of the basis on which the decisions are made. This produces results which cause high costs. I propose to you to use the models and the data you have, and see what the results are. If they are easy to work with, like no action or modest action, then that is fine. If, on the other hand, the costs are very high, you should reverse tracks. You should find the most sensitive parameters, look at the data for those parameters, and focus the research on that. The scientific community is not doing bad work. It is just that they are not being funded to do it correctly, so scientists often make do with what they have, not with what they should have.

Victor McFarland, U.S. ACE, Waterways Experiment Station:

I would like to mention a word about terminology. When we first started talking about the relationship between neutral chemicals and their concentration in sediments and in organisms based on organic carbon and lipid normalization, we called that a preference factor. Later, the preference factor was changed to an accumulation factor, which was essentially the same thing. Then it became a biota-sediment accumulation factor, and we were still talking about the same thing, although the term got stretched to include a disequilibrium situation for fish instead of applying it to just invertebrates and such. Now we are talking about a BSAF for metals, and I am unable to figure out how things are going to be done or what the rationale is.

Laura Weiss:

I assume you are addressing me regarding the slide that alluded to BSAFs from metals. That was probably an oversight in terms of the terminology. The report focused on the bioaccumulation of certain metals, such as methylmercury and tributyltin. These chemicals are of concern from a human health perspective as they are known to bioaccumulate in aquatic biota. The definition of BSAF includes TOC and lipid normalization, and clearly that is not appropriate for metals. Therefore, that was an error on my part since the term BSAF is not strictly applicable to metals.

Q (Victor McFarland): Is it too late to change it and call it something else like a biota/sediment metals factor?

Laura Weiss:

I am interested in some terminology for that. Like I mentioned, it seems that our biggest challenge is quantifying the factor so a sediment level can be determined to protect human health.

Q (Philip Cook, U.S. EPA, Office of Research and Development): Amy, I am concerned about one part of your presentation, which dealt with the comparison of predictions from the site-specific BSAFs to predictions from BAFs. You described the BAFs as being consistent with the Great Lakes Water Quality Initiative methodology. However, if I followed your equations correctly, my impression was that you were applying those BAFs to predicted concentrations in the water and that resulted from an assumption that there was an equilibrium between the sediments and the water. How did you calculate the water concentration so that you could apply the BAFs?

Amy Pelka:

I did not do those calculations. For Buffalo and Saginaw, I think we had water column concentrations that came from the ARCS program. The water column concentrations were measured.

Q (Philip Cook): Were those water concentrations based on freely dissolved chemicals?

Amy Pelka:

Both. When it was available, we looked at freely dissolved and total.

Q (Philip Cook): How did you calculate freely dissolved?

Amy Pelka:

I am not sure of that. I think that would have been operationally defined, where freely dissolved was what passed through a filter.

Q (Philip Cook): The point I would make, in conclusion, is that the BAFs that you used are based on freely dissolved chemicals. You have to be very concerned about what model you use to predict that exposure concentration. In this case, fluctuations in concentrations over time have to be considered, and, if there is a disequilibrium between the sediments and the water column, that would factor into the analysis. So, I have some concern that the high values you predicted for some of the fish may have resulted from improper use of BAFs.

Amy Pelka:

I would have to go back and look more carefully at those specific calculations. I want to make sure I understand your point. Your point is that the BAFs are based on the dissolved portions, and we should be consistent in terms of the water measurements that are used.

Philip Cook:

Yes.

Amy Pelka:

I believe that was taken into account, but those calculations were done a while ago. You are right, though. It is important to be consistent.

Laura Weiss:

I would like to address Mr. Watts. I believe you made a comment about cost, and I would like to reiterate that, in Puget Sound, cost does play a role in our decision-making process. In addition, before we can adopt criteria, we have to go through a cost-benefit analysis as required by our legislature. Cost is something we cannot ignore.

Amy Pelka:

Cost is also considered in the Superfund remedial process. The costs and implications for different cleanup goals and remediations are an important part of the negotiation discussions. They are, by no means, forgotten in any sense of the word.

Session Six: Ecological-Based Risk Assessment

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Use of Bioaccumulation Data in Aquatic Life Risk Assessment

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Introduction

Problems of sediment contamination are being viewed increasingly from a risk assessment perspective. Environmental managers with regulatory responsibility for hazardous waste sites, sediment dredging and disposal, and similar problems are beginning to use the tools and approaches of risk assessment to evaluate adverse effects associated with chemical concentrations measured in sediments. In the most general sense, accurate estimation of ecological risks requires both quantification of environmental exposure conditions and understanding of the biological and ecological effects resulting from that exposure. Ecological risk involving chemical stressors is a function of chemical concentration or dose at the site of toxic action (DSTA) and the biological or ecological effects occurring at that chemical concentration. Historically, external exposure has been used in aquatic toxicology as a surrogate for internal dose. Body burden and tissue residue data are thought to provide more direct measures of DSTA. Without complete understanding of the internal dynamics of chemical stressors and their mechanisms of toxic effect, however, these measures are still but estimates (although hopefully improved) of DSTA.

So what, then, are the uses of bioaccumulation and tissue residue data in assessing ecological risk? The value of this kind of information obviously is limited to assessments involving chemical stressors. Further, the data confer insight solely about exposure, just one part of the risk assessment puzzle. Sediment risk assessments in which bioaccumulation is an issue presently focus on the biological responses of individual organisms or their component cells and tissues. However, organismal response can be extrapolated to population-level impacts, and given the appropriate ecological relationships, bioaccumulation can be related to community and ecosystem responses. Assessments involving these levels of ecological organization require trophic transfer models and models involving species interactions.

The objectives of this presentation are threefold: (1) to describe ecological risk assessment and to present EPA's approach for conducting such assessments; (2) to identify how bioaccumulation and tissue residue data are used in each of the steps of ecological risk assessment with respect to aquatic life; and (3) to highlight some of the key uncertainties associated with uses of bioaccumulation data in making risk-based management decisions. Although EPA's framework is by no means the sole approach used to evaluate risks, description of this paradigm will help to illustrate the uses of bioaccumulation data in the various components of any risk assessment. By enumerating uncertainties, I hope to identify general areas of future research that could improve the utility of bioaccumulation information in evaluating the ecological risks of contaminated sediments. Much of the information provided here is obvious; yet it is important to keep these ideas in the forefront of discussions concerning bioaccumulation to ensure that misconceptions are not propagated as part of environmental management and the communication of risks.

Ecological Risk Assessment

Ecological risk assessment can be described as a process for estimating the likelihood of adverse ecological impact resulting from anthropogenic stress. Risk assessments can be retrospective, prospective, or a combination of both. In the context of sediment contamination, retrospective assessments attempt to quantify the impacts of past releases of contaminants on sediment-associated receptors to enhance understanding of current ecological condition. This often is the type of application used when evaluating impacts associated with hazardous waste sites. Prospective assessments involving sediment contamination attempt to predict future impacts based upon the nature and behavior of chemical stressors and potentially exposed ecological systems. Prospective assessments have utility, for example, when selecting among various dredged sediment disposal options. A combination of both retrospective and prospective approaches is useful for evaluating risks of in-place contaminated sediments when both current and future conditions (for example, under various remediation scenarios) are cogent.

As proposed by the U.S. Environmental Protection Agency's Risk Assessment Forum (USEPA, 1992, 1995), ecological risk assessment consists of three primary phases or steps: *Problem Formulation*, *Analysis*, and *Risk Characterization* (Figure 1). Some of the goals of Problem Formulation are (1) to evaluate existing information concerning stressors, receiving ecosystems, and potential ecological effects; (2) to identify assessment endpoints (valued ecological conditions or processes) to be protected; and (3) to develop a conceptual model describing potential risks to assessment endpoints. Discussions among the risk assessors, environmental managers, and other stakeholders are crucial in the process to ensure that the assessment addresses the important regulatory and societal concerns and that the information generated is useful in making environmental management decisions. The Analysis step involves characterization of exposure conditions in time and space, as well as evaluation of ecological effects potentially resulting from those levels of exposure. Analysis involves a variety of empirical and modeling activities, with the ultimate goal of developing profiles of exposure and effect. These profiles are synthesized into estimates of ecological risk during the Risk Characterization step. Characterization activities may be either qualitative or quantitative, and are directed toward providing the information necessary to make informed environmental management decisions. An analysis of the uncertainties associated with the assessment is a critical part of Risk Characterization. EPA's framework is intended to be general with respect to the nature of the stressor(s) and the ecological systems involved in any given assessment. It is therefore useful in assessments involving either chemical or nonchemical stressors, and all types of ecological systems. How bioaccumulation and tissue residue data are used in the specific steps of risk assessment is described in the next three sections.

Uses in the Problem Formulation Phase

Bioaccumulation and tissue residue data play three somewhat related roles in Problem Formulation: (1) to identify those stressors which may impact biological receptors, particularly at higher trophic levels (including humans); (2) to aid in initial descriptions of the potential extent and magnitude of sediment contamination; and (3) to assist in identifying the range of potential biological and ecological effects resulting from exposure. Information concerning important stressors, their concentration distributions, and the effects they potentially elicit supports development of a conceptual model that focuses the remainder of the risk assessment.

Appreciation of bioaccumulation potentials can lead to identification of contaminants that might be available to biological receptors, and may affect organisms at higher trophic levels. The potential for highly lipophilic organic compounds to bioaccumulate, for instance, identifies trophic transfer as an important exposure route when these compounds are present. Not only does this suggest that ecological receptors removed from immediate contact with the sediment should be considered, but it also leads to hypotheses concerning the biological transport of

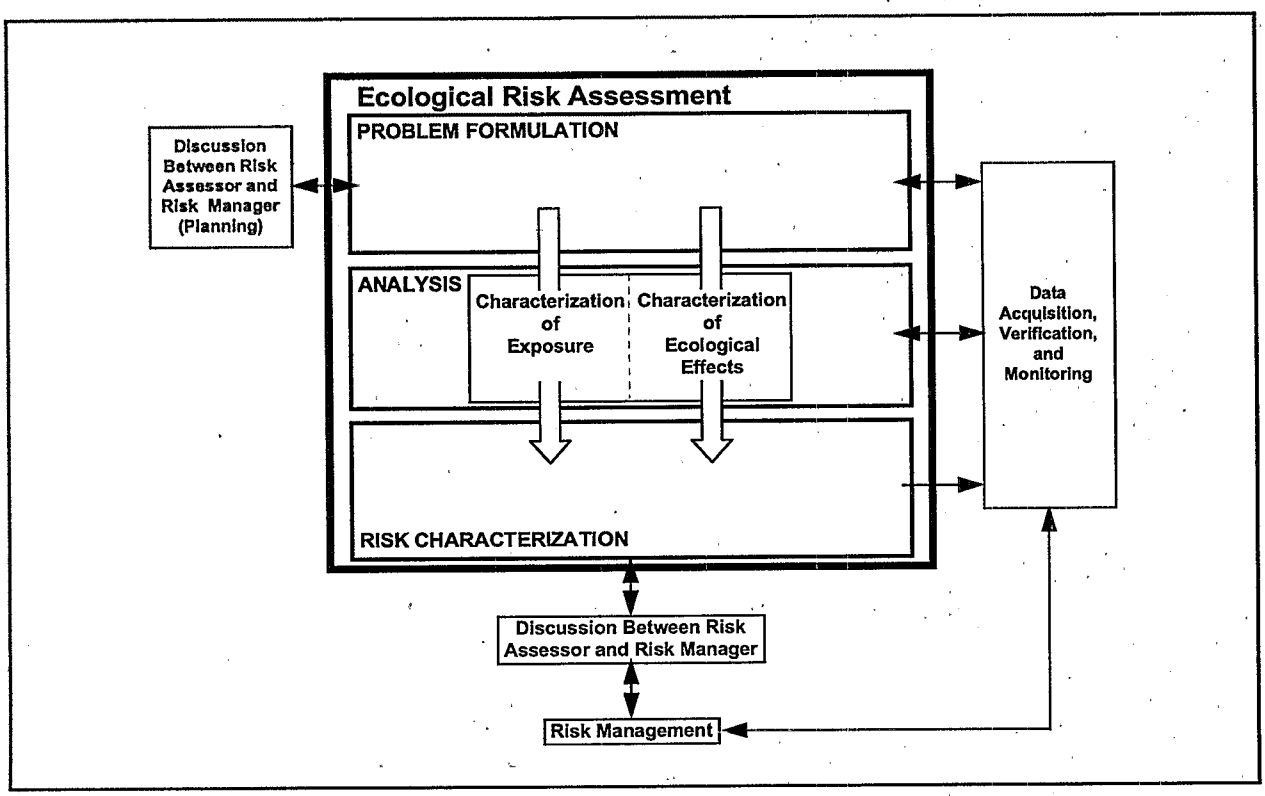

Figure 1. EPA's framework for ecological risk assessment (from USEPA, 1995).

contaminants away from the immediate site of contamination. Conversely, chemicals with low potential to bioaccumulate (some metals, for example) cannot be eliminated as stressors of concern, since the toxic effects of some chemicals are not tightly linked to body burden.

A requirement for adverse impact is the co-occurrence of the stressor with biological receptors. For chemical stressors associated with sediments, environmental exposure (external to biological receptors) is controlled by a number of geochemical factors. Historically, environmental exposure had been quantified as the chemical's bulk concentration in the sediment. Recent advances in understanding the partitioning of chemicals among various environmental phases has enhanced the accuracy of predictions of the availability of chemicals to biological receptors, particularly those in intimate contact with the sediment. In many cases, tissue residue data provide independent verification of these predictions and support description of the extent of contamination. As importantly, tissue residues support evaluation of the extent of exposure to those receptors somewhat removed from direct contact with the sediment. For example, elevated contaminant levels in deployed blue mussels or pelagic finfish indicate contaminant transport from the sediment to the water column or through trophic transfer. This in turn implies that ecological effects may not be limited to benthic organisms.

Knowledge of the degree to which contaminants bioaccumulate, and the tissues in which they accumulate, can provide insight to potential biological effects. PCBs, for example, are known to accumulate in lipid-rich tissues such as gonads and have been associated with reproductive impairment. They also can be transferred during oogenesis to potential offspring, and can cause a number of developmental and survival effects. Thus, in addition to effects resulting from trophic transfer, potential transgenerational effects may be possible.

In combination, the information above can be used to define a conceptual model of exposure leading to potential ecological effects. The conceptual model can incorporate hypotheses of how contaminants move through the physical environment and biotic food webs, thereby identifying key exposure pathways and exposure media for further evaluation. Figure 2 illustrates a generalized conceptual model relating potential contaminant sources in a watershed to ecological receptors in an estuary. In this model, a chemical released to the environment through anthropogenic activity enters the estuary via surface water, ground water, and atmospheric routes. Phase partitioning, water movement, and transport of particulates redistribute the chemical to various environmental compartments (including sediments) within the estuary, leading to potential exposure of a variety of aquatic organisms. Geochemical and biological processes influence uptake of the chemical by biological receptors, which in turn may result in its transfer to organisms at higher trophic levels. In addition to providing a description of environmental exposure pathways (primarily transport and fate), fully developed conceptual models communicate hypotheses concerning the potential adverse effects that may result from exposure. This often requires greater detail in

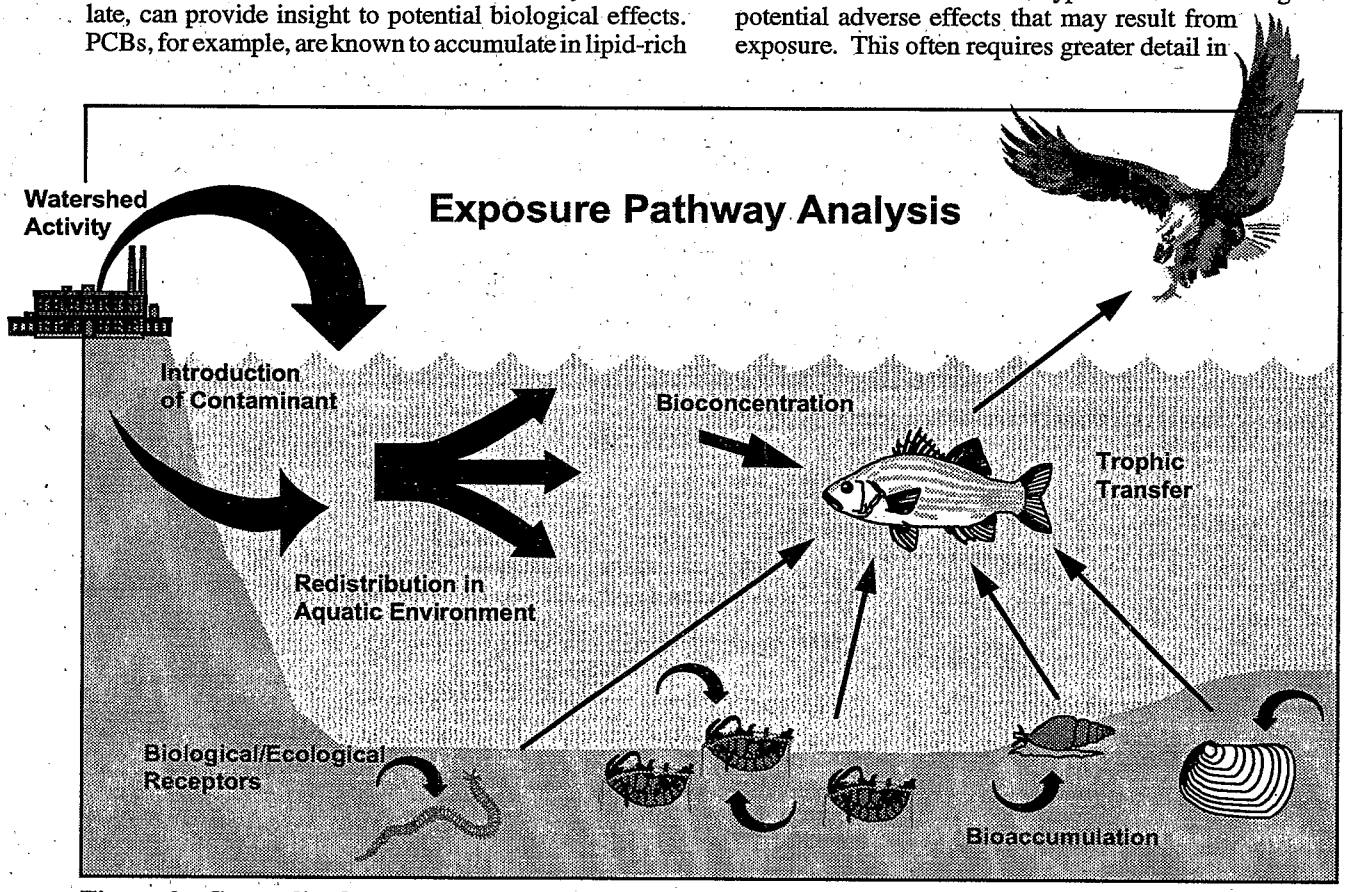

Figure 2. Generalized conceptual model relating contaminant sources to estuarine receptors.

the description of chemical uptake by receptors, and can involve conceptual understanding of the toxicokinetics and toxicodynamics of the contaminant within receptor organisms (Figure 3).

When developed on an assessment-specific basis, the conceptual model can be used to guide the activities in the Analysis phase. For instance, should the chemical of interest be persistent and display a high potential for biological uptake, the conceptual model would dictate analysis of trophic transfer (either empirically or through modeling efforts). Conversely, conceptual models hypothesizing little potential of risk to key consumer organisms, as a result of low bioaccumulation potential or the absence of important trophic pathways, may focus analysis activities on the direct toxicological effects on benthic organisms.

Uses in the Analysis Phase

The Analysis phase of ecological risk assessment involves characterization of exposure and characterization of ecological effects. Legitimately, bioaccumulation data are primarily restricted to evaluations of exposure. Tissue residues can be used, however, as independent variables in models relating exposure to effects.

As discussed previously, bioaccumulation data can help define the availability of chemicals to receptor organisms, improving the accuracy of estimates of exposure over bulk measures made in sediments. For some contaminants, tissue residues can be used to quantify exposure along pathways leading to consumer organisms (including humans). This information is most useful in models of trophic transfer. Residue data also support analysis of the fate of chemical stressors when significant biotransformation or biological transport is possible. These kinds of information support development of a profile describing the nature, extent, and severity of exposure to contaminants found in sediments.

Use of bioaccumulation data to characterize effects in the Analysis step is limited to quantifying internal dose in development of dose-response relationships. These models relate the degree of exposure to levels of biological/ecological response, typically generated through laboratory or field experimentation. Although true DSTA-response models theoretically provide the most accurate predictions of likely biological effect, exposure-response and residue-response models can also be useful. Promising approaches for developing relationships between residues and toxicity have been proposed by McCarty (see McCarty and Mackay, 1993) and others, and several efforts are under way to construct databases containing residue-effects information. Extrapolations to threshold residue values from ambient water quality criteria (Shephard, this conference) and similar toxicity-based benchmarks also hold promise. However, attempts to relate internal dose of chemical stressors to biological effects have met with varying degrees of success.

Bioaccumulation usually is considered a phenomenon relevant to individual organisms, and past assessments of contaminated sediments have tended to focus on effects manifested in individuals (mortality, reproduction, growth, and development). Residue data can also be linked to responses at higher levels of ecological organization. For instance, Munns et al. (1997) used a modeling approach to extrapolate survival and reproductive effects of PCBs on mummichogs (*Fundulus heteroclitus*) to estimates of population growth rate as

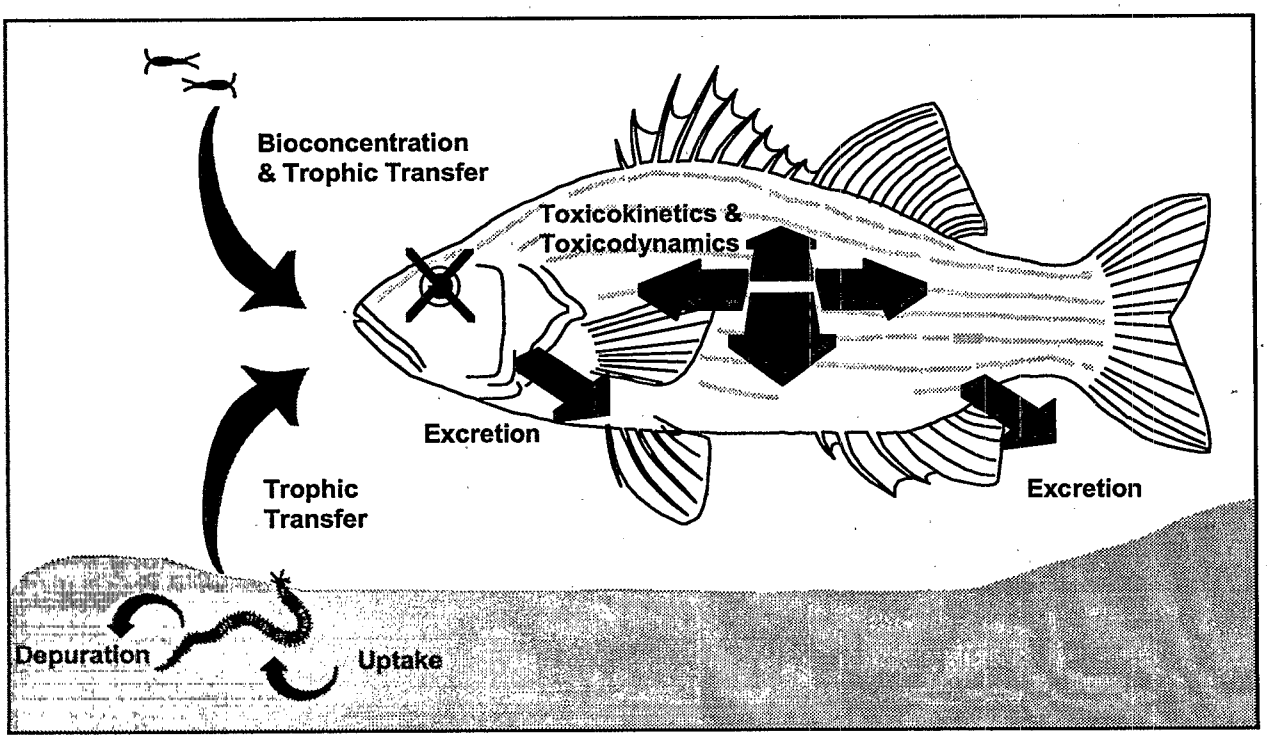

Figure 3. Detail of conceptual model showing contaminant kinetics within an organism.

a function of liver PCB burden (Figure 4). Despite involving many assumptions, such approaches can be subjected to verification through manipulative laboratory and field experimentation. Similar extrapolations can be made to responses manifested at higher levels of ecological organization (for example, shifts in community structure and function mediated by direct toxic effects on individuals and indirect effects resulting from changes in species interactions) by applying ecological models that incorporate residue-response relationships.

Uses in the Risk Characterization Phase

The exposure and ecological effects profiles developed during Analysis ultimately are used to develop understanding of the risks posed by contaminated sediments. The utility of bioaccumulation and tissue residue data in this process therefore plays out directly from the exposure and effects analyses. Similarly, the limitations on their use alluded to above also apply in Risk Characterization.

Both qualitative and quantitative methods have been used to characterize risk. One of the qualitative approaches involves calculation of simple ratios of the environmental exposure concentration (measured or modeled) to biological benchmark concentrations. Biological benchmarks can be receptor-specific toxicity thresholds, sediment quality criteria or standards, or other sediment quality assessment guidelines. Critical body residues or other derived toxicity thresholds based on residue burdens offer a means to incorporate bioaccumulation data into this characterization approach. This so-called risk or hazard quotient approach is most useful in screening-level assessments, since the magnitudes of likely impact are difficult to ascertain from simple ratios. Other qualitative techniques include weight-of-evidence approaches that base conclusions about contaminant-associated risks on the preponderance of information evaluated during the assessment. Residue data are often included as evidence of exposure.

Quantitative characterization methods attempt to provide information concerning the realized or expected severity of impact, often in terms of the probability of a particular level of effect. Dynamic simulation modeling, static assessment, and distributional analysis are examples of techniques providing quantitative estimates of impact. Most complete with respect to analysis of the full range of impact are simulation models that incorporate both direct (toxicity) and indirect (species interactions) responses potentially resulting from contaminant exposure. Trophic transfer submodels are useful in this context when the effects of oral dose can be described. Quantitative techniques are often useful for evaluating the consequences of various remediation alternatives.

Key Uncertainties and Areas for Future Research

As reported during this conference (and elsewhere), our understanding of environmental processes leading to contaminant availability and uptake has improved substantially over the past 5 to 10 years. We have a fairly firm grasp of the sediment factors and partitioning dynamics which influence bioavailability, can model uptake and depuration kinetics with plausible accuracy, and can describe transfer of contaminants from

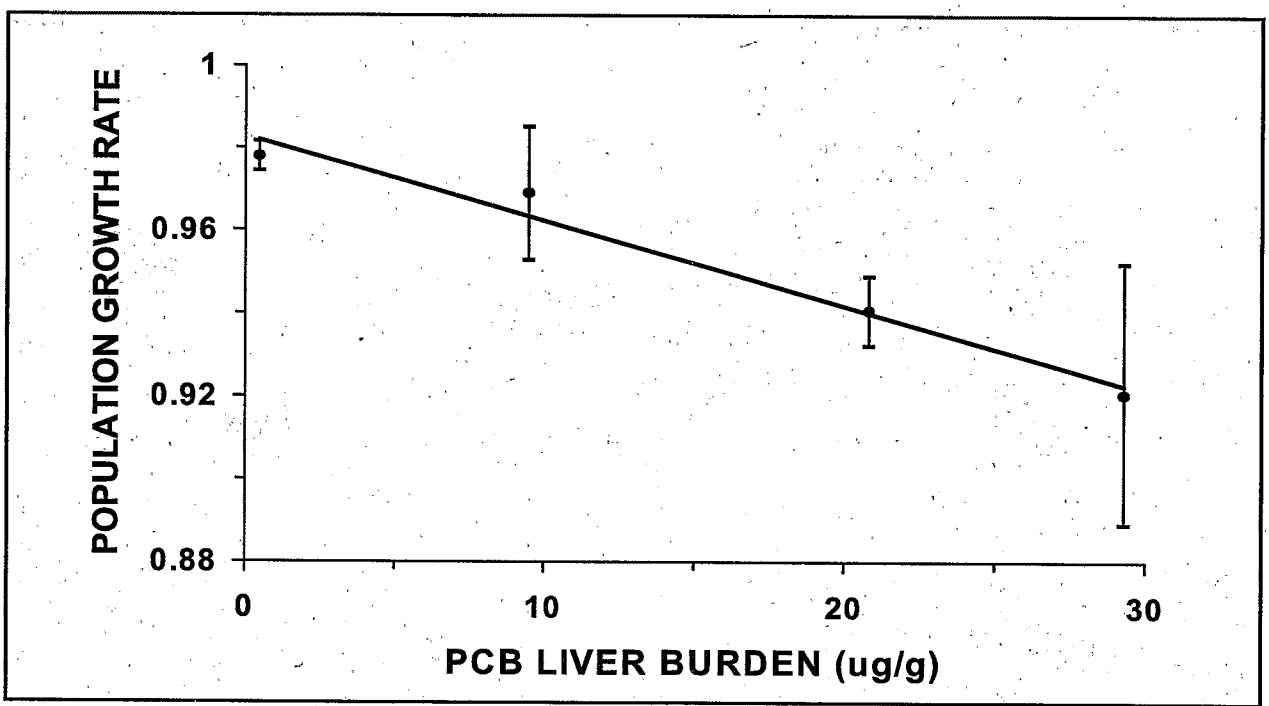

Figure 4. Population-level effects on mummichogs as a function of PCB liver burden (from Munns et al., 1997).

prey to predator with reasonable confidence. Although additional research is still needed in these areas, perhaps the greatest uncertainties associated with the use of bioaccumulation data in risk-based environmental management are associated with linking tissue residues to biological/ecological effect. While advances are being made in this area (for example, the critical body residue and similar empirical approaches), risk assessments will continue to rely primarily on appropriately normalized sediment concentrations as the measure of exposure until this nut is cracked.

In a general sense, factors that hinder our ability to develop residue relationships include: the rates at which contaminants are metabolized or eliminated; the toxicities of intermediate metabolites relative to parent compounds; dose-related induction of enzymatic systems; the modes and time course of toxic action; homeostatic processes resulting in immobilization/sequestration; and the environmental factors that mediate toxic effect. Toxicokinetic and toxicodynamic studies would appear to be fruitful approaches for addressing these issues. These, then, are some of the areas on which future research should focus.

Acknowledgments

W. Berry, D. Hansen, R. Johnston, and T. Dillon helped develop some of the ideas in this presentation. W. Berry, G. Pesch, and B. Brown kindly provided timely reviews of this paper. NHEERL AED Contribution no. 1807. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by USEPA.

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Wildlife Risk Assessment

David Charters
U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response,
Superfund Program Environmental Response Team Center, Edison, New Jersey

This presentation will cover the Superfund ecological risk assessment process and an ecological risk assessment conducted at the LCP Chemicals site in Brunswick, Georgia by the Superfund Program. The focus will be on the wildlife risk assessment part of the much larger ecological risk assessment conducted at the site under the Superfund removal program. Ecological risk assessment is the process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. The risk assessment process at this site involved applying the eight-step draft Superfund risk assessment guidance (USEPA, 1997) at the site to obtain information to answer important risk questions. This approach contrasts with the former process of collecting extensive data initially at a site and playing what was once referred to as "Million Dollar Jeopardy." The result of the past process was that the Superfund Program literally spent millions of dollars on sites to collect data, but often made decisions based on a small fraction of relevant data.

Risk Assessment Process

The current approach to risk assessment in the Superfund Program is for the risk assessors to work with risk managers during the initial site assessment to meet two objectives: (1) to identify the questions the risk managers need to have answered and (2) to build the scientific framework to support the risk assessment. This up-front planning will avoid the pitfall of reaching the end of the risk assessment process with only the general conclusion that it is a bad site where something needs to be done. The outcome of the ecological risk assessment should be cleanup goals, not lines around the site that define areas where everything inside the boundaries must be cleaned up.

In the Superfund ecological risk assessment process there are eight steps and five decision points which are referred to individually as a scientific management decision point (SMDP). At each decision point, it is the risk assessor's job to explain to the risk manager what has been done. It is critical that the risk manager understands all the information. This is also an appropriate time to share the assessment information with the community and other stakeholders in the process. The risk assessor must be able to communicate clearly to the public what is being done on the site and why it is being done.

The eight-step ecological risk assessment process for Superfund can be summarized as follows:

1. Preliminary problem formulation and ecological effects evaluation
2. Preliminary exposure estimates and risk calculation
3. Problem formulation
4. Study design and data quality objective process
5. Field verification of sampling design
6. Site investigation and analysis phase
7. Risk characterization
8. Risk management

Below is a brief description of each step in the risk assessment process.

The process begins with a preliminary risk assessment (steps 1 and 2) which is the equivalent of a screening risk assessment. The first step in the preliminary assessment includes the site visit, problem formulation, and toxicity evaluation; the second step involves development of preliminary exposure estimates and risk calculation. A preliminary assessment normally includes about 240 chemicals. These chemicals are run through a very preliminary and conservative screen initially to eliminate all the chemicals that show no indication that they need to be considered in the full risk assessment. In rare cases, a preliminary risk assessment can result in termination of the risk assessment if data indicates that there is no significant risk at the site. Both the risk assessors and the risk managers must agree on this determination. This determination is the first scientific management decision point.

The decision to continue the risk assessment initiates the full risk assessment process which is described in program-specific guidance, but follows the Agency framework for risk assessment (USEPA, 1997).

The decision to continue the risk assessment initiates the full risk assessment process, beginning with problem formulation (step 3). The problem formulation phase involves several activities including toxicity evaluation, selection of assessment endpoints, and development of the conceptual model and testable hypotheses. The second SMDP occurs at the end of problem formulation. At this point the involved parties must agree on the assessment endpoints, the conceptual model, the exposure pathways, and questions or risk hypotheses.

The risk asssessment continues with development of the study design and identification of data quality

objectives and statistical considerations (step 4). This process produces a work plan and sampling and data analysis plan. Agreement on the content of both plans constitutes the third SMDP.

The fifth step in the risk assessment process is verification of the field sampling plan. Field activities during this step will determine sampling feasibility. This step ends with the approval of the work plan and the sampling and analysis plan which is the fourth SMDP. With work plan approval, the risk assessment moves into the site investigation and analysis phase (step 6). The plans define how these activities will be conducted, but they may require modification depending on circumstances such as changing field conditions. An additional SMDP may be necessary to approve plan modifications.

The risk assessment process continues with the risk characterization (step 7). The three primary components of risk characterization are risk estimation, risk description (threshold for effects on assessment endpoints and other risk information), and uncertainty analysis. And finally, the process culminates in risk managers using the results of the risk characterization to make the record of decision (ROD). This decision is the final SMDP.

Site Summary

The LCP Chemicals site in Brunswick, Georgia offers a case study for Superfund wildlife risk assessment. The site initially contained a petroleum refinery which operated for more than 10 years. In 1955 it was sold to a chemical company that built a chloralkali plant on the property to manufacture caustic soda, chlorine, and hydrochloric acid. The plant continued to produce these chemicals at the site through the 1980s. Early in the risk assessment process some highly contaminated areas were identified within the site that all parties agreed needed remediation. PCBs measuring in the thousands of parts per million (ppm) were found in sediments from these areas. The Superfund removal program initiated some actual cleanups in these areas prior to the completion of the risk assessment. Elevated levels of mercury and lead were also found in sediments near the site.

The site is tidally influenced with a tidal range of about 8 feet. The tidal marsh at the site covers about 500 acres. It provides a very diverse habitat for wildlife, including a number of endangered species. Contaminants from the outfall area have been released through tributaries into the marsh. The question being addressed here was what needed to be done to restore the habitat not only for aquatic life, but also for wildlife. In this case, the primary objective was to restore and maintain the ecological health of the salt marsh community, particularly with respect to the structure and function of the marsh.

Assessment Endpoints and Results

The assessment endpoints are what risk assessors and managers target for ecological protection. Results from other sites indicate that current wildlife assessment endpoints are not particularly good. The Superfund Program is providing additional resources to ecologists to determine more specifically the critical ecological functions that require protection to maintain diverse wildlife habitats such as salt marshes. This information would allow risk assessors to derive better assessment endpoints and to define better measurement endpoints for wildlife risk assessment.

[Dr. Charters discussed how they evaluated contaminant concentrations in wildlife at the LCP Chemicals site, particularly for terrapins, otters, and racoons. His discussion focused on evaluation of mercury measurements. Selected data sets and related graphics presented at the conference are not available for publication in this paper. Please contact Dr. Charters directly for further information on the site-specific wildlife data for this Superfund site in Brunswick, Georgia. For additional information on EPA's Environmental Response Team, you may visit their website on the Internet at http://www.ert.org]

Summary

The LCP Chemicals site is one of the first Superfund sites to undergo evaluation using the eight-step risk assessment process. It is a site where the Superfund Program expanded the scope of the risk assessment to consider impacts to semi-aquatic and terrestrial animals living around the aquatic environment. That was a point of departure from the normal EPA approach of being heavily oriented to protection of aquatic organisms. The Superfund Program has recognized that a lot of sites have some terrestrial component associated with them. The program is working very hard to come up with appropriate numbers to use for screening aquatically or terrestrially. This would allow the program to do a better job of identifying semi-aquatic and terrestrial species that are significantly impacted by bioaccumulative compounds at these sites.

Going through the eight-step process, the Superfund Program is starting to address some of the issues that have been addressed in the aquatic community up to the fish level. The program is beginning to focus on higher species that utilize fish as forage and introduce contamination into the terrestrial environment, including ones that have not been evaluated in the past. Sediment bioaccumulation is an important part of the wildlife evaluation not only from a question of sediment uptake directly into these species, but also from the view that significant impacts can result from contaminant-induced changes in behavior.

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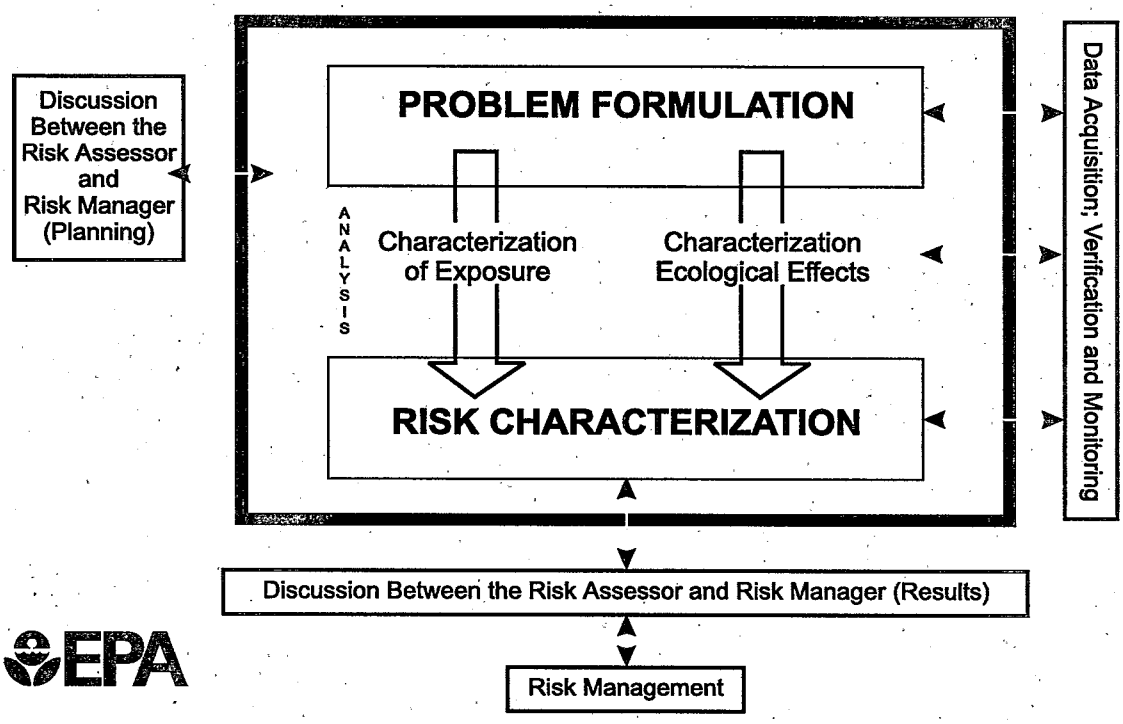
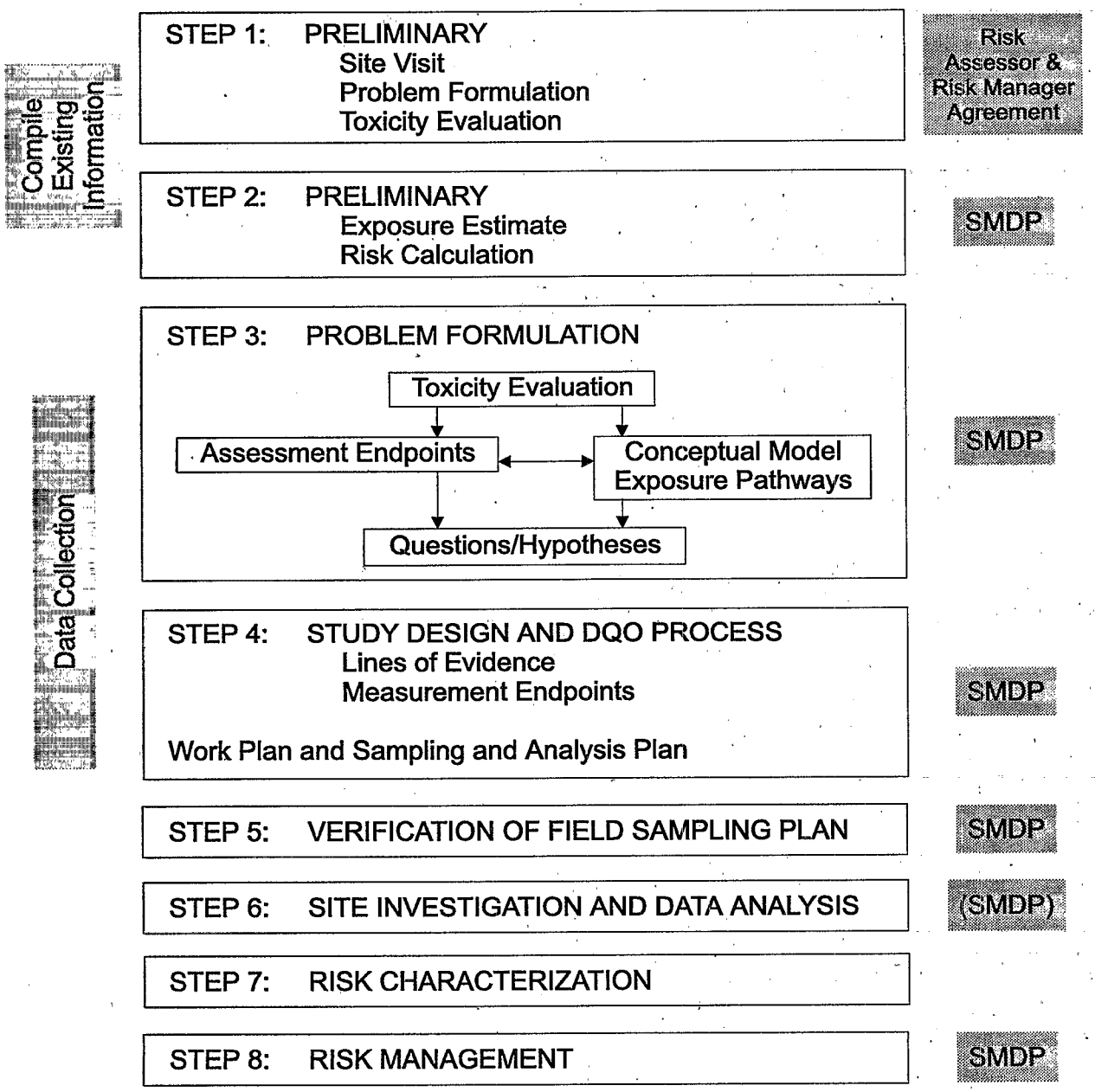
Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments

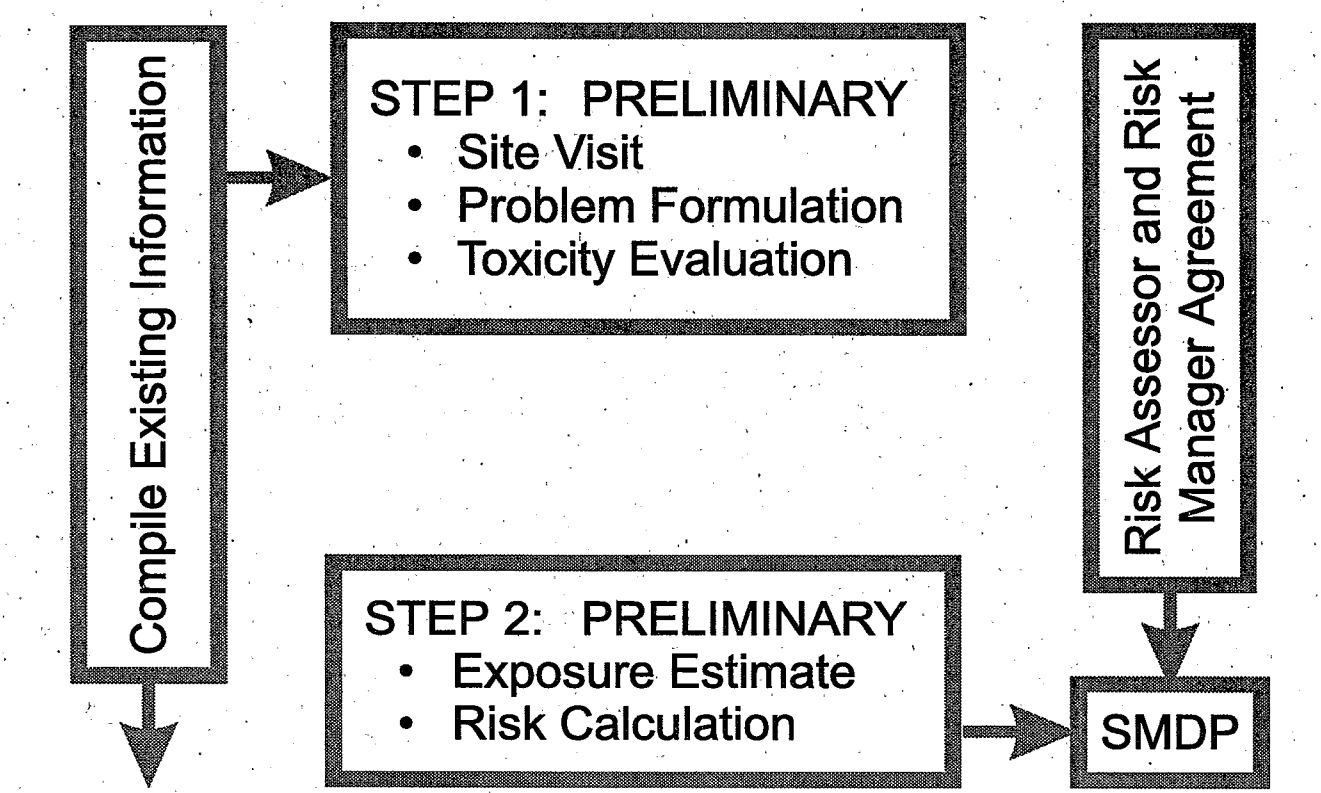
David W. Charters, Ph.D.
Mark D. Sprenger, Ph.D.

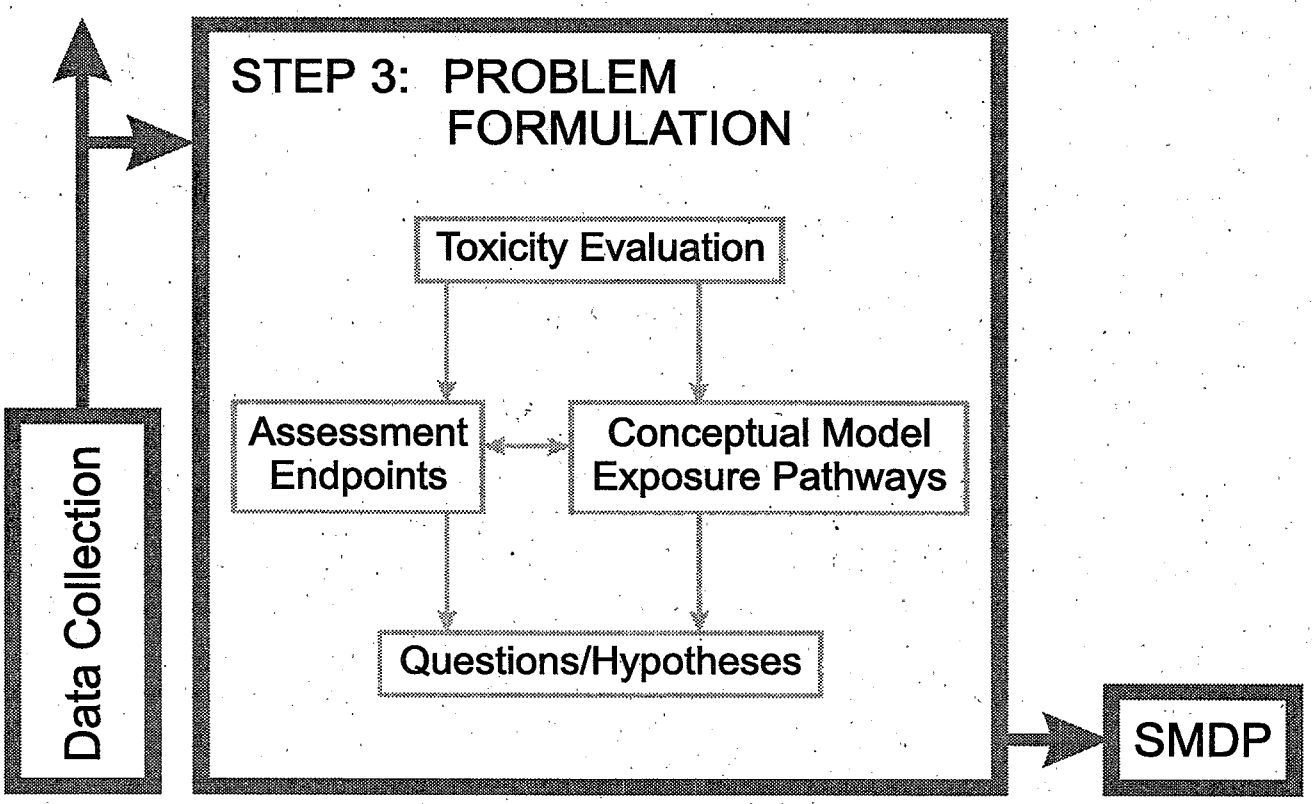
Ecological Risk Assessment Framework (U.S. EPA, 1992a)

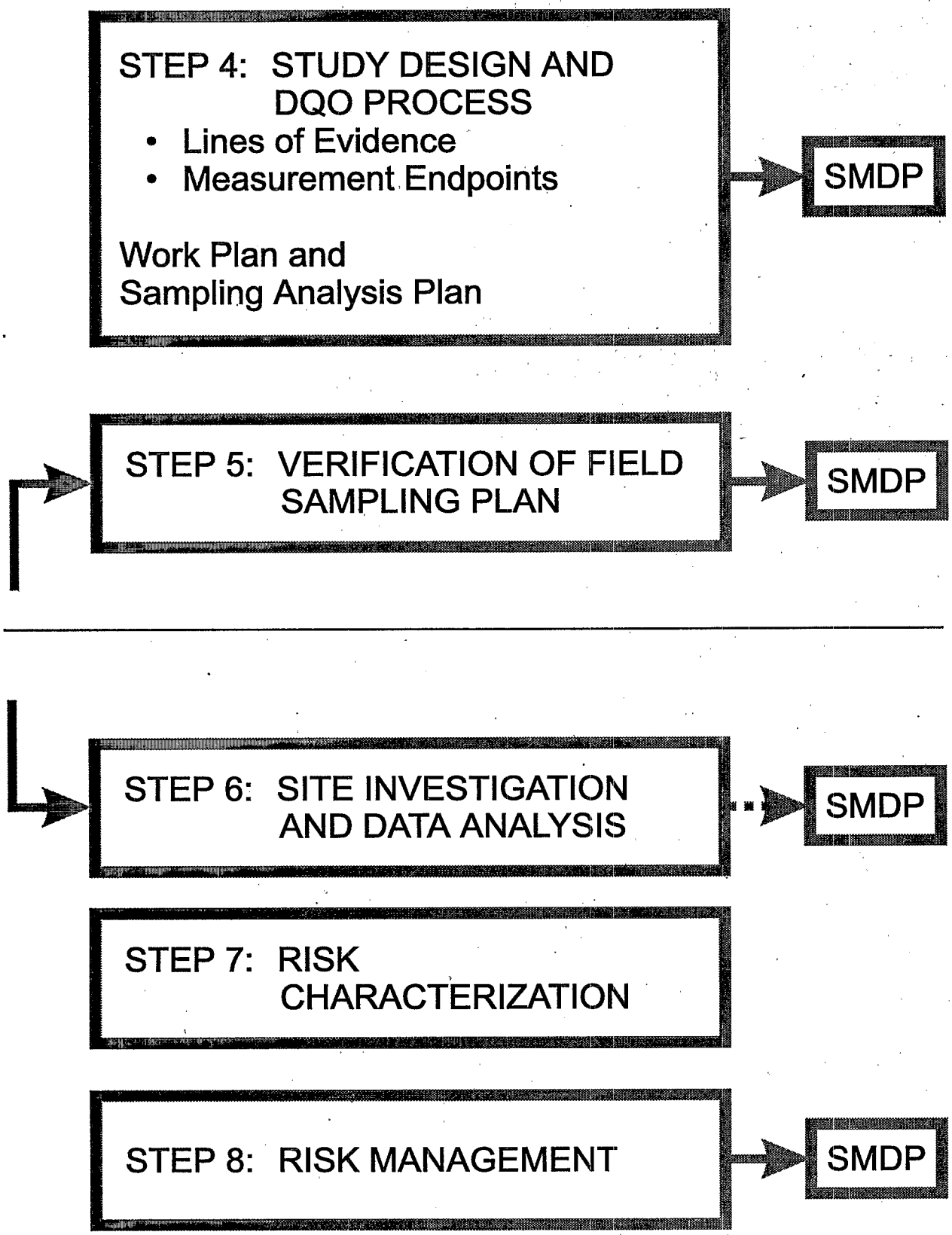


Eight-Step Ecological Risk Assessment Process for Superfund




Compile Existing Information
STEP 1: PRELIMINARY
• Site Visit
• Problem Formulation
• Toxicity Evaluation
Risk Assessor and Risk Manager Agreement
STEP 2: PRELIMINARY
• Exposure Estimate
• Risk Calculation
SMDP


STEP 3: PROBLEM FORMULATION
Toxicity Evaluation
Assessment Endpoints
Conceptual Model Exposure Pathways
Questions/Hypotheses
Data Collection
SMDP


STEP 4: STUDY DESIGN AND DQO PROCESS
• Lines of Evidence
• Measurement Endpoints
Work Plan and Sampling Analysis Plan
SMDP
STEP 5: VERIFICATION OF FIELD SAMPLING PLAN
SMDP
STEP 6: SITE INVESTIGATION AND DATA ANALYSIS
SMDP
STEP 7: RISK CHARACTERIZATION
STEP 8: RISK MANAGEMENT
SMDP

Day Two: September 12, 1996

Session Six: Questions and Answers

After each session, there was an opportunity for questions and answers and group discussions pertaining to the speakers' presentations.

Q (Arnold Kuzmack, U.S. EPA Office of Water): Dave, I could not tell on your mercury charts whether you were distinguishing inorganic from organic mercury. You really should.

David Charters:

We are not. That is a nice toxicological issue, but it is not a cleanup issue.

Q (Arnold Kuzmack): Whether or not the hazard quotient is really greater than one, I think, compels what kind of mercury it is. In terms of the process you are laying out, I noticed that at the step of risk characterization there was not a scientific management decision point (SMDP), and it strikes me as extremely important to have one.

David Charters:

The difference there is that the discussion on what is risk characterization is pretty much the risk assessors trying to figure out how it is done. In risk management, it is more consideration of the risk communication issues. We need to keep these separate. We do not want risk management and risk communication to be confused any more than has historically been done. So it is the same thing once again. It is whether it is in step seven or in step eight.

Q (Todd Bridges, COE Waterways): David, you did not say what you did with the threatened and endangered species. Did you treat them in the same way that you did the raccoon?

David Charters:

No, the threatened and endangered species were done through the two trust agencies, U.S. Fish and Wildlife and NOAA. We do not have a legal responsibility there, but we must comply with the spirit of that law in working with Fish and Wildlife and NOAA. It is not a unilateral EPA decision. At this point, those decisions have not been made.

Q (Participant): I noticed you used a LOAEL when you were calculating your hazard quotient rather than a NOAEL.

David Charters:

We had better information on the LOAEL. We also have the no-observed-adverse-effect level (NOAEL) and we can take it down to that. This was for illustration purposes. It is not the standard way that is taken. If you want to turn those into NOAEL numbers, multiply them by 10. What we are really looking for is not only to be reasonable, but also to use the best data available to address the question, "Is there a realistic probability of an adverse impact or not?" This is the real part of the risk assessment, not the screening anymore. We felt that we were working with the best information and that it was very solid.

Session Seven: Integrating Bioaccumulation Results into EPA's Decision-Making Process

Elizabeth Southerland, Panel Moderator
U.S. EPA, Office of Science and Technology,
Washington, DC
Opening Remarks

Michael Kravitz
U.S. EPA, Office of Science and Technology,
Washington, DC
Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment: Status and Needs

Opening Remarks

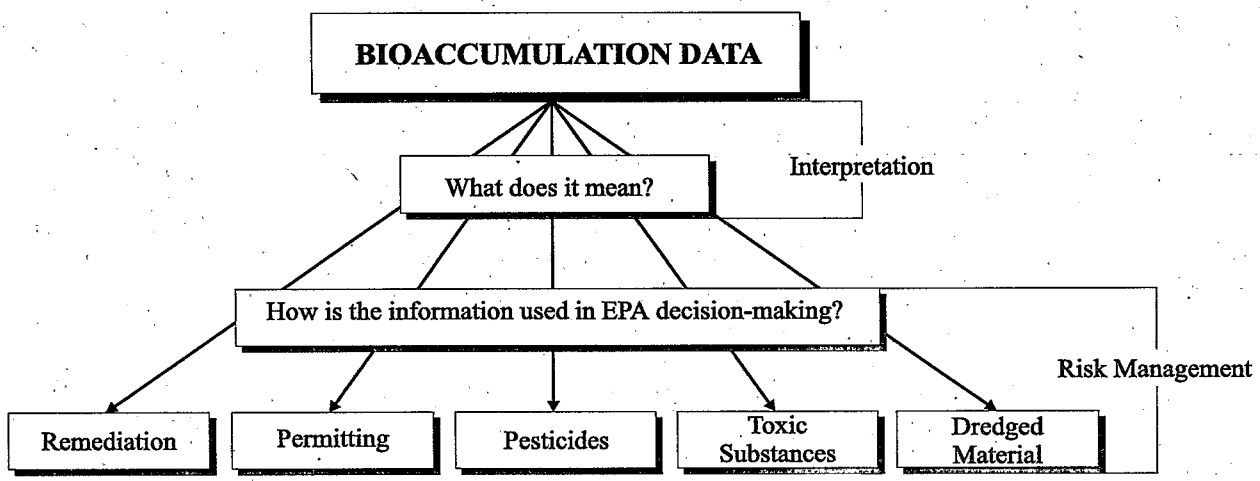
Elizabeth Southerland
U.S. Environmental Protection Agency, Office of Science and Technology, Washington, DC

During the first two days of the conference, we heard technical presentations. Today we will be focusing on policy. I would like to review today's agenda. We will open with speakers from four regulatory programs in EPA that use bioaccumulation data. The speakers will describe the current use of this data in their regulatory programs. After the program presentations, I will begin the panel discussion by asking the panelists to identify barriers to using bioaccumulation data in their programs and additional needs to allow more efficient use of bioaccumulation data. Once each panelist has had an opportunity to address those issues, I will open the discussion up to the entire audience. During this time you can share your ideas about future needs and comment on any of the information that has been presented.

Before we begin the panel presentations, I will give a brief overview of the programs in EPA that use bioaccumulation data. Then Mike Kravitz from my Division in the Office of Science and Technology will talk about an EPA report that is currently under development on bioaccumulation testing and interpretation for assessing sediment quality. The report is being developed by an Agency workgroup that is co-chaired by the Office of Water and the Office of Solid Waste.

The program representatives speaking today will explain how bioaccumulation data is used in EPA regulatory programs. The data may be from laboratory assays that we heard about on the first day of the conference, or it may be from field studies similar to those presented on the second day of the conference. Once these programs have collected the data, their primary concern is determining what it means in the context of their regulations. The interpretation issue has been raised a number of times at this conference. There are fundamental questions that we are still trying to answer about bioaccumulation data. One of the critical issues to help resolve these questions is to define tissue residue levels of concern for regulatory programs. All of us feel that we need to have a better idea of how to interpret this data. Once we can interpret it by applying better scientific information, we can move to the next step and determine how to use it in regulatory decision-making.

Our panelists today are representatives of the Superfund remediation program in the Office of Emergency and Remedial Response, the industrial chemicals program in the Office of Pollution Prevention and Toxics, and the permitting (National Pollutant Discharge Elimination System) and dredged material programs in the Office of Water. Again, they will be giving you an overview of how they currently use bioaccumulation data in these programs. But before we hear from our panelists, let me introduce Mike Kravitz, who is our Office of Water chair of the Bioaccumulation Analysis Workgroup. He will tell us about the bioaccumulation report that the workgroup is preparing.



Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment: Status and Needs

Michael Kravitz
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Problem Statement

Sediments serve as both a sink and a reservoir for persistent chemical contaminants, some of which are bioavailable or become bioavailable as conditions change naturally or anthropogenically. For instance, metals bioavailability can change in estuaries depending on seasonal changes in the influences of riverine flows and oceanic tides (Geesey et al., 1984). Bioturbation (mixing and movement of sediments by organisms) can also affect bioavailability of sediment contaminants by increasing oxygen and nutrient exchanges, and increasing exchange of contaminants with overlying water. Increased or sufficient bioavailability of contaminants can result in bioaccumulation and, depending on the contaminant and level of bioaccumulation, can also result in toxicity and/or in transfer to consumers through dietary uptake. In the case of certain contaminants (i.e., arsenic, mercury, methyl mercury, PCBs, DDT, DDE, toxaphene), biomagnification up the food chain can occur, affecting higher trophic levels (Suedel et al., 1994; USACE, 1995).

Bioaccumulation of toxic persistent organic contaminants by aquatic organisms is a concern for several federal agencies, including the U.S. Environmental Protection Agency (EPA), U.S. Army Corps of Engineers (USACE), National Oceanic and Atmospheric Administration (NOAA), U.S. Fish and Wildlife Service (FWS), and U.S. Geological Survey (USGS). EPA's National Sediment Quality Survey (USEPA, 1997a) has shown that these contaminants are widely distributed in sediments throughout the United States. The National Study of Chemical Residues in Fish (USEPA, 1992a) has demonstrated that these compounds are detectable in fish tissues, and many states have issued fish consumption bans as a result of bioaccumulative compounds reaching concentrations in fish tissue that may pose a threat to humans that consume them (USEPA, 1997b). For these reasons, EPA and other Federal and State agencies have identified a need to find solutions to problems associated with bioaccumulative compounds in sediments.

Scope of the Document

The EPA document, "*Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment: Status and Needs*" is intended to summarize the current status of our knowledge of bioaccumulation and recent developments in bioaccumulation research that might improve our ability to use bioaccumulation testing to evaluate sediment quality. Chapter 2 discusses the information compiled in chemical-specific summary tables (Appendix) that represent bioaccumulation research conducted during the past 10 years. The summary tables contain information associating the presence and quantity of potentially bioaccumulative chemicals in sediment with uptake in the tissues of aquatic and terrestrial organisms and with the effects of those chemicals on the organisms. Chapter 3 discusses factors affecting the bioavailability of sediment-associated contaminants. Chapter 4 describes methods and techniques that have been developed for measuring and modeling bioaccumulation. Chapter 5 presents brief synopses of current research on and uses of bioaccumulation testing in several Federal agency programs to support regulatory activities. Finally, Chapter 6 summarizes further research needs for the development of guidance for interpreting bioaccumulation of persistent organic pollutants to assist in protecting aquatic and terrestrial biota and humans from toxic effects of bioaccumulative chemicals in sediments.

Purpose

A number of sediment assessment methods have been developed to determine the bioaccumulation potential of contaminants in sediments, but overall guidance on interpretation of test results in the evaluation of ecological and human health effects is lacking. To begin to address this concern, EPA's Office of Science and Technology (OST) and Office of Solid Waste (OSW) formed a "Bioaccumulation Analysis Workgroup" consisting of 40 Headquarters and regional participants. This workgroup has overseen the production of the present "status and needs paper," the purpose of which is to provide background information and report on the status of bioaccumulation testing and interpretation in various EPA programs (and other federal agencies). EPA's Office of Water envisions that the paper will serve as the basis for EPA-wide cross program guidance on interpretation of bioaccumulation tests for the purpose of sediment quality assessment. Ultimately, integration of interpretable bioaccumulation tests into a regulatory decision-making framework will be required, with the understanding that this would be subject to case-specific modifications based on individual program needs.

Regulatory Uses

A brief synopsis of possible uses of bioaccumulation data in EPA programs implementing a variety of statutes is presented on page 7-9. More detailed information on how bioaccumulation data is used by various programs is provided in Chapter 5. Typical applications of bioaccumulation guidance might be the characterization of sediment contamination at Superfund sites, the verification of contaminants of concern in sediment for purposes of NPDES permitting, or the selection of disposal options for dredged material.

The Office of Enforcement and Compliance Assurance (OECA) is responsible for developing and implementing enforcement and compliance assurance strategies for the National Environmental Policy Act (NEPA) and other federal regulations. As such, OECA may use bioaccumulation data under a broad range of statutes to determine the environmental acceptability of proposed Federal actions.

The Office of Prevention, Pesticides, and Toxic Substances (OPPTS) uses the results of bioaccumulation tests to support review of new and existing chemicals under the Toxic Substances Control Act (TSCA) and the registration/re-registration of chemicals under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA). In addition biaccumulation information may be used to provide guidance on the design of new chemicals to reduce bioavailability and partitioning of toxic chemicals to sediment.

The Office of Solid Waste and Emergency Response (OSWER) is responsible for controlling hazardous wastes and remediating hazardous waste sites under the Resource Conservation and Recovery Act (RCRA) and the Comprehensive Environmental Response Compensation and Liability Act (CERCLA). Under CERCLA, the Office of Emergency and Remedial Response (OERR)—the Superfund Program—uses sediment assessment methods, including bioaccumulation data, as a standard part of initial sampling during the preliminary site assessment and the more in-depth remedial investigation/feasibility study for Superfund sites where sediment contamination may be present. Under RCRA, OSW is preparing a rule that addresses listed hazardous wastes, and mixtures of and residues derived from managing the hazardous wastes that pose low risks to human health and the environment. The rule will establish chemical-specific concentrations in wastes to be eligible for a self-implementing exemption from the hazardous waste management system requirements under Subtitle C of RCRA. A risk-based methodology is under development that will be used as the basis for the exit concentrations. The methodology considers the bioaccumulative potential of relevant chemicals in the evaluation of potential exposures from multiple pathways, in multimedia, and from a variety of waste management units.

In response to the Hazardous and Solid Waste Amendments of 1984 (HSWA), which amended RCRA, and the Pollution Prevention Act of 1990 (PPA), EPA released the Waste Minimization National Plan (WMNP) in November 1994. The WMNP focuses on reducing the generation and subsequent release to the environment of the most persistent, bioaccumulative, and toxic chemicals in hazardous wastes. One of the objectives of the WMNP was to develop a flexible risk-based screening tool that would assist stakeholders in identifying source reduction and recycling priorities. EPA committed to fulfill this objective by developing a tool that would prioritize chemicals based on their persistence, bioaccumulation potential, toxicity, and quantity. This screening tool—the Waste Minimization Prioritization Tool (WMPT)—has been developed by OSW and the Office of Pollution Prevention and Toxics (within OPPTS). It is currently under public review.

The Office of Water (OW) is responsible for EPA's water quality activities, which represent a coordinated effort to restore the nation's waters. The functions of this program include developing national programs, technical policies, and regulations relating to drinking water, water quality, and ground water; establishing environmental and pollution source standards; and providing for the protection of wetlands. In addition, this Office furnishes technical direction, support, and evaluation of regional water activities; enforces standards; and develops programs for technical assistance and technology transfer. The Office oversees the provision of training in the fields of water quality, economic and long-term environmental analysis, and marine and estuarine protection.

OW and the USACE developed joint technical guidance for evaluating the potential for contaminant-related impacts associated with the discharge of dredged material in the ocean under the Marine Protection, Research, and Sanctuaries Act (MPRSA) (USEPA and USACE, 1991). Similar updated guidance has been drafted for evaluating dredged material discharges in fresh, estuarine, and saline (near-coastal) waters under

Section 404 of the Clean Water Act (CWA) (USEPA and USACE, 1994). These documents employ a tiered testing protocol in which bioaccumulation data figures prominently.

Under Sections 301, 304, 306, and 307 of the CWA, the Office of Science and Technology (OST) (within OW) promulgates technology-based national effluent limitations guidelines that control the discharge of toxic chemicals and other pollutants by categories of industrial dischargers. Bioaccumulation data and modeling are used in support of this effort.

In response to the Water Resources Development Act (WRDA) of 1992 requirement that EPA conduct a national survey of data regarding sediment quality in the United States, OST prepared the National Sediment Quality Survey (NSQS) (USEPA, 1997a). For calculations related to bioaccumulation, the Survey makes use of fish tissue residue data, and models bioaccumulation from sediment using the theoretical bioaccumulation potential approach. A national database containing information in the NSQS, i.e., the National Sediment Inventory, will be maintained and updated on a regular basis so that it can be used to assess trends in both sediment quality and the effectiveness of existing regulatory programs at the Federal, State, and local levels.

Section 403 of the CWA requires determination of the quantities of and potential for bioaccumulation of released chemicals, the potential for pollutant transport, potential harm to biological communities, and direct and indirect effects on humans. The "*CWA Section 403: Procedural and Monitoring Guidance*" (USEPA, 1994) developed by the Office of Wetlands, Oceans, and Watersheds (OWOW) (within OW) discusses the qualities of target species and methods for assessing bioaccumulation; monitoring program design, including sampling of caged or indigenous indicator species; the type of tissue to be analyzed in invertebrates and fishes; and techniques for extracting and analyzing chemical contaminants. USEPA (1995) provides additional information on some of these topics.

EPA's National Estuary Program (NEP), authorized under CWA Section 320, is a national demonstration program that uses a comprehensive watershed management approach to address water quality and habitat problems in designated estuaries on the Atlantic, Gulf, and Pacific coasts and in the Caribbean. OWOW developed guidance for this program (USEPA, 1992b) which is similar to that for Section 403 (above) and which includes the design and conduct of bioaccumulation monitoring studies to link exposure and effects and to examine risks to target species and humans.

Section 402 of the CWA authorizes the National Pollutant Discharge Elimination System (NPDES) permitting program, administered by the Office of Wastewater Management (OWM) (within OW), to regulate the discharge of pollutants from point sources into navigable waters. Bioaccumulation screening methods can be used to identify chemicals of potential concern in the sediments, followed by chemical-specific analysis for confirmatory purposes. Until the States adopt numeric criteria into their standards for sediment contaminants based on bioaccumulation, the NPDES program would not require permitting authorities to include, in their NPDES permits, sediment bioaccumulation-based numeric limits. However, States have the discretion to include such limits in permits based on an interpretation of their narrative standards for toxics. To establish such permit limits, it will be necessary for permitting authorities to develop Waste Load Allocations (WLAs) for the relevant sediment contaminants.

Section 118(c)(2) of the CWA (Pub. L. 92-500 as amended by the Great Lakes Critical Programs Act of 1990 (CPA), Pub. L. 101-596, November 16, 1990) required EPA to publish proposed and final water quality guidance on minimum water quality standards, antidegradation policies, and implementation procedures for the Great Lakes System. In response to these requirements, EPA developed the Final Water Quality Guidance for the Great Lakes System; Final Rule, 40 CFR part 132; *Federal Register*, Thursday, March 23, 1995. The Guidance incorporates bioaccumulation factors (BAFs) in the derivation of criteria and values to protect human health and wildlife.

Section 118(c)(3) established the Assessment and Remediation of Contaminated Sediments (ARCS) Program to assess the extent of sediment contamination in the Great Lakes and to demonstrate bench- and pilot-scale treatment technologies for contaminated sediment. Under the ARCS program, the Great Lakes National Program Office (GLNPO) used bioaccumulation data and models to estimate comparative human health risks associated with direct and indirect exposures to contaminated sediments in the lower Buffalo River under selected remedial alternatives. It was shown that risks could be reduced under the different remedial alternatives compared to no action, particularly if dredging was the selected option.

Ongoing work in the State of Washington provides an example of the use of bioaccumulation data to implement a state regulation. Sediment Management Standards (SMS) for the State of Washington were promulgated by the Washington State Department of Ecology under Chapter 173-204 WAC in March 1991. The purpose of these standards is to "reduce and ultimately eliminate adverse effects on biological resources and significant human health threats" resulting from contaminated sediments. The State of Washington is developing human health sediment quality criteria for bioaccumulative compounds in Puget Sound sediments which will be incorporated into the State's existing SMS. These criteria (not to be confused with Sediment Quality Criteria for the Protection of Benthic Organisms proposed by EPA in the *Federal Register* in 1994) are based on standard risk assessment methodologies in conjunction with empirically derived biota-sediment accumulation factors (BSAFs).

Important Issues Involved in Generating and Interpreting Bioaccumulation Data

The following general and specific issues should be addressed before agencies can effectively consider bioaccumulative compounds when developing guidance on

sediment contamination. Each of these issues will be addressed in the "status and needs" paper.

General Issues

- What are the assumptions, applications, and limitations for each bioaccumulation methodology?
- What are the major uncertainties related to the assessment of bioaccumulation of sediment-associated contaminants?
- Do these uncertainties affect regulatory decisions? Will they be resolvable in the near term or will they require a much longer period for resolution?
- How can bioaccumulation assessment results be effectively applied to human health and ecological risk assessments?

Specific Issues

- What are the most appropriate definitions of terms related to bioaccumulation?
- What are the requirements for selecting species for bioaccumulation testing?
- What species are potentially available for use in testing?
- What are the most appropriate methods for testing bioaccumulation?
- Are there alternative tests that can be considered for assessing bioaccumulation?
- How can tissue-specific residue levels be coupled with chronic toxicity response data to develop dose-response relationships for bioaccumulative contaminants?
- How can bioaccumulation methods be used to assess population level effects (i.e., in order to allow for regulatory cost-benefit analysis)?
- How should we account for the bioaccumulation of metabolites of contaminants, such as PAHs?
- When should theoretical models be used rather than testing to assess bioaccumulation?
- How much site-specific information is required to apply models to predict bioaccumulation?
- What model parameters are more broadly applicable rather than site-specific?
- What bioaccumulation model components are essential for food chain modeling?
- Is contaminant partitioning behavior related to biomagnification?
- Can log K_{ow} help determine the trophic level at greatest risk from bioaccumulation of specific sediment contaminants?
- How should we account for differential partitioning of bioaccumulative contaminants among tissues?
- Do steady-state equilibrium model assumptions represent prevailing conditions in the long term for risk assessment purposes?
- What are the most sensitive exposure parameters that drive the outcome of human health and ecological risk assessments?
- How are different programs using bioaccumulation data and what do they need from the data to address their program responsibilities?

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Some Major Issues Involved in Generating and Interpreting Bioaccumulation Data

- Laboratory vs. field methods of assessing bioaccumulation
- Feeding during testing
- Uncertainties in test procedures
- Lack of test organisms
- Relationship between contaminant body burdens and adverse ecological effects
- Relationship between bioassay organisms and procedures and natural populations
- Relationship between bioassay organisms and human health

Some Major Issues Involved in Generating and Interpreting Bioaccumulation Data (cont.)

- Need to take home range/foraging area into account when estimating exposure concentrations
- Weight to give to various effects endpoints
- Presence of bioaccumulative chemicals in sediments may pose risks to aquatic life, wildlife, and humans
- Overall guidance on interpretation of bioaccumulation data in the evaluation of ecological and human health effects is lacking
- To begin to address this concern, EPA formed a Bioaccumulation Analysis Workgroup consisting of 40 Headquarters and regional participants

Bioaccumulation Analysis Workgroup is Overseeing the Production of:

- "Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment: Status and Needs"

Document will:

- Provide background information and summarize current research that might improve our ability to use bioaccumulation data to evaluate sediment quality
- Report on the status of bioaccumulation testing and interpretation in various EPA programs for the purpose of sediment quality assessment

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APPENDIX A. Chemical-specific Summaries of Bioaccumulation Information

Biaccumulation Summary Cadmium

Chemical Category: METAL (Divalent)

Chemical Name (Common Synonyms): CADMIUM **CASRN:** 7440-43-9

Chemical Characteristics

Solubility in Water: **Half-Life:**

Log K_{ow}: **Log K_{oc}:**

Human Health

Oral RfD: **Confidence:**

Critical Effect:

Oral Slope Factor: **Carcinogenic Classification:**

Wildlife

Partitioning Factors:

Food Chain Multipliers:

Aquatic Organisms

Partitioning Factors:

Food Chain Multipliers:

Toxicity/Bioaccumulation Assessment Profile

Summary of Biological Effects Tissue Concentrations for Tributyltin

Species:	Concentration, Units in:			Toxicity:	Ability to Accumulate[1]:			Source:	
Taxa	Sediment	Pore Water	Tissue (Sample Type)	Effects	BCF	BAF	BSAF	Reference	Comments[2]
Invertebrates									
Amphipod *Hyalella azteca*		4.8 nM[3]	110 nM/g dw (whole body)	4 week LC_{50}				[24]	L; 1 week to reach equilibrium in tissue
Polychaete *Nereis diversicolor*	455 ± 83 ng/g dw (n=5)	68.2 ± 40.6[3] ng/L (n=8)	479 ± 249 ng/g dw (pooled, whole body) (n=5)		7,023			[23]	F
Polychaete *Neanthes arenaceodentata*		100 ng/L[3]	6.27 µg/g dw (whole body)	Reduced growth and reproduction				[25]	L; 4
Polychaete *Neanthes arenaceodentata*		50 ng/L[3]	<3.0 µg/g dw (whole body)	No significant effect on survival, growth, or reproduction				[25]	L; 4
Polychaete *Neanthes arenaceodentata*		500 ng/L[3]	16.81 µg/g dw (whole body)	Significant effect on survival				[25]	L; 4
Gastropod mollusk (Common winkle) *Littorina littorea*	455 ± 83 ng/g dw (n=5)	68.2 ± 40.6[3] ng/L (n=8)	1009 ± 428 ng/g dw (pooled, whole body) (n=4)		14,795			[23]	F
Bivalve mollusk (American piddock) *Petricola pholadiformis*	455 ± 83 ng/g dw (n=5)	68.2 ± 40.6[3] ng/L (n=8)	838 ± 108 ng/g dw (pooled, whole body) (n=2)		12,287			[23]	F
Bivalve mollusk (Clam) *Scrobicularia plana*	455 ± 83 ng/g dw (n=5)	68.2 ± 40.6[3] ng/L (n=8)	3375 ± 232 ng/g dw (pooled, whole body) (n=4)		49,487			[23]	F

Summary of Agencies' Uses of Bioaccumulation Data for the Interpretation of Sediment Quality

Agency Mission and Mandates		Components of the Contaminated Sediment Management Strategy					
Program	Relevant Statutes	Research	Assessment	Remediation	Dredged Material Management	Prevention	Outreach
USEPA, Office of Enforcement and Compliance Assurance							
Office of Compliance						●	
Office of Federal Activities	CWA §404, NEPA §102, CAA §309, MPRSA §102 &§103, River & Harbors Act		●	●	●	●	●
USEPA, Office of Prevention, Pesticides, and Toxic Substances							
Office of Pesticide Programs	FIFRA	●	●			●	
Office of Pollution Prevention and Toxics	TSCA	●	●			●	
USEPA, Office of Research and Development							
National Health and Environmental Effects Research Laboratory							
Atlantic Ecology Div. (Narragansett)		●	●			●	●
Mid-Continent Ecology Div. (Duluth)		●	●			●	●

Session Seven: Panel Presentations

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Bioaccumulation Results and Decision-Making: The Superfund Program

James Pendergast
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Sediment Bioaccumulation—A National Pollutant Discharge Elimination System (NPDES) Program Perspective

Thomas Murray
U.S. EPA, Office of Pollution Prevention and Toxics,
Washington, DC
Integrating Bioaccumulative Results into EPA's Decision-Making Process

Maurice Zeeman
U.S. EPA, Office of Pollution Prevention and Toxics,
Washington, DC
U.S. EPA/OPPT and Sediments: Screening New and Existing Chemicals for Potential Environmental Effects

Craig Vogt
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Washington, DC
Dredged Material Management Program

Mario Del Vicario
U.S. EPA, Region 2,
New York, New York
Dredged Material Management: A Regional Perspective

Bioaccumulation Results and Decision-Making: The Superfund Program

Lawrence Zaragoza
U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC

I have been following sediment issues with Betsy Southerland and other people at EPA for a number of years. During that time I have been involved with a variety of sediment projects, including the Agency's Contaminated Sediment Management Strategy and sediment quality criteria. It has been an interesting experience. What I will try to do for the next several minutes is provide you with a reflection of my experience in working through these issues with the Office of Water and with the EPA Regions.

I am sure that many of you are familiar with the Superfund Program. The Superfund Program is designed to protect both public health and the environment, so we are interested in looking holistically at environmental issues for all media. This means we are concerned with air, water, and soil. In dealing with the environmental issues at Superfund sites, we try to be consistent in our approach to assessment of risks and cleanups. We also seek to be consistent with the activities of other program offices. That is one of the reasons why we look to the Office of Water to provide direction for the type of approaches we should apply for contaminated sediments.

The places where we tend to do some innovative things are generally areas that bridge between different program media. We especially need to consider how to focus between different program media and how to make sure that we are looking at everything in a consistent manner. Coming to the decision about what is actually going to be protective can also be challenging. You have to look at what happens under different programs with different statutes, and it can be difficult to reconcile issues from a technical perspective that have different legislative histories and requirements.

Superfund reauthorization has been a subject of discussion over the last couple of years. One of the most striking points that I have heard expressed at some of the reauthorization meetings is that Superfund is a program that provides significant cleanup leverage for other cleanup programs. We are told that the EPA's efforts complement state efforts and that EPA work is especially useful on some of the larger and more difficult sites.

Superfund reauthorization activities have been going on for several years. Bills have been drafted that did not make it through the political process. We hope to see a reauthorization bill passed in the next year.

What about bioaccumulation data? If we are really seeking to effectively assess risk to public health in the environment, we should be looking at all the available information to assist us in making better decisions. This would include bioaccumulation information, bioavailability data, information on exposure, and information on routes of exposure to either humans or sensitive species. One of the activities that we are encouraged to see moving forward is the work that Mike Kravitz in the Office of Science and Technology is undertaking to assess the different methodologies that exist for measuring bioaccumulation and to build scientific consensus on these methods so they can be used more broadly. Right now, each program is usually faced with having to synthesize that kind of information on a case-by-case basis.

I would like to briefly describe Superfund's hazard ranking system (HRS), which is the primary tool we use to screen sites for placement on the National Priorities List (NPL). The HRS was revised and published as a Federal Register Notice on December 14, 1990. We track the number of sites in the Superfund universe through a database called CERCLIS. Last year, we removed 28,000 sites from CERCLIS, which left about 13,000 sites in the database. As of June, 1997, we have further reduced the number of sites in CERCLIS to 10,735. These steps have helped to identify those sites that need additional assessment and possible cleanup. Within the CERCLIS universe, the NPL represents those sites that are likely to require long-term cleanup efforts. Over 1,200 sites are on the NPL. These sites tend to be complex and costly to clean up and many include sediment contamination. The HRS screening process for these sites relies on readily available information, because we want the process to be

cost effective in identifying sites that warrant further attention. Sites undergo an in-depth evaluation once they are listed. Bioaccumulation is included in this evalution, but only the surface water exposure pathway is evaluated for humans and other sensitive species.

Under the HRS, three factors are examined for contaminants at a site. We use bioconcentration data to determine whether contaminants accumulate up food chains. We look at water solubility data, particularly the logarithm of the octanol-water partition coefficient K_{ow}, to consider the potential for and consequences of chemical partitioning. We also distinguish between freshwater and saltwater environments and the resulting impacts on the receptors (humans or other sensitive organisms) being evaluated. Since the budget for an HRS evaluation is very limited, the data must be collected over a relatively short period of time.

Risk assessments for Superfund sites involve a longer and more costly process than a HRS evaluation. Depending on the site, a risk assessment can cost a few to several hundred thousand dollars. Human health risk assessments are conducted more frequently than ecological risk assessments. The risk assessment stage involves more extensive data collection and detailed analysis of the data. Results from standard bioassays and other assessment techniques discussed at this conference are used at this stage of analysis. These comprehensive studies are performed at NPL sites and other areas to assess the impact of contaminated materials on humans and other sensitive species.

Finally, I would like to mention a Government Accounting Office (GAO) study that was conducted a couple of years ago. The report included a summary of results that we got on risk information for a number of different pathways. We found it interesting for sediments which drove the cleanup for 14 out of about 200 sites that were evaluated. Traditionally, the Superfund Program has been driven by impacts to ground water. I definitely see some changes that have taken place in scoring sites using the revised HRS, which allows for more equal consideration of risks among all media. We are also trying to be much more comprehensive about evaluating all the potential risks from all the media. Some of the bases for cleanups at Superfund sites include maximum contaminant levels (MCLs), MCL goals, Resource Conservation and Recovery Act (RCRA) risk levels, and state standards. In some instances, state standards are more restrictive than federal criteria. You may end up being more protective in some states than you would be in others because of the differences in state standards. These are all issues that will be debated significantly during reauthorization. The costs and complications of dealing with sediment problems are often significantly more challenging than many of the other media. This is true not only from a risk assessment perspective, but also from the perspective of what to do with the material once you determine that it is a problem.

What is Superfund?

- Protect health and the environment as outlined in the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) as amended by the Superfund Amendment and Reauthorization Act of 1986 (SARA)
- To be consistent with the legislative direction and the policies of other EPA offices (e.g., the National Air Quality Standard for lead), OSWER lead policy has targeted protection
- Available scientific information to establish protective levels plays a key role in policy formulation
- Superfund complements State cleanup programs and has been reported to provide an incentive for responsible parties to clean up environmental problems
- Superfund is in the process of reauthorization—we hope within the next year

What is Needed to Use Bioaccumulation Data?

- The Superfund Program seeks to effectively assess risks to health and the environment
- Costs and feasibility may preclude some assessments for bioaccumulation or risks
- The expectation is that Superfund will seek to improve risk assessments and the confidence that can be placed in risk assessment results

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How Does the Superfund Program Use Bioaccumulation Data?

- Protect health and the environment as outlined in the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) as amended by the Superfund Amendment and Reauthorization Act of 1986 (SARA)
- Superfund is in the process of reauthorization—we hope within the next year

Assessing Bioaccumulation within the Hazard Ranking System

- The Hazard Ranking System (HRS) is a screening tool
- The HRS is the primary tool for supporting the addition of sites to the National Priorities List (NPL)
- The HRS employs readily available information and information that can be collected
- Bioaccumulation is evaluated in the human food chain treat and environmental threats within the Surface Water Pathway

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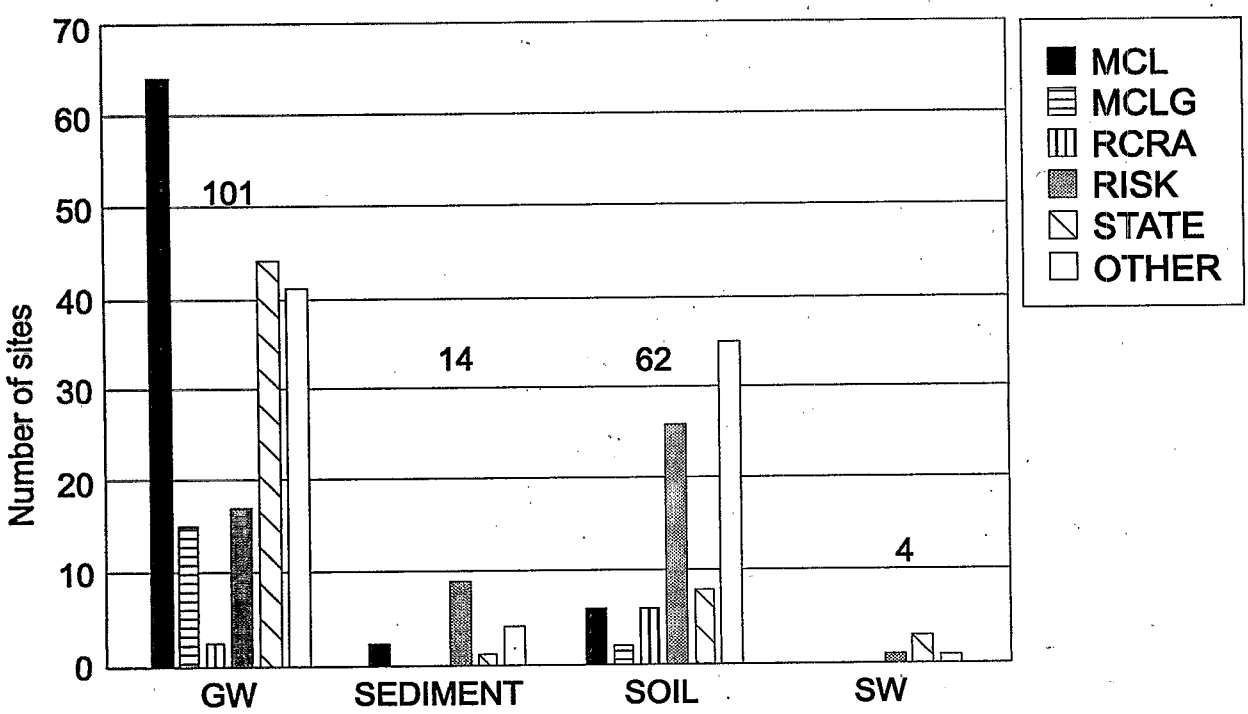
A Tiered System is Used to Determine Bioaccumulation Potential

- Logarithm of the n-octanol-water partition coefficient (log K_{OW})
- Water solubility data
- Direction is provided to distinguish between fresh and salt water

Risk Assessments Provide for a Detailed Examination

- Bioassays and site specific Sediment Quality Criteria are assessed
- The types of standard bioassays employed by others support risk assessments

SUMMARY OF THE BASIS FOR CLEANUP



Sediment Bioaccumulation—A National Pollutant Discharge Elimination System (NPDES) Program Perspective

James Pendergast
U.S. Environmental Protection Agency, Office of Wastewater Management, Washington, DC

I am going to talk briefly about what the National Pollutant Discharge Elimination System (NPDES) Program does in the area of sediment and bioaccumulation and what we think we are going to be doing in the future. For all of you who have dealt with the NPDES Program, you may believe that half of the program staff are engineers and the other half are attorneys. You could probably ask why these people are talking about bioaccumulation and whether they know enough to talk about it in the first place. We would respond that bioaccumulation is of interest to us both in terms of setting permit limitations and collecting information that may be needed for making future watershed assessments.

I am going to spend most of my time talking about both of these, but will preface my talk with a couple important points. One is that what I will be talking about is going to be in the future. And the future does not mean tomorrow and maybe not even next year. In a couple of instances, I will be referring to what is occurring now, but in most cases I will be talking about activities in the 5- to 10-year horizon. That is when the majority of permits may have to be dealing with these considerations where necessary. The second is a point of clarification that, in contrast to the Superfund Program which deals with remediation, the NPDES Program deals with discharges today. By that I mean we take care of current point source pollutant discharges in the waters of the United States. For example, a facility that had discharged high levels of pollutants in their wastewater years ago may be discharging very good quality wastewater today. We regulate their discharge today without taking into account discharges that polluted the environment 10 to 20 years ago.

In the ideal surface water protection program, water quality criteria are first developed by a combination of Agency research and program staff. The Office of Science and Technology's (OST's) Health and Ecological Criteria Division (HECD) is responsible for criteria development. States then use that information to adopt water quality standards. States are assisted in this process by the EPA Regions and program staff in OST's Standards and Applied Science Division (SASD). The next step is to take that information and devise a total maximum daily load (TMDL) and a waste load allocation (WLA) using water quality models. That process will produce a number in terms of how many pounds of a pollutant a facility can discharge without exceeding water quality standards. Finally, the NPDES permit writer incorporates that information into the terms of the permit to be issued.

The problem with the ideal world is that it is so seldom realized. In the last 22 years of my experience, the three steps prior to issuing permits are not always there. For example, we do not have water quality criteria for every pollutant. We especially do not have criteria for all the pollutants that bioaccumulate. We may have some raw data and results from some good studies conducted at universities, but we do not have criteria for every contaminant. Even where we have the criteria, the states may not have adopted a numeric water quality standard that includes all those criteria. There is a catch-all phrase in water quality standards called a narrative standard. It is worded something like this: "There should be no toxics in toxic amounts." This gives the states a standard that can be used, but it puts the burden on someone to figure out what "toxic" and "toxic amounts" mean.

Not many waste load allocations have been developed for the bioconcentratable, bioaccumulative types of pollutants. This means that permit writers now have to figure out what to do. All of a sudden, they have to become someone who knows water quality criteria, who knows how things bioconcentrate, and who knows the chemistry and biology behind that. They must also become a mathematical modeler and figure out how pollutants cycle through the environment. And, of course, they do not usually have the training for that, except for a 5-day course that the Permits Division conducts. But nevertheless, that is the type of information they must learn to integrate.

Let us consider how sediments fit into the permitting process. In dealing with sediments, we look at bioaccumulation data when we try to interpret the narrative for those "no toxics in toxic amounts" standards.

If you have sediment quality criteria, and you can figure out the fate and transport, you can walk through the same process of interpreting nontoxic and toxic amounts, figuring out the cause-effect relationships, and determining which facilities are discharging which pollutants that are causing problems in the sediments. But you also have to recognize that when you are dealing with the impact of discharges on sediments, you are having to take into consideration a number of different factors from discharges into water. For sediments, you have to consider important factors like the physical composition of the sediment and chemical interactions with sediments. If you find a hot spot below a facility, you also need to determine whether it is due to the facility's discharge or to something that happened 20 years ago that just got washed downstream in the last flood. Bioaccumulative pollutants in sediments may not always originate from point source discharges. They may originate from agricultural uses where there is runoff and erosion from farms. This is another factor that the permit writer must consider.

When we look at the information we have available to address sediments today in the NPDES Program, we have some challenges or barriers to overcome. No official sediment criteria have been published, although we have five proposed criteria. We do not find TMDLs that deal with sediments because you have to have sediment criteria to develop them. It is a difficult challenge to factor in unquantified nonpoint source contributions. It is also difficult to accurately define fate and transport when you deal with physical things that scour during flooding events. The calculations done by the typical NPDES permit writer are steady-state calculations, and storm events do not fit into a steady-state calculation. So what do we do?

Right now we are focusing more on the aqueous part of the equation. We are not doing much with sediments in our program today. On the aqueous side, we have put out some guidance on how to deal with that "no toxics in toxic amounts" phrase. We have two sources of information that we refer people to. One is a technical support document produced in 1991 for water quality-based toxics control. In there, we have laid out the equation similar to the water quality criteria program approach where they are primarily protecting human health from adverse affects for either a cancer or noncancer endpoint. We start out with how many ounces of fish a person can eat a year, the typical weight of an American who eats that fish, etc. From that information, you can determine what a water quality standard should look like. The technical support document also talks about a bioconcentration factor. Since that was 1991, we know the information needs to be updated. In the intervening years, we have been working on how to bring bioaccumulation into that evaluation process.

Through some work with our Office of Research and Development, we have worked out how to estimate or calculate bioaccumulation factors that can be added to the equation. The best source of that information today is the Great Lakes Initiative (GLI) rule. The rule only applies to the Great Lakes basin, but the science in it is universal and can be used across the country. So, this is the type of guidance that we point people to when they want to use bioaccumulation in making assessments in the NPDES Program. There is also some discussion in the GLI rule about what to do with fish tissue data. If you have nothing in your aqueous phase chemistry, but you find fish tissue that is high in a certain pollutant, can you make regulatory decisions based on the fish tissue data? The GLI rule shows how to translate the fish tissue data and determine what its equivalent water quality concentration would be.

That is where we are today. We are trying to work out a companion process for dealing with sediments in developing permits. Some of the focus is on simplifying modeling that currently works on a large mainframe computer to an assessment tool that an average permit writer can use. This could take five to ten years. Another important area of focus is to conduct evaluations on a watershed scale rather than the traditional approach of evaluating each individual facility. The NPDES Program was initiated under the Clean Water Act of 1972 to permit facilities to reduce pollutants. At this time, it was easier to identify major sources of pollutants. The program targeted the most obvious sources of pollution from the big pipes or from areas with signs of pollution like foam on the rivers. The program was successful in getting treatment technology established at plants to combat pollution. Now the program is in a more difficult stage. Every two years the states are required to provide assessments of their water quality that are compiled in the 305(b) reports. Lately, they are indicating that nontraditional sources are causing more problems. These are either urban point sources such as discharges from a storm sewer or a combined sewer overflow (CSO) or nonpoint source runoff from agricultural lands or forestry sites. Now we are having to open our doors for the first time and begin talking to programs in the Bureau of Land Management and the Forest Service to decide how we are going to apply a holistic approach in protecting our waters. Working with programs that are entirely voluntary to develop responsible water quality management is perhaps our biggest challenge. NPDES is still part of the equation, but no longer the primary part of the equation. Sediments come into play in this process. We need to identify watersheds where sediment contamination is the greatest source of impairment and to work with other agencies to deal with the sources of sediment problems. And we may find out that, in some cases, NPDES may not be the answer.

What do we expect today? We still expect that the permit writers use their best judgement on how to deal with bioaccumulation and sediments. And best judgement does not mean to find an ounce of science and start leaping miles ahead of that. What best judgement means is taking the current science, taking the current data, and using it responsibly to take a look at watershed-based water management decisions. If in looking at this information, a permit writer determines that a point source is not the cause of the problem, then their best professional judgement will say, "Don't deal with it." But if the permit writer finds that a point source is discharging a

bioconcentratable or bioaccumulative pollutant at levels causing impairment today, then they should use the information to include the right requirement in the permit to limit that pollutant.

We do believe that NPDES permits still need to be issued. We do not want to spend 25 years studying issues before we can take any action. But we also recognize that issuing permits on time does not necessarily mean every 5 years regardless of the situation. Issuing permits on a timely basis means doing it when you have enough data that it makes good sense. And that we will do.

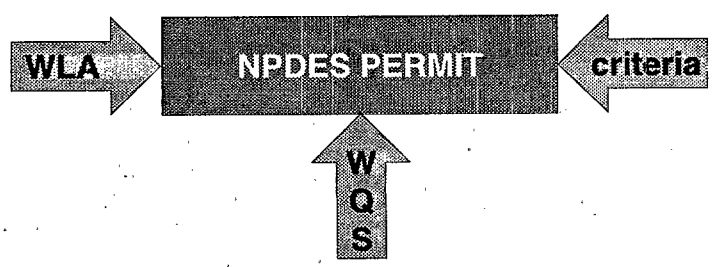
WHY IS A NPDES PROGRAM MANAGER TALKING ABOUT SEDIMENT BIOACCUMULATION?

- ⇨ Is sediment bioaccumulation a permitting issue?
- ⇨ Is sediment bioaccumulation strictly a monitoring issue?

PERMIT ISSUANCE (WITHOUT SEDIMENT CONTAMINATION FACTORED IN)

Permit Writing Factors:

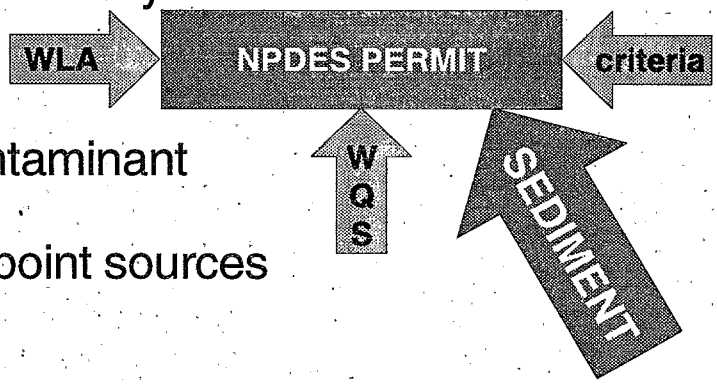
- Develop criteria
- Adopt Water Quality Standards (WQS)
- Develop Waste Load Allocation (WLA)
- Issue permits, based on the WLA, for each point source



PERMIT ISSUANCE (WITH SEDIMENT CONTAMINATION FACTORED IN)

Sediment Characteristics to Consider:

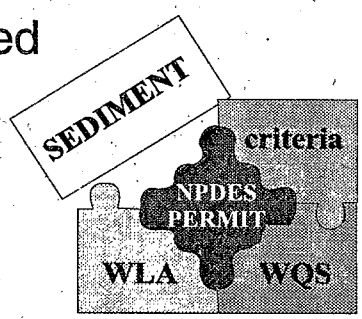
- Different sediment types (physical and chemical composition)
- Sediments are not stationary (Fate/Tranport Issue)
- Sediments may have pre-existing contaminant contributions from: natural, point or non-point sources



PERMIT WRITER'S CHALLENGE

Permit Writer's Challenges When Developing a Permit with Sediment Contamination Factored in:

- No sediment criteria, therefore no WQS for sediment
- No total maximum daily loads (TMDLs)
- Non-point source contributions not factored in
- Transport-Fate issues are not addressed for sediments

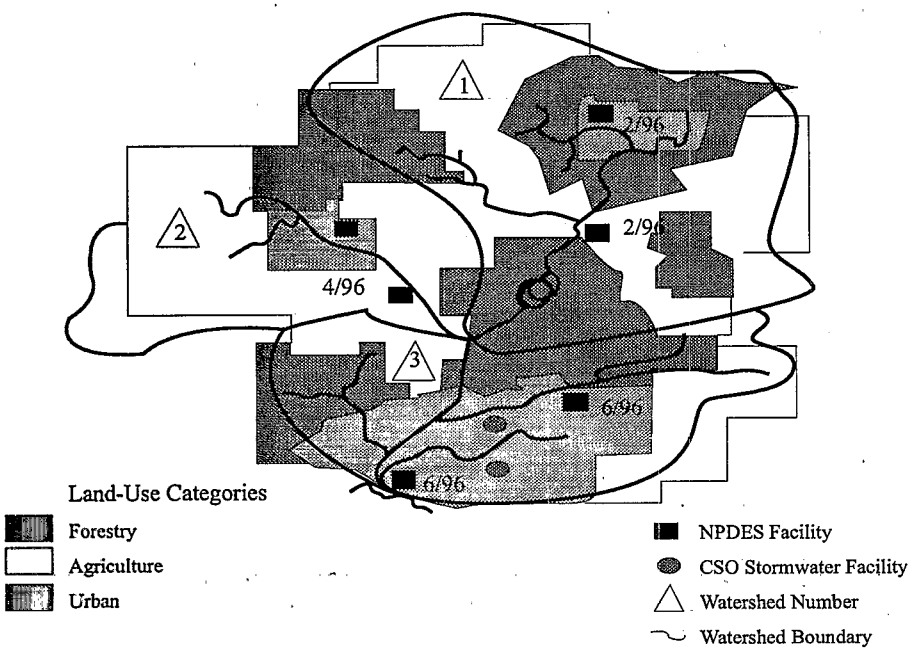


ADDRESSING THE CHALLENGES

What's Available:

- National Guidance
  - Technical Support Document (TSD) (pages 36-44)
  - Great Lakes Initiative (GLI) (pages 15400-15406)
- Watershed Permitting Approach

PERMITTING BY WATERSHEDS



WHAT IS EXPECTED FROM EPA, STATE, AND TRIBAL PERMIT WRITERS?

Minimum:

- Using all available EPA permitting guidance, issue permits based on Best Professional Judgement (BPJ).
- Issue NPDES Permits on time.

Integrating Bioaccumulative Results into EPA's Decision-Making Process

Thomas Murray
U.S. Environmental Protection Agency, Office of Pollution Prevention and Toxics, Washington, DC

Premanufacture Notifications (PMNs) are required under section 5 of the Toxic Substances Control Act (TSCA) from anyone who intends to manufacture or import a new chemical substance. A new substance is defined as one not on the Chemical Inventory created under TSCA section 9. Once the U.S. Environmental Protection Agency (EPA) receives a PMN document, Agency staff normally have 90 days to review it. First, the PMN substance is sent through an initial screening-level risk assessment process. A major objective of this review process is to determine whether the PMN substance may present an unreasonable risk of injury to human health or the environment during its lifetime.

The Office of Pollution Prevention and Toxics (OPPT) uses a similar process for those chemicals already in the marketplace. Priorities are established for selecting existing chemicals from the chemical inventory for review. There is no statutorily imposed time schedule for these reviews.

Coming out of the screening-level risk assessment process, those chemicals which are found to present the potential for unreasonable risk to human health or the environment are scrutinized more thoroughly. OPPT sometimes requires testing in response to identified concerns before reaching a decision for a chemical under review.

Professionals from many different disciplines take part in reviewing the various aspects of the life cycle of a chemical under review. In his remarks, Maurice Zeeman describes those things considered when assessing risk to ecological communities. My remarks will focus on the human health exposures.

When a chemical is introduced into OPPT's risk screening process, a Structure Activity Team convenes to discuss its potential health and ecotoxicity concerns. This Team provides direction to the review process that follows. Of concern to us in this discussion are those chemicals which result in human exposures through surface water. Here, OPPT engineers quantify surface water releases of these chemicals from point sources or estimate these releases where monitoring data are absent. We do not routinely assess nonpoint sources. OPPT chemists then assess the fate of these chemicals and describe likely or known transport processes. Assessors then use these fate estimates to assess the potential human health risks associated with the expected releases.

The process of evaluating each potential route of exposure can be complicated. To perform the calculations necessary for exposure evaluations, assessors make use of simple formulas, complex mathematical models, and a number of basic assumptions. Sound monitoring data are always preferred but are rarely available. Much of the work involved in the assessment process involves frequently encountered exposure scenarios. This has allowed OPPT to standardize most of the assessment process. A number of thresholds have been developed which help determine whether a given chemical or exposure route will present significant potential exposures. OPPT relies heavily on the efforts of other EPA program offices and academia in updating its exposure assumptions and models.

Sediment/Bioaccumulation Results

When a chemical under review is believed to have the potential to pose a risk to human health through the surface water pathway, several exposure assessment steps are set in motion. First, a fate review is done to determine, among other things, how much of the chemical is likely to be removed in the wastewater treatment process. With an estimate of chemical loading to the receiving stream, instream processes (hydrolysis, photolysis, biodegradation, etc.) are explored to assess the ultimate fate of the chemical. Exposure models are then deployed to track the chemical downstream to drinking water facilities. The fate chemists will also calculate the Bioconcentration Factor (BCF) for the chemical if exposure through fish ingestion is indicated. Potential Dose Rates are calculated through this pathway only when the log BCF is greater than 2. When the BCF is less than 100 (or the log BCF is less than 2), the resulting human exposure is considered negligible. If the log K_{ow} is greater than 8, human exposure is not calculated. Values for log K_{ow} that are higher than 8 are considered

unreliable; therefore, the BCF estimate is deemed unreliable and is not used.

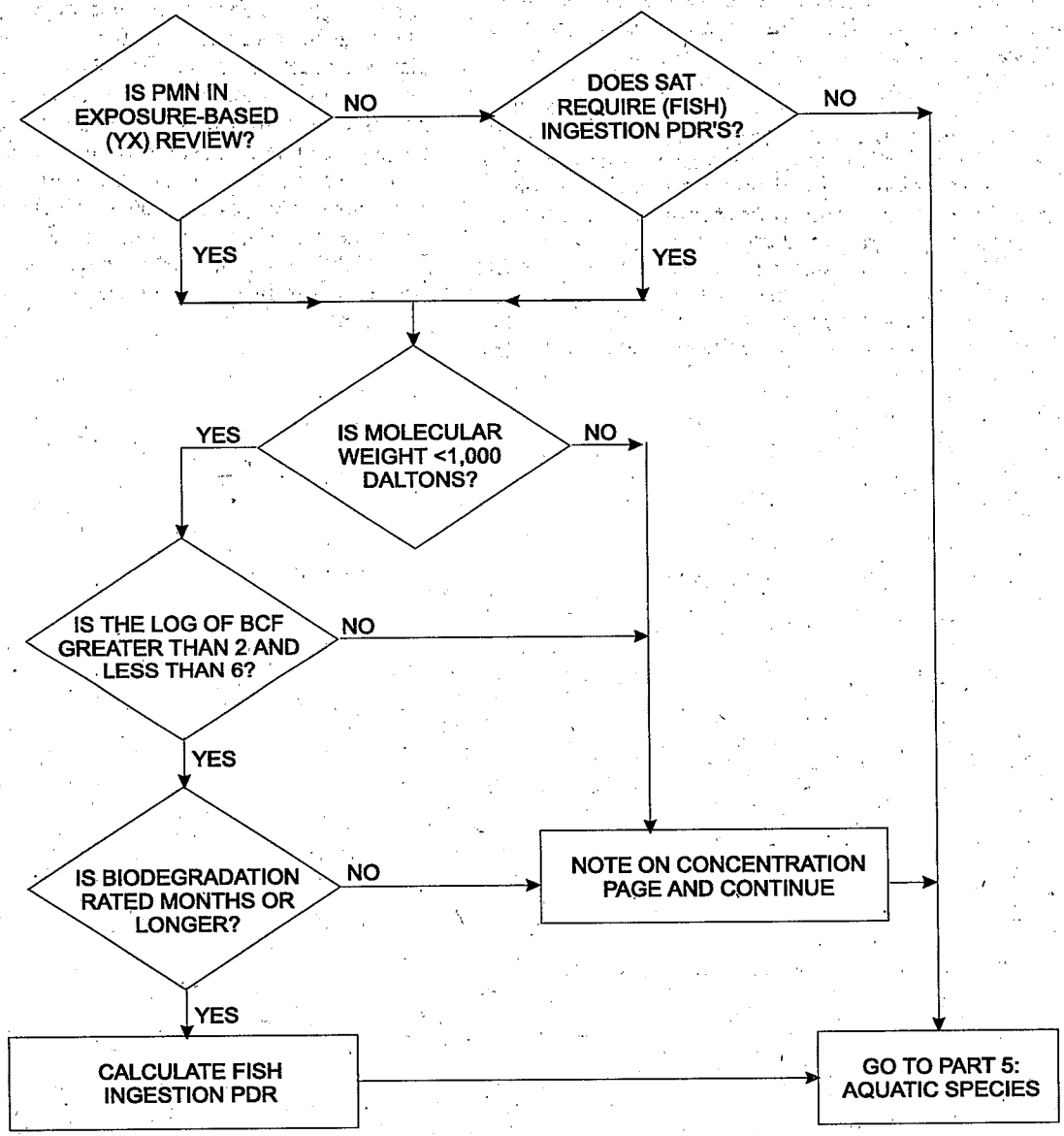
OPPT has also determined that in addition to having the potential to bioaccumulate, the chemical must also remain in the environment long enough to be absorbed into fish tissue. For our purposes, degradation of the chemical must be estimated to take months or longer to allow uptake by fish tissue to occur. Potential dose rates (PDR, a measure of exposure) for fish ingestion are calculated using the stream concentration of the chemical at mean flow (see pages 7-33 and 7-34). For discharges to saline waters, chronic mixing zone dilution factors are used, when available. OPPT currently assumes a fish ingestion rate of 16.9 grams per day. To be consistent with the most recent draft of the *Exposure Factors Handbook*, we will adjust this rate. For subsistence fishers a rate of 140 grams per day can be used; for sport fishers 39 grams per day is reasonable.

Where our calculations indicate that potential exists for unreasonable risk to humans through fish ingestion, we can reach several decisions. Oftentimes, we will require testing by the industry to refine our fate estimates. Dialogue with industry is routinely employed and frequently leads to other refinements in our assessment process. In extreme cases where sufficient risk is anticipated and uncertainty low, TSCA allows OPPT to initiate actions to limit or ban the production or use of a chemical.

Most recently, where potential risk is indicated, we engage industry in discussions designed to encourage voluntary efforts to reduce or eliminate the risk. These range from suggestions to recycle waste or seek disposal practices other than wastewater discharge to intensive efforts to work with industry to redesign their processes to eliminate problems upstream in the process through chemical substitution or process change.

As science advances our understanding of the fate and transport processes of chemicals in the environment and as new testing protocols are developed and standardized, OPPT will embrace these new developments and, where appropriate, will update its assessment process accordingly.

WATER RELEASE CYCLE
PART 4: FISH INGESTION PDR'S



NOTES:

Calculate fish ingestion PDRs for releases to saline, tidal, or "still" waters using chronic mixing zone dilution factors when available.

$$PDR_{fi} = C_{sw} * ING_{fis} * BCF * FQ * 1E^{-6}$$

where

C_{sw} = concentration in surface water (ug/l)

ING_{fis} = ingestion rate

BCF = As calculated

FQ = frequency of exposure (days/year)

Decisions

- Testing
- Dialogue
- Voluntary Actions
- TSCA Regulatory Actions

U.S. EPA/OPPT and Sediments: Screening New and Existing Chemicals for Potential Environmental Effects

Maurice Zeeman, Jerry Smrchek, Joseph Nabholz, and Donald Rodier
U.S. Environmental Protection Agency, Office of Pollution Prevention and Toxics, Washington, DC

Introduction

Since 1976, the Office of Pollution Prevention and Toxics (OPPT, formerly the Office of Toxic Substances, OTS) at the U.S. Environmental Protection Agency (U.S. EPA), has been the office that is responsible for assessing new and existing industrial chemicals. Prior to a reorganization within OPPT in 1997, the Environmental Effects Branch was the principal ecological group within that office. In the past, that group had been concerned primarily about the ecological effects of industrial chemicals in the water column (Zeeman and Gilford, 1993; Zeeman, 1995). However, in the last five to ten years, there has been more and more concern expressed about the possible effects of industrial chemicals upon organisms in the sediments (Clements et al., 1994; U.S. EPA, 1994; Nabholz et al., in press; Smrchek et al., 1995; Smrchek and Zeeman, 1997; Zeeman, 1993; Zeeman et al., 1995 and in press).

Following is a brief introduction to the U.S. EPA, the responsibilities of OPPT, especially the new and existing chemical programs, and the kind of assessments of these chemicals that are required under the Toxic Substances Control Act (TSCA, the law under which OTS [now OPPT] was formed in 1976). After this is a description of how OPPT went about requiring and getting sediment toxicity testing for a new industrial chemical. And finally, an example is presented on how OPPT developed a methodology that was used to screen thousands of existing chemicals to try to develop a list of higher priority chemicals for possible ecological testing concerns, including possible sediment testing.

Background

For those who think of the U.S. EPA as some sort of monolithic agency, it is not. It has about 17,000 to 18,000 employees in numerous programs and offices that were each set up at different times and for different purposes. The Agency's various programs and offices address responsibilities defined by several different environmental laws that were passed over decades by different legislative actions (Clean Air Act, Clean Water Act, TSCA, etc.). The Offices in the U.S. EPA were organized under broad legislative or environmental categories, such as Air and Radiation, Water, Pesticides and Toxic Substances, Solid Waste and Emergency Response (Superfund), and Research and Development. The figures on page 7-40 [U.S. EPA, OPPT] illustrate how the U.S. EPA is organized, with OPPT residing within the Office of Prevention, Pesticides, and Toxic Substances and the Environmental Effects Branch (EEB) residing within OPPT.

The ecological hazard and risk assessment functions of that Branch used to be in the Health and Environmental Review Division (HERD), and those functions now reside in the newly formed Risk Assessment Division (RAD, formed in 1997 from the combination of the staff and functions of both HERD and the Chemical Screening and Risk Assessment Division, CSRAD). The Economics, Exposure, and Technology Division (EETD) contains the staff responsible for performing the engineering and exposure assessment portions of the ecological risk assessments that are done in OPPT.

Why does/should OPPT care about bioaccumulation, bioconcentration, and sediments? Because there are industrial chemicals which adsorb strongly to sediments and which are quite likely to also bioaccumulate or bioconcentrate. When introduced into the food chain, they could be detrimental not only for those organisms in the sediment and water column environments, but also for humans. TSCA was passed in 1976 to regulate industrial chemicals, those chemicals used in commerce that were not regulated by other statutes, meaning that they are not pesticides, pharmaceuticals, food additives, or substances covered under other laws (see figure on page 7-41 [watering can]). For example, industrial

chemicals are substances like solvents, plastics, dyes, adhesives, detergents, monomers, and polymers. They are chemicals that are widely used in commerce and are around us all of the time. An independent calculation of a partial sum of the 1989 production volumes of industrial chemicals in the United States was around 6 trillion pounds and it is undoubtedly even more now (Zeeman, 1997).

One of the first activities after TSCA was passed in 1976 was to develop an inventory of the industrial chemicals already in commerce in the U.S., the existing chemicals (see figure on page 7-42 [TSCA (Passed 1976)]). After more than two years, over 60,000 chemicals were identified as being in commerce and these became the TSCA Inventory of existing chemicals. These chemicals were basically grandfathered and the Agency was to look at them for environmental concerns on an ongoing basis.

By definition, anything that was not an existing chemical (i.e., on the Inventory) was a new industrial chemical, and if anyone wanted to make it or import it into the U.S., they would have to submit an application for its assessment to the U.S. EPA. This assessment program is not a registration program. Industry (the potential manufacturer or importer) submits a notice to OPPT called a Premanufacture Notification (PMN). OPPT receives a large number of these notices from industry each year (see figure on page 7-42). They need to submit only the data that they have or know about. They do not have to do any testing. OPPT then has 90 days to make a decision based on the available information or provide risk-based reasons for not allowing it to be made or imported.

The number of new chemicals that have gone through this process in OPPT and have become existing chemicals on the TSCA Inventory is currently around 15,000, which brings the Inventory total to about 75,000. OTS/OPPT has received around 30,000 new chemical notices from 1979 to 1996. From 1986 to the present, the average has been about 2,300 new chemical notices per year. That is almost 50 notices per week or 10 per work day that are received and must be assessed in a short time frame.

As mentioned earlier, OPPT does not receive a lot of information. What is required to be submitted with these PMNs? Manufacturers must furnish information such as the name of the chemical, the chemical structure, the amount to be produced, disposal and human exposure information, and any available test data since testing is not required (see figure on page 7-43 [Required Submission]). Nevertheless, OPPT staff must make these risk-based decisions in 90 days or the manufacturer will be able to make or import that chemical unless a risk-based case is made by OPPT that they should not.

The consistent experience of OPPT is that we seldom receive the kind of data that is needed for ecological risk assessment. About 95 percent of the PMNs received by OPPT contain no ecotoxicity or environmental fate data (Zeeman and Gilford, 1993; Zeeman, 1995; Zeeman et al., 1995). The engineers and fate chemists in OPPT only get physical, chemical, or environmental fate data about 1 to 5 percent of the time (Zeeman et al., 1993 and 1995). They seldom get any bioaccumulation or bioconcentration data.

A retrospective was performed on the ecotoxicity data received for new chemicals and it was found that OPPT received ecotoxicity data less than five percent of the time, and the vast proportion of the data received was acute toxicity data (Zeeman, 1995). That means that 19 out of 20 times no ecotoxicity data are provided for new chemicals. Therefore, OPPT has been forced to rely heavily upon a modeling approach that uses structure-activity and quantitative structure-activity relationships (SAR/QSAR, or (Q)SAR) to estimate chemical properties, chemical fate, and the ecotoxicity of a chemical to organisms in the aquatic environment (Clements, 1988 and 1994; Nabholz et al., 1993; Zeeman et al., 1993 and 1995).

How does OPPT make decisions for ecological risk assessment in the new chemical program? A number of PMNs for new chemicals are received each week and they are generally assessed in groups. Within two or three weeks of receipt of a group of PMNs, OPPT holds what is called a FOCUS meeting (see figure on page 7-43 [Flow Chart]). That is the first level of risk assessment where a comparison is made of the hazard assessments, i.e., toxicity profile, and the exposure assessments for each chemical. What is important for ecotoxicity is whether the predicted environmental concentration (PEC, from the exposure assessment) exceeds what is called the concern concentration (CC, hazard assessment value(s) adjusted for uncertainty). In at least 90 percent of the cases, the PEC is not exceeded and a risk is not inferred. These chemicals are dropped from further evaluation. In the remaining five to ten percent of the cases, we proceed to standard review or a request for testing and perhaps through risk management, which can result in iterations to obtain additional data (Wagner et al., 1995).

Sediment Assessment of an OPPT New Chemical

Portions of the following specific new chemical sediment assessment example have been previously presented at a Society of Environmental Toxicology and Chemistry (SETAC) national meeting (Zeeman, 1993), and were also used as one case study of ecological risk assessment methods used by the Agency (U.S. EPA, 1994). Further elaborations of this sediment assessment will also be coming out soon as publications, in both a SETAC book on Uncertainty in Ecological Risk Assessment (Nabholz et al., in press), and in a White House OSTP publication on Ecological Risk Assessments in the Federal Government (Zeeman et al., in press).

The following describes the decision-making process for a specific new chemical where OPPT was able to get aquatic toxicity data, some generic and site-specific data (so predictions could be made on whether or not it would get into the sediments), and some sediment toxicity data. The figure on page 7-44 shows the physical and chemical properties of the chemical. This new chemical would be produced at the level of 100,000 kg per year, so it is not a trivial compound in terms of how much is going to be manufactured or imported. It is an alkylated diphenyl, and was classified into the SAR chemical class of neutral organic chemicals. OPPT is very comfortable in using its structure-activity relationship and quantitative

structure-activity relationship [(Q)SAR] models for making predictions on neutral organic chemicals (Clements, 1988 and 1994; Nabholz et al., 1993; Zeeman et al., 1993).

Based upon the chemical structure, an estimate was made of the K_{ow} value (the octanol to water partition coefficient, often used to estimate both ecotoxicity and bioconcentration ability) and it was fairly high. An estimate of the K_{oc} value (the water to organic carbon partition coefficient, often used to estimate ability to partition to sediments) was also made, and it was relatively high. The water solubility was estimated to be fairly low. These are all strong indications that this chemical will be very likely to distribute selectively from the water column and into the sediments.

A hazard assessment (stressor-response) or toxicity profile for this chemical was prepared (see the figure on page 7-44). OPPT's structure-activity methods (Clements, 1988 and 1994; Nabholz et al., 1993) were used to predict the acute and chronic aquatic toxicity and to make a risk-based case that eventually resulted in test data. From this estimation information, OPPT inferred that this new chemical would be a highly toxic compound on a chronic exposure basis and that it would be very likely to be transported to sediments. A case was subsequently made through refined exposure assessment estimates that sediment toxicity testing was needed, and it was eventually provided (see below).

How well did the OPPT (Q)SAR estimates predict aquatic toxicity to fish and aquatic invertebrates? It was estimated that there would be no effects at saturation for acute toxicities, which was confirmed by the testing. Such short-term tests do not provide a long enough exposure time to observe any acute effects. It was also estimated that the longer term chronic toxicity values would be somewhere in the low parts per billion (ppb) range. The test data subsequently received confirmed that the OPPT (Q)SARs were very accurate with the actual chronic values of 13 ppb for fish and 7 ppb for daphnids being very close to the predicted ecotoxicity results.

OPPT went through a number of iterations to characterize the risk associated with this new chemical (see the figure on page 7-45 [Summary...]). Initially it was thought that fish would be the most sensitive species. This did not turn out to be the case. However, the concern concentration (CC, the level in the water below which OPPT would not take any regulatory action) turned out to be around 1 ppb using either the fish or the aquatic invertebrate toxicity data. A risk-based case was then made through our initial and subsequent iterations assessing exposures. Generic production data and some site-specific data were then obtained to improve the input for our exposure models. The resulting estimates of exposure were contrasted with estimates of benthic toxicity and the estimates of risk to organisms indicated that the manufacturer should consider chronic testing in sediments. The manufacturer did perform a 28-day sediment toxicity test with chironomids, which showed that there would be a moderate level of toxicity.

What risk management decision was made about this new chemical? It was decided that this new chemical could be used only at the three sites for which they provided specific information and that they should limit the water column concentrations of the chemical to 1 ppb or less. OPPT also warned the manufacturer that this chemical would, over the years, be likely to accumulate in sediments.

Existing Chemicals and Sediments

The TSCA Inventory of existing chemicals consists of about 75,000 chemicals that include polymers, mixtures and salts, organometallics, and discrete organic chemicals. A focus on discrete organics was used for an approach that was developed by OPPT for screening possible persistent bioaccumulative chemicals on the TSCA Inventory. Using this approach, thousands of discrete existing chemicals were screened to identify those that might have the chemical characteristics to be persistent and bioaccumulate, and therefore, have potential for concern for selectively partitioning into sediments (Zeeman et al., 1995).

In 1990, OPPT asked the EPA research laboratory in Duluth, MN to screen the TSCA Inventory for discrete organic chemicals which could have log K_{ow} values greater than 3.5. For this screening effort, the Duluth lab created a database and estimated the log K_{ow} (also known as log P). They started the screening process with about 20,000 chemicals and ended it with 6,668 chemicals with log K_{ow} values greater than 3.5 (see figure on page 7-45 [Screening for Persistent Bioaccumulators]). The log K_{ow} value of 3.5 was chosen as a lower limit because it represents a fish bioconcentration factor of about 250, which was derived using an equation from research performed at the Duluth lab (Veith and Kosian, 1982). That limit approximately defines the subset of chemicals that should be cause for moderate or greater concern for bioconcentration in aquatic organisms such as fish.

After this initial screening, OPPT continued with a series of steps to reduce the list of chemicals to a more manageable size. The first step was to do an intersection between the Duluth database and the OPPT Chemical Update System (CUS) database, which includes about 8,600 industrial chemicals that industry reports producing in quantities of more than 10,000 pounds per site per year (see figure on page 7-46). The intersection resulted in the identification of about 1,000 chemicals with this level of production and log K_{ow} values greater than 3.5. The next level of assessment was to consider chemicals with log K_{ow} values between 3.5 (initially 4.3 was used) and 8.0 to focus on those chemicals that had the highest probability of causing the most critical problems. Then persistence was evaluated using a degradation half-life of greater than 30 days as the threshold. This reduced the list to about 80 chemicals. Finally, we considered chemicals that were being produced in quantities greater than 1 million pounds per year. Applying this final criterion, a list of 34 chemicals resulted and these chemicals were examined further because they could be of concern for persistence, bioaccumulation, and partitioning into sediments (Zeeman et al., 1995).

What are the next steps? Obviously, one of the things that would be important is to obtain information on actual environmental loadings. It has been suggested that

OPPT list some of these chemicals on the Toxics Release Inventory (TRI) to find out if those chemicals that were proposed to be only chemical intermediates were actually being released into the environment. When this list of 34 chemicals was presented at a SETAC Workshop on the Environmental Risk Assessment of Organochlorine Chemicals (Clements et al., 1994), one of the chemicals was supposed to be a chemical intermediate that would not be released into the environment. A researcher from Canada came up after the presentation and said that they had been finding this chemical in gull eggs along the Great Lakes.

OPPT has also developed a sequence of tests that persistent bioaccumulative chemicals should be considered for (see figure on page 7-46 [Potential Testing ...]). First, we would like to determine whether or not these chemicals are likely to degrade in a sediment and water system. Chemicals that do not degrade should be evaluated further for bioconcentration in fish. Finally, for the chemicals whose results indicate that they can bioconcentrate, then these chemicals need to be considered for testing to estimate their chronic toxicity to aquatic and sediment dwelling organisms (e.g., see figure on page 7-47 [Preliminary Testing Scheme ...], adapted from Smrchek and Zeeman, 1997).

Discussion and Summary

Sediment issues for both new and existing industrial chemicals are becoming more and more important to OPPT (see figure on page 7-48 [OPPT Sediment Issues]). OPPT has asked for, received, and evaluated the results of sediment toxicity testing for both new and existing industrial chemicals (Smrchek et al., 1995; Smrchek and Zeeman, 1997). It has also been interested in the sediment toxicity test methods being developed nationally through the American Society for Testing and Materials (ASTM) and the U.S. EPA, and internationally through the Organization for Economic Cooperation and Development — the OECD (Ingersoll, 1995; OECD, 1997; Smrchek and Zeeman, 1997). OPPT also continues to have an interest in and support the ongoing refinements in true chronic toxicity testing of sediment organisms, i.e., longer duration chronic testing, such as the 65-day life-cycle test recently developed for *Chironomus tentans* (Benoit et al., 1997).

In summary, sediment issues have been a concern for both OPPT's new and existing industrial chemicals for almost a decade. OPPT receives notices from industry for about 2,300 new chemicals per year. Most of the time there does not appear to be a problem related to sediments. But in those few cases where we are concerned that there could be such problems, OPPT has been able to either get results from sediment toxicity testing, or identify ways to mitigate the concerns for those chemicals. Sediment toxicity testing has also been requested and received for existing chemicals. However, the current focus with existing chemicals is likely to continue the use of screening methods to identify those high production volume chemicals that might have characteristics that result in concerns for partitioning into the sediments and having the potential to adversely affect organisms in the environment.

Disclaimer

This document has been reviewed by the U.S. EPA's Office of Pollution Prevention and Toxics, and approved for publication. Approval neither signifies that the contents necessarily reflect the official views and policies of the Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation of these products for use.

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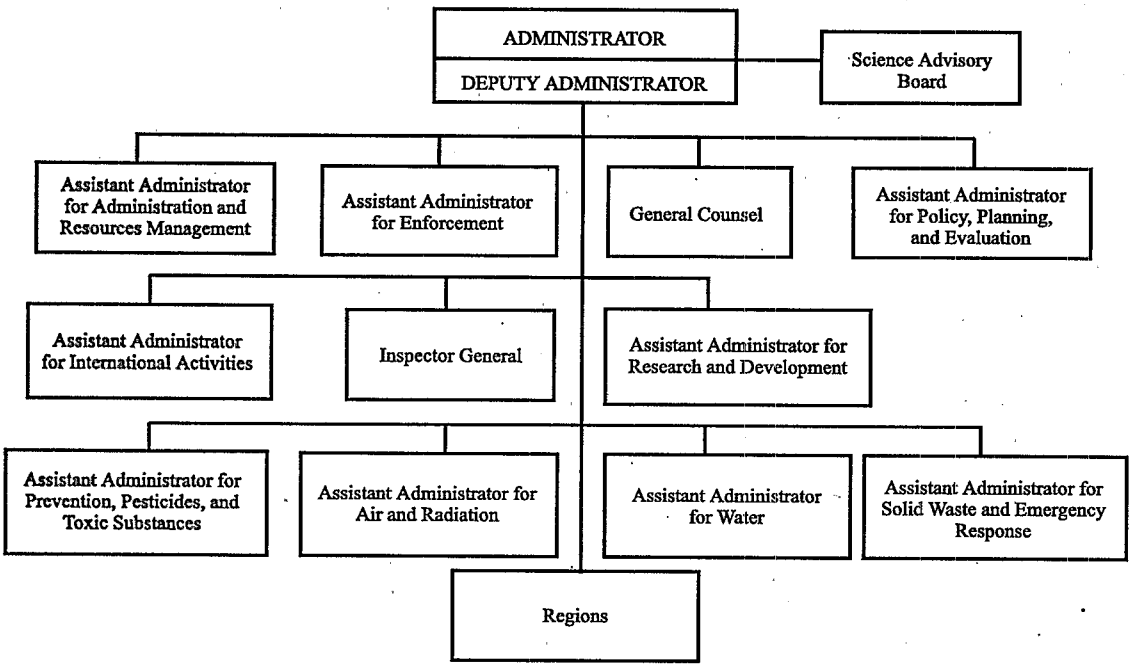
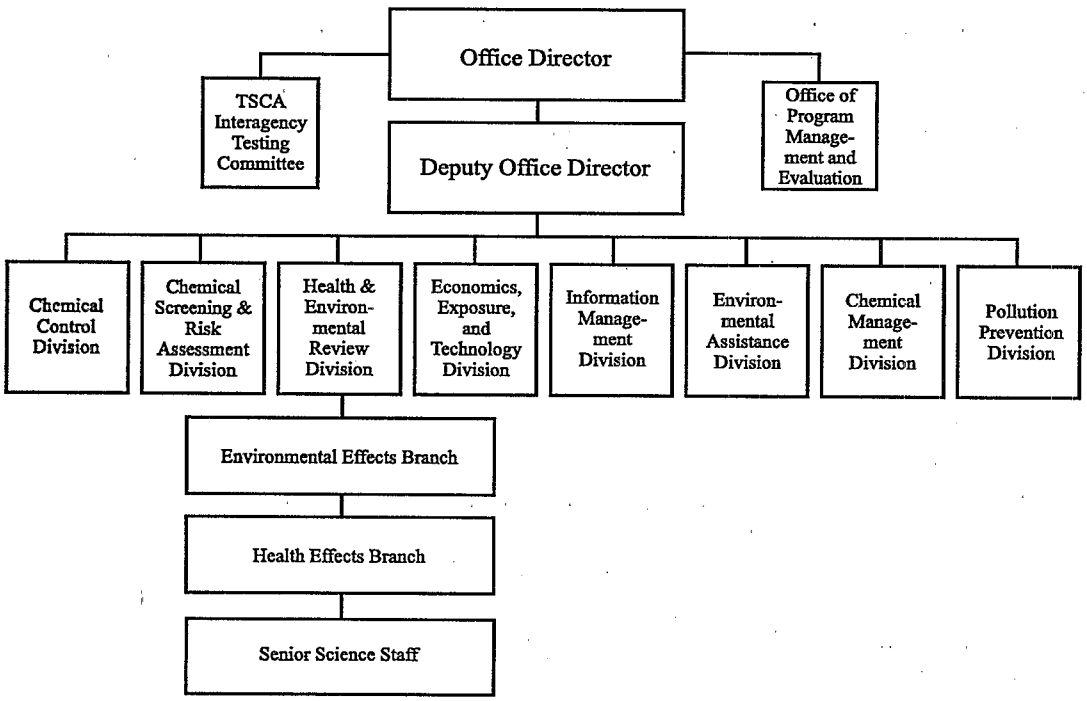
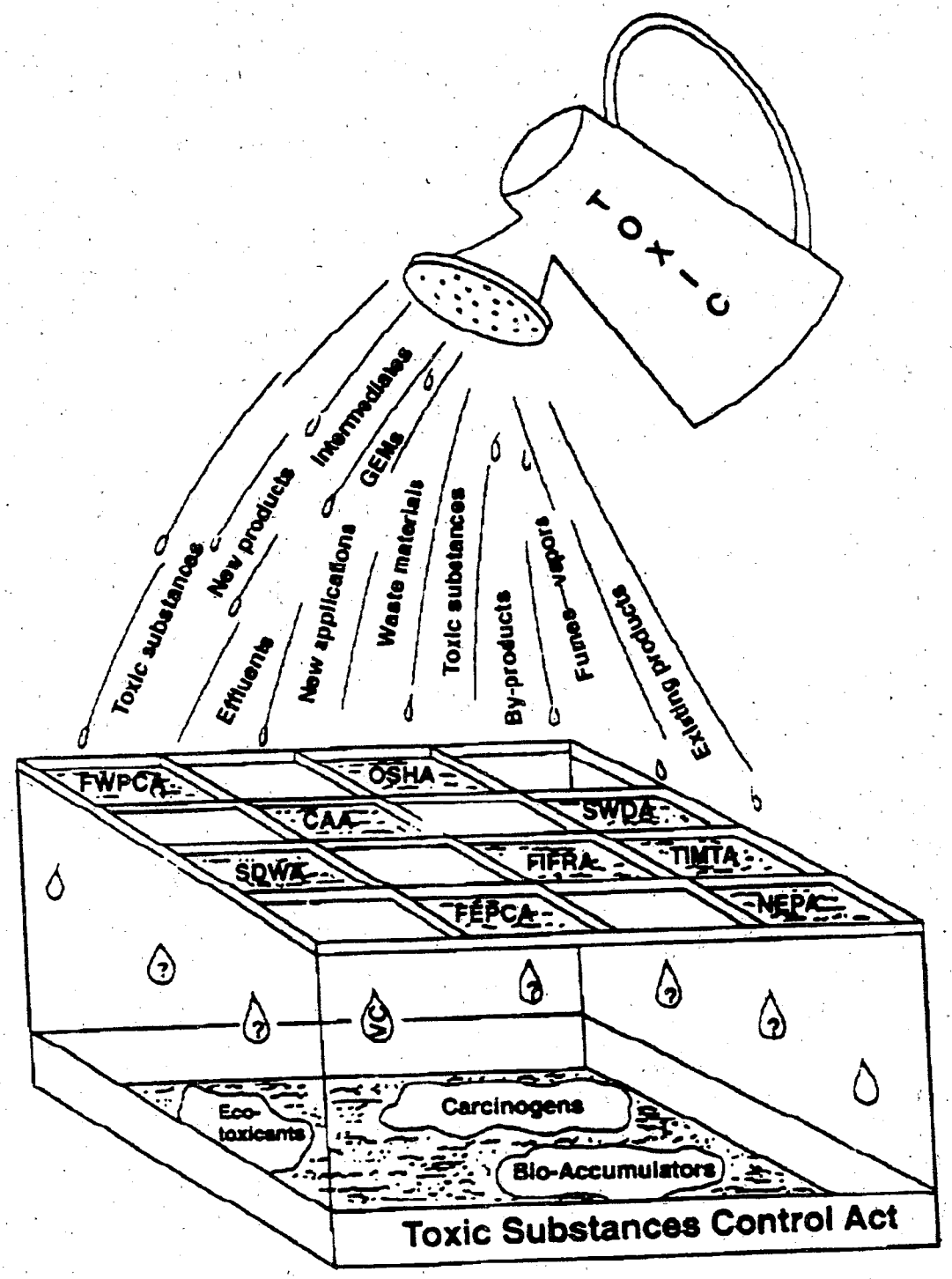
U.S. Environmental Protection Agency



Office of Pollution Prevention and Toxics



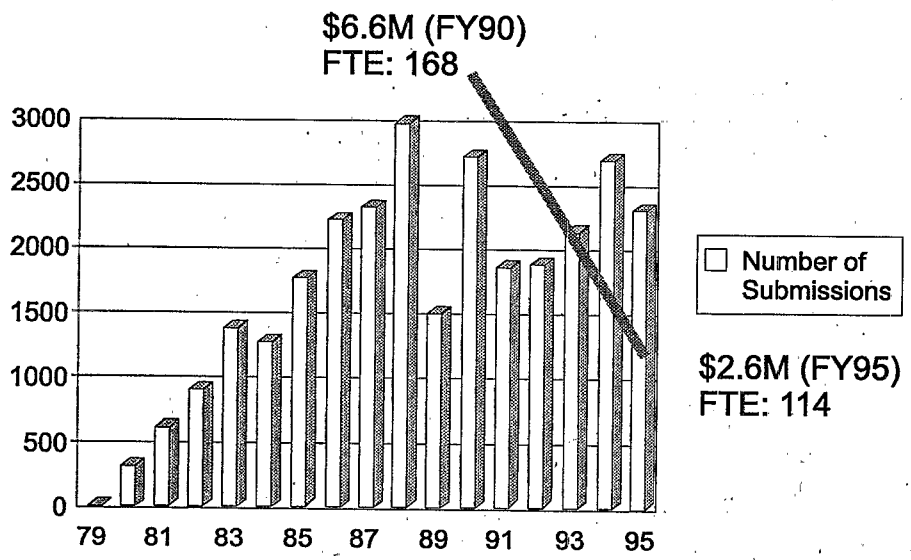
TSCA: The Gap Filling Law



TSCA (Passed 1976)

- Existing Chemicals Inventory (1979) [§8, etc.]
  - ~ 60,000 Chemicals (Grandfathered)
  - ~ 10,000 Chemicals Added (by 1991)
- New Chemicals (PMN's) [§5]
  - Chemical PMN's ~ 2,500/yr (currently)
  - Genetically Engineered Microbe PMN's
  - Submit Extant Data Only
  - 90-Day Clock (after submit PMN)

PMNs Received/Fiscal Year and FY90-FY95 Budget

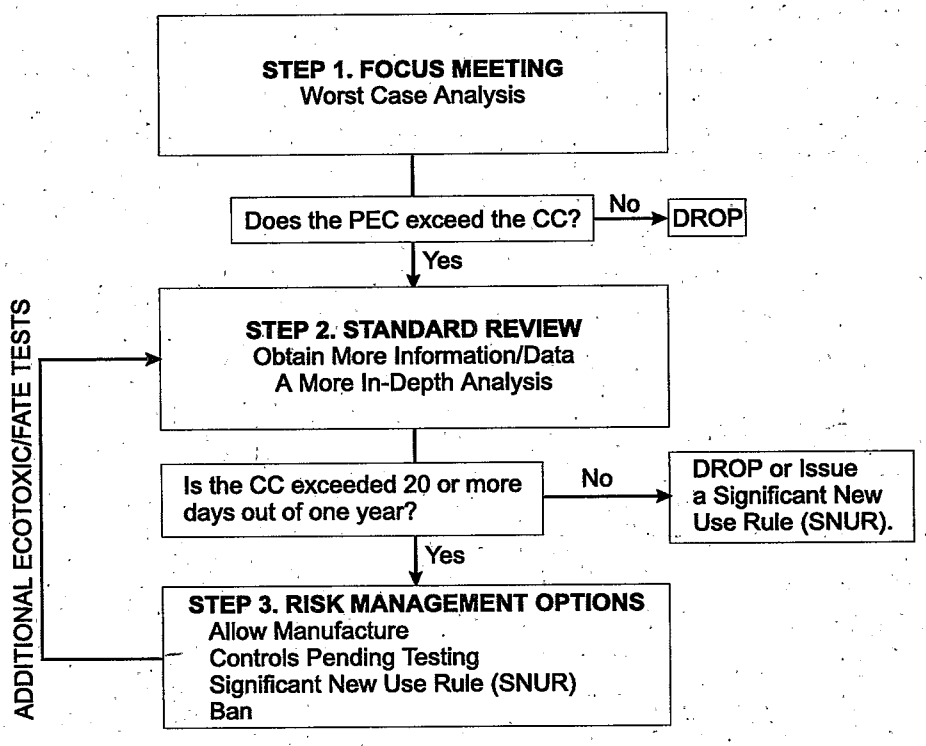


Required Submission in New Chemical Notice

- Chemical Name
- Chemical Structure
- Production Volume
- Uses and Disposal Methods
- Human Exposure Estimates
- Any Extant Test Data

(Testing *Not* Required)

Flow Chart and Decision Criteria for the Ecological Risk Assessment of a PMN Substance



PHYSICAL/CHEMICAL PROPERTIES OF PMN SUBSTANCE

Chemical Class: Neutral Organic

Chemical Name & Structure: CBI

Physical State: Liquid

Molecular Weight: 232

Log K_{OW}: 6.7 [Via CLOGP program (Leo and Weininger, 1985)]

Log K_{OC}: 6.56 [Via regression equation (Karickhoff et al., 1979)]

Water Solubility: 0.30 mg/L [Measured]

Vapor Pressure: < 0.001 Torr @ 20° C.

PMN SUBSTANCE STRESSOR-RESPONSE PROFILE

QSAR ESTIMATED TOXICITY

(Clements, 1988)

Fish 96 hr. LC_{50}: No effects at saturation

Daphnid 48 hr. LC_{50}: No effects at saturation

Green Algae 96 hr. EC_{50}: No effects at saturation

Fish Chronic Value: 0.002 mg/L

Daphnid Chronic Value: 0.004 mg/L

Algal NOEC: No effects at saturation

ACTUAL MEASURED TOXICITY

Fish Acute Test (FHM; 96 hr): No effects at saturation

Fish Early Life Stage Test (FHM; 31-day):

Chronic Value (growth, mean wet weight): 0.013 mg/L

Chronic Value (survival, growth (length)): 0.061 mg/L

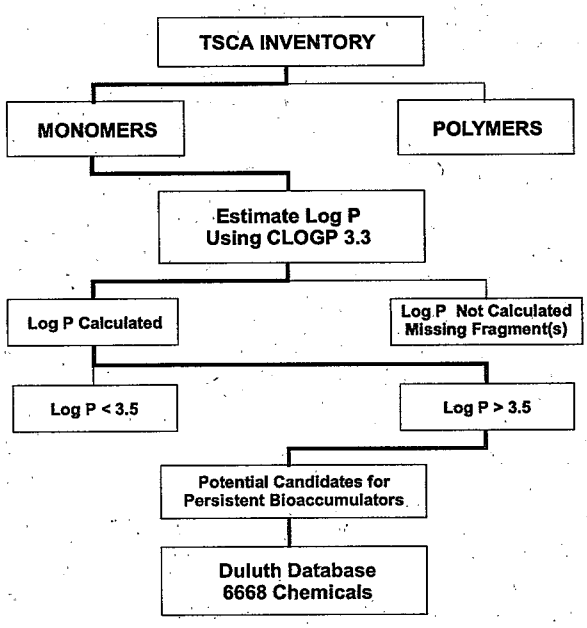
Daphnid Reproduction Test (D. magna; 21-day):

Chronic Value (survival, growth, reprod.): 0.007 mg/L

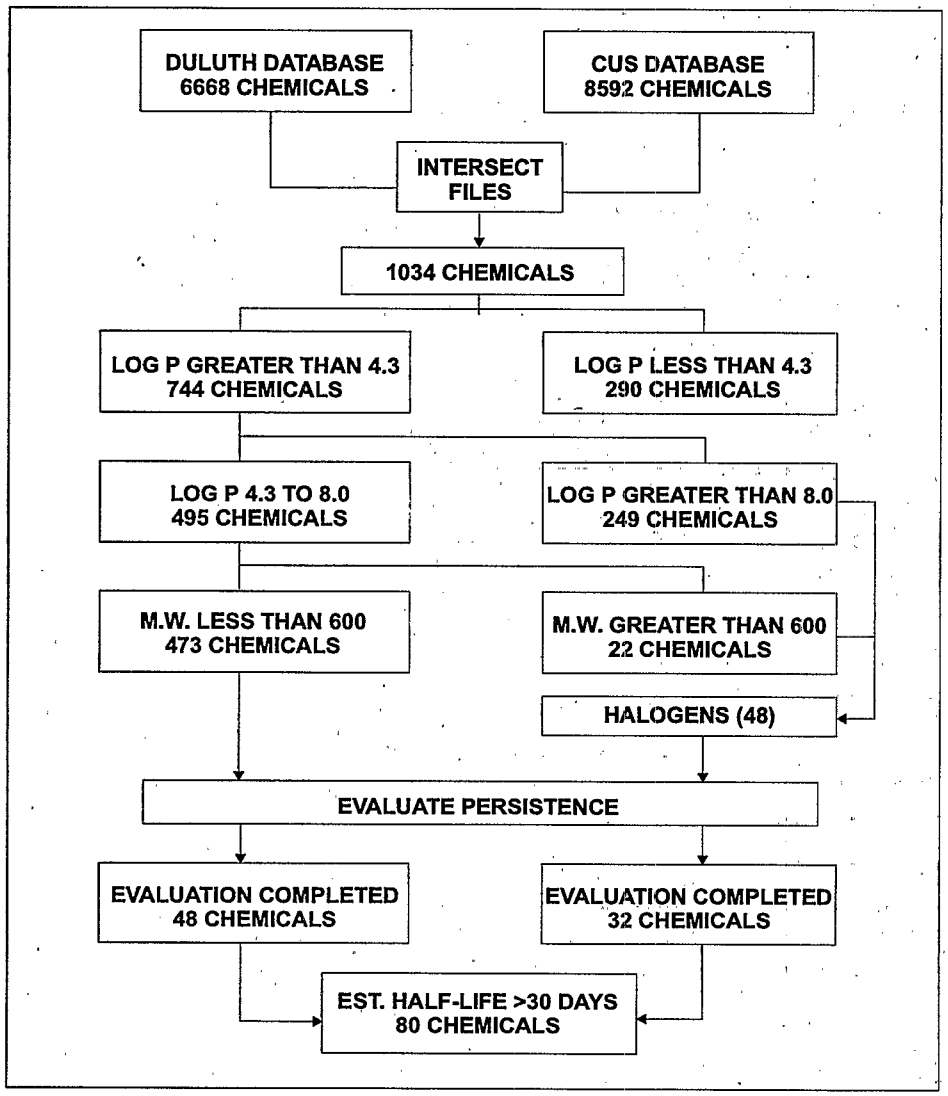
Summary of Five Risk Characterization Iterations

ITERATION	ESTIMATES / ASSUMPTIONS	UNCERTAINTY
1	Fish are the most sensitive species. CC = 1 ug/L. PMN substance mixes instantaneously in water. No losses.	Worst case. Actual test data not available.
2	Actual test data for Daphnids still yield a CC of 1 ug/L. Determine how often this concentration is exceeded using PDM3.	Worst case. Other species may be more sensitive.
3	Estimate risk to benthic organisms using Daphnid chronic value and mitigation by organic matter. EXAMS II used to estimate concentrations.	Generic production sites. Actual data for benthic invertebrates not available.
4	Site specific data obtained on use and disposal. EXAMS II rerun with new data.	Estimated toxicity for benthic invertebrates.
5	Actual test data for benthic organisms obtained.	Best estimates for identified sites. May not hold for other sites or uses.

Screening for Persistent Bioaccumulators



Initial screening process performed by ERL, Duluth to identify potential bioconcentrators.

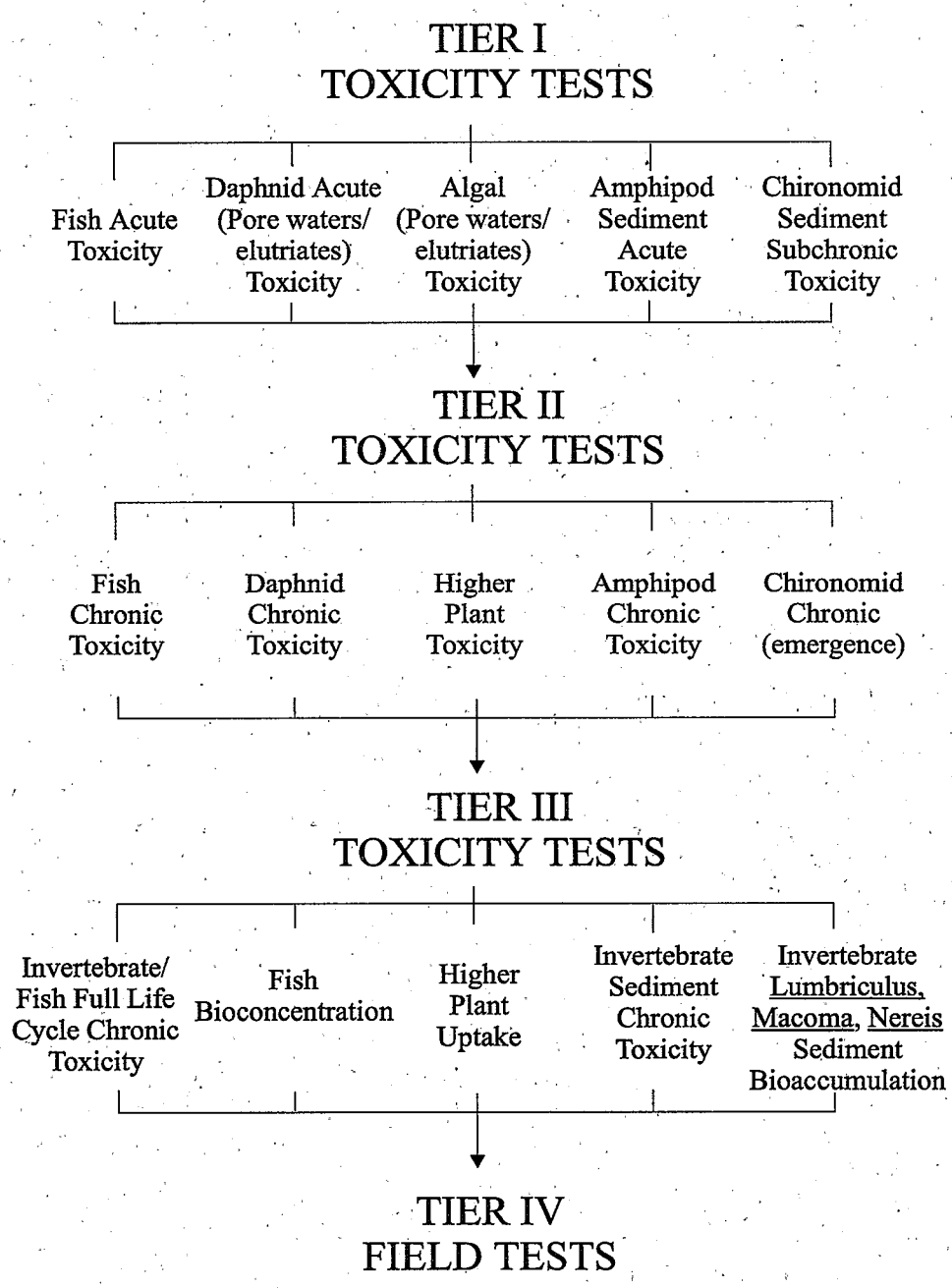


Application of criteria to screen the Duluth database for potential bioaccumulators

POTENTIAL TESTING OF PERSISTENT BIOACCUMULATORS

- Sediment/Water Biodegradation
  - Fish Bioconcentration
    - Chronic Aquatic Toxicity
      - Chronic Sediment Toxicity

Preliminary Testing Scheme For Determining Sediment Effects



(Adapted from Smrchek & Zeeman, 1997)

OPPT SEDIMENT ISSUES

NEW CHEMICALS

- 90 % Minimum Risks
- 5-10 % Further Review
- PMN Case Study — Sediments [EPA/630/R-94/003]

EXISTING CHEMICALS

- ITC Recommendations [e.g. OMCTS]
- Screening Methods [e.g., PB's see SAR/QSAR Env. Res. 3:179-201;1995]

"The work will teach you how to do it."

(Estonian Proverb)

Dredged Material Management Program

Craig Vogt
U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Washington, DC

The Dredged Material Management Program is jointly managed by EPA and the U.S. Army Corps of Engineers. Today, I want to give you a brief overview of the pertinent statutes, regulations, and guidelines and how we implement the program, including a discussion about testing, particularly what tests we do and how we make decisions based on the results.

In dredged material management, there are two statutes that we must follow: the Marine Protection, Research, and Sanctuaries Act (MPRSA) and the Clean Water Act (CWA). There is some overlap between the statutes, but generally speaking, the MPRSA applies to ocean waters and the CWA applies to inland waters, including estuaries and fresh waters. Related to this activity is the London Convention of 1972 which is the international treaty on ocean dumping. We implement the terms of the London Convention through the MPRSA, which is commonly referred to as the Ocean Dumping Act. The London Convention essentially sets forth a permitting regime that includes a black list and a gray list. The black list specifies chemicals that cannot be dumped into the ocean unless they are present only as trace contaminants or they are rapidly rendered harmless; examples include mercury, cadmium, crude oil, organohalogens, and chemical or biological warfare agents. The gray list includes a group of chemical contaminants that require a permit if they are present in dredged material (or any other material) that is proposed to be dumped into the ocean.

For dredged material management, there are specific regulations and guidelines on how to make decisions regarding issuance of a permit for disposal of dredged material in ocean or fresh/estuarine waters, and they include consideration of acute and chronic toxicity, including bioaccumulation. Under the two statutes, EPA is charged with developing the environmental criteria that form the basis for judgements on the acceptability of dredged material disposal in the aquatic environment. The Corps of Engineers is the permitting authority and the Corps determines whether or not the dredged material meets the environmental criteria. EPA has a review and concurrence role in the permit issuance process. For ocean waters, EPA has site designation responsibility which means that we designate an actual site in the ocean where dredged material can be disposed.

The statutes and regulations require that dumping of dredged material will not unreasonably degrade or endanger human health or the environment. Results of biological effects-based testing (i.e., bioassays) are the basis for decision-making in the Dredged Material Program. We consider both acute and chronic toxicity. EPA and the Corps jointly develop the testing procedures to assess the potential risks to human health and the environment of disposal of dredged material in aquatic environments. Testing manuals have been developed by the Corps and EPA; the manual for ocean disposal is commonly referred to as the Green Book. The Inland Testing Manual is a companion document currently in draft form; the final document should be available soon. Methods are included in the testing manuals for acute toxicity and bioaccumulation.

The overall framework that is used in the Dredged Material Management Program to make determinations on acceptability for aquatic disposal is provided below. We use a reference site approach. This approach has already been described by Norm Rubinstein in an earlier session at this conference; briefly, we compare the results of testing of the dredged material to those of a reference site. For acute toxicity, we use a criterion for decision-making of an increase of 20 percent acute toxicity due to the dredged material compared to the reference material. For bioaccumulation, we proceed through a logical progression of evaluations to determine the acceptability of dredged material for aquatic disposal. One tool that is available are U.S. Food and Drug Administration (FDA) action levels. If contaminant levels in the dredged material exceed the FDA action level, then the material would fail the criteria for open water disposal. If contaminant levels in the dredged material do not exceed the FDA action level, then further evaluation is required. The dredged material is then evaluated for additional factors, including such factors as how many contaminants have bioaccumulated, the magnitude of that bioaccumulation, the toxicological importance of the contaminants, and the potential for biomagnification. Evaluations of potential impacts upon both human and ecological health are carried out.

Progress is being made in regard to assessing the results of bioaccumulation tests to determine the acceptability for dredged material aquatic disposal. However,

there is a need for better information and data, such as has been presented in this conference. The Corps of Engineers and EPA sponsored a joint workshop a year ago on interpreting the consequences of bioaccumulation. Those proceedings should be out soon. The Corps of Engineers is also developing a bioaccumulation database that should be on the Internet in the next few months. EPA is contributing to that database. And finally, EPA has worked jointly with the Corps of Engineers to develop a manual for making better decisions on bioaccumulation. This manual will define a framework for decision-making focusing on a much more formalized risk assessment and risk management approach for dredged material containing bioaccumulative contaminants. EPA and the Corps of Engineers are also collaborating on development of a chronic test using an amphipod with survival, growth, and reproduction as endpoints.

In summary, we do need more information. I think this conference will help in pulling together important information.

Dredged Material Management: A Regional Perspective

Mario Del Vicario
U.S. Environmental Protection Agency Region 2, New York, New York

I would like to focus my talk on problems we are having with the dredged material program in the New York/ New Jersey area, particularly for the marine environment. I will be describing issues related to dredged material disposal at the ocean disposal site (known as the Mud Dump Site) located about 6 miles east of Sandy Hook, NJ. We are devoting considerable attention to this site because of its close proximity to a densely populated coastal area. The annual volume of dredged materials deposited at this site typically ranges from four to six million cubic yards.

Dredged material in our program is classified into three categories for ocean disposal. This regional classification scheme is based on requirements in the testing manual for Section 103 of the Marine Protection, Research, and Sanctuaries Act (MPRSA). This testing manual, which was developed jointly by the U.S. Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers (USACE) in 1977, is commonly referred to as the Green Book. We designate dredged material that is suitable for unrestricted ocean disposal as category 1 material. This material does not show acute toxicity or potential bioaccumulation. Category 2 material does not show acute toxicity, but does show some potential for bioaccumulation. This material requires capping with either category 1 material or sand for ocean disposal. Category 3 material shows both acute toxicity and bioaccumulation potential. There can be no disposal of category 3 material in the ocean environment.

In addition to testing for toxicity and bioaccumulation, we analyze bulk sediments to help us determine which chemicals we need to test for bioaccumulation. Based on the 1977 Green Book testing requirements, dredged material in our program was generally classified as about 95 percent category 1 material, 5 percent category 2 material, and less than 1 percent category 3 material. Before 1991, we also had matrix values for mercury, cadmium, PCBs, and DDTs that delineated category 1 and category 2 materials. Material that exceeded the matrix values was classified as category 2 rather than category 1. During this same period, toxicity test results and U.S. Food and Drug Administration (FDA) action levels delineated category 3 material.

The Green Book was revised in 1991. These revisions contained changes in testing, including more specificity for bioaccumulation testing and identification of more sensitive test species for toxicity testing. The new Green Book requirements had a dramatic impact on the dredged material program in our region. Based on the revised testing requirements, the distribution of material in each category changed from 95 percent to 40 percent in category 1, from 5 percent to 30 percent in category 2, and from less than 1 percent to 30 percent in category 3. We are still trying to address difficulties related to these changes, including the substantial increase in the cost of disposing of much larger quantities of category 2 and 3 material and the lack of availability of disposal sites for category 3 material. Today an annual volume of 1.7 million cubic yards of material is not being dredged because disposal sites are not available.

One cost concern is the cost of testing. We have seen the cost go up dramatically in recent years. It is fairly typical for someone that needs to dredge in the New York area to spend $80,000 to $150,000 to determine whether the material would be classified as category 1, 2, or 3. That price range covers the cost of sediment chemistry and conducting bioassays to generate toxicity information. We have tried to minimize the need to do extensive testing, but we still must meet the requirements in the Green Book and the program regulations. As more information becomes available linking sediment chemistry to biological impacts, we hope that application of this information will reduce the cost of testing.

Another cost concern is the cost of disposal. Before 1991, the cost of disposing unrestricted dredged material at the ocean disposal site was $5.00 per cubic yard. The current cost of disposing category 2 dredged material at this site, including capping, ranges from $30.00 to $50.00 per cubic yard. The cost differences span an order of magnitude. Disposal costs are even greater for dredged material at sites that serve as alternatives to ocean disposal. The cost of disposing at these alternative sites currently ranges between $60.00 and $120.00 per cubic yard.

Increasing costs, coupled with the shift in classification of dredged material to categories with more disposal restrictions, could pose a difficult dilemma for maintaining the viability of the ports in the New York/ New Jersey Harbor area and the regional economy tied to those port activities. We are working on a regional level to develop

contaminant guidance levels that we can use to make decisions on a case-by-case basis. Alex Lechich described our regional process in his earlier presentation at this conference. However rigorous our process is for defining values to categorize dredged material, we can expect to be challenged. There is too much at stake in this area. All the individual aspects of our work will likely be questioned and we need to be prepared to meet that challenge. The work we are doing, including derivation of biota-sediment accumulation factors (BSAFs) and an impact analysis on ecological and human health, must be sound and scientifically defensible.

You can help us meet that challenge. We need sound science to support our process in developing regional guidance levels and the decision-making that will follow from that process. We were very fortunate to have Bob Huggett and several of his senior scientists meet with us about our dredging issues and discuss how to address them. We hope to see national guidance levels developed, but will continue with our regional work in the interim. I have been hearing timelines mentioned like 5 years, 10 years, or even longer. But I appeal to you to make the progress we need in this area sooner, because the challenge is already there for us.

Day Three: September 13, 1996

Session Seven: Questions and Answers

After each session, there was an opportunity for questions and answers and group discussions pertaining to the speakers' presentations.

Q (Susan Svirsky, U.S. EPA Region 1): I would just like to make a couple of clarifying points, and perhaps the last one is a question for Larry Zaragoza to see if you have experienced what we have. First, in some parts of the country, we are doing ecorisk assessments with more frequency than human health risk assessments at this point because of the tailing off of issues related to human health. Now we are dealing with the tougher issues, and that leads into my second point concerning your slide. The number of sediment cleanups (14) is extremely misleading in that we have practiced avoidance behavior and put off the sediment cleanups to subsequent operable units for the sites. We are just now getting into the really tough problems, and we are going to be having a huge number of sediment-driven operable units to work with. I was wondering if that is your understanding as well?

Larry Zaragoza:

Our analysis included over 200 sites from all over the country. I would need to go back and see how many of those sites had final records of decision (RODs) at the time this analysis was done. I agree with you that basically we have found sediments to be very difficult to deal with. I have reviewed a lot of RODs and there are a number of them that have not yet dealt with the sediment issues because they were waiting for technological or other reasons. I am also encouraged by the comments you made that we are having more ecorisk assessments done. I think that is something probably everyone in this room is glad to hear.

Betsy Southerland:

Senator Levin is planning some amendments to the Superfund bill directed at encouraging more contaminated sediment cleanups. We do not know what the bill is going to look like, because they have not released the language yet. I know that was one of his prime concerns since he is from the Great Lakes area.

Q (Nelson Thomas, U.S. EPA, Office of Research and Development): Jim, 5 to 8 years ago your office was a leader in trying to deal with the whole subject of bioaccumulation. Now you say it is going to be 5 or 10 years before you can factor this information into the permitting process. Why do we need this 10- to 15-year delay to account for bioaccumulation in NPDES permits?

Jim Pendergast:

About six winters ago, I visited Nelson in Duluth to work with him to develop guidance on how to bring bioaccumulation into NPDES decisions. We can do things today if we are talking about bioaccumulation in the aqueous phase. When I was talking about 5 to 10 years in the future, I was referring to sediment bioaccumulation. What we have learned since that day in Duluth is that the fate and transport of sediments is a lot more difficult to grapple with than we originally thought. Back then we thought that it would work to add sediment into the model using a steady-state approach. When we did field studies in Lake Charles (LA) and the Blackstone River (MA and RI), the steady-state approach worked fine in the water column, but it did not work well in the sediment. Since then, we have learned from the modelers that the steady-state models do not deal well with sediments. Models for sediments must account for flooding events and other dynamic processes. That is the technology we need to develop. I think it is going to take 5 to 10 years to work that out in a way that is simple enough to put into a mass production process like NPDES permitting.

Q (Nelson Thomas): So, you do not see anything in the interim that will be able to handle bioaccumulation in the NPDES permits?

Jim Pendergast:

Not for the sediments.

Q (Phillip Rury, Arthur D. Little, Inc.): Tom, has EPA considered shifting this 90-day risk assessment burden to industry, which would put EPA in more of a review capacity? This might be a little more manageable.

Tom Murray:

Yes, we are actually involved in that right now. One of the things we have been trying to do over the last few years, with some degree of success, is to make the various tools that we use available to industry. We recently did a pilot project with Kodak where we provided them with about six different modeling packages. They applied them within their own business to make decisions about what the environmental impacts might be for the chemicals they use in their photographic processes. They felt that the pilot was a great success since the models allowed them to make their own decisions. They do not have to come into our complex program and wait for a decision. We are going through a process this year and next year to build on that experience and distribute information to other chemical manufacturers. We will probably work with the Chemical Manufacturers Association and other trade associations to see about getting more people involved with this. We are also going through a data integration process within OPPT to bring together the arsenal of tools that we use and create a context to educate a broader audience on how we use them. We are planning to make this information available on the Internet so people can access it, learn how to use it, and begin applying the models themselves.

Maurice Zeeman:

The 90-day time frame is statutory. It is not something that we chose. Nevertheless, we have been very creative in doing things to meet that deadline. For example, because our group relies so heavily on structure-activity relationships (SARs), we have developed a PC program called ECOSAR that is available from the National Center for Environmental Publication and Information (NCEPI) in Cincinnati (OH). If you know the chemical class of a compound, it will predict the toxicity of that compound and produce a hazard profile. The program is a tool for making predictions, not a data base. We have made this program available to industry and they use it to determine whether or not they should submit a chemical. If they find out that a substance is going to be really toxic on a chronic basis, then they might decide not to submit it. We have already started to see safer chemicals being submitted to us from that application.

Q (Mick DeGraeve, Great Lakes Environmental Center): Craig, how do you determine what types of bioaccumulative chemicals might potentially be present in dredged material? What process do you use?

Craig Vogt:

We go out and take a sample of dredged material, run a bulk chemistry analysis on it, determine what kind of chemical contaminants are there, and then take the next step of looking at the bioaccumulation.

Q (G. Fred Lee, G. Fred Lee & Associates): The Mud Dump Site has been studied a number of times. The last I heard, they have not found a problem after dumping from 5 to 10 million cubic yards a year for how many years?

Mario Del Vicario:

The Mud Dump Site has been an area for historical use since the 1930s. We have measured background levels for sediment chemistry and tissue analysis, both in the site and in areas outside the site that are not influenced by disposal in the site. What we have found is certain levels of contamination out there. Do we find terrible things going on? As far as bioaccumulation measurements go, a lot of the sediment and the tissue data are not at levels that you would say are startling, considering that a lot of material went out years ago without our having the knowledge that contaminants were even in the material. Dioxin is the classic example. Millions of cubic yards of dioxin-contaminated material went out to that disposal site in the past. We do not see tremendous accumulations out there in worm, lobster, or fish tissue. We do see accumulations, but not what you would expect based on what you would see if you went to our Superfund site. What we have seen when we go out to areas surrounding the Mud Dump Site are places where toxicity test results, particularly for amphipods, indicate that Category III material is sitting on the bottom. We plan to do remediation using Category I material to improve the areas where we see high levels of toxicity or accumulation. The Category I material will consist of clean dredged material from the harbor, and maybe some sand, depending on benthic community structure. I believe that much of the problem here is a perceived problem. Newspapers in the Northeast have described the terrible dilemma with sediment contaminants in the New York area. I think a lot of the news coverage was exaggerated, but unfortunately, we now have to prove that each regulatory decision made by EPA and the Corps of Engineers results in safe disposal of dredged material. That is why we need the science.

Q (G. Fred Lee): The science should be based on what you find at the site and not extrapolated from some other place.

Mario Del Vicario:

Well, we do look at the site conditions to make decisions, but part of what we have done in the last 10 years is use capping as a tool to minimize bioaccumulation at the site. I feel it has been a very effective tool.

Q (Betsy Southerland): I am going to have the panelists tell us their biggest problem with using bioaccumulation data today in their regulatory programs and what high priority needs they have to improve their use of bioaccumulation data. Then I am going to open it up to the audience to address what the future needs are. It is important to document this discussion because ORD is currently preparing its next big grant proposal for contaminated sediments. Any high priority needs that you identify at this conference will be included in the request

for application (RFA) for the grant. They will also be incorporated into the ORD Research Strategy to direct the use of their own in-house resources. If the panel does not mention a high priority need that you have perceived after listening to the discussion in the past two days, please speak up and let us add this issue to the priorities. I will ask Mario Del Vicario to begin.

Mario Del Vicario:

A critical need for the Dredged Material Program is to ensure that the information we have available to help us make decisions is truly supportable and defensible. We have been able to apply work completed on food chain multipliers and trophic models to generate information for decision-making in our program. We need to continue to use information like that and work to make the information better. We also need to continue to improve how we are looking at chronic values, aquatic effects data, dose-response relationships, and concentration effects data. A concern I have is being able to make good regulatory decisions without someone having to pay for elaborate and expensive tests when we can perhaps develop simpler ways to get a better, quicker answer. To the extent possible, we should attempt to provide easier ways for people to get decisions.

Tom Murray:

I think, programmatically speaking, we have very few barriers. If I had to describe some of the problems that we might have with the use of bioaccumulation information, one would be obtaining information that has good QA/QC. We are dealing directly with industry and with their economy in terms of producing and manufacturing chemicals. We want to try to provide as reasonable an assessment as we can. When you move into problematic areas such as bioaccumulation and sediment toxicity, it becomes more questionable. We are looking both for the availability of data and for good quality information. I know there is probably a lot of information in the EPA Regions and states that we do not have access to. It would be nice to figure out how to get those data. Another problem we have is that, unlike programs in the Regions, states, and the Office of Water to some extent, we do not do site-specific analyses. In some instances, we go into a particular area and do an analysis. If bioaccumulation data are available there, we might be able to use them. Much of the work we do on a chemical is really more of a generic population-type estimate, where we are looking at manufacturing/processing user sites all across the country. The sediment situations may differ at these sites and we are trying to come up with a generic answer. That would cause a problem for us with the bioaccumulation data.

Q (Betsy Southerland): Tom, when you say you need more data, do you want tissue residue effects levels in aquatic life, wildlife, and humans, or bioaccumulation factors and fate and transport information?

Tom Murray:

All of the above. We basically are scavengers looking for a way to get good information in our hands. Any and all of that data would be helpful.

Jim Pendergast:

Like Tom Murray's program, the NPDES Program does not have any legal regulatory barriers, but there are three data gaps that I see. First, we need to find a better way of being able to identify existing and future potential bioaccumulative problems. When Craig Vogt was asked how they determine the pollutants for the dredged material program, he responded that they use chemistry. That approach would not be practical for the NPDES Program where we must work with hundreds of thousands of chemicals. Instead, we are working with the Office of Research and Development to develop a screening methodology. We currently have a draft guidance that needs some refinements, but it is about 95 percent complete. We need to finish the guidance and get it distributed. Second, we need to have a better handle on the fate and transport of sediments. For programs involving remediation or dredging, you can identify where the pollutants are and determine appropriate ways to remove and dispose of the contaminated material. For TSCA programs, you work with a facility to keep pollutants from being released into the environment. But in the NPDES Program, we are working with mixtures and multiple sources in attempting to regulate close to 300,000 sources. We must be able to use fate and transport to link the sources with the problems, and we need a tool to do that. If possible, we need to develop something more simplistic than the WASP model that is a little bit easier to run and a little less data intensive. We began to develop this tool 3 or 4 years ago, but we need more time to complete the effort. The last problem is trying to look at things holistically. It is much easier to get data on point sources. Getting data on nonpoint sources can be more difficult. You have to put all that information together to try to get the picture of what we need to do. We really have to start using geographic information systems (GIS) more widely to determine what goes into the watershed and what sources are contributing to the predominant problems. A lot of work has already been done in this area, but much more still needs to be done. It will take some time not only to fit together the existing pieces of the puzzle, but also to fill in the missing puzzle pieces.

Larry Zaragoza:

I see a couple different needs for the Superfund Program. One is the kind of work Mike Kravitz is undertaking to understand the meaning of different testing. This will be very useful. It is difficult when you have several tests available, but you are not really sure how to interpret the results and determine what is significant. Traditionally, we have tended to look to other offices for guidance on that issue. Another issue for our program is determining

how the significant risks in the water column or sediments compare with risks in other media. We would like to be consistent in the way that we are addressing risks for a particular site. If we find that we are much more sensitive in one medium than another, we need to balance how we are addressing all the factors we consider. Better science could help us balance those factors, even with the added complication of having to incorporate cost/benefit analyses into our evaluations. Finally, there is the issue of what to do with contaminated sediment once it has been identified at a site. If it is dredged, the material may be difficult to dispose of. This is an issue because there are a lot of sites in our program with contaminated sediments.

Q (Betsy Southerland): Are you talking about developing new remedial technologies or improving existing technologies for remedial alternatives like capping or disposal in confined disposal facilities (CDFs)?

Larry Zaragoza:

I am talking about all those things. There are problems with existing technologies to deal with contaminated materials. Not only are there limitations in terms of places that can hold the sediments, but there are also debates about what would be an appropriate disposal site. And the question has been raised about whether CDFs are an appropriate place for disposal of certain types of sediments. We have been involved in intensive discussions on all of these issues. It is really important to have an open dialogue on this subject now, because Superfund legislation is being considered for reauthorization and it is likely that the reauthorization package will include amendments to address a number of different issues. One set of amendments will target facilitating the process for addressing contaminated sediments. It will probably include a research component that can help us resolve outstanding issues. But the cost/benefit question will come up again if we focus more on sediments or the water column than on soil.

Maurice Zeeman:

I will begin and end with the same question, "What sediment and bioaccumulation data?" I say that because a lot of monitoring is being done today for the same chemicals we have been looking at for over thirty years. These typically include organochlorine insecticides, PAHs, PCBs, dioxins, furans, and heavy metals. The Toxics Release Inventory, which was established as a result of Superfund, required industry to report on their emissions for only 300 chemicals. The reporting requirement focused on chemicals that were highly toxic, but not necessarily persistent and bioaccumulative. In the late 1980's, industry reported several billion pounds of emissions for just 300 of the 75,000 existing chemicals. These reports indicate that a lot of chemicals are being released into the air or water. Not all of them will necessarily be a problem, but a fair portion could end up in the sediment. We need to move beyond looking at the usual suspects. My group is responsible for conducting ecotoxicity assessments, but we rarely have bioaccumulation data to use in these assessments. We have a tiered testing approach in our program for both new and existing chemicals. We require testing for acute toxicity and if we think a chemical is likely to bioaccumulate, we move to chronic toxicity testing. There are cases, especially for high log P chemicals, where acute toxicity testing may not always be appropriate. In these cases, we need to consider moving right to chronic toxicity testing. So, I go back to my original question, "What sediment and bioaccumulation data?"

Craig Vogt:

There are a lot of needs for the Dredged Material Management Program. Mario Del Vicario summarized some of those needs very well, but I am going to expand on them in terms of tissue residue effects levels, chronic tests, and fate and transport models. We basically need more information to be able to get better answers for several important program questions. For example, we have a site designated in the ocean for disposal of dredged material. When dredged material is dumped there, where does it go? Does it stay on the site or does it move to another location? Better fate and transport models would help us answer these questions. The Dredged Material Management Progam is moving into a more formalized process for risk assessment that will involve looking at exposure, characterizing risk and managing based on the risk assessment results. We have the Green Book and the Inland Testing Manual, but we need help on exposure and exposure analyses. We also need to determine what tissue residue effects levels mean on an ecological scale. Without that type of information, decision-making gets difficult. As we continue to develop the science, we can rely more fully on science as the basis for decision-making, rather than politics and misinformation that has influenced public perception. The public currently considers the oceans to be a sacred place where nothing should be dumped. But disposal of dredged material at ocean sites is a very minor source of contaminants that enter the ocean compared to other sources such as surface water runoff and some point source discharges. It is a matter of balance and how you achieve that balance. I would like to see better risk assessments conducted for the various media. We can use this additional information to involve stakeholders and together make more scientifically sound decisions for managing dredged material.

Betsy Southerland:

I would like to open the panel discussion up to the audience. Please state your name and affiliation before you begin to address the panel.

Q (Susan Svirsky, U.S. EPA Region I, Superfund Program): Maurice Zeeman, do you have analytical

methods available for some of these chemicals that are outside the normal sweep of what we look for? If so, can you make them available to us so we can further our analytical capability and look for some of them?

Maurice Zeeman:

I am a biologist, so I am not a good source of information about analytical chemistry. Let me refer you to Bob Boethling of my office for that information. But I do know, for example, that chlorinated paraffins are compounds like PCBs that are a messy mixture of chemicals. The analytical chemistry for these mixtures can be very complex. I am convinced that these chemicals are extremely hazardous to organisms in the environment, but manufacturers are producing hundreds of millions of pounds of them each year worldwide. Some limited monitoring data from a few places around the world indicate that they are causing problems. However, there are a fair number of more simple compounds where the analytical methodology should not be so complex to develop and apply.

Betsy Southerland:

Susan, there are two studies I am aware of that might provide some useful information for you. One was a study that Bob Huggett conducted with the Virginia Water Control Board when he was still at the Virginia Institute of Marine Science. It was a big study where they collected sediments downstream of some urban industrial areas and analyzed the chemicals in the sediments using some kind of library scan. They were able to identify over 300 compounds that were persisting in the sediments. You can locate literature on the study to see what method they used. There is also the screening procedure for wastewater being developed by our Duluth laboratory that Jim Pendergast mentioned earlier. For this procedure, wastewater is fractionated based on octanol-water partition coefficients and the fractions are run across a library scan to identify any potential bioaccumulatives that could be in a waste stream. They are doing some additional testing before they release the protocol.

Steve Cibik, ENSR:

We have come a long way in taking the "pseudo" out of pseudo science and replacing it with good science, especially with recent developments for BCFs, bioaccumulation, trophic modeling, and food chain modeling. We made a big advance when EPA issued the Wildlife Exposure Handbook to help standardize risk assessments. What we need now is better information on tissue residue values, because I heard that we are approaching a factor of 2 or 3 in accuracy for many of these things, but adding one safety factor to a risk assessment will give you a factor of 5 or 10. Decisions in risk assessment have to be based on good tissue residue values. We already have a lot of aquatic toxicity numbers available. I hope EPA will be able to help develop more information for wildlife. We need tissue residue values for both terrestrial and avian wildlife.

Q (Catherine Fox, U.S. EPA, Office of Enforcement and Compliance Assurance): The 1992 mandate of the Water Resources Development Act (WRDA) called for the development of the National Sediment Inventory, a national data base which is now available. Many of you have evaluated it to identify hot spots based on a number of factors, including fish tissue data. What we can do right now is to look at these hot spots based on fish tissue data, and for the various programs, to address the current sources of contaminants to prevent the discharge of these contaminants from point sources, nonpoint sources, and sediments. I have looked at this using a GIS and found some really interesting things that are happening right now with discharges. I would like to go upstream and look at the sediments. I am asking the panel to consider this as a need that we could address now. We have the people and we have a lot of the tools. Jim Pendergast looked at this 5 years ago. He probably knows a lot that could help us now. This is an area of interest for enforcement and I am requesting assistance from other offices.

Tom Murray:

As Maurice has mentioned, there are many chemicals on the inventory that OPPT has not had a chance to look at. OPPT is setting up a priority system to look at these chemicals. A couple of things are happening in OPPT right now to advance that process. One is that we are moving away from a single chemical approach to life and trying to look at chemicals as use clusters or clusters of chemicals that might be found in a product. We have developed some systems within our organization to help us set those priorities. For example, we are looking at indoor air sources as one system. We are ranking various chemicals in this system to figure out which indoor air chemical products or chemical products that might lead to indoor air problems are the most important to look at. We have also developed the use cluster scoring system to look at a variety of information and help identify what areas or what chemicals we should focus on. Catherine, maybe we can help generate more information about the sediment contaminants you are concerned about. We could perhaps consider a cluster of chemicals that may find its way into sediments or tissue and factor that additional area into our priority-setting process.

Betsy Southerland:

Enforcement actions based on data in the National Sediment Inventory or fish advisory data base must be based on the fate and transport modeling which Jim Pendergast mentioned earlier. There is a lot of concern that it would be difficult to take the ambient data and trace it back to a source. None of the data that we have either in the sediment inventory or in our fish advisory data base will prove cause and effect. We need additional studies to be able to take the presence of a contamination problem and

work it back to a responsible source. If that responsible source is a point source in violation of permit limits on those toxicants, then it certainly would be appropriate to take an enforcement action. But again, the ambient information in our data bases will not allow you to link to sources. You will have to demonstrate cause and effect from additional studies.

Jim Pendergast:

I would like to add one other thing to that. Catherine, you may want to talk with Louise Wise in the Office of Wetlands, Oceans, and Watersheds. She helps chair a Division Director group on watersheds. All of the Office of Water divisions are working to try to figure out how to do things on a watershed basis, so we avoid the sector-by-sector approach and try to look at things holistically. A Minnesota example can illustrate one of the reasons why this is important when you are dealing with fish advisories and sediments. There are a number of lakes in Minnesota with fish advisories for mercury, but no identifiable point source for mercury. Mercury may be transported into the area in the air and deposited in the lakes when it rains. You should start to address these issues with the watershed group.

Q (John Haggard, General Electric Company): Robert Paulson brought up a new term that I think is particularly appropriate for sites where companies with liability and regulatory agencies are dealing with difficult sediment problems. That term is potentially reasonable parties or PRPs. We are one of the PRPs at a number of these sites and we clearly have concerns with the liability management. How do we manage these problems in a cost-effective way? These sites are presenting unique problems to regulatory agencies, because there are no simple solutions to sediment problems. Where are you going to put this vast volume of contaminated material? What is science telling us about what we can really achieve? What damage might we cause by removing the material? There is a lot of positive research going on to help reduce the uncertainty. This is very important to industry since we would like to see some actual benefits from the money expended. I would like to make a recommendation related to a chart Larry Zaragoza presented that showed the basis for decisions at sediment sites. I was surprised to see that 9 of the 14 sites involved risk-based decisions. We have looked at 80 or 90 contaminated sediment sites and found that only about a dozen have gone through a record of decision process and some remediation. When we look at sites, we can rarely figure out the basis for the decision. Documentation on these sites is a serious problem. In addition, when we look at how well the technology performed, we see clear problems with dredging. I would recommend strongly to this group that you look at the capabilities of the technology within a comparative risk framework at these contaminated sediment sites. It is an important issue that I hope somebody will take a shot at.

Betsy Southerland:

A group from our Office of Research and Development laboratory in Cincinnati, Ohio just informed me that they are organizing the National Conference on Management and Treatment of Contaminated Sediments to be held during May 1997 in Cincinnati. They are inviting speakers to present a series of case studies ranging from bench-scale remediation projects to full-scale remediation efforts. They are also inviting a large variety of vendors to display and demonstrate new equipment developed for remediation projects and equipment that has proven successful for previous projects. The conference will be a good opportunity to hear about experiences with sediment remediation. Since a number of the case studies will involve Superfund sites, we can request that the speakers include details in their presentations about how they reached the decision for remediation and how they determined the volume of sediment for removal.

Larry Zaragoza:

I think that one of the things we need to do is to begin operating in an environment where people can actually see the logic and consistency behind a particular program. We have been confronted by a whole host of challenges in the area of contaminated sediments. Susan Svirsky raised some of the issues in her earlier comments about people avoiding dealing with sediment problems to date because of these challenges. But for sediment sites that have been addressed, these efforts might be considered successful if all the parties involved in the site agree to clean it up and the community is pleased about the actions being taken at the site. That is very different from having a national profile that shows you benefits information and consistency in cleanup levels across the country. As a result of looking at how we have operated as a program and hearing comments like yours, we are seeking to collect that information in a more systematic and consistent manner. We have already collected some information, but we are trying to do a better job of obtaining more information. I think you can expect that in the future more comprehensive information about our sites will be readily available.

Maurice Zeeman:

We also have to realize that there has been a bias in many cases for "we have to do something" action. Certainly in the past, there have been cases where actions other than removal were appropriate. An example that comes to mind is the case in Triana, Alabama where people were being exposed to high levels of DDT. The highest levels were found on an Army base that produced a lot of DDT. The Army Corps of Engineers solution was to reroute part of the river and cover the most contaminated areas with topsoil. That may not have been the best solution, but it was the most cost-effective approach under the circumstances. I have been teaching a course in

environmental toxicology at the National Institutes of Health (NIH) since 1982. One of the things I have found really remarkable with that experience is how ignorant we can become over time. I have asked my classes recently about what happened with the Kepone spill in the James River, and virtually nobody is aware of the incident any more. Many people think the problem has gone away. It was going to cost many millions of dollars to dredge after the spill, so the lower river was closed to a number of activities and allowed to recover naturally with deposition of clean sediments. It is a solution that has worked over time. But, Bob Huggett has presented results of studies that show Kepone-contaminated sediments have been moving out into Chesapeake Bay. All we need is the right kind of hurricane to redistribute these sediments for the problem to occur all over again.

Q (Helder Costa, Inchcape Testing Services, Aquatec Laboratory): Several EPA-sponsored studies have shown close associations between alkylated PAHs in petroleum-influenced systems and adverse effects to benthic organisms and habitat. In the Delaware Estuary, for instance, the alkylated PAHs accounted for typically 50 to 70 percent, or even 80 percent, of the total PAH loading. I want to comment that we can only begin to understand the importance of PAHs as a chemical compound class when we begin to consider the alkylated PAHs in petroleum-influenced systems.

Betsy Southerland:

I know that Rick Swartz from our laboratory in Newport, Oregon will soon be publishing a total PAH analysis based on narcosis effects. He is trying to help us analyze PAHs as a whole chemical group instead of having to consider each PAH with its individual toxicity.

Q (Helder Costa): I particularly applaud the work Rick Swartz has done in developing the sum PAH model and publishing the results in the SETAC journal (Environmental Toxicology and Chemistry, *Vol. 14, No. 11, pp. 1977-1987, 1995). In his paper, he acknowledged the importance of alkylated PAHs in petroleum-influenced regimes as a temporary limitation to the model.*

Betsy Southerland:

An individual from the Metropolitan Washington Council of Governments has conducted some studies that show PAHs are coming not only from products of combustion, but also from ground water contaminated by leakage from automobiles and other sources. That is new information to me in terms of sources.

Q (John Zambrano, New York State Department of Environmental Conservation): If we agree that contaminated sediments are a problem that we want to do something about, we are going to have to know the relative amounts of various sources of contamination. We also need to know whether past contaminants that we may not be able to do much about are circulating within the system, or whether new contaminants are being introduced by current sources. Could these sources be point sources that we are not regulating sufficiently such as stormwater or combined sewer overflows? Could land runoff or atmospheric deposition be contributing to the problem? It seems to me that unless you know the amounts from various sources, you will not be able to make intelligent decisions within the Superfund Program, the NPDES Program, or the Dredged Material Program. I think we need to do more sampling and modeling to get that information.

Larry Zaragoza:

Your comment is very well taken. I agree that it is worthwhile to have that kind of information. For a Superfund site, we would specifically look for that kind of information to see what is going on at the site level. But that only speaks to a particular site. It does not tell you about what is going on in the rest of the country. Sediment assessment generally needs to be site-specific, but that complicates comparing a site to other sites across the country.

Betsy Southerland:

The best mass balance study done to date that I know of was the work conducted for PCBs on the Fox River in Wisconsin. I think they had a budget of about $15 million for monitoring. I know that EPA programs are trying to look at mass balances, particularly in the Office of Air and Radiation. The air program has worked with us in preparing their second report to Congress on deposition of air pollutants to the Great Waters. They are trying to do large-scale mass balances for substances like mercury and PCBs to determine how much mercury and PCB contamination can be attributed to air emissions.

Q (Don Porcella, Electric Power Research Institute): I have enjoyed this conference very much. There have been some new approaches developed and presented here, particularly the food chain modeling. Will these approaches be incorporated into future analyses? All the approaches we have heard about include the concept of models. For example, a bioassay is a physical model as opposed to a mathematical model. I think it is really important to do groundtruthing for these models, as well as to address the "So what?" question Gil Veith raised at the beginning of the conference. This question particularly relates to Fred Lee's comment about whether there has in fact been any damage in the New York Harbor area from dredged material disposal.

Mick DeGraeve:

I just want to reinforce the importance of what Betsy Southerland and Jim Pendergast mentioned about finalizing the screening procedure for bioconcentratable contaminants. Larry Burkhard, who is in the audience, is the principal author of the method. One thing that Betsy and Jim did not mention is that the method works not only for effluents in water, but also for tissues and sediments.

In terms of some of the dredging-related issues, it can be applied to address the likelihood of bioconcentratable contaminants being present in sediments and certainly in tissues. We have used the procedure commercially to address some very interesting questions and it has worked quite well.

Q (Betsy Southerland): Have you used it on sediments or just wastewater?

Mick DeGraeve:

We have used it on sediments. I am glad that efforts are going forward to finalize it.

Jim Pendergast:

My focus at EPA has been primarily on wastewater. But for one year in the 1980s, I worked with the Superfund Program. While I was there, it would have really helped to have had that methodology available to assess sediments at Superfund sites. The guidance also includes information about integrating fish tissue data in the decision-making process to determine whether or not there is a problem. We tried to make it a one-stop document to deal with multimedia.

Q (Lynn McCarty, L.S. McCarty Scientific Research & Consulting): My opinions are entirely my own since I do not represent an agency. There are two areas I would like to comment on. The first one is technical. If you continue to go down the toxicological route, you are eventually going to have to deal with molar units. Since toxicology is a function of the number of molecules of a compound, not its weight, you will find that many relationships that would be obscured by using weight-based measures will become much clearer when you use molar weight. The second issue is a philosophical one at the very end of the scale of what we are talking about today. I was pleased to hear Dr. Patton indicate in her discussion that both science and policy are included in the EPA risk assessment process. That is an explicit recognition of something that I think a number of people have realized for a long time, but it is an important distinction. We must keep the science and the policy separate or at least identify them as such. In this whole process of becoming open and transparent, making this distinction is particularly important. What we are trying to do in this whole exercise is risk management. We are trying to achieve some sort of environmental protection. As scientists, we need technical decision criteria to do that and to fit into the decision-making process. In other words, we need to be able to define what a significant adverse effect is. It is great to say that we have to protect the environment, but when it comes down to scientific measurements, what does that mean? We have some difficulties because we are trying to define future problems from an existing problem perspective and from a modeling perspective. The objective of risk assessment is providing information to make choices among alternatives. So, what is the framework? What are the effects that we are concerned about and what are the alternatives that we are choosing among? Developing information for risk assessments involves specific goals that change from the research emphasis being presented here to regulatory utility. If we do not have risk management directions that are framed in technical terms, we cannot make decisions. That is what this is all about. This is a policy issue, and the science is guiding the decisions. I found the list of questions Dr. Southerland provided for the speakers to be very useful because it addressed many of the important technical issues. Unfortunately, I do not think we have coalesced those into a useful decision-making framework. I would like to encourage the panel and EPA to consider that, because I think that many of the people at this meeting are looking for very specific directions. What is it that we want to do, and how can the science that we have available help us in making the decisions and choosing among those alternatives?

Alex Lechich:

As a scientist regulator, I sometimes feel that research scientists do not carry the same burden as we do, so I do not have any qualms about trying to lay a little more weight on their shoulders. In terms of the data bases that are being developed for the specific effects, I think it would be helpful to go beyond just assembling, compiling, and presenting this information. Have the people who are most familiar with the studies and the data itself become involved where there are conflicts. In some cases, we can look at the data and make fairly clear decisions. In other cases where the results conflict with each other, it would be nice to have the people closest to those studies review them and provide a recommendation as to where to go from there.

Phillip Rury, Arthur D. Little, Inc.:

I would like to thank and commend all of the people who contributed to this excellent conference. It is probably one of the better conferences I have been to in many years. I would also like to say that I feel it is long overdue and I hope that we can see a sequel to this very soon. In terms of some of the gaps that I see and the future research needs that some of the speakers have identified, those same gaps were reflected in the presentations here. I would hope that we could address them in future conferences. I would implore everyone to look in these directions. In a kind of evolutionary-speak sense, perhaps our research needs to crawl up onto the land again. The aquatic focus of this conference was obvious, but I am concerned that issues such as amphibian bioaccumulation are not being addressed. The questions concerning wetland species also need to be addressed more vigorously. For example, the data base presentations at this conference specifically excluded amphibians with the exception of tadpoles. We have found this lack of information on amphibians to be a major constraint in several risk assessments. Mink food chain exposure models, for example, are highly sensitive to the frog elements in the diet. We also have a dearth of bioaccumulation factors. Several that do exist are quite

high and they are driving the cleanup goals. I would also like to get on my soapbox about the lack of integration of ecological and human health risk assessment. I would hope that maybe in another 10 or 20 years, as the pace of regulatory change proceeds, that we might see a shift to an ecosystem-level approach to risk assessment that includes consideration of both human health and ecological risks. This approach should allow us to realize some efficiencies of costs and some enhancements of understanding by integrating humans into an ecosystem risk assessment as a mandatory indicator species in every case.

Burt Shephard, URS Greiner, Inc.:

There are two areas I would like to comment on. One is to respond to the remarks about the data bases. I think those of us who are putting the data bases together are very much aware of the conflicting data. You might not see it at the first level of building the data bases, but I think that shortly thereafter you will see some attempt by those of us putting the data bases together to resolve conflicts. Some of the data quality issues are very important to us. Some of the data are not very good, but some are excellent. Initially, we will be putting out summaries of the data with a lot of data qualifiers saying how things were done and let the end users make their own judgments about whether the data is good or bad for their own purposes. I certainly concur with the comments and think we will be making some efforts to resolve some of the conflicts within the data at some point down the road. Second, before we can really define a contaminated sediment, we need to know something about what an uncontaminated sediment is. There were some talks here about defining background reference area sediment concentrations. Within the risk assessment area, that is a very large need right now. The guidance today within the various EPA Regions for defining background or reference area concentrations is inconsistent. I think it is very important to have a defined statistical guidance that describes how to compile reference area data and compare site data to it to see if the site data are exceeding the reference or background data to a degree that indicates contamination problems. The RCRA guidance documents on statistical comparisons are the best ones that I have seen within EPA. They look at sample mean comparisons between sites and background areas. They also have, for lack of a better term, a hot spot out on the tail of the distribution to see if very high concentrations fall within a distribution or within a confidence limit of a distribution. I think some consistency with comparing backgrounds or calculating backgrounds would be very useful to everyone who has to do risk assessments.

Q: (Malcolm Watts, Zeneca, Inc.): I want to support the comments that have been made by Fred Lee, Lynn McCarty, and John Haggard. Lynn McCarty asked, "What do we want to do and how does science help us?" What we want to do is to maintain or clean up the environment. That is pretty obvious. We are all breathing out carbon dioxide and contributing to global warming. Shall we all remove ourselves from the planet? Clearly not. I suggest that it is a matter of cost-effectiveness. How much does it cost and what are the benefits? Fred Lee said this morning that there seemed to be very little damage from the New York Harbor dredging. So, I question the costs. I also question the effectiveness versus the benefits. I would like your comments on cost-effectiveness and the extent to which we have tried to look at the costs of doing nothing and the benefits of doing something.

Betsy Southerland:

We did do a few case studies where we looked at the costs and benefits of remediation. I am sure everyone here is aware that the science of analyzing benefits is an area where we lack monetization methods. We have three tiers of benefits analysis that we do for environmental projects. The first tier, which is a qualitative discussion of benefits, is the one we generally have to use. If you have more data you can move up to tier two, which is a quantitative description of benefits. Only at the third tier do you have monetization of benefits. No matter which of our programs is doing a regulation or other action, we always have a tough time at EPA doing monetization of benefits, whereas monetization of costs is a very well-defined area. We have disagreements with the regulated community about our cost estimates, but we have plenty of procedures to provide us with costs. At any rate, when we have done cost/benefit analyses on some remediation projects, we have been able to demonstrate that the benefits have met or exceeded the cost of the remediation. There were studies at three Superfund sites where the costs were in the area of $25 million to $50 million for the sediment remediation and the benefits were at or above $50 million. Those are the only cases that I know of where the focus has been on calculating the costs and benefits of sediment cleanup. Since they have state water quality sediment standards, Washington State is doing similar work.

Larry Zaragoza:

I did not want to give any site-specific information, but I did want to say that Superfund is not a cost/benefit statute. In making the request that decisions be put into a cost/benefit framework, I think you need to look at what the legislative direction is for the program. Basically, each of our programs has a different framework and a different legislative history. We are supposed to administer each program based on what the law says. We should also coordinate among programs to be efficient and to learn from each other. The last Congress had a lot of discussion about costs and benefits, and I expect that we are going to see that reflected in the next set of legislation. I would also like to reinforce what Betsy Southerland said about the monetization of benefits. Quantifying benefits in monetary terms is not something we have done a lot of. It is controversial, so that will make it challenging in many cases to develop a number that

may be accepted. There are a number of issues in this area that will continue to be debated.

Maurice Zeeman:

As Mother Nature says, "Pay me now or pay me later, but a lot more later." If the Great Lakes area is any example, the time frame for many ecological concerns is decades. If you are talking about the need to know this week how much it is going to cost and what the benefits are, you are going to end up with nearly intractable problems. The Chesapeake Bay is another example. We are seeing examples around us of problems where their scope and scale is virtually beyond all the small management and cost/benefit decisions we are making. I concur with what the prior two people said. We are getting better at it, but so far the environment has been virtually free. That is the reason that it is so contaminated.

Betsy Southerland:

If there is no further discussion, I will close this session with a few summary remarks. I want to reiterate that the future needs raised by speakers and other participants in this conference will be factored into ORD research planning and included in our bioaccumulation report. If you are not already on the mailing list, I also want to remind you to sign up for the *Contaminated Sediments News*, a newletter that we produce and distribute from our office. We will provide follow up information for several things mentioned during this conference in future issues of our newsletter.

It has been tremendously informative and enjoyable for all of us that have worked for more than a year to organize this conference. We especially appreciate the commitment of all the speakers and moderators to continue working with us when we had to reschedule the conference due to the Federal furlough in the fall of 1995. It was a challenge we were finally able to overcome. We thank you for your interest and participation in this conference. My staff will be available after the conference to listen to any additional comments and suggestions you may have before you leave. Again, thank you for coming. We will be keeping in touch with you through the *Contaminated Sediments News*.

Speakers' Biographies

James K. Andreasen, Ph.D.

Dr. Andreasen is the leader of the Ecological Assessment Team at the National Center for Environmental Assessment within the Office of Research and Development of the U.S. Environmental Protection Agency (EPA). He works in Washington, D.C. Dr. Andreasen received his Ph.D. from the Department of Fisheries and Wildlife at Oregon State University in 1975. Following a short teaching career at the University of Alaska, he secured a position with the Federal Government and has been conducting research on the effects of contaminants and other environmental stressors on ecosystems for the past 20 years. He came to the Office of Research and Development 3 years ago from the Fish and Wildlife Service, where he worked in the environmental contaminants program. Dr. Andreasen is active in the Society of Environmental Toxicology and Chemistry and currently serves as the president of the Chesapeake-Potomac Regional Chapter. During his free time he enjoys genealogy, family history research, and woodworking. His 17 grandchildren enjoy the toys and other creations he makes in his shop. One of his goals in life is to someday return to the wide open spaces of the West.

Lawrence P. Burkhard, Ph.D.

Dr. Burkhard is a research chemist in the Ecological Toxicology Research Branch of EPA's National Health and Environmental Effects Research Laboratory, Mid-Continent Ecology Division, in Duluth, Minnesota. He received his B.S. in Civil Engineering from Pennsylvania State University and his M.S. and Ph.D. in Water Chemistry from the University of Wisconsin-Madison. His research interests include the behavior and effects of bioaccumulative organic contaminants in aquatic ecosystems, analytical methodologies for the detection and quantification of known and unknown organic contaminants in environmental samples, and toxicologically based analytical methodologies for the detection and identification of unknown toxicants in environmental samples.

Peter M. Chapman, Ph.D.

Dr. Chapman is a senior principal at EVS Environment Consultants, North Vancouver, British Columbia. He received his B.Sc. in Marine Biology, his M.Sc. in Biological Oceanography, and his Ph.D. in Benthic Ecology from the University of Victoria (1979). His experience and expertise since graduation have centered on ecotoxicology and aquatic ecology, which he has combined into integrative assessments such as the Sediment Quality Triad. He has published over 90 peer-reviewed journal publications and book chapters and over 200 technical reports on a wide variety of subjects, including sediment bioaccumulation.

David W. Charters, Ph.D.

Dr. Charters is an environmental scientist with EPA's Environmental Response Team located in Edison, New Jersey. He attended undergraduate school at Syracuse University and continued his education at the State University of New York at Binghamton, where he received a Doctorate in Biology specializing in Environmental Pathology. Dr. Charters' dissertation study was conducted at Love Canal, Niagara Falls, New York, and he investigated the demographic structure of the small mammal populations in the area and the histopathological responses of these indigenous mammals. Dr. Charters joined EPA's Environmental Response Team in 1985 and has worked on ecological risk assessment since then in the Superfund program. He has conducted investigations on many of the National Superfund sites and has conducted studies and investigations internationally. He is presently completing Superfund guidance on conducting ecological risk assessments and is involved in the reauthorization of the Superfund law.

John P. Connolly, Ph.D., P.E.

Dr. Connolly is a principal engineer with HydroQual, Inc., in Mahwah, New Jersey. He received his B.E. in Civil Engineering and his M.E. in Environmental Engineering from Manhattan College. His Ph.D. in Environmental Health engineering is from the University of Texas at Austin. Prior to his Ph.D. studies, Dr. Connolly worked for more than 2 years at Manhattan College on the development and application of a model of eutrophication in Lake Erie. His thesis research on the fate of sediment-associated hydrophobic organics was conducted at the U.S. EPA Environmental Research Lab in Gulf Breeze, Florida, where he also was involved in

microcosm studies of the transport and degradation of toxic organics in sediment. Upon completion of his Ph.D., he accepted a faculty position in the Environmental Engineering and Science Program at Manhattan College. During his 14 years at Manhattan, Dr. Connolly conducted research in several areas, including the development of models of toxic chemical fate and bioaccumulation, eutrophication modeling, and modeling the fate and effects of genetically engineered microorganisms introduced to surface waters. While at Manhattan he consulted with HydroQual, Inc., on a wide range of problems. In June 1994 he left Manhattan to join HydroQual full-time. His recent work has included several bioaccumulation projects. Among these are detailed analysis of bioaccumulation field data for EPA and modeling assessments of DDE bioaccumulation from sediments on the Palos Verdes Shelf and PCB bioaccumulation from sediments in Green Bay.

Phillip M. Cook, Ph.D.

Dr. Cook is the Acting Chief of the Ecological Toxicology Branch at the Mid-Continent Ecology Division of EPA's National Health and Environmental Effects Laboratory in Duluth, Minnesota. Dr. Cook received a B.S. in Chemistry from Tufts University, an M.S. in Geochemistry from Colorado School of Mines, and a Ph.D. in Physical Inorganic Chemistry from the University of Wisconsin. Following graduate school in 1972, he accepted a position as a research chemist at the U.S. EPA National Water Quality Laboratory at Duluth, where his initial work involved evaluation of risks associated with mining wastes in Lake Superior. His management positions since then have included Chief of the Hazardous Waste Research Branch and Associate Director for Research Operations. Dr. Cook has diverse research experience that includes effects of fine particles such as asbestos; sediment contaminant bioavailability; ecological risks of organochlorine chemicals, especially PCDDs, PCDFs and PCBs; and methods for ecological risk assessment in the Great Lakes.

Judy L. Crane, Ph.D.

Dr. Crane is the Team Leader of the Toxics Unit in the Division of Water Quality at the Minnesota Pollution Control Agency (MPCA) in St. Paul, Minnesota. She received a B.S. in Animal Ecology from Iowa State University, an M.S. in Ecology and Behavioral Biology from the University of Minnesota-Minneapolis, and a Ph.D. in Water Chemistry from the University of Wisconsin-Madison. Her professional work experience includes contractual work with the U.S. EPA Environmental Research Laboratories in Duluth, Minnesota (1983-1985) and Athens, Georgia (1990-1992), as well as consulting work with EVS Consultants (1992-1995) in Vancouver, British Columbia. Dr. Crane was actively involved in the Great Lakes National Program Office's Assessment and Remediation of Contaminated Sediments (ARCS) Program from 1990 to 1995, where she was responsible for conducting baseline and comparative human health risk assessments for selected Areas of Concern. She has also been involved in developing ecological risk assessment guidance for contaminated sites in Canada and developing interim sediment quality criteria for the Province of British Columbia. Dr. Crane is currently working as a research scientist for the MPCA, where she is leading several federally funded contaminated sediment investigations in the Duluth/Superior Harbor.

Tudor T. Davies, Ph.D.

Dr. Davies joined EPA in 1972, in the Office of Research and Development (ORD). From 1975 to 1979 he was the Deputy Laboratory Director of the ORD Gulf Breeze Environmental Research Laboratory in Gulf Breeze, Florida. He then became the Director of the Narragansett Environmental Research Laboratory from 1979 to 1983; during that time he also served as the director of the EPA Chesapeake Bay Program. Prior to his current position as Director of the Office of Science and Technology, he was the Director for the Office of Water's Office of Marine and Estuarine Protection for 7 years. Dr. Davies attended the University of Wales in Swansea, and he holds a Bachelor of Science and a Doctorate in Geology. He was an Associate Professor of Geology at the University of South Carolina.

Mario P. Del Vicario, M.S.

Mr. Del Vicario serves as the Chief of the Place-Based Protection Branch (PBPB) for Region 2 of the U.S. Environmental Protection Agency. He began his employment at EPA in 1985 as the Assistant Chief of the Marine and Wetlands Protection Branch and was Chief of that branch from 1987 to 1996. Prior to his service with EPA, Mr. Del Vicario was employed by the U.S. Army Corp of Engineers in the New York District from 1975 to 1985, where he served as the Assistant Chief of the Environmental Analysis Branch. He was a microbiologist with the NYC Health Department from 1973 to 1975.

Mr. Del Vicario received a B.A. in biology from Adelphi University in 1971 and an M.S. in Marine Science in 1973. From 1980 to 1986, he was enrolled in a Ph.D. program in Ecology in the City University system.

Dominic M. Di Toro, Ph.D.

Dr. Di Toro is a principal consultant for HydroQual, Inc., in Mahwah, New Jersey. He received his B.E.E. in Electrical Engineering from Manhattan College, and his M.A. and Ph.D. in Electrical Engineering and Civil and Geological Engineering from Princeton University. Dr. Di Toro has specialized in the development and application of mathematical and statistical models to

stream, lake, estuarine, and coastal water quality and sediment problems. He has published over sixty technical papers and has participated as Expert Consultant, Principal Investigator, and Project Manager on numerous water quality studies for industry and governmental agencies. Recently, his work has focused on the development of sediment quality criteria and the development of eutrophication and sediment flux models for nutrients and metals.

L. Jay Field, M.S.

Mr. Field received a bachelor's degree from the University of Michigan and an M.S. in Fisheries from the University of Washington. Since 1986, he has worked as a marine biologist for the National Oceanic and Atmospheric Administration (NOAA), Office of Ocean Resources Conservation and Assessment, Hazardous Materials Response and Assessment Division, Coastal Resources Coordination Branch in Seattle, Washington. His responsibilities include providing technical support to NOAA Coastal Resource Coordinators and EPA in the evaluation of ecological risk to coastal marine resources from hazardous waste sites.

Christopher G. Ingersoll, Ph.D.

Dr. Ingersoll is a fisheries biologist with the U.S. Geological Survey at the Environmental Contaminants Research Center in Columbia, Missouri. He received his bachelor's (1978) and master's (1982) from Miami University in Oxford, Ohio, and his doctorate (1986) from the University of Wyoming in Laramie, Wyoming. His current research is focused on investigating the toxicity and bioavailability of contaminates in sediment. He has coordinated the development of chronic toxicity methods for the amphipod *Hyalella azteca* that have been used to evaluate contaminated sediments in several areas, including the Great Lakes, the upper Mississippi River, and the Clark Fork River in Montana. A second area of research that he has been working on with EPA is developing a sediment bioaccumulation test with the oligochaete *Lumbriculus variegatus*. Dr. Ingersoll also chairs the ASTM Subcommittee E47.03 on Sediment Toxicology. This subcommittee, in coordination with EPA and Environment Canada, has developed a variety of standard methods for evaluating sediment toxicity and bioaccumulation.

Michael J. Kravitz, M.A.

Mr. Kravitz is a biologist in the Standards and Applied Science Division of the Office of Science and Technology at U.S. EPA Headquarters. In this capacity he consults with, coordinates, and provides technical support to EPA personnel and programs related to contaminated sediment issues. As co-chair of EPA's Bioaccumulation Analysis Workgroup, Mr. Kravitz is helping to lead the development of an EPA report on the current status of bioaccumulation testing and interpretation for the purpose of sediment quality assessment. He has been responsible for the development of a number of EPA and EPA/Army Corps of Engineers guidance documents, including the *Draft Inland Testing Manual* and *QA/QC Guidance for Sampling and Analysis of Sediments, Water, and Tissues for Dredged Material Evaluations*. Prior to joining EPA, Mr. Kravitz worked in the field of marine/estuarine benthic ecology at laboratories in New York, Oregon, Florida, Virginia, and Massachusetts. Mr. Kravitz received a B.S. in Biology from the State University of New York at Stony Brook, and an M.A. in Marine Science (major in Biological Oceanography) from the College of William and Mary School of Marine Science in Gloucester Point, Virginia.

Peter F. Landrum, Ph.D.

Dr. Landrum is the head of the Biogeochemical Sciences Division of NOAA's Great Lakes Environmental Research Laboratory in Ann Arbor, Michigan, where he oversees research on aquatic contaminants, biogeochemistry, and ecosystems studies directed to examine the effects of anthropogenic impacts on the Great Lakes as well as the day-to-day administration for the division. He received his B.S. in Chemistry from California State College, San Bernadino, and Ph.D. in Pharmacology and Toxicology from the University of California, Davis. He spent the next 2 years working as a research associate for the University of Georgia at the Savannah River Ecology Laboratory, Aiken, South Carolina. His research focused on the fate, transport, and bioaccumulation of polycyclic aromatic hydrocarbons in freshwater stream systems. He then moved to the Great Lakes Environmental Research Laboratory as a research chemist to conduct research in the bioavailability and bioaccumulation of organic contaminants by aquatic invertebrates with an emphasis on the benthos. Over the last 15 years, his research has examined the role of dissolved organic matter on the bioavailability of waterborne contaminants and the influence of sediment characteristics on sediment-associated contaminants. More recent work has examined the utility of whole-body residue levels to determine the dose required to produce contaminant toxicity. In addition to his research, Dr. Landrum served as a part-time instructor in environmental toxicology at Eastern Michigan University.

Alex Lechich, M.S.

Mr. Lechich has been an environmental scientist with EPA Region 2 since 1988, principally working on ocean disposal, contaminated sediment, and dredging issues. Prior to that, he worked for 3 years with the New York District Corps of Engineers in its regulatory program. Mr. Lechich received a B.S. in Biology from the State University of New York at Stony Brook in 1983 and an M.S. in Marine Environmental Science from Stony

Brook's Marine Sciences Research Center in 1984. He served in the U.S. Army for 3 years in the late 1970s, where he collected and analyzed ambient air samples from nerve gas de-commissioning areas on a base near Denver. Prior to completing his undergraduate and graduate work, Mr. Lechich worked for several industrial chemical companies, mainly doing organic chemistry product development and quality control. Following his graduate work, he worked for the New York Power Authority.

Henry Lee II, Ph.D.

Dr. Lee is with the Pacific Northwest Estuarine Ecosystem Study of the Western Ecology Division of EPA, located at the Newport, Oregon, laboratory. He received his B.S. in Biology from Rollins College and his Ph.D. in Marine Sciences from the University of North Carolina. After a postdoctoral position at the University of Maryland and an NRC postdoctoral position with EPA, Dr. Lee joined the EPA staff at Newport, Oregon. He spent the next decade working on bioaccumulation and contaminated sediments, including investigations on the application of equilibrium partitioning and toxicokinetic models to benthic invertebrates, and the development of bioassay methods. His work on contaminated sediments culminated when Dr. Lee directed an ecological risk assessment of sediment-associated DDT. During this period, Dr. Lee also was the Program Manager for EPA's Marine Stratospheric Ozone Depletion Program. He is presently directing a research program to assess the cumulative effects of chemical and nonchemical (e.g., sedimentation, introduced species) stressors on an ecosystem scale.

Lynn Scott McCarty, Ph.D.

Dr. McCarty operates L.S. McCarty Scientific Research & Consulting, an ecotoxicological consulting business based in Oakville, Ontario, Canada. He received B.S. and M.S. degrees from Brock University and a Ph.D. from the University of Waterloo. He has spent over 18 years examining various aspects of environmental contamination, ecotoxicology, and environmental risk assessment. This included a number of years as an environmental scientist and ecological studies group manager at MacLaren-Plansearch (Lavalin), scientific consultant for the Health Studies Service of the Ontario Ministry of Labour, and senior scientist with CanTox, Inc. He has been involved in a wide variety of projects examining environmental impacts and/or human health effects for an assortment of situations and contaminants/stresses. This included the production or critical review of a number of air and water quality guidelines, as well as risk assessments in Canada and the USA.

Dr. McCarty's scientific interests include Quantitative Structure-Activity Relationships (QSAR), toxicokinetics, mixture toxicity, residue-based potency estimation, and risk assessment. He has been an invited expert dealing with human and environmental health at a number of workshops sponsored by CNTC, SETAC, U.S. EPA, and U.S. Army Corps of Engineers. In addition to reports for clients, he continues to publish in the primary scientific literature, contribute to book chapters, and make presentations at professional scientific meetings, as well as in courts, regulatory hearings, and public meetings.

David R. Mount, Ph.D.

Dr. Mount is a research fishery biologist with EPA's Office of Research and Development, Mid-Continent Ecology Division, Duluth, Minnesota. He received his B.A. in Biology with a statistics concentration from St. Olaf College in Northfield, Minnesota, and his Ph.D. in Zoology and Physiology (emphasis in aquatic toxicology) from the University of Wyoming. After a year of postdoctoral research at the Fish Physiology and Toxicology Laboratory at the University of Wyoming, Dr. Mount worked for 5 years at ENSR Consulting and Engineering as a senior aquatic toxicologist and as the manager of the Environmental Toxicology Department. Dr. Mount joined the federal government in 1993 as the Deputy Chief Biologist at the National Biological Service's National Fisheries Contaminant Research Center, before transferring to his current position with EPA. Dr. Mount's research interests include effluent and sediment toxicology, and the effects of major ions on freshwater organisms. His current research centers on sediments, including the development of Toxicity Identification Evaluation (TIE) procedures and evaluation of the bioavailability of sediment contaminants.

Wayne R. Munns, Jr., Ph.D.

Since 1995, Dr. Munns has been a Research Ecologist (Ecological Risk Assessor) and Leader of the Ecological Significance Team in EPA's National Health and Environmental Effects Research Laboratory, Narragansett, Rhode Island, which is responsible for conducting ecological effects research to reduce the uncertainties associated with risk assessment. He received his B.A. in Biology from the University of Washington and his Ph.D. in Biological Sciences from the University of Rhode Island. In 1983, he joined Science Applications International Corporation, ultimately accepting positions as senior scientist, division manager, and assistant vice president. Over the past 13 years, Dr. Munns has served as principal investigator in a number of marine and estuarine ecological risk assessment case studies involving contaminated sediments,

and has conducted research evaluating the effects of chemical stressors on the population dynamics of aquatic organisms.

Thomas M. Murray, M.S.

Mr. Murray is Chief of the Exposure Assessment Branch in the Office of Pollution Prevention and Toxics at EPA Headquarters. In this capacity, he is responsible for integrating the assessment of total chemical exposure to humans and the environment in support of OPPT's regulatory and nonregulatory program activities; supporting OPPT's testing and existing chemical programs by providing integrated assessments of chemical exposure to humans and the environment; providing scientific assessment of total chemical exposure, including chemical and biological fate; and providing exposure assessment and project management support to various OPPT Design for the Environment and Pollution Prevention Program activities. Mr. Murray has been with the Office of Pollution Prevention and Toxics since 1985. Prior to that he spent 14 years with EPA's Office of Water. He received his B.S. in Biology from Mt. St. Marys' College and an M.S. in Biology from the American University.

Dorothy E. Patton, Ph.D.

Dr. Patton holds several positions at the U.S. Environmental Protection Agency. She is the Executive Director of EPA's Science Policy Council, a new Agency organization established to address significant science policy issues that go beyond program and regional boundaries. She also directs the Office of Science Policy and the Office of Regulatory and Science Integration. From 1985 through July 1994, Dr. Patton was the Executive Director of EPA's Risk Assessment Forum, a standing committee of senior EPA scientists charged with developing Agency-wide guidance on selected risk science issues. She also chaired that group from 1989 to 1995. Dr. Patton began her EPA career in 1976 as an attorney in the Office of General Counsel, where she worked on legal and scientific issues arising under the laws relating to pesticides, toxic substances, and the air program. Before joining EPA, Dr. Patton was an Assistant Professor of Biology in the City University of New York (York College), and she did postdoctoral research in cellular and developmental biology at the Albert Einstein College of Medicine in New York. She has a J.D. from Columbia University School of Law, a Ph.D in developmental biology from the University of Chicago, and a bachelor's degree in chemistry from the University of Wisconsin.

Robert L. Paulson, M.S.

Mr. Paulson is an environmental toxicologist in the Water Quality Modeling Section of the Wisconsin Department of Natural Resources' Bureau of Watershed Management. He received his B.S. in Water Resources from the University of Wisconsin-Stevens Point and his M.S. in Fisheries from the University of Missouri. Mr. Paulson's thesis research focused on field validation and predictability of laboratory methods of assessing the effects of contaminants. Prior to joining WDNR, Mr. Paulson was a staff toxicologist with The Johns Hopkins University, Applied Physics Laboratory, where his work centered on effluent and single-chemical estuarine and freshwater toxicity testing. Mr. Paulson also spent a brief period of time in private consulting. Mr. Paulson joined WDNR as the coordinator of the whole effluent toxicity testing for the point source discharge permit program. His current assignment is to coordinate WDNR's technical staff in conjunction with the efforts of the Fox River Coalition to develop a whole-river sediment strategy.

Amy E. Pelka, M.S.

Ms. Pelka is an environmental health scientist in the Office of Strategic Environmental Analysis in EPA Region 5, Chicago, Illinois. She received her B.S. in Zoology (specialization in cellular biology) from the University of Wisconsin-Madison and her M.S. in Microbiology/Immunology from Northwestern University. After graduate school she began working with Region 5, in the Water Division, on sediments and water quality standards. She has worked for the Water Division and now the Strategic Analysis office for the past 5 years, primarily as a human health risk assessor. She has worked on several Superfund and other enforcement cases, as well as various technical and risk policy issues. Her primary areas of focus have been sediment sites, bioaccumulation, risk assessment policy, and community-based environmental protection.

James F. Pendergast, M.S.

Mr. Pendergast is currently Acting Director of the NPDES Permits Division at U.S. EPA Headquarters. In this capacity, he directs national activities and initiatives for the NPDES permits, pretreatment, and sludge programs. He has worked on reauthorization of the Clean Water Act as a special assistant to EPA's Assistant Administrator for Water and as Chief of the Water Quality and Industrial Permits Branch in the NPDES Permits Division. He has also worked in EPA

Region 6 in the NPDES permit and Superfund programs. Prior to joining EPA, Mr. Pendergast was a project manager at Limno-Tech, Inc., in Michigan, where he developed models of water quality impacts from nonpoint and point sources. He received a B.S. in Environmental Engineering and an M.S. in Water Resources Engineering from the University of Michigan and is a registered engineer.

Richard J. Pruell, Ph.D.

Dr. Pruell is a research chemist at the Atlantic Ecology Division (Narragansett, Rhode Island) of EPA's National Health and Ecological Effects Laboratory. He received a B.A. in Biology from Merrimack College, an M.S. in Marine Biology from Southeastern Massachusetts University, and a Ph.D. in Chemical Oceanography from the Graduate School of Oceanography of the University of Rhode Island. Since receiving his degree in 1984, Dr. Pruell has worked at the Narragansett EPA laboratory on issues related to the biogeochemistry of organic contaminants in marine ecosystems.

Mary C. Reiley, M.S.

Ms. Reiley is the Team Coordinator for the Ecological Criteria Team within the Health and Ecological Criteria Division. She received her B.S. in Biology from the College of William and Mary and her M.S. in Environmental Biology from George Mason University. Ms. Reiley joined the Office of Science and Technology in 1991 to coordinate the sediment quality criteria program. Prior to that she spent 7 years in the Office of Water Enforcement Program, where she was the Team Leader for the Toxics Team and involved in a variety of issues including permitting limits set below detection, whole effluent toxicity, and compliance negotiations. Her current role as Ecological Criteria Team Coordinator maintains her involvement in methodology and criteria development and technical support for aquatic life assessments, sediment quality assessment, and whole effluent toxicity testing. The team works closely with the Surface Water Assessment Team, on which Ms. Reiley serves as the program lead for integrated pathways analysis for criteria development.

Norman I. Rubinstein, M.S.

Mr. Rubinstein is the Acting Division Director for the National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, in Narragansett, Rhode Island. He received his B.S. in Biology from the City College of New York and his M.S. in Marine Science from the University of West Florida. He joined EPA in 1976 as a research aquatic biologist at the Environmental Research Laboratory, Gulf Breeze, Florida. In 1983 Mr. Rubinstein joined the research staff at the Environmental Research Laboratory, Narragansett, Rhode Island where he focused his studies on the bioavailability of contaminants in sediments. He became Chief of the Research Exposure Branch at the Narragansett Laboratory in 1989 and was appointed Deputy Director for Research in 1994. He has been the acting Division Director at Narragansett since the ORD laboratory reorganization in 1995. Mr. Rubinstein has been associated with EPA's research efforts in support of the Ocean Disposal Program since the development of the original dredged material testing manual (the "Green Book") in 1976.

Burt K. Shephard, M.S.

Mr. Shephard is a senior ecotoxicologist with URS Greiner, Inc., in Seattle, Washington. He received his B.S. in Chemistry and M.S. in Environmental Health from Purdue University. He also has completed course work toward a Ph.D. in Fisheries Biology from Iowa State University, where he is a past recipient of the Society of Environmental Toxicology and Chemistry Pre-Doctoral Fellowship Award. Mr. Shephard has nearly 20 years of experience working to evaluate the environmental impacts or ecological risks of metals, nutrients, PCBs, chlorinated insecticides, and PAHs for both environmental consulting firms and EPA. This work has taken him throughout the United States, as well as to several foreign countries. Mr. Shephard's current work involves defining ecological risks to aquatic biota from bioaccumulated chemicals.

Elizabeth Southerland, Ph.D.

Dr. Southerland is Acting Director of the Standards and Applied Science Division in EPA's Office of Water in Washington, D.C. Dr. Southerland received her Ph.D. in Environmental Science and Engineering from Virginia Polytechnic Institute and State University. Over the past 25 years, she has held a variety of positions in state and local government, consulting engineering, and EPA. Dr. Southerland is currently the Acting Director of EPA's Standards and Applied Science Division, which is responsible for the national water quality standards program and environmental assessments in support of water quality- and sediment quality-based controls.

Robert V. Thomann, Ph.D.

Dr. Thomann is Professor of Environmental Engineering at Manhattan College. He received a bachelor's degree in Civil Engineering from Manhattan College in 1956, a master's degree in Civil Engineering from New York University in 1960, and a Ph.D. in Physical Oceanography from New York University in 1963. He spent 10 years with the U.S. Public Health Service from 1956 to 1966, during which time he was Technical Director for the Delaware Estuary Study. Dr. Thomann joined the faculty of Manhattan College in 1966. His work has been in mathematical modeling of water quality and ecosystem

fate, transport, and transformation processes. He has published about 50 papers and two books, has lectured at a variety of institutions, and has received several honors for his research work. Dr. Thomann has worked on many major water bodies in the United States and abroad and is currently doing research on modeling bioaccumulation processes including the transfers of PCBs in the Hudson estuary aquatic food web.

Nelson A. Thomas, B.S.

Mr. Thomas received his B.S. in Natural Resources (fisheries) from the University of Michigan. He was the limnologist for the State of Ohio when Lake Erie was discovered to be greatly affected by eutrophication. He then joined the U.S. EPA and served as chief biologist for the Office of Enforcement and Standards. After 10 years, he assumed a research position to conceive, supervise, and conduct large lakes research. Currently, as senior advisor for national programs with EPA, he develops and coordinates freshwater research for the control of toxic chemicals. His areas of responsibility include water quality criteria, complex effluent toxicity testing, sediment quality criteria, and integrated watershed assessment for ecosystem protection.

Marc L. Tuchman, Ph.D.

Dr. Tuchman received his B.S. in Biology from Colgate University and his Ph.D. in Natural Resources from the University of Michigan. He has been with EPA for the past 12 years, working in the Waste Division and Water Division prior to his current position in the Great Lakes National Program Office (GLNPO). Over the past 10 years, Dr. Tuchman has been involved in a variety of contaminated sediment and dredging issues. For 6 years he was actively involved in EPA's Assessment and Remediation of Contaminated Sediments (ARCS) Program, having served as Program Manager for the latter 2 years of the program. Dr. Tuchman currently serves as team leader of GLNPO's Sediment Assessment and Remediation Team, where he is responsible for coordinating GLNPO's sediment assessment and remediation activities in the Great Lakes basin.

Laura B. Weiss, M.P.H.

Ms. Weiss is a toxicologist who has been the project manager for the development of health-based sediment criteria for the Washington State Department of Ecology in Olympia, Washington, for the past 3 years. She received a B.S. in Human Ecology/Public Policy from Rutgers University and an M.P.H. in Environmental Health from the University of California, Berkeley. Prior to working for the Department of Ecology, Ms. Weiss worked as a pesticide policy specialist and organizer for the environmental and consumer group Public Citizen in Washington, D.C.

Lawrence J. Zaragoza, Ph.D.

Dr. Zaragoza received his doctorate in Environmental Science and Engineering from the University of California at Los Angeles in 1982. He started with EPA in 1979 in the Office of Air Quality Planning and Standards. Today, Dr. Zaragoza works for EPA's Office of Emergency and Remedial Response, which administers the Headquarters component of the Superfund Program. His responsibilities at EPA have included representing the OERR on issues related to contaminated sediments, the identification of research needs for the air and hazardous waste programs, and Project Officer responsibilities for the Interagency Agreements with the Agency for Toxic Substances and Disease Registry (ATSDR) and the National Institute for Environmental Health Statistics (NIEHS). Dr. Zaragoza has experience in drafting of EPA regulations (e.g., National Ambient Air Quality Standards, air toxics, revisions to the Hazard Ranking System) and guidances (e.g., Risk Assessment Guidance for Superfund and Capacity Assurance Guidance).

Maurice Zeeman, Ph.D.

Dr. Zeeman is Chief of the Environmental Effects Branch in the Health and Environmental Review Division of OPPT at EPA Headquarters. He directs 16 professional experts in their assessment of the potential ecological hazards and risks of thousands of new and existing industrial chemicals evaluated under the Toxic Substances Control Act (TSCA). He also teaches a course in environmental toxicology in the Department of Pharmacology and Toxicology for the Graduate School at the National Institutes of Health. Prior to joining EPA, he was an expert toxicologist for the U.S. Food and Drug Administration, where he evaluated the effects of chemicals on humans (under the FD&C Act) and on the environment (under NEPA). Dr. Zeeman received a Master's in Zoology/Ecology from UCLA and a Ph.D. in Biology from Utah State University.

National Sediment Bioaccumulation Conference

September 11-13, 1996
Hyatt Regency
Bethesda, Maryland

FINAL AGENDA

Sponsored by:
Office of Water/Office of Science and Technology
Office of Research and Development

Wednesday, September 11

7:30-8:30 ***Registration***

8:30-8:45 ***Welcome and Introduction***
Dr. Elizabeth Southerland and
Dr. Thomas Armitage
Office of Science and Technology
U.S. Environmental Protection Agency

BIOACCUMULATION OVERVIEW AND APPROACHES

8:45-9:15 ***Contaminated Sediments: State of the Science and Future Research Directions***
Dr. Gilman D. Veith, Associate Director for Ecology
National Health and Environmental Effects Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency

Field and Laboratory Methods for Measuring Bioaccumulation
Moderator: Dr. Peter Chapman
EVS Environmental Consultants, Ltd.

9:15-9:40 ***Methods for Assessing Sediment Bioaccumulation in Marine/Estuarine Benthic Organisms***
Dr. Henry Lee
Office of Research and Development
U.S. Environmental Protection Agency

9:40-10:05 ***Methods for Assessing Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates***
Dr. Christopher Ingersoll
Environmental Contaminants Research Center
U.S. Geological Survey

10:05-10:20 Break

10:20-10:45 ***Kinetic Models for Assessing Bioaccumulation***
Dr. Peter Landrum
Great Lakes Environmental Research Laboratory
National Oceanic and Atmospheric Administration

10:45-11:15 ***Discussion/Question and Answer Session***

11:15-12:30 Lunch

Interpretation and Applications of Bioaccumulation Results
Moderator: Dr. Richard Pruell
Office of Research and Development
U.S. Environmental Protection Agency

12:30-12:55 ***Reference Sediment Approach for Determining Sediment Contamination***
Mr. Norman Rubinstein
Office of Research and Development
U.S. Environmental Protection Agency

12:55-1:20 ***Development of Tissue Residue Threshold Values***
Dr. David Mount
Office of Research and Development
U.S. Environmental Protection Agency

1:20-1:45 ***Use of Tissue Residue Data in Exposure and Effects Assessments for Aquatic Organisms***
Mr. L. Jay Field
National Ocean Service
National Oceanic and Atmospheric Administration

Final Agenda

1:45-2:10 ***Comments on the Significance and Use of Tissue Residues in Sediment Toxicology and Risk Assessment***
Dr. Lynn McCarty
LS McCarty Scientific Research & Consulting

2:10-2:35 ***Quantification of Ecological Risks to Aquatic Biota from Bioaccumulated Chemicals***
Mr. Burt Shephard
URS Greiner, Inc.

2:35-3:05 ***Discussion/Question and Answer Session***

3:05-3:25 Break

Modeling Bioavailability of Sediment Contaminants
Moderator: Mr. Nelson Thomas
Office of Research and Development
U.S. Environmental Protection Agency

3:25-3:50 ***Equilibrium Partitioning and Organic Carbon Normalization***
Dr. Dominic Di Toro
HydroQual, Inc.

3:50-4:15 ***Estimating Bioaccumulation Potential in Dredged Sediment Regulation***
Dr. Victor McFarland
Waterways Experiment Station
U.S. Army Corps of Engineers

4:15-4:40 ***Development of Bioaccumulation Factors for Protection of Fish and Wildlife in the Great Lakes***
Dr. Philip Cook
Office of Research and Development
U.S. Environmental Protection Agency

4:40-5:00 ***From Modeling to Criteria: Integrated Approach to Criteria Development***
Ms. Mary Reiley
Office of Science and Technology
U.S. Environmental Protection Agency

5:00-5:30 ***Discussion/Question and Answer Session***

6:00-7:00 ***Welcome Reception***

Thursday, September 12

Food Chain Models and Bioenergetics
Moderator: Dr. Lawrence Burkhard
Office of Research and Development
U.S. Environmental Protection Agency

8:30-9:00 ***Food Chain Models for Predicting Bioaccumulation***
Dr. Frank Gobas
Simon Fraser University

9:00-9:30 ***Use of Food Web Models to Evaluate Bioaccumulation Data***
Dr. John Connolly
HydroQual, Inc.

9:30-10:00 ***Bioaccumulation Modeling of PCBs in the Hudson Estuary: A Review and Update***
Dr. Robert Thomann
Manhattan College

10:00-10:30 ***Discussion/Question and Answer Session***

10:30-10:45 Break

BIOACCUMULATION AND RISK ASSESSMENT

10:45-11:15 ***Risk Assessment Overview***
Dr. Dorothy Patton
Office of Research and Development
U.S. Environmental Protection Agency

Human Health-Based Risk Assessment
Moderator: Dr. Marc Tuchman
Great Lakes National Program Office
U.S. Environmental Protection Agency

11:15-11:45 ***Methodology for Assessing Human Health-Based Risks***
Dr. Judy Crane
Minnesota Pollution Control Agency

11:45-1:00 Lunch

Final Agenda

Case Studies:

1:00-1:45 ***Bioaccumulation Models and Applications: Setting Sediment Cleanup Goals in the Great Lakes***
Ms. Amy Pelka
Region 5
U.S. Environmental Protection Agency

1:45-2:15 ***Use of Human Health- and Ecological-Based Goals in Developing a Whole River Sediment Strategy: Fox River, WI***
Mr. Robert Paulson
Wisconsin Department of Natural Resources

2:15-2:45 ***Development of Health-Based Sediment Criteria for Puget Sound***
Ms. Laura Weiss
Washington Department of Ecology

2:45-3:15 ***Development of Bioaccumulation Guidance for Dredged Material Evaluations in EPA Region 2***
Mr. Alex Lechich
Region 2
U.S. Environmental Protection Agency

3:15-3:45 ***Discussion/Question and Answer Session***

3:45-4:00 Break

Ecological-Based Risk Assessment
Moderator: Dr. James Andreasen
Office of Research and Development
U.S. Environmental Protection Agency

4:00-4:30 ***Use of Bioaccumulation Data in Aquatic Life Risk Assessment***
Dr. Wayne Munns
Office of Research and Development
U.S. Environmental Protection Agency

4:30-5:00 ***Wildlife Risk Assessment***
Dr. David Charters
Office of Solid Waste and Emergency Response
U.S. Environmental Protection Agency

5:00-5:30 ***Discussion/Question and Answer Session***

Friday, September 13

BIOACCUMULATION RESULTS AND DECISION-MAKING

Integrating Bioaccumulation Results into EPA's Decision-Making Process
Moderator: Dr. Elizabeth Southerland
Office of Science and Technology
U.S. Environmental Protection Agency

8:30-8:40 ***Opening Remarks***
Dr. Elizabeth Southerland
Office of Science and Technology
U.S. Environmental Protection Agency

8:40-9:00 ***Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment: Status and Needs***
Mr. Michael Kravitz
Office of Science and Technology
U.S. Environmental Protection Agency

9:00-10:30 ***Panel Presentations***

- Superfund Program
  Dr. Lawrence Zaragoza
- NPDES Program
  Mr. James Pendergast
- Office of Pollution Prevention and Toxics
  Mr. Thomas Murray
  Dr. Maurice Zeeman
- Dredged Material Program
  Mr. Craig Vogt
  Mr. Mario Del Vicario

10:30-10:45 Break

10:45-12:00 ***Panel Discussion***

- Common Elements Among Programs
- Barriers to Decision-Making
- Recommendations

12:00-12:30 ***Future Needs/Conference Wrap Up***

National Sediment Bioaccumulation Conference

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