Technical Support Document: Methodology Used to Generate Deposition, Fish Tissue Methylmercury Concentrations, and Exposure for Determining Effectiveness of Utility Emission Controls:

Analysis of Mercury from Electricity Generating Units:

1. Mercury Emissions from Power Plants

The specific controls anticipated to be adopted by utilities under the Clean Air Interstate Rule (CAIR) and Clean Air Mercury Rule (CAMR) are expected to preferentially reduce the forms of mercury that are of concern with respect to local deposition (non-elemental mercury) (See Control of Mercury Emissions from Coal Fired Electric Utility Boilers: An Update in the docket).

Based on the analysis of CAIR, EPA's modeling projects that mercury emissions would be 38.0 tons (12 tons of non-elemental mercury) in 2010, 34.4 tons in 2015 (10 tons of non-elemental mercury), and 34.0 tons in 2020 (9 tons of non-elemental mercury), about a 20 and 30 percent reduction (in 2010 and 2015, respectively) from a 1999 baseline of 48 tons. With respect to oxidized mercury, emissions in 2020 are 7.9 tons compared to 20.6 tons in 2001. This 62 percent drop in oxidized mercury emissions is particularly important because this species of mercury deposits more readily. Tables 1.1 and 1.2 highlight the changes in speciated mercury emissions from the 1999 ICR through the CAIR and CAIR plus CAMR regulatory approaches.

Elemental mercury has an atmospheric half-life of 6-12 months, which makes it an important contributor to the global pool. Coupled with the fact that elemental mercury is generally non-water soluble, it's impact on local deposition is minimized. On the other hand, oxidized mercury has a half-life on the order of days to weeks, is relatively water soluble, and contributes significantly to nearfield (< 25 km) deposition. Thus, an appropriately designed control strategy should focus on reductions in oxidized mercury to optimize its ability to impact local deposition in and around coal-fired power plants.

The graphs in Appendix A demonstrate the local or nearfield depositional aspect of oxidized and particulate mercury, and the relative unimportance of elemental mercury in the nearfield. Particulate mercury tends to form in the atmosphere from oxidized mercury in conjunction with relatively available atmospheric constituents, (e.g., dust and soot particles).

Given the settling characteristics of particle-bound mercury, this species of mercury also is important to nearfield deposition; however, not nearly as important in terms of magnitude as oxidized mercury.

EPA projected future mercury emissions from the power generation sector using the Integrated Planning Model (IPM). EPA uses IPM to analyze the projected impact of environmental policies on the electric power sector in the 48 contiguous States and the District of Columbia. IPM is a multi-regional, dynamic, deterministic linear programming model of the U.S. electric power sector. EPA used IPM to project both the national level and the unit level of utility unit mercury emissions under different control scenarios. EPA also used IPM to project the costs of those controls (see Chapter 7 of CAMR Final RIA for further discussion).

In these IPM runs, EPA assumed that States would implement the mercury requirements through the mercury cap and trade program that EPA is establishing in today's rulemaking. The cap-and-trade program is implemented in two phases, with a hard cap of 38 tons in 2010 (set at the co-benefits reduction under CAIR) and 15 tons in 2018. EPA modeling of section 111 projects banking of allowances due to excess mercury reductions in the 2010 to 2017 timeframe for compliance with the cap in 2018 and beyond timeframe. A cap-and-trade program assures that those reductions will be achieved with the least cost. For that reason, EPA believes it reasonable to assume that States will adopt the program even though they are not required to do so.

Under the CAMR scenario modeled by EPA, units are projected to install SCR and scrubbers to meet their SO_2 and NO_x requirements under CAIR and take additional steps to address the remaining mercury reduction requirements under section 111, including adding mercury-specific control technologies (model applies Activated Carbon Injection), additional scrubbers and SCR, dispatch changes, and coal switching. Many of these reductions are projected to result from large units installing controls and selling excess allowances. Under the cap-and-trade approach we are projecting that mercury reductions result from units that are most cost effective to control, which enables those units that are not cost effective to install controls to use other approaches for compliance including buying allowances, switching fuels, or making dispatch changes.

Table 1.1 below provides mercury emissions projections organized by plants with certain categories of total mercury emissions. As presented in the table, the largest emitting plants (those with total mercury emissions greater than 230 kg/yr) represented the largest total mercury emissions in 1999, but under CAIR and CAMR, this category of plants have the greatest reductions in ionic mercury emissions. In addition, as shown in Table 1 and 2, the smallest emitting category of plants (plant with Hg total less than 90 but greater than 10 kg per year) actually increases under the CAIR and CAMR options. This increase is consistent with the conclusion that reductions are projected to result from large units installing controls. As controls are installed, the large emitting units move to the category of lower emitting units.

Table 1.1: Mercury Emission Projections (kg/yr)

1999 ICR Emissions	% of total plants	Mercury (p)	Mercury (++)	Mercury (O)	Mercury total
Plants with Mercury total > 230 Kg/yr	15%	619	8,824	10,804	20,248
Plants with Mercury total < 230 but > 90 Kg/yr	29%	444	6,182	8,171	14,797
Plants with Mercury total < 90 but > 10 kg/yr	56%	263	3,371	4,540	8,174
Total National Emissions		1,346	18,514	23,673	43,533
CAIR 2020 Projected Emissions	**% of total plants**	**Mercury (p)**	**Mercury (++)**	**Mercury (O)**	**Mercury total**
Plants with Mercury total > 230 Kg/yr	6%	131	1,292	7,632	9,055
Plants with Mercury total < 230 but > 90 Kg/yr	22%	361	2,985	10,139	13,485
Plants with Mercury total < 90 but > 10 kg/yr	72%	331	2,866	7,134	10,331
Total National Emissions		751	7,139	23,337	31,227
CAMR 2020 Projected Emissions	**% of total plants**	**Mercury (p)**	**Mercury (++)**	**Mercury (O)**	**Mercury total**
Plants with Mercury total > 230 Kg/yr	3%	121	589	2,964	3,674
Plants with Mercury total < 230 but > 90 Kg/yr	17%	267	2,061	5,945	8,273
Plants with Mercury total < 90 but > 10 kg/yr	80%	315	2,839	6,570	9,725
Total National Emissions		753	5,960	16,000	22,713

Note: Total national emissions include emissions from plants less than 10 kg/yr.

Table 1.2: Percent Reduction from 1999 Emissions

CAIR 2020	Mercury (p)	Mercury (++)	Mercury (O)	Mercury total
Plants with Mercury total > 230 Kg/yr	79%	85%	29%	55%
Plants with Mercury total < 230 but > 90 Kg/yr	19%	52%	-24%	9%
Plants with Mercury total < 90 but > 10 kg/yr	-26%	15%	-57%	-26%
Total National Emissions	44%	61%	1%	28%
CAMR 2020	**Mercury (p)**	**Mercury (++)**	**Mercury (O)**	**Mercury total**
Plants with Mercury total > 230 Kg/yr	81%	93%	73%	82%
Plants with Mercury total < 230 but > 90 Kg/yr	40%	67%	27%	44%
Plants with Mercury total < 90 but > 10 kg/yr	-20%	16%	-45%	-19%
Total National Emissions	44%	68%	32%	48%

2. Analysis of Mercury Deposition from Power Plants

The Community Multi-Scale Air Quality model (CMAQ) is a three-dimensional grid-based Eulerian air quality model designed to estimate pollutant concentrations and depositions over large spatial scales (e.g., over the contiguous United States).   Because it accounts for spatial

and temporal variations as well as differences in the reactivity of Mercury emissions, CMAQ is best suited for evaluating the impacts of the CAMR on U.S. mercury depositions. This model accounts for the atmospheric reactions of specific mercury emissions and their significance to the levels of deposition as shown through our results here for CAMR. In addition, the boundary and initial species concentrations are provided by a three-dimensional global atmospheric chemistry and transport model, (i.e., Harvard's GEOS-CHEM model). The model simulations were performed based on plant-specific emissions of mercury by species as provided by the Integrated Planning Model (IPM).

2.1 Deposition Analysis

Regional deposition is best modeled using a grid model that accounts for atmospheric chemistry, meteorology, and large scale fate and transport, as well as impacts of global emissions of mercury. For this analysis, the gridded estimates of mercury deposition from application of the CMAQ model were used to evaluate the impacts of utility mercury controls. The CAMR Emissions Inventory and Air Quality Modeling Technical Support Document (U.S. EPA, 2005) discusses the development of the 2001 and 2020 emissions inventories and the CMAQ mercury deposition modeling in greater detail.

For this analysis, deposition at 36 km grid cells was aggregated to the eight-digit watershed using the ArcMap spatial join function (ESRI, 2004)[1,2]. The HUC, Hydrologic Unit Code, developed by the USGS, spatially delineates watersheds throughout the United States[3]. Hydrologic units are available at four levels of aggregation, ranging from a two-digit regional level (21 units nationwide) to the eight-digit HUC (2,150 distinct units). The eight-digit HUC-level designation is useful for this analysis because it provides a nationally consistent approach for grouping waterbodies on a sufficiently local scale (the average HUC area is 1,631 sq mi). The average deposition for the grid cells that intersect the HUC-8 polygon is then used as the deposition value for the HUC-8 unit. Averaging over grid cells may result in a smoothing out of areas of high and low deposition, because the CMAQ grid cells are smaller than many HUCs.

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[1] Just like states can be subdivided into counties, large watersheds can be subdivided into smaller and smaller watersheds. For example, the Chesapeake Bay Watershed is composed of 104 small 8–digit Hydrologic Unit Code (HUC). The 8-digit HUC is the smallest USGS Classification. We refer to 8-digit HUCs as 8-digit watersheds for clarity.

[2] While appropriate for regional scale analyses, use of horizontal grids finer than 36 km^2 would be expected to result in higher local dry deposition of reactive gaseous mercury emissions, especially those from surface-level emissions or from short exhaust stacks. Eulerian grid models such as CMAQ immediately dilute simulated emissions into the entire grid volume in which they are released. This causes an artificially fast dilution and under-represents direct deposition from air to surfaces near emission sources. The magnitude of this artificial dilution depends on a number of factors related to emission source characteristics and atmospheric variables.

[3] More information regarding these hydrological units can be found through the USGS Web site http://water.usgs.gov/GIS/huc.html.

This approach averages mercury deposition rates projected by CMAQ at the 36 km grid cell resolution across all model grid cells that occur within a given HUC-8. We consider the HUC-8 aggregation most appropriate for this regional deposition analysis in light of the following rationale. First, because much of the mercury deposited on the watershed of different ecosystems will eventually enter waterbodies through subsurface inflow and runoff, we consider a watershed scale analysis to be more appropriate than finer scale resolution that may only describe direct inputs to surface waters. Second, in larger waterbodies, (i.e., the Great Lakes) where there is substantial fishing activity, the higher trophic level fish species consumed by humans are likely migratory and the accumulation of mercury by these species will represent an aggregated signal from deposition over a wider area, (e.g., the entire waterbody). Third, many anglers catching these higher trophic level fish may not always fish in the same waterbody within a watershed. Since we are concerned about the cumulative dose over weeks and months from repetitive consumption of fish containing methylmercury, this fishing behavior should be considered in the exposure pathway. Based on the above considerations, we conclude that the HUC-8 watershed is the appropriate unit of measure for analyzing regional deposition patterns. As discussed in section 3, for the analysis of individual fish sample locations, we use the unaveraged 36 km CMAQ outputs to provide a better match with the spatial resolution of the sampling data.

Key deposition metrics:
> Total Deposition ($\mu g/m^2$)
>> A.   2001 all sources (base case)
>> B.   2001 utility zero out
>> C.   2020 utilities with CAIR (2020 base case)
>> D.   2020 utilities with CAIR plus CAMR Requirements
>> E.   2020 utilities with CAIR plus CAMR Alternative
>> F.   2020 utilities zero out
>
> Change in deposition
>> A.   2020 utilities with CAIR - 2001 base case (approximate estimate of benefits of CAIR)
>> B.   2020 utilities with CAIR plus CAMR Requirements - 2020 utilities with CAIR
>> C.   2020 utilities with CAIR plus CAMR Alternative - 2020 utilities with CAIR
>> D.   2020 utilities zero out - 2020 utilities with CAIR
>
> Utility Attributable Deposition
>> A.   2001 (2001 base case - 2001 utility zero out)
>> B.   2020 CAIR (2020 utilities with CAIR - 2020 utilities zero out)
>> C.   2020 CAIR plus CAMR Requirements (2020 utilities with CAMR Requirements - 2020 utilities zero out)
>> D.   2020 CAIR plus CAMR Alternative (2020 utilities with CAMR Alternative - 2020 utilities zero out)

Maps of deposition at the HUC-8 aggregation level are provided to show the spatial pattern of deposition from all sources and the portion of deposition that is attributable to utilities after implementation of CAIR and the CAMR requirements and alternative. In addition, maps of the change in deposition associated with the CAMR requirements and alternative are provided to show the incremental impact of each. Figure 2.1 displays a map of total mercury deposition from all air sources in the 2001 base case. Figure 2.2 displays a map of the deposition in 2001 that is attributable to utilities, based on comparing CMAQ deposition estimates with and without utility emissions of mercury. This map suggests that most of the current utility attributable deposition occurs in the Eastern U.S., especially in the Ohio Valley and Pennsylvania. Figure 2.3 displays a map of projected utility attributable mercury deposition after implementation of the Clean Air Interstate Rule[4]. This map (note the change in scale) shows that after CAIR is implemented, the utility attributable mercury deposition is greatly reduced throughout the Eastern U.S., with remaining utility attributable deposition highest in parts of the Ohio Valley and around the Southern Great Lakes.
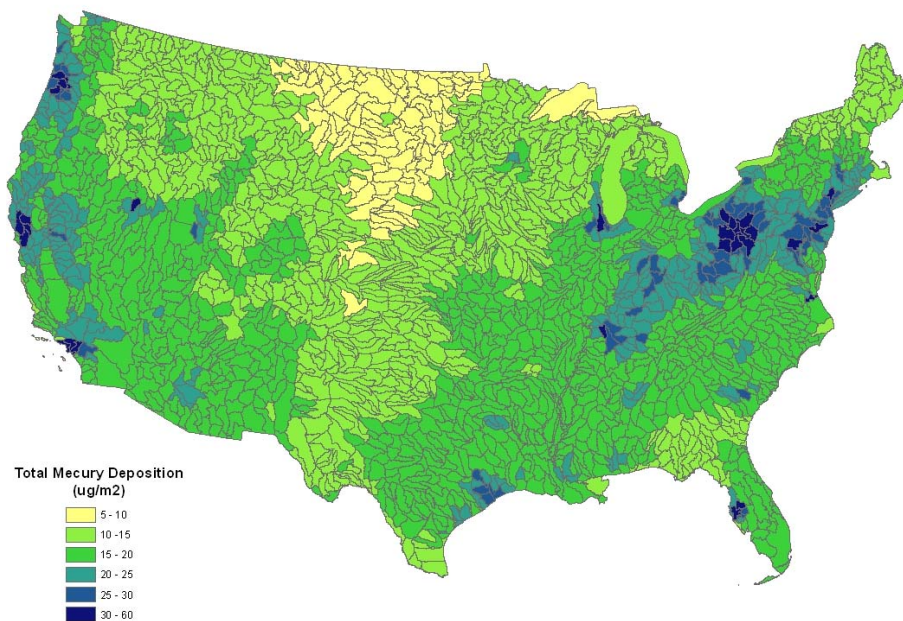

Figure 2.1. Regional Annual Deposition of Mercury in the 2001 Base Case

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[4] Note that the Clean Air Interstate Rule as promulgated does not include utilities in New Jersey, Delaware, or Arkansas. However, a rulemaking has been proposed to include these states in the CAIR region. The modeling conducted for the Clean Air Mercury Rule anticipated the inclusion of these states in the CAIR region, and thus included them in CAIR for the purpose of projecting future conditions in 2020. This may lead to a slight underestimation of utility attributable mercury deposition in the 2020 base case with CAIR. However, it is expected that implementation of the mercury cap and trade program would result in reductions in emissions such that the utility attributable deposition after implementation of both CAIR and CAMR would be the same.
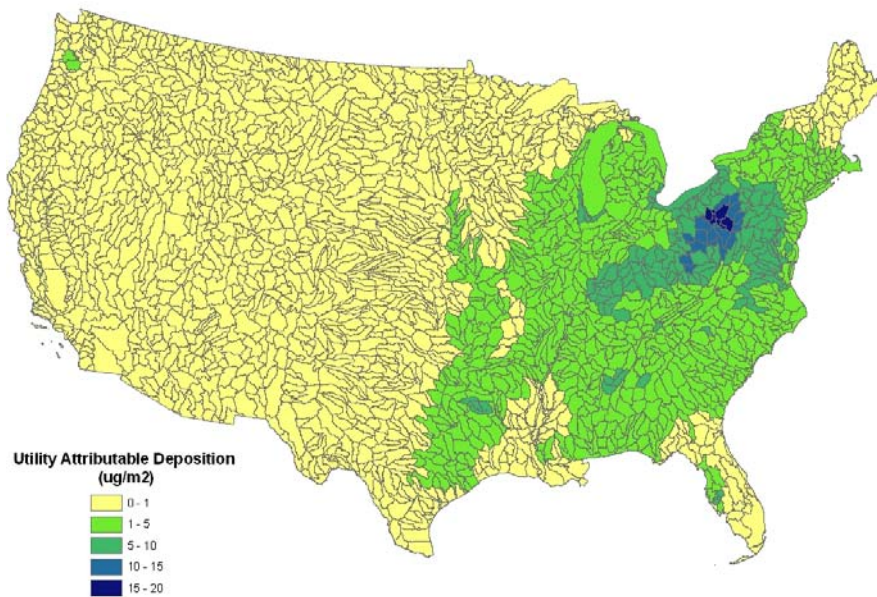
Utility Attributable Deposition
(ug/m2)
- 0 - 1
- 1 - 5
- 5 - 10
- 10 - 15
- 15 - 20

Figure 2.2. Regional Annual Deposition of Mercury Attributable to Electricity Generating Utilities in the 2001 Base Year

Utility Attributable Mecury Deposition
(ug/m2)
- 0 - 1
- 1 - 5
- 5 - 10
- 10 - 15
- 15 - 20

Figure 2.3a. Regional Annual Deposition of Mercury Attributable to Electricity Generating Utilities in the 2020 Base Case With Implementation of the Clean Air Interstate Rule (2001 Deposition Comparison Scale)

Utility Attributable Deposition (ug/m2)
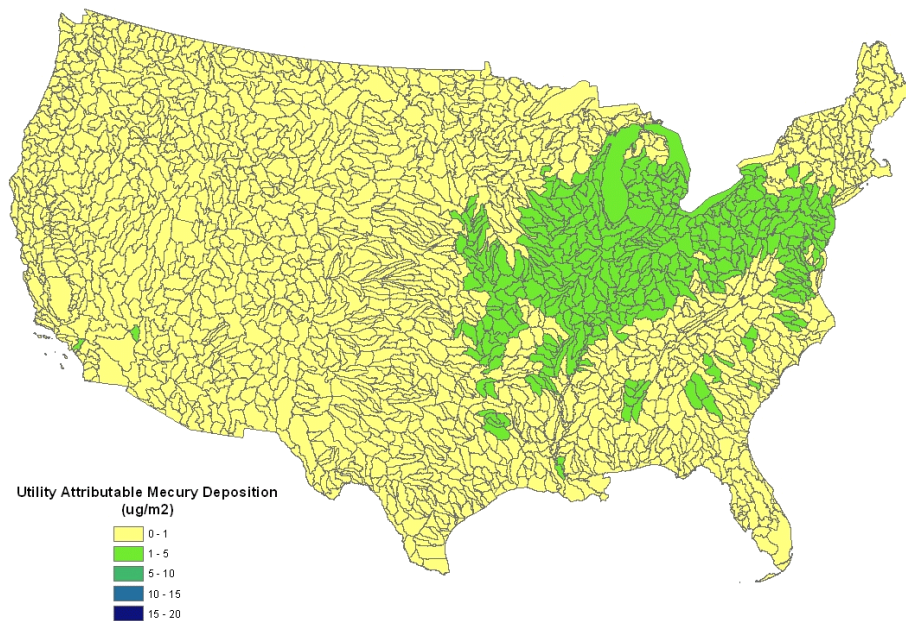- 0 - 1
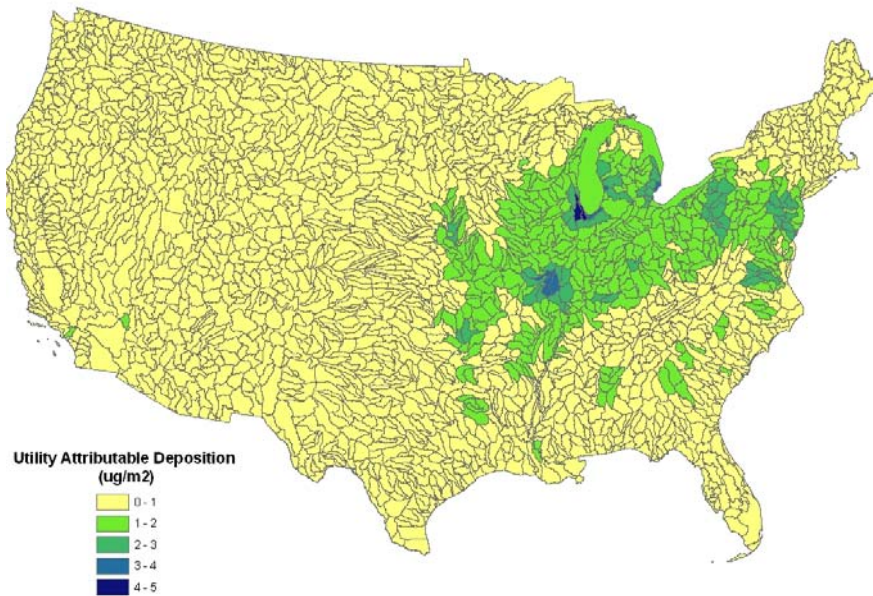- 1 - 2
- 2 - 3
- 3 - 4
- 4 - 5

Figure 2.3b. Regional Annual Deposition of Mercury Attributable to Electricity Generating Utilities in the 2020 Base Case With Implementation of the Clean Air Interstate Rule (2020 Deposition Scale)

Utility Attributable Deposition (ug/m2)
- 0 - 1
- 1 - 2
- 2 - 3
- 3 - 4
- 4 - 5

Figure 2.4. Regional Annual Deposition of Mercury Attributable to Electricity Generating Utilities in 2020 With Implementation of CAIR and the Clean Air Mercury Rule Requirement (2020 Deposition Scale)s
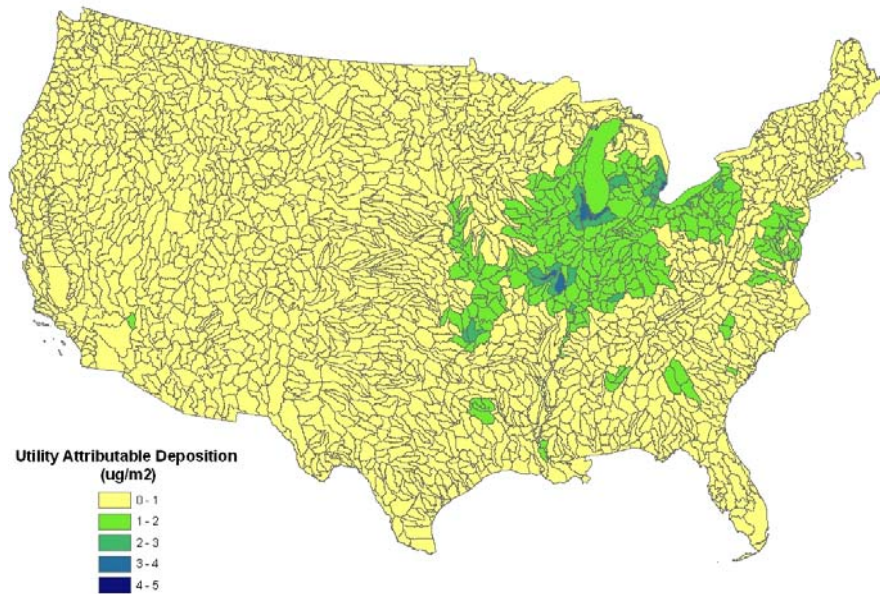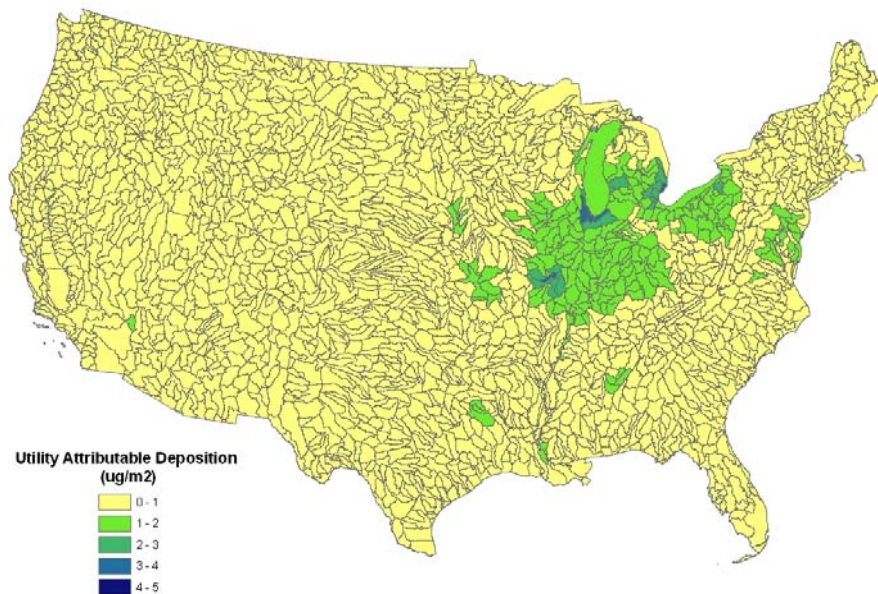
Figure 2.5. Regional Annual Deposition of Mercury Attributable to Electricity Generating Utilities in 2020 With Implementation of CAIR and the Clean Air Mercury Rule Alternative (2020 Deposition Scale)

Figure 2.6. Change in 2020 Regional Annual Deposition Due to Implementation of the

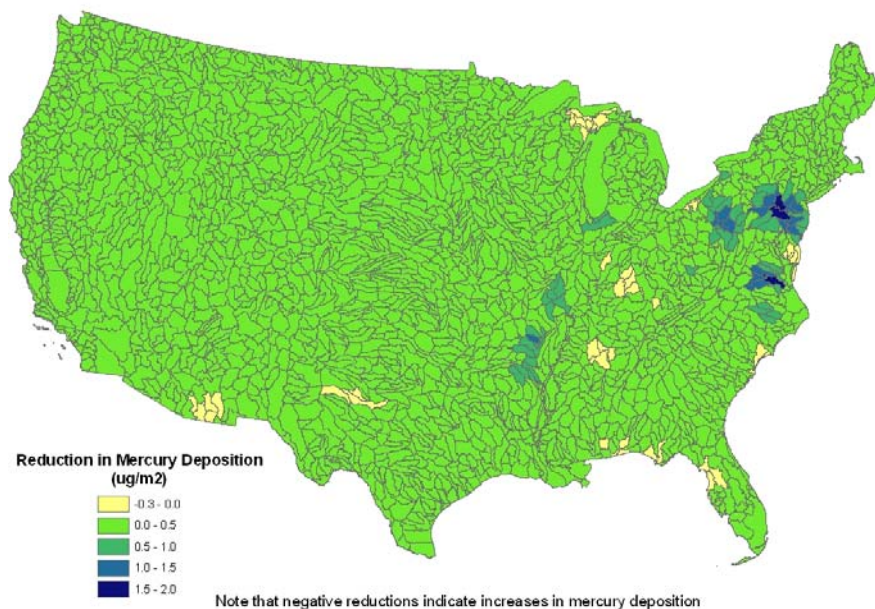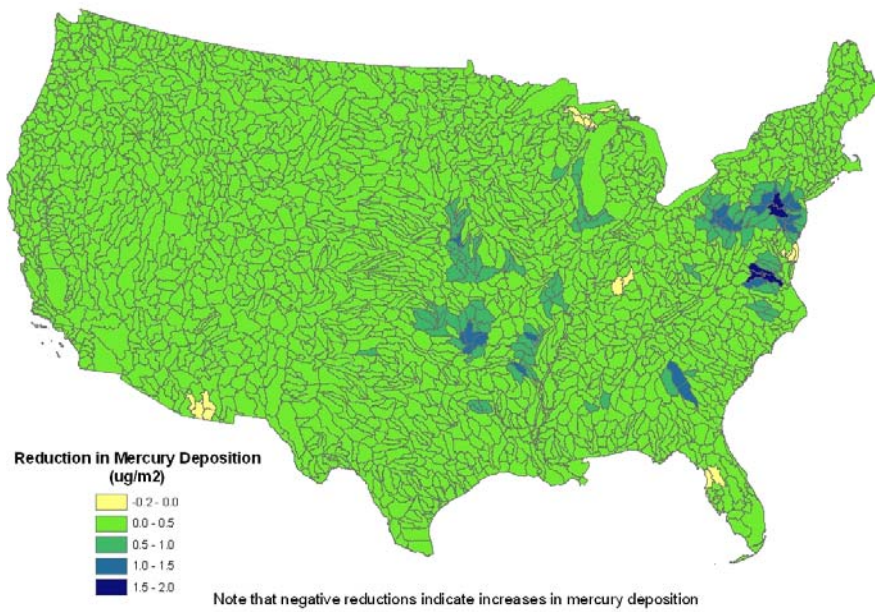**Clean Air Mercury Rule Requirements (incremental to the CAIR)**

Figure 2.7. Change in 2020 Regional Annual Deposition Due to Implementation of the Clean Air Mercury Rule Alternative (incremental to the CAIR)
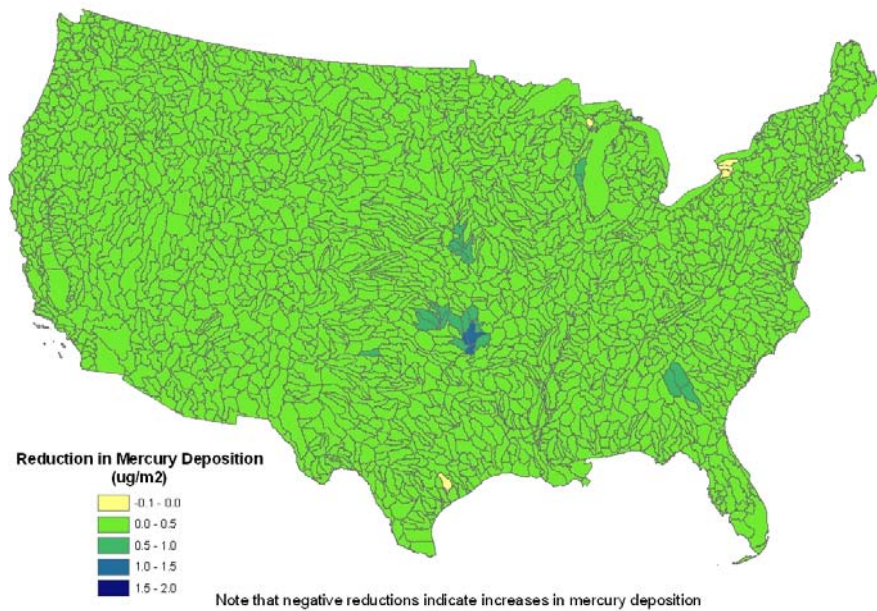

Figure 2.8. Incremental Difference in 2020 Regional Annual Mercury Deposition Between the Promulgated Clean Air Mercury Rule Requirements and the Alternative

Figure 2.4 shows the utility attributable mercury deposition remaining after implementation of Requirements for the Clean Air Mercury Rule (in addition to the CAIR). Figure 2.5 shows the utility attributable mercury deposition remaining after implementation of the CAMR alternative. To focus on the specific impacts of the CAMR relative to the 2020 base case with CAIR, Figures 2.6 and 2.7 show the projected reductions in mercury deposition from implementation of CAMR Requirements and Alternative, respectively. Figure 2.6 shows that requirements of the CAMR reduces deposition around the Great Lakes, in parts of the Ohio Valley and along the Mississippi River, and in Pennsylvania. Figure 2.7 shows that, similar to the CAMR requirements, the CAMR alternative results in reductions in deposition around the Great Lakes, the Ohio Valley, and Pennsylvania, as well as in parts of Georgia and the Midwest. Figure 2.8 shows the incremental reduction in mercury deposition comparing the CAMR requirements and CAMR alternative and that the CAMR alternative provides additional reductions in deposition primarily in the Midwest and in Georgia. However, in most HUCs, the difference between the two is very small. Note that in a few HUCs, deposition is expected to increase due to implementation of the CAMR. This occurs due to the trading provisions of the rule. However, it should also be noted that after implementation of both CAIR and CAMR, none of these HUCs have increased mercury deposition relative to the levels of utility deposition in the 2001 base case.

Cumulative distribution graphs are a useful way to summarize the overall impact of emission controls on mercury deposition throughout the U.S.   These graphs indicate the distribution of deposition across HUC-8 units. Moving from left to right on each graph indicates the cumulative percentage of HUC-8 units that have deposition less than the value on the x-axis. Figure 2.9 shows the impact of utilities on the distribution of total mercury deposition across HUC-8 units, while Figure 2.10 shows the impact of implementation of CAIR and CAMR on the distribution of utility attributable mercury deposition. Figure 2.11 show the impact of CAIR and CAMR on the distribution of the percent of mercury deposition attributable to utility mercury emissions.

We also provide tables showing specific percentiles of the cumulative distributions, and the percent of HUC-8 units that fall within particular ranges of total and utility attributable mercury deposition. These are provided in Tables 2.1 through 2.3.

Cumulative Distributions of Total Hg Deposition
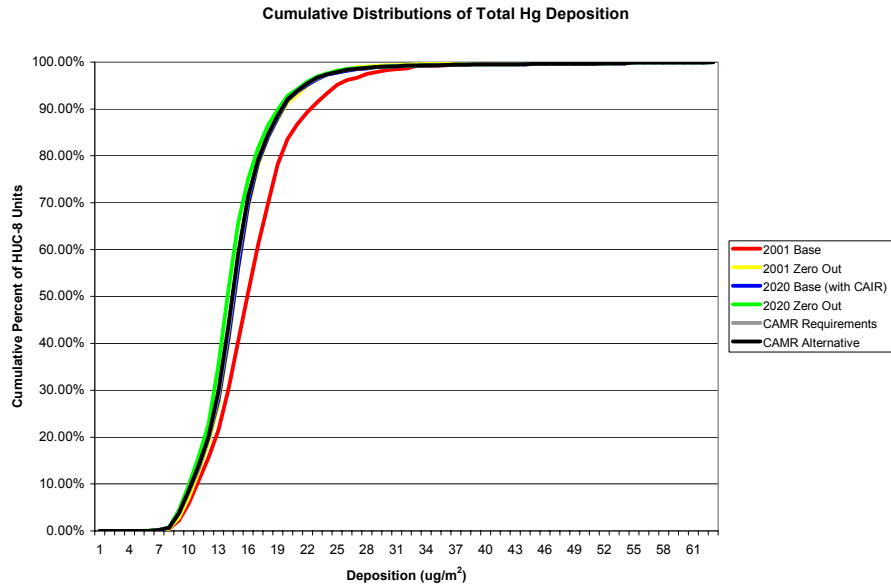
Figure 2.9. Cumulative Distributions of Total Annual Mercury Deposition (Across HUC-8 Units)

Cumulative Distributions of Utility Attributable Hg Deposition Across HUC-8 Units
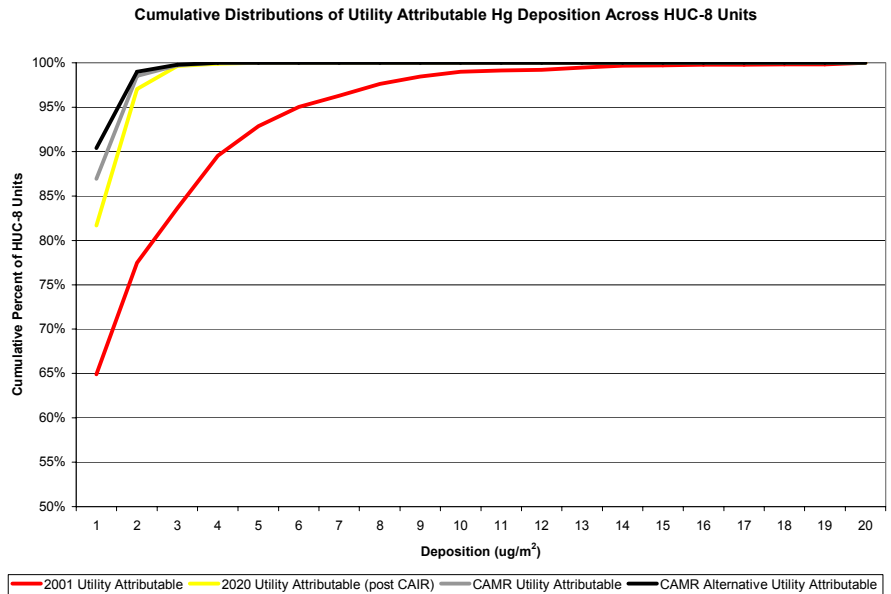

Figure 2.10. Cumulative Distributions of Annual Mercury Deposition Attributable to Utilities (Across HUC-8 Units)
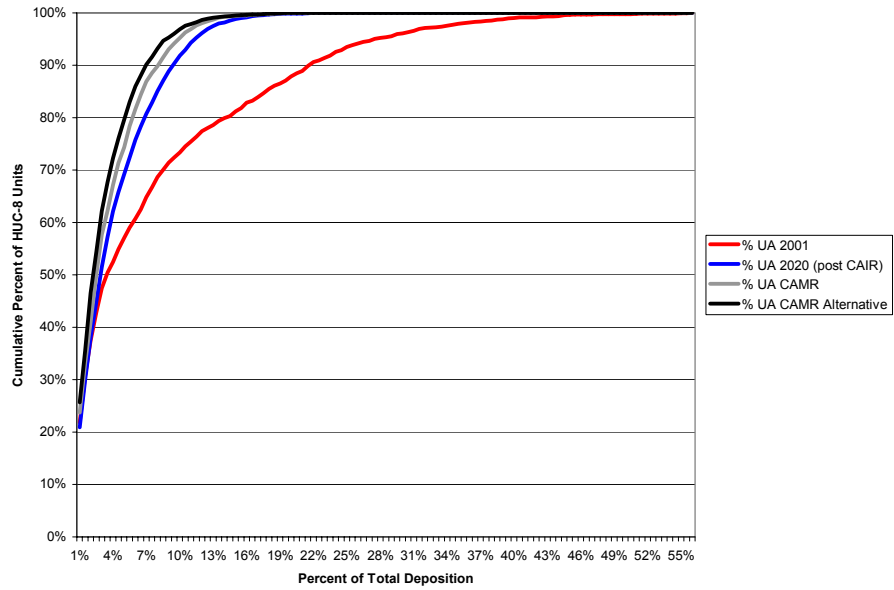
Figure 2.11. Cumulative Distributions of Percent Utility Attributable Mercury Deposition (Across HUC-8 Units)

Table 2.1. Distributions of Total Annual Mercury Deposition

Range (µg/m²)	2001 Base		2020 Base (with CAIR)		2020 CAMR Requirements		2020 CAMR Alternative	
	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %
0 - 5	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
5 - 10	5.71%	5.71%	8.26%	8.26%	8.26%	8.26%	8.49%	8.49%
10 - 15	34.93%	40.63%	47.98%	56.24%	49.98%	58.24%	50.86%	59.35%
15 - 20	43.00%	83.63%	35.64%	91.88%	33.83%	92.06%	32.71%	92.06%
20 - 25	11.50%	95.13%	5.89%	97.77%	5.80%	97.87%	5.80%	97.87%
25 - 30	3.15%	98.28%	1.30%	99.07%	1.21%	99.07%	1.21%	99.07%
Over 30	1.72%	100.00%	0.93%	100.00%	0.93%	100.00%	0.93%	100.00%

Table 2.2. Distributions of Utility Attributable Annual Mercury Deposition

Range (µg/m²)	2001		2020 (with CAIR)		2020 CAMR Requirements		2020 CAMR Alternative	
	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %
0 - 2	77.46%	77.46%	97.03%	97.03%	98.52%	98.52%	98.98%	98.98%
2 - 4	12.06%	89.52%	2.88%	99.91%	1.48%	100.00%	1.02%	100.00%
4 - 6	5.52%	95.04%	0.09%	100.00%	0.00%	100.00%	0.00%	100.00%
6 - 8	2.60%	97.63%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
8 - 10	1.35%	98.98%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
10 - 15	0.74%	99.72%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
15 - 20	0.28%	100.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
Over 20	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%

Table 2.3. Distributions of Percent Utility Attributable Mercury Deposition

Range (µg/m²)	2001		2020 (with CAIR)		2020 CAMR Requirements		2020 CAMR Alternative	
	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %
0 - 5%	59.04%	59.04%	72.36%	72.36%	78.39%	78.39%	82.98%	82.98%
5 - 10%	15.45%	74.49%	20.59%	92.95%	17.90%	96.29%	14.56%	97.54%
10 - 15%	7.33%	81.82%	6.08%	99.03%	3.20%	99.49%	2.04%	99.58%
15 - 20%	6.68%	88.50%	0.88%	99.91%	0.51%	100.00%	0.42%	100.00%
20 - 25%	5.33%	93.83%	0.09%	100.00%	0.00%	100.00%	0.00%	100.00%
25 - 30%	2.50%	96.34%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
30 - 40%	2.83%	99.17%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
40 - 50%	0.60%	99.77%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
50 - 60%	0.23%	100.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
Over 60%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%

Utility attributable mercury deposition has a relatively small effect on total mercury deposition even in the 2001 base case. After implementation of CAIR in 2020, the distribution of mercury deposition attributable to utilities shows a substantial shift to the left, indicating significant reductions in deposition attributable to utilities. The disproportionately large shift in the upper percentiles of the distribution indicates that HUCs with high levels of utility deposition are receiving a larger reduction in utility attributable mercury deposition relative to HUCs with a relatively small level of utility attributable deposition. As shown in Table 2.2, in 2001, 77 percent of HUCs have utility attributable deposition of 2 $\mu g/m^2$ or less. By 2020, after the implementation of CAIR, 97 percent of HUCs have utility attributable deposition of 2 $\mu g/m^2$ or less, and after implementation of the CAMR requirements, almost 99 percent have utility attributable deposition of 2 $\mu g/m^2$ or less. Additional reductions in mercury emissions due to the CAMR requirements or the alternative relative to CAIR result in relatively small additional shifts in the overall distribution of deposition. The incremental impact of the CAMR alternative relative to the promulgated requirements is very small.

In terms of the percent of deposition attributable to utility emissions, after implementation of CAIR, the distribution again shows a substantial shift to the left, indicating significant reductions in the number of HUCs with a large percentage of deposition attributable to utilities. As shown in Table 2.3, in 2001, 89 percent of HUCs have 20 percent or less mercury deposition attributable to utilities. In 2020, after implementation of CAIR, 99.9 percent of HUCs are projected to have 20 percent or less mercury deposition attributable to utilities, and 93 percent are projected to have 10 percent or less. Additional emissions reductions due to the CAMR requirements result in a small additional reduction in the number of HUCs with a high percentage of utility attributable emissions. Incremental impacts of the CAMR alternative relative to the promulgated requirements are small. In addition to the cumulative distribution graphs and tables, for each metric, we provide the following summary statistics: minimum, maximum, 50th percentile, 90th percentile, and 99th percentile for Total mercury (Table 2.4) and utility attributable mercury deposition (Table 2.5 and Table 2.6).

Table 2.4. Summary Statistics for Total Mercury Deposition (aggregated to the HUC-8 level)

Statistic ($\mu g/m^2$)	2001 Base Case	2001 Utility Zero Out	2020 Base Case (with CAIR)	2020 Utility Zero Out	2020 CAMR Requirements	2020 CAMR Alternative
Minimum	6.94	6.94	6.08	5.90	6.08	6.07
Maximum	54.54	54.38	62.76	62.72	62.76	62.75
50th percentile	15.92	14.60	14.59	13.92	14.44	14.39
90th percentile	22.16	19.48	19.46	19.04	19.37	19.33
99th percentile	32.35	27.20	29.15	28.93	28.96	28.95

Table 2.5. Summary Statistics for Utility Attributable Mercury Deposition (aggregated to the HUC-8 level)

Statistic ($\mu g/m^{2)}$	2001 Base Case	2020 Base Case (with CAIR)	2020 CAMR Requirements	2020 CAMR Alternative
Minimum	0.00	0.00	0.00	0.00
Maximum	19.71	4.03	3.85	3.80
50th percentile	0.39	0.31	0.26	0.22
90th percentile	4.08	1.38	1.16	0.99
99th percentile	10.15	2.56	2.17	2.04

Table 2.6. Summary Statistics for Percent Utility Attributable Mercury Deposition (aggregated to the HUC-8 level)

Statistic	2001 Base Case	2020 Base Case (with CAIR)	2020 CAMR Requirements	2020 CAMR Alternative
Minimum	0.01%	0.02%	0.01%	0.01%
Maximum	55.21%	19.21%	18.79%	18.79%
50th percentile	2.92%	2.39%	2.00%	1.69%
90th percentile	21.14%	8.88%	7.58%	6.45%
99th percentile	39.16%	14.82%	12.81%	12.10%

Summary statistics reveal very similar information. The median deposition level is reduced by only 8 percent when utilities emissions are zeroed out in 2001, suggesting that utilities are not a major source of mercury deposition in most HUCs. At HUCs with the highest deposition levels, zeroing out utilities reduces the 99th percentile deposition level by 15 percent, suggesting that there are relatively larger impacts of utilities in high deposition areas. CAIR shifts the distribution of attributable deposition significantly, resulting in a 75 percent reduction in the 99th percentile of utility attributable deposition, and a 20 percent reduction in the 50th percentile. CAMR results in small additional reductions in attributable deposition relative to 2001 levels. CAIR also shifts the distribution of percent attributable deposition. In the 2001 base case, 10 percent of HUCs had greater than 20 percent of deposition attributable to utilities. In the 2020 post-CAIR base case, no HUCs had greater than 20 percent of deposition attributable to utilities, and 90 percent had less than 9 percent of deposition attributable to utilities.

Table 2.7 presents the frequency and cumulative distributions of the reductions in deposition associated with the CAMR requirements and the CAMR alternative. We also provide the reduction in deposition from the 2020 base case with CAIR implemented relative to the 2001 base case. This change (2001 Base - 2020 CAIR) shows that there are both increases and decreases in deposition. Negative reductions (increases) are due to growth in non-utility mercury emissions, and growth in utility emissions in areas unaffected by CAIR. Reductions in deposition are largely due to the implementation of CAIR controls at utilities.

Both the promulgated CAMR requirements and the CAMR alternative result in small reductions in mercury deposition beyond CAIR. Less than 1 percent of HUCs have a reduction

in deposition due to the CAMR requirements that is greater than 1 $\mu g/m^2$. A small number of HUCs have increased mercury deposition, due to shifting of mercury emissions due to the trading provisions of the rule. The CAMR alternative results in slightly more HUCs with a greater than 1 $\mu g/m^2$ reduction in mercury deposition, and a somewhat smaller number of HUCs with increased mercury deposition. Over 97 percent of HUCs in both cases have reductions in deposition between 0 and 1 $\mu g/m^2$. Table 2.8 provides summary statistics for the change in mercury deposition for the promulgated CAMR requirements and the CAMR alternative.

Table 2.7. Distributions of Reductions in Total Mercury Deposition

Range ($\mu g/m^2$)	2001 Base - 2020 Base (with CAIR)		2020 Base (with CAIR) - 2020 CAMR Requirements		2020 Base (with CAIR) - 2020 CAMR Alternative		2020 CAMR Requirements - 2020 CAMR Alternative	
	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %
<=0	6.59%	6.59%	2.13%	2.13%	0.83%	0.83%	0.28%	0.28%
0 - 1	58.02%	64.61%	97.03%	99.17%	97.87%	98.70%	99.58%	99.86%
1 - 2	12.06%	76.67%	0.83%	100.00%	1.30%	100.00%	0.14%	100.00%
2 - 3	7.33%	84.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
3 - 4	5.10%	89.10%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
4 - 5	3.71%	92.81%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
5 - 10	6.08%	98.89%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
10 - 15	0.88%	99.77%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
15 - 20	0.23%	100.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%

Table 2.8. Summary Statistics for Reductions in Total Mercury Deposition

Statistic (($\mu g/m^2$)	2001 Base - 2020 Base (with CAIR)	2020 Base (with CAIR) - 2020 CAMR Requirements	2020 Base (with CAIR) - 2020 CAMR Alternative	2020 CAMR Requirements - 2020 CAMR Alternative
Minimum	-22.44	-0.26	-0.18	-0.06
Maximum	17.29	1.65	1.70	1.28
50th percentile	0.57	0.04	0.07	0.03
90th percentile	4.21	0.22	0.37	0.13
99th percentile	10.23	0.88	1.08	0.51

2.2 How would regional deposition results differ between 2020 and 2015?

Although EPA did not directly model the effects of CAIR on mercury deposition in 2015, the impacts are not expected to differ too much from the modeling for the 2020 baseline with CAIR. The estimated total mercury emissions of just over 34 tons from EGUs in 2015 with CAIR will be virtually the same as the estimated total in 2020 with CAIR. The readily deposited non-elemental mercury emissions from EGUs are estimated to be 10 tons in 2015 but roughly 9 tons in 2020. The non-elemental mercury emissions consists of the sum of the reactive gaseous mercury and particulate mercury species. It could be inferred that although 2015 was not modeled here that the mercury deposition levels that are estimated to occur with CAIR in 2020 as shown in Figure 2.3 are similar to those that would occur in 2015 with CAIR. Thus, the difference in mercury deposition from 2001 to CAIR in 2020 as shown in Figures 2.9 and 2.10

should also be indicative of the change between 2001 and CAIR in 2015. The similarity between these scenarios will depend upon the following three
factors:

• The spatial distribution of mercury emissions reductions across these years will influence the spatial nature of mercury deposition. Despite the fact that the level of total mercury emissions is virtually the same for both years, the spatial distribution of these emissions reductions across EGUs will likely differ between 2015 and 2020. This difference should lead to the spatial coverage of reductions in mercury deposition to be somewhat less in 2015 than 2020 similar to the reduced spatial coverage observed in the modeling for the 2020 baseline with CAIR compared to 2020 CAMR options.

• The level of mercury emissions reductions by species across these years would effect the modeled levels of mercury deposition. Despite the fact that the level of total mercury emissions is virtually the same for both years, the more readily deposited non-elemental emissions are different by roughly 10 percent, or 10 vs 9 tons respectfully for 2015 and 2020. The mercury emissions from sectors other than EGUs are also expected to be different between these years. In addition to the spatial differences, these differences in emissions will contribute to an undetermined difference in the spatial coverage of mercury deposition reductions in 2015 than 2020.

• The levels of criteria pollutant emissions are different across these years and would effect the mercury deposition through the atmospheric reactions accounted for by CMAQ. However, the potential for these interactions to cause notable differences is limited as the emission differences are not significant enough for these interactions to be more than a second-order impact.

3. Analysis of Fish Tissue Methylmercury Concentrations

While deposition changes will occur immediately after reductions in emissions, changes in fish tissue methylmercury concentrations will not change immediately. Case studies of individual ecosystems show that the time necessary for aquatic systems to reach a new steady state after a reduction in mercury deposition rates can be as short as 5 years or as long as 50 years or more. The medium response scenarios also varied widely but were generally on the order of one to three decades. Overall, we conclude that the most likely appropriate response times for freshwater ecosystems to be considered in the national scale assessment range between five and 30 years, while recognizing that some systems will likely take more than 50-100 years to reach steady state. It is important to note that the analysis presented in this document does not account for this lag. Results for methylmercury concentrations and corresponding exposure estimates should be interpreted being at steady state, not in any particular year.

Fish tissue methylmercury concentrations were developed using fish tissue samples collected as part of two different sampling programs, the National Listing of Fish Advisories (NLFA) and the National Lake Fish Tissue Survey (NLFTS). These two databases combined

provide the largest existing database of fish tissue mercury levels. The NLFA collects data from state sampling of fish (U.S. EPA, 2004a). According to the EPA NLFA website (http://www.epa.gov/ost/fish/advisories), states monitor their waters by sampling fish tissue for long-lasting pollutants that bioaccumulate. States issue their advisories and guidelines voluntarily and have flexibility in what criteria they use and how the data are collected. As a result, there are significant variations in the numbers of waters tested, the pollutants tested for, and the threshold for issuing advisories. Based on self-reporting, the national trend is for states to monitor different waters each year, generally without retesting waters monitored in recent years. State sampling programs are based on different sampling criteria and may include samples from locations suspected to be contaminated by mercury from a variety of sources, including non-air deposition sources such as gold or mercury mines, as well as industrial locations such as chloralkalai plants[5]. In general, the States historically sampled waterbodies in areas of suspected contamination. More recently, States also have focused sampling efforts on areas of elevated fishing pressure.

The NLFTS is an ongoing program intended to provide a more geographically representative set of fish tissue data at lakes and reservoirs in the U.S. (U.S. EPA, 2004b,c)[6]. The NLFTS provides samples from 500 randomly selected lakes and reservoirs in the U.S. during the period 2000-2003. The NLFTS excluded the great lakes, and randomly selected lake sites from over 270,000 possible sites. The sampling design used a geographically stratified design, and oversampled large lakes to avoid a preponderance of small lakes in the sample. The NLFTS is a carefully designed sampling protocol and includes consistent sampling methods and quality assurance procedures.

The NLFA database includes data collected from 1990 to 2003, while the NLFTS includes data collected from 2000 to 2003. Given the significant changes in emissions of mercury between 1990 and 1999 due to existing Clean Air Act programs (reductions of almost 50%) and the potential for changes in the methylation rates due to changes in lake acidity affected by the acid rain program, there could be significant differences in the mercury concentrations found in fish sampled prior to 1999 and fish sampled after 1999[7]. Given that

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[5] For example, the Pennsylvania Department of Environmental Protection lists the following protocol for determining sample locations: "Stations are chosen based on: 1) the need for verification (second) samples at selected sites for possible new advisories or de-listings, 2) the demand of the Water Quality Network (WQN) rotation, and 3) the need to follow-up on existing advisories. In addition, DEP issues a request for suggested sampling station locations and target species to DEP Regional Biologists, PA Fish and Boat Commission Area Fisheries Mangers (AFMs) and the Erie County Department of Health (ECDH). (Pennsylvania Department of Environmental Protection, 2004)" The Georgia Department of Natural Resources identifies as a key objective the need "to identify areas where fish tissue contamination may present a health or environmental risk. To satisfy this objective, sampling sites should target areas suspected of having high contamination. (Fish Tissue Advisory Committee, 1992)"

[6] For more information, visit the EPA NLFTS website at http://epa.gov/waterscience/fishstudy/

[7] In most locations, it is not possible to test for this type of trend effect, because many states do not resample locations on a regular basis. As such, for many sampling sites, there is only one or a few sampling dates.

CAIR and CAMR rule provisions will not go into effect until 2010, we determined that using the more current data on fish tissue concentrations will provide a more reasonable characterization of the current and future mercury risks attributable to utility Mercury emissions. As such, we use the NLFTS and NLFA samples collected from 1999 forward for the majority of the analyses. However, we recognize that because many states do not frequently resample locations, excluding samples collected prior to 1999 will result in some loss of spatial coverage. To examine the potential impact of this loss in spatial coverage, we also present several sensitivity analyses using the full set of NLFA data collected between 1990 and 2003.

Many samples in the NLFA databases are for inedible fish, either due to the species type or due to the size of the sampled fish. In order to prevent data artifacts associated with using samples from fish that are of a size that is rarely consumed, we filtered the NLFTS and NLFA data further to remove samples for fish under 7 inches in length, which we determined to be a reasonable edible size. We also excluded saltwater species and crustaceans. These species were excluded due to concerns about the methods to attribute methylmercury in these species to utility mercury emissions. We recognize that all of the filtering criteria will reduce the geographic coverage of the data, however, it will ensure that only the most reliable data are used in determining the impacts of the rule.[8]

For included locations, samples for the same species are averaged across all available years (post 1998), and then the highest averaged per species concentration is used to represent the methylmercury concentration for that sample location. For example, if there are two species at a location, walleye and pike, with three sampling dates for each species, we would first average over the three sample dates for each species, and then select walleye if the average for walleye is highest, or select pike if the average for pike is highest. To the extent that an angler consumes several different species or focuses on a fish type that does not have the highest concentrations, exposure will be overestimated. This algorithm is used to reflect the variability in methylmercury within a species at a particular location, and to allow for the possibility that some anglers may prefer to catch and consume one species of fish over another.   Assignment of the maximum average species concentration recognizes the greater risk to an individual consuming species with higher accumulation of mercury while respecting the fact that each sample for an individual species is only an estimate of the true mean concentration in that species.

The NLFTS and NLFA data were used to determine the 2001 base case methylmercury levels. Methylmercury levels for other scenarios were calculated as the 2001 base case multiplied by the ratio of deposition in the scenario to deposition in the base case. For 2020 control options, this indicates a series of multiplications, for example methylmercury in the CAMR Requirements case is calculated as:

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[8] Note that sensitivity analyses using the full 1990 - 2003 database did not lead to substantially different results. There were a few sample locations that had higher methylmercury values, however, these were determined to be points that were likely heavily influenced by non-air sources. See all_fish_sample_location_analysis.xls in the docket for this rule for more information on the sensitivity analysis.

$$MeHg\ CAMR = \left(MeHg\ 2001\ Base \right) \times \left(\frac{2020\ Base\ Deposition\ with\ CAIR}{2001\ Base\ Deposition} \right) \times \left(\frac{2020\ CAMR\ Deposition}{2020\ Base\ Deposition\ with\ CAIR} \right)$$

In assigning deposition levels to sample locations, we used the predicted CMAQ deposition for the 36 km grid cell containing the sample location. We use the CMAQ results in this case rather than the HUC averages because of the point nature of the sampling data. Using a HUC average would tend to limit the extremes of deposition, and given that many waterbodies where samples were taken are much smaller than 36 km grid size, it is more appropriate in this case to match the scale of the waterbody with the scale of the modeling grid. This does not imply that the deposition at any specific water body should be used to characterize regional deposition patterns (see previous section).

The scaling method assumes a proportional relationship between reductions in air deposition of mercury and methylmercury concentrations in fish. This relationship has been documented in the Mercury Maps Approach (Cocca, 2001). Mercury Maps implements a simplified form of the IEM-2M model applied in EPA's Mercury Study Report to Congress (USEPA, 1997b).[9] By simplifying the assumptions inherent in the freshwater ecosystem models that were described in the Report to Congress, the Mercury Maps model showed that these models converge at a steady-state solution for methylmercury concentrations in fish that are proportional to changes in mercury inputs from atmospheric deposition, (e.g., over the long term, fish concentrations are expected to decline proportionally to declines in atmospheric loading to a waterbody). This solution only applies to situations where air deposition is the only significant source of mercury to a water body, and the physical, chemical, and biological characteristics of the ecosystem remain constant over time. EPA recognizes that concentrations of methylmercury in fish across all ecosystems may not reach steady state and that ecosystem conditions affecting mercury dynamics are unlikely to remain constant over time. EPA further recognizes that many water bodies, particularly in areas of historic gold and mercury mining in western states, contain significant non-air sources of mercury. In addition, some areas have soils with relatively high mercury levels that contribute to mercury levels in waterbodies. Finally, EPA recognizes that Mercury Maps does not provide for a calculation of the time lag between a reduction in mercury deposition and a reduction in the methymercury concentrations in fish. Despite these limitations, EPA is unaware of any other tool for performing a national-scale assessment of the change in fish methylmercury concentrations resulting from reductions in atmospheric deposition of mercury.

As is stated above, the relationship between changes in mercury deposition from air to the change in fish tissue concentration holds only when air deposition is the predominant source of the mercury load to a waterbody. Due to this requirement in the model, the national application of the Mercury Maps approach used for the benefits assessment screened out watersheds in which sources of mercury other than air deposition were significant. These

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[9] Note that SERAFM described elsewhere also employs a similar simplifying assumption that, with all other conditions remaining constant, methylmercury concentrations will respond proportionally to deposition at steady state.

watersheds were not screened out for the purposes of the analysis presented in this document. This may result in higher concentrations of methylmercury in fish being attributed to power plants than would be the case had we been able to account for the non-air sources. This bias is in the direction of overestimating exposure attributable to power plants and therefore is a conservative simplification.

Key fish tissue methymercury metrics:

      A.   Total methylmercury concentrations
            A.   2001 all sources (base case)
            B.   2001 utility zero out
            C.   2020 utilities with CAIR (2020 base case)
            D.   2020 utilities with CAIR plus CAMR Requirements
            E.   2020 utilities with CAIR plus CAMR Alternative
            F.   2020 utilities zero out

      B.   Total methylmercury concentration attributable to utilities
            A.   2001 (2001 base case - 2001 utility zero out)
            B.   2020 CAIR (2020 utilities with CAIR - 2020 utilities zero out)
            C.   2020 CAIR plus CAMR Requirements (2020 utilities with CAMR Requirements - 2020 utilities zero out)
            D.   2020 CAIR plus CAMR Alternative (2020 utilities with CAMR Alternative - 2020 utilities zero out)

      C.   Change in methylmercury
            A.   2020 utilities with CAIR - 2001 base case (approximate estimate of benefits of CAIR)
            B.   2020 utilities with CAIR plus CAMR Requirements - 2020 utilities with CAIR
            C.   2020 utilities with CAIR plus CAMR Alternative - 2020 utilities with CAIR
            D.   2020 utilities zero out - 2020 utilities with CAIR

Fish Sampling Site Maps

      Maps of fish tissue methylmercury concentrations at sampling locations are provided (Fig. 3.1 - 3.3) to show the geographic coverage of sampling sites and spatial patterns in methylmercury concentrations. Maps and analysis are provided for the post-1999 combined NLFTS and NLFA database and for the estimated utility attributable fish tissue methylmercury concentrations in 2001, including after the implementation of CAIR and CAMR.

      Figure 3.1 displays a map of the methylmercury concentrations in fish at sampling locations based on the 1999-2003 NLFTS and NLFA sampling data. This map shows that, based on the sampled data, many locations currently have methylmercury levels above the EPA water

quality criterion of 0.3 ppm (mg/kg). Figure 3.2 displays a map of methylmercury concentrations in fish that are attributable to utility emissions of mercury in the 2001 base year. This map was derived by reducing the sampled methylmercury concentrations by the ratio of mercury deposition with utility emissions to mercury deposition without utility emissions. This map shows that current utility mercury emissions may contribute more than the water quality criterion level (0.3 ppm) to total fish tissue methylmercury concentrations in a few locations in South Carolina, New York, and Ohio. Most locations have utility attributable methylmercury concentrations well below the 0.3 ppm (mg/kg) water quality criterion. Figure 3.3 displays a map of methylmercury concentrations in fish that are attributable to utility emissions of mercury in 2020, after implementation of the Clean Air Interstate Rule, which achieves a substantial reduction in mercury emissions from utilities. This map was derived by reducing the sampled methylmercury concentrations by the ratio of mercury deposition in 2001 to mercury deposition in 2020 (with CAIR implemented). This map shows that after implementation of CAIR in 2020, there are no sample locations with utility attributable methylmercury concentrations above 0.3 ppm (mg/kg). Significant reductions in utility attributable methylmercury occur in the Ohio Valley, the Northeast, and parts of South Carolina.

Cumulative distribution graphs

These graphs indicate the distribution of methylmercury fish tissue concentrations across sampling locations. Moving from left to right on each graph indicates the cumulative percentage of sampling locations that have methylmercury concentrations less than the value on the x-axis. These graphs are provided only for the filtered set of post-1999 NLFA and NLFTS samples. Figure 3.4 shows the impact of utilities on the distribution of total fish tissue methylmercury concentration across sampling locations, while Figure 3.5 shows the impact of implementation of CAIR and CAMR on the distribution of fish tissue methylmercury attributable to utility mercury emissions.

We also provide tables (Tables 3.1 and 3.3) showing specific percentiles of the cumulative distributions, and the percent of sample locations that fall within particular ranges of total and utility attributable methylmercury concentrations. In addition to the cumulative distribution graphs and tables, for each metric, we provide the following summary statistics: minimum, maximum, 50th percentile, 90th percentile, and 99th percentile. Statistical Analysis: For each metric, we provide the following summary statistics: minimum, maximum, 50th percentile, 90th percentile, and 99th percentile (Tables 3.2 and 3.4). Together, these statistics provide a relatively complete picture of the expected impacts of the CAIR and CAMR regulations on fresh water fish tissue concentrations of methylmercury in the U.S.

Fish Tissue Methylmercury (ppm)

- 0.0 - 0.3
- 0.3 - 0.5
- 0.5 - 1.0
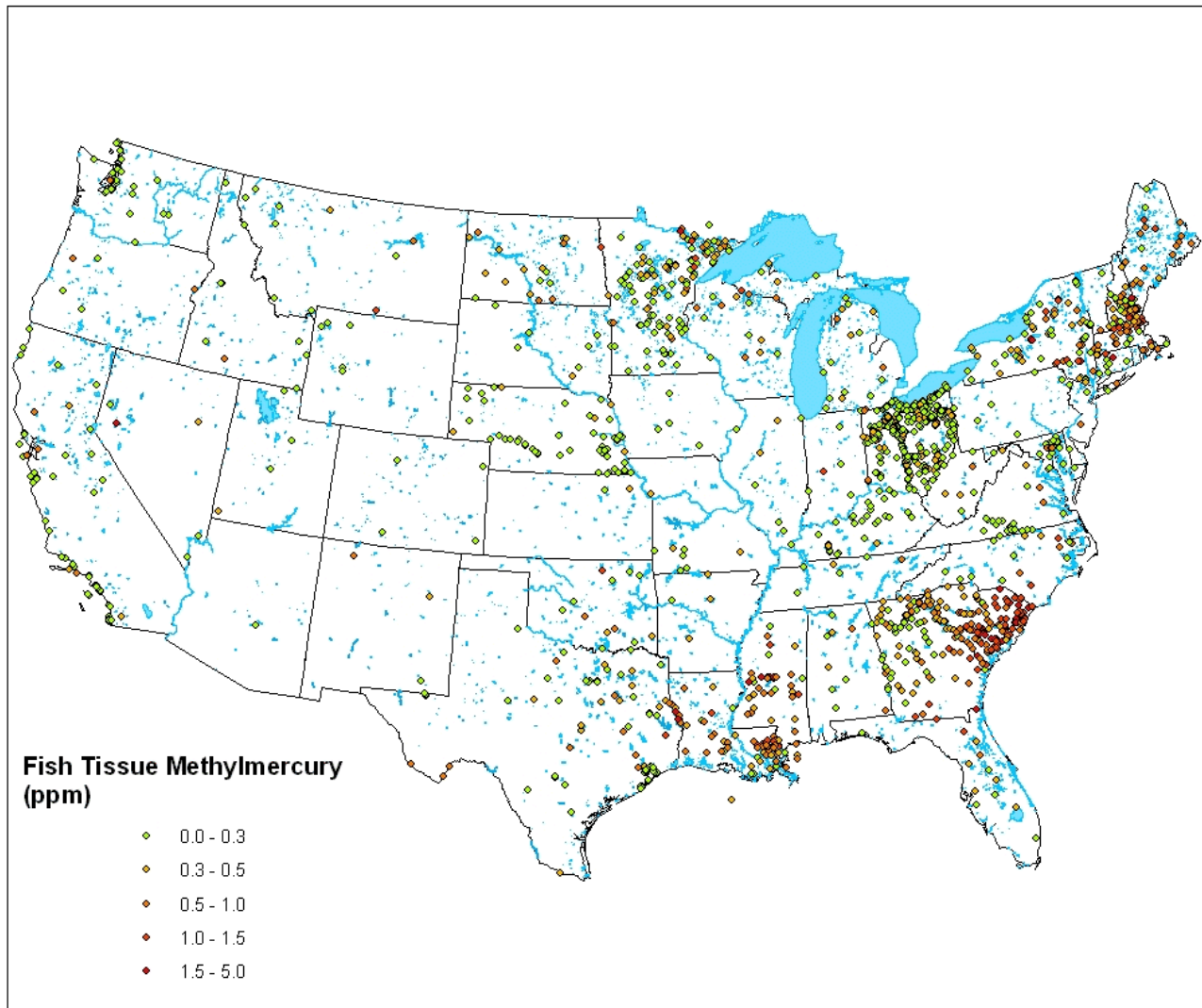- 1.0 - 1.5
- 1.5 - 5.0

Figure 3.1. Fish Tissue Methylmercury Concentrations at NLFA and NLFTS Sampling Locations (1999-2003 samples)

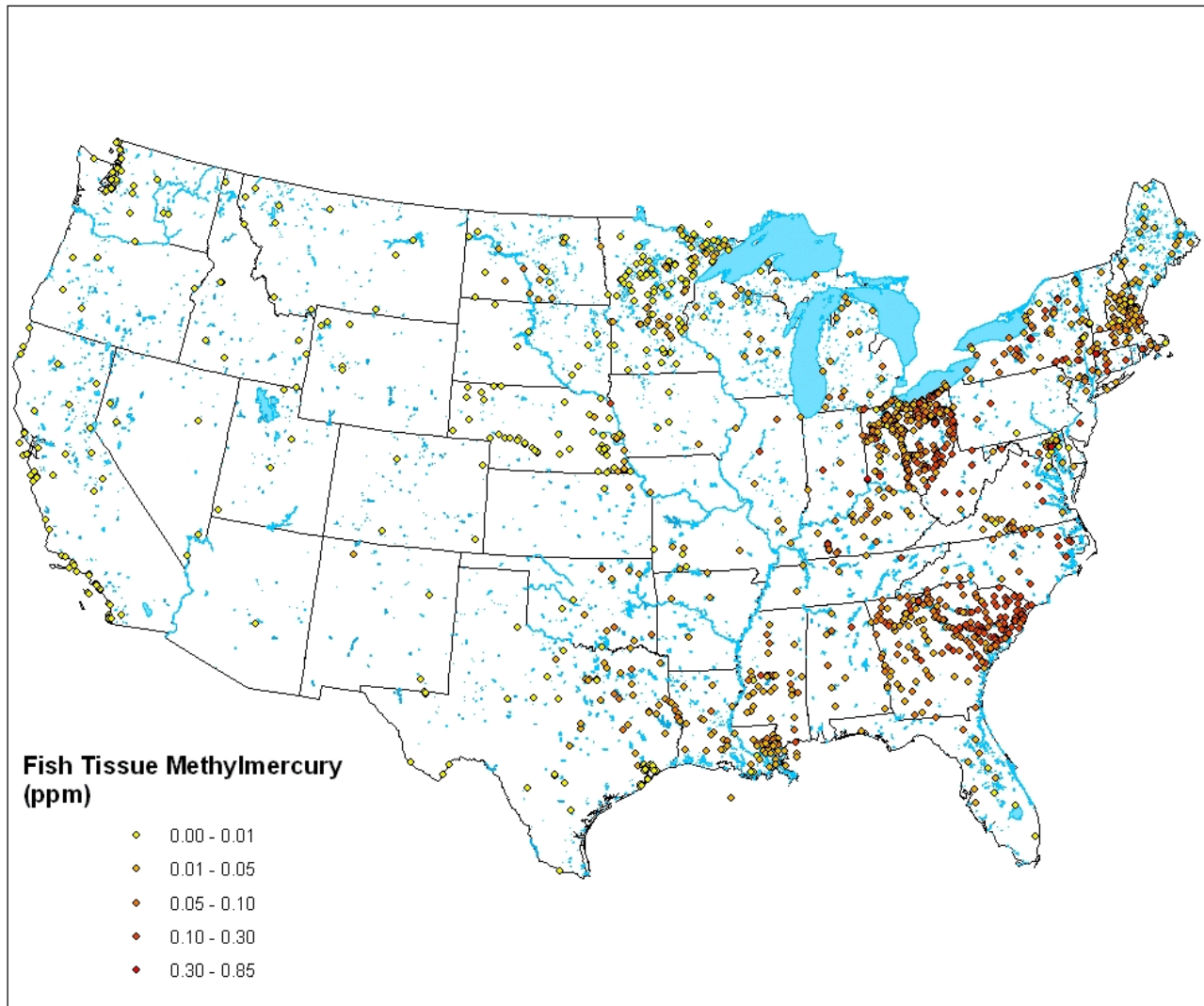**Figure 3.2. Fish Tissue Methylmercury Concentrations Attributable to Utilities in 2001 Base Year**

Fish Tissue Methylmercury (ppm)

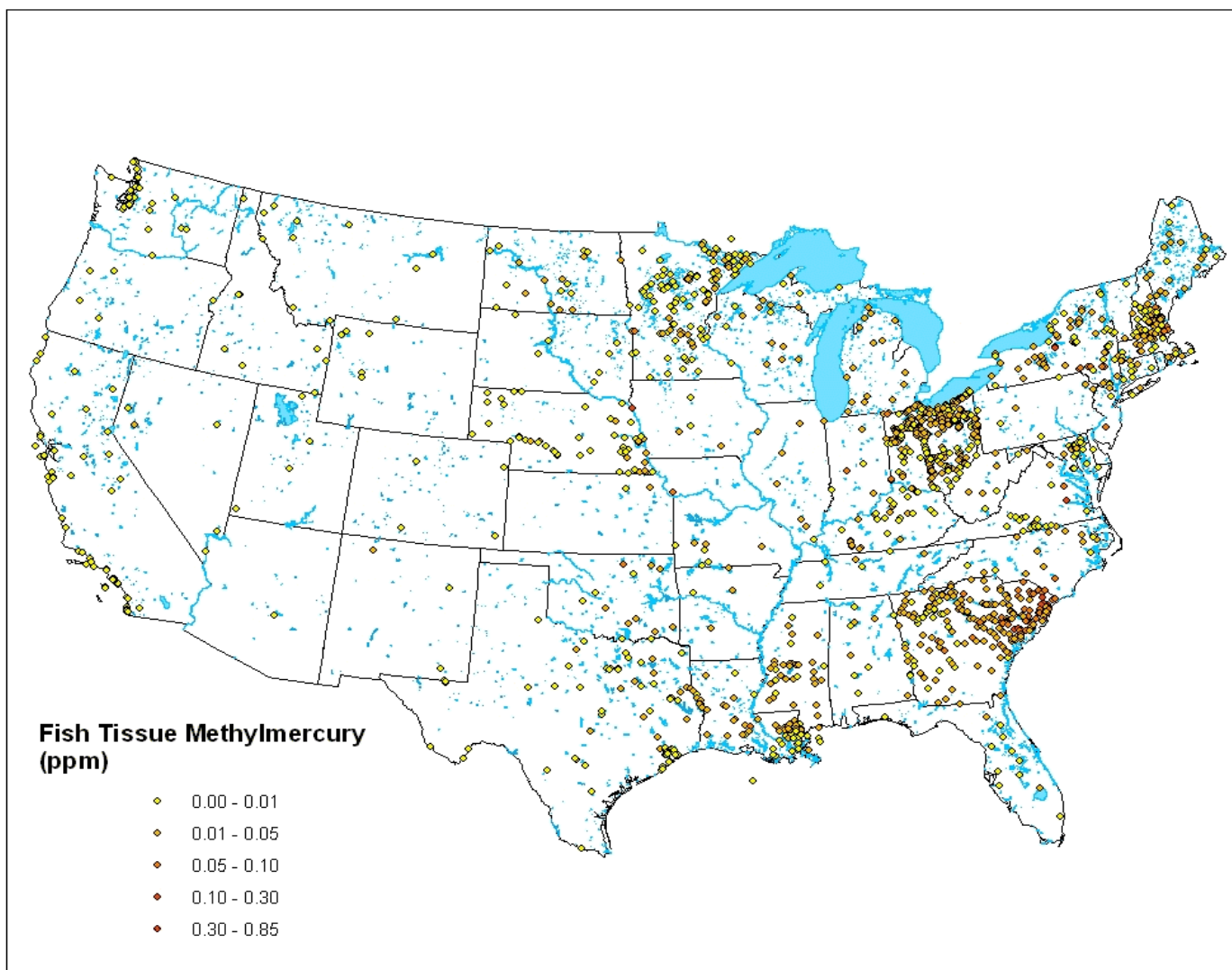- 0.00 – 0.01
- 0.01 – 0.05
- 0.05 – 0.10
- 0.10 – 0.30
- 0.30 – 0.85

Figure 3.3. Fish Tissue Methylmercury Concentrations Attributable to Utilities in 2020 After Implementation of the Clean Air Interstate Rule

Cumulative Distributions of Fish Tissue Concentrations Across Sampling Locations
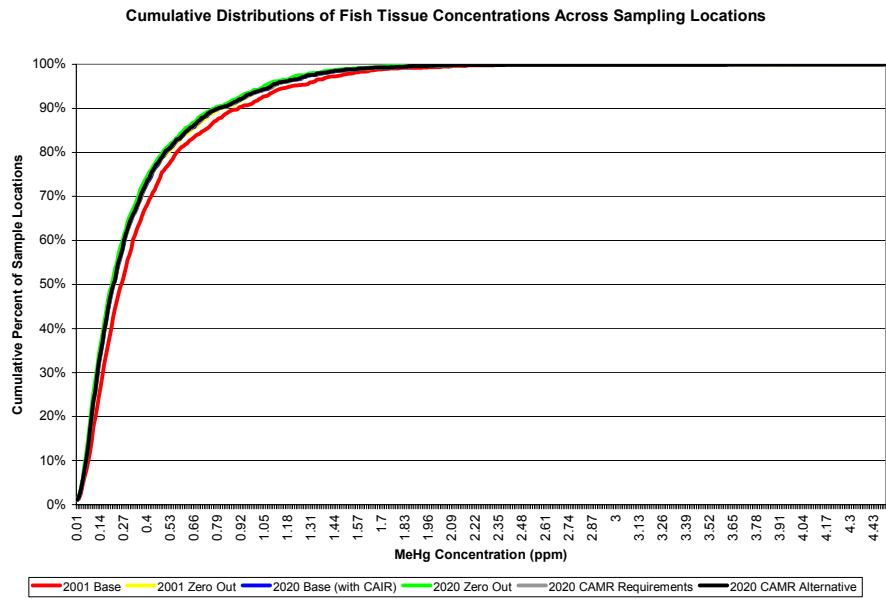

Figure 3.4. Cumulative Distributions of Total Fish Tissue Methylmercury (Across Sampling Locations)

Table 3.1. Distributions of Total Fish Tissue Methlymercury

Range (mg/kg)	2001 Base		2020 Base (with CAIR)		2020 CAMR Requirements		2020 CAMR Alternative	
	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %
0.10 - 0.19	17.6%	17.6%	23.2%	23.2%	23.3%	17.6%	23.5%	23.5%
0.20 - 0.29	22.7%	40.3%	24.9%	48.1%	25.0%	40.3%	25.0%	48.5%
0.30 - 0.39	16.4%	56.7%	15.5%	63.6%	15.7%	56.7%	15.6%	64.1%
0.40 - 0.49	11.5%	68.2%	9.6%	73.1%	9.5%	68.2%	9.6%	73.6%
0.50 - 0.59	8.3%	76.5%	6.8%	79.9%	6.8%	76.5%	6.7%	80.3%
1.00 - 1.99	15.1%	91.6%	13.8%	93.6%	13.6%	91.6%	13.4%	93.8%
2.00 - 2.99	7.7%	99.3%	6.1%	99.7%	5.9%	99.3%	6.0%	99.8%
3.00 - 4.99	0.6%	99.9%	0.3%	99.9%	0.3%	99.9%	0.2%	99.9%
5.00 - Max	0.1%	100.0%	0.1%	100.0%	0.1%	100.0%	0.1%	100.0%

Table 3.2. Summary Statistics for Total Fish Tissue Methylmercury (Sample Locations)

Statistic (mg/kg)	2001 Base Case	2001 Utility Zero Out	2020 Base Case	2020 Zero Out	2020 CAMR Requirements	2020 CAMR Alternative
Minimum	0.00	0.00	0.00	0.00	0.00	0.00
Maximum	4.49	3.64	3.65	3.46	3.63	3.61
50th percentile	0.25	0.21	0.21	0.20	0.21	0.21
90th percentile	0.90	0.81	0.79	0.77	0.79	0.78
99th percentile	1.80	1.65	1.64	1.57	1.63	1.63

Consistent with deposition patterns, utilities emissions of mercury have relatively little impact on overall fish tissue concentrations in the 2001 base case for the sample locations where recent samples have been collected. Zeroing out all utility emissions in 2001 results in a less than 10 percent decrease in the 99th percentile fish tissue concentration and less than a 17 percent decrease in median fish tissue concentrations. As such, reductions in utility emissions do not affect the overall distribution of fish tissue methylmercury concentrations. At the 2020 base case, which includes reductions in utility mercury emissions due to CAIR, the 99th percentile methylmercury concentration has been reduced by 10 percent, and the median methylmercury concentration has been reduced by 17 percent. Additional reductions due to either CAMR requirements or the CAMR alternative result in small additional shifts in the distribution of methylmercury fish tissue concentrations.
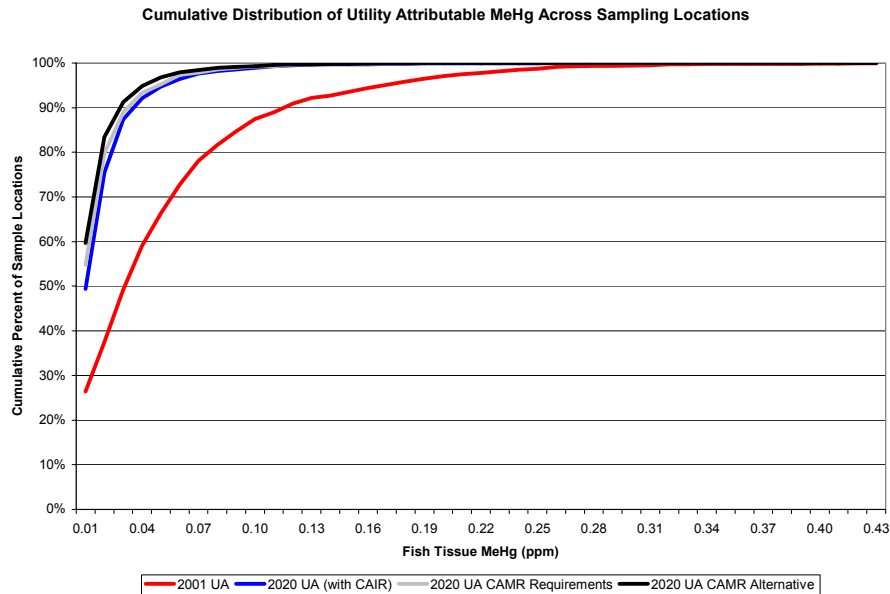
Cumulative Distribution of Utility Attributable MeHg Across Sampling Locations

Figure 3.5. Cumulative Distributions of Utility Attributable Fish Tissue Methylmercury (Across Sampling Locations)

Table 3.3. Distributions of Utility Attributable Fish Tissue Methylmercury Concentrations

Range (mg/kg)	2001 Percent	2001 Cumulative %	2020 (with CAIR) Percent	2020 (with CAIR) Cumulative %	2020 CAMR Requirements Percent	2020 CAMR Requirements Cumulative %	2020 CAMR Alternative Percent	2020 CAMR Alternative Cumulative %
0.0 - 0.1	87.50%	87.50%	98.94%	98.94%	99.19%	99.19%	99.25%	99.25%
0.1 - 0.2	9.56%	97.06%	1.00%	99.94%	0.81%	100.00%	0.75%	100.00%
0.2 - 0.3	2.38%	99.44%	0.06%	100.00%	0.00%	100.00%	0.00%	100.00%
0.3 - 0.4	0.44%	99.88%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
0.4 - 0.5	0.06%	99.94%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
More	0.06%	100.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%

Table 3.4. Summary Statistics for Utility Attributable Fish Tissue Methylmercury (Across Sampling Locations)

Statistic (mg/kg)	2001 Base	2020 (with CAIR)	2020 CAMR Requirements	2020 CAMR Alternative
Minimum	0.00	0.00	0.00	0.00
Maximum	0.85	0.25	0.19	0.18
50th percentile	0.03	0.01	0.01	0.01
90th percentile	0.11	0.03	0.03	0.03
99th percentile	0.26	0.10	0.09	0.08

CAIR results in a substantial shift in the distribution of utility attributable fish tissue

concentrations, resulting in a 60 percent reduction in the 99th percentile attributable fish tissue concentration and a 67 percent reduction in the 50th percentile attributable fish tissue concentration (Table 3..3). However, it should be noted that this results in only a 0.02 ppm (mg/kg) reductions in median fish tissue concentrations. Additional reductions due to either the promulgated CAMR requirements or the CAMR alternative result in small additional reductions in median utility attributable concentrations (Table 3.4), however, the CAMR requirements result in an additional 9 percent reduction in the 99th percentile concentration and the CAMR alternative results in an additional 20 percent reduction in the 99th percentile concentration. However, this equates to only a 0.01 ppm (mg/kg) and 0.02 ppm (mg/kg) reduction for the promulgated CAMR requirements and the alternative, respectively.

Certain important findings can be drawn from this analysis. These include:

- No sample locations have utility attributable fish tissue concentrations exceeding 0.3 ppm (mg/kg) after CAIR in 2020.
- Less than 1 percent of locations have utility attributable fish tissue concentrations exceeding 0.1 ppm (mg/kg) after CAIR in 2020.
- Less than 10 percent of locations have utility attributable fish tissue concentrations exceeding 0.05 ppm (mg/kg) after CAIR in 2020.

Table 3.5 presents the frequency and cumulative distributions of the reductions in fish tissue methylmercury concentrations associated with the CAMR requirements and the CAMR alternative. We also provide the reduction in methylmercury concentrations from the 2020 base case with CAIR implemented relative to the 2001 base case. This change (2001 Base - 2020 CAIR) shows that there are both increases and decreases in methylmercury concentrations. Negative reductions (increases) are due to growth in non-utility mercury emissions, and growth in utility emissions in areas unaffected by CAIR. Reductions in methylmercury concentrations are largely due to the implementation of CAIR controls at utilities. The reductions between 2001 and 2020 with CAIR are small for most of the sample locations (the median change is only 0.03 ppm). However, 15 percent of locations show an improvement of 0.1 ppm or greater.

Both the promulgated CAMR requirements and the CAMR alternative result in small reductions in fish tissue methylmercury concentrations. Less than 1 percent of sample locations have a reduction in fish tissue concentrations due to the CAMR requirements that is greater than 0.05 ppm. A small number of sample locations have increased mercury deposition, due to shifting of mercury emissions due to the trading provisions of the rule. The CAMR alternative results in slightly more sample locations with a greater than 0.05 ppm reduction in fish tissue concentrations. Over 99 percent of sample locations in both cases have reductions in fish tissue concentrations between 0 and 0.05 ppm. Table 3.6 provides summary statistics for the change in fish tissue methylmercury for the promulgated CAMR requirements and the CAMR alternative. From this table it is clear that even at the 99th percentile, the CAMR requirements do not result in a substantial reduction in fish tissue concentrations over that achieved by CAIR.

Table 3.5. Distributions of Reductions in Fish Tissue Methylmercury Concentrations

Range (mg/kg)	2001 Base - 2020 Base (with CAIR)		2020 Base (with CAIR) - 2020 CAMR Requirements		2020 Base (with CAIR) - 2020 CAMR Alternative		2020 CAMR Requirements - 2020 CAMR Alternative	
	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %
<0.00 - 0.05	65.75%	65.75%	99.94%	99.94%	99.81%	99.81%	99.94%	99.94%
0.05 - 0.10	20.19%	85.94%	0.00%	99.94%	0.13%	99.94%	0.06%	100.00%
0.10 - 0.20	11.00%	96.94%	0.00%	99.94%	0.00%	99.94%	0.00%	100.00%
0.20 - 0.30	2.44%	99.38%	0.06%	100.00%	0.06%	100.00%	0.00%	100.00%
0.30 - 0.40	0.25%	99.63%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
0.40 - 0.50	0.19%	99.81%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%
0.50 - 1.00	0.19%	100.00%	0.00%	100.00%	0.00%	100.00%	0.00%	100.00%

Table 3.6. Summary Statistics for Reductions in Fish Tissue Methylmercury Concentrations

Statistic (mg/kg)	2001 Base - 2020 Base (with CAIR)	2020 Base (with CAIR) - 2020 CAMR Requirements	2020 Base (with CAIR) - 2020 CAMR Alternative	2020 CAMR Requirements - 2020 CAMR Alternative
Minimum	-0.22	-0.03	-0.03	0.00
Maximum	0.84	0.20	0.21	0.05
50th percentile	0.03	0.00	0.00	0.00
90th percentile	0.12	0.00	0.01	0.00
99th percentile	0.25	0.01	0.02	0.02

Note: There has been some discussion about whether inclusion of pre-1999 samples would alter the conclusions. We have also analyzed the data with pre-1999 samples included, and the results are substantively unchanged. Tables 3.7 and 3.8 show the distributions and sample statistics for utility attributable fish tissue methylmercury using the full post-1990 set of NLFA and NLFTS data. There is one sample location in New Jersey in 2020 with a utility attributable concentration of 0.42 mg/kg, however, it is located in an area that may be influenced by a chloralkali plant, so the magnitude of the utility attributable portion may be overstated. One other location in South Carolina has a utility attributable portion that exceeds 0.3 mg/kg in the pre-1999 sample set. This location is located near several mining operations and thus is likely to have been influenced by non-air sources. As noted earlier in this document, the proportionality assumption based on Mercury Maps is not valid for locations that are significantly influenced by non-air sources. In addition, the higher value is influenced mainly by samples collected in 1993. After 1999, the samples all have much lower values. Given changes in mercury emissions and mining practices, it is not recommended that earlier samples be used when more recent data are available. Excluding these two locations, the conclusions remain generally the same.

Table 3.7. Sensitivity Analysis: Distributions of Utility Attributable Fish Tissue Methylmercury Concentrations Using All Post-1990 NLFA and NLFTS Samples

Range (mg/kg)	2001		2020 (with CAIR)		2020 CAMR Requirements		2020 CAMR Alternative	
	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %	Percent	Cumulative %
<0.00	0.1%	0.1%	0.1%	0.1%	0.1%	0.1%	0.1%	0.1%
0.00 - 0.05	72.0%	72.1%	94.0%	94.1%	95.2%	95.3%	96.2%	96.3%
0.00 - 0.10	17.0%	89.2%	4.5%	98.6%	3.7%	99.0%	2.9%	99.2%
0.10 - 0.20	8.0%	97.1%	1.3%	99.8%	0.9%	99.9%	0.7%	99.9%
0.20 - 0.30	2.1%	99.3%	0.1%	100.0%	0.1%	100.0%	0.1%	100.0%
0.30 - 0.40	0.3%	99.6%	0.0%	100.0%	0.0%	100.0%	0.0%	100.0%
0.40 - 0.50	0.1%	99.7%	0.0%	100.0%	0.0%	100.0%	0.0%	100.0%
0.50 - 1.00	0.3%	100.0%	0.0%	100.0%	0.0%	100.0%	0.0%	100.0%

Table 3.8. Sensitivity Analysis: Summary Statistics for Utility Attributable Fish Tissue Methylmercury Concentrations Using All Post-1990 NLFA and NLFTS Samples

Statistic (mg/kg)	2001	2020 (with CAIR)	2020 CAMR Requirements	2020 CAMR Alternative
Minimum	0.00	0.00	0.00	0.00
Maximum	1.63	0.42	0.36	0.34
50th percentile	0.02	0.01	0.01	0.01
90th percentile	0.10	0.04	0.03	0.03
99th percentile	0.28	0.12	0.10	0.09

4. Fish Consumption

Modeling exposure to methylmercury through fish consumption requires characterizing dietary fish consumption rates for specific populations such as recreational anglers and subpopulations exhibiting higher fish consumption rates including Native American subsistence fishers or other high-consumption (subsistence-like) groups. The primary concern is for the consumption of self-caught freshwater fish, (i.e., fish caught and consumed from rivers and lakes by members of the public). This reflects the fact that mercury emitted from coal-fired power plants is likely to primarily impact domestic inland waterbodies in the eastern part of the country through local and regional deposition (see Section 2). In addition, because the RfD for mercury is based on a long-term average daily exposure rate for methylmercury, studies that characterize long-term fish consumption patterns are most relevant to modeling exposure to methylmercury. Many available studies on fish consumption focus on short-term consumption patterns (i.e., 2 day dietary recall) without considering meal frequencies over longer periods ranging from

months to a year or more. These short-term studies can over-estimate consumption rates for higher percentiles of the population (i.e., higher consuming individuals) compared with longer-term studies that consider meal frequencies in characterizing high-end consumption rates[10].

Two broad categories of fishers can be considered in the context of modeling exposure to methylmercury through fish consumption: (a) recreational freshwater anglers,(i.e., the majority of fishers in the nation who typically purchase fishing licenses) and (b) higher consuming individuals who through choice, socio-cultural practices or necessity, (i.e., dietary supplementation) consume relatively larger amounts of freshwater fish. It is important to note that these two populations are not mutually exclusive and there is some degree of overlap. For example, available fish consumption data suggests that the higher percentiles of the recreational fisher population, (i.e., >95th %) may have consumption rates that are similar to the average subsistence consumer such as Native American subsistence fishing populations. Therefore, while members of group "b" above (higher consuming individuals) are generally considered to be of greatest concern for exposure to methylmercury due to their high fish consumption rates, recreational anglers with extreme high-end consumption rates may also consume large amounts of self-caught freshwater fish; in the same range as subsistence fishers[11].

As indicated above, our analysis focused on assessing exposure to self-caught freshwater fish because available information indicated that U.S. utility mercury emissions might contribute to the methylmercury concentrations in these fish. EPA also considered the following fish consumption pathways: consumption of fish from commercial sources (including marine, freshwater, and estuarine fish from domestic and foreign sources), consumption of recreationally caught marine fish, consumption of recreationally-caught estuarine fish, and consumption of commercial fish raised at fish farms (aquaculture). For a number of reasons, EPA does not believe that these latter pathways would result in significant exposure to utility-attributable methylmercury. In addition, an important analytical tool, the Mercury Maps approach that has been applied in freshwater systems to correlate the amount of deposition with level of fish tissue methylmercury, is more uncertain in saltwater systems (estuaries, coastal, and deep ocean).

Specifically, EPA does not believe that commercial fish are adversely affected by mercury emissions from U.S. utilities to any significant degree. Imports account for over half of

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[10] EPA notes that, while mean consumption rates for dietary items characterized using short-term dietary recall data are reasonable and will typically match rates generated using longer-term surveys, high-end consumption rates based on short-term dietary recall studies can be biased high when used to represent upper percentile long-term average consumption rates. This applies only for dietary items which are consumed sporadically, (i.e., are not consumed day-to-day at a relatively constant rate) which would seem to apply to most types of self-caught fish consumption except high-end subsistence which may approach a meal a day. (Exposure Factors Handbook, Section 9.2.1, EPA, 1997)

[11] Surveys of licensed recreation anglers have shown that extreme high-end consumption rates for self-caught freshwater fish can meet or exceed the EPA's recommended mean value for freshwater subsistence fishers. Fiore et al (1999) as reported in U.S. EPA (1997) provides a 100th percentile rate of 150 g/day for recreational anglers in Wisconsin which exceeds the EPA's recommended subsistence freshwater value of 60 g/day.

the U.S. commercial fish supply. Moreover, of the commercial fish landed domestically, the majority (61%) are caught in the deep ocean (3-200 miles offshore) in the Pacific and Atlantic Oceans, and the Gulf Coast. In 2002, the Pacific coast region alone accounted for 65% of U.S total commercial landings, with Alaska and California accounting for the largest portion. In contrast, our modeling shows that U.S. utility-attributable mercury deposition is concentrated in the midwestern and northeastern portion of the U.S., that is, in areas geographically removed from the majority of commercial fishing activity. For the same reasons, EPA does not expect that self-caught, (e.g., recreational) fish from marine waters will be affected significantly by U.S. utilities.

Individuals also may consume estuarine fish and fish derived from aquaculture. We determined that it was not critical to assess the impact of utility mercury emissions on commercial or self-caught estuarine fish. Based on our deposition modeling, we expect that any impact of utility-attributable mercury on estuarine environments will be limited.

Similarly, we believe that the overall impact of utility-derived deposition on U.S. aquaculture will be minimal. U.S. aquaculture is very small part of commercial fish production, comprising, in 2002, about 5% of total commercial fish production. Much of U.S. aquaculture occurs outside of the northeast and midwest, and therefore is not much impacted by utility attributable mercury emissions. We expect that potentially significant health impacts could only occur in the event that individuals consistently consume a particular aquaculture-derived fish type that is supplied from a single geographic location significantly impacted by utility mercury emissions. We do not have adequate information to allow us to predict the number of individuals who would be subject to such a confluence of events, and we suspect that the number would be quite small. In any event, even under these circumstances, we do not have sufficient information to characterize the impact of utility emissions on aquaculture. We do not believe that there will be a clear or consistent relationship between the mercury deposition onto waters supporting aquaculture and the methylmercury concentrations in fish. Such an analysis would be complicated by the fact that the diets of fish raised in aquacultural environments often consist of, or are supplemented with, commercial feeds[12]. For all these reasons, we believe that the focusing our analysis on non-commercial freshwater fish is reasonable and appropriate.

Recreational Angler Consumption Rates

EPA has recommended both average and high-end consumption rates for self-caught freshwater fish for recreational anglers (8 g/day and 25 g/day as the mean and 95th percentile, respectively) (EPA, 1997a). These values are based on several of the best available peer-

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[12] The belief that farm-raised fish often contain less contamination than wild caught fish is currently an issue under debate in the published literature. The main issue in determining the difference in tissue methylmercury concentrations between farm raised and wild fish appears to be the duration of exposure and source of food consumed by the fish.

Farm raised fish species bred for human consumption are fed a controlled food source. Recent studies have found varying levels of contamination in fish feed (Hites et al. 2004a & b). In situations where the food source is not contaminated, the threat of bioaccumulation of contaminants in farm raised fish should be lessened.

reviewed studies focusing on freshwater recreational fisher populations located in the Midwest and Northeast. All of these studies are based on surveys of licensed fishers and therefore, exclude non-licensed fishers which may include individuals engaging in high-end subsistence-like activity.

Because these studies support characterization of high-end consumption by recreational anglers (through the 95th percentile of 25 g/day), it is possible to fit a consumption rate distribution for recreational anglers using these recommended values (lognormal distributions as well as other statistical models can be fitted using these datapoints, although the discussion here references the fitting of a lognormal distribution[13]). A lognormal distribution fitted to these datapoints can be used both for (a) interpolating percentile consumption rates between these two percentiles (the mean and 95th percentile) and (b) extrapolating to higher rates of consumption beyond the 95th %. The former application is important in supporting population-level modeling of recreational fisher exposure where individual fish consumption rates are sampled from this distribution and assigned to modeled fishers. The latter application allows projections of high-end percentiles (e.g., 99th percentile and above), which allows modeling of extreme high-end recreational angler consumption that can approach or even reach subsistence levels of consumption. Values in the range of 50-60 g/day have been identified for the 99th percentile recreational fisher using this *fitted lognormal distribution approach*. This "derived" 99th percentile value is in-line with similar percentiles obtained from specific survey's of freshwater recreational anglers (e.g., the 41 g/day to 150 g/day range reported for the 98th percentile to 100th % in a study of recreational anglers in Wisconsin, (Fiore et al, 1989 as reported in USEPA, 1997a)).

Use of the 95th percentile recreational freshwater consumption rate recommended by the EPA (25 g/day) provides coverage for the majority of recreational fishers. However, based on consideration of the lognormal distribution described above, as well as published data (i.e., the 98th-100th percentile values published by Fiore referenced above), some recreational anglers would have fish consumption rates above this level, including individuals with consumption rates near or at subsistence-levels, (i.e., at or above the 60 g/day mean level recommended by the EPA for freshwater subsistence fishers - see below).

In the context of CAMR, which is focused on recreational consumption of self-caught freshwater fish, the 17.5 g/day (11.7 g/day cooked weight equivalent) value recommended by the Office of Water (as representing the 90th percentile of general population fish consumption) and used in calculating the Ambient Water Quality Criteria for mercury (0.3 mg/kg) may over-estimate exposure for the average recreational angler and underestimate exposure for the

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[13] The lognormal model is often used in fitting distributions to point value data (e.g., means, 90th %, 95th %) for purposes of more completely characterizing inter-individual variability in dietary consumption for modeled populations. There is precedent for using the lognormal distribution for representing inter-individual variability in fish consumption rates in supporting risk assessment conducted as part of national-scale regulatory development by EPA (Section 6.3.2 of Human Health and Ecological Risk Assessment Support to the Development of Technical Standards for Emissions from Combustion Units Burning Hazardous Waste, Background Document, US Environmental Protection Agency, Office of Solid Waste, Washington, DC, July 1999.)

high-end recreational anglers. This is because the 17.5 g/day (11.7 g/day cooked weight equivalent) value is larger than the EPA's recommended freshwater self-caught fish mean consumption rate of 8 g/day and smaller than the EPA's recommended 95th percentile for that same pathway of 25 g/day. It is important to note also that the 17.5 g/day (11.7 g/day cooked weight equivalent) includes consumption of commercially-purchased fish and estuarine fish (in addition to self-caught freshwater fish), both of which are not being considered in this analysis.

EPA recognizes that recreational fisher consumption rates will vary geographically reflecting different patterns of fishing activity. Ideally, studies covering activity at the local or regional level would be combined to develop an integrated set (i.e., patchwork) of fish consumption rates that, when viewed collectively, provided a more representative characterization of national recreational fisher behavior. However, many of these regional/local studies do not meet the criteria described above, (i.e., characterizing long-term daily consumption rates of self-caught freshwater fish). In addition, available local- and regional-scale studies that meet these criteria do not, as a whole, provide reasonable coverage for the entire study area. Consequently, EPA favors the use of the recommended recreational fisher consumption rates of 8 g/day (mean) and 25 g/day (95th percentile) for our analyses.

High-Level Fish Consumption Rates

Characterization of fish consumption rates for high fish consuming people, (e.g., certain segments of Native American and other ethnic populations exhibiting high-end consumption) in the context of a larger regional or national analysis is technically challenging for a number of reasons. Peer reviewed study data on these populations are relatively limited in geographical coverage, especially when subjected to the criteria outlined above, (i.e., meal frequencies averaged over longer periods). This means that, while subsistence-level fish consumption has been characterized for some populations, many portions of the country where these populations may be present are not covered by existing data. In addition, many of the high consumption populations that have been studied are located near the ocean and consequently have a significant fraction of their overall exposure comprised of saltwater fish. Since analyses conducted in support of CAMR focus on self-caught freshwater fish consumption by various populations, inclusion of saltwater fish consumption in many of these studies prevent them from being considered (since the fractional contribution of saltwater relative to freshwater in these studies can not be determined). In addition, some of these studies provide details on seasonal consumption rates, but do not integrate these rates to provide an overall mean annual-averaged consumption rate relevant to an analysis of longer term exposure to methylmercury.

In addition, while many of these studies provide mean consumption rates, few have identified specific high-end percentile values (e.g., 90th, 95th or 99th percentile consumption rates). Instead, many studies, including a number of non-peer reviewed sources, cite non-specific high-end or bounding point estimates, (e.g., the range of consumption rates for the Ojibwe submitted for the CAMR NODA - see below). While these point values can be used in developing high-end bounding scenarios for evaluating risk to these sub populations, they do not support population-level analysis of exposure since they can not be used to fit distributions

characterizing variability in fish consumption rates across these sub-populations (as noted above, modeling of population-level exposures requires that distributions characterizing fish consumption rates across a particular population be developed).

An additional challenge in characterizing high-level fish consumption is that care needs to be taken in extrapolating study results from one sub-population to other sub-populations. This reflects the fact that high-level fish consumption is often tied to socio-cultural practices and consequently consumption rates for a study population can not be easily transferred to other populations which may have different practices, (e.g., practices for one Native American tribe may not be relevant to another and consequently behavior regarding fish consumption may not be generalized).

Despite these challenges in characterizing high-level consumption, EPA has developed a recommended high-level fish consumption rates of 60 g/day (mean) and 170 g/day (95th percentile) (EPA, 1997a). These values are based on a study of several Native American tribes located along the Columbia River in Washington state that exhibit high fish consumption behavior. Although these consumption rates are specific to the tribes included in the study and reflect their particular socio-cultural practices (including seasonality and target fish species), EPA believes that this study does provide a reasonable characterization of high-consuming subsistence-like freshwater fishing behavior. Therefore, in the absence of data on local practices, EPA recommends that these consumption rates be used to model high-consuming (subsistence) groups in other locations. It is important to note that, as explained above, application of these consumption rates outside of the original Columbia River study area to populations that could have different fishing behavior prevents these consumption rates from being used in modeling population-level exposure. As noted earlier, population-level exposure modeling requires a fish consumption variability distribution matched to the population under consideration. Consequently, because it is not known how representative the percentile consumption rates from the Columbia River tribes are of other high-consumption groups, they can not be transferred to other groups and used in population-level modeling. However, it is possible to use the Columbia River consumption rates to conduct scenario-based analysis of consumption rate groups, (e.g., modeling average or high-end exposures without attempting to specify a true range of exposures across the modeled sub-population). As stated on page 10-27 of the Exposure Factors Handbook (EPA, 1997a), "[i]t should be emphasized that the above recommendations refer only to Native American subsistence fishing populations, not the Native American general population."

Although these high-level consumption rates are recommended by EPA, a number of sources (including NODA comments obtained for this rule), have identified alternate consumption rates for specific high-consuming groups (including Native Americans) that are in some instances, higher than these recommended values. For example, a survey by the Great Lakes Indian Fish and Wildlife Commission (GLIFWC) (as referenced in comments to the CAMR NODA) indicates that consumption rates by members of Ojibwe Great Lakes tribes during fall spearing season may range from 155.8-240.7 g/day and may range from 189.6 - 393.8 g/day during the spring. EPA has reviewed these data, and the Agency does not believe they are

suitable for use in this analysis. The Ojibwe consumption data, while useful in providing perspective on subsistence-fishing, can not be readily translated into an annual-averaged consumption rate which can be associated with a specific percentile of the population. By contrast, the EPA's recommended subsistence consumption value of 170 g/day is a longer-term averaged value and is identified as clearly representing the high-end of a particular subsistence fishing group, (i.e., the 95th percentile of the Columbia River groups). In addition , the locations where the Ojibwe tribes reside do not appear to be significantly impacted by utility emissions of mercury, decreasing the relevance of the Ojibwe data in covering subsistence fishing groups located in those areas of the country with higher relative rates of EGU deposition after the rule. Therefore, despite these higher consumption rates referenced by GLIFWC, EPA believes that the Columbia River-based consumption rates provide reasonable coverage for high-level consumption sub-populations, such as Native American subsistence fishing groups, in other locations. Also, we can not readily translate these ranges into clear population percentiles characterizing long-term (annual-averaged) fish consumption rates for Native Americans located in that part of this country or for other subsistence Native Americans. This final point prevents these NODA-related data from being used as part of a population-level analysis of exposure in this analysis.

In the context of CAMR with its emphasis on self-caught freshwater fish consumption, EPA has concluded that the 142 g/day value recommended by the Office of Water for subsistence populations (calculated as the 99th percentile of the general population) is not appropriate for this analysis for two reasons. First, it includes commercial-source fish and estuaries fish, and the current analysis is focused on freshwater fish. Second, the methodology used in calculating the distribution of fish consumption rates has a known bias that causes the tails (90th and 99th percentiles) of the distribution to be overestimated by an unknown but possibly large amount or possibly small amount. The methodology used a distribution of short-term consumption rates to extrapolate to distribution of long-term (annual) consumption rates without adjusting for the larger variability that occurs in short-term consumption rates.[14] This bias does not occur for the mean values.

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[14] This bias was noted in the Estimated Per Capita Fish Consumption in the United States (U.S. EPA, 2002) . See http://www.epa.gov/waterscience/fish/consumption_report.pdf at page ix: "The CSFII surveys have advantages and limitations for estimating per capita fish consumption. The primary advantage of the CSFII surveys is that they were designed and conducted by the USDA to support unbiased estimation of food consumption across the population in the United States and the District of Columbia. One limitation of the CSFII surveys is that individual food consumption data were collected for only two days—a brief period which does not necessarily depict "usual intake." Usual dietary intake is defined as "the long–run average of daily intakes by an individual." Upper percentile estimates may differ for short–term and long–term data because short term food consumption data tend to be inherently more variable. It is important to note, however, that variability due to duration of the survey does not result in bias of estimates of overall mean consumption levels."

5. Human Health Benchmarks

The primary route of methylmercury exposure in the US is through the consumption of fish. It is important to understand that mercury must be transformed into methylmercury for it to bioaccumulate in fish and for it to be bioavailable to humans.[15]

EPA has set a health-based ingestion rate for chronic oral exposure to methylmercury, termed an oral Reference Dose (RfD). The RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime (EPA 2002a). EPA believes that exposures near or below the RfD are very unlikely to be associated with appreciable risk of deleterious effects. It is important to note, however, that the RfD does not define an exposure level corresponding to zero risk; mercury exposure at or below the RfD could pose a very low level of risk which EPA deems to be non-appreciable. It is also important to note that the RfD does not define a bright line above which individuals are at risk of adverse effect.

In 1995, EPA set an oral RfD for methylmercury at 0.0001 mg/kg-day (0.1 ug/kg-day) based on a study of the Iraqi poisoning episode (Marsh et al. 1987). Subsequent research from large epidemiological studies in the Seychelles, Faroe Islands, and New Zealand added substantially to the body of knowledge on neurological effects from methylmercury exposure. Per Congressional direction via the House Appropriations Report for Fiscal Year 1999, the National Research Council (NRC) of the National Academy of Science was contracted by EPA to examine these data and, if appropriate, make recommendations for deriving a revised RfD. NRC's analysis concluded that the Iraqi study should no longer be considered the critical study for the derivation of the RfD and also provided specific recommendations to EPA regarding methylmercury based on analyses of the three large epidemiological studies (NRC 2000). EPA's current assessment of the methylmercury RfD, revised in 2001, relied on the quantitative analyses performed by the NRC.

In their analysis, NRC examined in detail the epidemiological data from the Seychelles, the Faroe Islands, and New Zealand, as well as other toxicological data on methylmercury (Crump et al. 1998; Grandjean et al., 1997; Myers et al., 2003). In determining a recommended point of departure (i.e., the specific dose on which health criteria should be based), NRC recommended a benchmark dose approach, which applies mathematical models to the available data to identify the point of departure. The BMD is the exposure level at which a particular level of response, (i.e., the benchmark response, or BMR) for some outcome of concern is predicted to occur. In their assessment of the epidemiological data, NRC proposed that the Faroe Islands cohort was the most appropriate study for defining an RfD, and specifically selected children's performance on the Boston Naming Test (a neurobehavioral test) as the key endpoint. They recommended a BMR of 0.05, (i.e., the level at which would result in a doubling in the number

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[15] Note that as such, the relevant risk that we are addressing is not from exposure to mercury vapors, but is the exposure to methylmercury through the consumption of fish.

of children with a response at the 5th percentile of the population).[16] On the basis of this study cohort and that test, NRC identified a BMD of 85 ppb in cord blood. The NRC also estimated the 95% lower confidence limit for the BMD, (i.e., the BMDL) for this endpoint to be 58 ppb. The BMDL is a conservative estimate which is used as a point of departure in risk assessment. Although this BMDL was specifically recommended by NRC as appropriate for deriving the RfD, NRC also conducted BMD analyses on other endpoints in the Faroe cohort and several endpoints in the other two populations, as well as an integrative analysis of data from all three studies (NRC 2000).

In updating the RfD, EPA considered BMD analyses completed by NRC involving endpoints of neuropsychological development from the Faroe Islands cohort (including results for the Boston Naming Test), the New Zealand cohort, and the NRC's integrative analysis of all three studies. The BMDLs for these endpoints, measured as concentrations of mercury in umbilical cord blood, were considered. For the purposes of calculating the RfD, EPA converted these BMDLs to maternal daily dietary intake in mg/kg-day using a one-compartment model.[17] The BMDLs for these analyses (measured in terms of mercury in cord blood) were all observed to be within a relatively close range, and, after application of 10x factor to account for variability and uncertainty, the calculated RfDs converge at about 0.0001 mg/kg-day. Specifically, BMDLs for a number of neurological endpoints based on tests that gauge a child's ability to learn and process information, (i.e., Boston Naming Test, Continuous Performance Test, California Verbal Learning Test, McCarthy Perceived Performance, and McCarthy Motor Test) were calculated by NRC to range from about 25 to 100 ppb mercury in cord blood. These exposures were converted to dietary exposures of about 0.0005 mg/kg-bw/day to 0.0019 mg/kg-day, with most dietary exposures estimated to be about 0.001 mg/kg-bw/day. The integrative BMDL (taking into account data from all three studies) was calculated by NRC to be 32 ppb mercury in cord blood, or an exposure of about 0.6 ug/kg-day. All of these results were considered in defining the RfD; as stated in the IRIS summary for methylmercury[18]:

> "Rather than choose a single measure for the RfD critical endpoint, EPA based this RfD for this assessment on several scores from the Faroes measures, with supporting analyses from the New Zealand study, and the integrative analysis of all three studies."

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[16] As noted by NRC in reference to data from the Seychelles, Faroe Islands, and New Zealand, "because those data are epidemiological, and exposure is measured on a continuous scale, there is no generally accepted procedure for determining a dose at which no adverse effects occur." The NRC chose a 5% response level in the BMD analysis for test results in the lower 5% of the distribution.

[17] The one-compartment toxicokinetic model employed by EPA is described by NRC (2000); it represents all maternal body compartments as a single pool with a relatively small set of parameters, and assumes steady-state conditions in the maternal system. Methylmercury dose levels were measured as concentrations in umbilical cord blood (analysts have assumed that methylmercury concentration in cord blood is roughly equal to that in maternal blood).

[18] The IRIS summary may be found at the following website;
http://www.epa.gov/iris/subst/0073.htm#reforal

After identifying a BMDL, consistent with NAS guidance, EPA then applied an uncertainty factor of 10 to account for interindividual toxicokinetic variability and pharmicodynamic variability and uncertainty. On this basis, EPA defined the updated RfD of 0.0001 mg/kg-day (or, equivalently, 5.8 ppb in blood) in 2001. Although derived from a more complete data set and with a somewhat different methodology, the current RfD is the same as the previous (1995) RfD.

In addition, to put these exposure levels in perspective, it is useful to consider typical mercury exposure levels in the U.S. measured in the National Health and Nutrition Examination Survey (NHANES, (CDC, 2001)). This survey is conducted by the National Center for Health Statistics via standardized interviews to provide continuous health data for the general U.S. population, and it has included measurements of mercury in blood and hair as biomarkers of mercury exposure. Based on NHANES data for blood collected for 1999-2002, the overall distribution of blood mercury concentrations for women of child-bearing age, (i.e., between 16 and 49 years of age) has been estimated for the U.S. population (see Figure 5.1). The RfD and BMDL derived from the Faroe cohort effect level are included on this chart for reference. Although all observed exposures are below the BMDL, and most of the exposures fall below the RfD, about 6% of the population exposures were above the RfD[19]. The geometric mean blood mercury concentration in the NHANES data for 1999-2002 is 0.92 ppb, and the range of observed concentrations was from 0.07 to 38.90 ppb.

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[19] CDC Morbidity Mortality Weekly Report (MMWR) Vol. 53/NO. 43

Figure 5.1. Probability Distribution Function of Blood Mercury Levels in US Women of Childbearing Age (NHANES Data 1999-2002)
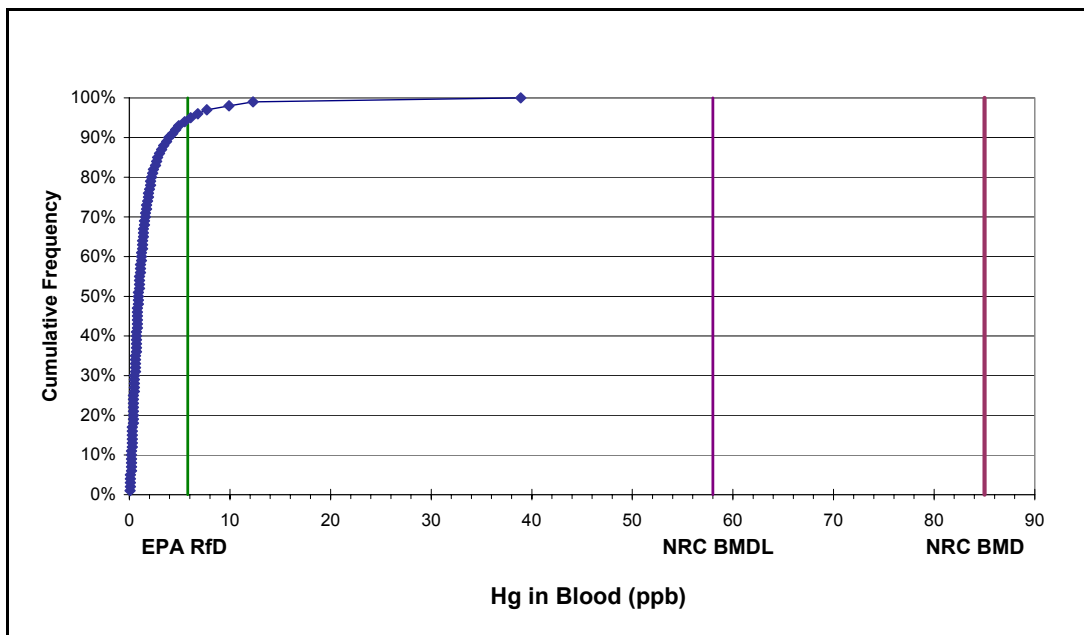
Note: Cumulative frequency (y-axis) refers to the fraction of the population exposed at or below a given blood mercury level. EPA's RfD for methylmercury is 0.1 ug/kg-day, which is approximately equivalent to a concentration of 5.8 ppb in blood.

Methylmercury is a developmental neurotoxicant and the greatest biological sensitivity is *in utero* exposure. The developing fetus is most sensitive to mercury exposure because methylmercury easily passes the placenta and the blood-brain barrier and because, in general, the developing nervous system is more susceptible to toxicants. However, there are currently no studies designed specifically to identify the relationship between childhood fish consumption and developmental outcomes. Thus, we do not have data upon which to determine whether childhood exposures alone (without fetal exposure) contributed to neurodevelopmental deficits. Data from what is considered to be the most sensitive subpopulation were used as the basis for the RfD; thus, its use is thought to be protective of all life stages without additional uncertainty factors or adjustments. The exposed subpopulations of interest to this rulemaking are women of childbearing age who consume large quantities of fish caught from vulnerable water bodies with significant utility attributable mercury deposition.

Using the 2001 RfD and information on mercury exposure routes, EPA published a recommended ambient water quality criterion for the States' and Tribes' use in setting water quality standards designed to protect human health. EPA issued the methylmercury water quality criterion in 2001. (EPA 2001a.) Because of the wide variability in methylmercury bioaccumulation among waterbodies, EPA set the criterion as a fish tissue level rather than as a water concentration. The criterion is 0.3 mg/kg (milligram methylmercury per kilogram of wet-weight fish tissue). The criterion is a risk assessment number which States and authorized Tribes

may use in their programs for protection of designated uses.

6. Human Exposure to Methylmercury Through Fish Consumption

As described above in Section 3, it is important to note that the analysis presented in this document does not account for the time lag between changes in deposition and changes in methylmercury fish tissue concentrations. Results for methylmercury concentrations and corresponding exposure estimates should be interpreted being at steady state, not in any particular year.

The RfD provides a useful reference point for comparisons with measured or modeled exposure. The Agency defines the RfD as an exposure level below which the Agency believes exposures are likely to be without an appreciable risk over a lifetime of exposure. For the purposes of assessing population exposure due to power plants, we create an index of daily intake (IDI). The IDI is defined as the ratio of exposure due solely to power plants to an exposure of 0.1ug/kg bw/day. The IDI is defined so that an IDI of 1 is equal to an incremental exposure equal to the RfD level, recognizing that the RfD is an absolute level, while the IDI is based on incremental exposure without regard to absolute levels. Note that an IDI value of 1 would represent an absolute exposure greater than the RfD when background exposures are considered.

The available data on fish tissue methylmercury concentrations and fish consumption in the population of recreational anglers does not support an analysis of the specific number of individuals that are exposed to any particular IDI value. In order to calculate these population estimates, we would require additional data on the specific fishing locations for individual anglers, matched with the consumption rates for freshwater fish for those anglers. We do not currently have this type of detailed, location specific information. In the absence of such information, we can calculate the likelihood of an individual angler at particular IDI values by examining the distributions of fish tissue methylmercury concentrations and fish consumption rates. We can also estimate the general size of the population that might be exposed to various IDI values by examining the conditional probabilities given specific percentiles of methylmercury concentrations or consumption rates. The data on consumption rates differs between the general population of freshwater anglers and subsistence fishing populations. As such, this discussion is presented in two parts dealing with consumption by the general freshwater fishing population and the subsistence fishing population.

6.A. General Population of Freshwater Anglers

The distribution of utility attributable methylmercury concentrations in 2001 shows a 99th percentile concentration of 0.26 mg/kg (or ppm), representing 16 water bodies (see Table 3.4). The distribution of utility attributable methylmercury concentrations after implementation of CAIR in 2020 shows a 99th percentile concentration of 0.102, whereas after CAMR, the 99th percentile methylmercury concentration is 0.093 (see Table 6.2). By combining these upper end concentrations with estimates of high-end consumption, it is possible to determine whether there

is a low probability of an individual exceeding the IDI. For the general population of female recreational anglers and spouses of male anglers of child-bearing age (around 10.5 million), the 99th percentile consumption level is 47 g/day, based on a lognormal distribution with mean of 8 and 95th percentile at 25 (see section 4 above for more discussion).

In order to calculate the IDI value associated with a particular combination of methylmercury concentration and fish consumption rate, we use the following equation:

$$IDI = \frac{\text{consumption rate x methylmercury concentration x 1.5}}{\text{bodyweight}},$$

where 1.5 is an adjustment factor to reflect the fact that cooking tends to reduce fish mass by approximately one-third. Because methylmercury is not volatile and is contained primarily in the muscle, this translates into a factor increase of 1.5 for concentration of mercury in the cooked fish (Morgan, Berry, and Graves, 1997). For example, if a person with the 99th percentile of consumption were to catch and consume fish with the 99th percentile methylmercury concentration after implementation of CAIR in 2020, that individual would have an exposure level calculated as:

$$IDI(60, 0.10) = \frac{47 \text{ g/day} * 0.1 \text{ µg/g} * 1.5}{64\text{kg} * 0.1 \text{ µg/kg/day}} = 1.11$$

Table 6.1 shows the IDI associated with the 99th percentile of utility attributable methylmercury and 99th percentile of consumption rates for the 2001 base case, 2020 after implementation of CAIR, and 2020 after implementation of the CAMR requirements. It also shows the percentile of methylmercury concentrations at which the IDI value would be 1 holding the consumption rate constant, and the percentile of consumption rates at which the IDI value would be 1 holding the methylmercury percentile constant.

Table 6.1. Index of Daily Intake (IDI) Levels Associated with Upper Percentile Methylmercury Concentrations and Recreational Angling Fish Consumption Rates

Analytical Scenario	IDI values at 99th Percentile of Methylmercury Concentration and 99th percentile Consumption Rates (47 g/day)	Percentile of Methylmercury Distribution at Which the IDI Value is Less Than 1 (holding consumption rates at the 99th percentile)	Percentile of Consumption Rate Distribution at Which the IDI Value is Less than 1 (holding methylmercury concentrations at the 99th percentile)
2001 Base Case	2.81	85.5th	89.7th
2020 Implementation of CAIR	1.11	98.7th	98.6th
2020 Implementation of CAMR	1.03	99th	99th

In the 2001 base case, at the 99th percentiles of concentrations and consumption rates, the IDI value would be almost 3. The IDI value would exceed 1 as long as the methylmercury concentration exceeded the 85th percentile (holding consumption rates at the 99th percentile) or the fish consumption rate exceeded the 90th percentile (holding the methylmercury concentrations at the 99th percentile). There is still a very low probability that an individual angler would exceed an IDI value of 1 (at the highest consumption levels, the number of anglers would be 15 percent of 1 percent, or 0.15 percent). However, for those individuals where this occurs, the IDI value may exceed 1 by several fold. After implementation of CAIR in 2020, the IDI value at the 99th percentiles of methylmercury and consumption rates is much closer to 1. In fact, the methylmecury concentrations would only have to be at the 98.7 percentile (holding consumption at the 99th percentile) or consumption rates at the 98.6th percentile (holding methymercury concentrations at the 99th percentile) for the IDI value to be reduced to 1. If consumption is not correlated with the methylmercury concentrations in fish (there is no data to suggest it is), then the IDI value of 1 would be potentially exceeded only 1 percent of the time in 1 percent of the 10.5 million female consumers of recreationally caught fish. This implies that it is highly unlikely under this modeling scenario. After implementation of the CAMR requirements, even at the 99th percentile of methylmercury concentrations and consumption rates, the IDI value does not exceed 1. It is important to note that these calculations are based on a mean bodyweight that is not correlated with consumption rates. Increasing the bodyweight (to reflect a correlation between higher consumption and higher bodyweight) by only 10 percent (less than one standard deviation from the mean) would reduce the IDI value to 1.

The probability that the IDI value after CAIR or CAMR would exceed 1 is thus much smaller than 1 percent, as it would require high consumption rates, high fish tissue

concentrations, and no correlation between bodyweight and consumption rates, all of which are very unlikely. EPA recognizes that for those few hypothetical individuals where this confluence of high consumption and high methylmercury concentrations occurs, there may be exposures above the IDI. In assessing the health implications for exposures for exposure above an IDI of 1, EPA also looks to the severity of effects which the RfD is designed to prevent as well as the conservatism built into the parameters used to derive the RfD.

To better understand the implications of the potential effects near or above the RfD it is helpful to consider the study effects used in establishing the RfD. As mentioned in Section 5, in updating the RfD, EPA considered endpoints from a battery of neuropsychological tests that focus on subtle neurological endpoints. We must then interpret what those endpoints will translate to. The methylmercury-associated performance decrements on the neuropsychological tests administered in the Faroe Islands and New Zealand studies suggest that prenatal methylmercury exposure is likely to be associated with poorer school performance. Thus, children who are exposed to low concentrations of methylmercury prenatally may be at increased risk of poor performance on neurobehavioral tests, such as those measuring attention, fine motor function, language skills, visual-spatial abilities (like drawing), and verbal memory.

The underlying parameters chosen to establish the RfD introduce a degree of conservatism. Choice of the nature and extent of change that would be the appropriate focus for establishing a reference dose also involved key science policy decisions. Prior to determining a level that would result in an impact or change in the expression of an adverse effect, the population baseline rate of the effect must first be identified, (i.e., the probability of the effect occurring in the general population). After identifying that rate, another informed policy decision must be made to identify the benchmark response (BMR). In general, risk assessments for various toxicants based on animal studies have used a BMR of 0.1, because it usually represents the low range of the observed exposure data. Crump et al. (2000) used a BMR of 0.1, (i.e., 10% of the population is at risk) in their analyses of the New Zealand and Seychelles studies. For the end points studied, the baseline rate in the population is 0.05 (P0). Selection of a BMR of 0.1, therefore, could result in as much as a tripling of the percentage of the population falling into the abnormal range of neurological performance. In its assessment, the NRC committee, and subsequently EPA, chose a more health protective P0 of 0.05, and BMR of 0.05, (i.e., 5% of the population is at risk).

The choice of an "acceptable" risk level is one of policy informed by science. The RfD does not represent a "bright line" above which individuals are at risk of significant adverse effects. Rather, it reflects a level where EPA can state with reasonable certainty that risks are not appreciable. The Agency further notes that a number of other national and international scientific bodies have assessed the health effects of mthylmercury and have adopted other health-based benchmarks greater than EPA's RfD. Health Canada established its Tolerable Daily Intake (TDI) level at twice the EPA's RfD. Their benchmark is 0.2 ug/kg bw/day. The Agency for Toxic Substances and Disease Registry (ATSDR) has set a Minimal Risk Level (MRI) of 0.3 ug/kg bw/day – three times EPA's RfD level. The World Health Organization's (WHO) benchmark is set at 0.23 ug/kg bw/day. Of these major agencies, EPA's RfD has established the

lowest risk benchmark to define levels of exposure that are without appreciable risks.As exposure levels increase beyond the RfD, the possibility of deleterious effects increases, but the point at which they become "unacceptable" must be determined on a case-by-case basis. In making this determination, the Agency considers a number of factors including[20]:

• Confidence in the risk estimate: How certain is the scientific information supporting the link between possible health effects and exposures?

• The effects of concern: How serious are the health effects?

• The size of the population at risk, as well as the distribution of risk within the population.

The Agency has considered these factors in the case of mercury and has concluded that the exposures above the IDI described elsewhere in this chapter do not constitute an unacceptable risk.

To examine further the potential impact of correlation between bodyweights and consumption rates, Table 6.2 shows the distributions of fish consumption rates, bodyweights, and methylmercury concentration after implementation of CAIR and CAMR. It also shows the estimated IDI values if the distributions were perfectly correlated, i.e. if the highest percentiles of each distribution occurred in the same individuals. If this holds, then even at the 99th percentile of consumption and methylmercury, after either CAIR or CAMR, individuals are not exposed at an IDI value exceeding 1.

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[20] See Residual Risk Report to Congress, USEPA, March 1999 EPA-453/R-99-001), page 119.

Table 6.2. Implied Index of Daily Intake (IDI) Values Associated with Percentiles of the Distributions of Fish Tissue Methylmercury, Consumption Rates, and Bodyweight

Percentile	(A) Consumption Rate (g/day)	(B) Bodyweight (kg)	(C) 2020 Residual Utility Attributable Fish Tissue Methylmercury After CAIR	(D) 2020 Residual Utility Attributable Fish Tissue Methylmercury After CAMR Requirements	2020 Estimated IDI Value of Methylmercury After CAIR (A*C*1.5/B/0.1)	2020 Estimated IDI Value of Methylmercury After CAMR (A*D*1.5/B/0.1)
5th	0.95	47.4	0.000	0.000	0.00	0.00
10th	1.38	49.6	0.001	0.001	0.00	0.00
15th	1.76	51.4	0.002	0.002	0.00	0.00
25th	2.50	54.3	0.004	0.004	0.00	0.00
50th	4.88	60.9	0.010	0.009	0.01	0.01
75th	9.48	69.6	0.020	0.017	0.04	0.03
85th	13.61	78.4	0.027	0.024	0.07	0.06
90th	17.29	84.1	0.035	0.032	0.11	0.10
95th	24.46	93.5	0.052	0.047	0.20	0.18
99th	46.6	93.5*	0.102	0.093	0.76	0.70

* The exposure factors handbook does not report the 99th percentile of the bodyweight distribution.

6.B. Subsistence Fishing Population

As discussed in Section 3 of this document, consumption rates for subsistence fishers are likely to be much higher than for the general population of recreational anglers. As such, these populations are more likely to experience an IDI value above 1 if they fish in areas that have a higher utility attributable methylmercury level. Data on consumption rates for the population of subsistence fishers is much more limited. In addition, because these populations are located in very specific areas of the U.S., the general distribution of fish tissue methylmercury concentrations across the U.S. may not be representative of the fish tissue concentrations in fish consumed by these population. Specific data on concentrations in fish at waterbodies frequented by subsistence fishing populations has not been generated. As such we use the overall distribution of fish tissue methylmercury concentrations, recognizing that it may not capture the specific distribution for the subsistence subpopulation.

Based on the EPA Exposure Factors Handbook, Table 10-85, the recommended mean consumption rate for Native American subsistence fishers is 59 grams/day, with a 95th percentile consumption rate at 170 grams/day. This can be fit to a lognormal distribution which has a mean of 59 g/day and a standard deviation of 63 g/day. We use this fitted distribution for the following analysis.

Table 6.3 shows the IDI value for subsistence fishing groups associated with the 99th percentile of utility attributable methylmercury and 99th percentile of consumption rates, modeled to be 295 g/day, for the 2001 base case, 2020 after implementation of CAIR, and 2020 after implementation of the CAMR requirements. It also shows the percentile of methylmercury concentrations at which the IDI value would be equal 1 holding the consumption rate constant, and the percentile of subsistence fishing consumption rates at which the IDI value would be equal to 1 holding the methylmercury percentile constant.

Table 6.3. Average Index of Daily Intake (IDI) Levels Associated with Upper Percentile Methylmercury Concentrations and Native American Subsistence Fish Consumption Rates

Analytical Scenario	IDI values at 99th percentiles of Methylmercury Exposure and Consumption Rates	Percentile of Methylmercury Distribution at Which the IDI Values Do Not Exceed 1 (holding consumption rates at the 99th percentile)	Percentile of Consumption Rate Distribution at Which the IDI Value Do Not Exceed 1 (holding methylmercury concentrations at the 99th percentile)
2001 Base Case	17.8	32.5th	16.5th
2020 Implementation of CAIR	7.05	66.5th	52.0th
2020 Implementation of CAMR	6.43	71.8th	55.2th

In the 2001 base case, at the 99th percentiles of concentrations and Native American subsistence fisher consumption rates the IDI value for subsistence populations would be almost 18. The IDI value would exceed 1 as long as the methylmercury concentration exceeded the 32nd percentile (holding consumption rates at the 99th percentile) or the fish consumption rate exceeded the 16th percentile (holding the methylmercury concentrations at the 99th percentile). Thus for these highly consuming groups, there is a reasonable chance that the IDI value will exceed 1 in 2001, if those groups are in locations that are moderately affected by mercury deposition from utilities.   After implementation of CAIR, the estimated IDI values associated with the 99th percentile of methylmercury and consumption rates falls by 60 percent, but is still an IDI value of 7. After CAIR, the IDI values would exceed 1 as long as the methylmercury concentrations exceeded the 66th percentile (holding consumption rates at the 99th percentile) or the fish consumption rate exceeds the 52nd percentile. Thus, even after CAIR, there remains a chance that the IDI value will exceed 1, if high fish consumption groups fish in locations highly affected by utility emissions.   Using 2000 Census data, Figure 6-1 shows the location of Native Americans in tribal census tracts relative to utility attributable deposition after implementation of CAIR in 2020. Tribal census tracts are small, relatively permanent statistical subdivisions of a federally recognized American Indian reservation and/or off-reservation trust land. The optimum size for a tribal census tract is 2,500 people; it must contain a minimum of 1,000 people. Visual inspection shows very few locations where Native Americans live where there is also high residual deposition due to utilities. This suggests that the 99th percentile of the utility attributable methylmercury concentrations is likely inappropriate as an upper bound for Native Americans' subsistence fishing exposures, and thus the IDI based on combining the 99th percentile utility attributable methylmercury concentration and the Native American subsistence fishers' consumption rates will overstate the IDI for Native Americans subsistence fishers'.
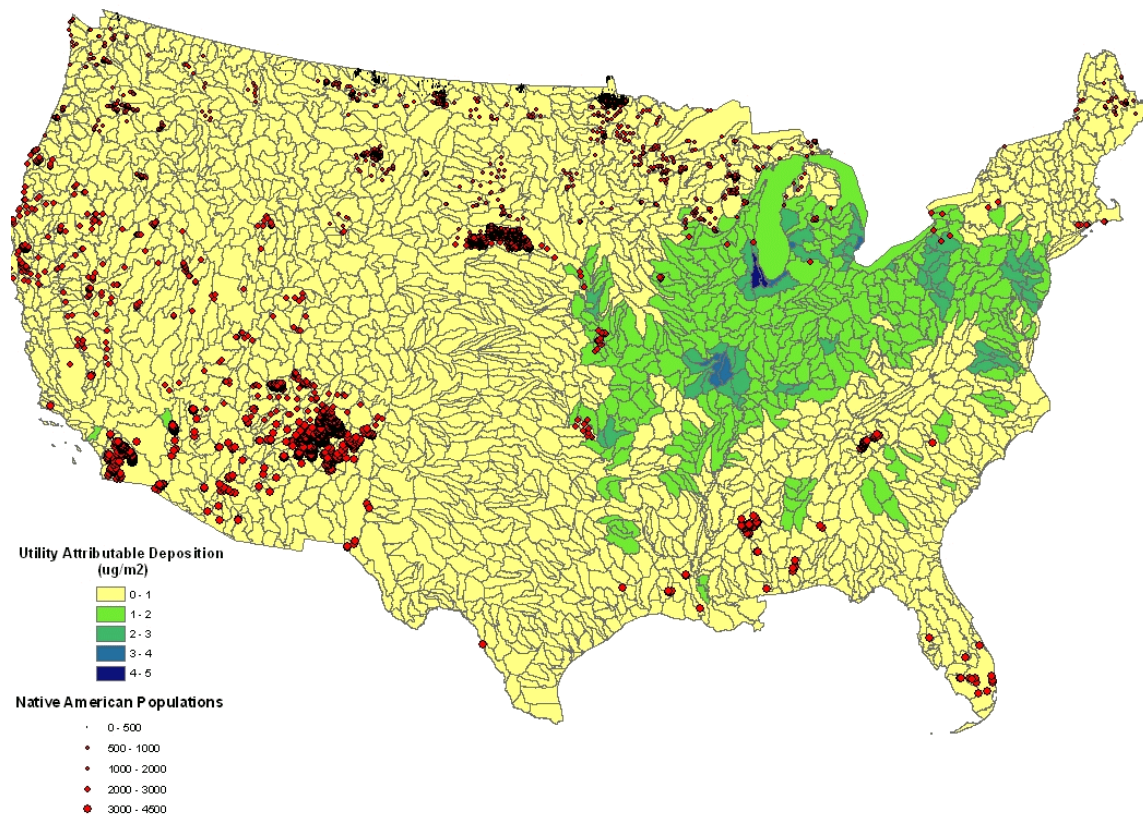
Figure 6.1. Locations of Native American Populations in Tribal Census Tracts Relative to Utility Attributable Mercury Deposition After Implementation of CAIR in 2020

After CAMR, the IDI value exceeds 1 for the cases when the 99th percentile consumption rate is combined with the 99th percentile methylmercury concentration, and the IDI value exceeds 1 as long as methylmercury concentrations are greater than the 72nd percentile (holding consumption rates at the 99th percentile) or consumption rates are greater than the 55th percentile (holding methylmercury concentrations at the 99th percentile). However, as with CAIR, the 99th percentile of overall utility attributable methylmercury concentrations likely exceeds the 99th percentile of methylmercury concentrations for fishing locations frequented by Native American subsistence fishing populations.

It is important to note that these calculations are based on a mean bodyweight that is not correlated with consumption rates. Increasing the bodyweight (to reflect a correlation between higher consumption and higher bodyweight) by only 10 percent (less than one standard deviation from the mean) would reduce the IDI value to 1. In addition, the potential for exposures above an IDI of 1 should be viewed with consideration of the conservativism built into the derivation of

the RfD which is the basis for the definition of the IDI reference point.

Table 6.4 shows the estimated IDI for various combinations of alternative consumption rates and percentiles of methylmercury concentrations. This table suggests that after implementation of CAIR in 2020, there are only a few circumstances where the IDI exceeds 1.0. With CAMR there is less chance of an exceedance. These cases occur only at the highest percentiles of methylmercury and consumption rates. It should be noted that in many elements of this analysis, we have made assumptions that would likely lead to over-estimates of possible exposure to utility attributable fish tissue methylmercury. These include the decision to not screen out sampling locations that may be influenced by non-air deposition sources of mercury, the use of the maximum averaged species concentration at each sampling location, and the focus on the extreme percentiles of the methylmercury and consumption rate distributions.

Table 6.4. Fish Tissue Concentration Percentiles and EPA EFH and OW Fish Consumption Rates and Resulting IDI values

	Consumption Rate	Fish Tissue Methylmercury Percentile									
		5th	10th	15th	25th	50th	75th	85th	90th	95th	99th
2001 Base Case Methylmercury		0.000	0.002	0.004	0.008	0.030	0.064	0.09	0.113	0.164	0.257
EPA EFH Mean Recreational Fisher	**8**	0.00	0.00	0.01	0.02	0.06	0.12	0.17	0.21	0.31	0.48
EPA OW 90th Percentile General Population	**17.5/11.7***	0.00	0.01	0.02	0.02	0.08	0.18	0.25	0.31	0.45	0.70
EPA EFH 95th Percentile Recreational Fisher	**25**	0.00	0.01	0.02	0.05	0.18	0.38	0.53	0.66	0.96	1.51
EPA EFH 99th Percentile Recreational Fisher	**47**	0.00	0.02	0.04	0.09	0.33	0.71	0.99	1.24	1.81	2.83
EPA EFH Mean Subsistence Native American	**60**	0.00	0.03	0.06	0.11	0.42	0.90	1.27	1.59	2.31	3.61
EPA EFH 95th Percentile Subsistence Native American	**170**	0.00	0.08	0.16	0.32	1.20	2.55	3.59	4.50	6.53	10.24
EPA EFH 99th Percentile Subsistence Native American	**295**	0.00	0.14	0.28	0.55	2.07	4.43	6.22	7.81	11.34	17.77
2020 with CAIR Methylmercury		0.000	0.001	0.002	0.004	0.010	0.020	0.027	0.035	0.052	0.102
EPA EFH Mean Recreational Fisher	**8**	0.00	0.00	0.00	0.01	0.02	0.04	0.05	0.07	0.10	0.19
EPA OW 90th Percentile General Population	**17.5/11.7***	0.00	0.00	0.01	0.02	0.03	0.06	0.07	0.09	0.14	0.28
EPA EFH 95th Percentile Recreational Fisher	**25**	0.00	0.01	0.01	0.02	0.06	0.12	0.16	0.21	0.30	0.60
EPA EFH 99th Percentile Recreational Fisher	**47**	0.00	0.01	0.02	0.04	0.11	0.22	0.30	0.39	0.57	1.12
EPA EFH Mean Subsistence Native American	**60**	0.00	0.01	0.03	0.06	0.14	0.28	0.38	0.49	0.73	1.43
EPA EFH 95th Percentile Subsistence Native American	**170**	0.00	0.04	0.08	0.16	0.40	0.80	1.08	1.39	2.07	4.06
EPA EFH 99th Percentile Subsistence Native American	**295**	0.00	0.07	0.14	0.28	0.69	1.38	1.87	2.42	3.60	7.05
2020 with CAIR + CAMR Requirements Methylmercury		0.000	0.001	0.002	0.004	0.009	0.017	0.024	0.031	0.047	0.092
EPA EFH Mean Recreational Fisher	**8**	0.00	0.00	0.00	0.01	0.02	0.03	0.05	0.06	0.09	0.17
EPA OW 90th Percentile General Population	**17.5/11.7***	0.00	0.00	0.01	0.02	0.03	0.05	0.07	0.09	0.13	0.26
EPA EFH 95th Percentile Recreational Fisher	**25**	0.00	0.01	0.01	0.02	0.05	0.10	0.14	0.18	0.28	0.54
EPA EFH 99th Percentile Recreational Fisher	**47**	0.00	0.01	0.02	0.04	0.10	0.19	0.26	0.34	0.52	1.01
EPA EFH Mean Subsistence Native American	**60**	0.00	0.01	0.03	0.06	0.13	0.24	0.34	0.44	0.66	1.29
EPA EFH 95th Percentile Subsistence Native American	**170**	0.00	0.04	0.08	0.16	0.36	0.68	0.96	1.24	1.87	3.67
EPA EFH 99th Percentile Subsistence Native American	**295**	0.00	0.07	0.14	0.28	0.62	1.18	1.66	2.14	3.25	6.36

Note: EFH is Exposures Factors Handbook. See http://www.epa.gov/ncea/pdfs/efh/front.pdf. *The 17.5 g/day value is uncooked weight. We use the cooked-weight equivalent value of 11.7 g/day for the calculation of IDI values. Due to this calculation, some rounding errors may occur. However, errors will be in the

direction of overstating IDI values.

7. Utility Report to Congress Modeling

In the Study of Hazardous Air Pollutant Emissions from Electric Utility Steam Generating Units -- Final Report to Congress (U.S. EPA, 1998), referred to hereafter as the Utility RtC, EPA conducted a screening level analysis of deposition patterns likely to result from utility emissions of mercury. That analysis found that some mercury emissions have the potential to deposit locally and accumulate in fish tissue of local water bodies. However, the analysis was based on hypothetical situations and was not suitable for estimating the frequency of any given fish tissue or exposure level. The Utility RtC presented results from modeling that are superseded by the modeling results presented above. In particular, the Agency views the application of a more robust modeling approach as critical and required for assessing the mercury deposition associated with CAMR because of the density and properties of mercury and its complex transport and reactions in the atmosphere. The recently developed CMAQ modeling system meets our requirements and the recommendations of the Report to Congress for a "single air quality model" to address Mercury deposition.

Based on our current modeling we now recognize that the Utility RtC model plant analysis modeling does not appear to represent the central tendency nor did it attempt to describe the distribution of impacts nationally. The conclusion of the Utility RtC (with regard to the 230kg/day model facility) is currently thought to overstate the impact in the majority of locations nationally. In some (small) set of locations, however, conditions may be similar to that projected by the Utility RtC scenario. As described below, the Agency is pursuing additional research to better understand these locations and any potential for "utility hotspots."

In responding to the numerous comments received by the Agency on the January 2004 proposal, the March 2004 Supplemental Notice of Proposed Rulemaking (SNPR) and the December 2004 Notice of Data Availability (NODA), we performed model plant analyses similar to those done in the Utility RtC in that they employed the same dispersion model (ISC3), but these new analyses differ from the Utility RtC analysis in certain ways intended to help us better understand important factors influencing near-field deposition. The Utility RtC assessment also included watershed modeling of the projected local deposition. That modeling and more recent watershed modeling of mercury deposition are described in ecosystem modeling below.

7.1 Speciation Profile of Mercury Emissions

The Agency has learned much more about mercury emissions from the power plant sector since the Utility RtC. Pursuant to areas of further research identified in the Utility RtC in 1998, the Agency conducted an Information Collection Request (ICR) for all coal-fired power plants in the U.S. which were greater than 25 MW. To that end, the Agency sampled incoming coal shipments for approximately 450 coal-fired power plants for calendar year 1999 to determine the concentration of mercury and chlorine in these coals, and to quantify the heat

content (British Thermal Units, Btu) and ash content. Additionally, the Agency selected 81 individual plants for emissions testing during 1999, which were representative of the larger fleet of sources. These two pieces of data greatly refined the Agency's understanding of mercury emissions from coal-fired power plants. Among the most notable was the overall speciation of mercury emissions from coal-fired power plants. Table 7.1. summarizes the differences in the speciation profiles for both the Utility RtC and the 1999 ICR.

Table 7.1: Speciation Assumptions in Utility Report compared to results of 1999 ICR

	Utility Report values for model plant scenarios	1999 ICR (average of 81 plants)
Percent Mercury(0)	50%	54%
Percent Mercury(++)[1]	30%	43%
Percent Mercury(p)[1]	20%	3%

[1] Divalent mercury occurs in the ionic form, abbreviated here as Mercury(++) and in particle-bound compounds, abbreviated here as Mercury(p).

As can be seen from Table 7.1, the mean percentage of elemental mercury from the 1999 ICR matches fairly well with the values used in the Utility model plant, while the percentages of the ionic and particulate divalent mercury differ somewhat.

7.2 Industrial Source Complex Version 3 (ISC) modeling

In the Industrial Source Complex Version 3 (ISC) modeling done at the time of the Utility RtC, it was found that the highest deposition and associated concern about localized exposure occurred when a lake was located 2.5 km from a large coal-fired power plant theoretically sited in a humid eastern U.S. location, (i.e., subject to the meteorology of this location). This lake and surrounding watershed was estimated to receive an average deposition rate of 15.5 ug per square meter per year from the large coal-fired power plant[21]. One concern raised in the Utility RtC is summarized in the Table 7-10 (page 7-37) of that report. The report found that, under a hypothetical but not impossible scenario, a subsistence fisher could be exposed to an IDI level of 3.7. Our more recent modeling allows us to better assess the probability of this event (See Section 6).

In the Utility RtCt, a model plant was sited in two different locations, one a humid eastern site and the second an arid western site.   In the more recent work, we modeled air

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[21] See table 7-10. 17.9ug/m^2/year from all power plants, 87% of which is from the single facility. 17.9*.87=15.5 ug/m^2/year.

quality and deposition for four locations representing four different meteorological regimes across the U.S. - Phoenix, Arizona; Kansas City , Missouri; Indianapolis, Indiana; and, Tampa, Florida.   Phoenix, Arizona is located in the Sonoran Desert in the Southwestern U.S., and is representative of an arid climatic regime. Temperatures range from very hot in the summer months to relatively mild during winter, and generally experience large diurnal temperature fluctuations and light winds. Kansas City, Missouri is located near the geographical center of the continental U.S. and exhibits a modified continental climate. Indianapolis, Indiana is located in the western reaches of the Ohio Valley, and has a classic temperate climatic regime - very hot during summer, and often bitterly cold during winter. Tampa, Florida is located in the central Gulf Coast region of the Florida peninsula, and exhibits a strong subtropical climatic regime. The local climate is modified extensively by the existence of daily land and sea breezes. Thus, these 4 climatic regimes typify the range of conditions expected at the majority of major U.S. power plant sites.

The 1997 Mercury Study Report to Congress (EPA, 1997b) noted that "a single air quality model which was capable of modeling both the local as well as regional fate of mercury was not identified." Thus, the modeling approach used for the Mercury RtC employed two models: 1) the Regional Lagrangian Model of Air Pollution (RELMAP) to address regional-scale atmospheric transport, and 2) the Industrial Source Code model (ISC3) to address local-scale analyses (i.e., within 50 km of source). The ISC3 model discussed here is a Gaussian plume dispersion model routinely used by the Agency for local-scale air quality modeling for air toxics. It differs in many ways from other models used for modeling mercury distribution and deposition nationally. The current state-of-the-science national models include three-dimensional eulerian grid models such as the Community Multi-Scale Air Quality (CMAQ) model and the Regional Modeling System for Aerosols and Deposition (REMSAD). As described in the December 2004 NODA, the Agency has continued to refine its models and scientific understanding of the fate, transport and deposition of mercury and the subsequent cycling through the terrestrial and aquatic ecosystems. See the CAMR RIA Chapter 3 (Ecosystem) and Appendix A (Case studies) for a discussion of the ecosystem modeling.

As mentioned above the objective in the more recent ISC3 modeling was to assess the relative importance of geography, emissions profile and stack height on localized/nearfield deposition patterns. Specifically, we have applied these modeling scenarios to the four climatic regimes described above. The model results were based on a hypothetical coal-fired power plant emitting 1 kilogram (kg) of either speciated mercury (mercury(++), mercury(p) or mercury(0)) per day from either a 50 m, 250 m or 500m stack. The design of the scenarios was such that the results would yield evidence on the relative importance of the species of mercury being emitted, and the sensitivity of the local/nearfield (< 50 km) deposition to both stack height and local meteorological conditions.

The results show that the highest deposition occurs when 100% mercury(++) is emitted from the stack in the more humid climatic regions (e.g., Kansas City, Indianapolis and Tampa) and this deposition is dominated by wet deposition in the 5-10 km downwind distance. In general, beyond 10-15 km downwind, dry deposition becomes the dominant form of local

deposition (See Figures 1,2 and 3 in Appendix A). In general, the deposition patterns associated with a 100 % release of mercury(p) showed depositional patterns approximately half of the rates observed for the oxidized mercury analyses. Consistent with discussion in the Utility Report, deposition patterns for emission of 100 % mercury(0) yielded extremely low (< 5 g/m2/yr) deposition totals for all four sites. In assessing the changes in deposition in arid climatic conditions, dry deposition becomes the dominant mode of deposition, as would be expected, and the total deposition is somewhat reduced (see Figure 1 in Appendix A). These results are consistent with the Utility RtC which indicated that wet deposition was believed to be the dominant form of mercury deposition in the continental U.S.

Discussion of stack height importance:

In analyzing atmospheric transport and deposition, several physical parameters are of key importance - wind speed, stack temperature and stack exit velocity. Stack temperature and exit velocity contribute to plume rise, or the buoyancy in the plume as it enters the ambient atmosphere. In determining the importance of a source to contributing to local deposition, plume buoyancy and stack height are of key importance.

Wind profiles near the surface are logarithmic in nature, with wind speeds increasing exponentially as one moves away from the surface. Thus, the higher the stack height (or source of the emissions) the greater the wind speed encountered by the mercury emissions being released from the stack. The impact of higher wind speeds on deposition is to effectively carry the emissions further downwind, or farther from the source. Holding stack temperature and exit velocity constant/similar (constant buoyancy), the height of the mercury emissions release is a key factor in defining the ambient wind speed encountered by the mercury emissions and thus the downwind transport and deposition of the emissions.

Appendix A documents the importance of these concepts by examining the depositional difference for oxidized mercury (the most important species of mercury for local-scale deposition) given 3 different stack heights - 50 m, 250 m and 500 m. In the figures, for four distinct geographic regions (Tampa, Florida; Phoenix, Arizona; Indianapolis, Indiana; Kansas City, Missouri), it is evident that at the lower stack heights (50 m), holding other emissions characteristics constant, nearfield deposition is increased by an order of 3 times (based on the mercury deposition index, explained elsewhere in this document). When reviewing the figures for the higher stack height (500 m), the maximum deposition is shifted downwind because of the higher release. The figures for the 250 m stack are generally representative of a normal coal-fired power plant in the U.S.

Assessing the impact of stack height on local/nearfield deposition indicated that for the scenario of 100 % mercury(++), a stack height of 50 m resulted in 3-5-fold increase in nearfield deposition (< 15 km downwind) over the deposition observed with a 250m stack height and deposition was completely dominated by the dry deposited fraction. On the other hand, an increase in stack height from 250m to 500 m resulted in only minor changes in the downwind deposition patterns for the three eastern-most sites. For the Phoenix location, the higher stack

resulted in approximately a 50 % reduction in nearfield deposition. In general, the results of these revised analyses are supportive of the conclusions reached in the Utility RtC which indicated that localized deposition was significantly impacted by the physical parameters of the coal-fired power plant (e.g., stack height, emissions rate, etc.), local climatic conditions and intensity of precipitation events.

7.3 Ecosystem Modeling

Assessment of the IEM-2M RtC Model and other Ecosystem Scale Modeling
In the Utility RtC, EPA modeled a "hypothetical" ecosystem at both an eastern and a western site. In the recent work (Chapter 3 of the RIA), EPA considers the relationship between declines in deposition and fish mercury concentrations at five real ecosystems and considers a range of ecological characteristics that affect the magnitude and response times of different ecosystems. The ecosystems modeled include: Eagle Butte, SD; Lake Waccamaw, NC; Lake Barco, FL; Brier Creek, GA; and Pawtuckaway Lake, NH. Since the RtC was released in 1997, EPA has developed and refined a set of watershed, waterbody, and food web models that describe the speciation, transport, and bioaccumulation of mercury as a function of the physical, chemical and biological properties of different ecosystems. For consistency with previous work, an updated version of the original IEM-2M model applied in the RtC (SERAFM) was applied with several other models to describe the fate and bioaccumulation of mercury in the systems listed above. We also compared projected fish mercury levels at the "hypothetical ecosystem" described in the RtC using the IEM model to the SERAFM forecasted values. These results are described in greater detail below.

State variables in both the IEM-2M and SERAFM models include three mercury species, mercury(0), mercury(++), and methylmercury. SERAFM includes four solids types and dissolved organic carbon, DOC. Both IEM-2M and SERAFM simulations are driven by external mercury loadings delivered from the atmosphere, from watershed tributaries, and from point sources, or by internal loadings from contaminated sediments. However, at the time the IEM-2M model was developed, inputs of methylmercury from the watershed were not quantified. This can be a significant source of methylmercury in may aquatic ecosystems (see Chapter 2, RIA for more detail).

The SERAFM model incorporates more recent advances in scientific understanding described above and implements an updated set of the IEM-2M solids and mercury fate algorithms described in detail in the Mercury Study RtC (USEPA, 1997b). These updates provide more realistic representations of the processes governing mercury fate and transport in aquatic systems. Major differences between the SERAFM model and the IEM-2M model are as follows:

• Dynamic calculations: SERAFM can describe the temporal response of fish mercury concentrations to changes in mercury loading, while the IEM-2M model calculated expected fish tissue mercury concentrations at steady state.

• <u>Watershed Loading</u>: Both IEM-2M and SERAFM model soil erosion into the water body using the Revised Universal Soil Loss Equation (RUSLE). However, in SERAFM mercury loading from the watershed to the water body is modeled using run-off coefficients. SERAFM defines four land-use types: impervious, upland, riparian, and wetland/forest. The user defines the percentage of each type in the watershed. The model uses run-off coefficients to describe mercury from atmospheric deposition to each land type as loadings to the water body. IEM-2M calculates mercury concentrations in soils, and calculates erosion and transport to the water body.

• <u>Two-Layer</u>: SERAFM has the capability to model a layered lake system with an epilimnion and hypolimnion, while IEM-2M used a single, well mixed layer to represent the water column.

• <u>Photo-reactions</u>: Recent research has demonstrated the photo-reactions of mercury. These have been incorporated into SERAFM but were not part of the original IEM-2M model. The oxidation and reduction of mercury as functions of visible and UV-B light are included.

• <u>Speciation</u>: Speciation of mercury with hydroxides, chlorides, and sulfides has been included in the SERAFM model but was not incorporated in the IEM-2M model. The abiotic oxidation rate constant for Mercury++ is multiplied by the fraction of dissolved divalent mercury and the fraction of Mercury++ present as Mercury(OH)$_2$.

• <u>Equilibrium Partitioning</u>: SERAFM models equilibrium partitioning between multiple compartments or phases: aqueous phase, abiotic particles (silts/fines), biotic particles (phytoplankton, zooplankton, seston), and DOC-complexation. In SERAFM, the biotic demethylation rate constant is multiplied by the sum of the fraction dissolved and the fraction DOC-complexed, as suggested by previous research (Matilainen and Verta, 1995).

• <u>Trophic status</u>: Trophic status of the lake has been incorporated into the SERAFM model and was not a component of the IEM-2M model. Trophic status is used to calculate visible light attenuation in the lake, the turnover of biomass, and the phytoplankton and zooplankton concentration in the SERAFM model framework.

• <u>Suspended particle types in the water column</u>: The SERAFM model accounts for both zooplankton and phytoplankton as biotic materials in the system, while IEM-2M only accounted for one biotic particle type.

• <u>Reaction rates</u>: The SERAFM model incorporates more recent reaction rate coefficients, and the understanding of the variability of these rates with different conditions.

• <u>Partition coefficients</u>: The SERAFM model incorporates more recent values for

mercury partition coefficients for each mercury species. Future versions of the SERAFM model will calculate site-specific partitioning as a function of sediment organic matter and the organic carbon content of suspended materials.

A preliminary comparison of the SERAFM model to the IEM-2M model using the parameter values for the model ecosystem described in the RtC suggests that updates to the IEM-2M model incorporated into the SERAFM model result in lower values for fish mercury concentrations (Table 7.2), by up to a factor of two. However, the model ecosystem described in the RtC uses a lower dry deposition rate than estimated based on more recent understanding and assumes that there is no watershed methylmercury loading (discussed above). At the time the RtC modeling was conducted, best estimates of dry deposition were approximately 50% the magnitude of wet deposition. More recent modeling described in the RIA suggests that on average, dry deposition is better approximated as 50% of total deposition, a significant increase over previous estimates. This is clearly a major source of uncertainty when attempting to quantify inputs of mercury to different aquatic systems and characterize the contribution of different sources to fish mercury concentrations. When the parameters are updated to reflect current knowledge, forecasted fish mercury concentrations are higher than the original IEM-2M results (Table 7-2). We note, however, that because we did not run IEM-2M with these updated parameters, a clear comparison of the IEM-2M and SERAFM models themselves cannot be made in this instance (original IEM-2M versus updated SERAFM model runs). The differences in the predicted concentrations reflect, in part, input assumptions that are not related to these ecosystem models (e.g. differences in assumptions regarding air deposition). Although this preliminary comparison provides some perspective on the differences between these two models, it is difficult to draw from this comparison general conclusions that are applicable across all ecosystems and scenarios. In part, for these reasons, we conclude, at this point, that the ecosystem models (e.g. SERAFM) and case studies (described below) are primarily useful as a tool for evaluating the national-scale assessment described in sections 1- 6 of this document.

Table 7.2. Comparison of SERAFM and IEM-2M

Parameters	RtC Model Ecosystem	RtC Model Ecosystem	Updated Parameters
Model	IEM-2M	SERAFM	SERAFM
Water Column Me mercury Unfiltered	0.08	0.031 NG L-1	0.12 nG L-1
Water Column MercuryT Unfiltered	1.16 nG L-1	2.50 nG L-1	1.17 nG L-1
Trophic Level 4 Fish	0.44 ug g-1	0.21 ug g-1	0.80 ug g-1

Trophic Level 4 Fish BAF: FishMercury/Me mercury	6.8×10^6	6.8×10^6	6.8×10^6

Note: Comparison of SERAFM and IEM-2M forecasted mercury concentrations using parameter values for model ecosystem described in the Mercury Study RtC (EPA, 1997b) and a 50% reduction in atmospheric deposition. The "Updated Parameters" column refers to modification of the original model ecosystem described in the RtC to incorporate more recent knowledge on the magnitude of dry deposition and inputs of Me mercury from the catchment.

Based on the BAFs considered, the hypothetical ecosystem described in the RtC is more sensitive than three out of four ecosystems modeled as case studies (see Table 7-3 below and Chapter 3, RIA), and is less sensitive than one.

Table 7.3. Empirically derived BAFs for each of the ecosystem case studies

BAFs (L/kg)	MeHg (nG/L)	Mercury T (nG/L)	BAF-MeHg	BAF - MercuryT
Eagle Butte	0.82	10.2	8.90E+05	8.73E+04
Lake Barco	0.018	1.03	3.06E+07	5.34E+05
Lake Pawtuckaway	0.19	2.26	1.11E+06	9.29E+04
Lake Waccamaw	0.48	4.79	1.24E+06	3.86E+04
Hypothetical Ecosystem (USEPA 1997)	--	--	6.80E+06	5.30E+05

Comparing these case studies to empirically derived BAFs characterized by the Office of Water indicates that modeled fish tissue responses in three of four case studies has empirically derived BAFs that fell between the 5th and 50th percentiles of the geometric mean of field-measured BAFs for trophic level four species obtained from the published literature (EPA 2000). The model ecosystem described in the RTC fell between the 50th and 95th percentile for the BAFs identified in the OW Methylmercury Water Quality Criterion Appendix A (U.S. EPA, 2001) (see http://www.epa.gov/waterscience/criteria/methylmercury/document.html) and one of the case studies (Lake Barco) exceeded the 95th percentile. Some limitations to the BAF approach deserve mention. Because methylmercury concentrations in the water column are highly variable, empirically-derived BAFs are inherently undetermined and have limited predictive power. A more credible approach based on our current knowledge is to forecast changes in fish mercury concentrations using information on the food web dynamics ("bioenergetics") of different ecosystems. Such a model (BASS) was applied in one of the case studies described in the RIA. Results showed that the BAFs calculated from the outputs of the bioenergetics based bioaccumulation model were within a factor of two of the empirically derived BAF used in the SERAFM model.

Acknowledging the many uncertainties in this analysis, results generally suggest that the

forecasted contribution of US power plants to fish mercury concentrations may have been overpredicted in the hypothetical ecosystems described in the original RtC relative to the majority of aquatic systems that could potentially be affected by mercury deposition from US power plants. However, it is important to note that fish tissue methylmercury concentrations due to power plants may be higher in some ecosystems (for example, ecosystems similar to Lake Barco described in Chapter 3 of the RIA).

8.0   Data Limitations, Uncertainty, and the Need for Further Work

Based on the analyses presented in this document, several areas exist where additional data may provide for more clear understanding of the potential for continued exposures to utility attributable methylmercury concentrations after implementation of CAIR and CAMR. Figure 8.1 shows the location of post-1999 fish sampling locations relative to utility attributable mercury deposition after implementation of CAIR and CAMR in 2020. In areas with residual utility attributable deposition, including portions of the Ohio Valley, the Southern Great Lakes, including the area around Chicago, and portions of Missouri and Southern Illinois, including watersheds surrounding the Missouri and Mississippi Rivers, there are many gaps in the fish sampling data. As such, there is uncertainty as to the methylmercury fish tissue concentrations due solely to power plants in these areas.
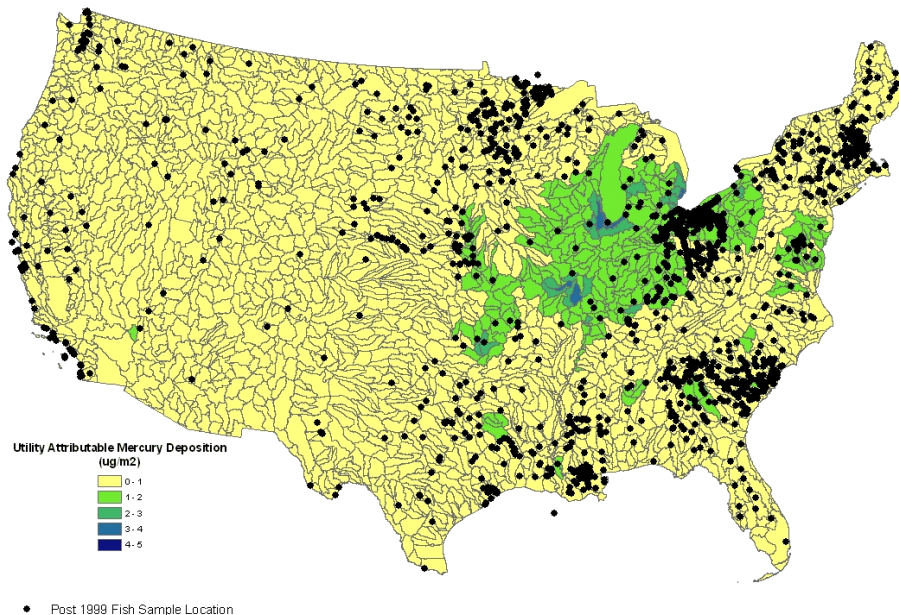

Figure 8.1. Location of Post-1999 NLFA and NLFTS Sample Sites Relative to 2020 Utility Attributable Mercury Deposition After Implementation of CAIR and CAMR Requirements

As with any analysis based on limited data, and especially analyses of future conditions, there is inherent uncertainty in the estimates of all analytical outputs of our modeling. EPA typically classifies the major areas of uncertainty in risk assessments as parameter uncertainty, scenario uncertainty, and model uncertainty. Parameter uncertainty is simply uncertainty about some parameter used in the analysis. The sources of parameter uncertainty are measurement errors, sampling errors, variability, and use of generic or surrogate data. Scenario uncertainty is uncertainty about missing or incomplete information needed to fully define the exposure and dose. Scenario uncertainty results from the general impracticality of making actual measurements of receptors' exposure. The sources of scenario uncertainty are descriptive errors, aggregation errors, errors in professional judgment, and incomplete analysis. Model uncertainty results from the fact that models and their mathematical expressions are simplifications of reality that are used to approximate real-world conditions and processes, and their relationships. Models do not include all parameters or equations necessary to express real-world conditions because of the inherent complexity of the natural environment, and the lack of sufficient data to describe the natural environment. Consequently, models are based on numerous assumptions and simplifications, and reflect an incomplete understanding of natural processes.

In conducting this analysis, EPA endeavored to use state of the science models and algorithms, as well as the best available parameter data of known and documented quality. Nevertheless, EPA understands that all analysts who conduct complex assessments of the type presented in this document are faced with numerous sources of uncertainty. Throughout this document we describe sources of uncertainty in our analysis, as well as our efforts to more fully understand and mitigate those uncertainties.   In cases where sophisticated modeling tools or complete data were not available, we undertook additional effort to identify and understand their potential impact on our analysis. The analysis described in this document represents a reasonable and considered approach to evaluating a highly complex problem. We believe that the decisions we have made in the face of unavoidable uncertainty have reduced the chance that we have either significantly over- or underestimated the methylmercury exposures to consumers of noncommercial freshwater fish. In cases where over- or under-estimates exist, we have made these estimates transparent.

Additional Monitoring Needs

Based on the demonstrated gaps in available fish tissue sampling data, EPA is encouraging states, utilities, and other interested parties to work with the Agency in collecting data on mercury deposition and fish samples in and around waterbodies near where utilities are projected to have residual emissions of those mercury species that contribute to near-field or regional deposition. Collection of this data will ensure that CAIR and CAMR are resulting in the reductions in utility attributable deposition and fish tissue methylmercury concentrations as anticipated. Furthermore, the ongoing collection of data by state agencies under the NLFA and the EPA NLFTS will continue to provide fish sampling data that can be used to assess the effectiveness of CAIR and CAMR over time.

EPA also recognizes the potential for certain populations, including subsistence fishers

within the population of Native Americans, to experience greater exposure to utility attributable methylmercury due to high consumption of fish. While the CAIR and CAMR cannot be demonstrated to reduce all exposure from utility mercury emissions, Table 6.4 shows that under most circumstances, the chances of incremental exposures above an IDI value of 1 are small. EPA will continue to work with Native American tribes to provide more complete information about levels of methylmercury in fish caught in waterbodies frequented by Native American subsistence fishing populations.

EPA also recognizes that there are other subsistence fishing populations outside of the Native American community, including low income and minority populations located in urban and rural areas who eat large amounts of self-caught freshwater fish. These populations may have higher consumption rates then the general population of recreational anglers. However, very little information is available on the fishing behaviors of this population. As such, while additional collection of fish samples will allow for a more complete understanding of the extent of residual fish-tissue methylmercury levels due to utilities, population surveys of fish consumption in these subpopulations will also be useful in helping to characterize remaining exposure to utility attributable methylmercury.

Mercury Vulnerability Analysis

Our current national-scale analysis uses the Mercury Maps proportionality assumption to relate changes in deposition to changes in fish tissue. This approach has a number of limitations as described above . In part to help address some of these limitations, EPA is currently engaged in ongoing work to analyze the vulnerability of different ecosystems across the United States to methylmercury exposure. Different ecosystems exhibit dramatically different responses to changes in mercury loading depending on their chemical and physical attributes. Using georeferenced empirical databases that describe some of the watershed and waterbody characteristics across the United States, we are exploring a preliminary approach to such an analysis by presenting the gradient of values for some attributes known to be important for methylmercury formation and bioaccumulation in fish. In this analysis, we will consider sulfate deposition, organic matter in soils, percent wetland coverage, and total mercury deposition forecasted using the CMAQ model as proxy indicators for methylation potential. We define "methylation potential" as a relative indicator of the likelihood that mercury deposited in a given ecosystem will be converted to methylmercury and bioaccumulate in fish. A number of variables important for methylmercury formation have not yet been included in this analysis and the results must therefore be considered a work in progress and are by no means conclusive. We, therefore, do not present the results here and do not rely on them for this rulemaking.

Different variables and combinations of variables will produce different results for the most vulnerable regions or enhanced areas of methylmercury production. Developing broad categories of ecosystem types based on their propensity for methylmercury formation and bioaccumulation in fish and their frequency of occurrence is an iterative effort. Such work will help us better characterize case studies and empirical about particular ecosystems within the range of ecosystems found across the US. It also will allow us to focus better on individual

watersheds or waterbodies that may be of concern as "utility hotspots."

In addition to the above analysis, EPA's Regional Vulnerability Assessment Program (ReVa) has developed a pilot internal version of an environmental decision toolkit (EDT) to assess mercury vulnerability across the United States. While recognizing that both our process based understanding of the mercury exposure pathway and the availability of data to make this assessment are currently imperfect, the ReVA program develops effective ways to make use of data and information that currently exist. Given that there is no obvious "right" way to assess the risk from methylmercury, a toolkit with the flexibility to consider and compare alternative data, model inputs, and assumptions, and alternative ways to combine these inputs into indices of relative risk will allow a broader understanding of where the greatest uncertainties lie and where there is agreement among data and methods.

The EDT is a statistical toolkit that displays information spatially. The advantage of using a statistical package over a GIS is that it allows rapid reanalysis of data such that different combinations of variables can be displayed and compared quickly. This makes it ideal for problems that have a great deal of uncertainty or where a number of "what if" scenarios might be explored. Within the Mercury-EDT:

- The raw data can be viewed and explored

- Choices can be made as to which data or model results are used in determining overall risk when multiple options exist

- Different weights for influential parameters can be set for estimating a methylation potential index

- Comparisons can be made between estimated values and monitored data, and

- Distributions of sensitive populations, estimated indices of methylation potential, and estimated mercury deposition can be integrated into relative rankings of risk from mercury generated from EGUs.

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