SECTION 5

SCREENING VALUES FOR TARGET ANALYTES

For the purpose of this guidance document, screening values are defined as concentrations of target analytes in fish or shellfish tissue that are of potential public health concern and that are used as threshold values against which levels of contamination in similar tissue collected from the ambient environment can be compared. Exceedance of these SVs should be taken as an indication that more intensive site-specific monitoring and/or evaluation of human health risk should be conducted.

The EPA-recommended risk-based method for developing SVs (U.S. EPA, 1989d) is described in this section. This method is considered to be appropriate for protecting the health of fish and shellfish consumers for the following reasons (Reinert et al., 1991):

- It gives full priority to protection of public health.
- It provides a direct link between fish consumption rate and risk levels (i.e., between dose and response).
- It generally leads to conservative estimates of increased risk.
- It is designed for protection of consumers of locally caught fish and shellfish, including susceptible populations such as sport and subsistence fishers who are at potentially greater risk than the general adult population because they tend to consume greater quantities of fish and because they frequently fish the same sites repeatedly.

At this time, the EPA Office of Water is recommending use of this method because it is the basis for developing current water quality criteria. A detailed discussion of the flexibility of the EPA risk-based method and the use of EPA's SVs as compared to FDA action levels is provided in Section 1.2. Further discussion of the EPA Office of Water risk-based approach, including a detailed description of the four steps involved in risk assessment (hazard identification, dose-response assessment, exposure assessment, and risk characterization) is provided in the second guidance document in this series, *Volume 2: Risk Assessment and Fish Consumption Limits*.

5.1 GENERAL EQUATIONS FOR CALCULATING SCREENING VALUES

Risk-based SVs are derived from the general model for calculating the effective ingested dose of a chemical m (E_m) (U.S. EPA, 1989d):

$$E_m = (C_m \bullet CR \bullet X_m) / BW \tag{5-1}$$

where

E_m = Effective ingested dose of chemical *m* in the population of concern averaged over a 70-yr lifetime (mg/kg-d)

C_m = Concentration of chemical *m* in the edible portion of the species of interest (mg/kg; ppm)

CR = Mean daily consumption rate of the species of interest by the general population or subpopulation of concern averaged over a 70-yr lifetime (kg/d)

X_m = Relative absorption coefficient, or the ratio of human absorption efficiency to test animal absorption efficiency for chemical *m* (dimensionless)

BW = Mean body weight of the general population or subpopulation of concern (kg).

Using this model, the SV for the chemical *m* (SV_m) is equal to C_m when the appropriate measure of toxicologic potency of the chemical *m* (P_m) is substituted for E_m. Rearrangement of Equation 5-1, with these substitutions, gives

$$SV_m = (P_m \bullet BW) / (CR \bullet X_m) \tag{5-2}$$

where

P_m = Toxicologic potency for chemical *m*; the effective ingested dose of chemical *m* associated with a specified level of health risk as estimated from dose-response studies; **dose-response variable**.

In most instances, relative absorption coefficients (X_m) are assumed to be 1.0 (i.e., human absorption efficiency is assumed to be equal to that of the test animal), so that

$$SV_m = (P_m \bullet BW) / CR . \tag{5-3}$$

However, if X_m is known, Equation 5-2 should be used to calculate SV_m.

Dose-response variables for noncarcinogens and carcinogens are defined in Sections 5.1.1 and 5.1.2, respectively. These variables are based on an assessment of the occurrence of a critical toxic or carcinogenic effect via a specific route of exposure (i.e., ingestion, inhalation, dermal contact). Oral dose-response variables for the recommended target analytes are given in Appendix G. Because of the fundamental differences between the noncarcinogenic and carcinogenic dose-response variables used in the EPA risk-based method, SVs

must be calculated separately for noncarcinogens and potential carcinogens as shown in the following subsections.

5.1.1   Noncarcinogens

The dose-response variable for noncarcinogens is the **reference dose**. The RfD is an estimate of a daily exposure to the human population (including sensitive subpopulations) that is likely to be without appreciable risk of deleterious effects during a lifetime. The RfD is derived by applying uncertainty or modifying factors to a subthreshold dose (i.e., lowest observed adverse effects level [LOAEL] if the no observed adverse effect level [NOAEL] is indeterminate) observed in chronic animal bioassays. These uncertainty or modifying factors range from 1 to 10 for each factor and are used to account for uncertainties in:

- Sensitivity differences among human subpopulations
- Interspecies extrapolation from animal data to humans
- Short-term to lifetime exposure extrapolation from less-than-chronic results on animals to humans when no long-term human data are available
- Deriving an RfD from a LOAEL instead of a NOAEL
- Incomplete or inadequate toxicity or pharmacokinetic databases.

The uncertainty (UF) and modifying (MF) factors are multiplied to obtain a final UF•MF value. This factor is divided into the NOAEL or LOAEL to derive the RfD (Barnes and Dawson, 1988; U.S. EPA, 1989d).

The following equation should be used to calculate SVs for noncarcinogens:

$$SV_n = (RfD \bullet BW)/CR \tag{5-4}$$

where

SV_n = Screening value for a noncarcinogen (mg/kg; ppm)
RfD = Oral reference dose (mg/kg-d)

and BW and CR are defined as in Equation 5-1.

5.1.2   Carcinogens

According to *The Risk Assessment Guidelines of 1986* (U.S. EPA, 1987f), the default model for low-dose extrapolation of carcinogens is a version (GLOBAL 86) of the linearized multistage no-threshold model developed by Crump et al. (1976). This extrapolation procedure provides an upper 95 percent bound risk estimate (referred to as a q1*), which is considered by some to be a conservative estimate of cancer risk. Other extrapolation procedures may be used when justified by the data.

Screening values for carcinogens are derived from: (1) a carcinogenicity potency factor or **cancer slope factor**, which is generally an upper bound risk estimate; and (2) a **risk level** (RL), an assigned level of maximum acceptable individual

lifetime risk (e.g., RL = 10^{-5} for a level of risk not to exceed one excess case of cancer per 100,000 individuals exposed over a 70-yr lifetime) (U.S. EPA, 1997b). The following equation should be used to calculate SVs for carcinogens:

$$SV_c = [(RL / CSF) \bullet BW] / CR \qquad (5\text{-}5)$$

where

SV_c = Screening value for a carcinogen (mg/kg; ppm)
RL = Maximum acceptable risk level (dimensionless)
CSF = Oral cancer slope factor $(mg/kg\text{-}d)^{-1}$

and BW and CR are defined as in Equation 5-1.

5.1.3 Recommended Values for Variables in Screening Value Equations

The default values for variables used in Equations 5-4 and 5-5 to calculate SVs are based on assumptions for the general adult population. These default values are consistent with values included in the *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000)* (EPA-822-B-00-004). For risk management purposes (e.g., to protect sensitive populations such as pregnant and nursing women), states may choose to use alternative values for consumption rates, etc. different from those recommended in this section.

5.1.3.1 Dose-Response Variables—

EPA has developed oral RfDs and/or CSFs for all of the recommended target analytes in Section 4 (see Appendix G). These are maintained in the EPA Integrated Risk Information System (IRIS, 1999), an electronic database containing health risk and EPA regulatory information on approximately 400 different chemicals. IRIS is available online at:

http://www.epa.gov/iris/subst/index.html

The IRIS RfDs and CSFs are reviewed regularly and updated as necessary when new or more reliable information on the toxic or carcinogenic potency of chemicals becomes available.

When IRIS values for oral RFDs and CSFs are available, they should be used to calculate SVs for target analytes from Equations 5-4 and 5-5, respectively. It is important that the most current IRIS values for oral RfDs and CSFs be used to calculate SVs for target analytes unless otherwise recommended.

In cases where IRIS values for oral RFDs or CSFs are not available for calculating SVs for target analytes, estimates of these variables may be derived from the most recent water quality criteria (U.S. EPA, 1992e) according to procedures described in U.S. EPA (1991a, p. IV-12), or from the Classification

List of Chemicals Evaluated for Carcinogenicity Potential (U.S. EPA 1999b) from the Office of Pesticide Programs Health Effects Division.

5.1.3.2 Body Weight and Consumption Rate—

Values for the variables BW and CR in Equations 5-4 and 5-5 are given in Table 5-1 for various subpopulations including recreational and subsistence fishers. **Note:** In this third edition of this document, EPA's Office of Water uses a BW of 70 kg, a default CR of 17.5 g/d to calculate the SV for the general populations and recreational fishers, and a default CR of 142.4 g/d to calculate the SV for subsistence fishers. The CR values have been revised since the release of the previous edition.

Table 5-1. Recommended Values for Mean Body Weights (BWs) and Fish Consumption Rates (CRs) for Selected Subpopulations

Variable	Recommended value	Subpopulation
BW	70 kg	All adults (U.S. EPA, 1999a)
	78 kg	Adult males (U.S. EPA, 1985b, 1990a)
	65 kg	Adult females (U.S. EPA, 1985b, 1990a)
	12 kg	Children <3 yr (U.S. EPA, 1985b, 1990a)
	17 kg	Children 3 to <6 yr (U.S. EPA, 1985b, 1990a)
	25 kg	Children 6 to <9 yr (U.S. EPA, 1985b, 1990a)
	36 kg	Children 9 to <12 yr (U.S. EPA, 1985b, 1990a)
	51 kg	Children 12 to <15 yr (U.S. EPA, 1985b, 1990a)
	61 kg	Children 15 to <18 yr (U.S. EPA, 1985b, 1990a)
CR[a]	17.5 g/d (0.0175 kg/d)	Estimate of the 90th percentile of recreational or sport fishers (USDA/ARS, 1998) and of the average consumption of uncooked fish and shellfish from estuarine and fresh waters by recreational fishers (U.S. EPA, 2000c)
	142.4 g/d (0.1424 kg/d)	Estimate of the 99th percentile of subsistence fishers (USDA/ARS, 1998) and of the average consumption of uncooked fish and shellfish from estuarine and fresh waters by subsistence fishers (U.S. EPA, 2000c)

[a] These are recommended default consumption rates only. **Note:** When local consumption rate data are available for recreational and subsistence fishers, they should be used to calculate SVs for noncarcinogens and carcinogens by subsistence fishers, as described in Sections 5.1.1 and 5.1.2, respectively.

The default CR of 6.5 g/d used in the previous edition of Volume I was based on data from a fish consumption survey conducted in 1973 and 1974 by the National Purchase Diaries and funded by the Tuna Institute. This value represented the estimated mean per capita freshwater/estuarine finfish and shellfish consumption rate for the general U.S. population (Jacobs et al., 1998). This value has been revised based on new data from the combined 1994, 1995, and 1996 Continuing Survey of Food Intake by Individuals (CSFII) survey (USDA/ARS, 1998). The

CSFII survey is a national food consumption survey conducted by the U.S. Department of Agriculture, consisting of multistage, stratified-cluster area probability samples from all states except Alaska and Hawaii.

These data are collected over 3 consecutive days. On the first day of the survey, participants give information to an in-home interviewer, and on the second and third days, data are taken from self-administered dietary records. Meals consumed both at home and away from home are recorded. Average daily individual consumptions of fish in a given fish-by-habitat category were calculated by summing the amount of fish eaten by the individual across 3 reporting days for all fish-related food codes in a given fish-by-habitat category. The total individual consumption was then divided by three to obtain an average daily consumption rate. The 3-day individual food consumption data collection period is one during which a majority of sampled individuals did not consume any finfish or shellfish. The nonconsumption of finfish or shellfish by a majority of individuals, combined with consumption data from high-end consumers, resulted in a wide range of observed fish consumption rates. This range of fish consumption data would tend to produce distributions of fish consumption with larger variances than would be associated with a longer survey period, such as 30 days. The larger variances would reflect greater dispersion, which results in larger upper-percentile estimates, as well as upper confidence intervals associated with parameter estimates. It follows that estimates of the upper percentiles (90th and 99th percentiles) of per capita fish consumption based on 3 days of data will be consecutive with regard to risk (U.S. EPA, 1998a).

If states and tribes do not have site-specific fish consumption information concerning their recreational and subsistence fishers, it is EPA's preference that they use as fish intake assumptions the default values from the most recent 1994-1996 CSFII study (USDA/ARS, 1998). The fish consumption default values of 17.5 g/d for the general adult population and recreational fishers and 142.4 g/d for subsistence fishers used in this document are representative of fish intake for these different population groups. These values are based on risk management decisions that EPA has made after evaluating numerous fish consumption surveys (U.S. EPA, 2000c). These default values represent the uncooked weight intake of freshwater/estuarine finfish and shellfish. EPA recognizes the data gaps and uncertainties associated with the analysis of the 1994-1996 CSFII survey conducted in the process of making its default consumption rate recommendations. The estimated mean of freshwater/estuarine fish ingestion for adults is 7.50 g/d, and the median is 0 g/d. The estimated 90th percentile is 17.53 g/d; the estimated 95th percentile is 49.59 g/d; and the estimated 99th percentile is 142.41 g/d. The median value of 0 g/d may reflect the portion of individuals in the population who never eat fish as well as the limited reporting period (2 days) over which intake was actually measured. By applying as a default consumption rate the 17.5-g/d value for the general adult population, EPA intends to select a consumption rate that is protective of the majority of the population (the 90th percentile of consumers and nonconsumers according to the 1994-1996 CSFII survey data). EPA further considers this rate to be indicative of the average consumption among recreational fishers based on

averages in the studies reviewed (U.S. EPA, 2000c). Similarly, EPA believes that the assumption of 142.4 g/d is within the range of average consumption estimates for subsistence fishers based on the studies reviewed. Experts at a 1992 National Water Quality Workshop acknowledged, however, that the national survey high-end values are representative of average rates for highly exposed groups such as subsistence fishers, specific ethnic groups, or other high-risk populations. EPA is aware that some local and regional studies indicate greater fish consumption among Native Americans, Pacific Asian Americans, and other subsistence consumers and recommends the use of those studies in appropriate cases.   States and tribes have the flexibility to choose fish consumption rates higher than an average value for these populations groups. If a state has not identified a separate well-defined population of high-end consumers and believes that the national data from the 1994-1996 CSFII are representative, they may choose these consumption rates.

With respect to consumption rates, EPA recommends that states always evaluate any type of consumption pattern they believe could reasonably be occurring at a site. Evaluating additional consumption rates involves calculating additional SVs only and does not add to sampling or analytical costs.

EPA has published a review and analysis of survey methods that can be used by states to determine fish and shellfish consumption rates of local populations (U.S. EPA, 1992b, 1998b). States should consult these documents to ensure that appropriate values are selected to calculate SVs for site-specific exposure scenarios.

For any given population, there can be a sensitive subpopulation composed of individuals who may be at higher-than-average risk due to their increased exposure or their increased sensitivity to a contaminant or both. For Native American subsistence fishers, there are several exposure issues of concern that should be addressed as part of a comprehensive exposure assessment:

• **Consumption rates and dietary preferences**. Harris and Harper (1997) surveyed traditional tribal members in Oregon with a subsistence lifestyle and determined a consumption rate of 540 g/d, which included fresh, dried, and smoked fish. They also confirmed that the parts of the fish (heads, fins, tails, skeleton, and eggs) eaten by this group were not typically eaten by other groups. Another study conducted of four tribes in the Northwest that also surveyed tribal members in Oregon but did not target subsistence fishers, reported a 99th percentile ingestion rate of 390 g/d for tribal members (CRITFC, 1994). These consumption rates are much higher than the default consumption rates provided in this document for subsistence fishers and emphasize the need for identifying the consumption rate of the Native American subsistence population of concern.

• **Community characteristics** - It is important to consider family-specific fishing patterns in any exposure scenario, and attention should be paid to the role of the fishing family with respect to the tribal distribution of fish, the

sharing ethic, and providing fish for ceremonial religious events. Entire communities are exposed if fish are contaminated, and the community contaminant burden as a whole must be considered, not just the maximally exposed individual.

- **Multiple contaminant exposure** - Multiple contaminant exposure is significant for Native American subsistence fishers. A large number of contaminants are often detected in fish tissues and their combined risk associated with the higher consumption rates and dietary preferences for certain fish parts could be very high even if individual contaminants do not exceed the EPA reference dose (Harper and Harris, 1999).

- **Other exposure pathways -** For Native American subsistence fishers, overall exposure to a contaminant may be underestimated if it fails to take into account nonfood uses of fish and other animal parts that may contribute to overall exposure, such as using teeth and bones for decorations and whistles, animal skins for clothing, and rendered fish belly fat for body paint (Harper and Harris, 1999). If other wildlife species (e.g., feral mammals, turtles, waterfowl) that also live in or drink from the contaminated waterbody are eaten, or if the contaminated water is used for irrigation of crops or for livestock watering or human drinking water, the relative source contribution of these other pathways of exposure must also be considered. As with fish and wild game, plants are used by Native Americans for more than just nutrition. Daily cleaning, preparation, and consumption of plants and crafting of plant materials into household goods occurs throughout the year (Harris and Harper, 1997).

As in the general population, increased sensitivity to a chemical contaminant for Native Americans can result from factors such as an individual's underlying health status and medications, baseline dietary composition and quality, genetics, socioeconomic status, access to health care, quality of replacement protein, age, gender, pregnancy, and lactation. These factors are only partially considered in the uncertainty factor(s) used to develop the RfD (Harper and Harris, 1999).

Other important issues that need to be considered concern risk characterization and risk management. For Native American subsistence fishers, the use of an acceptable risk level of 1 in 100,000 (10^{-5}) may not be acceptable to all tribes. Each tribe has the right to decide for themselves what an acceptable level of risk is, and, in some cases, it may be zero risk (zero discharge) to protect cultural resources and uses. Ecological well-being or health is another key issue. Human and ecological health are connected in many ways and the ripple effects are often not recognized. For example, human health may be affected by injury to the environment, which affects the economy and the culture (Harper and Harris, 1999).

Native American subsistence fishers should be treated as a special high-risk group of fish consumers distinct from fishers in the general population and

distinct even from other Native American fish consumers living in more suburbanized communities. Table 5-2 compares fish consumption rates for various fisher populations within the general population and in several surveys of specific Native American tribal populations. EPA currently recommends default fish consumption rates of 17.5 g/d for the general and recreational fishers and 142.4 g/d for subsistence fishers. However, the tribal population fish consumption studies show that some Native American tribal members living in river-based communities (CRITFC, 1994) eat from 3 to 22 times more fish (from 59 g/d up to 390 g/d) than do recreational fishers, but that traditional Native American subsistence fishing families may eat up to 30 times more fish, almost 1.2 lb/d (540 g/d) (Harris and Harper, 1997). The fish consumption rate from Harris and Harper (1997) for Native American subsistence fishers is also 3.8 times higher than the EPA default consumption rate for subsistence fishers (142.4 g/d) in the general population. The difference in fish consumption is due to the fact that the Native American subsistence fisher's lifestyle is not the same as a recreational fisher's lifestyle with additional fish consumption added, nor is it the same as the "average" Native American tribal member living in a fairly suburbanized tribal community. In addition to exposures from direct consumption of contaminated fish, Native American subsistence fishers also receive more exposure to the water and sediments associated with catching and preparing fish and possibly from drinking more unfiltered river water than more suburbanized tribal community members as well. The Native American subsistence fishing population should be treated as a separate group with a unique lifestyle, distinct from recreational and subsistence fishers in the general U.S. population and also distinct from other Native American fisher populations.

5.1.3.3 Risk Level (RL)—

In this guidance document, EPA's Office of Water uses an RL of 10^{-5} to calculate screening values for the general adult population. However, states have the flexibility to choose to use an appropriate RL value typically ranging from 10^{-4} to 10^{-7}. This is the range of risk levels employed in various U.S. EPA programs. Selection of the appropriate RL is a risk management decision that is made by the state.

5.2   SCREENING VALUES FOR TARGET ANALYTES

Target analyte SVs, and the dose-response variables used to calculate them, are given in Tables 5-3 and 5-4. The SVs are provided as default values for the states to use when site-specific information on variables such as consumption rates are not available for local recreational or subsistence fisher populations.

Table 5-2. Fish Consumption Rates for Various Fisher Populations

Source	Recreational Fishers (g/d)	Subsistence Fishers (g/d)	Native American Subsistence Fishers (g/d)	Native Americans (g/d)	Basis for Consumption Rate
U.S. EPA	17.5 [a]	142.4 [a]	70 (mean) [b] 170 (95th percentile)[b]	NA	Fish consumption rate from 1994 and 1996 Continuing Survey of Food Intake by Individuals (CSFII)
Harris and Harper (1997)	NA	NA	540 (fresh, smoked and dried)	NA	Surveyed members of the Confederated Tribes of the Umatilla Indian Reservation
CRITFC (1994)	NA	NA	NA	59 (mean) 170 (95th percentile) 390 (99th percentile)	Surveyed members of the Umatilla, Nez Perce, Yakama, and Warm Springs Tribes
Toy et al. (1996)	NA	NA	NA	53 (median, males) 34 (median, females)	Surveyed members of the Tulalip Tribe
				66 (median, males) 25 (median, females)	Surveyed members of the Squaxin Island Tribe

[a] These values were revised in this 3rd edition of Volume 1 of this series (USDA/ARS, 1998)
[b] These values are from EPA's Exposure Factors Handbook (U.S. EPA, 1997b)

These SVs were calculated from Equations 5-4 or 5-5 using the following values for BW, CR, and RL and the most current IRIS values for oral RfDs and CSFs (IRIS, 1999) unless otherwise noted:

• **For noncarcinogens:**

BW = 70 kg, average adult body weight
CR = 17.5 g/d (0.0175 kg/d), estimate of average consumption of uncooked fish and shellfish from estuarine and fresh waters by recreational fishers, or
= 142.4 g/d (0.1424 kg/d), estimate of average consumption of uncooked fish and shellfish from estuarine and freshwaters by subsistence fishers.

• **For carcinogens:**

BW and CR, as above

RL = 10^{-5}, a risk level corresponding to one excess case of cancer per 100,000 individuals exposed over a 70-yr lifetime.

If both oral RfD and CSF values are available for a given target analyte, SVs for both noncarcinogenic and carcinogenic effects are listed in Table 5-2 for recreational fishers and Table 5-3 for subsistence fishers. Unless otherwise indicated,

Table 5-3. Dose-Response Variables and Recommended Screening Values (SVs) for Target Analytes - Recreational Fishers[a]

Target analyte	Noncarcinogens RfD (mg/kg-d)	Carcinogens CSF (mg/kg-d)$^{-1}$	SV[b] (ppm) Noncarcinogens[b]	SV[b] (ppm) Carcinogens[b] (RL=10^{-5})
Metals				
Arsenic (inorganic)[c]	3×10^{-4}	1.5	1.2	0.026
Cadmium	1×10^{-3}	NA	4.0	-
Mercury (methylmercury)[d]	1×10^{-4}	NA	0.4	-
Selenium	5×10^{-3}	NA	20	-
Tributyltin[e]	3×10^{-4}	NA	1.2	-
Organochlorine Pesticides				
Total chlordane (sum of cis- and trans-chlordane, cis- and trans-nonachlor, and oxychlordane)[f]	5×10^{-4}	0.35	2.0	0.114
Total DDT (sum of 4,4'- and 2,4'- isomers of DDT, DDE, and DDD)[g]	5×10^{-4}	0.34	2.0	0.117
Dicofol[h]	4×10^{-4}	NA[i]	1.6	2.5
Dieldrin	5×10^{-5}	16	0.2	2.50×10^{-3}
Endosulfan (I and II)[j]	6×10^{-3}	NA	24	-
Endrin	3×10^{-4}	NA	1.2	-
Heptachlor epoxide	1.3×10^{-5}	9.1	5.2×10^{-2}	4.39×10^{-3}
Hexachlorobenzene	8×10^{-4}	1.6	3.2	2.50×10^{-2}
Lindane (g-hexachlorocyclohexane; g-HCH)[k]	3×10^{-4}	1.3	1.2	3.07×10^{-2}
Mirex	2×10^{-4}	NA[l]	0.8	-
Toxaphene[j,m]	2.5×10^{-4}	1.1	1.0	3.63×10^{-2}
Organophosphate Pesticides				
Chlorpyrifos[n]	3×10^{-4}	NA	1.2	-
Diazinon[o]	7×10^{-4}	NA	2.8	-
Disulfoton	4×10^{-5}	NA	0.16	-
Ethion	5×10^{-4}	NA	2.0	-
Terbufos[p]	2×10^{-5}	NA	0.08	-
Chlorophenoxy Herbicides				
Oxyfluorfen[q]	3×10^{-3}	7.32×10^{-2}	12	5.46×10^{-1}
PAHs[r]	NA	7.3	-	5.47×10^{-3}
PCBs				
Total PCBs[s]	2×10^{-5}	2.0	0.08	0.02
Dioxins/furans[t]	NA	1.56×10^{5}	-	2.56×10^{-7}

NA = Not available in EPA's Integrated Risk Information System (IRIS, 1999).
DDD = p,p'-dichlorodiphenyldichloroethane
DDT = p,p'-dichlorodiphenyltrichloroethane
DDE = p,p'-dichlorodiphenlydichloroethylene

PAH = Polycyclic aromatic hydrocarbon
PCB = Polychlorinated biphenyl
RfD = Oral reference dose (mg/kg-d)
CSF = Cancer slope factor (mg/kg-d)$^{-1}$

Table 5-3. (continued)

[a] Based on fish consumption rate of 17.5 g/d, 70kg body weight and, for carcinogens, 10^{-5} risk level and 70-yr lifetime. Unless otherwise noted, values listed are the most current oral RfDs and CSF in EPA's IRIS database (IRIS, 1999).

[b] The shaded screening value (SV) is the recommended SV for each target analyte. States should note that the screening values listed may be below analytical detection limits achievable for some of the target analytes. Please see Table 8-4 for detection limits.

[c] Total inorganic arsenic rather than total arsenic should be determined.

[d] Because most mercury in fish and shellfish tissue is present primarily as methylmercury (NAS, 1991;Tollefson, 1989) and because of the relatively high cost of analyzing for methylmercury, it is recommended that total mercury be analyzed and the conservative assumption be made that all mercury is present as methylmercury. This approach is deemed to be most protective of human health and most cost-effective. The National Academy of Sciences conducted an independent assessment of the RfD for methylmercury. They concluded that "On the basis of its evaluation, the committee's consensus is that the value of EPA's current RfD for methylmercury, 0.1μg/kg per day, is a scientifically justifiable level for the protection of human health".

[e] The RfD value listed is for tributyltin oxide (IRIS, 1999).

[f] The RfD and CSF values listed are derived from studies using technical-grade chlordane (IRIS, 1999) for the *cis*- and *trans*-chlordane isomers or the major chlordane metabolite, oxychlordane, or for the chlordane impurities *cis*- and *trans*-nonachlor. It is recommended that total chlordane be determined by summing the concentrations of *cis*- and *trans*-chlordane, *cis*- and *trans*-nonachlor, and oxychlordane.

[g] The RfD value listed is for DDT. The CSF value (0.34) is for total DDT sum of DDT, DDE and DDD); the CSF value for DDD is 0.24. It is recommended that the total concentration of DDT include the 2,4'- and 4,4'-isomers of DDT and its metabolites, DDE and DDD.

[h] The RfD value is from Office of Pesticide Programs Reregistration Eligibility Decision (RED) for Dicofol (EPA, 1998c).

[i] The CSF for dicofol was withdrawn from IRIS pending further review by the CRAVE Agency Work Group (IRIS, 1999).

[j] The RfD value listed is from the Office of Pesticide Program's Reference Dose Tracking Report (U.S. EPA, 1997).

[k] IRIS (1999) has not provided a CSF for lindane. The CSF value listed for lindane was calculated from the water quality criteria (0.063 mg/L) (U.S. EPA, 1992f).

[l] No CSF or cancer classification is available for mirex. This compound is undergoing further review by the CRAVE Agency Work Group (IRIS, 1999)

[m] The RfD value has been agreed upon by the Office of Pesticide Programs and the Office of Water.

[n] Because of the potential for adverse neurological developmental effects from chlorpyrifos, EPA recommends the use of a Population Adjusted Dose (PAD) of 3×10^{-5} for infants, children under the age of 6 years, and women ages 13 to 50 years (U.S. EPA, 2000b).

[o] The RfD value is from a memorandum dated April 1, 1998, Diazinon:-Report of the Hazard Identification Assessment Review Committee. HED Doc. No. 012558.

[p] The RfD value listed is from a memorandum dated September 25, 1997; Terbufos-FQPA Requirement- Report of the Hazard Idenification Review.

[q] The CSF value is from the Office of Pesticide Programs List of Chemicals Evaluated for Carcinogenic Potential (U.S. EPA, 1999b).

[r] The CSF value listed is for benzo[*a*]pyrene. Values for other PAHs are not currently available in IRIS (1999). It is recommended that tissue samples be analyzed for benzo[*a*]pyrene and 14 other PAHs, and that the order-of-magnitude relative potencies given for these PAHs (Nisbet and LaGoy, 1992; U.S. EPA, 1993c) be used to calculate a potency equivalency concentration (PEC) for each sample (see Section 5.3.2.4).

[s] Total PCBs may be determined as the sum of congeners or Aroclors. The RfD is based on Aroclor 1254 and should be applied to total PCBs. The CSF is based on a carcinogenicity assessment of Aroclors 1260, 1254, 1242, and 1016. The CSF presented is the upper-bound slope factor for food chain exposure. The central estimate is 1.0 (IRIS, 1999).

[t] The CSF value listed is for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) (HEAST, 1997). It is recommended that the 17 2,3,7,8-substituted tetra- through octa-chlorinated dibenzo-*p*-dioxins and dibenzofurans and the 12 dioxin-like PCBs be determined and a toxicity-weighted total concentration be calculated for each sample, using the method for estimating toxicity equivalency concentrations (TEQs) (Van den Berg et al., 1998).

Table 5-4. Dose-Response Variables and Recommended Screening Values (SVs) for Target Analytes - Subsistence Fishers[a]

Target analyte	Noncarcinogens RfD (mg/kg-d)	Carcinogens CSF (mg/kg-d)$^{-1}$	SV[b] (ppm) Noncarcinogens[b]	SV[b] (ppm) Carcinogens[b] (RL=10^{-5})
Metals				
Arsenic (inorganic)[c]	3×10^{-4}	1.5	0.147	3.27×10^{-3}
Cadmium	1×10^{-3}	NA	0.491	-
Mercury (methylmercury)[d]	1×10^{-4}	NA	0.049	-
Selenium	5×10^{-3}	NA	2.457	-
Tributyltin[e]	3×10^{-4}	NA	0.147	-
Organochlorine Pesticides				
Total chlordane (sum of cis- and trans-chlordane, cis- and trans-nonachlor, and oxychlordane)[f]	5×10^{-4}	0.35	0.245	1.40×10^{-2}
Total DDT (sum of 4,4'- and 2,4'- isomers of DDT, DDE, and DDD)[g]	5×10^{-4}	0.34	0.245	1.44×10^{-2}
Dicofol[h]	4×10^{-4}	NA[i]	0.196	-
Dieldrin	5×10^{-5}	16	0.024	3.07×10^{-4}
Endosulfan (I and II)[j]	6×10^{-3}	NA	2.949	-
Endrin	3×10^{-4}	NA	0.147	-
Heptachlor epoxide	1.3×10^{-5}	9.1	6.39×10^{-3}	5.40×10^{-4}
Hexachlorobenzene	8×10^{-4}	1.6	0.393	3.07×10^{-3}
Lindane (γ-hexachlorocyclohexane; γ-HCH)[k]	3×10^{-4}	1.3	0.147	3.78×10^{-3}
Mirex	2×10^{-4}	NA[l]	0.098	-
Toxaphene[j,m]	2.5×10^{-4}	1.1	0.122	4.46×10^{-3}
Organophosphate Pesticides				
Chlorpyrifos[n]	3×10^{-4}	NA	0.147	-
Diazinon[o]	7×10^{-4}	NA	0.344	-
Disulfoton	4×10^{-5}	NA	0.019	-
Ethion	5×10^{-4}	NA	0.245	-
Terbufos[p]	2×10^{-5}	NA	0.009	-
Chlorophenoxy Herbicides				
Oxyfluorfen[q]	3×10^{-3}	7.32×10^{-2}	1.474	6.71×10^{-2}
PAHs[r]	NA	7.3	-	6.73×10^{-4}
PCBs				
Total PCBs[s]	2×10^{-5}	2.0	9.83×10^{-3}	2.45×10^{-3}
Dioxins/furans[t]	NA	1.56×10^{5}	-	3.15×10^{-8}

NA	= Not available in EPA's Integrated Risk Information System (IRIS, 1999).	PAH	= Polycyclic aromatic hydrocarbon
		PCB	= Polychlorinated biphenyl
DDD	= p,p'-dichlorodiphenyldichloroethane	RfD	= Oral reference dose (mg/kg-d)
DDT	= p,p'-dichlorodiphenyltrichloroethane	CSF	= Cancer slope factor (mg/kg-d)$^{-1}$
DDE	= p,p'-dichlorodiphenlydichloroethylene		

Table 5-4. (continued)

[a] Based on fish consumption rate of 142.4 g/d, 70kg body weight and, for carcinogens, 10^{-5} risk level and 70-yr lifetime. Unless otherwise noted, values listed are the most current oral RfDs and CSF in EPA's IRIS database (IRIS, 1999)

[b] The shaded screening value (SV) is the recommended SV for each target analyte. States should note that the screening values listed may be below analytical detection limits achievable for some of the target analytes. Please see Table 8-4 for detection limits.

[c] Total inorganic arsenic rather than total arsenic should be determined.

[d] Because most mercury in fish and shellfish tissue is present primarily as methylmercury (NAS, 1991;Tollefson, 1989) and because of the relatively high cost of analyzing for methylmercury, it is recommended that total mercury be analyzed and the conservative assumption be made that all mercury is present as methylmercury. This approach is deemed to be most protective of human health and most cost-effective. The National Academy of Sciences conducted an independent assessment of the RfD for methylmercury. They concluded that "On the basis of its evaluation, the committee's consensus is that the value of EPA's current RfD for methylmercury, $0.1\mu g/kg$ per day, is a scientifically justifiable level for the protection of human health".

[e] The RfD value listed is for tributyltin oxide (IRIS, 1999).

[f] The RfD and CSF values listed are derived from studies using technical-grade chlordane (IRIS, 1999) for the *cis-* and *trans-*chlordane isomers or the major chlordane metabolite, oxychlordane, or for the chlordane impurities *cis-* and *trans-*nonachlor. It is recommended that total chlordane be determined by summing the concentrations of *cis-* and *trans-*chlordane, *cis-* and *trans-*nonachlor, and oxychlordane.

[g] The RfD value listed is for DDT. The CSF value (0.34) is for total DDT sum of DDT, DDE and DDD); the CSF value for DDD is 0.24. It is recommended that the total concentration of DDT include the 2,4'- and 4,4'-isomers of DDT and its metabolites, DDE and DDD.

[h] The RfD value is from Office of Pesticide Programs Reregistration Eligibility Decision (RED) for Dicofol (EPA, 1998c).

[i] The CSF for dicofol was withdrawn from IRIS pending further review by the CRAVE Agency Work Group (IRIS, 1999).

[j] The RfD value listed is from the Office of Pesticide Program's Reference Dose Tracking Report (U.S. EPA, 1997).

[k] IRIS (1999) has not provided a CSF for lindane. The CSF value listed for lindane was calculated from the water quality criteria (0.063 mg/L) (U.S. EPA, 1992f).

[l] No CSF or cancer classification is available for mirex. This compound is undergoing further review by the CRAVE Agency Work Group (IRIS, 1999)

[m] The RfD value has been agreed upon by the Office of Pesticide Programs and the Office of Water.

[n] Because of the potential for adverse neurological developmental effects from chlorpyrifos, EPA recommends the use of a Population Adjusted Dose (PAD) of 3×10^{-5} for infants, children under the age of 6 years, and women ages 13 to 50 years (U.S. EPA, 2000b).

[o] The RfD value is from a memorandum dated April 1, 1998, Diazinon:-Report of the Hazard Identification Assessment Review Committee. HED Doc. No. 012558.

[p] The RfD value listed is from a memorandum dated September 25, 1997; Terbufos-FQPA Requirement- Report of the Hazard Idenification Review.

[q] The CSF value is from the Office of Pesticide Programs List of Chemicals Evaluated for Carcinogenic Potential (U.S. EPA, 1999b).

[r] The CSF value listed is for benzo[*a*]pyrene. Values for other PAHs are not currently available in IRIS (1999). It is recommended that tissue samples be analyzed for benzo[*a*]pyrene and 14 other PAHs, and that the order-of-magnitude relative potencies given for these PAHs (Nisbet and LaGoy, 1992; U.S. EPA, 1993c) be used to calculate a potency equivalency concentration (PEC) for each sample (see Section 5.3.2.4).

[s] Total PCBs may be determined as the sum of congeners or Aroclors. The RfD is based on Aroclor 1254 and should be applied to total PCBs. The CSF is based on a carcinogenicity assessment of Aroclors 1260, 1254, 1242, and 1016. The CSF presented is the upper-bound slope factor for food chain exposure. The central estimate is 1.0 (IRIS, 1999).

[t] The CSF value listed is for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) (HEAST, 1997). It is recommended that the 17 2,3,7,8-substituted tetra- through octa-chlorinated dibenzo-*p*-dioxins and dibenzofurans and the 12 dioxin-like PCBs be determined and a toxicity-weighted total concentration be calculated for each sample, using the method for estimating toxicity equivalency concentrations (TEQs) (Van den Berg et al., 1998).

the lower of the two SVs (generally, the SV for carcinogenic effects) should be used for the respective fisher population. EPA recommends that the SVs in the shaded boxes (Tables 5-3 and 5-4) be used by states when making the decision to implement Tier 2 intensive monitoring. However, states may choose to adjust these SVs for specific target analytes for the protection of sensitive populations (e.g., pregnant women, nursing mothers, and children or for recreational or subsistence fishers based on site-specific consumption rates). EPA recognizes that states may use higher CRs that are more appropriate for recreational and subsistence fishers in calculating SVs for use in their jurisdictions rather than the EPA default values of 17.5 g/d CR for recreational fishers used to calculate the SVs shown in Table 5-3 and the 142.4 g/d CR for subsistence fishers used to calculate the SVs shown in Table 5-4.

Note: States should use the same SV for a given target analyte in both screening and intensive studies. Therefore, it is critical that states clearly define their program objectives and accurately characterize the target fish-consuming population(s) of concern to ensure that appropriate SVs are selected. If the selected analytical methodology is not sensitive enough to reliably quantitate target analytes at or below selected SVs (see Section 8.2.2 and Table 8-4), program managers must determine appropriate fish consumption guidance based on the lowest detectable concentrations or provide justification for adjusting SVs to values at or above achievable method detection limits.   It should be emphasized that when SVs are below method detection limits, the failure to detect a target analyte cannot be assumed to indicate that there is no cause for concern for human health effects.

States should recognize the importance of ensuring that the analytical method selected for quantification of any target analyte must have a method detection limit (MDL) lower than the risk-based screening values calculated using the EPA methodology for noncarcinogenic and carcinogenic effects of the target analyte. If the method detection limit for a specific target analyte is higher than the target analyte SV, the following procedure is recommended as a means to reduce the problem of interpreting data results for chemicals that fall in this category. For example, if fish tissue residue values for several replicate samples are above the MDL while other data values are reported as below the method detection limit (<MDL) including not detected (e.g., no observed response), the state may make a risk management decision to use a value of one-half the MDL as the residue concentration in their risk assessment for those data below the MDL rather than using a value of zero. In this way, the calculated mean target analyte concentration for a group of replicate samples may be higher than the SV. If all of the replicate samples from a particular monitoring site are below the MDL or are not detected, the state may choose to use one-half MDL value for all not detected values rather than a value of zero. The use of one-half MDL rather than zero for these data (< MDL) is a risk management policy decision that should be made by the state.

For noncarcinogens, adjusted SVs should be calculated from Equation 5-4 using appropriate alternative values of BW and/or CR. For carcinogens, adjusted SVs

should be calculated from Equation 5-5 using an RL ranging from 10^{-4} to 10^{-7} and/or sufficiently protective alternative values of BW and CR. Examples of SVs calculated for selected populations of concern and for RL values ranging from 10^{-4} to 10^{-7} are given in Table 5-5.

The need to accurately characterize the target fisher population of interest in order to establish sufficiently protective SVs cannot be overemphasized. For example, the recommended consumption rate of 142.4 g/d for subsistence fishers may be an underestimate of consumption rate and exposures for some subsistence populations such as Native American subsistence fishers (see Section 5.1.3.2). In a recent study of a Native American subsistence fishing population, an average daily consumption rate for these subsistence fishers was estimated to be 540 g/d (Harris and Harper, 1997). Using this average consumption rate and an estimated average body weight of 70 kg, the SV for cadmium (RfD = 1 x 10^{-3} mg/kg/d) is, from Equation 5-4,

$$SV = (0.001 \text{ mg/kg-d} \bullet 70 \text{ kg}) / (0.540 \text{ kg/d}) = 0.129 \text{ mg/kg (ppm)}. \quad (5\text{-}7)$$

This value is almost four times lower than the SV of 0.491 ppm for cadmium based on the EPA default consumption rate of 142.4 g/d for subsistence fishers, as shown in Table 5-4.

5.3   COMPARISON OF TARGET ANALYTE CONCENTRATIONS WITH SCREENING VALUES

As noted previously, the same SV for a specific target analyte should be used in both the screening and intensive studies. The measured concentrations of target analytes in fish or shellfish tissue should be compared with their respective SVs in both screening and intensive studies to determine the need for additional monitoring and risk assessment.

Recommended procedures for comparing target analyte concentrations with SVs are provided below. Related guidance on data analysis is given in Section 9.1.

5.3.1   Metals

5.3.1.1 Arsenic—

Most of the arsenic present in fish and shellfish tissue is organic arsenic, primarily pentavalent arsenobetaine, which has been shown in numerous studies to be metabolically inert and nontoxic (Brown et al., 1990; Cannon et al., 1983; Charbonneau et al., 1978; Bos et al., 1985; Kaise et al. 1985; Luten et al., 1982; Sabbioni et al., 1991; Siewicki, 1981; Bryce et al., 1982; Vahter et al., 1983; Yamauchi et al., 1986). Inorganic arsenic, which is of concern for human health effects (ATSDR, 1998a; WHO, 1989), is generally found in seafood at concentra-

Table 5-5. Example Screening Values (SVs) for Various Target Populations and Risk Levels (RLs)[a]

Chemical	Target population[b]	CR[c]	BW	RfD	CSF	RL	SV (ppm)
Noncarcinogens							
Chlorpyrifos	Recreational fisher	17.5	70	3×10^{-4}	—	—	1.2
	Children (<6 yr)	6.5	17[d]	3×10^{-5e}	—	—	0.078
	Subsistence fisher	142.4	70	3×10^{-4}	—	—	0.147
Cadmium	Recreational fisher	17.5	70	1×10^{-3}	—	—	4.0
	Children	6.5	17[d]	1×10^{-3}	—	—	2.6
	Subsistence fisher	142.4	70	1×10^{-3}	—	—	0.491
Carcinogens							
Lindane	Recreational fisher	17.5	70	—	1.3 1.3 1.3 1.3	10^{-4} 10^{-5} 10^{-6} 10^{-7}	3.07×10^{-1} 3.07×10^{-2} 3.07×10^{-3} 3.07×10^{-4}
	Children	6.5	17[d]	—	1.3 1.3 1.3 1.3	10^{-4} 10^{-5} 10^{-6} 10^{-7}	1.98×10^{-1} 1.98×10^{-2} 1.98×10^{-3} 1.98×10^{-4}
	Subsistence fisher	142.4	70	—	1.3 1.3 1.3 1.3	10^{-4} 10^{-5} 10^{-6} 10^{-7}	3.78×10^{-2} 3.78×10^{-3} 3.78×10^{-4} 3.78×10^{-5}
Toxaphene	Recreational fisher	17.5	70	—	1.1 1.1 1.1 1.1	10^{-4} 10^{-5} 10^{-6} 10^{-7}	3.63×10^{-1} 3.63×10^{-2} 3.63×10^{-3} 3.63×10^{-4}
	Children	6.5	17[d]	—	1.1 1.1 1.1 1.1	10^{-4} 10^{-5} 10^{-6} 10^{-7}	2.35×10^{-1} 2.35×10^{-2} 2.35×10^{-3} 2.35×10^{-4}
	Subsistence fisher	142.5	70	—	1.1 1.1 1.1 1.1	10^{-4} 10^{-5} 10^{-6} 10^{-7}	4.6×10^{-2} 4.6×10^{-3} 4.6×10^{-4} 4.6×10^{-5}

CR = Mean daily fish or shellfish consumption rate (uncooked weight), averaged over a 70-yr lifetime for the population of concern (g/d).
BW = Mean body weight, estimated for the population of concern (kg).
RfD = Oral reference dose for noncarcinogens (mg/kg-d).
CSF = Oral slope factor for carcinogens $(mg/kg-d)^{-1}$.
RL = Maximum acceptable risk level for carcinogens (dimensionless).

[a] See Equations 5-4 and 5-5.
[b] See Tables 5-1, 5-2, 5-3 and 5-4 for information on target populations.
[c] To calculate SVs, the CRs given in this table must be divided by 1,000 to convert g/d to kg/d.
[d] BW used is for children 3 to <6 yr (see Table 5-1).
e Because of the potential for adverse neurological developmental effects, EPA recommends the use of a Population Adjusted Dose for chlorpyrifos of 3×10^{-5} mg/kg-d for infants, children to the age of 6, and women ages 13 to 50 years (U.S. EPA, 2000b).

tions ranging from <1 to 20 percent of the total arsenic concentration (Edmonds and Francesconi, 1993; Nraigu and Simmons, 1990). It is recommended that, in both screening and intensive studies, total inorganic arsenic tissue concentrations be determined for comparison with the recommended SV for chronic oral exposure. This approach is more rigorous than the current FDA-recommended method of analyzing for total arsenic and estimating inorganic arsenic concentrations based on the assumption that 10 percent of the total arsenic in fish tissue is in the inorganic form (U.S. FDA, 1993). Although the cost of analysis for inorganic arsenic (see Table 8-5) may be three to five times greater than for total arsenic, the increased cost is justified to ensure that the most accurate data are obtained for quantitative assessment of human health risks.

5.3.1.2 Cadmium, Mercury, and Selenium—

For cadmium, mercury, and selenium, the total metal tissue concentration should be determined for comparison with the appropriate target population SV.

Because most mercury in fish and shellfish tissue is present as methylmercury (Kannan et al., 1998; NAS, 1991; Tollefson, 1989), and because of the relatively high analytical cost for methylmercury, it is recommended that total mercury be determined and the conservative assumption be made that all mercury is present as methylmercury. The determination of methylmercury in fish tissue is not recommended even though methylmercury is the compound of greatest concern for human health (NAS, 1991; Tollefson, 1989) and the recommended SVs are for methylmercury (see Tables 5-3 and 5-4). This approach is deemed to be most protective of human health and most cost-effective.

5.3.1.3 Tributyltin—

Tissue samples should be analyzed specifically for tributyltin for comparison with the recommended target population SVs for this compound (see Tables 5-3 and 5-4).

5.3.2 Organics

For each of the recommended organic target analytes that are single compounds, the determination of tissue concentration and comparison with the appropriate SV is straightforward. However, for those organic target analytes that include a parent compound and structurally similar compounds or metabolites (i.e., total chlordane, total DDT, endosulfan I and II) or that represent classes of compounds (i.e., PAHs, PCBs, dioxins/furans, or toxaphene), additional guidance is necessary to ensure that a consistent approach is used to determine appropriate target analyte concentrations for comparison with recommended SVs.

5.3.2.1 Chlordane—

The SVs for total chlordane are derived from technical-grade chlordane. Oral cancer slope factors are not available in IRIS (1999) for *cis-* and *trans-*chlordane, *cis-* and *trans-*nonachlor, and oxychlordane. At this time, as a conservative approach, EPA recommends that, in both screening and intensive studies, the concentrations of all chlordane constituents (*cis-* and *trans-*chlordane, *cis-* and *trans-*nonachlor) and the metabolite of chlordane (oxychlordane) be determined and summed to give a total chlordane concentration for comparison with the recommended SVs (see Tables 5-3 and 5-4).

5.3.2.2 DDT—

DDT and its metabolites (i.e., the 4,4'- and 2,4'-isomers of DDE and DDD) are all potent toxicants, DDE isomers being the most prevalent in the environment. As a conservative approach, EPA recommends that, in both screening and intensive studies, the concentrations of 4,4'- and 2,4'-DDT and their 4,4' and 2,4'-DDE and DDD metabolites be determined and a total DDT concentration be calculated for comparison with the recommended SVs for total DDT (see Tables 5-3 and 5-4).

5.3.2.3 Endosulfan—

Endosulfan collectively refers to two stereoisomers designated I and II. At this time, for both screening and intensive studies, EPA recommends that the concentrations of the two endosulfan constituents (endosulfan I and II) be determined and summed to give a total endosulfan concentration for comparison with the recommended SVs for total endosulfan.

5.3.2.4 Toxaphene—

The SVs for toxaphene are derived from technical-grade toxaphene, a mixture of approximately 670 chlorinated camphenes (ATSDR, 1996). At this time, determination of total toxaphene is recommended rather than individual congener analysis. Research is currently under way to determine the relative health risks of the toxaphene congeners. In the future, it may be possible to develop a congener-specific quantitative risk assessment approach for toxaphene similar to that for PCBs and dioxins/furans. The total toxaphene concentration should be analyzed for comparison with the recommended SVs for toxaphene (see Tables 5-3 and 5-4).

5.3.2.5 PAHs—

Although several PAHs have been classified as B2 carcinogens (probable human carcinogens), benzo[*a*]pyrene is the only PAH for which a CSF is currently available in IRIS (1999). As a result, EPA quantitative risk estimates for PAH mixtures have often assumed that all carcinogenic PAHs are equipotent to benzo[*a*]pyrene. The EPA Office of Health and Environmental Assessment has

issued guidance for quantitative risk assessment of PAHs (Nisbet and LaGoy, 1992; U.S. EPA, 1993c) in which an estimated order of potential potency for 14 PAHs relative to benzo[*a*]pyrene is recommended, as shown in Table 5-6. Based on this guidance, EPA recommends that, in both screening and intensive studies, tissue samples be analyzed for the PAHs shown in Table 5-6 and that a potency-weighted total concentration be calculated for each sample for comparison with the recommended SVs for benzo[*a*]pyrene (see Tables 5-3 and 5-4). This potency equivalency concentration should be calculated using the following equation:

$$PEC = \sum_{i} (RP_i \cdot C_i) \tag{5-8}$$

where

RP_i = Relative potency for the ith PAH (from Table 5-6)
C_i = Concentration of the ith PAH.

Table 5-6. Toxicity Equivalency Factors for Various PAHs

Compound	Toxicity Equivalency Factor (TEF)
Dibenz[*a,h*]anthracene	5
Benzo[*a*]pyrene	1
Benz[*a*]anthracene	0
Benzo[*b*]fluoranthene	0.1
Benzo[*k*]fluoranthene	0.1
Indeno[*1,2,3-cd*]pyrene	0.1
Anthracene	0.01
Benzo[*g,h,i*]perylene	0.01
Chrysene	0.01
Acenaphthene	0.001
Acenaphthylene	0.001
Fluoranthene	0.001
Fluorene	0.001
Phenanthrene	0.001
Pyrene	0.001

Source: Nisbet and LaGoy (1992).

5.3.2.6 PCBs—

Using the approach for PCB analysis recommended by the EPA Office of Water (see Section 4.3.6), total PCB concentrations may be determined as the sum of Aroclor equivalents in screening studies. For intensive studies, the total PCB concentration should be determined as the sum of PCB congeners or the sum of homologue groups. The total PCB concentration should be compared with the recommended SVs for PCBs (see Tables 5-3 and 5-4). The EPA Office of Water recognizes the potential problems associated with PCB congener analysis (i.e.,

standard methods are not yet available but are under development, relatively high analytical cost, and limited number of qualified laboratories), but is recommending these methods for intensive studies because Aroclor analysis does not adequately represent bioconcentrated PCB mixtures found in fish tissue. EPA has developed a draft method for selected PCB congeners (Method 1668) (U.S. EPA, 1997a). This method is being tested and may be revised to include all PCB congeners. Currently, Method 680 is available for PCB homologue analysis.

5.3.2.7 Dioxins and Dibenzofurans—

Note: At this time, EPA's Office of Research and Development is reevaluating the potency of dioxins/furans. Consequently, the following recommendation may change pending the results of this reevaluation.

It is recommended in both screening and intensive studies that the 17 2,3,7,8-substituted tetra- through octa-chlorinated PCDDs and PCDFs and the 12 coplanar congeners with dioxin-like effects be determined and that a toxicity-weighted total concentration be calculated for each sample for comparison with the recommended SVs for 2,3,7,8-TCDD (see Tables 5-3 and 5-4).

The method for estimating total TEQ (Van den Berg et al., 1998) should be used to estimate TCDD equivalent concentrations according to the following equation:

$$TEQ = \sum_i (TEF_i \bullet C_i) \tag{5-9}$$

where

TEF_i = Toxicity equivalency factor for the ith congener (relative to 2,3,7,8-TCDD)

C_i = Concentration of the ith congener.

TEFs for the 2,3,7,8-substituted tetra- through octa-PCDDs and PCDFs and the 12 dioxin-like PCBs are shown in Table 5-7. Note: TEFs for five congeners have changed over those TEFs recommended by Barnes and Bellin (1989).

Table 5-7. Toxicity Equivalency Factors (TEFs) for Tetra-through Octa-Chlorinated Dibenzo-p-Dioxins and Dibenzofurans and Dioxin-Like PCBs

Analyte	Old TEF-89	TEF-98
Dioxins[a]		
2,3,7,8-TCDD	1.00	1.00
1,2,3,7,8-PeCDD	0.50	1.00*
1,2,3,4,7,8-HxCDD	0.10	0.10
1,2,3,6,7,8-HxCDD	0.10	0.10
1,2,3,7,8,9-HxCDD	0.10	0.10
1,2,3,4,6,7,8-HpCDD	0.01	0.01
OCDD	0.001	0.0001*
Furans[a]		
2,3,7,8-TCDF	0.10	0.10
1,2,3,7,8-PeCDF	0.05	0.05
2,3,4,7,8-PeCDF	0.50	0.50
1,2,3,4,7,8-HxCDF	0.10	0.10
1,2,3,6,7,8-HxCDF	0.10	0.10
1,2,3,7,8,9-HxCDF	0.10	0.10
2,3,4,6,7,8-HxCDF	0.10	0.10
1,2,3,4,6,7,8-HpCDF	0.01	0.01
1,2,3,4,7,8,9-HpCDF	0.01	0.01
OCDF	0.001	0.0001*
PCBs		
3,3',4,4'-TetraCB (77)	0.0005	0.0001*
3,4,4',5-TetraCB (81)	not available	0.0001*
2,3,3',4,4'-PentaCB (105)	0.0001	0.0001
2,3,4,4',5-PentaCB (114)	0.0005	0.0005
2,3',4,4',5-PentaCB (118)	0.0001	0.0001
2',3,4,4',5-PentaCB (123)	0.0001	0.0001
3,3',4,4',5-PentaCB (126)	0.1	0.1
2,3,3',4,4',5-HexaCB (156)	0.0005	0.0005
2,3,3',4,4',5'-HexaCB (157)	0.0005	0.0005
2,3',4,4',5,5'-HexaCB (167)	0.00001	0.00001
3,3',4,4',5,5'-HexaCB (169)	0.01	0.01
2,3,3',4,4',5,5--HexaCB (189)	0.0001	0.0001

Sources: Barnes and Bellin, 1989; Van den Berg et al., 1998.

*Note: TEF-98 value changed from TEF-89 value.

[a]TEFs for all non-2,3,7,8-substituted congeners are zero.