*Appendix B*

# Description of Evaluation Parameters Used in the NSI Data Evaluation

Chapter 2 of this document presented the methodology used in the evaluation of the NSI data. This appendix describes in greater detail the screening values and other parameters used in the NSI data evaluation. The actual parameter values used are presented in Appendix D. For the purpose of discussion, the sediment evaluation parameters have been placed into three groups: (1) those used to assess potential impacts on aquatic life, (2) those used to assess potential impacts on human health, and (3) those used to assess potential impacts on wildlife. The uncertainties associated with the use of these parameters in the NSI data evaluation are discussed in Chapter 5.

## Aquatic Life Assessments

To evaluate the potential threat to aquatic life from chemical contaminants detected in sediments, measured concentrations of contaminants were compared to sediment chemistry screening levels. The results of toxicity tests to indicate the actual toxicity of sediment samples to species of aquatic organisms, when available, were also evaluated for the NSI.

Sediment chemistry screening levels are reference values above which sediment contaminant concentrations could pose a significant threat to aquatic life. Several different approaches, based on causal or empirical correlative methodologies, have been developed for deriving screening levels of sediment contaminants. Each of these approaches attempts to predict contaminant concentration levels that could result in adverse effects to benthic species, which are extrapolated to represent the entire aquatic community for this evaluation. For the purpose of this analysis, the screening levels selected include the following:

- EPA's draft sediment quality criteria (SQCs) for five nonionic organic chemicals, developed using an equilibrium partitioning approach (USEPA, 1992a, 1993a).
- Sediment quality advisory levels (SQALs) for selected nonionic organic chemicals, developed using an equilibrium partitioning approach (USEPA, 1992a, 1993a).
- The sum of simultaneously extracted divalent transition metals concentrations minus the acid-volatile sulfide concentration ([SEM] - [AVS]), also based on an equilibrium partitioning approach.
- Effects range-median (ERM) and effects range-low (ERL) values for selected nonionic organics and metals developed by Long et al. (1995).
- Probable effects levels (PELs) and threshold effects levels (TELs) for selected nonionic organics and metals developed for the Florida Department of Environmental Protection (FDEP, 1994).
- Apparent effects thresholds (AETs) for selected organics and metals developed by Barrick et al. (1988).

The principles behind the development of each of these sediment chemistry screening values are discussed below. The sediment toxicity tests are also briefly described in this section.

## *Equilibrium Partitioning Approaches*

The potential toxicity of sediment-associated nonionic organic chemicals and divalent metals is indicated by the amount of the contaminant that is uncomplexed or freely available in the interstitial (pore) water. The bioavailability and toxicity of nonionic organic chemicals and divalent metals in sediments are mediated by several physical, chemical, and biological factors, including sediment grain size, particulate and dissolved organic carbon, and sulfide produced by sulfate-reducing bacteria (Di Toro et al., 1991, 1992; Howard and Evans, 1993). For nonionic organic chemicals, sorption to the organic carbon dissolved in the interstitial water and bound to sediment particles is the most important factor affecting bioavailability. Sulfide, specifically the reactive solid-phase sulfide fraction that can be extracted by cold hydrochloric acid (acid-volatile sulfide, or AVS), appears to control the bioavailability of most divalent metal ions because of the sulfide ions' high affinity for divalent metals, resulting in the formation of insoluble metal sulfides in anaerobic sediments.

When the concentrations of nonionic organic chemicals and divalent metals were measured in pore water extracted from spiked sediment and field-collected sediment used in toxicity tests, the biological effects observed in those tests occurred at similar pore water concentrations, even when different types of sediments were used, typically within a factor of 2 (Di Toro et al., 1991, 1992). Biological effects also occurred at similar concentrations in tests with different sediment types containing different amounts of organic carbon (OC) when (1) the dry-weight sediment concentrations of nonionic organic chemicals were normalized for organic carbon content (i.e., µg chemical/$g_{OC}$) and (2) when the difference between molar concentrations of simultaneously extracted metals ([SEM]) in the sediment exceeded the molar concentration of AVS ([AVS]) in the sediments by similar amounts (the mortality of sensitive species increases in the range of 1.5 to 12.5 µmol of SEM per µmol of AVS). Most importantly, the effects concentrations in the sediment could be predicted from the effects concentrations determined in water-only exposures to these chemicals. Most measurements of sediment chemical concentrations are made from whole sediment samples and converted to units of chemical per dry-weight of sediment, because of the difficulties in extracting the pore water. However, when dry-weight concentrations of nonionic organics and metals were used to plot concentration-response curves of the toxicity of different sediments, biological effects occurred at different dry-weight concentrations when measured in different sediments (Luoma, 1983; USEPA, 1993a). To develop criteria or advisory levels for comparing the toxicity of different chemicals in different sediments, it was necessary to examine the role of organic carbon and other complexing factors in the bioavailability of chemicals in sediment.

In sediment, the partitioning of a nonionic organic chemical between organic carbon and pore water and the partitioning of a divalent metal between the solid and solution phases are assumed to be at equilibrium. The fugacity (activity) of the chemical in each of these phases is the same at equilibrium. Fugacity describes mathematically the rates at which chemicals diffuse or are transported between phases (Mackay, 1991). Hence, an organism in the sediment is assumed to receive an equivalent exposure from water only or from any equilibrated phase. The pathway of exposure might include pore water (respiration), sediment carbon (ingestion), sediment organism (ingestion), or a mixture of routes. The biological effect is produced by the chemical activity of the single phase or the equilibrated system (Di Toro et al., 1991). The equilibrium partitioning approach uses this partitioning theory to relate the dry-weight sediment concentration of a particular chemical that causes an adverse biological effect to the equivalent free chemical concentration in pore water and to the concentration sorbed to sediment organic carbon or bound to sulfide. The theoretical causal resolution of chemical bioavailability in relation to chemical toxicity in different sediments differentiates equilibrium partitioning approaches from purely empirical correlative assessment methods (described later in this section).

The processes that govern the partitioning of chemical contaminants among sediments, pore water, and biota are better understood for some kinds of chemicals than for others. Partitioning of nonionic hydrophobic organic compounds between sediments and pore water is highly correlated with the organic carbon content of sediments, but it does not account for all of the toxicity variation observed between sediment and water-only experimental exposures. Other factors that can affect biological responses are not considered in the model. The equilibrium partitioning approach has been tested using only nonionic organic chemicals with octanol/water partition coefficients (log $K_{ow}$s) between 3.8 and 5.3. However, because the theory should be applicable to nonionic organic chemicals with log $K_{ow}$s from 2.0 to 5.5 (Dave Hansen, EPA/ORD-Narragansett, pers. commun., April 17, 1995), nonionic organic chemicals with log $K_{ow}$s in this range were evaluated for the analysis of NSI data. For trace metals, concentrations of sulfides and organic carbon have been identified as important factors that control the phase associations and, therefore, the bioavailability of trace metals in

anoxic sediments. However, models that can use these factors to predict the bioavailability of trace metals in sediments are not fully developed (see below). Mechanisms that control the partitioning of nonionic and nonpolar organic compounds with log $K_{ow}$s of less than 2.0 or greater than 5.5 and polar organic compounds in sediments, and affect their toxicity to benthic organisms, are less well understood. Models for predicting biological effects from concentrations of such compounds have not yet been developed; therefore, these chemicals have not been evaluated using equilibrium partitioning approaches.

*Draft Sediment Quality Criteria*

The equilibrium partitioning model was selected for the development of sediment quality criteria because it can be applied to predict sediment contaminant concentrations below which biological effects are not expected to occur based on the toxicity of individual nonionic organic chemicals—and hence can protect benthic aquatic life in bedded, permanently inundated, or intertidal sediments—while accounting for sediment characteristics that affect the bioavailability of the chemical (Di Toro et al., 1991; USEPA, 1993a). The predominant phase for sorption of nonionic organic chemicals to sediment particles appears to be organic carbon, for sediments in which the fraction of organic carbon ($f_{oc}$) is greater than 0.2 percent.

The partitioning of a chemical between the interstitial water and sediment organic carbon is explained by the sediment/pore water partition coefficient for a chemical, $K_p$, which is equal to the organic carbon content of the sediment ($f_{oc}$) multiplied by the sediment particle organic carbon partition coefficient ($K_{oc}$). $K_p$ is the ratio of the concentration of the chemical in the sediment to the concentration of the chemical in the pore water. Normalizing the dry-weight concentration of the chemical in sediment to organic carbon is as appropriate as using the interstitial water concentration of the chemical because organic carbon in the sediment can also bind the chemical and affect its bioavailability and toxicity. The particle organic carbon partition coefficient ($K_{oc}$) is related to the chemical's octanol/water partition coefficient ($K_{ow}$) by the following equation (Di Toro et al., 1991):

$$\log K_{oc} = 0.00028 + 0.983(\log K_{ow})$$

The octanol/water partition coefficient for each chemical can thus predict the likelihood of the chemical to complex or sorb to organic carbon, when measured with modern experimental techniques that provide the most accurate estimate of this parameter. The concentration of the chemical on sediment particles ($C_s$) is then equal to the dissolved concentration of chemical ($C_d$) multiplied by the organic carbon content of the sediment ($f_{oc}$) and the particle organic carbon partition coefficient ($K_{oc}$), when $f_{oc}$ is greater than 0.2 percent (USEPA, 1993a), thus normalizing the dry-weight sediment concentration of the chemical to the organic carbon content of the sediment.

$$C_s = C_d f_{oc} K_{oc}$$

The criterion for the dissolved concentration of chemical ($C_d$) is derived from the final chronic value (FCV) of EPA's water quality criteria (USEPA, 1985). Freshwater and saltwater FCVs are based on the results of acceptable laboratory tests conducted to determine the toxicity of a chemical in water to a variety of species of aquatic organisms, and they represent the highest levels of a chemical to which organisms can be exposed without producing toxic effects. This level is predicted to protect approximately 95 percent of aquatic life under certain conditions. An evaluation of data from the water quality criteria documents and benthic colonization experiments demonstrated that benthic species have chemical sensitivities similar to those of water column species (Di Toro et al., 1991). Thus, if the concentration of a chemical in sediment, measured with respect to the sediment organic carbon content, does not exceed the sediment quality criterion, then no adverse biological effects from that chemical would be expected (USEPA, 1992a, 1993a).

EPA has developed and published draft freshwater sediment quality criteria (SQCs) for the protection of aquatic life for five contaminants: acenaphthene, dieldrin, endrin, fluoranthene, and phenanthrene. These draft SQCs are based on the equilibrium partitioning approach (USEPA 1993b, c, d, e, f) using the aquatic life water quality criterion final chronic value (FCV, in mg/L) and the partition coefficient between sediment and pore water ($K_p$, in L/g sediment) for the chemical

of interest (Di Toro et al., 1991; USEPA, 1993a). Thus, SQC = $K_p$ FCV. On a sediment organic carbon basis, the sediment quality criterion, $SQC_{oc}$, is:

$$SQC_{oc}(\mu g / g_{oc}) = FCV(\mu g / L) \times K_{oc}(L / kg) \times (10^{-3} kg_{oc} / g_{oc})$$

where:

FCV = EPA aquatic life water quality criterion final chronic value and
$K_{oc}$ = organic carbon-water partitioning coefficient.

$K_{oc}$ is presumed to be independent of sediment type for nonionic organic chemicals, so that the $SQC_{oc}$ is also independent of sediment type. Using a site-specific organic carbon fraction, $f_{oc}$ ($g_{oc}$/g sediment), the $SQC_{oc}$ can be expressed as a sediment-specific value, the SQC:

$$SQC = (SQC_{oc})(f_{oc})$$

*Sediment Quality Advisory Levels*

EPA intends to develop sediment quality criteria for additional chemicals in the future. In the interim, EPA's Office of Science and Technology developed equilibrium partitioning-based sediment quality advisory levels (SQALs) using the following equation:

$$SQAL_{oc}(\mu g / g_{oc}) = [FCV,\ SCV(\mu g / L)] \times K_{oc}(L / kg) \times (10^{-3} kg_{oc} / g_{oc})$$

where:

$SQAL_{oc}$ = calculated sediment quality advisory level;
FCV, SCV = EPA aquatic life chronic criterion (final chronic value, FCV), or other chronic threshold water concentration (secondary chronic value, SCV); and
$K_{oc}$ = organic carbon-water partitioning coefficient.

As noted in Chapter 2, EPA has proposed sediment quality criteria (SQCs) for five chemicals based on the highest quality toxicity and octanol/water partitioning ($K_{ow}$) data, which have been reviewed extensively. This section describes the sources of data used to calculate the values used in the SQAL equations: log $K_{ow}$s (used to derive $K_{oc}$s) and chronic threshold water concentrations. A detailed description of the methods and data used to develop SQALs for specific chemicals using the equilibrium partitioning approach will be published by EPA as a separate document.

SQALs for use in the NSI data evaluation were developed in conjunction with other programs at EPA (established under the Resource Conservation and Recovery Act, RCRA, and the Superfund Amendments and Authorization Act, SARA) to provide the same values for conducting screening-level evaluations of sediment toxicity for these programs. The SQALs (as well as the other sediment chemistry threshold levels) are meant to be used *for screening purposes only*. The screening values are not regulatory criteria, site-specific cleanup standards, or remediation goals. The screening levels are set to be appropriately conservative, so samples that do not exceed the screen would not be expected to exhibit adverse effects from the action of the specific chemical evaluated; exceeding the screening levels does not indicate the level or type of risk at a particular site, but can be used to target additional investigations. EPA's Office of Research and Development (ORD), including staff from Environmental Research Laboratory, Athens, Georgia; Environmental Research Laboratory, Duluth, Minnesota; and Environmental Research Laboratory, Narragansett, Rhode Island, provided guidance and assisted in the development of the necessary values. The SQALs used for the NSI data evaluation are presented with other screening values in Table D-1 of Appendix D.

**Method for Determination of Log $K_{ow}$s.** Log $K_{ow}$ values were initially identified in summary texts on physical-chemical properties, such as Howard (1990) and Mackay et al. (1992a, b) and accompanying volumes. Additional compendia of log $K_{ow}$ values were also evaluated, including De Kock and Lord (1987), Doucette and Andren (1988), Klein et al. (1988), De Bruijn et al. (1989), Isnard and Lambert (1989), Leo (1993), Noble (1993), and Stephan (1993). To supplement these sources, on-line database searches were conducted in ChemFate, TOXLINE, and Hazardous Substances Data Bank (HSDB) (National Library of Medicine); Internet databases such as CARL UNCOVER; and EPA

databases such as ASTER, OLS, and the ORD BBS. Original references were identified for the values, and additional values were identified. In cases where log $K_{ow}$ values varied over several orders of magnitude or measured values could not be identified, detailed on-line searches were conducted using TOXLIT, Chemical Abstracts, and DIALOG. Values identified from all of these sources and the method used to obtain each log $K_{ow}$ value were compiled for each chemical. A few chemicals lacked experimentally measured log $K_{ow}$s, and no log $K_{ow}$ data were available from any source for butachlor, DCPA/Dacthal, and Ethion/Bladen.

The determination of $K_{ow}$ values was based on experimental measurements taken primarily by the slow-stir, generator-column, and shake-flask methodologies. The SPARC Properties Calculator model was also used to generate $K_{ow}$ values, when appropriate, for comparison with the measured values. Values that appeared to be considerably different from the rest were considered to be outliers and were not used in the calculation.

For each chemical, the available value based on one of these methods was given preference. If more than one such value was available, the log $K_{ow}$ value was calculated as the arithmetic mean of those values (USEPA, 1994). Recommended log $K_{ow}$s were finalized by ORD-Athens based on recommended criteria, and the justification for selection of each value was included in the report (Karickhoff and Long, April 10, 1995, report).

**Selection of Chronic Toxicity Values.** A hierarchy of sources for chronic toxicity values to develop the SQALs was prepared. The following sources were identified and ranked from most to least confidence in the chronic values to be used:

1. Sediment quality criteria (SQCs).
2. Final chronic values from the Great Lakes Initiative (USEPA, 1995c).
3. Final chronic values from the National Ambient Water Quality Criteria documents.
4. Final chronic values from freshwater criteria documents.
5. Final chronic values developed from data in EPA's Aquatic Toxicity Information Retrieval database (AQUIRE) and other sources.

6a. Secondary chronic values developed from data in AQUIRE and other sources.

6b. Secondary chronic values from Suter and Mabrey (1994)

EPA SQCs were available for five chemicals: acenapthene, dieldrin, endrin, fluoranthene, and phenanthrene. There were no final chronic values (FCVs) obtained by the aquatic life criteria methodology (referred to as "Tier I") described in USEPA (1995c) available for the remaining chemicals in the NSI. Two SQALs were based on the FCVs from National Ambient Water Quality Criteria documents, for gamma-BHC/Lindane and toxaphene. No FCVs were available from criteria documents.

Thirteen SQALs were based on work conducted by Oak Ridge National Laboratories (Suter and Mabrey, 1994) using the USEPA (1995c) methodology for obtaining secondary chronic values ("Tier II"). This methodology was developed to obtain whole-effluent toxicity screening values based on all available data, but the SCVs could also be calculated with fewer toxicity data than are required for the criteria methodology. The SCVs are generally more conservative than those which can be produced by the FCV methodology, reflecting greater uncertainty in the absence of additional toxicity data. The minimum requirement for deriving an SCV is toxicity data from a single taxonomic family (Daphnidae), provided the data are acceptable. Only those values from Suter and Mabrey (1994) that included at least one daphnid test result in the calculation of the SCV were included for the NSI. SCVs from Suter and Mabrey (1994) were used to develop SQALs for the following chemicals:

benzene
chlorobenzene
delta-BHC
dibenzofuran
diethyl phthalate
di-n-butyl phthalate
ethylbenzene
napthalene
1,1,2,2-tetrachloroethane
tetrachloroethene
toluene
1,1,1-trichloroethane
trichloroethene

A preliminary search of data records in EPA's AQUIRE database indicated that the following chemicals might have sufficient toxicity data for the development of SCVs:

| | |
|---|---|
| biphenyl | fluorene |
| 4-bromophenyl phenyl ether | hexachlorethane |
| butyl benzyl phthalate | malathion |
| diazinon | methoxychlor |
| 1,2-dichlorobenzene | pentachlorobenzene |
| 1,3-dichlorobenzene | tetrachloromethane |
| 1,4-dichlorobenzene | tribromomethane |
| endosulfan mixed isomers | 1,2,4-trichlorobenzene |
| alpha-endosulfan | trichloromethane |
| beta-endosulfan | m-xylene |

Insufficient toxicity test data were found in AQUIRE for acenapthylene, endosulfan sulfate, heptachlor epoxide, and trichlorofluoromethane. In addition, review of AQUIRE data records indicated that no daphnid acute toxicity tests had been conducted for hexachlorobutadiene. These chemicals were dropped from further development of SQALs.

*Acid-Volatile Sulfide Concentration*

The use of the total concentration of a trace metal in sediment as a measure of its toxicity and its ability to bioaccumulate is not supported by field and laboratory studies because different sediments exhibit different degrees of bioavailability for the same total quantity of metal (Di Toro et al., 1990; Luoma, 1983). These differences have been reconciled by relating organism toxic response (mortality) to the metal concentration in the sediment pore water (Adams et al., 1985; Di Toro et al., 1990). Metals form insoluble complexes with the reactive pool of solid-phase sulfides in sediments (iron and manganese sulfides), restricting their bioavailability. The metals that can bind to these sulfides have sulfide solubility parameters smaller than those of iron sulfide and include nickel, zinc, cadmium, lead, copper, and mercury. Acid-volatile sulfide (AVS) is one of the major chemical components that control the activities and availability of metals in the pore waters of anoxic sediments (Meyer et al., 1994).

AVS is operationally defined as the sulfide liberated from a sediment sample to which hydrochloric acid has been added at room temperature under anoxic conditions (Meyer et al., 1994). The metals concentrations that are extracted during the same analysis are termed the simultaneously extracted metals (SEM). SEM is operationally defined as those metals which form less soluble sulfides than do iron or manganese (i.e., the solubility products of these sulfides are lower than that of iron or manganese sulfide) and that are at least partially soluble under the same test conditions in which the AVS content of the sediment is determined (Allen et al., 1993; Di Toro et al., 1992; Meyer et al., 1994).

Laboratory studies using spiked sediments and field-collected metal-contaminated sediments demonstrated that when the molar ratio of SEM to AVS [SEM]/[AVS] was less than 1 (excess AVS remained), no acute toxicity (mortality greater than 50 percent) was observed in any sediment for any benthic test organism. When [SEM]/[AVS] was greater than 1 (excess metal remained), the mortality of sensitive species (e.g., amphipods) increased in the range of 1.5 to 2.5 µmol of SEM per µmol AVS (Casas and Crecelius, 1994; Di Toro et al., 1992).

Experimental studies indicate that the lower limit of applicability for AVS is approximately 1 ㎜ol AVS/g sediment and possibly lower; other sorption phases, such as organic carbon, probably become important for sediments with smaller AVS concentrations and for metals with large partition coefficients and large chronic water quality criteria (Di Toro et al., 1990). In addition, studies indicate that copper, as well as mercury, might be associated with another phase in sediments, such as organic carbon, and AVS alone might not be the appropriate partitioning phase for predicting its toxicity. Pore-water concentrations of metals should also be evaluated (Allen et al., 1993; Ankley et al., 1993; Casas and Crecelius, 1994). However, the AVS approach can be used to predict when a sediment contaminated with metals is not acutely toxic (Ankley et al., 1993; Di Toro et al., 1992).

There are several important factors to consider in interpreting the [SEM]-[AVS] difference. First, all toxic SEMs present in amounts that contribute significantly to the [SEM] sum should be measured. However, because mercury presents special problems, it is not included in the current SEM analysis. Second, if the AVS content of sediment is low, as in fully oxidized sediments, the metal-binding capacity of the sediment decreases and the method will not work

(Adams et al., 1992; Zhuang et al., 1994). Most benthic macroorganisms, including those used in toxicity tests, survive in sediments that have a thin oxidized surface layer and then an anoxic layer. The anoxic layer can have significant AVS concentrations that would reduce the metal activity to which these organisms are exposed (Di Toro et al., 1992). Third, AVS varies spatially in sediment—vertically with depth and horizontally where patches of an appropriate carbon source occur under low oxygen conditions for the sulfate-reducing bacteria. Lastly, AVS can vary when sediments are oxgenated during physical disturbance and seasonally as changes in the productivity of the aquatic ecosystem alter the oxidation state of sediment and oxidize metal sulfides; therefore, the toxicity of the metals present in the sediment also changes over time (Howard and Evans, 1993).

Selection of an [SEM] - [AVS] difference sufficiently high to place a sediment in the Tier 1 classification requires careful consideration because the relationship between organism response and the [SEM] - [AVS] difference of sediment depends on the amount and kinds of other binding phases present. Using freshwater and saltwater sediment amphipod toxicity data, researchers at EPA's Environmental Research Laboratory in Narragansett, Rhode Island, plotted [SEM] - [AVS] versus the percentage of sediments with a higher [SEM] - [AVS] value that were toxic. For this analysis, the researchers defined toxicity as greater than 24 percent mortality. Analysis of these data reveals that between 80 percent and 90 percent of the sediments were toxic at [SEM] - [AVS] = 5. The running average mortality at this level was between 44 percent and 62 percent (Hansen, 1995). EPA's Office of Science and Technology selected [SEM] - [AVS] = 5 as the demarcation line between the higher (Tier 1) and intermediate (Tier 2) probability categories.

## *Biological Effects Correlation Approaches*

Biological effects correlation approaches are based on the evaluation of paired field and laboratory data to relate incidence of adverse biological effects to the dry-weight sediment concentration of a specific chemical at a particular site. Researchers use these data sets to identify level-of-concern chemical concentrations based on the probability of observing adverse effects. Exceedance of the identified level-of-concern concentrations is associated with a likelihood of adverse organism response, but it does not demonstrate that a particular chemical is solely responsible. Consequently, correlative approaches do not indicate direct cause-and-effect relationships. In fact, a given site typically contains a mixture of chemicals that contribute to observed adverse effects to some degree. These and other potentially mitigating factors tend to make screening values based on correlative approaches lower than screening values based on effects caused by a single chemical. However, correlative procedures differ from one another by design and, subsequently, in how they relate to sediment toxicity. For example, ERMs are levels usually associated with adverse effecs, whereas AETs are levels intended to always be associated with adverse effects. Thus, when in error, ERMs minimize false negatives relative to AETs and AETs minimize false positives relative to ERMs (Ingersoll et al., 1996).

### *Effects Range-Medians and Effects Range-Lows*

The effects range approach for deriving sediment quality guidelines involves matching dry-weight sediment contaminant concentrations with associated biological effects data. Long and Morgan (1990) originally developed informal guidelines using this approach for evaluation of NOAA's National Status and Trends (NS&T) data. Data from equilibrium partitioning modeling, laboratory, and field studies conducted throughout North America were used to determine the concentration ranges that are rarely, sometimes, and usually associated with toxicity for marine and estuarine sediments (Long et al., 1995). Effects range-low (ERL) and effects range-median (ERM) values were derived by Long et al. (1995) for 28 chemicals or classes of chemicals: 9 trace metals, total PCBs, 13 individual polynuclear aromatic hydrocarbons (PAHs), 3 classes of PAHs (total low molecular weight, total high molecular weight, and total PAH), and 2 pesticides (p,p'-DDE and total DDT). For each chemical, sediment concentration data with incidence of observed adverse biological effects were identified and ordered. The authors identified the lower 10th-percentile concentration as the ERL and the 50th-percentile concentration as the ERM. In terms of potential biological effects, sediment contaminant concentrations below the ERL are defined as in the "minimal-effects range," values between the ERL and ERM are in the "possible-effects range," and values above the ERM are in the "probable-effects range." Data entered into this biological effects database for sediments (BEDS) were expressed on a dry-weight basis.

The accuracy of these guidelines was evaluated using the data in the database not associated with adverse effects and noting whether the incidence of effects was less than 25 percent in the minimal-effects range, increased consistently with increasing chemical concentrations, and was greater than 75 percent in the probable-effects range. Long et al.

(1995) reported that these sediment quality guidelines were most accurate for copper, lead, silver, and all classes of PAHs and most of the individual PAHs; however, accuracy was low for nickel, chromium, mercury, total PCBs, and DDE and DDT. The guidelines generally agreed within factors of 2 to 3 with other guidelines, including the freshwater effects-based criteria from Ontario. The authors attributed variability in the concentrations associated with effects to differences in sensitivities of different taxa and physical factors that affect bioavailability, but they argued that because of the synergistic effects of multiple toxicants, the inclusion of data from many field studies in which mixtures of chemicals were present in sediments could make the guidelines more protective than guidelines based on a single chemical. The authors also emphasized that ERLs and ERMs were intended to be used as informal screening tools only.

Although the ERL and ERM guidelines were not based upon deterministic or cause-effects studies, their accuracy in correctly predicting nontoxicity and toxicity has been determined empirically among field-collected samples (Long et al., in press). Analyses were performed with matching laboratory bioassay data and chemical data from 989 samples collected in regions of the Atlantic, Pacific, and Gulf coasts. Data were gathered from results of amphipod survival tests (*Ampelisca abdita* and *Rhepoxynius abronius*) for all 989 samples. Data from a battery of sensitive bioassays (fertilization success of urchin gametes, embryological development of mollusc embryos, and microbial bioluminescence) were gathered for 358 of these samples. The percentages of samples indicating non-toxicity (not significantly different from controls, $p > 0.05$), significant toxicity ($p < 0.05$), and high toxicity ($p < 0.05$ and mean response >20 percent difference from controls) were determined for the results of the amphipod tests alone and for the results of any one of the tests performed.

Results of the analyses (summarized in Table B-1) suggest that highly toxic responses occurred in 12 percent of the samples in the amphipod tests and 28 percent of the samples in any one of the tests performed when all chemical concentrations were less than their respective ERL values. These samples were analogous to those classified as Tier 3 in this report (i.e., all chemical concentrations less than the screening values). When one or more chemicals exceeded ERL concentrations, but all concentrations were lower than the ERM concentrations (analogous to Tier 2), the percentages of samples indicating high toxicity were 19 percent in the amphipod tests and 64 percent in any one of the tests performed. The incidence of high toxicity in the amphipod tests increased from 10 percent when only one ERL value was exceeded to 58 percent when 20-24 ERLs were exceeded. The incidence of toxicity in any one of the tests increased from 29 percent when only one ERL was exceeded to 91 percent when 20-24 ERLs were exceeded. In samples analogous to those classified as Tier 1 (one or more ERMs exceeded), the incidence of high toxicity was 42 percent in amphipod tests and 80 percent in any one of the battery of tests performed. If both the significant and highly toxic results were combined in the Tier 1 samples, the percentage of samples indicating toxicity increases to 55 percent in amphipod tests and 87 percent in any one of the tests. As with the ERLs, the incidence of toxicity increased with increasing number of chemicals that exceeded the ERMs.

### *Probable Effects Levels and Threshold Effects Levels*

A method slightly different from that used by Long et al. (1995) to develop ERMs and ERLs was used by the Florida Department of Environmental Protection (FDEP, 1994) to develop similar correlative, effects-based guidelines for Florida's coastal waters. Modifications to the Long et al. (1995) approach increased the relevance of the resultant guidelines to Florida's coastal sediments by making information in the database more consistent and by expanding the information

**Table B-1. Incidence of Toxicity in Amphipod Survival Tests Alone and Any One of 2-4 Tests Performed in Samples Analogous to Those Classified as Tier 1, 2, or 3 (from Long et al., in press)**

| Chemical Concentrations | Analogous Tier | Amphipod Tests Alone | | | Any Test Performed | | |
|---|---|---|---|---|---|---|---|
| | | % Not Toxic | % Signif. Toxic | % Highly Toxic | % Not Toxic | % Signif. Toxic | % Highly Toxic |
| all < ERLs | Tier 3 | 64 | 23 | 12 | 67 | 5 | 28 |
| > 1 or more ERLs | Tier 2 | 59 | 22 | 19 | 20 | 15 | 64 |
| > 1 or more ERMs | Tier 1 | 45 | 13 | 42 | 13 | 7 | 80 |

used to derive sediment quality assessment guidelines with additional data from other locations in the United States and Canada, particularly Florida and the southeastern and Gulf of Mexico regions (FDEP, 1994). Three effects ranges were developed with a method that used both the chemical concentrations associated with biological effects (the "effects" data) and those associated with no observed effects (the "no-effects" data). In this method, the threshold effects level (TEL) is the geometric mean of the lower 15th-percentile concentration of the effects data (the ERL) and the 50th-percentile concentration of the no-effects data. The probable-effects level (PEL) is the geometric mean of the 50th-percentile concentration of the effects data (the ERM) and the 85th-percentile concentration of the no-effects data. Essentially, the PEL and TEL reflect the ERM and ERL values adjusted upward or downward depending on the degree of overlap between the distributions of "effects" and "no effects" data. TELs and PELs have been developed for 33 chemicals: 9 trace metals, total PCBs, 13 individual polynuclear aromatic hydrocarbons (PAHs), 3 classes of PAHs (total low molecular weight, total high molecular weight, and total PAH), 6 pesticides (chlordane, dieldrin, p,p' -DDD, p,p' -DDE, p,p' -DDT), and total DDT (FDEP, 1994).

As was the case with the Long et al. (1995) approach, in the FDEP (1994) approach the lower of the two guidelines for each chemical (i.e., the TEL) was assumed to represent the concentration below which toxic effects rarely occurred. In the range of concentrations between the TEL and PEL, effects occasionally occurred. Toxic effects usually or frequently occurred at concentrations above the upper guideline value (i.e., the PEL). TEL and PEL values were developed on a sediment dry-weight basis.

Although the extensive database and evaluation of effects data make this approach applicable to many areas of the country, the available data still have limitations. For example, FDEP (1994) noted that there is a potential for underprotection or overprotection of aquatic resources if the bioavailability of sediment-associated contaminants and other factors affecting toxicity are not included. Most of the TELs and PELs were within a factor of 2 to 3 of other sediment quality guideline values. Most were deemed reliable for evaluating sediment quality in Florida's coastal waters, with less confidence in the values for mercury, nickel, total PCBs, chlordane, lindane, and total DDT. An evaluation of independent sets of field data from Florida, the Gulf of Mexico, California, and New York showed that TELs and PELs correctly predict the toxicity of sediment in 86 percent and 85 percent of the samples, respectively.

As with ERLs and ERMs, the accuracy of TEL and PEL guidelines to correctly predict nontoxicity and toxicity has been determined empirically among field-collected samples (Long et al., in press). Analyses were performed with matching laboratory bioassay data and chemical data from 989 samples collected in regions of the Atlantic, Pacific, and Gulf coasts. Data were gathered from results of amphipod survival tests (*Ampelisca abdita and Rhepoxynius abronius*) for all 989 samples. Data from a battery of sensitive bioassays (fertilization success of urchin gametes, embryological development of mollusc embryos, and microbial bioluminescence) were gathered for 358 of these samples. The percentages of samples indicating nontoxicity (not significantly different from controls, $p > 0.05$), significant toxicity ($p < 0.05$), and high toxicity ($p < 0.05$ and mean response >20 percent difference from controls) were determined for the results of the amphipod tests alone and for the results of any one of the tests performed.

Results of the analyses (summarized in Table B-2) suggest that highly toxic responses occurred in 10 percent of the samples in the amphipod tests and 5 percent of the samples in any one of the tests performed when all chemical concentrations were less than their respective TELvalues. These samples were analogous to those classified as Tier 3 in this report (i.e., all chemical concentrations less than the screening values). When one or more chemicals exceeded TEL concentrations, but all concentrations were lower than the PEL concentrations (analogous to Tier 2), the percentages of samples indicating high toxicity were 17 percent in the ampipod tests alone and 59 percent in any one of the tests performed. The incidence of high toxicity in the amphipod tests increased from 13 percent when only one TEL value was exceeded to 52 percent when 20-27 TELs were exceeded. The incidence of toxicity in any one of the tests increased from 31 percent when 1-5 TELs were exceeded to 63 percent when 20-27 TELs were exceeded. In samples analogous to those classified as Tier 1 (one or more PELs exceeded), the incidence of high toxicity was 38 percent in amphipod tests and 78 percent in any one of the battery of tests performed. If both the significant and highly toxic results were combined in the Tier 1 samples, the percentage of samples indicating toxicity increases to 51 percent in amphipod tests and 86 percent in any one of the tests. As with the TELs, the incidence of toxicity increased with increasing number of chemicals that exceeded the PELs.

*Apparent Effects Thresholds*

**Table B-2. Incidence of Toxicity in Amphipod Survival Tests Alone and Any One of 2-4 Tests Performed in Samples Analogous to Those Classified as Tier 1, 2, or 3 (from Long et al., in press)**

| Chemical Concentrations | Analogous Tier | Amphipod Tests Alone | | | Any Test Performed | | |
|---|---|---|---|---|---|---|---|
| | | % Not Toxic | % Signif. Toxic | % Highly Toxic | % Not Toxic | % Signif. Toxic | % Highly Toxic |
| all < TELs | Tier 3 | 61 | 29 | 10 | 90 | 5 | 5 |
| > 1 or more TELs | Tier 2 | 62 | 21 | 17 | 22 | 19 | 59 |
| > 1 or more PELs | Tier 1 | 49 | 13 | 38 | 14 | 8 | 78 |

The AET approach is another empirical data evaluation approach to defining concentrations in sediment associated with adverse effects. Barrick et al. (1988) reported that AETs can be developed for any measured chemical (organic or inorganic) with a wide concentration range in the field. The AET concept applies to matched field data for sediment chemistry and any observable biological effects (e.g., bioassay responses, infaunal abundances at various taxonomic levels, bioaccumulation). By using these different biological indicators, application of the resulting sediment quality values enables a wide range of biological effects to be addressed in the management of contaminated sediments. Using sediment samples from Puget Sound in Washington State, AET values have been developed for 52 chemicals: 10 trace metals, 15 individual polynuclear aromatic hydrocarbons (PAHs), 3 pesticides (p,p'-DDD, p,p'-DDE, p,p'-DDT), 6 halogenated organics, and 18 other compounds.

The focus of the AET approach is to identify concentrations of contaminants that are associated exclusively with sediments exhibiting statistically significant biological effects relative to reference sediments. AET values were based on measured chemical concentrations per dry weight of sediment. AETs for each chemical and biological indicator were developed using the following steps (Barrick et al., 1988).

1. Collected “matched” chemical and biological effects data—Conducted chemical and biological effects testing on subsamples of the same field sample.

2. Identified “impacted” and “nonimpacted” stations—Statistically tested the significance of adverse biological effects relative to suitable reference conditions for each sediment sample and biological indicator.

3. Identified the AET using only “nonimpacted” stations—For each chemical, the AET was identified for a given biological indicator as the highest *detected* concentration among sediment samples that did not exhibit statistically significant effects.

4. Verified that statistically significant biological effects were observed at a chemical concentration higher than the AET; otherwise, the AET was only a preliminary minimum estimate.

5. Repeated steps 1-4 for each biological indicator.

For a given data set, the AET value for a chemical is the sediment concentration above which a particular adverse biological effect for individual biological indicators (amphipod bioassay, oyster larvae bioassay, Microtox bioassay, and benthic infaunal abundance) is always significantly different statistically relative to appropriate reference conditions. Two thresholds were recognized in the evaluations conducted in this report, when possible, based on the different indicators. EPA defined the AET-low as the lowest AET among applicable biological indicators, and the AET-high as the highest AET among applicable biological indicators. The use of the high/low AET values is not a recommendation of the authors of the approach; rather it was developed for the NSI evaluation. The two thresholds were used in this evaluation to give a range of effects values (as with the ERL/ERMs and TEL/PELS). AET values based on Microtox bioassays were not used for the NSI evaluation.

### *Sediment Toxicity Approaches*

Sediment toxicity tests provide important information on the effects of multiple chemical exposures to assist in the evaluation of sediment quality. Methods for testing the acute and chronic toxicity of sediment samples to benthic freshwater and marine organisms have been developed (see reviews in API, 1994; Burton et al., 1992; Lamberson et al., 1992; USEPA, 1994b, c) and used primarily for dredged material evaluation (USEPA and USACOE, 1994). The NSI data contain acute sediment toxicity results from tests in which organisms were exposed to field-collected sediments and mortality was recorded. Results of whole sediment and elutriate toxicity tests were used in the evaluation of the NSI.

Variations in observed toxicity from tests of the same sediment sample may be attributed to the relative sensitivities of the species used in the tests; disruption of geochemistry and kinetic activity of bedded sediment contaminants during sampling, handling, and bioturbation; and laboratory-related confounding factors (Lamberson et al., 1992). Recent studies indicate that aqueous representations of whole sediment (e.g., elutriate) do not accurately predict the bioavailability of some contaminants compared to whole-sediment exposures (Harkey et al., 1994). Acute sediment toxicity tests have been widely accepted by the scientific and regulatory communities and the results can be readily interpreted, although more work is needed on chronic testing (Thomas et al., 1992). Appendix G presents the methodology for evaluating sediment toxicity tests as applied in the NSI data evaluation.

## Human Health Assessments

In the evaluation of NSI data, two primary evaluation parameters were used to assess potential human health impacts from sediment contamination: (1) sediment chemistry theoretical bioaccumulation potential and (2) tissue levels of contaminants in demersal, nonmigratory species.

### *Theoretical Bioaccumulation Potential*

The theoretical bioaccumulation potential (TBP) is an estimate of the equilibrium concentration of a contaminant in tissues if the sediment in question were the only source of contamination to the organism (USEPA and USACOE, 1994). The TBP calculation is used as a screening mechanism to represent the magnitude of bioaccumulation likely to be associated with nonpolar organic contaminants in the sediment. At present, the TBP calculation can be performed only for nonpolar organic chemicals; however, methods for TBP calculations for metals and polar organic chemicals are under development (USEPA and USACOE, 1994).

The environmental distribution of nonpolar organic chemicals is controlled largely by their solubility in various media. Therefore, in sediments they tend to occur primarily in association with organic matter (Karickhoff, 1981) and in organisms they are found primarily in the body fats or lipids (Bierman, 1990; Geyer et al., 1982; Konemann and van Leeuwen, 1980; Mackay, 1982). Bioaccumulation of nonpolar organic compounds from sediment can be estimated from the organic carbon content of the sediment, the lipid content of the organism, and the relative affinities of the chemical for sediment organic carbon and animal lipid content (USEPA and USACOE, 1994). It is possible to relate the concentration of a chemical in one phase of a two-phase system to the concentration in the second phase when the system is in equilibrium. The TBP calculation focuses on the equilibrium distribution of a chemical between the sediment and the organism. By normalizing nonpolar organic chemical concentration data for lipid in organisms, and for organic carbon in sediment, it is possible to estimate the preference of a chemical for one phase or the other (USEPA and USACOE, 1994).

The TBP can be calculated relative to the biota-sediment accumulation factor (BSAF), as in the following equation (USEPA and USACOE, 1994):

$$TBP = BSAF(C_s / f_{oc})f_l$$

where TBP is expressed on a whole-body basis in the same units of concentration as $C_s$ and

TBP = theoretical bioaccumulation potential (ppm);

| | | |
|---|---|---|
| $C_s$ | = | concentration of nonpolar organic chemical in sediment (ppm); |
| BSAF | = | biota-sediment accumulation factor (ratio of the concentration of a chemical in tissue, normalized to lipid, to the concentration of the chemical in surface sediment, normalized to organic carbon (in kg sediment organic carbon/kg lipid)); |
| $f_{oc}$ | = | total organic carbon (TOC) content of sediment expressed as a decimal fraction (i.e., 1 percent = 0.01); and |
| $f_l$ | = | organism lipid content expressed as a decimal fraction (e. g., 3 percent = 0.03) of fillet or whole-body dry weight. |

BSAF values used in the TBP evaluation are discussed in Appendix C. If TOC measurements were not available at a site, $f_{oc}$ was assumed to be 0.01 (1 percent).

For the evaluation of NSI data, EPA selected a 3 percent lipid content in fish fillets for the TBP calculation for assessing human health effects from the consumption of contaminated fish. Lipid normalization is now part of the EPA guidance on bioaccumulation, and the current national methodology uses a 3 percent value for human health assessments. The *Great Lakes Water Quality Initiative Technical Support Document for the Procedure to Determine Bioaccumulation Factors* (USEPA, 1995b) uses a 3.10 percent lipid value for trophic level 4 fish and 1.82 percent for trophic level 3 fish in its human health assessments.

As part of the NSI TBP evaluation, EPA also evaluated percent lipid measurements included in the STORET database, the *National Study of Chemical Residues in Fish* (NSCRF; USEPA, 1992b), and other published sources, and compared those values to the value selected for the NSI evaluation (Appendix C). The mean fillet percent lipid content for various groups of fish species in the STORET database ranged from 0.753 to 4.49 percent; in the NSCRF, mean fillet values ranged from 1.6 to 4.9 percent. The mean whole-body percent lipid content for various groups of fish species in the STORET database ranged from 3.757 to 6.33 percent; in the NSCRF, mean whole-body values ranged from 4.6 to 8.8 percent.

In the NSI data evaluation approach, TBP values were compared to U.S. Food and Drug Administration tolerance/action/guidance levels and EPA risk levels. These parameters are discussed below.

### *FDA Tolerance/Action/Guidance Levels*

The U.S. Food and Drug Administration (FDA) is responsible for the safety of the Nation's commercial food supply, including fish and shellfish, for human consumption. Under the authority of the Federal Food, Drug and Cosmetic Act (FFDCA), FDA ensures that regulated products are safe for use by consumers. The FFDCA authorizes FDA to conduct assessments of the safety of ingredients in foods. The key element of the FFDCA, and the source of FDA's main tools for enforcement, is the prohibition of the "adulteration" of foods. FDA can prescribe the level of contaminant that will render a food adulterated and, therefore, can initiate enforcement action based on scientific data. The establishment of guidance and action levels (informal judgments about the level of a food contaminant to which consumers can be safely exposed) or tolerances (regulations having the force of law) is the regulatory procedure employed by FDA to control environmental contaminants in the commercial food supply.

During the 1970s, the available detection limits were considered to demonstrate elevated contamination and were used as action levels. Since that time, FDA has focused on using risk-based standards. These standards have been derived by individually considering each chemical and the species of fish it is likely to contaminate. FDA also considered (1) the amount of potentially contaminated fish eaten and (2) the average concentrations of contaminants consumed. FDA has established action levels in fish for 10 pesticides and methylmercury, tolerance levels for polychlorinated biphenyls (PCBs), and guidance for 5 metals.

### *EPA Risk Levels*

Potential impacts on humans are evaluated by estimating potential carcinogenic risks and noncarcinogenic hazards associated with the consumption of chemically contaminated fish tissue. In this assessment it was assumed that the only source of contamination to fish is contaminated sediment. The procedures for estimating human health risks due to the consumption of chemically contaminated fish tissue are based on *Risk Assessment Guidance for Superfund*

(USEPA, 1989) and *Guidance for Assessing Chemical Contamination Data for Use in Fish Advisories, Volume II: Development of Risk-Based Intake Limits* (USEPA, 1994a).

EPA human health risk assessment methods were used in this assessment to determine the levels of contamination in fish that might result in a $10^{-5}$ cancer risk (1 in 100,000 extra chance of cancer over a lifetime) or a noncancer hazard in humans. A $10^{-5}$ risk level exceeds the lower bound (i.e., $10^{-6}$) but is lower than the upper bound (i.e., $10^{-4}$) of the risk range accepted by EPA (USEPA, 1990).

Human health cancer risks and noncancer hazards are based on the calculation of the chronic daily intake (CDI) of contaminants of concern:

$$CDI = \frac{(EPC)(IR)(EF)(ED)}{(BW)(AT)}$$

where:

| | | |
|---|---|---|
| CDI | = | chronic daily intake (mg/kg/day); |
| EPC | = | exposure point concentration (contaminant concentration in fish); |
| IR | = | ingestion rate (6.5 g/day); |
| EF | = | exposure frequency (365 days/year); |
| ED | = | exposure duration (70 years); |
| BW | = | body weight (70 kg); and |
| AT | = | averaging time (70 years x 365 days/year). |

These are the same parameter values used by EPA to develop human health water quality criteria. Carcinogenic risks are then quantified using the equation below:

$$\text{Cancer risk}_i = CDI_i \times SF_i$$

where:

| | | |
|---|---|---|
| $\text{Cancer risk}_i$ | = | the potential carcinogenic risk associated with exposure to chemical *i* (unitless); |
| $CDI_i$ | = | chronic daily intake for chemical *i* (mg/kg/day); and |
| $SF_i$ | = | slope factor for chemical *i* $(mg/kg/day)^{-1}$. |

The hazard quotient, which is used to quantify the potential for an adverse noncarcinogenic effect to occur, is calculated using the following equation:

$$HQ_i = \frac{CDI_I}{RfD_I}$$

where:

| | | |
|---|---|---|
| $HQ_i$ | = | hazard quotient for chemical *i* (unitless); |
| $CDI_i$ | = | chronic daily intake for chemical *i* (mg/kg/day); and |
| $RfD_i$ | = | reference dose for chemical *i* (mg/kg/day). |

If the hazard quotient exceeds unity (i.e., 1), an adverse health effect might occur. The higher the hazard quotient, the more likely that an adverse noncarcinogenic effect will occur as a result of exposure to the chemical. If the estimated hazard quotient is less than unity, noncarcinogenic effects are unlikely to occur.

Using these formulas, the fish tissue concentration (EPC) of a contaminant that equates to a cancer risk of $10^{-5}$ or a hazard quotient that exceeds unity can be back-calculated.

*Cancer risk:*

$$EPC = \frac{(10^{-5})(BW)(AT)(C_1)}{(IR)(EF)(ED)(SF_i)}$$

*Noncancer hazard:*

$$EPC = \frac{(BW)(AT)(RfD_i)(C_1)}{(IR)(EF)(ED)}$$

where:

$C_1$ = conversion factor ($10^3$ g/kg).

### *Tissue Levels of Contaminants*

In addition to sediment chemistry TBP values, measured levels of contaminants in the tissues of resident aquatic species were used to assess potential human health risk. As was the case with the evaluation of TBP values, the NSI evaluation approach compared contaminant tissue levels to FDA tolerance/action/guidance levels and EPA risk levels. Each of these parameters was discussed in the previous section. In such a comparison it is assumed that contaminant concentrations in tissue result from bioaccumulation of contaminants in the sediment.

## Wildlife Assessments

In addition to the evaluation parameters described above for the assessment of potential aquatic life and human health impacts, EPA also conducted a separate analysis of potential wildlife impacts resulting from exposure to sediment contaminants.

Wildlife criteria based on fish tissue concentrations were derived using methods similar to those employed for deriving EPA wildlife criteria, as presented in the *Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife* (USEPA, 1995a). EPA has developed Great Lakes Water Quality Wildlife Criteria for four chemicals: DDT, mercury, 2,3,7,8-TCDD, and PCBs. A Great Lakes Water Quality Wildlife Criterion (GLWC) is the concentration in the water of a substance that, if not exceeded, protects avian and mammalian wildlife populations from adverse effects resulting from the ingestion of surface waters and aquatic prey (USEPA, 1995a). Wildlife values are calculated using the equation:

$$WV = \frac{(NOAEL \times SSF) \times Wt_A}{W_A + (F_A \times BAF)}$$

where:

WV = wildlife value (mg/L);
NOAEL = no-observed-adverse-effect level, as derived from mammalian or avian studies (mg/kg-d);
$Wt_A$ = average weight for the representative species identified for protection (kg);
$W_A$ = average daily volume of water consumed by the representative species identified for protection (L/d);
SSF = species sensitivity factor, an extrapolation factor to account for the difference in toxicity between species;
$F_A$ = average daily amount of food consumed by the representative species identified for protection (kg/d); and
BAF = bioaccumulation factor (L/kg), the ratio of the concentration of a chemical in tissue, normalized to lipid, to the concentration in ambient water. Chosen using guidelines for wildlife presented in appendix B to part 132, Methodology for Development of Bioaccumulation Factors (*Federal Register*, Vol. 58, No. 72, April 16, 1993).

In the development of the four GLWCs, wildlife values for five representative Great Lakes basin wildlife species (bald eagle, herring gull, belted kingfisher, mink, and river otter) were calculated, and the geometric mean of these values within each taxonomic class (birds and mammals) was determined. The GLWC is the lower of two class-species means (USEPA, 1995a).

The wildlife values are considered to be generally protective of wildlife species. However, it should be noted that the approach is not based on the most sensitive wildlife species, but rather a typical class of either avian or mammalian piscivores. Despite this limitation, this approach is still considered appropriate and conservative because of the many conservative assumptions used to derive these wildlife values (e.g., species sensitivity factors, assumption that animals consume only contaminated fish).

Proposed EPA wildlife criteria are based on surface water contaminant levels protective of potential wildlife exposure. Thus, the proposed EPA wildlife criteria cannot be compared directly to the NSI fish tissue concentrations (either the calculated TBPs or fish tissue monitoring data). Therefore, it was necessary to develop an approach for estimating wildlife criteria for fish tissue based on the same toxicity and exposure parameter assumptions that were used to derive the surface water wildlife criteria. First, wildlife values (i.e., fish tissue concentrations protective of wildlife) were derived for the most sensitive mammalian species (i.e., otter and mink) and avian species (i.e., kingfisher, herring gull, and eagle)—the same species used to derive the proposed EPA wildlife criteria. The equation used to estimate wildlife values for fish tissue is presented below. (Exposure assumptions used for each species are presented in USEPA, 1995a.)

$$WV_{fish} = \frac{[NOAEL \times SSF] \times Wt_A}{F_A}$$

where:

| | | |
|---|---|---|
| $WV_{fish}$ | = | wildlife value for fish tissue (mg/kg); |
| NOAEL | = | no-observed-adverse-effect level (mg/kg-day); |
| SSF | = | species sensitivity factor |
| $Wt_A$ | = | average weight of animal in kilograms (kg); and |
| $F_A$ | = | average daily amount of food consumed (kg/day). |

Secondly, the geometric mean of the wldlife values was calculated for the mammal group, as well as for the avian group. Finally, the lower of the two geometric mean values was considered the wildlife criterion for fish tissue for a given chemical.

It should be noted that direct ingestion of surface water was included when developing proposed EPA wildlife criteria for surface water. This exposure route, however, was not considered when evaluating NSI data, even though sediment contamination might result in contamination of surface water available for wildlife consumption. A sensitivity analysis was conducted to evaluate the impact of excluding the surface water ingestion exposure route. Based on this analysis, ingestion of surface water contributes less than 0.0001 percent of the total exposure (i.e., ingestion of fish and water). Therefore, excluding the water ingestion exposure route had no significant impact on the evaluation of NSI data with regard to potential wildlife impacts.

Wildlife criteria derived for DDT, mercury, 2,3,7,8-TCDD, and PCBs based on fish tissue concentration are presented below.

| Chemical | Fish Tissue Criterion (mg/kg) |
|---|---|
| DDT | 3.93E-2 |
| Mercury | 5.73E-2 |
| 2,3,7,8-TCDD | 5.20E-7 |
| PCBs | 1.60E-1 |

The wildlife criteria were compared to measured fish tissue residue data contained in the NSI and to TBPs calculated for DDT, 2,3,7,-TCDD, and PCBs. Mercury is not a nonpolar organic chemical, and thus a TBP for mercury was not calculated. A whole-body lipid value of 10.31 was assumed for the TBP evaluation of potential wildlife impacts, based on the *Great Lakes Water Quality Technical Support Document for the Procedure to Determine Bioaccumulation Factors* (USEPA, 1995b).

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