

Appendix I. Fish Bioaccumulation Factors

I.1 Introduction

The algorithm used in the AB-2588 risk assessment to estimate exposure to contaminants via intake of angler-caught fish contains a chemical-specific variable known as a bioaccumulation factor (BAF). Fish are exposed to chemicals that are deposited into their aqueous environment from airborne sources. Only a small subset of Hot Spots chemicals are wholly or partially in the particulate phase and thus subject to deposition. These chemicals include semivolatile organic chemicals and toxic metals and semi-metals. Table I-1 presents the chemical-specific BAF values derived by OEHHA for the Hot Spots program. This appendix outlines the methods used for estimating BAFs and summarizes the available literature used for deriving the chemical-specific BAFs recommended in Table I-1.

Table I-1. Recommended Default Fish BAFs for Edible (Muscle) Tissue^a

Organic Chemicals^b	
Diethylhexylphthalate (DEHP)	40
Hexachlorobenzene (HCB)	80,000
Hexachlorocyclohexanes (HCH)	3000
Pentachlorophenol	^c
Polycyclic aromatic hydrocarbons (PAH)	800
Polychlorinated biphenyls (PCB)	2,000,000
Polychlorinated dibenzo-p-dioxins and furans (PCDD/F)	300,000
Inorganic Metals and Semi-Metals^d	
Arsenic	20
Beryllium	40
Cadmium	40
Chromium	20
Lead	20
Inorganic mercury	80
Nickel	20
Selenium	1000

^a All BAFs were rounded to the nearest whole number.

^b Lipid-normalized to adult rainbow trout with 4% lipid content in muscle tissue, and based on the freely dissolved fraction of organic chemical in water under conditions of average POC and DOC in U.S. lakes and other water bodies.

^c To be assessed for bioaccumulation in fish

^d Based on wet weight muscle tissue concentration, and on the total water concentration of the metal or semi-metal in water.

Accumulation of a chemical in fish is a physical-chemical process by which chemicals tend to apportion themselves between the fish and the fish's contact with its environment. The environment in this case is defined broadly to include the water, food that the fish eats, and contact with materials other than water. Accumulation of

chemicals in fish may result in human exposure from fish consumption, which may be significant relative to other exposure pathways considered in the Hot Spots Program.

The Hot Spots program previously only considered the physical-chemical transfer of chemicals from the water column to the fish. This approach does not address other potentially important sources of toxic contaminant contributions to fish and can thus underestimate human exposure for some chemicals. This issue is discussed in more detail below.

The BAF reflects the uptake and retention of a chemical by fish from all surrounding media (e.g., water, food, sediment) when a steady-state concentration has been reached between the fish and the media. The BAF will vary depending on the organ or tissue of interest, but is also often expressed as the chemical accumulation in the whole fish. The BAF is defined under the Hot Spots program as representing the ratio of a concentration of a chemical in edible tissue, specifically the whole muscle tissue or muscle lipid fraction, to its concentration in the surrounding water in situations where the organism and its food are exposed and the ratio does not change substantially over time. The BAF is calculated as:

$$\text{BAF} = C_t / C_w \quad (\text{Eq. I.1})$$

where:

C_t = concentration of the chemical in wet tissue

C_w = concentration of chemical in water

Lipophilic, organic chemicals tend to concentrate in the lipid fraction of fish and the resulting BAF is often lipid normalized to express the concentration of chemical in lipid (see below). The concentration of a chemical in water is often expressed in milligrams or micrograms of chemical per liter of water (i.e., mg/L or $\mu\text{g/L}$) and the concentration in tissue is often expressed in μg of chemical per kg tissue ($\mu\text{g/kg}$, or ppb). The BAF can be represented as a unitless factor through conversion of a volume of water to a mass (1 L water \approx 1 kg), or simply represented in L/kg.

In some instances, the BAF may be based on a bioconcentration factor (BCF). The BCF is defined as representing the ratio of a concentration of a chemical in tissue to its concentration in the surrounding water only when a steady-state concentration has been reached between the two media. Potential fish exposure via food sources is not included. Laboratory accumulation studies often determine BCFs due to the simplicity of the test and easier comparison with other BCF studies. Currently, U.S. EPA (2003a) recommends use of BCFs only for exposure to inorganic metals, presumably because intake of inorganic metals by fish via food sources is minor compared to uptake from water. However, a review of the literature by OEHHA suggests contaminated food sources can also be an important source of metal accumulation in fish tissues. Thus, reliance on BCFs to estimate fish exposure may also underestimate the actual accumulation of a metal in fish.

For semi- or non-volatile organic chemicals that are highly persistent and hydrophobic (generally with a $\log K_{ow} > 4$), the magnitude of bioaccumulation by fish via food sources can be substantially greater than the magnitude of bioaccumulation via exposure to water. For such chemicals, only true BAFs adequately assess accumulation of the chemical in fish tissues. For many of these persistent organic chemicals, biomagnification can occur. Biomagnification is the process through which chemical concentrations in fish increase as the chemical moves up the food chain, essentially through food sources. This process occurs because there are fewer organisms feeding off of more organisms at each level in the food chain, thus concentrating the chemical contaminants.

Numerous variables can affect uptake of persistent organic chemicals and inorganic metals in fish, therefore literature sources that reflected potential chemical accumulation as might occur under the "Hot Spots" program were our primary focus. That is, BCF/BAFs were primarily based on the edible portion (i.e., muscle tissue) of freshwater sport fish common to California lentic environments. Lentic environments consist mainly of standing water bodies including lakes, reservoirs and ponds. Sport fish that are caught and consumed in California are predominantly in trophic levels 3 and 4. These fish are typically of highest economic value and include predatory and carnivorous fish that feed on lower trophic level animals. BAF values for trophic level 2 organisms (e.g., zooplankton and larval fish stages) and non-sport fish, such as mosquito fish and the fathead minnow, were not considered unless there was a lack of accumulation data for higher trophic level sport fish.

The muscle tissue is defined here as the edible tissue of fish, although some ethnic groups may also eat various organs of fish. OEHHA's California fish advisories recommend against eating the liver and other organs of fish, because they may have higher concentrations of organic contaminants than the muscle tissue (OEHHA, 2003). In addition, most inorganic metals will also concentrate in the organs, particularly the kidney and liver. Thus, the BAFs derived in this document cannot be used for estimating accumulation of chemicals in organs other than muscle tissue, as doing so could seriously underestimate the dose received by consuming fish organs and tissues other than muscle.

In California, common freshwater sport fish caught for consumption include various species of trout, catfish, bass, perch, sunfish and carp (CDFG, 2007). Mean muscle lipid content and trophic level data for some sport-fish are shown in Table I-2. In general, the size of the sport fish should be representative of the size being consumed by the target human population. Thus, the mean values are based on fish sizes that are caught and consumed by anglers. As Table I-2 shows, both muscle lipid content and trophic level can increase with increasing length (and age) of the fish. In some instances, lipid content or trophic level based on fish length, in cm, is provided.

Table I-2. Percent Muscle Lipid Content and/or Mean Trophic Level for some Freshwater Sport-Fish Found in California

Common Name	Mean % Muscle Lipid	Mean Trophic Level
<u>Carp (<i>Cyprinus carpio</i>)</u>	<u>4.45</u>	<u>3 (10-23 cm)^a</u> <u>2.4 (>23 cm)</u>
<u>Catfish</u>		
<u>Black bullhead</u>	<u>1.12</u>	<u>3</u>
<u>Brown bullhead</u>	<u>2.79</u>	<u>3</u>
<u>Channel catfish</u>	<u>5.00</u>	<u>3.1 (5-30 cm)</u> <u>2.8-4 (36-54 cm)</u>
<u>White catfish</u>	<u>2.15</u>	
<u>Yellow catfish</u>	<u>0.75</u>	
<u>Blue catfish</u>		<u>3</u>
<u>Flathead catfish</u>		<u>3.8</u>
<u>Perch</u>		
<u>Yellow perch</u>	<u>0.66</u>	<u>3.4</u>
<u>Trout</u>		
<u>Rainbow trout</u>	<u>4.00</u>	<u>3 (<30 cm)</u> <u>3.6 (30-50 cm)</u> <u>4 (>50 cm)</u>
<u>Brook trout</u>	<u>1.51</u>	<u>3.2</u>
<u>Brown trout</u>	<u>3.81</u>	
<u>Cutthroat trout</u>	<u>1.23</u>	<u>3 (<40 cm)</u> <u>3.2 (>40 cm)</u>
<u>Lake trout</u>	<u>10.90</u>	<u>3.7 (20-30 cm)</u> <u>3.9 (30-40 cm)</u> <u>4.2 (>40 cm)</u>
<u>Bass</u>		
<u>Smallmouth</u>	<u>1.1</u>	
<u>Largemouth</u>	<u>1.03 (35-48 cm)</u> <u>3.1 (54 cm)</u>	
<u>Black crappie</u>	<u>0.57 (14-23 cm)</u>	

Sources: U.S. EPA (1998); OEHHA (1999); SFBRWQCB (2005); Morrison et al. (1997)

^a Length of fish shown in parentheses

1.1.1 Uptake and Accumulation of Semi- or Non-Volatile Organic Chemicals in Fish Tissues

Much of the field data for BAFs of organic chemicals comes from studies in the Great Lakes region (Eisenreich et al., 1981). The large surface area of the lakes, long hydraulic residence times, and major pollution sources near and upwind of the lakes have a significant impact on airborne deposited trace organic inputs.

For lipophilic, bioaccumulative organic chemicals, U.S. EPA (1998) recommends calculating a BAF based on the concentration of freely dissolved chemical in the

ambient water and the lipid-normalized concentration in tissue. Regarding lipid normalization, the BAF of lipophilic organic chemicals is usually directly proportional to the percent lipid content in the tissue of interest (U.S. EPA, 1998). For example, a fish with four percent lipid content would accumulate twice the amount of a chemical as a fish with two percent lipid content, all else being equal. Normalizing BAFs or BCFs to lipid content allows comparison between different fish species on the basis of factors other than percent lipid content. The lipid-normalized concentration is expressed as:

$$C = C_t / f \quad \text{(Eq. I.2)}$$

where:

C_t = Concentration of chemical in wet tissue (either whole fish or specified tissue)

f = Fraction lipid content in the organism

The lipid fraction of the edible muscle tissue is generally estimated because this is where the lipophilic chemicals will reside. However, the lipid content of muscle tissue can vary considerably among freshwater sport fish species (see Table I-1) as well as among the same species of different sizes and in different habitats. For this document, the rainbow trout lipid muscle content (4%) is used as the basis for point estimate BAFs for lipophilic organic chemicals. The rainbow trout is a common freshwater sport fish species caught and consumed in California and represents a reasonable "average" lipid content value among California sport fish. However, muscle lipid content can increase well above 10% in some fish species (carp, lake trout, and certain catfish) as they reach maximum size and age. The BAFs determined in this document may underestimate chemical intake if proportionally high consumption rates of such fish occur.

The tendency of an organic compound to bioconcentrate has been shown to be related to its lipophilicity and inversely related to the chemical's water solubility. However, correlations between bioconcentration and physical properties are poor for very large molecules of high molecular weight and for chemicals metabolized by fish (Oliver and Niimi, 1985). Large molecules (about 300 to 500 MW) appear to be less efficiently transferred from water and food to fish tissues, but can have very long half lives in lentic/lotic environments (U.S. EPA, 2003a). Comparison of laboratory and field bioaccumulation studies in fish show that use of laboratory BCFs (kinetic and steady state studies), in which water was the only media for bioconcentration, would severely underestimate the field residue levels of large organic molecules in fish, particularly if they are poor substrates for metabolic enzymes. This is a clear indication that water is not the primary route of fish exposure for these chemicals; consumption of contaminated food is likely the major chemical source.

U.S. EPA (1998) derived some BAFs from field measured biota-sediment accumulation factors (BSAFs) for very hydrophobic, organic compounds such as PCDD/Fs. The BSAF is the ratio of the lipid-normalized concentration of a chemical in tissue to its organic carbon-normalized concentration in surface sediment. Water concentrations of highly hydrophobic compounds can be difficult to measure accurately for field-measured BAFs, so U.S. EPA (2003a) recommends the BSAF as the only field-based method that can be used to estimate the concentration of certain organic compounds in ambient

water. The California “Hot Spots” PCDD/F BAF point estimates discussed below in Section I.3.1.6 were derived from field-measured BSAF data by U.S. EPA (1998).

U.S. EPA (1998) recommends that for organic chemicals with a log K_{ow} greater than four, the concentrations of particulate organic carbon (POC) and dissolved organic carbon (DOC) in the ambient water should be either measured or reliably estimated. For these chemicals, the concentration of the chemical that is dissolved in ambient water excludes the portion sorbed onto particulate or dissolved organic carbon. The freely dissolved concentration is considered to represent the most bioavailable form of an organic chemical in water and, thus, is the form that best predicts bioaccumulation. The freely dissolved concentration is calculated as:

$$C^{fdw} = (f_{fd}) \times (C^{tw}) \quad \text{(Eq. I.3)}$$

Where:

C^{fdw} = freely dissolved concentration of the organic chemical in ambient water

f_{fd} = fraction of the total chemical in ambient water that is freely dissolved

C^{tw} = total concentration of the organic chemical in ambient water

If f_{fd} is not known, it may be calculated using the equation:

$$f_{fd} = \frac{1}{1 + POC \times K_{ow} + DOC \times 0.08 \times K_{ow}} \quad \text{(Eq. I.4)}$$

For the California BAFs, DOC and POC were sometimes based on U.S. EPA (2003a) national default estimates of 2.9 mg/L for DOC and 0.5 mg/L for POC. These values reflect the central tendency estimated for DOC and POC for lakes and reservoirs distributed throughout the United States.

Field-based estimates of the freely dissolved concentration of an organic chemical in water (C^{fdw}) are preferred in order to predict BAF point estimates. However, Eq. I.4 was used to estimate f_{fd} in a number of instances when sufficient data were lacking in studies used to estimate a BAF.

1.1.2 Uptake and Accumulation of Inorganic Metals in Fish Tissues

In aquatic systems the availability of a metal to fish depends on many physico-chemical as well as biological factors. As summarized by Dallinger et al. (1987), availability is influenced by the chemical speciation of the ionic forms. The chemistry of the water including factors such as pH, hardness, and the presence of organic compounds and suspended particles may change the activity of free metal ions and influence the speciation of heavy metals. Binding to, and release from the sediment also affects the availability of metals to fish. Among the biological factors affecting metal availability, species-specific differences like feeding behavior and habitat preferences play a dominant role. These basic features are modified by physiological factors, such as accumulation rates and the binding capacity in various fish species. The three ways by

which inorganic metals may enter fish include body surface, the gills, and the alimentary tract. However, fish seem to be able to homeostatically regulate some heavy metals that they are exposed to. Thus, BCFs and BAFs for metals will generally be smaller compared to BCFs and BAFs for persistent bioaccumulative organic chemicals.

In general, soluble metal fractions may accumulate preferentially via the gills, and particulate metal fractions via the alimentary tract (Dallinger et al., 1987). Unlike persistent, hydrophobic organic chemicals, bioconcentration and biotransference factors of metals tend to decrease with increasing trophic level up to fish, although the organometal methylmercury is an exception. However, even if biomagnification is not observed, or bioconcentration factors are small, the amount of metal transferred via food or water can be high enough to reach levels that are harmful to humans. This is because under chronic exposure of a water system, very high metal levels may occur in sediments, macrophytes and benthic animals in relation to the water levels. Thus, ingestion of sediment and sediment-dwelling invertebrates by bottom-dwelling fish species may be an important route of metal uptake by these fishes.

The wet weight muscle tissue concentrations of metals are used for determination of the BAF values. If the reference data are expressed only as a dry weight muscle tissue concentration, the tissue concentration was adjusted to a wet weight concentration using a factor of 0.24 (i.e., water content of fish muscle is roughly 75-76% by weight) if specific conversion data are not presented in the reference to calculate the adjustment.

An inverse relationship between metal accumulation and weight/size of the fish has been observed; metal in tissues decreases with increasing size or weight of fish (Liao et al., 2003). This effect has been attributed to growth dilution, increased metabolic rate in juvenile fish and increased ability to depurate the metals as the fish matures. As a result, metal uptake studies in fingerlings or juvenile fish may overestimate bioaccumulation of mature sport fish caught and consumed by anglers and were usually not used in this document to derive accumulation factors.

Another factor to take into account is exposure duration. Numerous accumulation studies summarized below have observed long exposure times, on the order of months, before steady-state levels of a metal are reached in fish tissues. Thus, short-term exposure studies may underestimate bioaccumulation of a metal in fish.

Based on the bioaccumulation literature for metals of interest in the "Hot Spots" program, some general statements can be made. Waterborne exposure to an inorganic metal will result in greatest metal accumulation in gill, kidney and liver. Metals in the diet will increase levels in the gut as well. Muscle tissue will have the lowest accumulation of the metals. Basing BAFs on whole body concentrations of a metal may overestimate metal intake, as the concentration of an inorganic metal can be quite high in the viscera (e.g., kidney and liver), with organ-specific BAFs of 1000 or greater. Where sufficient data were present, laboratory-measured BCFs were lower for a metal than those derived using data from field studies. BCF studies often did not account for intake via contaminated food, which in some studies summarized below was shown to be an important route of exposure for inorganic metals. Also, many of the laboratory

BCF studies likely did not attain steady-state concentrations because exposures were too short.

In almost all instances, acidic water bodies (generally with a pH of 6.5 or lower) will increase accumulation of the cationic metals and oxy-anionic chromium in fish organs and tissues compared to pH neutral (7.0 to 7.5) water bodies. The default BAFs in this document are primarily based on pH neutral lentic water bodies, as these are the most common in California. Consequently, the default BAFs may underestimate the actual accumulation of a metal in fish if the water body is acidic.

I.2 Derivation of Fish BAFs

I.2.1 Semi- or Non-Volatile Organic Chemicals

I.2.1.1 Diethylhexylphthalate (DEHP)

DEHP has been detected in marine and lake sediments, as well as in marine and freshwater sport fish (Stalling et al., 1973; McFall et al., 1985; Camanzo et al., 1987; Mackintosh et al., 2004). However, the source of the DEHP found in these marine and lake sediments is not likely to be solely from air emissions. The very high K_{ow} of 7.73 and model calculations suggest that DEHP could readily bioaccumulate in fish and that dietary uptake would be an important route of exposure (Staples et al., 1997; Gobas et al., 2003). However, bioaccumulation and biomagnification studies of DEHP in fish show roughly three orders of magnitude lower BCFs/BAFs than predicted based on the K_{ow} of DEHP. This finding is a result of trophic dilution and lack of biomagnification through the aquatic food web, primarily due to the metabolic transformation of DEHP in fish (Staples et al., 1997; Mackintosh et al., 2004). The term trophic dilution means that the BAF tends to decrease as the trophic level increases.

The only freshwater study from which a field-measured BAF was developed was based on a Dutch study investigating the occurrence of DEHP in the freshwater and fish throughout the Netherlands (Peijnenburg and Struijs, 2006). Twenty-five samples of bream and roach fish and 66 freshwater samples from 23 sites were collected throughout the country. Based on the geometric mean DEHP concentration of 1.8 $\mu\text{g}/\text{kg}$ wet fish and the dissolved freshwater DEHP concentration of 0.33 $\mu\text{g}/\text{L}$, a BAF of 5.5 is calculated (Table I.3). We corrected for the lipid fraction in the whole fish samples (median: 0.5% lipid), generating a lipid-normalized DEHP BAF of 1.1×10^3 . Finally, we also corrected for the muscle lipid content of rainbow trout (4%), which is approximately eight times greater than that of the bream and roach fish, generating a BAF of 44.

An assumption used for this BAF is that the influence of collecting fish and water samples at different times and from different locations on this BCF is not large. Another factor to consider is that the fish in the Dutch study were collected from both lentic and lotic water bodies. Lentic environments are characterized by still (not flowing) water, as in lakes and reservoirs. But the lotic environments are characterized by flowing water, as in streams and rivers.

Gobas et al. (2003) and Mackintosh et al. (2004) conducted a saltwater field study to assess the food-web bioaccumulation of a range of phthalate esters including DEHP. The calculated lipid-normalized BAF for the staghorn sculpin, a forage fish, and the dogfish, a predatory species, were 16,000 and 580, respectively (Table I.3). The larger dogfish (3 kg BW) has a smaller BAF than the sculpin (0.1 kg BW) due to gill elimination and fecal egestion rates dropping with increasing organism size and becoming negligible compared to growth rates.

Table I.3. BAF Values for DEHP in Fish

Fish Species	Total BAF^a	BAF(fd)^b	BAF(rt)^c
Staghorn Sculpin	ND ^d	16,000	640
Spiny Dogfish	ND	580	23
Bream & Roach	5.5	1091	44

^a Total concentration in whole fish divided by the total concentration of chemical in water

^b Freely dissolved, lipid-normalized concentration

^c BAF(rt) for sport-sized rainbow trout (rt) based on muscle lipid content of 4%

^d No data

Supporting studies from other laboratories report BCFs in small sport and non-sport fish. Whole-fish BCFs of 17 and 30 were estimated in separate studies in small rainbow trout (Mehrlé and Mayer, 1976; Tarr et al., 1990). Mayer (1976) estimated a BCF of 594 in fathead minnows, and Karara and Hayton (1984) estimated a BCF of 637 in sheepshead minnows. The estimated BCF values are based on the parent compound (i.e., they did not estimate a total BCF including DEHP and its metabolites) and did not include data that appeared to suffer from water solubility problems or lack of steady state attainment.

Basing the bioaccumulation of DEHP on BCF values does not take into account accumulation of DEHP from food or sediment sources, which may result in an underestimation of the BAF. In addition, basing a BAF on fingerlings or small fish may overestimate BAFs for sport-sized fish. Until field-based bioaccumulation studies for specific lentic water bodies are published for DEHP, we recommend that the BAF of 44, based on the Dutch freshwater field study, be used in the “Hot Spots” program as the default point estimate for DEHP accumulation in sport fish.

I.2.1.2 Hexachlorobenzene

HCB in the atmosphere is predicted to be predominantly in the vapor phase (see Appendix E). HCB concentrations in the vapor phase averaged 96.6% (range: 92-100%) of the total HCB concentration in air samples over Ontario, Canada (Lane et al., 1992). This finding would suggest that airborne deposition of HCB into water bodies would be small enough to disregard. However, due to the extreme persistence of HCB in air, water and soil, accumulation of HCB into water bodies by both dry and wet deposition can be significant (Eisenreich et al., 1981; Kelly et al., 1991). Field studies at Lake Superior, a relatively pristine water body in which organics deposit primarily from

atmospheric sources, report HCB in water, sediment and fish tissue samples (Eisenreich et al., 1981).

Niimi and Oliver (1989) determined the percent lipid content and HCB concentration in muscle tissue of four salmonid species (brown, lake, and rainbow trout and coho salmon) collected from Lake Ontario. Based on the published water concentration of HCB in Lake Ontario, the researchers calculate a total BAF of 101,333. The total BAF was lipid-normalized based on 4% muscle lipid content in the fish, and adjusted for the concentration of freely dissolved HCB in water, assuming a DOC content of 0.25 mg/L in Lake Ontario from Gobas (1993). The resulting BAF(fd) is 2.6×10^6 .

We did not adjust the BAF(fd) to the muscle lipid fraction of rainbow trout (0.04) used in the California "Hot Spots" program because it is the same as the fish investigated by Niimi and Oliver (1989). We calculated the freely dissolved HCB fraction in water (0.78) from Eq. H.4 using the national default DOC and POC content of lakes and reservoirs (U.S. EPA, 2003a). A final BAF point estimate of 81,120 ($2.6 \times 10^6 \times 0.04 \times 0.78$) is recommended for California fish.

U.S. EPA (1998) calculates a similar BAF(fd) of log 6.40 (2.5×10^6) using Lake Ontario whole fish HCB data from Oliver and Niimi (1988). This BAF(fd) is similar to that estimated by Niimi and Oliver (1989) using only the muscle HCB concentration (BAF(fd) = 2.6×10^6) of the fish presented. U.S. EPA (1998) also calculated a mean log BAF(fd) of 5.70 (5.0×10^5) derived from BSAF data for HCB. Pereria et al. (1988) and Burkhard et al. (1997) determined a similar log BAF(fd) in the range of 6.03 to 6.68 for bioaccumulation of HCB in small, mostly non-sport fish in estuarine environments.

I.2.1.3 Hexachlorocyclohexanes

Technical grade hexachlorocyclohexane (HCH) generally consists of five isomers, including α -, β -, γ -, δ -, and ϵ -HCH. α -HCH is the most common isomer in technical grade HCH, and γ -HCH, also known as lindane, is most often isolated and used for its insecticidal action. Consequently, most environmental fate and bioaccumulation studies have investigated the α - and γ -isomers.

Lindane is a relatively small MW compound with a short half-life in fish, so rapid equilibrium occurs between the chemical concentration in fish and the water (Oliver and Niimi, 1985). The short half-life is probably a result of its log $K_{ow} < 4$. The high chlorine content of HCHs prevents metabolism of the isomers by rainbow trout (Konwick et al., 2006). The half-life of lindane in sport-sized fish (11-13 days) is longer than in juvenile fish (about 4 days). However, Geyer et al. (1997) report that α -HCH has a longer half-life of 14.8 days in juvenile rainbow trout. In addition, they observed a positive correlation for fish lipid content and the BCF for lindane.

The major factor governing residue levels for HCHs appears to be the chemical concentration in the water (Oliver and Niimi, 1985). Thus, good agreement between field BAFs and laboratory BCFs in rainbow trout is achieved. For lindane, the whole-fish laboratory BCF was 1200 and the whole-fish field BAF in Lake Ontario fish was 1000.

For α -HCH, the whole-fish laboratory BCF was 1600 and the whole-fish BAF in Lake Ontario fish was 700.

In a subsequent comprehensive investigation at Lake Ontario, Oliver and Niimi (1988) report total BAFs for α -HCH and lindane of 5357 and 9333, respectively. The lipid-normalized whole fish BAFs shown in Table I.4 were based on a weighted average lipid content of 11% for the four fish species examined (i.e., brown, lake, and rainbow trout, coho salmon).

Normalizing the BAFs to represent the freely dissolved fraction in water based on the national default DOC and POC values for lakes and reservoirs had little effect on the freely dissolved fraction of the HCHs, as chemicals with $\log K_{ow} < 4$ (the lindane and α -HCH $\log K_{ow}$ s are 3.67 and 3.78, respectively) will not partition significantly to OC. Normalizing the muscle concentration of the HCHs based on the muscle lipid content of rainbow trout (4%) results in point estimate BAFs of 3394 for lindane, and 1948 for α -HCH.

Table I.4. BAF Values Based on Lake Ontario Salmonids

HCH Isomer	Total BAF ^a	BAF(fd) ^b	BAF(rt) ^c
Lindane (γ -HCH)	9333	84,845	3394
α -HCH	5357	48,700	1948

^a Total concentration in whole fish divided by the total concentration of chemical in water

^b Freely dissolved, lipid-normalized concentration based on 11% lipid content in whole fish

^c BAF point estimates based on muscle lipid content of 4% for sport-sized rainbow trout

Niimi and Oliver (1989) determined the percent lipid content and HCH concentrations in muscle tissue, rather than only whole fish (apparently from the same fish examined in their previous study). The HCH concentrations in muscle adjusted for an average muscle lipid content of 4% for rainbow trout are 5.7 and 1.4 $\mu\text{g}/\text{kg}$ for α -HCH and lindane, respectively. Using the water concentrations of 2.8 and 0.3 ng/L for α -HCH and lindane, respectively, from Oliver and Niimi (1988) provides BAFs of 2036 (α -HCH) and 4667 (lindane).

Because the muscle HCH concentration data in Niimi and Oliver (1989) was at or below the limit of detection for some fish, particularly for lindane, the California BAF point estimate is based on the Oliver and Niimi (1988) data presented in Table I.4. We recommend a BAF(rt) point estimate of 2671 for the "Hot Spots" program, which is the arithmetic average of the muscle tissue BAF(rt)s for the two major HCH isomers in Table I.4.

I.2.1.4 Polycyclic Aromatic Hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs) are compounds with two or more fused benzene rings and often contain alkyl side groups. In water and sediment, low molecular weight PAHs (i.e., containing two or three aromatic rings) are more easily degraded by microbes, whereas the high molecular weight PAHs (i.e., containing four or more aromatic rings), including benzo[a]pyrene (BaP), tend to persist (Meador et al., 1995).

Bioaccumulation of PAHs in fish has not been rigorously studied, in part because PAHs undergo liver metabolism in fish resulting in low to non-detectable concentrations of the parent PAHs in fish tissues (Meador et al., 1995). Bioaccumulation of PAHs tends to decline with increasing K_{ow} , probably due to low gut assimilation efficiency and increased metabolism. However, low molecular weight PAHs tend to be less persistent in fish than the high molecular weight PAHs, probably due to more ready diffusion in and out of lipid pools.

BaP has been shown to be extensively metabolized in fish. In small bluegill sunfish (4 to 12 g wet weight) exposed to ^{14}C -labelled BaP in water, only 5% of the radiolabel in whole fish samples at the end of 24 hr exposure was found to be the parent compound (McCarthy and Jimenez, 1985). In their risk assessment, Boyce and Garry (2003) estimated a whole fish BCF of 14 for BaP based on the average value reported from relevant laboratory bioaccumulation studies in the literature.

Using the assumption that a typical lipid fraction of whole fish is 0.05 (Staples et al., 1997), and a muscle/whole body lipid ratio of 0.20 for adult rainbow trout (Niimi and Oliver, 1983), we calculated the lipid-normalized muscle tissue BCF as 56 for BaP. Adequate data for the DOC and POC water concentrations were not supplied by the studies used to derive the BCF, so the influence of this factor on the BAF could not be accounted for in the final estimate.

Burkhard and Lukasewycz (2000) determined field-measured BAFs for several PAHs found in water, sediment and lake trout muscle lipid of Lake Superior. The total BAF and BAF(fd) in Table I.5 were calculated by the researchers for lake trout in Lake Superior. The BAF(rt) was calculated by OEHHA for PAHs in rainbow trout (4% muscle lipid content) using default DOC + POC content for U.S. lakes and reservoirs. The relative order of metabolism was obtained by dividing the BAF of the chemical by its corresponding K_{ow} . By increasing rate of metabolism in the fish, the relative order was pyrene, benz[a]anthracene, chrysene/triphenylene, fluoranthrene, and phenanthrene. Thus, metabolism of the parent PAH compound appears to primarily control accumulation in the muscle tissue.

Table I.5. BAF Values for Polycyclic Aromatic Hydrocarbons

PAH congener (# of rings) ^a	PEF ^b	Total BAF ^c	BAF(fd) _d	BAF(rt) ^e
Phenanthrene (3)	ND ^f	18	89	4
Fluoranthrene (4)	ND	331	1660	62
Pyrene (4)	ND	10,471	52,481	2067
Benz[a]anthracene (4)	0.1	9550	53,703	1573
Chrysene/triphenylene (4)	0.01 (chrysene only)	759	4074	124 ^g

^a Number of benzene rings per PAH compound shown in parentheses

^b Potency Equivalency Factor for carcinogenicity, using benzo[a]pyrene as the index PAH compound with a PEF=1.

^c Total concentration in fillet of lake trout divided by the total concentration of chemical in water

^d Freely dissolved, lipid-normalized concentration based on 20.5% lipid content in fish fillet samples

^e BAF point estimates based on muscle lipid content of 4% for rainbow trout and default DOC + POC content for U.S. lakes and reservoirs from U.S. EPA (2003a).

^f Not determined, as a result of inadequate or no evidence for carcinogenicity in animals.

^g Assumed to represent BAF(rt) for both chrysene and triphenylene

The data in Table I.5 suggest that PAHs with four rings are more likely to accumulate in fish than PAHs with three rings. A study by Zabik et al. (1996) found some five- and six-ring PAHs in muscle fat of lake trout from Lake Superior. This study did not detect BaP in the fish tissue, but did find dibenzo[ah]pyrene which has a potency equivalency factor (PEF) value of 10. BAFs could not be calculated for any PAHs with five or more rings, either because dissolved levels of these congeners could not be detected in the water, or because the congener could not be detected in the fish (Baker and Eisenreich, 1989; 1990; Zabik et al., 1996). Another reason is that the individual PAHs quantified in water and fish were not all the same between various studies.

We calculated an average BAF(rt) of 849 from the congener groups in Table I.5 that have PEFs (i.e., benz[a]anthracene and chrysene), and is recommended as the default point estimate of BAF(rt) for PAHs. Considering that measurable levels of high molecular weight carcinogenic PAHs have been detected in fish muscle (although not enough data are present to estimate BAFs), but that a BAF for BaP is likely below the BAF(rt) of 849, a point estimate based on the most bioaccumulative carcinogenic PAHs should be sufficiently health protective to avoid underestimation of a BAF for the carcinogenic PAHs.

I.2.1.5 Polychlorinated biphenyls (PCBs)

PCBs are a group (209 congeners) of organic chemicals, based on various substitutions of chlorine atoms on a basic biphenyl molecule. However, probably less than 100 congeners are found at concentrations of significance in commercial PCB mixtures and environmental samples, and fewer represent a toxicological concern (Niimi, 1996). Solubilities and octanol-water partition coefficients (K_{ow}) for PCB congeners range over several orders of magnitude. The K_{ow} s, which are often used as estimators of the potential for bioconcentration, are highest for the most chlorinated PCB congeners.

Since log K_{ow} values of most PCB congeners are higher than 5, biomagnifications through trophic transfer is the primary mechanism governing the accumulation of these compounds in fish (Oliver and Niimi, 1985; van der Oost et al., 2003). Thomann and Connolly (1984) demonstrated that more than 99% of PCBs in Lake Michigan lake trout came from food. A food web bioaccumulation PCB study by Morrison et al. (1997) noted that over 99% of PCB 153 accumulated in fish through consumption of contaminated food and 79.9% of PCB 42 accumulation was through food (PCB 42 has a lower K_{ow}).

Food-web relationships and biomagnification may be more related to the PCBs in sediment rather than water. Therefore, biota sediment accumulation factors (BSAF) have been developed for PCBs as an indicator of bioavailability to fish because sediment is an important source for hydrophobic chemicals such as PCBs (Niimi, 1996). However, the PCBs found in the highest concentrations in fish generally reflected their high concentrations in water and sediment (Oliver and Niimi, 1988).

In the comprehensive field study by Oliver and Niimi (1988), the most common classes of PCB isomers in various salmon and trout species from Lake Ontario were the penta- and hexachlorobiphenyls, making up about 65% of the total isomeric composition. The tetra- and heptachlorobiphenyls made up another 30% of the isomeric composition. Eleven single and co-eluting PCB congeners (153, 101, 84, 138, 110, 118, 180, 87 + 97, 149, 187 + 182, and 105) constituted over half the PCBs in fish. The single most common congener was 153 (2,2', 4,4',5,5'-hexachlorobiphenyl). The tri, tetra, and penta congeners comprised a much higher fraction in water than in the fish. Thus, the PCB accumulation pattern in fish is not an accurate reflection of the aqueous composition of the mixture found in the lake.

Because the calculated total BAFs for the most common PCBs accumulating in fish gave a roughly 10-fold range for the values, a weighted average total BAF was calculated for the four most common chlorinated classes of PCB congeners in fish from the study by Oliver and Niimi (1988). These were the tetra-, penta-, hexa-, and hepta-CBs, which constituted about 95% of the overall PCBs accumulated in whole fish. The resulting weighted-average total BAF was 6.12×10^6 .

We calculated a lipid-normalized BAF of 5.56×10^7 based on the whole fish lipid content of 11% determined in the study by Oliver and Niimi (1988). The mean percent contribution of PCB congeners was similar for whole fish and muscle among the

species even though total concentrations vary widely (Niimi and Oliver, 1989). Consistency among congener contribution in whole fish and muscle was also demonstrated by cumulative percent of the more common PCB congeners. The freely dissolved PCB portion in water is based on data by Gobas (1993) who found about half of total PCBs in Lake Ontario water was in the freely dissolved form. The resulting calculated lipid-normalized, freely dissolved BAF, or BAF(fd), is 1.11×10^8 .

Next, we adjusted the BAF(fd) to generate a BAF point estimate to be used in the California "Hot Spots" program. Correcting the BAF(fd) for the muscle lipid fraction of 0.04 in rainbow trout, and correcting for the freely dissolved PCB fraction in water (0.25, or 50% of that calculated for Lake Ontario) gives a final BAF point estimate of 2.22×10^6 ($1.11 \times 10^8 \times 0.04 \times 0.50$).

I.2.1.6 Polychlorinated Dibenzo-p-Dioxins and Dibenzofurans (PCDDs and PCDFs)

PCDDs and PCDFs are two groups of toxic compounds composed of 135 and 75 individual isomers, respectively. Most studies have focused on the 17 congeners with lateral Cl substitutions at the 2,3,7,8 positions (Niimi, 1996). These congeners appear to be primarily responsible for the accumulation and toxicity of PCDD/Fs. The 2,3,7,8-TCDD, 1,2,3,7,8-PCDD, 2,3,7,8-PCDF and 2,3,4,7,8-PCDF congeners were common in four fish species (brown trout, lake trout, rainbow trout, coho salmon) examined from Lake Ontario. Dietary uptake of PCDD/Fs appears to be of more importance than waterborne uptake, although dietary absorption efficiencies in fish are consistently lower and more variable compared to PCBs.

The two main lateral substituted PCDDs, 2,3,7,8-TCDD and 1,2,3,7,8-PCDD, constituted about 89% of the sum of all PCDDs in the fish (Niimi, 1996). The two main PCDFs, 2,3,7,8-PCDF and 2,3,4,7,8-PCDF, constituted 51% of the sum of all PCDFs in the fish. Since these congeners are the most bioaccumulative and have the greatest toxicity concern, the PCDD/F BAFs will be representative of these four congeners.

U.S. EPA (1998) derived lipid-normalized, freely dissolved BAFs (i.e., BAF(fd)) from field measured BSAFs. The high hydrophobic nature of PCDD/Fs makes it difficult to accurately determine field-measured BAFs (i.e., based on water concentrations) for this group of chemicals. U.S. EPA (2003a) recommends the BSAF as the only field-based method that can be used to estimate the concentration of these compounds in ambient water. Using a weighted-average approach for the main congeners found in fish, the BAF(fd)s were 1.00×10^7 and 5.50×10^6 for PCDDs and PCDFs, respectively.

We then adjusted the BAF(fd)s to generate BAF point estimates to reflect the muscle lipid fraction of rainbow trout (0.04) for the "Hot Spots" program. The final BAF point estimates of 400,000 and 220,000 were calculated for PCDDs and PCDFs, respectively, for California fish. The average BAF of these two values, 310,000, is the recommended BAF point estimate for the "Hot Spots" program.

1.2.2 Derivation of Fish BCFs – Inorganic Metal and Semi-Metal Chemicals

1.2.2.1 Arsenic

Inorganic arsenic (As), either as As(III) or As(V), are the predominant forms in aquatic ecosystems such as sediment and water, but organoarsenic compounds may be present at significant levels in freshwater fish. Average concentrations of As in ambient freshwater are generally <1 to 10 µg/L (U.S. EPA, 2003b). U.S. EPA (2003b) states that recent research shows each of the major inorganic and organic As species, including As(III), As (V), arsenobetaine (AsB), dimethylarsenic acid (DMA), and monomethylarsonic acid (MMA), may exhibit different toxicities, and it may be important to take into account the fraction of total As present in the inorganic and organic forms when estimating the potential risk posed through consumption of As-contaminated fish. Ideally, the most appropriate BAFs would incorporate the most bioavailable and toxic form(s). This is currently not possible, so the point estimate BAF in this document will be based on total As in sport fish muscle tissue.

Direct accumulation of As in tilapia was proportional to the concentration of arsenicals in water (Suhendrayatna et al., 2002). Approximately 25% of absorbed arsenic from water in whole fish as either As(III) or As(V) was transformed to methylated arsenic, primarily methyl-, dimethyl-, and trimethyl- forms. Whether absorbed as As(III) or As(V) from water, metabolism in fish resulted in roughly equivalent concentrations of both inorganic arsenic species in whole fish, although As(III) was absorbed more easily than As(V).

Accumulation and transformation of As in the food chain has been investigated. In a three-step freshwater food chain (algae-shrimp-tilapia), exposure to As(III) in water resulted in total As concentrations decreasing in the organisms with each step up the food chain (Suhendrayatna et al., 2002). Inorganic As species were the predominant forms in each organism (As(III), 9-41%; As(V), 50-90%), with only a limited degree of As methylation at each step in the food chain. However, when As(V) was the dominant As species in water, mouthbreeder fish raised long-term in aquaculture ponds contained predominantly organoarsenic species in muscle tissue, with inorganic As equaling only 7.4% of total As (Huang et al., 2003).

Predicted and measured As concentrations in major organs of tilapia from culture ponds high in As observed highest As concentrations in the alimentary canal, blood and liver, and lowest concentrations in muscle tissue (Liao et al., 2005). Steady-state concentration of As in muscle tissue took up to 300 days to be achieved.

Arsenic bioaccumulation studies in fish have been conducted in laboratory, aquaculture pond, and field investigations, although exposure durations to achieve steady-state concentrations in fish tissues were only observed for the aquaculture and field studies. The BAFs findings are presented in Table I.6.

In aquaculture studies, an average BCF of 8.2 (range: 5.4 to 11) was determined for bioconcentration of As in muscle of mouthbreeder fish raised long-term in ponds from three different regions in Taiwan (Huang et al., 2003). The fish were collected from

ponds containing 14.4 to 75.8 µg/L As in water. A BCF of 3.5 was recorded for As in muscle tissue of large-scale mullet raised in a Taiwanese aquaculture pond (Lin et al., 2001). In farmed tilapia fish exposed to As in water for 300 days, a muscle BCF = 4 was calculated (Liao et al., 2005). In a similar study, BCFs of 15 and 53 were obtained for As from tilapia muscle raised in two aquaculture ponds containing 49.0 and 17.8 µg/L As in water, respectively (Liao et al., 2003). Because the fish in these aquaculture studies were fed with artificial bait that did not contain As, the accumulation factors may better represent BCF values rather than BAF values.

Only two field studies were located that presented data to determine a muscle tissue BAF for fish in As-contaminated lentic water bodies. A BAF of 28 was determined from muscle tissue of the common carp exposed to As in four wastewater treatment basins in Pennsylvania (Skinner, 1985). Channel catfish and large-mouth bass from a reservoir impacted by mining and agricultural runoff had muscle BAF values of 12.5 for As (Baker and King, 1994).

Table I.6. BAFs for Arsenic in Muscle Tissue of Fish from Lentic Water Bodies

Location	Species	Arsenic Water Concentration	Arsenic Muscle Concentration	BAF	Reference
Taiwanese Aquaculture Studies					
Putai Pond	mouthbreeder	75.8 µg/L	0.41 µg/g	5.4	Huang et. al., 2003
Yichu Pond	mouthbreeder	15.1	0.12	7.9	Huang et. al., 2003
Hsuehchia Pond	mouthbreeder	14.4	0.16	11.1	Huang et. al., 2003
Putai Pond	large-scale mullet	169.7	2.41	14.2	Lin et. al., 2001
Hsuehchia Pond	tilapia	17.8	0.95	53.4	Liao et. al., 2003
Yichu Pond	tilapia	49.0	0.75	15.3	Liao et. al., 2003
Tilapia farms	tilapia	94	1.5	16	Liao et al., 2005
Field Studies					
San Carlos Reservoir, AZ	large-mouth bass	8	0.1	12.5	Baker & King, 1994
San Carlos Reservoir, AZ	channel catfish	8	0.1	12.5	Baker & King, 1994
Wastewater treatment basins, PA	common carp	3.0 – 16.0	0.22 - <0.05	28	Skinner, 1985

Among the studies presented in Table I.6, average BCF/BAFs were calculated for six fish species: 8.1 for mouthbreeder, 14.2 for large-scale mullet, 28 for tilapia, 12.5 for

large-mouth bass and channel catfish, and 28 for common carp. The arithmetic average BAF combined for all species is 17, which we recommend as the BAF point estimate for As.

I.2.2.2 Beryllium

Little information could be found for bioaccumulation of beryllium in fish. U.S. EPA (1980) estimated a BCF of 19 in whole bluegill after 28 days of exposure in water. It is unknown if steady state levels were attained in the fish, although the whole-body elimination half-life was observed to be one day. Limited data by Eisler (1974) suggest that whole-fish accumulation of inorganic beryllium in mummichogs from seawater is similar to some other cationic metals such as cadmium, in that whole fish uptake of beryllium appears to be a passive process.

No information could be found regarding the accumulation of beryllium in muscle tissue of fish. Based on BCF and BAF studies of other cationic metals discussed in this appendix, steady state levels were probably not reached in bluegills during the 28-day exposure (U.S. EPA, 1980). The muscle BAFs for other cationic metals (i.e., cadmium, inorganic mercury, lead, nickel) presented in Table H.2 range from 20 to 80. We recommend that a mean cationic metal BAF of 40 be used for beryllium in sport fish until more comprehensive bioaccumulation studies are conducted.

I.2.2.3 Cadmium

A considerable number of cadmium (Cd) bioaccumulation studies have been carried out in fish. Freshwater sport fish accumulate Cd mainly in gills, kidney, liver, and gastrointestinal tract (Sangalang and Freeman, 1979; Harrison and Klaverkamp, 1989; Spry and Wiener, 1991; Szebedinszky et al., 2001). However, Cd does not accumulate as appreciably in muscle tissue of exposed sport fish and the concentration is generally low relative to other tissues and organs.

The Cd concentration in fish varies with the proportion of free divalent Cd in water, typically increasing with increasing water concentration (Camusso et al., 1995). Direct uptake across the gills has been generally considered the primary influx of the metal for fish in dilute waters (Spry and Wiener, 1991). However, absorption of Cd from contaminated food sources can be a significant route of exposure, and may be the dominant source of Cd in bodies of water with high pH and calcium levels (Ferard et al., 1983; Harrison and Klaverkamp, 1989; Farag et al., 1994; Kraal et al., 1995; Thomann et al., 1997).

The main characteristics of lakes that enhance bioaccumulation of Cd in fish include low pH ($\text{pH} \leq 6$), low aqueous calcium (often $< 2 \text{ mg/L}$), and low DOC (usually $< 3 \text{ mg/L}$) (Spry and Wiener, 1991). In the eastern U.S., whole-body Cd levels in bluegill fish from low pH lakes were as much as 10-fold higher compared to cadmium in bluegills from circumneutral-pH lakes. In addition, accumulation of Cd in fish is more sensitive to changes in water hardness, usually expressed in mg/L CaCO_3 , rather than changes in DOC (Wiener and Giesy, 1979).

Steady-state equilibrium of Cd in muscle and other tissues was obtained in brook trout at about 20 weeks exposure in a three-generation exposure study by Benoit et al. (1976). Benoit et al. (1976) also recorded a muscle BCF = 3.5 in brook trout exposed to aqueous Cd in Lake Superior water for 70 weeks. Equilibrium of Cd in tissues was also reached at 20 weeks of exposure.

Perhaps significantly, the numerous laboratory studies that measured muscle Cd content show an inverse relationship with water hardness. In several laboratory studies, BCFs varied between 1.6 to 4.8 for Cd in muscle of rainbow trout, carp and brook trout with a water hardness between 33 and 93 mg /L CaCO₃ (Benoit et al., 1976; Giles, 1988; Harrison and Klaverkamp, 1989; de Conto Cinier et al., 1997). Exposure durations for these studies ranged from 3 to 17 months, and tissue and organ Cd concentrations increased with increasing exposure duration. Two other laboratory studies that recorded somewhat higher BCFs of 17-19 in muscle of rainbow and brook trout also had the lowest water hardness (19-22 mg /L CaCO₃) (Sangalang and Freeman, 1979; Kumada et al., 1980). The exposure duration of fish to Cd-contaminated water for both of these studies was about 3 months. Alternatively, laboratory studies exposing rainbow trout to Cd in water with considerably higher hardness (140-320 mg/L CaCO₃) at circumneutral-to-high pH (7.4-8.2) for up to 80 weeks recorded BCFs from 0 to 2 in muscle tissue (Roberts et al., 1979; Calamari et al., 1982; Brown et al., 1994; Szebedinszky et al., 2001).

The level of DOC in the water of the laboratory BCF studies above were not discussed, but were likely low. Low DOC levels would allow water hardness to be the main factor affecting bioaccumulation of Cd.

Although comparatively few field studies have been published that investigated Cd accumulation in muscle tissue of sport fish, the field study by Wiener and Giesy (1979) supports the assumption that water hardness (and perhaps pH) is a more important factor in controlling tissue accumulation than the DOC content. In this study, a Cd muscle BAF = 12 was determined in bluegill stocked in an acidic (pH = 4.6), highly organic pond for 511 days. Measured total organic carbon of the pond was anywhere from 15 to >30 mg/L, but the CaCO₃ content of the pond was very low, averaging 2.1 mg/L.

Two field studies examined the effect of acidified water in New York lakes on fish tissue levels of various heavy metals as a result of acid deposition (i.e., acid rain) (Heit et al., 1989; Stripp et al., 1990). In general, higher BAFs were recorded for Cd in muscle tissue of yellow perch and white sucker from the most acidic lentic water body, Darts Lake, compared to two other lakes, Rondaxe and Moss lakes, with higher pH values (Table I.7). All three lakes were clear-water lakes with comparable concentrations of DOC.

Table I.7. BAFs for Cadmium in Muscle Tissue of Fish from U.S. Lakes

Location	Species	Lake pH	Cd Water Concentration (µg/L)	Cd Muscle Concentration (µg/g)	BAF
Darts Lake (1)	White sucker	4.9-5.4	0.7	0.062	89
Darts Lake (1)	Yellow perch	4.9-5.4	0.7	0.048	69
Darts Lake (2)	White sucker	5.1-5.4	0.26	0.038	146
Darts Lake (2)	Yellow perch	5.1-5.4	0.26	0.028	108
Rondaxe Lake (1)	White sucker	5.8-6.7	1.1	0.024	22
Rondaxe Lake (1)	Yellow perch	5.8-6.7	1.1	0.024	22
Rondaxe Lake (2)	White sucker	5.8-6.7	0.61	0.025	41
Rondaxe Lake (2)	Yellow perch	5.8-6.7	0.61	0.038	62
Moss Lake (1)	White sucker	6.5-6.8	0.6	0.022	36
Moss Lake (1)	Yellow perch	6.5-6.8	0.6	0.034	56
Skinface Pond, SC (3)	Bluegill	4.6	0.17	0.0021	12

Sources: (1) Stripp et al., (1990); (2) Heit et al., (1989); (3) Wiener and Giesy (1979).

The few field studies examining muscle tissue levels of Cd in contaminated lakes indicate that basing a BAF on laboratory BCF studies would underestimate the accumulation potential of Cd in fish. However, it is probably not appropriate basing a BAF on data from highly acidified lakes (i.e., Darts Lake and Skinface Pond), as California generally does not have the lake acidification problem that exists in the northeastern U.S. Thus, we recommend default BAF point estimate for Cd of 40 based on fish from the variable pH (Rondaxe Lake) and circumneutral lakes (Moss Lake), which is the arithmetic average BAF combining both fish species (white sucker and yellow perch, which represent trophic level 3 and 4 fish, respectively) from these lakes.

I.2.2.4 Chromium

Hexavalent chromium (Cr(VI)) in water readily penetrates the gill membrane of fish and is the main route of uptake (Holdway, 1988). Organs and tissues that accumulate Cr(VI) include gills, spleen, kidney, gall bladder, gastrointestinal tract, opercular bone, and brain. Accumulation in muscle tissue is minor compared to these other tissues. No biomagnifications occur at higher trophic levels. Cr(VI) uptake is a passive process with resulting tissue concentrations directly proportional to exposure concentrations. Chromium bioavailability to fish increases with decreasing pH (7.8 to 6.5), resulting in increased bioaccumulation in tissues and organs (Van der Putte et al., 1981).

In a laboratory study, six-month exposure of rainbow trout to Cr(VI) as potassium dichromate (K₂Cr₂O₇) in water resulted in a muscle tissue BCF of 3 (Calamari et al., 1982).

A small freshwater aquatic ecosystem containing adult catfish was created in a small tank, and a single dose of potassium dichromate was added to the system (Ramoliya et al., 2007). After 21 days of exposure, a muscle tissue BCF <1 was calculated for the

catfish based on the average water concentration of Cr(VI) over the 21 days. However, the Cr(VI) content in the catfish had not reached equilibrium at the end of exposure, and was still increasing with increasing exposure duration. High levels of Cr(VI) in the intestine of the catfish suggest Cr(VI) may be absorbed via food sources.

Rainbow trout that were reared for two years in either a hatchery or river water that was contaminated with low levels of sodium dichromate had muscle tissue BCFs of 40 and 12, respectively (Buhler et al., 1977). Exposing the same fish to high concentrations of Cr(VI) (2.5 mg/L) for 22 days increased muscle levels of Cr(VI), but the resulting BCF was only 0.1-0.2.

Two field studies from South Africa determined the bioaccumulation of chromium in muscle tissue of fish. In adult African sharptooth catfish, muscle tissue BAFs of 10 and 16 were calculated for fish kept in a treated sewage maturation pond and in a reservoir, respectively, for 12 months (Van den Heever and Frey, 1996). Nussey et al. (2000) calculated an average muscle tissue BAF of 23.6 in the moggel, a cyprinid fish, collected from a different reservoir over a period of 15 months.

Based on the long-term field exposure studies, an average muscle BAF of 26 was calculated for rainbow trout in the Buhler et al. study, and an average muscle BAF of 13 was calculated for the African sharptooth catfish in the van den Heever and Frey study. Combined with the muscle tissue BAF of 23.6 in the moggel from Nussey et al. (2000), we calculate an arithmetic mean BAF of 21 and recommend this value as the BAF point estimate for Cr.

1.2.2.5 Lead

Similar to Cd, factors that may increase accumulation of cationic metals such as lead in fish include low pH (6.0-6.5 or less) in the water body, low concentrations of aqueous calcium that compete with lead for absorption through the gills, and low DOC (Varanasi and Gmur, 1978; Spry and Wiener, 1991; Lithner et al., 1995). Pb appears to have a greater tendency than Cd to associate with DOC and particulate matter in lake water, with accumulation in fish varying inversely with the concentration of dissolved organics in water (Wiener and Giesy, 1979). When Merlini and Pozzi (1977a) added a Pb salt to lake water, only 8% remained in the ionic form with the remainder presumably associating with dissolved organics.

Accumulation of Pb by fish typically increases with increasing exposure concentration in water, although Pb does not biomagnify in aquatic food chains (Spry and Wiener, 1991). Pb chiefly accumulates in the bone, scales, gill, kidney, and liver. Pb does not accumulate as appreciably in skeletal muscle tissue of fish. Primary mode of absorption has been suggested to be direct uptake of Pb in the ionic state across the gills, with lead from food sources being minor or insignificant (Merlini and Pozzi, 1977a; Spry and Wiener, 1991; Farag et al., 1994). On the other hand, another laboratory study found that lead uptake in fish via food was significant, if not more important than uptake via water (Vighi, 1981).

In a three-generation laboratory study, a BCF of 2 to 3 was estimated for Pb in muscle tissue of first and second generation brook trout (Holcombe et al., 1976). Exposure to Pb in water was for 38 and 70 weeks in first and second generation fish, respectively. The concentration of Pb in muscle had reached equilibrium at about 20 weeks of exposure.

Whole bluegill Pb concentrations have been shown to be as much as 10 times higher in bluegills from low-pH lakes ($\text{pH} \leq 6.0$) compared to bluegills from circumneutral-pH lakes ($\text{pH} 6.7-7.5$) (Spry and Wiener, 1991). In another study, whole-fish Pb levels in sunfish increased almost three-fold when lake water pH was decreased from 7.5 to 6.0 (Merlini and Pozzi, 1977b).

In other field studies, Pb accumulated to greater extent in muscle of white suckers and yellow perch from an acidic lake compared to more neutral lakes (Heit et al., 1989; Stripp et al., 1990) (Table I.8). With increasing lake acidity, muscle bioaccumulation of Pb became increasingly higher in bottom-dwelling, omnivorous white suckers compared to carnivorous yellow perch. Thus, contact with sediments by bottom-dwelling fish increases Pb bioaccumulation.

A considerably greater concentration of Pb was found in surface sediments (880-1005 $\mu\text{g/g}$) of the lakes compared to the water (2.0-3.0 ng/g) (Stripp et al., 1990). It was postulated that higher levels in fish tissues from acidic lakes result from increased mobilization of the cationic Pb species from sediments coupled with an increase in the cationic Pb species in the acidic water.

The field data indicate higher muscle BAFs in fish from highly acidified lakes (Table I.8). California generally does not have the acidification problem that exists in the northeastern U.S. Thus, a BAF point estimate for Pb was based on fish from the variable pH and circumneutral lakes. The BAF data from Nussey et al. (2000) was also included, although water pH data were not provided in the report. We calculate an arithmetic average BAF of 19 combining all fish species (white sucker, yellow perch and moggel) from these lakes and recommend this value as the Pb BAF point estimate.

Table I.8. BAFs for Lead in Muscle Tissue of Fish from Lentic Ecosystems

Location	Species	Lake pH	Pb Water Concentration	Pb Muscle Concentration	BAF
Acidic water bodies					
Darts Lake (1)	White sucker	4.9-5.4	3.0 µg/L	0.13 µg/g	43
Darts Lake (1)	Yellow perch	4.9-5.4	3.0 µg/L	0.058	19
Darts Lake (2)	White sucker	4.9-5.4	1.5	0.13	87
Darts Lake (2)	Yellow perch	4.9-5.4	1.5	0.055	37
Acidic lakes & ponds, NJ (3)	Yellow perch	3.7-4.6	0.8 – 3.6	0.067 – 0.11	40
Variable and circumneutral water bodies					
Rondaxe Lake (1)	White sucker	5.8-6.7	2.0	0.048	24
Rondaxe Lake (1)	Yellow perch	5.8-6.7	2.0	0.058	29
Rondaxe Lake (2)	White sucker	5.8-6.7	2.3	0.050	22
Rondaxe Lake (2)	Yellow perch	5.8-6.7	2.3	0.050	22
Moss Lake (1)	White sucker	6.5-6.8	2.5	0.031	12
Moss Lake (1)	Yellow perch	6.5-6.8	2.5	0.024	10
Witbank Dam, South Africa (4)	Moggel	ND*	140	2.00	14

Sources: (1) Stripp et al. (1990), (2) Heit et al. (1989), (3) Sprenger et. al. (1988), (4) Nussey et al. (2000)

* No data

I.2.2.6 Mercury (inorganic) and Methylmercury

Mercury, like other metals deposited into water, can occur in a number of physical and chemical forms. Physically, mercury can be freely dissolved or bound to organic matter or particles suspended in water. Mercury can be found as elemental mercury (Hg⁰), inorganic ionic mercury (primarily Hg⁺⁺), or organic mercury (e.g., methylmercury (MeHg) or dimethylmercury).

Mercury (Hg) enters aquatic ecosystems primarily as inorganic Hg, but MeHg is the dominant form of Hg found in muscle tissue of freshwater fish (Spry and Wiener, 1991). MeHg has been shown to constitute virtually all, about 99% or greater, of the total Hg in muscle of trophic level 3-4 freshwater sport fish even though much of the Hg analyzed in the water was in inorganic Hg (Bloom, 1992; Kuwabara et al., 2007). In whole fish, the proportion of inorganic Hg is greater (5% or more of total Hg) because whole body samples include visceral tissue, such as kidney and liver, which is the principal site of inorganic Hg accumulation in fish (Hill et al., 1996; Watras et al., 1998). BAFs discussed for MeHg in this document are for informational purposes only and are not specific to the Hot Spots program. Mercury compounds emitted by facilities are almost exclusively in the elemental or inorganic form, so MeHg is not directly applicable to the Hot Spots program.

As summarized by Southworth et al. (2004), MeHg is produced in aquatic environments by the action of microorganisms on inorganic Hg. It can also be removed from the

aquatic systems by microorganisms that demethylate MeHg. Once formed, MeHg is taken up by microorganisms, primary producers, aquatic invertebrates, and fish. MeHg in the organisms shows the classical biomagnification process, with MeHg concentration increasing with trophic level. The concentrations of MeHg that are accumulated in fish are greatly affected by the nature of the aquatic food chain, and are sensitive to factors such as aquatic community composition and productivity. In many waters, minute concentrations (<10 ng/L) of waterborne inorganic Hg are capable of sustaining MeHg production at rates high enough to support bioaccumulation of MeHg in fish to levels warranting fish consumption advisories. The concentrations of MeHg and inorganic Hg are positively related in natural waters, which would appear to support expressing a BAF for MeHg in fish as a ratio based on total or dissolved inorganic Hg in water. Calculating MeHg bioaccumulation in fish using such a ratio (i.e., estimate the concentration of dissolved MeHg in water based in the total Hg concentration deposited in water), introduces another level of uncertainty compared to development of BAFs directly from published reports.

Using the dissolved MeHg fraction in water to derive BAFs is recommended, as this is the primary form of MeHg that is bioaccumulated in fish. MeHg is also more toxic than other forms of mercury. However, dissolved MeHg was not always the form measured in the studies U.S. EPA (2001) identified for inclusion in their database. Thus, translators were necessary to convert between other forms of Hg measured in water and dissolved MeHg for BAF calculations. For lentic systems (i.e., lakes, reservoirs and ponds), the translators that may be used in the Hot Spots program include dissolved MeHg (MeHg_d) over the total Hg (Hg_t) and the MeHg_d over the total MeHg (MeHg_t). The lentic U.S. EPA translators are $\text{MeHg}_d / \text{Hg}_t = 0.032$ and $\text{MeHg}_d / \text{MeHg}_t = 0.61$.

U.S. EPA (2001) derived the mean dissolved MeHg/total Hg translator of 3.2% for lentic ecosystems, and used it to convert between other forms of Hg measured in water and dissolved MeHg for BAF calculations. Thus it can be interpreted that 3.2% of inorganic Hg that has deposited into a lake will be converted by microorganisms and found in the form of dissolved MeHg.

Table I.9 presents various BAFs for methylmercury from U.S. EPA (2001) and California data (OEHHA, 2006). Although U.S. EPA presents the geometric means of BAFs, OEHHA recommends the use of arithmetic means of the BAFs to provide a more health protective estimate. In developing their BAFs, U.S. EPA assumed that 100 percent of the mercury measured as total mercury in both trophic levels 3 and 4 was MeHg. This assumption provides a more health protective estimate.

Table I.9. Methylmercury BAFs for Lentic/Lotic^a Ecosystems from U.S. EPA and California Data

Agency	Environment/Comments	Mean	Trophic Level	
			3	4
U.S. EPA	Lentic Only	Geometric	1.1×10^6	5.7×10^6
U.S. EPA	Lentic Only	Arithmetic	1.5×10^6	6.2×10^6
California	Lentic Alternative	Geometric	NP	NP
California	Lentic Alternative	Arithmetic	NP	NP
U.S. EPA	Lotic Only	Geometric	5.7×10^5	1.2×10^6
U.S. EPA	Lotic Only	Arithmetic	1.3×10^6	3.9×10^6
California	Lotic Alternative	Geometric	6.8×10^5	1.1×10^6
California	Lotic Alternative	Arithmetic	1.4×10^6	3.5×10^6
U.S. EPA	Lentic/Lotic Combined	Arithmetic	1.4×10^6	5.0×10^6

^a Lentic environments are characterized by still (not flowing) water, as in lakes and reservoirs. Lotic environments are characterized by flowing water, as in streams and rivers.

In California, using a MeHg BAF developed by U.S. EPA is complicated by the large number of Hg point sources originating from legacy mining activities, a situation somewhat unique to California. Atmospheric deposition of Hg into water bodies may be overshadowed by the existing Hg already present due to legacy mining. In addition, very little published data exist for California lentic ecosystems in order to determine if total Hg concentrations are good predictors of MeHg concentration. The BAFs and translators developed by U.S. EPA were based primarily on atmospheric deposition of Hg into water bodies. Hg speciation in water and fish may be quite different depending on whether the Hg originated from mining or atmospheric deposition.

Nevertheless, OEHHA (2006) found that the national values predicted California fish MeHg concentrations very well except for some water bodies where Hg concentrations in water were statistically higher. Hg concentrations (≥ 0.2 ng/L) in these water bodies were found to be more than one standard derivation from the mean for other data used in these tests. We concluded that the national default values for BAFs and translators may not work well for all water bodies in California. However, based on the limited comparisons possible, BAFs and translators based on the California data and international studies (U.S. EPA database) were found to be similar. Thus, a MeHg BAF = 6,200,000 (log 6.79) from Table I.9 for sport fish caught and consumed from lentic ecosystems, and a translator of 3.2% to convert total Hg deposited in water to dissolved MeHg in water may be relevant MeHg variates to use in California.

In partial support, Kelly et al. (1995) observed that total Hg concentration was not a good predictor of MeHg concentration in stream water or in lakes in general, but it appeared to be a good predictor for lakes within individual geographic areas. In lotic ecosystems, Southworth et al. (2004) concluded that it is not valid to assume that the fraction of total waterborne Hg comprised by MeHg would remain constant while total Hg varies at high total Hg concentrations (roughly >50 ng/L) typical of systems affected by point-source or legacy contamination. However, at total Hg concentrations less than 10 ng/L, the %MeHg varies little. They postulated that such a relationship results from

saturation of the ecosystems capacity to methylate inorganic Hg at high total Hg concentrations.

Inorganic Hg is absorbed by fish less efficiently than MeHg from both food and water, but if absorbed, is eliminated more rapidly. For example, rainbow trout fed inorganic Hg-contaminated prey resulted in Hg predominantly accumulating in the intestines, and the Hg was not significantly absorbed into the body (Boudou and Ribeyre, 1985). During the decontamination phase, Hg that had accumulated in the intestines was rapidly excreted.

In water, the most important route for uptake of inorganic Hg in fish is likely the gills, with accumulation of Hg mainly in the gills, kidney and liver (Allen et al., 1988; Gottofrey and Tjalve, 1991). Whole-body accumulation of inorganic Hg in rainbow trout and carp increases with decreasing water pH from 9 to 5, but did not reach equilibrium during a 17-day exposure in water (Wakabayashi et al., 1987).

MeHg is the primary concern for estimating Hg bioaccumulation. Since relatively little of the Hg in fish muscle is in the inorganic form, there are very little field data to estimate a BAF for inorganic Hg.

In a laboratory tank study investigating the relationship between inorganic Hg body burden levels and toxicity, a mean muscle BCF of 84 was calculated in rainbow trout exposed to HgCl in water for 60 to 130 days (Niimi and Kissoon, 1994). Steady-state levels in muscle tissue were reached by 60 days of exposure to high levels of HgCl (64 µg/L); these levels were eventually lethal to the fish. Since most lakes of concern contain inorganic Hg levels in the ng/L to low µg/L range, such high exposure conditions may not reflect an ideal situation for estimating an inorganic Hg BAF. In addition, it has been found that food sources containing inorganic Hg are also important for fish Hg bioaccumulation (Hill et al., 1996).

U.S. EPA (2001) has used a national criteria of 51 ng/L of total Hg in water as a measure that may result in the MeHg concentration of concern of 0.3 µg/g in fish. Using the assumption that, at most, 1% of the MeHg concentration in fish muscle is actually inorganic Hg, a BAF of 59 for inorganic Hg is calculated ($0.3 \mu\text{g/g} (0.01) \div 51 \text{ ng/L}$). Although this BAF derivation is a rather crude estimate of the inorganic Hg BAF, the value is near that calculated from the BCF study (BCF = 84) by Niimi and Kissoon (1994). OEHHA recommends using the inorganic Hg BAF point estimate = 84 (rounded to 8×10^1) derived from the Niimi and Kissoon study.

I.2.2.7 Nickel

In aquarium tank studies, brown trout exposed to water containing radioactive nickel (^{63}Ni) showed the greatest accumulation of the metal in the gills, kidneys and liver, with relatively low accumulation in muscle tissue (Tjalve et al., 1988). The Ni concentration in muscle was related to the water concentration of Ni (Van Hoof and Nauwelaers, 1984). Similar to other cationic metals, increasing the acidity of water increases accumulation of Ni in fish.

A muscle BCF of 1.5 was recorded in the brown trout following 3 week exposure to Ni in a water tank. However, equilibrium of Ni between water and fish tissues had not been attained. Rainbow trout exposed to Ni in hard water (hardness = 320 mg CaCO_3/L) for six months accumulated little or no Ni in muscle tissue (BCF = 0.8-1.1) (Calamari et al., 1982).

In a field study, Nussey et al. (2000) calculated an average muscle tissue BAF of 19 in the moggel, a cyprinid fish, collected from a reservoir containing various heavy metals, including Ni, over a period of 15 months. Average muscle BAFs of 4 and 39 were calculated in common carp collected from two different wastewater treatment basins in Pennsylvania (Skinner, 1985). The acidity of the treatment basin water was not discussed, so it is unknown if water acidity played a role for the variation in BAF values.

In laboratory studies, accumulation of Ni in fish muscle tissues is relatively low compared to other inorganic metals discussed in this document. There are also relatively few published reports investigating fish bioaccumulation of Ni. Based on the BAFs from the two field studies by Nussey et al. (2000) and Skinner (1985), we calculated an arithmetic mean average BAF of 21 and recommend this value as a point estimate BAF for Ni.

I.2.2.8 Selenium

Selenium (Se) occurs in the environment in several oxidation states with different physicochemical and biological properties (Besser et al., 1993). Se from both natural and anthropogenic sources enters surface waters primarily as the highly soluble Se(IV) and Se(VI) oxidation states, which form selenite, SeO_3^{2-} , and selenate, SeO_4^{2-} , respectively. Organic selenides, Se(-II), including Se-amino acids and Se-proteins, methyl selenides, and other Se-substituted analogs of organosulfur compounds, are produced by biological reduction of selenite and usually occur at lower concentrations in water than inorganic Se species. Little information is available for organic selenides, so the BAF is based on total Se.

Se is an essential micronutrient for most aquatic organisms but is also toxic at relatively low environmental concentrations. It is reported that Se concentrations in fish muscle rarely exceed 1 ppm (wet weight) in the absence of exposure to Se from geologic sources or from industrial wastes (Cumbie and Van Horn, 1979).

Four-month exposure of juvenile bluegill and largemouth bass to selenite (Na_2SeO_3) in water resulted in BCF values of 288 and 153, respectively, and was independent of

water temperature and hardness (Lemly, 1982). Accumulation of Se in muscle was relatively slow, reaching a steady-state concentration after 90 days of exposure in both fish species. Accumulation of Se in fish skeletal muscle was presumed to be a result of the high affinity of Se for sulfhydryl groups found on many organic molecules in muscle tissue. However, bioconcentration in muscle was quite low compared to BCF values for other organs and tissues. Lemly (1982) observed higher bioconcentration of Se in the spleen, heart, liver, kidney, gill, and erythrocytes.

In a food-chain study (algae-daphnids-bluegill), whole bluegill fry accumulated greater Se concentrations from food than from water in selenite-based exposures, and aqueous and food-chain Se bioaccumulation were approximately additive (Besser et al., 1993). However, in both aqueous and food-chain exposures based on selenite and selenate, Se bioaccumulation was greatest in algae and least in bluegills. Se concentrations in whole bluegill fry did not differ significantly between selenite and selenate treatments in either aqueous or food-chain exposures. Inorganic Se BCF values ranged from 13 to 106 in whole blue gill fry with 30- to 40-day exposures, although a steady-state concentration was not attained.

In a field study, Cumbie and van Horn (1979) analyzed muscle Se levels in various species of fish, primarily bluegill, other sunfish, carp and bullhead, during spring and summer from a reservoir with a high Se concentration. The range of muscle BAFs among all fish was 632 to 5450 with an arithmetic average of about 1780. Further research at the same reservoir observed muscle BAFs in warmwater sportfish (primarily various species of perch, catfish, sunfish and crappie) ranging from 739 to 2019 with an arithmetic average of 1351 (Lemly, 1985). There was evidence of biomagnification of Se through the food-chain, although when considering only muscle tissue of fish, levels of Se appeared to be similar to that of molluscs, insects, annelids and crustaceans found at the reservoir.

Lower Se BAFs of 124 and 216 were calculated in muscle of white suckers and yellow perch, respectively, from an acidic lake in New York (Stripp et al., 1990). Based upon geochemistry, Se would be expected to be less soluble in acidic lakes. BAFs of 454 and 490 were determined for Se in muscle tissue of crappie and carp, respectively, collected from a wastewater treatment basin in Pennsylvania (Skinner, 1985).

The accumulation data indicate Se uptake from both food and water results in accumulation of Se in muscle tissue, and that BAF/BCF values can be quite variable even between different fish species within the same water body. The two related field studies investigating Se accumulation in fish from a North Carolina reservoir (Cumbie and Van Horn, 1979; Lemly, 1985) gave an average BAF of 1566 ($1351 + 1780 / 2$) combining all trophic level 3 and 4 fish. Not including the data from the acidic lake, we calculate an arithmetic mean BAF of 1019 when the average BAF from the North Carolina reservoir is combined with the average fish BAF from the Pennsylvania wastewater treatment basin from Skinner (1985). In support, the BAF is within the predicted intervals (at water Se concentrations above 0.5 $\mu\text{g/L}$) of the Se whole fish bioaccumulation model for lentic systems developed by Brix et al. (2005). We recommend a default point estimate BAF of 1000 for selenium for use in the Hot Spots program.

I.3 Non-Bioaccumulated Chemicals

Some organic “Hot Spot” chemicals in which a significant airborne fraction can be found in the particle phase do not appear to be bioaccumulated in fish. For example, although data show that methylenedianiline (MDA) exists partly in the particle phase and is persistent in soils, the low log Kow of 1.59 (HSDB, 2008) and rapid metabolism in higher trophic level animals (ATSDR, 1998) indicate this chemical will likely not bioaccumulate in fish tissues. In addition, unpublished evidence summarized in ATSDR (1998) suggests that MDA does not bioaccumulate in carp. Until published evidence shows otherwise, a fish BAF for MDA will not be included in the fish pathway in the “Hot Spots” program.

In addition, OEHHA is proposing that fluoride should not be included in the fish pathway because fresh weight fluoride concentrations in muscle or the fillet portion of fish were found to be less than the water concentration, regardless of the weight of the fish (Gikunju, 1992; Mwaniki and Gikunju, 1995).

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