APPENDIX A

SECTION I. DRAFT NATIONAL METHYLMERCURY BIOACCUMULATION FACTORS

This appendix is a brief summary of the initial effort conducted to determine the feasibility of deriving draft National bioaccumulation factors for methylmercury. This appendix is based on the draft bioaccumulation report. The complete version of the original draft bioaccumulation factor report, with more in-depth discussions of the methodology, a list of the references cited, rationales for using data, and an uncertainty discussion can be obtained from the Water Docket W-00-20.

This appendix does not reflect comments or changes suggested by the peer reviewers. No changes were made to the draft report that served as the basis for this appendix. Data interpretations, findings, or conclusions discussed in this appendix are preliminary and may be changed in the future.

Introduction

The methylmercury bioaccumulation factors (BAFs) were estimated using guidance presented in the *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (U.S. EPA, 2000a; hereafter "the 2000 Human Health Methodology") and supplemented with methods presented in the Mercury Study Report to Congress (MSRC; U.S. EPA, 1997c). The generalized equation for estimating a BAF is as follows:

$$BAF = \frac{C_t}{C_w} \qquad \textbf{Equation-1}$$

where:

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

C_w = Concentration of chemical in water

Literature searches were conducted to obtain data on bioaccumulation, concentrations of different forms of mercury in water, percent methylmercury in tissue, and mercury predator-prey data. The data sources primarily included articles from peer reviewed journals published between 1990 and April of 1999 and publicly available reports (e.g., State, Federal, or trade/industry group reports; dissertations;

proceedings from professional meetings). Data from a variety of aquatic ecosystems (i.e., lakes, rivers, estuaries) and on lower trophic levels was specifically looked for since the MSRC focused only on lakes (primarily northern oligotrophic lakes) and trophic levels 3 and 4 fish.

BAFs are used in the ambient water quality criteria (AWQC) equation to estimate human mercury exposure from consumption of contaminated fish. Equation 2 is the generalized AWQC equation for a noncarcinogen and shows where the BAF fits into the calculation.

$$AWQC = RfD \times RSC\left[\frac{BW}{DI + \sum_{i}^{x}(FI \times BAF_i)}\right] \qquad \textbf{Equation 2}$$

Where:

RfD = reference dose for noncancer human health effects

RSC = relative source contribution to account for non-water sources of exposure

BW = human body weight

DI = drinking water intake

FI = fish intake

BAF_i = bioaccumulation factor for chemical "i".

The methylmercury BAFs that would be used in the above equation are presented in the accompanying table A-9, and are calculated as the geometric mean BAF of all BAFs calculated for a given trophic level.

Attachment A at the end of this appendix also contains the general comments made by the external peer reviewers on the draft national methylmercury BAFs.

Methods for Estimating Bioaccumulation Factors

Three approaches were used to derive draft BAFs that could be used to derive draft national methylmercury BAFs. These are direct, indirect, and conversion (modified direct) approaches. Each of

these approaches has its own limitations, biases, and uncertainties associate with it. These approaches and the BAFs derived using them are summarized below.

EPA's BAF derivation guidance is based on a data hierarchical preference approach. Under the hierarchy, the preferred method for deriving a BAF for an organometallic compound such as methylmercury is to use field-measured data to directly calculate a BAF (i.e., the direct method). BAFs estimated using this direct approach are calculated using the simple ratio of the chemical concentration in tissue and water. When such field data do not exist, or if the available field data are considered unreliable, the next preferred method in the hierarchy estimates a BAF by multiplying a bioconcentration factor (BCF) by a food chain multiplier (FCM) (i.e., the indirect method). The FCM is a factor used to account for food chain interactions and biomagnification. EPA has used this indirect method to estimate BAFs to support the development of wildlife criteria values in the Great Lakes Water Quality Initiative or GLWQI (EPA, 1993) and in the MSRC (EPA, 1997). With few exceptions, field-derived FCMs were calculated using concentrations of methylmercury in predator and prey species using the following equations:

$$FCM_{TL2} = BMF_{TL2} \qquad \text{Equation-3}$$

$$FCM_{TL3} = (BMF_{TL3})\,(BMF_{TL2}) \qquad \text{Equation-4}$$

$$FCM_{TL4} = (BMF_{TL4})\,(BMF_{TL3})\,(BMF_{TL2}) \qquad \text{Equation-5}$$

where:

FCM = Food chain multiplier for designated trophic level (TL2, TL3, or TL4)
BMF = Biomagnification factor for designated trophic level (TL2, TL3, or TL4)

The basic difference between FCMs and BMFs is that FCMs relate back to trophic level one, whereas BMFs always relate back to the next lowest trophic level. Biomagnification factors are calculated from methylmercury tissue residue concentrations determined in biota at a site according to the following equations:

$$BMF_{TL2} = C_t, TL2) / (C_t, TL1) \qquad \text{Equation-6}$$

$$BMF_{TL3} = (C_t, TL3) / (C_t, TL2) \quad \text{Equation-7}$$

$$BMF_{TL4} = (C_t, TL4) / (C_t, TL3) \quad \text{Equation-8}$$

where:

C_t = concentration of chemical in tissue of appropriate biota that occupy the specified trophic level (TL2, TL3, or TL4).

With the indirect BAF approach, it is important that when either selecting predator prey field data from the literature or when conducting a site-specific field study to obtain such data, that the feeding relationships between predator and prey are based on functional feeding relationships. It should be verified that a given predator is feeding on a given prey item at the location in question so that the BMFs and FCMs reflect actual trophic transfer of the chemical as close as possible. Usually, it is not enough to simply know that organisms are from two different trophic levels. Unfortunately, for the analyses presented here, much of the available data obtained from the published literature were insufficient to document functional feeding relationships. Thus, BAFs derived using the indirect approach were not used in determining the draft national methylmercury BAFs, but are presented only for comparison purposes.

In the MSRC, in cases where the direct empirical BAF derivation method could be used, but the available data was for a form of mercury other than dissolved methylmercury, a modified direct approach was also used. The modified direct approach was used when either the water data or organism tissue data was not in the methylmercury form (e.g., total mercury, dissolved total mercury, total methylmercury) but could be converted to methylmercury using translating factors. Data for mercury in water was converted to dissolved methylmercury by using chemical translators (see Section II of this Appendix). Mercury in tissue reported as total mercury was converted to methylmercury by multiplying by a factor that estimates the fraction of total mercury present in the methylated form (i.e., fmmf translator). The fmmfs were developed from field studies where both total mercury and methylmercury were measured in biota tissue.

Using the methods outlined above, BAFs were estimated initially by trophic level for lakes (lentic aquatic systems), rivers and streams (lotic aquatic systems), and estuaries. An ecosystem-based approach to deriving the BAFs was used because differences in general bioaccumulation trends would be expected among the aquatic ecosystems due to inherent differences in methylation processes, food web dynamics,

mercury loadings, and watershed interactions, among other factors. However, due to the lack of data in terms of both quality and quantity, no clear differences in bioaccumulation trends were observed between lentic and lotic ecosystems based on the available data (see Figure A-3). Based on qualitative and semi-quantitative comparisons of the data, no significant difference was found between the lentic and lotic BAFs. Thus, they were combined for each trophic level to obtain the trophic level-specific draft national BAFs. A near complete lack of adequate data prohibited derivation of draft national BAFs for estuarine systems.

Summary of BAFs for Methylmercury in Lentic Ecosystems

Table A-1 compares the BAFs estimated using the two primary approaches (direct and indirect) methods for estimating BAFs for trophic levels 2, 3, and 4 species. Although the BAFs based on the indirect approach are not used in the national draft BAF calculations because they are not based on verifiable functional predator-prey feeding relation ships, they are nonetheless useful for comparing and assessing general tends in bioaccumulation. Other than the BAF_2, the BAFs are within a factor of two of one another. Both the direct and indirectly estimated BAFs show an expected increase in methylmercury bioaccumulation with increasing trophic position. This suggests that if functional predator-prey feeding relationships can be developed, that indirect BAFs could provide reasonably good approximations of methylmercury bioaccumulation in organisms in the field.

Table A-1: Summary of Bioaccumulation Factors for Methylmercury Mercury in Lentic Ecosystems

Parameter	Methylmercury (1)	
	Direct ($L \cdot kg^{-1}$)	Indirect ($L \cdot kg^{-1}$)
BCF	5.9×10^4	NA
BAF_2	8.6×10^4	3.1×10^5
BAF_3	1.3×10^6	2.2×10^6
BAF_4	6.8×10^6	1.1×10^7

(1) All values are geometric means

Summary of BAFs for Methylmercury in Lotic Ecosystems

Table A-2 compares the lotic BAFs estimated using the direct and indirect methods. The BAFs based on the indirect approach are not used in the draft national BAF calculation because they are not based on verifiable functional predator-prey feeding relation ships; they are nonetheless useful for comparing and assessing general tends in bioaccumulation. As was the case with the lentic indirectly estimated BAFs, the indirect lotic BAFs are close approximations of the directly estimated BAFs (within a factor of 3 or less). Also, as was observed for lentic ecosystems, both the direct and indirectly estimated lotic BAFs show an expected increase in methylmercury bioaccumulation with increasing trophic position. This suggests that if functional predator-prey feeding relation ships can be developed,

Table A-2: Summary of Dissolved Methylmercury Bioaccumulation Factors for Lotic Ecosystems

Parameter	Methylmercury (1)	
	Direct ($L \cdot kg^{-1}$)	Indirect ($L \cdot kg^{-1}$)
BCF	1.2×10^4	NA
BAF_2	4.4×10^5	1.9×10^5
BAF_3	1.6×10^6	5.6×10^5
BAF_4	2.5×10^6	3.2×10^6

(1) values are geometric means

that indirect BAFs could provide reasonably good approximations of methylmercury bioaccumulation in organisms in the field.

Methylmercury BAFs Translated from Other Mercury Forms

Converted BAFs (that is, in terms of other mercury forms) were derived for dissolved methylmercury using translator factors (see Section II, Chemical Translators for Mercury and Methylmercury) and by using factors to convert total mercury measured in organism tissues to methylmercury in tissues.

Mercury Translators

For those studies that met the data quality objectives but did not analyze or report water mercury concentrations in the dissolved methylmercury form, the reported form of mercury was converted to the mean fraction of dissolved methylmercury (f_d $MeHg_d$) by using one or more of the "translators" listed in Table A-3. Section II below discusses the methodology and data used to derive the translators. Section II of this appendix also provides partition coefficients (K_D) that were not necessary for this analysis, but that can be used along with total suspended solids information to estimate the desired fraction of mercury in water.

Table A-3: Summary of Mercury Translators for Mercury in Water

f_d value	**Lentic**	**Lotic**
f_d Hg_d/Hg_t	0.600	0.370
f_d $MeHg_d/Hg_t$	0.032	0.014
f_d $MeHg_d/MeHg_t$	0.613	0.490

Conversion Factors for Mercury in Organism Tissue

Similar to the water data, if mercury in biota tissue (muscle or whole body) was reported as total mercury then the appropriate mean (arithmetic) estimate of the fraction present in the methylated form (fmmf) for the respective trophic level was used to convert it to methylmercury. Table A-4 summarizes the fmmfs used to estimate converted BAFs.

Table A-4: Summary of fmmfs for Lentic and Lotic Ecosystems

Trophic Level	**Lentic**	**Lotic**
1	0.18	0.05
2	0.44	0.49
3	1.00	1.00
4	1.00	1.00

Summary and Comparison of Converted BAFs and BCFs derived for Lentic and Lotic Ecosystems

Methylmercury translator factors (see Section II, *Chemical Translators for Mercury and Methylmercury*) were used to estimate dissolved methylmercury BCFs and BAFs in lotic and lentic ecosystems. Table A-5 summarizes the converted BAFs. The converted lentic BAFs range from approximately 2 to 37 times greater than the converted lotic BAFs.

Figures A-1 and A-2 compare the direct and converted estimates of BAFs and BCFs for lentic and lotic ecosystems, respectively. Although the data sets are relatively small, the ranges of converted BAFs are in agreement with BAFs directly estimated. Tables A-6 and A-7 summarize and compare the point estimates of each data set. In lentic ecosystems, the difference between the mean directly estimated BAFs and mean converted BAFs is generally less than a factor of two. For lotic ecosystems, the difference is slightly larger, ranging from a factor of two to a factor of seven, with an overall mean difference of four. This information suggests that the converted BAFs in each ecosystem are good estimates of directly measured BAFs for all trophic levels. However, because the set of BAFs estimated using the two different approaches are small for each ecosystem, insufficient data were available to perform any rigorous statistical evaluation to determine if a significant difference exists between the BAFs of each system. Nonetheless, graphically the data suggest that the direct and converted BAFs can be combined to derive overall BAFs for each trophic level in each ecosystem. The BAFs based on the combined data sets are presented in Table A-8.

Figure A-3 compares the combined data sets (e.g., directly-measured and converted BAFs and BCFs) for lentic and lotic ecosystems. While the lotic BAFs clearly span a greater range than the lentic BAFs, the differences between the mean lotic BAFs and the mean lentic BAFs for each trophic level are fairly small (differences range between 1 and 5). To investigate if there were significant differences between the BAFs for the two ecosystems significant, a student's T-test was performed on the combined data for each trophic level-specific BAF and BCF using the computer software WINKS (Texasoft, 1999). Although differences in mercury bioaccumulation between lentic and lotic ecosystems could be expected due to differences in mercury loading characteristics, bioavailability, food web dynamics, and methylation processes, among other factors, no significant statistical differences ($p>0.05$) were found between the lentic and lotic BAFs and BCFs. Furthermore, a closer inspection of the converted lentic BAF_4 data for several Minnesota Lakes (Glass et al., 1999) suggests that, given a larger sample size, the lower range of field-measured lentic BAF_4 values could be similar to the lower range of values observed for lotic ecosystems. Whether these observations are artifacts of the available data or trends due to real

Table A-5: Comparison of Converted Bioaccumulation Factors for Methylmercury in Lotic and Lentic Ecosystems

Parameter	Lentic ($L \cdot kg^{-1}$)	Lotic ($L \cdot kg^{-1}$)
BCF	4.3×10^4	6.1×10^3
$_{MD}BAF_2$	1.5×10^5	6.2×10^4
$_{MD}BAF_3$	1.3×10^6	3.5×10^4
$_{MD}BAF_4$	4.1×10^6	1.4×10^6

Table A-6: Comparison of Direct and Converted Methylmercury BAFs and BCFs for Lentic Ecosystems

	$_{MD}BCF$		$_{MD}BAF_2$		$_{MD}BAF_3$		$_{MD}BAF_4$	
Value[a]	direct	converted	direct	converted	direct	converted	direct	converted
5th	12,300	13,400	16,700	47,500	322,000	466,000	3,270,000	3,800,000
50th (GM)	58,700	43,000	85,600	150,000	1,260,000	1,330,000	6,800,000	4,080,000
95th	281,000	138,000	439,000	474,000	4,900,000	3,820,000	14,200,000	4,380,000
GSD	2.59	2.26	2.70	2.01	2.29	1.90	1.56	1.04

[a] GM = geometric mean; GSD = geometric standard deviation.

Table A-7: Comparison of Direct and Converted Methylmercury BAFs and BCFs for Lotic Ecosystems

	$_{MD}BCF$		$_{MD}BAF_2$		$_{MD}BAF_3$		$_{MD}BAF_4$	
Value[a]	direct	converted	direct	converted	direct	converted	direct	converted
5th	340	1,200	15,600	3,400	261,800	45,800	283,000	55,400
50th (GM)	5,400	6,000	179,000	61,900	1,640,000	346,000	2,520,000	1,380,000
95th	85,800	29,800	2,000,000	1,130,000	10,200,000	2,620,000	22,500,000	30,300,000
GSD	5.38	2.63	4.40	3.39	3.05	3.42	3.78	6.80

[a] GM = geometric mean; GSD = geometric standard deviation**.**

processes is not distinguishable. Because the range of available BAF values for lentic and lotic systems overlap one another, the individual BAFs for the two systems were combined in one data set to derive the trophic level-specific draft national methylmercury BAFs.

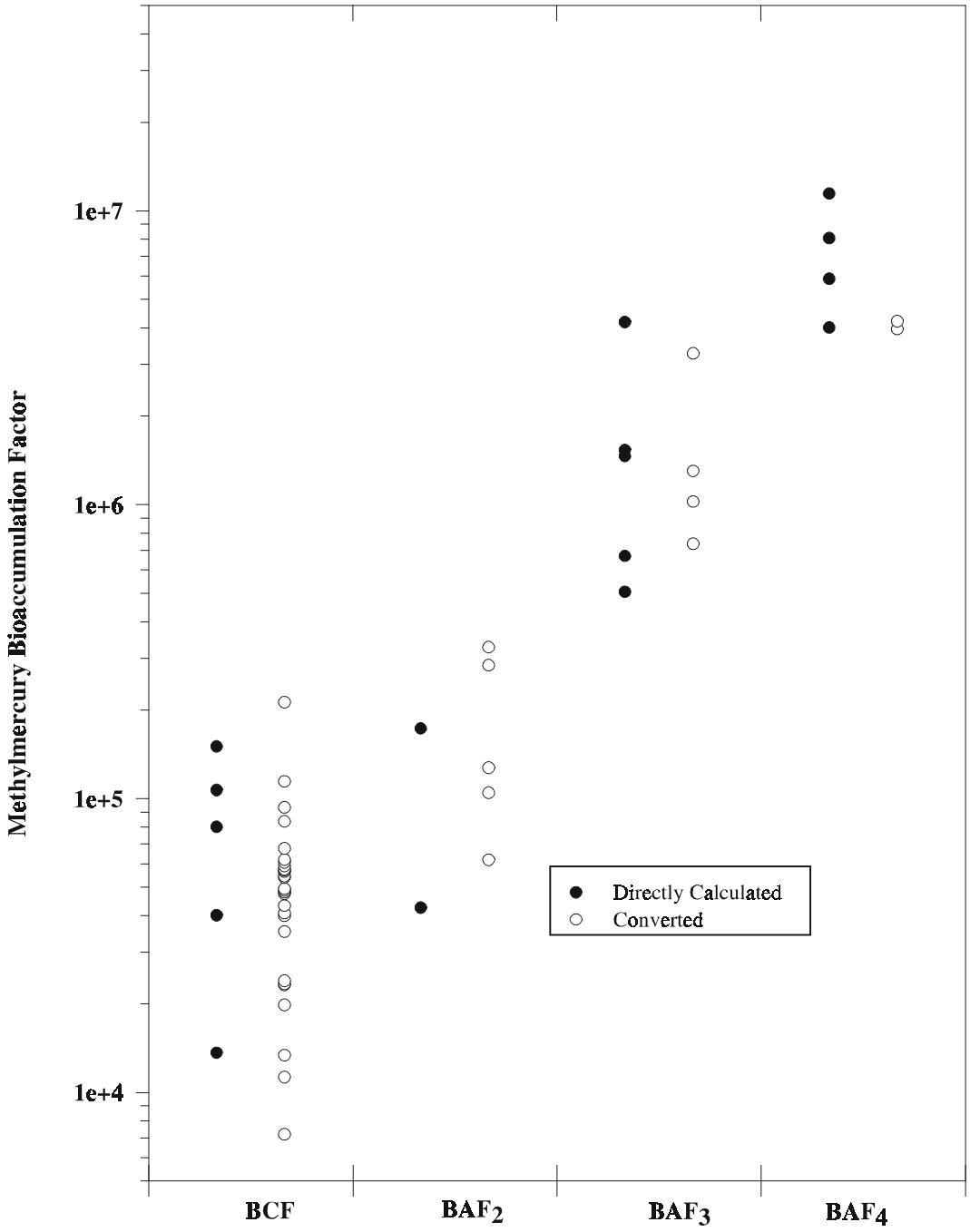

Figure A-1. Comparison of direct field-measured and converted field-measured methylmercury BCFs and BAFs for lentic ecosystems.

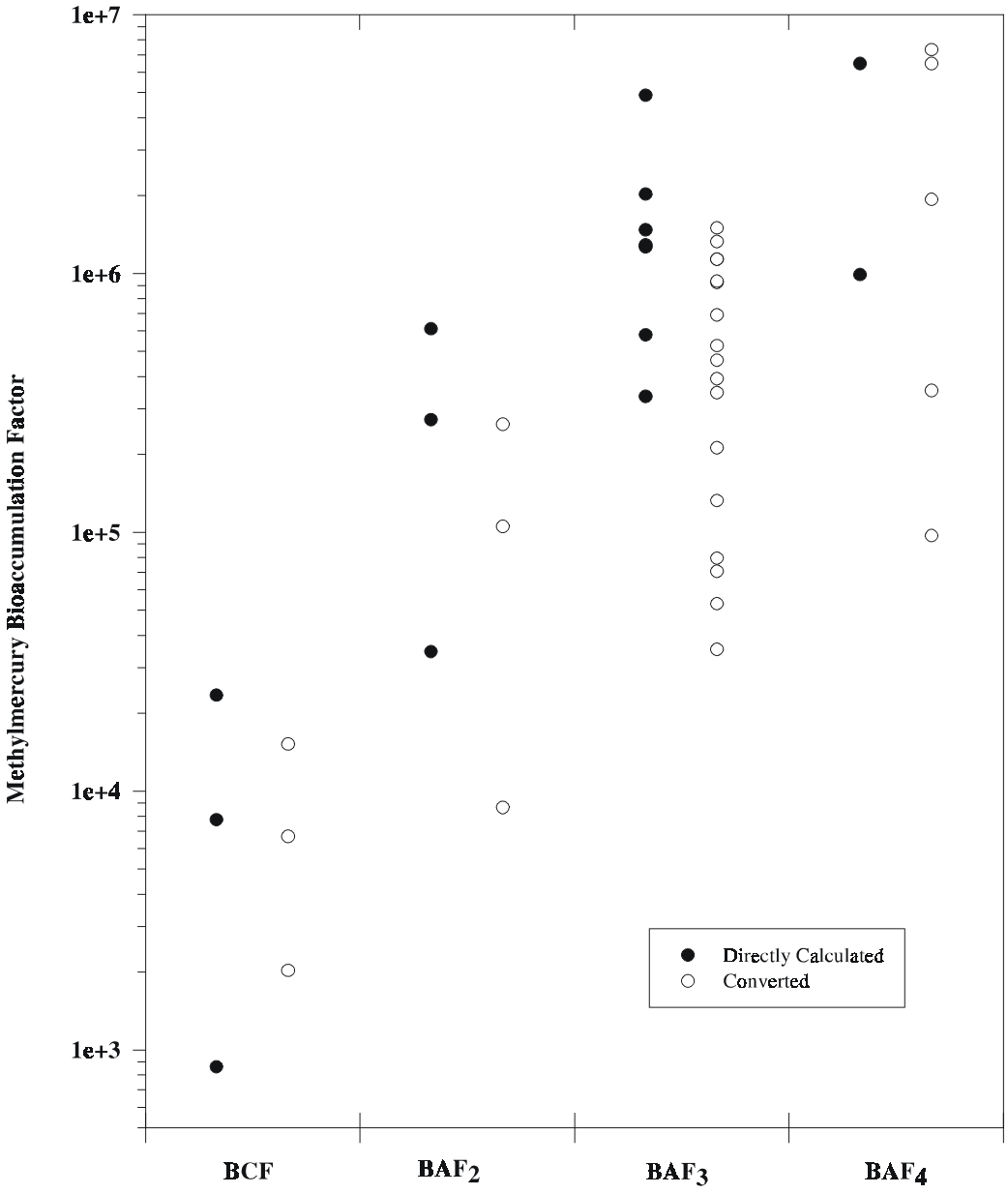

Figure A-2. Comparison of direct field-measured and converted field-measured methylmercury BCFs and BAFs for lotic ecosystems.

Table A-8: Summary of Lentic and Lotic Methylmercury BAFs and BCFs

	$_{MD}BCF$		$_{MD}BAF_2$		$_{MD}BAF_3$		$_{MD}BAF_4$	
Value[(1)a] (%)	Lentic	Lotic	Lentic	Lotic	Lentic	Lotic	Lentic	Lotic
5th	13,300	800	37,000	8,000	423,000	46,000	2,800,000	73,400
50th (GM)	45,000	5,700	127,800	105,000	1,115,000	517,000	5,740,000	1,240,000
95th	153,000	43,200	440,000	1,390,000	2,930,000	5,820,000	11,800,000	20,900,000
GSD	2.10	5.14	2.12	4.80	2.02	4.36	1.55	5.57

(1) Values are based on combined direct and converted BAFs and BCFs.

[a] GM = Geometric Mean; GSD = geometric standard deviation.

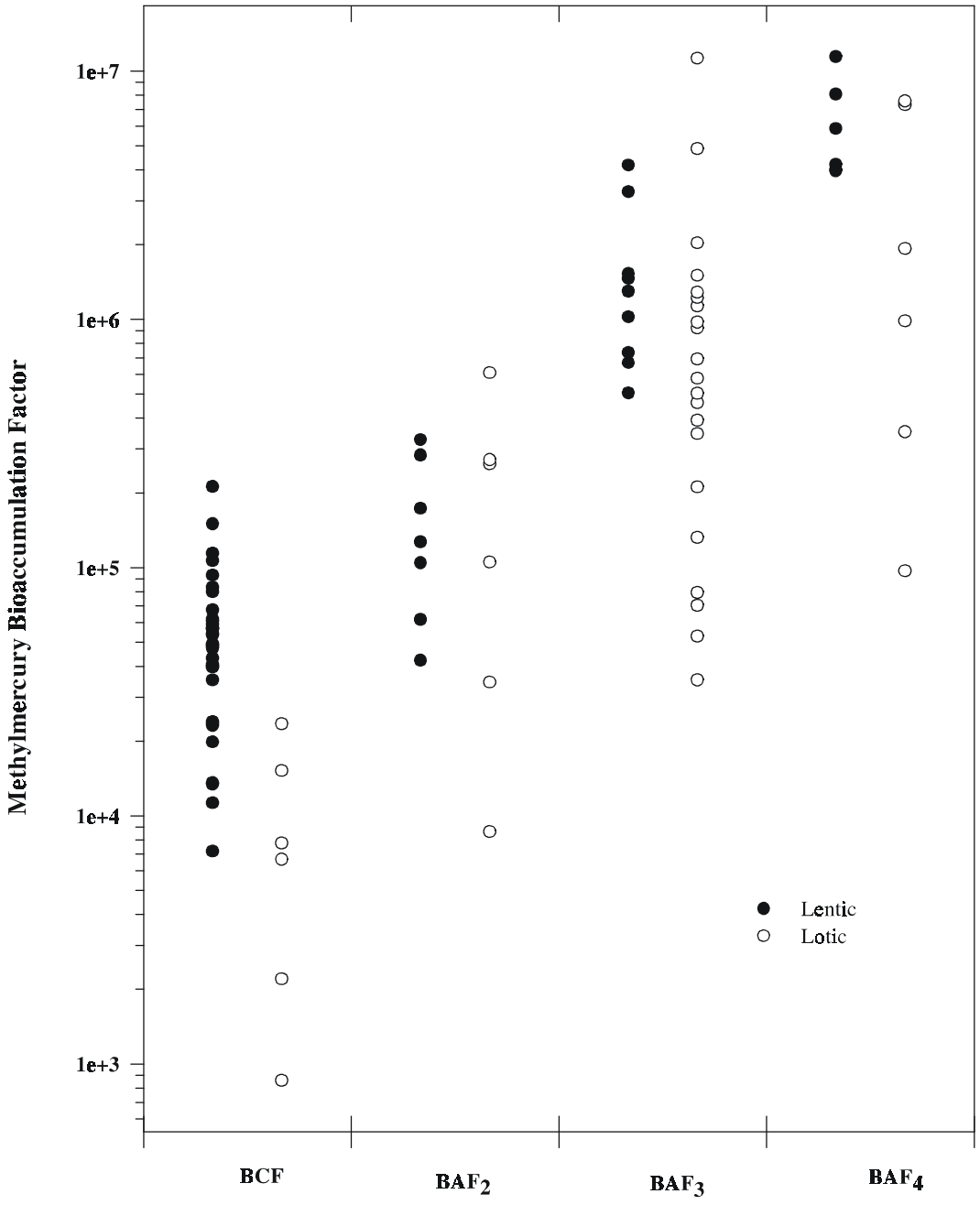

Figure A-3. Comparison of lentic and lotic methylmercury BAFs. Data includes both direct field-measured BAFs and converted field-measured BAFs.

Draft National Bioaccumulation Factors for Methylmercury

Based on the data presented above, and because the goal of the draft national BAFs is to be applicable under as many circumstances and to as many water bodies as possible, the BAFs based on the combined data sets (e.g., direct and converted, lentic and lotic) were chosen to be the empirically-derived draft national BAFs for methylmercury. The draft National BAFs, along with the draft BCF, and their empirical distributions are presented in Table A-9.

Table A-9: Summary of Draft National BAFs and BCF for Dissolved Methylmercury

Value[a]	BCF	BAF_2	BAF_3	BAF_4
5th percentile	5,300	18,000	74,300	250,000
50th (GM) percentile	33,000	117,000	680,000	2,670,000
95th percentile	204,000	770,000	6,230,000	28,400,000
GSD	3.03	3.15	3.84	4.21
Draft National Values	**3.3×10^4**	**1.2×10^5**	**6.8×10^5**	**2.7×10^6**

[a]GM = geometric mean; GSD = geometric standard deviation.

Discussion of Uncertainty and Variability in the BAF Estimates

The BAFs in this document were designed to estimate the central tendency of the concentration of mercury in fish of a given trophic level from an average concentration of dissolved mercury for water bodies located in the continental U.S. As shown in figures A1-A3, there is at least an order of magnitude in the variability of the individual BAF estimates for a given trophic level, which leads to uncertainty in the overall central tendency estimate. This is further reflected in the range of 90 percent (5th and 95th percentiles) confidence intervals. Although the empirical range of any given 90 percent confidence interval may largely overestimate the true extent of variability, the distributions do provide a rough estimate of the total uncertainty in the aggregate processes and an idea of the precision (or lack thereof) of the BAF estimates. The uncertainty in the BAF estimates is related to two basic sources. First is the uncertainty arising from natural variability, such as size of individual fish or differences in metabolic processes. Second is the uncertainty due to measurement error, such as error in measurements of mercury in water and fish samples or lack of knowledge of the true variance of a process (e.g., methylation). These two sources of uncertainty are generally referred to as "variability" and

"uncertainty", respectively. In this analysis, there was no distinction made between variability and uncertainty; they are aggregated in the final BAF distributions and point estimates. Thus, it cannot be determined where natural variability stops and uncertainty starts. However, some of the more important sources of variability and uncertainty are highlighted below in order to assist risk managers in understanding what the limitations are surrounding the BAFs, to see how the uncertainty in the BAF estimates might be reduced should they derive more data, and to assist them in decisions on development of site-specific BAFs.

Uncertainty Due to Sampling and Chemical Analysis

In many cases, water methylmercury concentrations reported in the available studies incorporated limited or no cross-seasonal variability, incorporated little or no spacial variability, and were often based on a single sampling event. Because fish integrate exposure of mercury over a life time, comparing fish concentrations to a single sample or mean annual concentrations introduces bias to the estimates. The geographic range represented by the water bodies is also limited. The available lentic data are biased towards northern oligotrophic lakes, primarily located in the Great Lakes region. The lotic BAFs are primarily based on data from canals of the Everglades (assumed to act as flowing aquatic ecosystems) and from a point-source-contaminated stream in Tennessee. Because of this general lack of data, a few studies on water bodies in other countries were included in the analysis, requiring one to assume that biotic and abiotic processes in these lakes are similar to lakes in the continental U.S.

The same sampling and analytical methods for water and tissue samples were not used in each acceptable study. Although all studies used met general requirements for data quality, studies with different analytical detection limits were combined to estimate the BAFs. The range of species used in the BAF estimates is relatively small compared to the suite of fish and invertebrates consumed by the general human population. Much of the available trophic level 4 data for both lentic and lotic ecosystems is limited to walleye, pike, or bass. For trophic level 3 much of the data is for bluegill and perch. For trophic level 2, most of the data was for zooplankton in lentic waters and for planktivorous fish in lotic waters. The lack of data complicated comparisons between the two aquatic ecosystems and introduces uncertainty into application of the BAFs.

Uncertainty Due to Estimation Method

Each of the approaches use to estimate BAFs have their own inherent uncertainties. Both the direct and indirect approaches assume that the underlying process and mechanisms of mercury bioaccumulation are the same for all species in a given trophic level and for all water bodies. The indirect approach deals with this assumption more specifically by assuming that the translators and fmmfs used to convert BAFs are equally applicable to all ecosystems. In reality, these factors are based on a limited set of data. Although the translators and fmmfs used in the analysis are consistent with those reported elsewhere (Porcella, 1994), they may over- or underestimate bioavailability and bioaccumulation in specific water bodies. Ideally, site-specific conversion factors would be used to estimate BAFs more reflective of conditions in a given water body. The approach used here aggregates all of the species-specific BAFs into a single trophic level-specific BAF; this also increases the over all variability in the BAF estimates.

Uncertainty Due to Biological Factors

Other than deriving BAFs based on organism trophic level, and initially by general water body type (i.e., lentic and lotic), there were no distinctions in the BAFs as to size/age of fish, water body trophic status, or underlying mercury uptake processes. It has been shown that methylmercury bioaccumulation for a given species can vary as a function of the ages (body size) of the organisms examined (Glass et al, 1999; Watras et al., 1998; Suchanek et al., 1993; Lange et al. 1993). As a result, it has been suggested that to reduce some of the lake-to-lake variability seen in BAFs for a given species, comparisons between water bodies should be made using "standardized" fish values (i.e., a value for a hypothetical 1 kg northern pike; Glass et al., 1999). Typically such data "normalization" is derived by linear regression of residue data collected from individuals of varying size and/or age. However, the currently available data are too limited to perform this kind of normalization; most of the water body-specific BAFs, and resulting trophic level distributions, are based on "opportunity" (whatever you catch, you include) and do not report age or size of individuals sampled.

Uncertainty Due to Universal Application of BAFs

Perhaps the greatest source of variability is that of model uncertainty. That is, uncertainty introduced by failure of the model (in this analysis a single trophic level-specific BAF) to represent significant real-world processes that vary from water body to water body. The simple linear BAF model relating methylmercury in fish to total mercury in water simplifies a number of nonlinear processes that

lead to the formation of bioavailable methylmercury in the water column and subsequent accumulation. Much of the variability in field data applicable to the estimation of mercury BAFs can be attributed to differences in biotic factors (e.g., food chain, organism age/size, primary production, methylation/demethylation rates), and abiotic factors (e.g., pH, organic matter, mercury loadings, nutrients, watershed type/size) between aquatic systems. As an example, in lake surveys conducted within a relatively restricted geographic region, large differences can exist between lakes with respect to mercury concentrations in a given species of fish (Cope et al., 1990; Grieb et al., 1990; Sorenson et al., 1990; Jackson, 1991; Lange et al., 1994; Glass et al., 1999). These observations have led to the suggestion that a considerable portion of this variability is due to differences in within-lake processes that determine the percentage of total mercury that exists as the methylated form. Limited data also indicate that within a given water body, concentrations of methylmercury are likely to vary with depth and season. Unfortunately, while the concentration of methylmercury in fish tissue is presumably a function of these varying concentrations, published BAFs are generally estimated from a small number of measured water values, whose representativeness of long-term exposure is poorly known. Furthermore, although it is known that biotic and abiotic factors control mercury exposure and bioaccumulation, the processes are not well understood, and the science is not yet available to accurately model bioaccumulation on a broad scale.

Summary

Three different approaches were use to estimate methylmercury bioaccumulation factors for use in deriving national 304(a) ambient water quality criteria for mercury. All three approaches resulted in BAFs with central tendency point estimates in good agreement with one another. Based on data comparability and EPA's national guidance for deriving BAFs, methylmercury BAFs estimated using directly measured and converted field data were used as the basis for deriving the draft national BAFs. Given the large range in the data, at this time lotic BAFs can not be distinguished from lentic BAFs, though the data suggests slightly reduced methylmercury accumulation may occur in higher trophic level organisms in lotic/wetland environments. The same trend is observed when BAFs are compared on a total mercury basis. Some of this difference might be accounted for by the lower accumulation of methylmercury at the base of the food chain in lotic/wetland ecosystems. A plausible explanation for this difference is the observation that the bioavailability of methylmercury in lentic environments (usually a low dissolved organic carbon content) may exceed the bioavailability of methylmercury in lotic/wetland environments (usually a high dissolved organic carbon content). Methylmercury and mercury have a high binding capacity to dissolved organic carbon which can affect their bioconcentration in

phytoplankton/periphyton. Watras et al. (1998) used modeling to show that BAFs based on the bioavailable fraction of methylmercury in water exceed BAFs based on the operationally defined (filtered) dissolved methylmercury in water. Bioavailability is perhaps the single most important factor affecting BAFs for mercury.

EPA fully recognizes that the approach taken to derive mercury BAFs collapses a very complicated non-linear process, which is affected by numerous physical, chemical, and biological factors, into a rather simplistic linear process. EPA also recognizes that uncertainty exists in applying a National BAF universally to all water bodies of the United States. Therefore, in the revised 2000 Human Health Methodology (EPA , 2000) we encourage and provide guidance for States, Territories, Authorized Tribes, and other stakeholders to derive site-specific field-measured BAFs when possible. In addition, should stakeholders believe some other type of model may better predict mercury bioaccumulation on a site-specific basis they are encouraged to use one, provided it is scientifically justifiable and clearly documented with sufficient data.

SECTION II. CHEMICAL TRANSLATORS FOR MERCURY AND METHYLMERCURY

Introduction

By regulation (40 CFR 122.45(c)), the permit limit, in most instances, must be expressed as total recoverable metal. Because chemical differences between the discharged effluent and the receiving water are expected to result in changes in the partitioning between dissolved and adsorbed forms of metal, an additional calculation using what is called a translator is required.

The translator is used to convert the dissolved concentration of a metal to a total metal concentration for use in waste load limit calculations. The translator is the fraction of the total recoverable metal in the downstream water that is dissolved, f_d. The translator can be used to estimate the concentration of total recoverable metal in a water body.

Methods

Two procedures were used to develop site-specific translators. The most straightforward approach for translating from a dissolved water quality criterion to a total recoverable effluent concentration is to analyze directly the dissolved and total recoverable fractions. The translator is the fraction of total recoverable metal that is dissolved. It may be determined directly by measurements of dissolved and total recoverable metal concentrations in water samples taken from the well mixed effluent and receiving water (i.e., at or below the edge of the mixing zone). In this approach, a number of samples are taken over time and an f_d value is determined for each sample:

$$f_d = Cd/Ct \quad [Eqn. 1]$$

where:

Cd = the dissolved concentration, and
Ct = the total metal concentration.

The translator is then calculated as the geometric mean (GM) of the dissolved fractions.

The second approach derives an f_d from the use of a partition coefficient K_D where usually the coefficient is determined as a function of total suspended solids (TSS) (although some other basis such as humic substances or particulate organic carbons may be used). The partition coefficient is the ratio of the particulate-sorbed and dissolved metal species multiplied by the adsorbent concentration, i.e. Cd + TSS ⇌ Cp, where Cp is the bulk particulate-sorbed concentration, and is expressed as:

$$K_D = Cp/(Cd \cdot TSS) \qquad [Eqn.2].$$

The dissolved fraction and the partition coefficient are related as shown in equation 3.

$$f_d = (1 + K_D \cdot TSS)^{-1} \qquad [Eqn.3]$$

As in the first approach, numerous samples are collected over time, and the f_d and TSS values found at the site are fit to a least squares regression, the slope of which is K_D. The established K_D is then used to determine the translator using Eqn. 3 with a TSS value representative of some critical condition, e.g., low flow conditions.

Although development of site-specific translators is recommended, EPA also envisions the possible need for national or default translators for use in translating dissolved mercury and dissolved methylmercury criteria into total mercury and methylmercury water quality permit limitations. Translators and/or related K_D values can be generated from an acceptable existing literature-derived data base. EPA's MSRC (U.S. EPA, 1997) contains extensive data, obtained primarily from lake systems, that are relevant to developing translators for mercury (e.g., percent total as methylmercury, percent total as dissolved mercury). Supplementation of these translators with additional, acceptable data from lotic and estuarine systems and update of lentic systems provides the necessary data base for the translators. To gather this data base, peer-reviewed literature papers from 1990 to present, were searched and reviewed. Since awareness of the contamination problems with mercury at low levels and the existence of analytical methods capable of accurately and precisely measuring mercury and methylmercury at low levels are relatively recent, the literature review was not conducted for publications prior to 1990. All data from the literature for use in developing the translators were required to meet the following criteria:

- Clean techniques, or equivalent, to reduce contamination were used in sampling and analysis.
- Adequate QA/QC procedures were used.
- Analytical methods used provided sufficiently low enough detection level.

Draft Translators

Table A-10 summarizes the numerous tables from the EPA internal draft BAF report (see Water-Docket W-00-20). These results are presented separately for lake, river and estuarine systems, and for each system, where sufficient data were available, both f_d and K_D values were tabulated. The K_D values were calculated using Eqn. 2. The K_D values could not be derived using the f_d-TSS correlation approach due to the limited data, i.e., multiple sampling events over time with measurements of both f_d and TSS were not conducted in most of the studies. The results are presented separately for both mercury and methylmercury. Table A-10 provides a summary of the GM values calculated for each system for f_d and K_D values, again for both mercury and methylmercury.

It is possible to calculate a "pseudo" K_D value for the partitioning of dissolved methylmercury with particulate total mercury using f_d and K_D data for a waterbody utilizing the following equation (see Attachment B for derivation and example calculation):

$$\text{"Pseudo" } K_D \text{ MeHgd/Hgt} = K_D \text{ MeHgd} \cdot \text{MeHgt} \bullet \text{Ratio Hgd/MeHg}_p \bullet \text{Ratio MeHgd/MeHg}_p$$

[Eqn. 4]

Table A-10: Summary of F_d and K_d Values for Lakes, Rivers, and Estuaries[a]

f_d and K_D Values	Lakes	Rivers	Estuaries
f_d Hg	0.60	0.37	0.353
f_d $MeHg_d/Hg_t$	0.032	0.014	0.190[b]
f_d $MeHg_d/MeHg_t$	0.613	0.49	0.612[b]
Log K_D Hg	5.43	5.06	5.52
Log K_D MeHg	5.53	4.81	NF[c]
"pseudo" Log K_D $MeHg_d/Hg_t$	6.83	6.44	NC[d]

a Values calculated as GM
b Only two sites
c No data found from the literature search
d Not able to calculate due to insufficient data

The K_D so derived is a "pseudo" value since dissolved methylmercury partitioning with particulate total mercury is just a synthetic or functional type description. These values are also given in Table A-10. The "pseudo" K_D values, however, allow for direct translation of dissolved methylmercury criteria to total mercury permit limits employing some designated TSS level. Insufficient data were found, e.g., K_DMeHg, to allow for calculation of "pseudo" K_Ds for estuaries. It should be understood that all values in Table A-10 represents values generated from the above-described literature-gleaned data base. Insufficient data were obtained to provide either reliable f_d (translator) or K_D "default" values for methylmercury for estuarine systems (only two sites). Examination of the translator values for lakes and rivers shows that in all instances the river values for both f_ds and K_Ds are lower than the lake values. The lower translator values can be generally explained by the generally higher TSS levels found in rivers as compared to lakes. For example, typical TSS values for eastern Washington state lakes are 0.5 to 5 mg/L, whereas river levels can be typically 5-50 mg/L (Pankow and McKenzie, 1991). Higher TSS levels lead to lower f_d values.

The lower K_D values for rivers vs. lakes are not as readily explainable. K_D values are not constant and are sensitive to environmental conditions and water chemistry (Sung, 1995). Inclusion of the colloidal fraction in the dissolved phase that is used in determining the K_D has been used to explain variation of K_D values and for deviation of the values from any true K_D (Pankow and McKenzie, 1991; Sung, 1995). Higher colloidal contents or higher DOC levels in the river samples compared with lake samples would produce lower apparent (as measured) K_D values. However, the following other factors have been suggested to play major roles in K_D determinations, and one or all of these may contribute significantly to the reason why the river K_Ds are less than the lake K_Ds for both mercury and methylmercury:

- Biotic or organic content of the TSS
- Dissolved organic content of the water
- Geochemistry and residual metal content of the TSS
- TSS particle size
- Pollution level existing in the waters

Regardless of the reason(s) for the differences between the lake and river values, differences do exist and are sufficiently significant that it is recommended that the two systems be treated separately with regard to translator values. Until additional data are available for estuarine systems, and a satisfactory comparison to lake and river systems can be made, it is recommended that separate values be retained for estuaries also.

One can estimate the TSS level that is represented by the f_d values for each system through the use of Eqn. 3 and employing the default K_D values provided in Table A-10. The results of calculations of these estimated levels and an example calculation are presented in Table A-11. The data show the following:

- In lakes, the f_d for mercury (0.60) would reflect TSS levels of 2.5 mg/L. The f_d for methylmercury (0.032) would reflect TSS levels of 1.8 mg/L. At TSS levels lower than these values, a greater fraction of the mercury and methylmercury would be expected to be dissolved than indicated by the f_d.

- In rivers, the f_d for mercury (0.37) would reflect TSS levels of 14.8 mg/L. The f_d for methylmercury (0.014) would reflect TSS levels of 16.3 mg/L. At TSS levels lower than these values, a greater fraction of the mercury and methylmercury would be expected to be dissolved than indicated by the f_d.

- In estuaries, the f_d for mercury (0.35) would reflect TSS levels of 5.5 mg/L.

Existing TSS levels less than those above would, in any instance, that the dissolved fraction present in the water could be greater than the value suggests.

Use of the partition coefficient approach may provide advantages over the dissolved fraction. EPA suggests (EPA, 1996) that when using dynamic simulation for Waste Load Allocation (WLA) or the Total Maximum Daily Load (TMDL) calculations and permit limit determinations, K_D allows for greater mechanistic representation of the effects that changing environmental variables have on f_d (the significance of the TSS variable has been shown in Table A-11 data and discussed above, and this variable is addressed or can be handled in the K_D approach).

Table A-11: Estimation of TSS Level at f_d Values

	Lakes		Rivers		Estuaries	
	f_d	**Est. TSS, mg/L**	**f_d**	**Est. TSS, mg/L**	**f_d**	**Est. TSS, mg/L**
Mercury[a]	0.60	2.5*	0.37	14.8	0.35	5.5
Methylmercury[b]	0.032	1.8	0.014	16.3	0.190	NC[c]

(a) Calculated using default K_D values and equation: $f_d = 1/(1+K_D \times TSS)$

(b) Calculated using default "pseudo" K_D values and equation: $f_d = 1/(Hg_d/HgMe_d + K_D \times TSS)$

(c) Not able to calculate; insufficient data.

* Calculation:

$f_d = 1/(1 + K_D \times TSS \times 10^{-6})$ note: 10^{-6} used to provide TSS in mg/L units

default K_DHg (lakes) = 269,153

substituting: $0.60 = 1/(1 + 269{,}153 \times TSS \times 10^{-6})$

$0.60 + 0.161 \times TSS = 1$

$0.161 \times TSS = 0.40$

$TSS = 2.5$

Although the K_D approach may be advantageous in use, employment of a default K_D value has inherent problems as does the use of a f_d. For example, mercury K_Ds have been shown to range from about 10^4 to about 10^6 (Watras et al., 1995). At an average K_D value of about 10^5 (the value found for rivers), and a critical TSS level of 10 mg/L, a translator value of 0.5 is derived from the K_D approach.

However, if the site K_D, for example, is close to the lower end of the K_D range, the translator value should be about 0.9. Thus the value is inaccurate at this site. Only at sites where the existing K_D is 10^5 or greater (at 10 mg/L TSS) would the use of the default K_D yield a translator value that does not underestimate the dissolved mercury level.

An additional problem with the use of the K_D approach is that even at a given site, K_D values can vary. Usually, K_D values decrease at a site as TSS increases, as has been shown recently for mercury and methylmercury in a Virginia river (Mason and Sullivan, 1998). In addition, the K_D translator approach necessitates that f_d correlate with TSS. A poor correlation, however, has been found to exist for many metals in a recent analysis of data obtained from State of Michigan surface waters (MDEQ, 1996).

Although the K_D approach has its advantages, the f_d approach is the most straightforward. Both approaches have their disadvantages, as discussed previously. The K_D is derived from f_d values and so the two approaches are truly linked. Therefore, preferential recommendation of either one approach over the other at present cannot be made.

Use of either f_d or K_D default values can be made as long as one recognizes the short comings of the approach taken. Perhaps the approach taken should be the one with the stronger data base, if a clear difference exists. As additional data appears in the literature, it is reasonable to assume that a fine-tuning of both the f_d and K_D default values will result. EPA recommends that translators be derived from site-specific studies when possible, but the values in Table A-10 could be used in absence of any site-specific data.

ATTACHMENT A: BAF PEER REVIEWERS' GENERAL COMMENTS

The following was excerpted from the BAF Peer Review Comments Report, August 23, 2000. See Water Docket W-00-20 for a complete version of the peer review report.

2.0 REVIEWERS' COMMENTS

2.1 General Comments

Nicolas Bloom

Overall, I found the document quite clear and well written compared to other EPA mercury documents that I have recently reviewed, a fact that made my job considerably easier. On the other hand, it seems quite clear that there is insufficient data currently available for the EPA to make any more than the broadest generalizations about methyl mercury bioaccumulation factors. The current greater than one order of magnitude spread in estimated BAFs will not be very useful in any actual case, although it serves to describe the situation in general terms. The EPA should be impelled to proceed by instigating research and/or requiring site-specific bioaccumulation factors to be developed until such time that a sufficient database is accumulated to allow some meaningful resolution between BAFs from different water body types, climates, and trophic levels.

I oppose the general use of the confusingly similar terms "lentic" and "lotic," which although probably clear to fish ecologists, never-the-less provide endless confusion to the rest of us. I conducted a poll of the 51 employees of our aquatic sciences research company, and no one could define these words correctly, although a few did say that they had heard of them back in college. Additionally, even though physically, the term "lentic" can be used to lump together the Everglades with a swiftly moving glacial stream, I see no logical biogeochemical reason to do so.

There is also the overwhelming sense, in the description of the trophic levels considered, that the only valid food chain model being considered is the water to plankton to zooplankton to fish model. However, many systems (i.e., Lavaca Bay, TX) are dominated by a sediment porewater to benthic invertebrates to fish model, which means that sediment issues (methyl concentrations, methylation depth profiles, redox condition, seasonality, etc.) loom way more important that water column concentrations.

James Hurley

First and foremost, the development of a national AWQC for methylmercury must be based on sound data with strict quality control/quality assurance to ensure that the calculation of bioaccumulation factors (BAFs) is scientifically valid. This is a difficult task when conducting literature searches for data that form the backbone of the report. Among the data chosen, methods must be comparable to allow transferability. Individual investigators also apply different definitions of biological assemblages and food chain pathways. This makes the task of synthesizing appropriate data a difficult task at best.

My overall concern with data used for determination of the national BAF is that not one study from which data was obtained for this report was actually with the specific purpose of generating MeHg-based BAFs through all trophic levels. I fully understand that EPA also recognizes this problem and commend them for assembling the data presented. However, I do think that EPA should consider a research effort designed to produce results directly related to their MeHg BAF goals. This would ensure that sample types and methodologies were consistent with the overall goal of development of national BAFs for methylmercury. Development of a scientifically sound BAF is a critical step in development of a management plan for this Level I contaminant in the U.S.

In addition to developing a field effort, EPA should also consider development of dedicated laboratory studies that address Hg and MeHg partitioning and transport in trophic levels 1 and 2. Although EPA decided to choose an approach that incorporates field-derived BAFs, laboratory studies using cultures of phytoplankton and zooplankton, coupled with key contrasting water chemistries, would certainly aid in reducing the variability that is inherent in using field-derived data on partitioning. Results of these studies alone would avoid the ambiguity that is inherent in using the terms “seston” and “phytoplankton” interchangeably for BCFs.

The current report divides the data into two environments (lentic and lotic) but then combines BAFs to determine a national BAF in the final section of the report. I strongly encourage EPA to establish a series of National BAFs that are watershed-type based, in slightly more detail than a simple lentic/lotic division. Data from lotic systems in the report combine wetlands with flowing rivers. As a result, the lotic grouping contains high dissolved organic carbon (DOC) systems such as wetlands, with low DOC headwater streams. This type of grouping of sites with such disparate Hg-cycling environments most likely accounts for both the spread of data for directly-calculated BCFs and the lack of agreement between directly calculated and converted BCFs depicted in Figure 5-2.

While I agree that translators are appropriate in some instances, they too should be calculated on a more site-specific basis. Use of the translators to calculate the fraction (f_d) of total Hg as MeHg should be refined to address factors such as trophic state and watershed type. The grand mean of 3.2% for this translator encompasses a range from 0.2% to 13.9% in lake waters. Similarly, the grand mean from rivers of 1.4% encompasses a range from 0.2 to 5.11% in rivers. Better grouping of the data would reduce variability for this data set. For instance, Kd's for several contaminants have been shown to decrease with increasing DOC. The processes controlling methylation and particle partitioning are site-specific, and the current report attempts to define complex chemical and biological processes across gradients by the use of a simple fraction. Since this factor (the amount of inorganic Hg that is converted to the bioaccumulative methyl form) is perhaps the most critical step in developing a BCF, a simple default conversion factor is not the best approach.

Finally, development of an acceptable model is mentioned within the report as a future goal, but I feel that model development and acceptance should be fast-tracked along with development of a National MeHg BAF. Models, such as the recent revisions of the Mercury Cycling Model (MCM), that incorporate processes such as methylation, aquatic speciation, and bioenergetics are keys to validation of the BAFs among contrasting sites. Having worked specifically with the MCM Model, I am confident that is has been tested on a number of contrasting environments (northern Wisconsin lakes, Everglades, Great Lakes) and could be used to validate BAFs for differing aquatic environments.

David Krabbenhoft

Overall, I found the document to be in very good order structurally, grammatically, and was of an appropriate length for the subject matter; my compliments to the authors. A quality manuscript makes the reviewer's job much easier, and a better technical review results when he or she is not "put off" for having to do editorial service too. I heartily support the U.S. EPA's decision to pursue changes to the AWQC for mercury and have methylmercury (MeHg) be the basis for such regulations. Although this has been a long time in coming, I do recognize that the peer reviewed data for this type of proposed change has been limited to just a few study locations until the past few years. That being said, however, I have serious reservations as to whether enough high quality data has been made available by the scientific community for the EPA to make an important decision like assigning "National BAF's". The authors of this report have largely done an admirable job with what is available, but it may be slightly ahead of its time. It may be that with the very recent release of the National Academy of Sciences report on human health and mercury, and the proposed decision time line of the EPA to enact emissions

regulations in the 5-year time frame, that a well-conducted, national-synoptic study to for the proper basis for a MeHg BAF's is in order.

David Maschwitz/Edward Swain

1. An update of the mercury bioaccumulation factor (BAF) is very much needed, for the reasons cited on page 1 of Section I. The new analytical methods that can measure ambient mercury in water at sub-nanogram per liter levels, and the large number of recent studies that provide field measured BAF data make the determination of a new BAF a necessity, if EPA plans to update the human health-based mercury criterion. The BCFs/BAFs used in previous EPA mercury criteria are clearly outdated. A new mercury BAF and criterion will be a great help to states and tribes (hereinafter, state). The determination of a BAF is often the biggest road block to the calculation of a human health-based water quality standard for state regulatory agencies.

2. The following comments are on *National Bioaccumulation Factors for Methylmercury* (Section I) and *Default Chemical Translator for Mercury and Methylmercury* (Section II). We have not reviewed for comment the background document.

3. The overall organization of Sections I and II, is logical, straight forward and easy to follow.

4. The EPA search for both available published and unpublished BAF data uncovered a substantial amount of new information; and, short of carrying out an independent literature search to confirm this comment, it should be reasonably complete and current.

5. The discussion of uncertainty associated with the final recommended BAFs (beginning on page 73, Section I), including a discussion of the limitations associated with reducing highly variable BAF data to a single national BAF (for each trophic level), and the myriad of variables that can affect BAFs, is appropriate. Further, EPA's rationale that, in spite of the uncertainty (actually, because of it), the recommendation of a single default BAF for each trophic level is valid. The recommendation that states should use local BAF data is good as well, but EPA must realize that local BAF data is not likely to be available in many situations. Thus, the default BAFs will get substantial use.

6. The decision to use only the preferred, field measured, BAF data (including the converted direct BAFs) and not use the indirectly determined BAFs (BCFs or BAFs times a FCM or BFM) is appropriate given the quality and quantity of the former. This is consistent with the proposed new EPA human health criteria methodology (EPA, 1998). However, including the comparison of direct and indirect BAFs in Section I (Tables 3-10 and 4-11) is valuable information.

7. To eliminate any uncertainty about the proper application of the translators listed in Tables 5-1 and 5-2, it is suggested that EPA include in Tables 5-3 through 5-10 columns showing the translators used in the conversions, and/or a column showing the "raw" as well as the converted BAFs. An alternative to expanding these tables is to add to the summary information at the beginning of each subsection (i.e., Variable, Definition, Estimate, Distribution) a section on "Translators" or "Conversion" that shows the translator(s) and conversion calculations (this option assumes the translators used and all the conversion calculations are the same for all the individual BCFs/BAFs). A third, but less desirable alternative, is to provide example calculations in the introductory discussion of converted methylmercury BAFs, beginning on page 49 of Section I.

8. Overall, we believe the final recommended BAFs (Table 5-15) are supported and a reasonable conclusion of the data analysis.

9. The introduction to Section II (page 1) talks about EPA's policy to use dissolved analyses for trace metals to measure compliance with the standard. This policy was developed in the context of the toxicity of particulate and chemically bound, versus the toxicity of "dissolved" or ionic forms, of trace metals to aquatic life. The science behind EPA's dissolved metal policy may not be as relevant to a highly bioaccumulative metal like mercury, for which the concern is the methyl form, and the risk is to human health through fish consumption rather than to aquatic life directly. EPA should expand this section to discuss if and how mercury differs from non-bioaccumulative trace metals with regard to the need or desirability of measuring dissolved metal in water.

10. EPA discusses in the "Background" part of Section II, total to dissolved metal conversion factors. Along the lines of comment number nine, the conversion factor of 0.85 for the current mercury criteria (CMC and CCC) are applicable to toxicity-based mercury criteria, not the human health-based chronic criterion (*Federal Register* 63: 68354-68364). The conversion factor for the chronic human health-based mercury criterion is 1.0 (see also *Federal Register* 60: 15392). EPA needs to

revise their discussion of conversion factors to reflect the conversion factor for the human health criterion, and to address the points made in comment number nine.

11. Separate average translators and K_D values for lakes, rivers and estuaries as derived in Section II seem to be reasonable and supported by the data presented.

Darell Slotton

I found the reports to be clear in their intent and in their explanation of approaches used. I especially appreciated the straightforward acknowledgment of the myriad sources of uncertainty and variability. My overall response to the entire exercise is that those sources of uncertainty and variability (geographic, water quality, water trophic status, analytical, individual organism, true trophic "level", food web complexity, etc.) make this a very difficult if not impossible proposition. I strongly support the development of tissue-based mercury criteria as the preferred mechanism for addressing mercury risk assessment and regulatory concerns throughout the huge range of aquatic systems affected. That said, if EPA has a legal charge to also develop the best predictive relationships it can as defaults, etc., the approach being used is probably as good as can be expected. It may be significantly more useful as a regional tool, though (e.g., northern midwestern lake systems, California rivers, Florida, etc). A truly applicable, nation-wide set of factors may be unattainable. I strongly concur with the suggestion that site-specific research is preferable in the event that BAFs are to be used.

ATTACHMENT B: DERIVATION AND CALCULATION OF "PSEUDO" K_DS FOR METHYLMERCURY

Derivation

$MeHg_d$ + TSS $\rightleftharpoons$ Hg_p, including $MeHg_p$

"Pseudo"K_DMeHg / Hg = $\frac{Hg_p}{MeHg_D \bullet TSS}$ [Eqn. A.1]

Also: K_DMeHg = $\frac{MeHg_p}{MeHg_D \bullet TSS}$ [Eqn. A.2]

Equating TSS and combining Eqn. A.1. and Eqn. A.2 yields:

"Pseudo"K_DMeHg / Hg • $\frac{MeHg_d}{Hg_p}$ = K_DMeHg • $\frac{MeHg_d}{MeHg_p}$

Rearranging:

"Pseudo"K_DMeHg / Hg = $\frac{Hg_p}{MeHg_d}$ • KMeHg • $\frac{MeHg_d}{MeHg_p}$ [Eqn. A.3]

Example Calculation for Lakes (see text of original draft report for source of data)

- K_DMeHg = 338,844
- When Hg_T = 1, $MeHg_d$ = 0.032, Hg_d = 0.60 and therefore Hg_p = 0.40
  and the ratio $Hg_p/MeHg_d$ = 0.40/0.032 = 12.5
- When MeHg = 1, $MeHg_d$ = 0.613, and therefore $MeHg_p$= 0.387
  and the ratio $MeHg_d/MeHg_p$ = 0.613/0.387 = 1.58
- Substituting the above values in Eqn. A.3 gives:
  "Pseudo" K_DMeHg / Hg = 12.5 • 338,844 • 1.58 = 6,692,169
  Log "Pseudo" K_DMeHg / Hg = 6.83

Example Calculation for Rivers (see text of draft report for source of data)

- K_DMeHg = 64,565
- When Hg_T = 1, $MeHg_d$ = 0.014, Hg_d = 0.37 and therefore Hg_p = 0.63
  and the ratio $Hg_p/MeHg_d$ = 0.63/0.014 = 45.0
- When MeHg = 1, $MeHg_d$ = 0.49, and therefore $MeHg_p$= 0.51
  and the ratio $MeHg_d/MeHg_p$ = 0.49/0.51 = 0.96
- Substituting the above values in Eqn. A.3 gives:
  "Pseudo" K_DMeHg / Hg = 45.0 • 64,565 • 0.96 = 2,789,208
  Log "Pseudo" K_DMeHg / Hg = 6.44