Regulatory Impact Analysis of the Final Clean Air Mercury Rule

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U.S. Environmental Protection Agency
Office of Air Quality Planning and Standards
Air Quality Strategies and Standards Division
Innovative Strategies and Economics Group (MD 339-01)
Research Triangle Park, N.C. 27711

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ACRONYMS AND ABBREVIATIONS

ACI	activated carbon injection
ACS	American Cancer Society
ADHD	attention deficit hyperactivity disorder
ADI	acceptable daily intake
ADP	adenosine diphosphate
AERMOD	American Meteorological Society/EPA Regulatory Model
AHRQ	Agency for Healthcare Research and Quality
AMI	acute myocardial infarction
ANL	Argonne National Laboratory
ASA	American Sportfishing Association
atm	atmosphere
ATSDR	Agency for Toxic Substance and Disease Registry
BAF	bioaccumulation factor
BASS	Bioaccumulation and Aquatic System Simulator
BC	boundary conditions
BEA	Bureau of Economic Analysis
BenMAP	Benefits Mapping and Analysis Program
BLS	Bureau of Labor Statistics
BMD	benchmark dose
BMDL	BMD lower statistical confidence limit
BMI	body mass index
BMR	benchmark response
BOC	Bureau of Census
C-R	concentration-response
CAAA	Clean Air Act Amendments
CAIR	Clean Air Interstate Rule
CAMR	Clean Air Mercury Rule
CDF	Cumulative Distribution Function
CEEPR	Center for Energy and Environmental Policy Research
cfs	cubic feet per second
CH3Hg	monomethyl mercury
CHD	coronary heart disease
CI	confidence interval
cm	centimeter
CMAQ	Community Multi-Scale Air Quality
CPS	Current Population Survey
CRDM	Climatological Regional Dispersion Model
CRF	capital recovery factor
CRST	Cheyenne River Sioux Tribal
CSFII	Continuing Survey of Food Intake by Individuals
CVD	cardiovascular disease
D-MCM	Dynamic Mercury Cycling Model
DEP	Department of Environmental Protection
DHHS	Department of Health and Human Services

DDT	dichloro-diphenyl-trichloroethane
DHA	docosahexaenoic acid
DOC	dissolved organic carbon
DOE	Department of Energy
DOI	Department of the Interior
DOM	dissolved organic matter
DPA	docosapentaenoic acid
E-MCM	Everglades Mercury Cycling Model
ECG	electrocardiogram
EFH	Exposure Factors Handbook
EGRID	Emissions & Generated Resource Integrated Database
EGU	electric generating unit
EIA	Economic Impact Analysis
EIA	Energy Information Administration
EKG	electrocardiogram
ELA	Experimental Lakes Area
EMF	emission modification factor
EMMA	Environmental Monitoring and Measurement Advisor
E.O.	Executive Order
EPA	Environmental Protection Agency
EPRI	Electric Power Research Institute
ESP	electrostatic precipitator
EU	European Union
EURAMIC	The European Multicenter Case Control Study on Antioxidants, Myocardial Infarction and Cancer of the Breast
EXAMS2	Exposure Analysis Modeling System, Version 2
F	Fahrenheit
FAO	Food and Agriculture Organization
FDA	Food and Drug Administration
FERC	Federal Energy Regulatory Commission
FF	fabric filters
FGD	flue gas desulfurization
FGETS	Food and Grill Exchange of Toxic Substances
FL	fork length
ft	feet
g	gram
GDP	gross domestic product
GIS	Geographic Information System
GRU	Gainesville Regional Utilities
GW	gigawatt
GWh	gigawatt hours
H-PAC	Hazard Prediction and Assessment Capability
HDL	high-density lipoprotein
Hg	mercury
Hg^0	elemental mercury
Hg(II)	inorganic divalent mercury
HgCl2	mercuric chloride

HgP	particulate mercury
HgT	total mercury
hrs	hours
HUC	hydrologic unit code
HYSPLIT	Hybrid Single Particle Lagrangian Integrated Trajectory
ICR	Information Collection Request
IEM-2M	Indirect Exposure Model, Version 2
IGCC	Integrated Gasification Combined Cycle
IMT	intima-media thickness
in	inch
IOM	Institute of Medicine
IOU	investor-owned utility
IQ	intelligence quotient
IPM	Integrated Planning Model
IRIS	Integrated Risk Information System
ISC3	Industrial Source Complex
K	Kelvin
kg	kilogram
KIHD	Kuopio Ischemic Heart Disease
km	kilometer
kWh	kilowatt hour
lb	pound
lbs	pounds
L	liter
LC50	lethal concentration for 50% percent of the population
LC omega-3 PUFA	long chain omega-3 polyunsaturated fatty acids
LDL	low-density lipoprotein
m	meter
M	molar mass
mm	millimeter
MACT	Maximum Achievable Control Technology
MAS/MILS	Mineral Availability System/Mineral Industry Location System
MCM	Mercury Cycling Model
MDN	Mercury Deposition Network
ME	Midwest/Northeast
MeHg	methylmercury
METAALICUS	Mercury Experiment to Assess Atmospheric Loading in Canada and the United States
mg	milligram
MI	myocardial infarction
MIT	Massachusetts Institute of Technology
MM5	Mesoscale Model
MMAPS	Mercury Maps
mmBtu	million British thermal units
mol	mole
MRFSS	Marine Recreational Fishing Statistical Survey
MRL	minimal risk level

MW	megawatt
MWC	municipal waste combustor
MWh	megawatt hour
MWI	medical waste incinerator
NAAQS	National Ambient Air Quality Standards
NAICS	North American Industry Classification System
NAS	National Academy of Sciences
NASS	National Agriculture Statistics Service
NDMMF	National Descriptive Model of Mercury in Fish
NEI	National Emissions Inventory
NERC	North American Electric Reliability Council
NES	Neurobehavioral Evaluation System
NFTS	National Fish Tissue Survey
ng	nanogram
NHD	National Hydrography Database
NHANES	National Health and Nutrition Examination Survey
NH3	ammonia
NLCD	National Land Cover Data
NLFA	National Listing of Fish and Wildlife Advisories
NLSY	National Longitudinal Study of Youth
NLFTS	National Lake Fish Tissue Survey
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observable Adverse Effects Concentration
NOAEL	No Observed Adverse Effect Level
NODA	Notice of Data Availability
NOx	nitrogen oxides
NPR	Notice of Proposed Rulemaking
NPV	net present value
NRC	National Research Council
NRS	National Recreation Survey
NSFHWR	National Survey of Fishing, Hunting and Wildlife-Associated Recreation
NSPS	New Source Performance Standards
NSR	New Source Review
NSRE	National Survey on Recreation and the Environment
NW	Northwest
O&M	operation and maintenance
OM	organic matter
OMB	Office of Management and Budget
OR	odds ratio
ORD	Office of Research and Development
P-PUFA	plasma polyunsaturated fatty acids
PAC	powder activated carbon
PCB	polychlorinated biphenyl

PCS	Permit Compliance System
pH	potential of hydrogen
PM	particulate matter
POTW	Publicly Owned Treatment Works
ppb	parts per billion
ppm	parts per million
RARE	Regional Applied Research Effort
RELMAP	Regional Lagrangian Model of Air Pollution
REMI	Regional Economic Models, Inc.
REMSAD	Regulatory Modeling System for Aerosols and Deposition
RFA	Regulatory Flexibility Act
RfD	Reference Dose
RFF	Resources for the Future
RGM	Reactive Gaseous Mercury
RIA	Regulatory Impact Analysis
RNA	ribonucleic acid
RQ	risk quotient
RR	relative risk
RtC	Report to Congress
RUSLE	Revised Universal Soil Loss Equation
SAB-HES	Science Advisory Board Health Effects Subgroup
SCR	selective catalytic reduction
SD	standard deviation
SE	Southeast
sec	second
SIC	standard industrial classification
SIP	state implementation plan
SMR	standard mortality rate
SNCR	selective non-catalytic reduction
SO2	sulfur dioxide
sq km	square kilometer
SR-MATRIX	Source Receptor Matrix
SRB	sulfate reducing bacteria
st dev	standard deviation
SW	Southwest
SWAT	Soil and Water Assessment Tool
TDI	tolerable daily intake
TL	total length
TMDL	Total Maximum Daily Load
TOLD-SL	Test of Language Development - Spoken Language
TRUM	Technology, Retrofit and Upgrading Model
TSD	Technical Support Document
UF	uncertainty factor
ug	microgram

uM	micromole
um	micrometer
UMRA	Unfunded Mandates Reform Act
U.S.	United States
USACE	United States Army Corps of Engineers
U.S.C.	United States Code
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UV-B	ultraviolet light, type B
VMI	Visual-Motor Integration
VSL	Value of Statistical Life
WASP	Water Quality Analysis Simulation Program
WCS	Watershed Characterization System
WHO	World Health Organization
WISC	Wechsler Intelligence Scales for Children
WISC-III	Wechsler Intelligence Scales for Children administered in the Seychelles Islands
WISC-R	Wechsler Intelligence Scales for Children administered in New Zealand and the Faroe Islands
WRAML	Wide-Range Assessment of Memory Learning
WTP	willingness to pay
yr	year

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SECTION 1

INTRODUCTION

1.0 Introduction

This report provides an analysis of the benefits and costs of the final Clean Air Mercury Rule (CAMR). In Section 2, we discuss the potential health effects of mercury. Section 3 provides a detailed discussion of mercury in the environment, including how mercury deposited to water bodies transforms into methylmercury in fish tissue. This section also provides an assessment of the response time for systems after a change in mercury deposition. Because fish consumption is the primary pathway for exposure to methylmercury, Section 4 provides a profile of fishing activity in the United States. Section 5 presents information on concentrations of mercury **in** fish. Because this regulation requires control on coal-fired power plants, Section 6 provides a profile of the power sector in the United States, while Section 7 describes the emissions, control requirements, control options considered for CAMR, and the regulatory costs of the final CAMR. In addition, Section 7 also provides an assessment of impacts on small businesses and government entities. Section 8 describes the resulting change in mercury deposition from air quality modeling of the CAMR regulatory options. Section 9 presents a derivation of a dose-response function that relates mercury consumption in women of childbearing with changes in IQ seen in children that were exposed prenatally. IQ is used as a surrogate for the neurobehavioral endpoints that EPA relied upon for setting the methylmercury reference dose (RfD). Chapter 10 presents exposure modeling and benefit methodologies applied to a no-threshold model (i.e., a model that assumes no threshold in effects at low doses of mercury exposure). Chapter 11 presents the final benefit analysis numbers of CAMR giving consideration to established health benchmarks (i.e., consideration of potential thresholds on effects at low doses of mercury exposure). Finally, Chapter 12 presents a benefit analysis of reductions in PM as a result of controls applied for mercury. Table 1-1 below summarizes the benefits, costs, and net benefits of the CAMR.

Figures

SECTION 2

IMPACT OF MERCURY ON HUMAN HEALTH, ECOSYSTEMS, AND WILDLIFE

2.1 Introduction

This section discusses the potential human health and ecological effects due to exposure to methylmercury. The material in this section is based upon the National Research Council (NRC) of the National Academies of Science report titled "Toxicological Effects of Methylmercury," which provides a thorough review of the effects on mercury on human health (NRC 2000), augmented by other related and more recent publications regarding the effects of methylmercury exposure. Many of the peer-reviewed articles cited in this section are publications originally cited in the NRC report (all secondary citations are clearly noted in the reference section).

The section starts with a short account of mercury poisoning episodes in Japan and Iraq (Section 2.2), which provide much of the basis for research on human health at very high exposure levels. Next, the reference dose and benchmark dose for methylmercury exposure are discussed in Section 2.3 to provide context for the ensuing descriptions of specific health effects (Sections 2.4 – 2.8). The section concludes with a discussion of ecological effects (Section 2.9) and conclusions (Section 2.10).

2.2 Mercury Poisoning Episodes

Instances of methylmercury poisoning have made it clear that adults, children, and developing fetuses are at risk from ingestion exposure to methylmercury. These episodes resulted in exposures well above those observed in any US subpopulations, however, they provided early motivation for risk management of mercury. Two of these high-dose mercury poisoning occurred in Japan and Iraq. In Japan, industrial by-products containing organic mercury were discharged into Minamata Bay between 1953 and 1960, contaminating fish and resulting in methylmercury poisoning of the local population via consumption of fish. The central nervous system was the primary target; symptoms of exposure included paresthesia (a burning or prickling sensation in the skin), ataxia (failure of muscle control), sensory disturbances (e.g., impaired vision, hearing, and smell), tremors, difficulty in walking, irritability, and others, including death (NRC 2000). Children of exposed women displayed a higher incidence of symptoms than did exposed adults. Some victims were born with a condition resembling cerebral palsy, with severe disturbances of nervous function, and affected offspring were very late in reaching developmental milestones (EPA 1997, UNEP 2002).

Maternal hair mercury concentrations in this population ranged from 3.8 to 133 ppm (mean of 41 ppm). There is significant uncertainty associated with these exposure estimates, primarily because measurements of methylmercury exposure were not taken until several years after the poisoning episode had begun and identification of cases was incomplete; however, it is clear that the exposures were quite high.

Another series of acute mercury poisonings occurred in Iraq in the late 1950s and 1972 following consumption of bread made with seed grain treated with fungicides containing alkylmercury compounds, affecting thousands of people in total. Symptoms in the exposed population were similar to those observed in Minamata, with severely affected individuals exhibiting paresthesia, ataxia, blurred vision, slurred speech, hearing difficulties, blindness, deafness, and death. Toxicity was observed in many adults and children who had consumed this bread over a three-month period, but the population that showed greatest sensitivity were offspring of women who had eaten contaminated bread during pregnancy. Maximum maternal hair mercury levels during pregnancy for mothers of affected children ranged to over 600 ppm, with some effects in children (e.g., delayed ability to walk) possibly associated with maternal hair mercury levels less than 100 ppm.

In both Iraq and Japan, the effects in offspring prenatally exposed to methylmercury were more serious, and in some cases seen at lower doses, than in adults (EPA 1997, NRC 2000).

These instances of methylmercury poisoning have made it clear that adults, children, and developing fetuses are at risk from ingestion exposure to methylmercury. In both episodes, mothers with few or no symptoms of nervous system damage gave birth to infants with severe disabilities, and it became clear that the developing nervous system of the fetus is more vulnerable to methylmercury than the adult nervous system. Even though these episodes resulted in exposures well above those observed in any US subpopulations, they provided early motivation for risk management of mercury, and the U.S. FDA first proposed an administrative guideline for mercury levels in fish and shellfish in 1969 in response to the poisonings in Japan (EPA 1997). In the years since these episodes, much research has been undertaken to more fully understand the effects associated with high-dose methylmercury poisoning as well as more common lower-dose exposures, and these data have been used by EPA and others in mitigating potential human health effects.

2.3 Reference and Benchmark Doses

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 2002). EPA believes that exposures at or below the RfD are 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 near 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 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 NRC 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 (EPA 2002).

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. 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 of children with a response at the 5th percentile of the population).[1] 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.[2] The BMDLs for these analyses (measured in terms of mercury in cord blood) were all observed to be within a relatively close range, and 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-

[1] 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.

[2] 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).

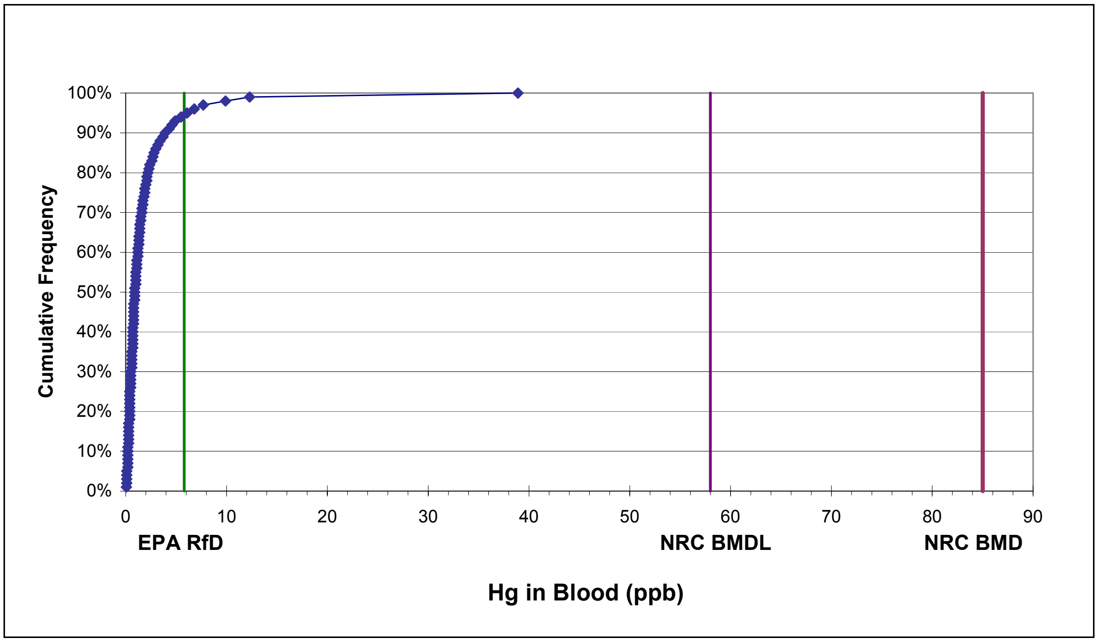
bw/day to 0.0019 mg/kg-day, with most dietary exposures estimated to be about 0.001 mg/kg-bw/day. The integrative BMDL (taking into account data from all three studies) was calculated by NRC to be 32 ppb mercury in cord blood, or an exposure of about 0.6 ug/kg-day. All of these results were considered in defining the RfD; as stated in the IRIS summary for methylmercury:

"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." (EPA 2002)

EPA used the various BMDLs and then applied an uncertainty factor of 10 to account for interindividual toxicokinetic variability and pharmacodynamic variability and uncertainty. On this basis, EPA defined the updated RfD of 0.0001 mg/kg-day 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.

The levels at which these key neurological effects were observed – in the study populations on which the updated RfD is based – provide a useful frame of reference for considering other, non-neurological effects described below (see above for observed exposure levels). It is important to note that although these populations were exposed to elevated levels of methylmercury via fish or marine mammal consumption, the exposure levels of interest in these studies are far below those associated with the Minamata and Iraqi poisoning episodes mentioned previously. 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). 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 2-1). 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 at or above the RfD (MMWR Vol. 53 / No. 43). 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.[3]

[3] The NHANES data summarized above suggests that exposures of women of child-bearing age in the U.S. exceed the RfD.



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.

Figure 2-1. Probability Distribution Function of Blood Mercury Levels in US Women of Childbearing Age (NHANES Data 1999-2002)

NRC notes in their analysis that a biomarker conversion factor of about 5 ppb in blood per 1 ppm in hair can be used to estimate the corresponding hair mercury values. Using this approach, the RfD of 5.8 ppb in blood corresponds to a hair mercury concentration of about 1 ppm, and the BMDL and BMD are equivalent to about 12 ppm and 17 ppm, respectively. Analyses of hair mercury for U.S. women of child-bearing age have been conducted using NHANES data from 1999-2000. A geometric mean hair mercury concentration of 0.20 ppm was reported for this population, and the geometric mean of the concentration of organic mercury was 0.80 ppb in blood (Mahaffey et al. 2004). Among frequent fish consumers (i.e., study participants who reported consuming fish three or more times in the previous 30 days),[4] geometric mean hair mercury levels were three-fold higher compared with nonconsumers (viz., 0.38 ppm vs. 0.11 ppm). Higher percentiles of exposure were also reported, with the 95th percentile hair mercury levels corresponding to 1.73 ppm and 2.75 ppm for all women and women frequently consuming fish, respectively (McDowell et al. 2004).

In general, the primary route by which the U.S. population is exposed to mercury is through the consumption of fish containing methylmercury. Exposure to methylmercury may result in a variety of health effects. The various categories of health effects, and the evidence on their significance, are described in the following pages.

2.4 Neurologic Effects

[4] Fish consumption rates were collected by questionnaire at the time of the survey; no information was collected regarding portion size or preparation methods.

In their review of the literature, NRC found neurodevelopmental effects to be the most sensitive endpoints and appropriate for establishing an RfD (NRC, 2000). Three large-scale epidemiological studies have examined the effects of low dose prenatal mercury exposure and neurodevelopmental outcomes through the administration of numerous tests of cognitive functioning. These studies were conducted in the Faroe Islands (Grandjean et al. 1997), New Zealand (Kjellstrom et al. 1989, Crump et al. 1998), and the Seychelles Islands (Davidson et al. 1998, Myers et al. 2003). The NRC noted that deficiencies of the magnitude observed in those studies were likely to be associated with difficulty with vocabulary, verbal learning, attention, and motor functions (NRC 2000). The NRC also concluded that children exposed at the levels reported in those studies are likely to struggle to keep up in class and may need special education, or other remedial help with school. Studies involving animals found sensory effects and support the conclusions reached by studies involving human subjects, with a similar range of neurodevelopmental effects reported (NRC 2000). As noted by the NRC, the clinical significance of some of the more subtle endpoints included in the human low-dose studies is difficult to gauge due to the quantal nature of the effects observed (i.e., subjects either display the abnormality or do not) and the rather low occurrence rate of these effects.

Little is known about the effects of low level chronic methylmercury exposure in children that can be linked to exposures *after* birth. The difficulty in identifying a cohort exposed after birth but not prenatally, or separating prenatal from postnatal effects, makes research on the topic complicated. These challenges were present in the three large epidemiologic studies used to derive the RfD, as in all three studies there was postnatal exposure as well.

Several studies have also examined the effects of chronic low-dose methylmercury exposures on adult neurological and sensory functions (e.g., Lebel et al. 1996, Lebel et al. 1998, Beuter and Edwards 1998). Research results suggest that elevated hair methylmercury concentrations (i.e., up to 50 ppm, and possibly as low as 20 ppm, though the NOAEL was not always be clearly estimated) in individuals are associated with visual deficits, including loss of peripheral vision and chromatic and contrast sensitivity. These individuals also exhibited a loss of manual dexterity, hand-eye coordination, and grip strength; difficulty performing complex sequences of movement; and (at the higher doses) tremors, although expression of some effects was sex-specific. Although additional data would be needed to quantify a dose-response relationship for these effects, it is noteworthy that the effects occurred at doses lower than the Japanese and Iranian poisoning episodes, via consumption of mercury-laden fish in riverine Brazilian communities (where extensive mercury contamination has resulted from small-scale gold mining activities begun in the 1980s); however these doses are above the EPA's RfD equivalent level for hair mercury. In regard to the Lebel et al. (1998) study, NRC states that "the mercury exposure of the cohort is presumed to have resulted from fish-consumption patterns that are stable and thus relevant to estimating the risk associated with chronic, low-dose methylmercury exposure" (NRC 2000). NRC noted, however, "that the possibility cannot be excluded that the neurobehavioral deficits of the adult subjects were due to increased prenatal, rather than ongoing, MeHg exposure." More recent studies in the Brazilian communities provide some evidence that the adverse neurobehavioral effects may in fact result from postnatal exposures (e.g., Yokoo et al. 2003); however, additional longitudinal study of these and other populations is required to resolve questions regarding exposure timing and fully characterize the potential neurological impacts of methylmercury exposure in adults.

2.5 Cardiovascular Impacts

While important, the weight of evidence for cardiovascular effects is not as strong as it is for childhood neurological effects and the state of the science is still being evaluated. However, in some recent epidemiological studies in men, methylmercury exposure is associated with a higher risk of acute myocardial infarction, coronary heart disease and cardiovascular disease in some populations (e.g. Salonen et al. 1995; Guallar et al.2002). Other recent studies have not observed this association (e.g. Yoshizawa et al. 2002; Hallgren et al. 2001). The studies that have observed an association suggest that the exposure to methylmercury may attenuate the beneficial effects of fish consumption. Studies investigating the relationship between methylmercury exposure and cardiovascular impacts have reached different conclusions. The findings to date and the plausible biologic mechanisms warrant additional research in this arena (Stern 2005; Chan and Egeland 2004).

The potential for adverse cardiovascular effects due to consumption of fish containing methylmercury is of particular interest given the evidence for the *protective* cardiovascular effect believed to occur from an increased dietary fish intake. Strong evidence indicates that consumption of fish, particularly fatty fish, has a cardio-protective effect (Wang et al. 2004; 2005 Dietary Guidelines Advisory Committee 2004; NRC 2000, Kris-Etherton et al. 2002). Thus, consumption of fish containing methylmercury is not necessarily detrimental even though some evidence suggests that the cardiovascular system may be a target system for methylmercury exposure .

A more robust discussion of multiple studies evaluating the association between methylmercury exposure via fish consumption and acute myocardial infarction and other cardiovascular effects as well as a description of the association between fish consumption and cardioprotective effects is presented in Appendix B.

2.6 Genotoxic Effects

The NRC concluded that evidence that human exposure caused genetic damage is inconclusive. However, in one recent study of adults living in the Tapajós River region in Brazil Amorim et al. (2000) reported a direct relationship between methylmercury concentration in hair and cytogenetic damage in lymphocytes, with polyploidal aberrations and chromatid breaks observed at mercury hair levels around 7.25 ppm and 10 ppm, respectively. Long-term methylmercury exposures in this population were believed to occur through consumption of fish, suggesting that cytotoxic effects may result from dietary, chronic methylmercury exposures similar to and above those seen in the Faroes and Seychelles populations.

2.7 Immunotoxic Effects

Although exposure to some forms of mercury can result in a decrease in immune activity or an autoimmune response (ATSDR 1999), evidence for immunotoxic effects of methylmercury is scarce (NRC 2000). However, a recent study of fish-consuming communities in Amazonian Brazil has identified a possible association between methylmercury exposure and immunotoxic effects, although the authors noted that this may reflect interactions with infectious disease and other factors (Silva et al. 2004). Exposures to these communities occurred via fish consumption (some community members were also exposed to inorganic mercury through gold mining activities). The researchers assessed levels of specific antibodies that are markers of mercury-induced autoimmunity. They found that both prevalence and levels of these antibodies were higher in a population exposed to methylmercury via fish consumption compared to a reference (unexposed) population. Median hair mercury concentration was 8 ppm in the more exposed population (range 0.29-58.47 ppm) and 5.57 ppm in the less exposed reference population (range 1.19-16.96 ppm). The ranges of mercury hair concentrations reported in this study are within an order of magnitude of the concentration corresponding to the methylmercury RfD. Overall, there is a relatively small body of evidence from human studies that suggests exposure to methylmercury can result in immunotoxic effects.

2.8 Other Human Toxicity Data

Based on limited human and animal data, methylmercury is classified as a "possible" human carcinogen by the International Agency for Research on Cancer (IARC 1994) and in the Integrated Risk Information System (IRIS) (EPA 2002). The existing evidence supporting the possibility of carcinogenic effects in humans from low-dose chronic exposures is tenuous. Multiple human epidemiological studies have found no significant association between mercury exposure and overall cancer incidence, although a few studies have shown an association between mercury exposure and specific types of cancer incidence (e.g., acute leukemia and liver cancer; NRC 2000). The MSRC observed that "Methylmercury is not likely to be a human carcinogen under conditions of exposure generally encountered in the environment" (p 6-16, Vol V). This was based on observation that tumors were noted in one species only at doses causing sever toxicity to the target organ. While some of the human and animal research suggests that a link between methylmercury and cancer may plausibly exist, more research is needed.

There is also some evidence of reproductive and renal toxicity in humans from methylmercury exposure. For example, a smaller than expected number of pregnancies were observed among women exposed via contaminated wheat in the Iraqi poisoning episode of 1956 (Bakir et al. 1973); other victims of that same poisoning event exhibited signs of renal damage (Jalili and Abbasi 1961); and an increased incidence of deaths due to kidney disease was observed in women exposed in Minamata Bay via contaminated fish (Tamashiro et al. 1986). Other data from animal studies suggest a link between methylmercury exposure and similar reproductive and renal effects, as well as hematological toxicity (NRC 2000). Overall, human data regarding reproductive, renal, and hematological toxicity from methylmercury are very limited and are based on either studies of the two high-dose poisoning episodes in Iraq and Japan or animal data, rather than epidemiological studies of chronic exposures at the levels of interest in this analysis. Note that the U.S. EPA Mercury Study Report to Congress provides an assessment of methylmercury cancer risk using the 1993 version of the Revised Cancer Guidelines. For hazard identification, these are similar to the current EPA revisions.

2.9 Ecological Effects

Deposition of mercury to water bodies can also have an impact on ecosystems and wildlife. While the benefit of further reducing mercury emissions cannot be quantified for ecosystems at this time, we find it useful to qualitatively describe this benefit for context.

Mercury contamination is present in all environmental media with aquatic systems experiencing the greatest exposures due to bioaccumulation. Bioaccumulation refers to the net uptake of a contaminant from all possible pathways and includes the accumulation that may occur by direct exposure to contaminated media as well as uptake from food. Elimination of methylmercury from fish is so slow that long-term reductions of mercury concentrations in fish are often due to growth of the fish ("growth dilution"), whereas other mercury compounds are eliminated relatively quickly. Piscivorous avian and mammalian wildlife are exposed to mercury mainly through the consumption of contaminated fish and, as a result, accumulate mercury to levels greater than those in their prey items (EPA 1997).

Numerous studies have generated field data on the levels of mercury in a variety of wild species. Many of the data from these environmental studies are anecdotal in nature rather than representative or statistically designed studies. The body of work examining the effects of these exposures is growing but still incomplete given the complexities of the natural world. A large portion of the adverse effect research conducted to date has been carried out in the laboratory setting rather than in the wild; thus, conclusions about overarching ecosystem health and population effects are difficult to make at this time. Nevertheless, numerous adverse effects have been identified. Further reducing the presence of mercury in the environment may help to alleviate the potential for adverse ecological health outcomes.

A full discussion of potential ecosystem effects updated since the 1997 Mercury Report to Congress is provided in Appendix C.

2.10 Conclusions

In summary:

- Children who are exposed to low concentrations of methylmercury prenatally may be at risk of poor performance on neurobehavioral tests, such as those measuring attention, fine motor function, language skills, visual-spatial abilities and verbal memory.

- Some recent epidemiological studies in men suggest that methylmercury is associated with a higher risk of acute myocardial infarction, coronary heart disease and cardiovascular disease in some populations. Other recent studies have not observed this association. The studies that have observed an association suggest that the exposure to methylmercury may attenuate the beneficial effects of fish consumption. The findings to date and the plausible biologic mechanisms warrant additional research in this arena (Stern 2005; Chan and Egeland 2004).

- The exposure levels at which neurological effects have been observed may occur via consumption of fish (rather than high-dose poisoning episodes). Exposure levels of concern for these effects are generally within two orders of magnitude of typical exposures for women of child-bearing age based on NHANES data, and within approximately an order of magnitude of the high end of the US exposure distribution.

- There is some recent evidence that exposures of methylmercury may result in genotoxic or immunotoxic effects. Other research with less corroboration suggest that reproductive, renal, and hematological impacts may be of concern. There are insufficient human data to evaluate whether these effects are consistent with levels in the U.S. population.

- Plant and aquatic life, as well as fish, birds, and mammalian wildlife can be affected by mercury exposure, however overarching conclusions about ecosystem health and population effects are difficult to make at this time.. Ecological effects are discussed in greater detail in Appendix C.

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SECTION 3

ECOSYSTEM SCALE MODELING FOR MERCURY BENEFITS ANALYSIS

Executive Summary

In the United States, humans are exposed to methylmercury (MeHg) mainly by consuming fish that contain MeHg. Aquatic ecosystems respond to changes in mercury deposition in a highly variable manner as a function of differences in their chemical, biological and physical properties. Depending on the characteristics of a given ecosystem, methylating microbes convert a small but variable fraction of the inorganic mercury in the sediments and water derived from human activities and natural sources into MeHg. Methylmercury is the only form of mercury that biomagnifies in the food web. Concentrations of MeHg in fish are generally on the order of a million times the MeHg concentration in water. In addition to mercury deposition, key factors affecting MeHg production and accumulation in fish include the amount and forms of sulfur and carbon species present in a given waterbody. Thus, two adjoining water bodies receiving the same deposition can have significantly different fish mercury concentrations.

For the Utility Mercury Reduction Benefits Assessment, EPA applied the Mercury Maps (MMaps) model to estimate changes in freshwater fish mercury concentrations resulting from changes in mercury deposition after regulation of mercury emissions from U.S. coal-fired power plants. MMaps, a simplified form of the IEM-2M model applied in EPA's 1997 *Mercury Study Report to Congress*, is a static model that assumes a proportional relationship between declines in atmospheric mercury deposition and concentrations in fish at steady state. This means, for example, that a 50% decrease in mercury deposition rates is projected to lead to a 50% decrease in mercury concentrations in fish. MMaps does not consider the dynamics of relevant ecosystem specific factors that can affect the methylation and bioaccumulation in fish in different water bodies over time, nor does it consider the inputs of non-air sources to the watershed. In all cases, the MMaps model does not address the lag time of different ecosystems to reach steady state (i.e., when fish mercury concentrations reflect changes in atmospheric deposition). In addition, applying the MMaps model assumes that atmospheric deposition is the principle source of mercury to the waterbodies being investigated and environmental factors that affect MeHg production and accumulation in organisms will remain constant, allowing each ecosystem to reach steady state. While MMaps has several limitations, EPA knows of no alternative tool for performing a national-scale assessment of such changes.

The objectives of this chapter are to: (a) provide information on the response times of different ecosystems to declines in mercury deposition, and (b) characterize some of the key sources of uncertainty around the proportional relationship used by the MMaps model. To do this, EPA applied dynamic, ecosystem scale waterbody, watershed, and bioaccumulation models to five freshwater systems spanning a range of types across the United States. We used model results to investigate the magnitude and timing of changes in fish mercury concentrations associated with changes in atmospheric mercury deposition after regulation of coal-fired power plants. While important advances have been made in recent years to enhance scientific understanding of the behavior of mercury in the environment, our ability to effectively model the range in response times for different systems is constrained by our limited knowledge of how methylation and bioaccumulation occur in various ecosystems. Because of these uncertainties, no modeling framework can be considered *a priori* predictive of ecosystem responses at this time. In recognition of the above, we calibrated models applied in the ecosystem case studies to monitoring data from each of the specific ecosystems studied. In addition, when choosing the locations of these case studies it was essential to rely on well-studied ecosystems where there were sufficient empirical data available to parameterize ecosystem scale models.

Sites investigated can be characterized as follows:

(1) Small, southern seepage lake with negligible watershed (Lake Barco, FL);
(2) Watershed dominated, coastal plain river (Brier Creek, GA);
(3) Large, shallow, well-mixed southern lake (Lake Waccamaw, NC);
(4) Medium sized, stratified seepage lake with a moderate sized drainage basin in the Northeast (Pawtuckaway Lake, NH); and
(5) Shallow, well-mixed farm pond in the Midwest (Eagle Butte, SD).

When considering other ecosystem variables that may affect MeHg production (e.g., sulfate deposition, percent wetland coverage, and organic carbon), these case studies represent ecosystem types of moderate methylation potential across the United States. Fish tissue concentrations and atmospheric deposition rates measured in these regions also do not represent the extremes observed on a national scale. Therefore, while these ecosystem case studies cover the bulk of the distributions of the key environmental characteristics that will affect MeHg production, they may miss the tails of the distributions for some characteristics.

For each of the above system types, we characterized a range of response times by varying key parameters in the modeling scenarios known to drive the temporal response. 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. Forecasted response times to changes in mercury inputs were longest for Brier Creek, a system strongly influenced by the watershed mercury loading, and Pawtuckaway Lake, a stratified cold water lake that had significant watershed mercury inputs. Shallow, well-mixed systems like Lake Waccamaw and Lee Dam, which receive most of their mercury inputs from the atmosphere are projected to respond to changes in atmospheric deposition in less than a decade, although watershed loading dynamics could introduce significant lag times in reaching the full response. These findings are consistent with those observed in the Florida Everglades, which can be characterized as a shallow, well-mixed, dynamic system and accordingly shows a measurable response within a decade. Results from Brier Creek, the watershed dominated system, qualitatively concur with findings from the METALLICUS study showing a much longer response time for mercury deposited in the watershed than direct atmospheric deposition to the surface of a waterbody.

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. This assessment is based on the "medium" or moderate estimates across the several system types considered in this study. Because our modeling scenarios include two extremes of rapidly and slowly responding system types (e.g., a watershed dominated system and warm, shallow, well-mixed systems), we expect that the range in responsiveness of freshwater bodies in the United States will be captured within the "fast, medium and slow" scenarios presented in the report. One additional uncertainty in our calculations related to preliminary results from the METAALICUS study showing newly deposited mercury is converted to methylmercury more rapidly than legacy mercury. These results, if extrapolated to other freshwater sites, imply that the response time of some freshwater ecosystems may be more rapid than predicted by our best available models at this time.

To appreciate the importance of the adjustment time for the benefits analysis, consider the following scenario: Suppose that (a) current benefits are proportional to the reduction in fish tissue concentrations from baseline levels, (b) fish tissue concentrations decrease exponentially to eventually reach 90% of the total reduction, and (c) the "adjustment time" is measured as the time required to reach 90% of the eventual total reduction. Under these conditions, using discount rates between 3% and 7% the net present values (NPV) of benefits if the adjustment times were 5 years would be between 1.6 and 2.1 times the NPV if the adjustment time were 50 years, even if the same eventual reduction were reached.

To investigate some of the key sources of uncertainty around the proportional relationship used by the MMaps model, we modeled the magnitudes of expected changes in fish mercury concentrations locations after removal of utilities as an emissions source. At each of the case study locations considered, removal of coal fired utilities as a source of mercury reduced atmospheric deposition rates between four and fifteen percent. These values are relatively small in magnitude compared to maximum values observed across the country where the maximum model forecasted difference in deposition in this "zero-out" scenario exceeds 70%. To place this in context, this rule is expected to reduced total mercury emissions in the U.S. by up to 70% for the "cap and trade" alternative. At the locations considered in this study, EPA estimates that the reductions in emissions associated with the cap and trade alternative will eventually reduce loading rates by approximately 10% from current conditions (see Table 3-8 below). This decline in deposition can be contrasted with areas most highly affected by the proposed regulation, where declines in deposition are more likely to be on the order of 50% according to modeling projections. Accordingly, we modeled a 50% decline in atmospheric loading in all of the case studies to investigate responses of similar ecosystem types to large declines in mercury deposition.

Our analysis suggests that differences between results from the ecosystem scale waterbody models and MMaps model across all sites are mainly a function of the initial fish mercury concentrations at steady state. Because error bars around each species of piscivorous fish investigated in the case studies from a single water body are large, we expect that on a national scale, techniques used to normalize fish mercury data will be a major source of uncertainty in the MMaps model. Accordingly, both the completeness of national fish tissue monitoring data coverage and the statistical techniques used to normalize these data among different trophic levels and ages will have a major impact on how well the MMaps approach captures the true variability in fish mercury concentrations across the country. Ecosystem models may be used to supplement limited fish tissue data with forecasted fish mercury concentrations given other information on mercury concentrations and dynamics in a given waterbody. Overall, it is clear that the magnitude of the uncertainty in the atmospheric and ecosystem models at this time is much greater than the signal derived from a change in loading following removal of the coal fired utilities at the sites investigated in this report.

Another source of uncertainty in the MMaps forecasts are the atmospheric deposition rates used to forecast changes in fish mercury concentrations. For each case study site, deposition rates in the corresponding CMAQ and REMSAD grid cells were compared to empirically derived loading rates. At the locations chosen for these case studies, site specific data suggest somewhat higher deposition rates than the CMAQ and REMSAD models. This would result in an overestimate of the relative change in atmospheric deposition and changes in fish mercury concentration by the MMaps model. These findings reinforce the need for additional data sets that can be used to test model-forecasted atmospheric mercury deposition rates.

The effect of epistemic uncertainty (i.e., lack of knowledge) about key mercury process variables, such as the functional form of equations used to quantify methylation rate constants, is a major contributor to overall uncertainty in the MMaps and ecosystem models that cannot be quantified at this time. In addition, a preliminary assessment of the expected effect of land-use changes on fish mercury concentrations for a watershed dominated system illustrates changes like urbanization within a watershed can alter the magnitude and timing of fish mercury concentrations. Accordingly, EPA's Office of Research and Development views this study as part of an iterative modeling exercise.

3.1 Introduction -- Rule Background

As described in the NODA (FR 69864-69877, December 1, 2004), EPA's revised benefits analysis estimates the extent to which adverse human health effects will be reduced as a result of reducing mercury (Hg) emissions from coal-fired power plants. In the United States, humans are exposed to methylmercury (MeHg) mainly by consuming fish that contain MeHg. Accordingly, to estimate changes in human exposure EPA must analyze how changes in Hg deposition from U.S. coal-fired power plants translate into changes in MeHg concentrations in fish. This proposed rule is expected to reduce total mercury emissions in the U.S. by up to 70% using the "cap and trade" program. In the case studies described later in this section, EPA estimates that the reductions in emissions associated with the cap and trade program will eventually reduce loading rates to the locations considered in this study by approximately 10% from current conditions (see Table 3-4 below). This decline in deposition can be contrasted with areas most highly affected by the proposed regulation, where declines in deposition are more likely to be on the order of 50% according to modeling projections. Quantifying the linkage between different levels of Hg deposition and fish tissue MeHg concentration is an important step in the benefits methodology and the focus of the material described in this chapter.

To effectively estimate fish MeHg concentrations in a given ecosystem, it is important to understand that the behavior of Hg in aquatic ecosystems is a complex function of the chemistry, biology, and physical dynamics of different ecosystems. The majority (95 to 97 percent) of the Hg that enters lakes, rivers, and estuaries from direct atmospheric deposition is in the inorganic form (Lin and Pehkonen, 1999). Microbes convert a small fraction of the pool of inorganic Hg in the water and sediments of these ecosystems into the organic form of Hg (MeHg). MeHg is the only form of Hg that biomagnifies in organisms (Bloom, 1992). Ecosystem-specific factors that affect both the bioavailability of inorganic Hg to methylating microbes (*e.g.*, sulfide, dissolved organic carbon) and the activity of the microbes themselves (*e.g.*, temperature, organic carbon, redox status) determine the rate of MeHg production and subsequent accumulation in fish (Benoit et al., 2003). The extent of MeHg bioaccumulation is also affected by the number of trophic levels in the food web (*e.g.*, piscivorous fish populations) because MeHg biomagnifies as large piscivorous fish eat smaller organisms (Watras and Bloom, 1992; Wren and MacCrimmon, 1986). These and other factors can result in considerable variability in fish MeHg levels among ecosystems at the regional and local scale.

3.1.1 Use of Mercury Maps (MMaps) to Project Changes in Fish Tissue Concentrations

To analyze the relationship between Hg deposition and MeHg concentrations in fish in freshwater aquatic ecosystems across the U.S. for the national scale benefits assessment, EPA applied EPA's Office of Water's Mercury Maps (MMaps) approach. MMaps implements a simplified form of the IEM-2M model applied in EPA's Mercury Study Report to Congress (USEPA, 1997). By simplifying the assumptions inherent in the freshwater ecosystem models that were described in the Report to Congress, the MMaps model showed that these models converge at a steady-state solution for MeHg concentrations in fish that are proportional to changes in Hg 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 Hg to a water body, and the physical, chemical, and biological characteristics of the ecosystem remain constant over time. EPA recognizes that concentrations of MeHg 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 Hg mining in western states, contain significant non-air sources of Hg. Finally, EPA recognizes that MMaps does not provide for a calculation of the time lag between a reduction in Hg deposition and a reduction in the MeHg concentrations in fish. Despite these limitations, EPA is unaware of any other tool for performing a national-scale assessment of the change in fish MeHg concentrations resulting from reductions in atmospheric deposition of Hg.

MMaps has several limitations:

1. The MMaps approach is based on the assumption of a linear, steady-state relationship between concentrations of methylmercury in fish and present day air deposition mercury inputs. We expect that this condition will likely not be met in many waterbodies because of recent changes in mercury inputs and other environmental variables that affect mercury bioaccumulation. For example, the US has recently reduced human-caused emissions while international emissions have increased.

2. The requirement that environmental conditions remain constant over the time required to reach steady state inherent in the MMaps methodology may not be met, particularly in systems that respond slowly to changes in mercury inputs.

3. Many water bodies, particularly in areas of historic gold and mercury mining in western States, contain significant nonair sources of mercury. MMaps methodology cannot be applied to these waterbodies.

4. Finally, MMaps does not provide for a calculation of the time lag between a reduction in mercury deposition and a reduction in the methylmercury concentrations in fish.

The following paragraphs provide additional details on the above limitations, as well as a brief assessment of the degree to which conditions match those assumptions. The MMaps model (US EPA, 2001) assumes that for long-term steady state conditions, reductions in fish tissue concentrations are expected to track linearly with reductions in air deposition watershed loads. The MMaps model represents a *reduced form* of the IEM-2M and MCM models used in the Mercury Study Report to Congress (USEPA, 1997), as well as the subsequent Dynamic MCM (D-MCM) model (Harris et al., 1996). That is, the equations of these mercury fate and transport models are reduced to steady state and consolidated into a single equilibrium equation equating the ratio of future/current air deposition rates to future/current fish tissue concentrations. At certain sites, the MMaps model has been shown to produce results equivalent to those of these complex models over the long term, under a specific set of conditions.

Though plainly stated, the steady state assumption is a compilation of a number of individual conditions. For example, fish tissue data may not represent average, steady state concentrations for two major reasons:

- Fish tissue and deposition rate data for the base period are not at steady state. Where deposition rates have recently changed, the watershed or waterbody may not have had sufficient time to fully respond. The pool of mercury in different media could be sufficiently large relative to release rates, and thus needs more time to achieve a new equilibrium. This is more likely to occur in deeper lakes and lakes with large catchments where turnover rates are longer and where the watershed provides significant inputs of mercury.

- Fish tissue data do not represent average conditions (or conditions of interest for forecast fish levels). Methylation and bioaccumulation are variable and dynamic processes. If fish are sampled during a period of high or low methylation or bioaccumulation, they would not be representative of the average, steady state or dynamic equilibrium conditions of the waterbody. This effect is significantly more pronounced in small and juvenile fish. Examples include tissue data collected during a drought or during conditions of fish starvation. Other examples include areas in which seasonal fluctuations in fish mercury levels are significant, due for example from seasonal runoff of contaminated soils from abandoned gold and mercury mines or areas geologically rich in mercury. In such a case, MMaps predictions would be valid for similar, conditions (e.g. wet year/dry year, or season) in the future, rather than typical or average conditions. Alternatively, sufficient fish tissue would need to be collected to get an average concentration that represents a baseline dynamic equilibrium.

Other ecosystem conditions might cause projections from the MMaps approach to be inaccurate for a particular ecosystem. Watershed and waterbody conditions can undergo significant changes in capacity to transport, methylate, and bioaccumulate mercury. Examples of this include regions where sulfate and/or acid deposition rates are changing (in turn affecting methymercury production independently of total mercury loading), and where the trophic status of a waterbody is changing. A number of other water quality parameters have been correlated with increased fish tissue concentrations (e.g. low pH, high DOC, lower algal concentrations), but these relationships are highly variable among different waterbodies. MMaps will be biased when waterbody characteristics change between when fish were initially sampled, and the new conditions of the waterbody.

As is stated above, the relationship between the change 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 MMaps approach screened out watersheds in which sources of mercury other than air deposition were significant.[1] Therefore, fish tissue concentrations are assumed to remain unchanged if located in watersheds that contain potentially significant nonpoint sources such as: historic mercury mining locations (as a surrogate for mercury bedrock deposits) (MAS/MILS database for mercury mines (US EPA, 2001b); significant producer gold mines (USGS Database for Significant Deposits; Long, et al, 1998); or mercury cell chlor-alkali facilities (USEPA – PCS database). Watersheds are also screened from the analysis where the sum estimated mercury loads from other sources, e.g. Publicly Owned Treatment Works (POTW) exceeds an arbitrary level of significance (e.g. 5% of total load).

To the degree to which the applicability of the above conditions is unknown, MMaps is a screening level estimate of the changes in fish tissue as a result of changes in air deposition rates. Where these specific conditions do apply, the results from MMaps will be equivalent. The above criteria, for assessing the validity of the steady state assumption, were used to evaluate this benefits analysis application (in the same order as above):

- Changes in sulfate deposition and waterbody sulfate concentration and pH. This has been shown to be the cause for 50% of the reduction in fish tissue mercury levels in a lake in Minnesota. Significant changes in the presence of riparian wetlands in the Adirondack park in New York, thought due to the resurgence in beaver population since the turn of the century, has led to a dramatic increase of mercury export from watersheds to waterbodies. Increased development and urbanization is associated with depressed bioaccumulation rates. Thus in some areas, the predictions using this approach will be somewhat inaccurate due to other confounding factors.

- While the base year for deposition modeling was 2001, the bulk of the fish tissue data was collected in the early 1990's. We know that emissions in 2001 were 50% of that in 1990. Given a short time lag (5 years) the available fish tissue would reflect these higher emissions from 1990. Thus, projected changes in fish tissue concentrations will be applied to higher fish tissue concentrations, and thus be somewhat higher than is actually

[1] An alternative approach, presented in US EPA, 2001, allows for taking into account other significant sources where loads from these sources can be quantified and are not expected to change with time.

expected. In this benefits context, this mismatch between air deposition and fish tissue data results in an underestimate of benefits.

- It is unknown to which degree the NLFA and NFTS data reflect those that are commonly fished. While state monitoring programs now generally focus on areas of high fishing pressure, early monitoring programs (prior to 1995) used to develop fish advisories focused their sampling efforts on areas near industrial outfalls and agricultural runoff, looking primarily for organochlorine (e.g. PCBs, DDT, chlordane) contaminants. For this reason, only data collected after 1999 were used in the RIA. Use of the Wente covariance model removes bias that might be introduced by fish samples by correcting for variability in mercury concentrations attributable to differences in trophic level, length and age from those species most typically consumed by humans.

It should be noted that MMaps was designed to address an important, but very specific issue – that of eventual response of fish tissue to air deposition reductions. As such it responds to a need to understand how mercury reductions, independent of other changes in the environment, will impact fish contamination and human health. More complex models are required in cases where more complete descriptions are needed. A dynamic model is essential for modeling waterbody recovery during the period in which waterbody response lags reductions in mercury loads. A dynamic model is also essential for understanding seasonal fluctuations, as well as year-to-year fluctuations due to meteorological variability. Finally, a more complex model would be essential for assessing the impact of other watershed and water quality changes (e.g. erosion, wetlands coverage, and acid deposition) that might affect mercury bioaccumulation in fish. These complex models are used to derive the MMaps approach, and are themselves based on a number of assumptions. The science of mercury fate and transport in the environment is an actively evolving area of research (e.g. see US EPA, 2003c). While these assumptions are considered reasonable given the state of the science of environmental modeling and mercury in the environment, the validity of assumptions inherent in both the MMaps approach and dynamic ecosystem scale models will need to be reevaluated as the science of mercury fate and transport evolves.

The MMaps methodology was peer reviewed by a set of national experts in the fate and transport of mercury in watersheds. While two reviewers felt it could be used to predict future fish tissue concentrations, a third cautioned it should not be considered a robust predictor until scientific data can be generated to validate the approach. Reviewers systematically identified a set of implicit assumptions that compose the steady state assumption in the MMaps approach. They pointed out that due to evolving and complex nature of the science of mercury, some features of the complex models are assumptions themselves, and thus cannot be wholly relied upon as ultimate predictors of mercury fate and transport. The reviewers pointed out that there is limited scientific information to directly verify this approach, and that some scientific data appears to refute individual components of the overall steady state assumption. One reviewer did perform a D-MCM and MMaps comparison, and found that, under these assumptions, MMaps model did produce comparable steady-state results as the D-MCM model. There was considerable discussion about how best to aggregate the data, to scale up to a deposition reduction requirement, from fish-specific and waterbody specific information. The peer review report has not been released because the document that it relates to has not yet been approved for release by EPA. However, the description of the approach, and the methodologies as applied in this RIA, are largely consistent with the peer review recommendations.

The MMaps report (US EPA, 2001) presented a national-scale application of Mercury Maps to determine the percent reductions in air deposition that would be needed in watersheds across the country for average fish tissue concentrations to achieve the national methylmercury criterion. In this national scale assessment, fish tissue concentrations were aggregated at the scale of large watersheds, thus presenting average results for each watershed. The use of other scales of aggregation, e.g., waterbody specific, is consistent with the Mercury Maps approach to the degree to which different mercury loads can be discerned.

3.1.2 Goal/Purpose of Ecosystem Case Studies

To supplement the MMaps methodology, this report explores the range in temporal responses of different ecosystems following reductions in atmospheric Hg emissions and some of the sources of uncertainty around the proportional relationship used by the MMaps model. To do this, we provide quantitative examples from five case studies of a range of freshwater ecosystem types across the Eastern and Midwestern United States. For all of these systems, we applied an updated dynamic version of the IEM-2M model originally used in the *Mercury Study Report to Congress* (USEPA, 1997) to forecast the time lag of fish mercury concentrations in each ecosystem to different atmospheric mercury input scenarios. We present our results in the context of recent scientific findings related to the temporal response of different ecosystems and the factors affecting accumulation of mercury in fish.

Because different ecosystems exhibit dramatically different responses to changes in mercury loading depending on their chemical and physical attributes, results from individual case studies must be qualified by their representativeness of ecosystem variability across the United States. Using georeferenced empirical databases that describe some of the watershed and waterbody characteristics across United States, we describe a preliminary assessment of the variability in some factors known to be important for MeHg formation and bioaccumulation in fish. Although this analysis has not been completed, the concept is demonstrated by a *preliminary qualitative* assessment of how much of the ecosystem variability in MeHg formation and bioaccumulation has been captured by the modeling case studies. Developing broad categories of ecosystem types based on their propensity for MeHg formation and bioaccumation in fish and their frequency of occurrence is an iterative effort. By combining the frequency of each category of ecosystem type with the magnitude and time lag in fish tissue reductions modeled using dynamic, ecosystem scale models, such an analysis could ultimately provide an alternate methodology for a national scale assessment of expected changes in fish MeHg concentrations resulting from reductions in atmospheric mercury deposition.

EPA acknowledges that present modeling capabilities do not allow *a priori* predictions of ecosystem responses due to considerable uncertainties in the science. These epistemic uncertainties limit the predictive power of both the MMaps approach and the models applied in this exercise (see Peer Review Comments, Appendix 8). Because of these limitations, modeling scenarios employed in this study are first calibrated to real ecosystem and rely heavily on empirical data to develop credible rate constants and flux terms for each of the case studies investigated. EPA's Office of Research and Development views this modeling exercise as part a series of iterative modeling development phases that will eventually allow us to achieve our goal of *a priori* modeled responses. Accordingly, this report highlights advances in our modeling capabilities since the publication of the *Mercury Study Report to Congress* in 1997. EPA will continue to develop the models described in this report by incorporating the latest scientific knowledge on the factors that control the distribution and accumulation of mercury in freshwater and coastal marine food web. This goal is consistent with EPA's Mercury Research Strategy (U.S. EPA, 2000).

3.2 Recent Advances in Mercury Science

This modeling exercise is based on our understanding of how mercury cycles through ecosystems and accumulates in fish. The set of physical, chemical, and biological processes controlling mercury fate in watersheds and water bodies can be synthesized into a general conceptual model that guides our model selection, refinement, and application. These processes can be grouped into specific categories: mercury cycle chemistry; mercury processes in the atmosphere, soils and water; bioavailability of mercury in water; and mercury accumulation in the food web. The following is a narrative of our conceptual model, discussing the recent scientific developments that have added to our understanding of mercury processes. This review builds upon the work previously summarized in EPA's Mercury Report to Congress (USEPA, 1997). The end of this section concludes with the conceptual model summary.

3.2.1 Mercury Cycle Chemistry

Mercury occurs naturally in the environment as several different chemical species. The majority of mercury in the atmosphere (95-97%) is present in a neutral, elemental state (Hg^0) (Lin and Pehkonen, 1999), while in water, sediments and soils the majority of mercury is found in the oxidized, divalent state (Hg(II)) (Morel et al., 1998). A small fraction (percent) of this pool of divalent mercury is transformed by microbes into methylmercury ($CH_3Hg(II)$/ MeHg) (Jackson, 1998). Methylmercury is retained in fish tissue and is the only form of mercury that biomagnifies in aquatic food webs (Kidd et al., 1995). As a result, methylmercury concentrations in higher trophic level organisms such as piscivorous fish, birds and wildlife are often 10^4-10^6 times higher than aqueous methylmercury concentrations (Jackson, 1998). Transformations among mercury species within and between environmental media result in a complicated chemical cycle. Mercury emissions from both natural and anthropogenic sources are predominantly as Hg(II) species and Hg^0 (Landis and Keeler, 2002; Seigneur et al., 2004). Anthropogenic point sources of mercury consist of combustion (e.g., utility boilers, municipal waste combustors, commercial/industrial boilers, medical waste incinerators) and manufacturing sources (e.g., chlor-alkali, cement, pulp and paper manufacturing) (USEPA, 1997). Natural sources of mercury arise from geothermic emissions such as crustal degassing in the deep ocean and volcanoes as well as dissolution of mercury from geologic sources (Rasmussen, 1994).

3.2.2 Mercury Processes in the Atmosphere

The relative contributions of local, regional and long range sources of mercury to fish mercury levels in a given water body are strongly affected by the speciation of natural and anthropogenic emissions sources. Elemental mercury is oxidized in the atmosphere to form the more soluble mercuric ion (Hg(II)) (Schroeder et al., 1989). Particulate and reactive gaseous phases of Hg(II) are the principle forms of mercury deposited onto terrestrial and aquatic systems because they are more efficiently scavenged from the atmosphere through wet and dry deposition than Hg^0 (Lindberg and Stratton, 1998). Because Hg(II) species or reactive gaseous mercury (RGM) and particulate mercury (Hg(p)) in the atmosphere tend to be deposited more locally than Hg^0, differences in the species of mercury emitted affect whether it is deposited locally or travels longer distances in the atmosphere (Landis et al., 2004). Recent research indicates that certain meteorological conditions and atmospheric constituents can result in the rapid oxidation of Hg^0 to RGM, potentially increasing the fraction of Hg^0 from anthropogenic sources that is deposited more locally (Landis, Pers. Comm., 2004).

Atmospheric models use various mathematical frameworks to describe how meteorology and atmospheric chemistry interact with different mercury species from a variety of sources to determine mercury deposition (e.g., (Bullock and Brehme, 2002; Cohen et al., 2004). Modeling the atmospheric fate and transport of mercury is outside of the scope of this project, although outputs of several atmospheric models will be used in combination with available empirical data to estimate atmospheric deposition of mercury to different water bodies under different regulatory scenarios.

3.2.3 Mercury Processes in Soils

A portion of the mercury deposited in terrestrial systems is re-emitted to the atmosphere. On soil surfaces, sunlight may reduce deposited Hg(II) to Hg^0, which may then evade back to the atmosphere (Carpi and Lindberg, 1997; Frescholtz and Gustin, 2004; Scholtz et al., 2003). Significant amounts of mercury can be co-deposited to soil surfaces in throughfall and litterfall of forested ecosystems (St. Louis et al., 2001), and exchange of gaseous Hg^0 by vegetation has been observed (e.g., (Gustin et al., 2004).

Hg(II) has a strong affinity for organic compounds such that inorganic Hg in soils and wetlands is predominantly bound to dissolved organic matter (Mierle and Ingram, 1991). MeHg likewise forms stable complexes with solid and dissolved organic matter (Hintelmann and Evans, 1997). These complexes can dominate MeHg speciation under aerobic conditions (Karlsson and Skyllberg, 2003). Truly dissolved and dissolved organic carbon (DOC)-complexed Hg(II) and MeHg are transported by percolation to shallow groundwater, and by runoff to adjacent surface waters (Ravichandran, 2004). Sorbed Hg(II) and MeHg are transported by erosion fluxes to depositional areas on the watershed and to adjacent surface waters (e.g., (Hurley et al., 1998).

Concentrations of MeHg in soils are generally very low. In contrast, wetlands are areas of enhanced MeHg production and account for a significant fraction of the external MeHg inputs to surface waters that have watersheds with a large portion of wetland coverage (e.g., (St. Louis et al., 2001). Accordingly, there is a positive relationship between MeHg yield and percent wetland coverage (Hurley et al., 1995). Hydrology exerts an important control on the magnitude and flux of MeHg in wetland ecosystems (Branfireun and Roulet, 2002), as well as the transport of inorganic mercury deposited in a given watershed to surface waters (Babiarz et al., 2001).

It should also be noted that there are exceptions to the predominace of wetlands as an external source of MeHg to surface waters. For example, preliminary simulations with a multicell version of EPRI's Dynamic Mercury Cycling Model (D-MCM®) for Lake Superior generated a model result where direct atmospheric deposition of methylmercury was predicted to be the largest single source of methylmercury to the waterbody (Harris et al., 2002). This was not because methylmercury deposition rates were unusually high. It was instead because in-situ production and terrestrial loading of MeHg were predicted to be low.

3.2.4 Mercury Processes in Water

In a water body, deposited Hg(II) is reduced to Hg^0 by ultraviolet and visible wavelengths of sunlight as well as microbially mediated reduction pathways (Amyot et al., 2000; Mason et al., 1995). In turn, Hg^0 is oxidized back to Hg(II), driven by sunlight as well as by "dark" chemical or biochemical processes (Lalonde et al., 2001; Zhang and Lindberg, 2001). Driven by wind and water currents, dissolved Hg^0 in the water column is volatilized, which can be a significant removal mechanism for mercury in surface waters and a net source of mercury to the atmosphere (Siciliano et al., 2002).

In the water column and sediments, Hg(II) partitions strongly to silts and biotic solids, sorbs weakly to sands, and complexes strongly with dissolved and particulate organic material. The abundance of various inorganic ligands (e.g., OH^-, Cl^-, S^{2-}, DOC) in freshwater and saltwater ecosystems plays an important role in both oxidation and reduction of inorganic mercury as well as its bioavailability to methylating microbes. For example, reduction of Hg(II) is hypothesized to be a function of the predominance of $Hg(OH)_2$, which is inversely correlated with pH (Mason et al., 1995). Reduction of Hg(II) to Hg^0 and subsequent volatilization from the water column is important because it effectively reduces the pool of inorganic mercury that could potentially undergo conversion to MeHg.

Hg(II) and MeHg sorbed to solids settle out of the water column and accumulate on the surface of the benthic sediment layer. Surficial sediments interact with the water column via resuspension and bioturbation. The burial of sediments below the surficial zone can be a significant removal mechanism for contaminants in surface sediments (e.g., (Gobas et al., 1998; Gobas et al., 1995). The depth of the active sediment layer is a highly sensitive parameter for predicting the temporal response of different ecosystems to changes in mercury loading in environmental fate models. This is because the reservoir of Hg(II) potentially available for conversion to MeHg in the sediments is a function of the depth and volume of the active sediment layer. The.compartment conducive for methylation is similarly affected (Harris and Hutchison, 2003; Sunderland et al., 2004). Physical characteristics of different ecosystem types affect estuarine mixing and sediment resuspension, which also affect the production of MeHg in the water and sediments (Rolfhus et al., 2003; Sunderland et al., 2004; Tseng et al., 2001).

3.2.5 Bioavailability of Inorganic Mercury to Methylating Microbes

The amount of bioavailable MeHg in water and sediments of aquatic systems is a function of the relative rates of mercury methylation and demethylation. In the water, MeHg is degraded by two microbial processes and sunlight (Barkay et al., 2003; Sellers et al., 1996). Recent research has shown that demethylating Hg-resistant bacteria may adapt to systems that are highly contaminated with total mercury, helping to explain the paradox of low MeHg and fish Hg levels in these systems (Schaefer et al., 2004).

Mass balances for a variety of lakes and coastal ecosystems show that *in situ* production of MeHg is often one of the main sources of MeHg in the water and sediments (Benoit et al., 1998; Bigham and Vandal, 1994; Gbundgo-Tugbawa and Driscoll, 1998; Gilmour et al., 1998; Mason et al., 1999). Sulfate reducing bacteria (SRB) are thought to be the principle agents responsible for the majority of MeHg production in aquatic systems (Beyers et al., 1999; Compeau and Bartha, 1987; Gilmour and Henry, 1991). SRB thrive in the redoxocline, where the maximum gradient between oxic and anoxic conditions exists (Hintelmann et al., 2000). Thus, in addition to the presence of bioavailable Hg(II), MeHg production and accumulation in aquatic systems is a function of the geochemical parameters that enhance or inhibit the activity of methylating microbes, especially sulfur concentrations, redox potential (Eh) and the composition and availability of organic carbon.

A number of factors affect the bioavailabilty of Hg(II). A strong inverse relationship between complexation of Hg(II) by sulfides and MeHg production has been demonstrated in a number of studies (Benoit et al., 1999; Benoit et al., 1999; Craig and Bartlett, 1978; Craig and Moreton, 1986). Passive diffusion of dissolved, neutral inorganic mercury species is hypothesized as one of the main modes of entry across the cell membranes of methylating microbes (Benoit et al., 1999; Benoit et al., 2003; Benoit et al., 1999). Thus, the formation of neutral, dissolved mercury species such as $HgCl_2$, $Hg(OH)_2$, HgClOH, and HgS^0(aq.), which depend on the availability of constituent ligands in the surface and interstitial waters, may strongly influence the availability of inorganic mercury to SRB, although our understanding of the forms of mercury that are bioavailable to methylating microbes is currently incomplete (Benoit et al., 2001; Benoit et al., 1999; King et al., 2001). The availability of the pool of inorganic mercury in the water and sediments for methylation is also dominated by the binding of Hg(II) with dissolved organic matter (DOM) complexes (Ravichandran, 2004).

Changes in the bioavailability of inorganic mercury and the activity of methylating microbes as a function of sulfur, carbon and ecosystem specific characteristics mean that ecosystem changes and anthropogenic "stresses" that do not result in a direct increase in mercury loading to the ecosystem but alter the rate of MeHg formation may also affect mercury levels in organisms (e.g., (Grieb et al., 1990). Because mercury concentrations in fish can increase even when there has been no change in the total amount of mercury deposited in the ecosystem, environmental changes such as eutrophication, which may alter microbial activity and the chemical dynamics of mercury within an ecosystem, must be considered together with emission control strategies to effectively manage mercury accumulation in the food web.

Recent research indicates that the bioavailability or reactivity of newly deposited Hg(II) may be greater than older "legacy" mercury in the system (Hintelmann et al., 2002). These results suggest that lakes receiving the bulk of their mercury directly from deposition to the lake surface (e.g., some seepage lakes) would see fish mercury concentrations respond more rapidly to changes in atmospheric deposition than lakes receiving most of their mercury from watershed runoff. The implications of these data are also that systems with a greater surface area to watershed area ratio that receive most of their inputs directly from the atmosphere (e.g., seepage lakes) may respond more rapidly to changes in emissions and deposition of mercury than those receiving significant inputs of mercury from the catchment area.

3.2.6 Mercury Accumulation in the Food Web

Dissolved Hg(II) and MeHg accumulate in aquatic vegetation, phytoplankton, and benthic invertebrates. Unlike Hg(II), MeHg biomagnifies though each successive trophic level in both benthic and pelagic food chains such that mercury in predatory, freshwater fish is found almost exclusively as MeHg (Bloom, 1992; Watras et al., 1998). Thus, trophic position and food-chain complexity plays an important role in MeHg bioaccumulation (Kidd et al., 1995).

The chemical and physical characteristics of different ecosystems affect MeHg uptake at the base of the food chain, driving bioaccumulation at higher trophic levels. At the base of pelagic freshwater food-webs, MeHg uptake by plankton is thought to be a combination of passive diffusion and facilitated transport (Laporte et al., 2002; Watras et al., 1998). Uptake of MeHg by plankton can be enhanced or inhibited by the presence of different ligands bound to MeHg (Lawson and Mason, 1998). Similarly, the assimilation efficiency of MeHg at the base of the food chain is also affected by the type of dissolved MeHg-complexes in the water and sediments. This may be a function of differences in the ability of organisms to solubilize MeHg through digestive processes with different MeHg complexes (Lawrence and Mason, 2001; Leaner and Mason, 2002). The presence of organic ligands and high concentrations of DOC in aquatic ecosystems are generally thought to limit MeHg uptake by biota (Driscoll et al., 1995; Sunda and Huntsman, 1998; Watras et al., 1998).

In fish, MeHg bioaccumulation is a function of several uptake (diet, gills) and elimination pathways (excretion, growth dilution) (Gilmour et al., 1998; Greenfield et al., 2001). As a result, the highest mercury concentrations for a given fish species correspond to smaller, long-lived fish that accumulate MeHg over their life span with minimal growth dilution (e.g., (Doyon et al., 1998). In general, higher mercury concentrations are expected in top predators, which are often large fish relative to other species in a waterbody.

3.2.7 Summary of Findings in the METAALICUS Study

METAALICUS is a whole-ecosystem experiment examining the relationship between atmospheric mercury deposition and fish mercury concentrations (Harris et al., 2004). Stable, non-radioactive isotopes of inorganic Hg(II) are being added to an 8.3-ha lake and its 44-ha watershed in the Experimental Lakes Area (ELA), Ontario, Canada. Using isotopes provides the ability to follow newly deposited mercury separately from background mercury. Different Hg(II) isotopes are being applied to uplands, wetlands and the lake surface to distinguish the contributions of each of these sources to fish mercury levels. Beginning in 2001, and continuing each year since (3 years to-date), annual wet deposition of atmospheric Hg(II) has been increased experimentally 3-4 fold relative to long term average wet deposition rates to the area. Annual mercury additions to the lake surface were 22 $\mu g\ m^{-2}$ each year. Upland and wetland areas received isotopes at average annual application rates of 21-25 and 25-28 $\mu g\ m^{-2}\ yr^{-1}$ respectively for the 2001-2003 period.

During the first season of additions (2001), concentrations of inorganic mercury in the surface waters of Lake 658 nearly doubled as a result of the mercury isotope (^{202}Hg) added directly to the lake surface. This represented a nearly proportional response for inorganic mercury, relative to the increase in mercury loading to the lake as a result of the spikes to the lake surface. Inorganic ^{202}Hg added to the lake surface was also detected as MeHg in the first season in the water column, sediments and biota, including fish. Concentrations of ^{202}Hg-labelled MeHg in water, sediments and biota continued to increase in 2002 and 2003. Different response dynamics were observed for the buildup of added mercury as MeHg in different compartments. Results to-date suggest that the system has not yet stabilized in response to the annual isotope additions directly to the lake surface. Inorganic mercury isotopes added to the terrestrial system were measured in the uplands and wetlands and observed at near-detection levels in lake waters by late 2002 (upland isotope only), but were not yet detectable in fish as of 2003. Initial efforts to simulate the L658 experiment with a mass balance model of aquatic Hg cycling were unable to match the rate at which the isotope applied to the lake as inorganic mercury was observed as MeHg in the system. The apparent higher bioavailability for methylation of newly added mercury compared to "older" mercury may be a factor.

3.2.8 Summary of Florida Everglades Study

The Florida Everglades TMDL Pilot Study is one of the best-known investigations of the temporal response of an aquatic system to reduced mercury loading. In the Everglades, elevated mercury concentrations in fish are caused by a combination of atmospheric loading, net methylation in water column periphyton, and food web dynamics(Atkeson, 2003). Periphyton and macrophytes influence fish levels through their control of available divalent and methyl mercury in the water column.

Incinerator mercury emissions in southern Florida have declined approximately 99% since the mid-1980's as a result of pollution prevention and control policies. In general accord, mercury in fish and wildlife of the Everglades has declined by approximately 60% since the mercury peaked in biota in the mid-1990's.

In 1999 Florida DEP and USEPA began a modeling analysis of the environmental cycle of mercury to explore the tools and data needed to perform a Total Maximum Daily Load analysis (TMDL) for an atmospherically derived pollutant. Extensive Everglades specific data are available to support a linked, multi-media modeling analysis through the auspices of the South Florida Mercury Science Program, a 10-year multi-agency program of research, modeling and monitoring studies.

The dynamic mercury cycling model (E-MCM) was applied to investigate changes in fish tissue Hg (at site at WCA 3A-15) with declines in atmospheric mercury deposition as part of the Pilot TMDL study for that site (Tetra Tech Inc., 2000). Model simulations showed that regardless of the magnitude of the load reduction, fish mercury concentrations were predicted to change by 50% of the ultimate response within 8-9 years. Within 25-30 years, 90% of the ultimate predicted response has occurred. In all cases, the actual magnitude of the change in fish Hg was dependent on the magnitude of the load reduction. In the above simulations, a 3 cm thick active sediment layer was assumed and the model did not distinguish between new and old or "legacy" Hg.

At steady state, E-MCM forecasts a linear relationship between atmospheric mercury deposition and mercury concentrations in largemouth bass, with a small residual mercury concentration in fish at zero atmospheric mercury deposition: for any reduction in mercury inputs to the Everglades a slightly lesser reduction in fish mercury concentrations may be anticipated. Furthermore, the E-MCM predicts near equivalence between the change in atmospheric mercury deposition rate and the change in largemouth bass mercury concentration over the likely range for current estimates of atmospheric deposition of mercury. The slight offset from a 1:1 relationship results from slow mobilization of historically deposited mercury from deeper sediment layers to the water column. Until buried below the active zone, this mercury can continue to cycle through the system. In addition, because mercury is a naturally occurring element, fish tissue mercury concentrations can never be reduced to zero. Further, the model showed that absent changes to the system other than mercury loading (e.g. sulfur or nutrient cycling, or hydrology), an ~80% reduction from the ca. 1996 peak total annual mercury atmospheric deposition would be needed for mercury concentrations in a 3-year old largemouth bass in the central Everglades to be reduced to less than Florida's present fish consumption advisory action level of 0.5 mg/kg.

Despite the quality of the modeling done as part of the Everglades study, this analysis has limited applicability to other aquatic systems because of unusual attributes of the Everglades. These attributes are listed below.

- Physiography of the waterbody — As a flat, shallow, vegetated marshland the Everglades is atypically vulnerable to atmospheric deposition because of its great surface-to-volume ratio.
- Climate — Year-round high temperature and insolation stimulates chemical and physical processes, promoting rapid aquatic cycling and unusually high production of MeHg.
- Meteorology — Easterly trade winds typify the synoptic transport regime during the summer when ~ 85% of rainfall and ~ 90% of mercury deposition occurs. This pattern efficiently brings emissions from the southeast coastal counties of Florida out over the Everglades where frequent thunderstorms focus deposition there.
- Sources — Incineration was the largest emissions category in south Florida through the mid-1990's. A predominance of emissions was 'reactive gas-phase mercury' (RGM or Hg(II)) which tends to deposit on a local scale.
- Synergy — Coupled with meteorology described above, the dominance of emissions as RGM has resulted in an unusually tight local-scale coupling between emissions in southern Florida and local-scale deposition.

3.3 Overview of Models Used in This Study

The general approach taken in this project was to couple outputs from atmospheric fate and transport models with a set of watershed and water body models that are parameterized using empirical data from well-characterized ecosystems.

3.3.1 Atmospheric Models

Over the past decade, EPA has used a variety of analytical and numerical simulation tools to project the atmospheric transport, chemistry, and deposition of both criteria (*e.g.*, ozone, fine particles, etc.) and toxic (*e.g.*, Hg) air pollutants. These models range in complexity from simple, one-layer Gaussian dispersion models (e.g., Industrial Source Complex (ISC3) model) to more complex, multi-layer Lagrangian puff-type trajectory models (e.g., Hybrid Single Particle

Lagrangian Integrated Trajectory (HYSPLIT) model), and finally to complex three dimensional (3–D) Eulerian grid models (e.g., Community Multiscale Air Quality (CMAQ) model). EPA and others have been using a suite of complex numerical models to assess the transport and fate of Hg emissions in the local, regional, and global atmosphere. In the *Utility Report to Congress*, EPA relied heavily on the ISC3 dispersion model to assess nearfield Hg deposition effects. The HYSPLIT model has also been used extensively in the Great Lakes and Chesapeake Bay watersheds to analyze source-receptor relationships for Hg deposition in these areas (Cohen et al., 2004).

A review of the strengths and weaknesses of atmospheric mercury fate and transport modeling is beyond the scope of this project. However, projected deposition scenarios for different water bodies obtained from the models described above serve as inputs to the aquatic fate and transport modeling described in this chapter. Modeled deposition rates (2001) using the CMAQ and REMSAD models for each of the five ecosystems were compared to site-specific data. Local monitoring data and sedimentary records of total mercury deposition obtained from dated sediment cores (where available) were used obtain the best possible estimates of overall loading to individual ecosystems. Outputs from atmospheric mercury cycling models (REMSAD/CMAQ) were then used to forecast the relative change in mercury inputs to each ecosystem modeled when contributions from coal-fired utilities were removed (e.g., the percent difference in atmospheric deposition for each ecosystem between 2001 and scenario removing coal-fired utilities as a source) to isolate their contribution the fraction of mercury accumulation in fish. Details of the atmospheric deposition scenarios are described below in Section 3.4.

3.3.2 Ecosystem Models

EPA has developed a set of watershed, water body, and food web models that describe the speciation, transport, and bioaccumulation of mercury as a function of the physical and chemical properties of a specific ecosystem. The selected watershed and water body models have been recently applied to various case studies. In this project, they were refined for consistency and used to construct ecosystem specific mass balances for the three principle mercury components – inorganic divalent mercury, Hg(II), elemental mercury, Hg^0, and monomethyl mercury, $CH_3Hg(II)$ (or MeHg). We compare the results from the SERAFM and WASP waterbody fate and transport models (Section 3.4) as an internal check on consistency of the modeling results. For more information on the specific models described below, please see http://www.epa.gov/athens and www.epa.gov/crem.

3.3.2.1 Overview of the SERAFM Model

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 Report to Congress* (USEPA, 1997). 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.
- *Watershed Loading*: 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.
- *Two-Layer*: 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.
- *Photo-reactions*: 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.
- *Speciation*: 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 HgII is multiplied by the fraction of dissolved divalent mercury and the fraction of HgII present as $Hg(OH)_2$.
- *Equilibrium Partitioning*: 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).
- *Trophic status*: 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.
- *Suspended particle types in the water column*: 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.
- *Reaction rates*: The SERAFM model incorporates more recent reaction rate coefficients, and the understanding of the variability of these rates with different conditions.
- *Partition coefficients*: 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.

State variables in both the IEM-2M and SERAFM models include three mercury species, Hg^0, Hg(II), and MeHg. As mentioned above, SERAFM includes four solids types (abiotic solids, phytoplankton solids, zooplankton solids, and detrital solids) 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. SERAFM calculates the time-dependent solids and mercury species concentrations in the water column and sediments of the specified water body reach. Hg(II) and MeHg are partitioned to suspended and benthic solids and to dissolved DOC with user-specified partition coefficients for each sorbent type.

In the SERAFM model, mercury species are subject to several transformation reactions, including photo-oxidation and dark oxidation of Hg^0 in the water column, photo-reduction and methylation of Hg(II) in the water column and sediment layers, and photo-degradation and demethylation of MeHg in the water column and sediment layers. Water column oxidation, reduction and demethylation reactions are driven by sunlight, and so their input rate constants are attenuated through the water column using specified light extinction coefficients. Hg^0 is subject to volatile exchange between the water column and the atmosphere governed by a transfer rate calculated from velocity and depth, and by its Henry's Law constant.

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 3-1). 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 MeHg loading. When these parameters are updated to reflect current knowledge, forecasted fish mercury concentrations are higher than the original IEM-2M results (see Table 3-1).

Table 3- 1. Comparison of SERAFM and IEM-2M Forecasted Mercury Concentrations Using Parameter Values for Model Ecosystem Described in the Mercury Study Report to Congress (RtC) and a 50% Reduction in Atmospheric Deposition

Parameters	RtC Model Ecosystem	RtC Model Ecosystem	Updated Parameters
Model	**IEM-2M**	**SERAFM**	**SERAFM**
Water Column MeHg Unfiltered	0.08	0.031 ng L^{-1}	0.12 ng L^{-1}
Water Column HgT 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: FishHg/MeHg		6.8×10^6	

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 MeHg from the catchment.

3.3.2.2 Overview of the WASP Model

WASP (Water Quality Analysis Simulation Program) is a dynamic, mass balance framework for modeling contaminant fate and transport in surface water systems. This model helps users interpret and predict water quality responses to natural phenomena and man-made pollution for various pollution management decisions. WASP is an enhancement of the original WASP (Ambrose, 1987; Ambrose, 1988; Connolly and Thomann, 1985; Di Toro et al., 1983) and allows the user to investigate 1, 2, and 3 dimensional systems, and a variety of pollutant types. The time-varying processes of advection, dispersion, point and diffuse mass loading and boundary exchange are represented in the model. WASP also can be linked with hydrodynamic and sediment transport models that can provide flows, depths, velocities, and temperature, salinity and sediment fluxes.

The WASP7 mercury module simulates three mercury species, Hg^0, Hg(II), and MeHg, as well as three solids types (silt, sand, and biotic solids) (e.g., (Ambrose and Wool, 2001). Simulations are driven by the speciated mercury loadings delivered from the atmosphere, from watershed tributaries, and from point sources. Throughout the simulation period, WASP calculates solids and mercury species concentrations in the water column and sediments of each reach. Transport processes simulated include advection, dispersion, and sediment-water column exchange. Hg(II) and MeHg are partitioned to silt, sand, and biotic solids, and to dissolved organic carbon (DOC).

3.3.2.3 Overview of the WCS Model

Although significant progress has been made in recent years on estimations of mercury transport fluxes in watershed areas within a region in which atmospheric deposition is presumably constant (Balogh et al., 1998; Hurley et al., 1995; Lawson et al., 2001; Lee et al., 1995; Tsiros, 1999), only a few watershed studies have focused on the importance of indirect anthropogenic sources of Hg such as terrestrial runoff, compared to direct atmospheric deposition. The EPA Region 4 Watershed Characterization System (WCS) is a GIS-based modeling system for calculating soil particle transport and pollutant fate in watersheds (Greenfield et al., 2002). Its mercury transport module was developed from the IEM-2M model, which calculated mercury species concentrations in an idealized watershed and water body based on steady atmospheric mercury deposition and long-term average hydrology. Similarly, the WCS calculates long-term average hydrology and sediment yield, but simulates total Hg(II) in a more realistic, distributed sub-watershed network. A second-generation WCS that operates on a finer computational grid is under development. Initial background soil mercury concentrations along with wet and dry atmospheric mercury deposition fluxes are input to the model. For pervious subwatershed grid elements, WCS calculates surficial soil mercury concentrations over time using a mass balance. Calculated total mercury in the surficial soil layers is partitioned between the dissolved and particulate phases (in the soil water and on the soil solids) assuming local equilibrium, governed by a partition coefficient. Dissolved mercury is lost from the surficial soil layers through percolation and runoff. Particulate mercury is lost through water runoff erosion. No wind resuspension is included in the model calculations. A fraction of the soil mercury is reduced and volatilized back to the atmosphere. Subwatershed mercury loadings in runoff water and runoff erosion particles are delivered to the watershed tributary system. For impervious areas and water surface areas within the subwatersheds, atmospheric mercury deposition is delivered to the tributary system without loss.

Surficial soil concentrations for each of the three mercury components are calculated in WCS on a daily basis using a mass balance equation driven by an input term for atmospheric deposition. Mercury output terms include gaseous flux from soil to air by volatilization, vertical hydrologic transport through soil by percolation, and horizontal hydrologic transport from surficial soil by runoff water and runoff erosion particles. A source/sink term for mercury cycling with first-order kinetics is used to represent oxidation of Hg^0, methylation of Hg(II), demethylation of MeHg, and abiotic reduction of Hg(II) to Hg^0, with a functional dependence of the reduction rate on soil moisture and vegetation cover shading (Tsiros, 2002).

3.3.2.4 Overview of the BASS Model

BASS (Bioaccumulation and Aquatic System Simulator) describes the dynamics of mercury bioaccumulation in the food chain using algorithms that account for mercury accumulation among different species and different age classes using species-specific uptake and elimination terms such as diet composition and growth dilution (Barber, 2001; Barber et al., 1987; Barber et al., 1988; Barber et al., 1991).

BASS simulates the population and bioaccumulation dynamics of age-structured fish communities. Although BASS was specifically developed to investigate the bioaccumulation of chemical pollutants within a community or ecosystem context, it can also be used to explore population and community dynamics of fish assemblages that are exposed to a variety of non-chemical stressors such as altered thermal regimes associated with hydrological alterations or industrial activities, commercial or sports fisheries, and introductions of non native or exotic fish species. Contaminants entering each fish through gill exchange and ingestion are partitioned internally to water, lipid, and non-lipid organic material. Internal equilibrium among these phases is assumed to be rapid in comparison with external exchanges. Stability coefficients are specified for the binding of MeHg to available sulfhydryl groups in the fish's non-lipid organic material.

BASS's model structure is very generalized and flexible. Users can simulate both small, short-lived species (e.g., daces, minnows, etc.) and large, long-lived species (e.g., bass, perch, sunfishes, trout, etc.) by specifying either monthly or yearly age classes for any given species. The community's food web is defined by identifying one or more foraging classes for each fish species based on body weight, body length, or age. The dietary composition of each of these foraging classes is then specified as a combination of benthos, incidental terrestrial insects, periphyton/attached algae, phytoplankton, zooplankton, and/or other fish species, including its own. One of the strengths of the BASS model relative to other bioaccumulation model frameworks is that there are no restrictions on the number of chemicals or the number of fish species that can be simulated, the number of cohorts/age classes that fish species may have, or the number of foraging classes that fish species may have (Barber, 2003).

3.4 Overview of Case Studies

Case studies presented in this section are used to explore the range in temporal responses of different ecosystems following reductions in atmospheric Hg emissions and some of the sources of uncertainty around the proportional relationship used by the MMaps model. To do this, we provide quantitative examples from five case studies that span a range of freshwater ecosystem types across the Eastern and Midwestern United States.

The five case studies selected for this project were constrained to reasonably well-studied ecosystems where there were sufficient empirical data to parameterize time-dependent aquatic cycling models. Our goal was to select well-characterized sites that would display a range in the magnitude and timing of responses to changes in mercury loading because of differences in ecosystem specific factors that affect the transport and transformation of mercury in different water bodies. The sites selected (see Section 3.5) cover a range of ecosystem types, sizes and latitudes.

3.4.1 Ecosystem Characteristics

A brief description of each site is given below and a summary of ecosystem characteristics used to parameterize the models are detailed in Table 3-2.

- *Eagle Butte, South Dakota:* Livestock lakes and ponds on the Cheyenne River Sioux Tribal Lands. The site modeled (Lee Dam), is a shallow, well-mixed system with a water surface area of 0.2 km^2 and a catchment to lake area ratio of 22.6. There are power plants in the vicinity of this site, although the prevailing meteorology likely transports most emissions east of the site. Currently, consumption advisories are in place on reservation lands due to high levels of mercury in piscivorous fish. Mercury dynamics at this site are currently being studied as part of an EPA Regional Office RARE Grant awarded in 2003 (http://www.epa.gov/osp/regions/RARE_Region8.pdf). Atmospheric deposition data, total mercury and MeHg concentrations in sediments, water and biota have all been collected as part of this study. To model this system, we used the empirical data collected at this site to parameterize both the SERAFM and WASP models. The BASS model was applied to investigate mercury residue attenuation in length classes of co-dominant fishes.
- *Pawtuckaway Lake, New Hampshire:* Medium sized, seepage lake in Nottingham, New Hamsphire. This lake has a water surface area of 3.6 km^2 and a catchment to lake area ratio of 13.7. Pawtuckaway Lake is characteristic of undisturbed lakes within the Northeastern Highlands Ecoregion (Omernik, 1987; US EPA, 2000). This lake was part of a recent study of mercury dynamics across a number of Vermont and New Hampshire Lakes funded by EPA's Office of Research and Development under the Regional Environmental Monitoring and Assessment Program (Kamman et al., 2004). Local mercury sources in the region include a number of utility units (See Figure 3-8) and several incinerators.
- *Lake Waccamaw, North Carolina:* Large, bay lake in southeastern North Carolina. Lake Waccamaw has a water surface area of almost 35 km^2 and a catchment to lake area ratio of slightly more than six (Table 3-2). In 1992, a survey of fish mercury concentrations North Carolina's Department of Environment, Health and Natural Resources in this region revealed that fish mercury concentrations exceeded 1 ppm in over 60% of the samples. Waccamaw is a popular destination for recreational fishing and a fish consumption advisory is currently in place. The area surrounding Lake Waccamaw is typical of the region: flat terrain with ubiquitous wetlands and waterways. Very little commercial or industrial activity takes place in the area immediately surrounding the park, population density is relatively low and roadways are lightly traveled. The nearest town is Whiteville, NC, located approximately 15 kilometers to the west-northwest of Lake Waccamaw. A variety of mercury sources are located in this region including at

least two coal-fired electric utility boilers, a large municipal waste incinerator, several large coal or oil-fired industrial boilers, and a pulp and paper mill. By far the largest historic source of mercury emissions was the HoltraChem mercury cell chlor-alkali operation located in Riegelwood, NC, approximately 25 kilometers east-northeast of Lake Waccamaw, which ceased operation in the last decade.

- *Brier Creek, Georgia:* Brier Creek is a coastal plain river, dominated by a watershed that contains several different types of land uses and is modeled using 11 different sub-watersheds located in central/eastern portion of Georgia. Unlike the lakes modeled in this report, the watershed response drives mercury dynamics in Brier Creek. Brier Creek is a popular destination for recreational fishing. Fish consumption advisories are in place in this region due to high mercury levels.
- *Lake Barco, Florida:* Small, seepage lake in northeast Florida. Lake Barco is located approximately 35 km east of Gainesville, Florida on the Ordway Preserve that is operated by the University of Florida. The Ordway is protected from direct human impacts, although some recreational fishing does take place. Lake Barco has a water surface area of 0.12 km^2 and a negligible catchment area. Hydrology and geochemistry of Lake Barco have been well characterized by past studies (EPRI, 2003; Pollman et al., 1991) and there are several nearby mercury sources. For example, Gainesville Regional Utilities (GRU) operates a medium-size coal-fired power plant approximately 40 km northwest of Lake Barco. Emissions of Hg from the Deerhaven Unit No. 2 facility averaged approximately 30 kg yr^{-1} between 1998 and 2002 (range 13 to 47 kg yr^{-1}).

Table 3- 2. Summary of Ecosystem Characteristics Used To Parameterize Mercury Models

Parameter	Lake Pawtuckaway	Lake Waccamaw	Lake Barco	Eagle Butte	Brier Creek
Watershed Area (m^2)	5.00×10^7	2.17×10^8	0	4.21×10^6	2.19×10^9
Percent Impervious	1%	1%	n/a	40%	2%
Percent Forest	88%	72%	n/a	0%	47%
Percent Riparian	10%	0%	n/a	20%	12%
Percent Upland	1%	27%	n/a	40%	39%
Lake Area (m^2)	3.64×10^7	3.47×10^7	1.18×10^5	1.86×10^5	n/a
Catchment/Lake Ratio	13.7	6.3	0	22.6	n/a
Epilimnion Depth (m)	2.0	2.3	3.7	2.0	0.3 – 2.0
Hypolimnion Depth (m)	3.0	n/a	n/a	n/a	n/a
Hypolimnion Anoxia	Yes	n/a	n/a	n/a	n/a
Hydraulic Residence Time (days)	165	241	n/a	n/a	12
Inflow/Outflow (m^3 yr^{-1})	4.05×10^7	1.20×10^8	0	0	$3.3 – 7.4 \times 10^8$
Water pH	6.45	4.3	4.5	9.0	n/a
Epilimnion DOC (mg L^{-1})	5.5	25.9	0.8	27.0	5.0-8.0
Hypolimnion DOC (mg L^{-1})	5.6	n/a	n/a	n/a	n/a
Trophic Status	Dystrophic	Mesotrophic	Oligotrophic	Eutrophic	n/a

3.4.2 Baseline Atmospheric Deposition at Each Site

For each site, measured empirical data were used to characterize the current level of atmospheric deposition (Table 3-3). Lake Pawtuckaway data were obtained from Kamman and Engstrom (2002). We estimated total deposition at Lake Waccamaw and Lake Barco by assuming dry deposition was approximately 50% of total deposition. Wet deposition at Lake Waccamaw represented averaged cumulative wet deposition at MDN site NC-O8 between 1998 and 2000. Wet deposition of mercury at Lake Barco deposition were from a technical report (EPRI, 2003). Atmospheric deposition at Eagle Butte was calculated as from ongoing empirical measurements and data for Brier Creek were from a number of MDN sampling sites. In Pawtuckaway Lake, Lake Waccamaw, Lake Barco and Eagle Butte, MeHg was assumed to represent approximately 3% of total deposition (Fitzgerald et al., 1994; Iverfeldt, 1991), which falls within the range of measured values.

Table 3- 3. Baseline Atmospheric Deposition For Each Model Ecosystem

	Lake Pawtuckaway	Lake Waccamaw	Lake Barco	Eagle Butte	Brier Creek
Annual Precipitation (cm yr^{-1})	102.0	120.4	134.8	43.0	120 .0
HgT Precipitation (ng L^{-1})	10.0	12.0	11.5	21.9	12.2
Wet Deposition (ng L^{-1})	10.2	14.4	15.5	9.4	14.7
Dry Deposition ($\mu g\ m^2\ yr^{-1}$)	10.2	14.4	15.5	10	12.1
Wet Deposition (MeHg) $\mu g\ m^2\ yr^{-1}$	0.15	0.22	0.23	0.20	n/a
Dry Deposition (MeHg) $\mu g\ m^2\ yr^{-1}$	0.15	0.22	0.23	0.15	n/a
Total Deposition (HgT) $\mu g\ m^2\ yr^{-1}$	20.7	29.2	31.5	19.8	26.8

3.4.3 Atmospheric Loading Scenarios Investigated

For each site, we compared deposition rates in the 36x36 km grid cell corresponding to the ecosystem locations forecasted using the CMAQ and REMSAD atmospheric fate and transport models (Table 3-4) to the empirically derived loading rates presented in Table 3-4. Overall, the site specific data suggest somewhat higher deposition rates than the CMAQ and REMSAD models at the locations chosen for these case studies. This reinforces the need for additional data sets that can be used to test model-forecasted atmospheric mercury deposition rates. In Table 3-4, 2001 base case deposition rates are compared to two atmospheric deposition scenarios. The 2020 projection describes anticipated loading rates at the end of the proposed rule and the zero out scenario was calculated by removing coal fired utilities from the atmospheric models as sources of mercury emissions.

Projected differences in atmospheric deposition (Table 3-4) with the zero-out scenario are similar to 2020 projected deposition rate under the Clean Air Interstate Rule (CAIR) presumably because the 2020 scenario includes reductions in emissions from mercury sources other than coal-fired utilities. We therefore modeled only the change in fish tissue concentrations associated with the zero out deposition scenarios. To do this, we calculated a maximum percent difference between 2001 base case deposition the zero out scenarios for the REMSAD and CMAQ numbers, and investigated the anticipated response of fish mercury concentrations to these changes. Changes in atmospheric loading under these scenarios across all ecosystems range from four to fifteen percent. These values are relatively small in magnitude compared to maximum values observed across the country.

Accordingly, we also considered the range in deposition changes projected to occur across the country. The highest forecasted relative change in deposition across all the grid cells covering the United States in the CMAQ and REMSAD atmospheric models after removal of power plants as a source of mercury was approximately 70% (50 µg m^{-2} yr^{-1} to 15 µg m^{-2} yr^{-1}). In addition to the zero-out scenarios run using the SERAFM model, we modeled a standard 50% decline in atmospheric loading for each case study to compare the expected response across sites.

Table 3- 4. Forecasted Atmospheric Deposition Rates in Case Study Areas Using the CMAQ and REMSAD Models

Ecosystem	CMAQ Deposition Projections (µg m^{-2} yr^{-1})			REMSAD Deposition Projections (µg m^{-2} yr^{-1})		
	Baseline (2001)	2020 Projection	Zero Out Utilities	Baseline (2001)	2020 Projection	Zero Out Utilities
Brier Creek, GA	14.2	12.6	12.7	13.2	11.8	12.0
Eagle Butte, SD	8.6	8.2	8.4	6.7	6.5	6.5
Lake Barco, FL	15.6	14.3	14.8	16.7	14.9	15.5
Pawtuckaway Lake, NH	16.1	15.0	15.2	10.3	8.9	9.4
Lake Waccamaw, NC	16.3	14.2	14.1	19.0	16.2	16.2

Note: The gradients of mercury deposition around model ecosystem watersheds are fairly low, even with a known EGU source nearby with the exception of Brier Creek. Two CMAQ grid cells that overlap the Brier Creek watershed are twice the magnitude of the average grid value of 16 µg m^{-2} yr^{-1}.

3.4.4 Summary of Model Evaluation

Models applied to each case study system are listed in Table 3-5. Irrespective of the quality of their process algorithms, none of the models can be considered *a priori* predictive tools. It is understood that all environmental models are under-determined, so that different combinations of model parameters can cause simulated concentrations to match the limited set of observations. It is also understood that observed datasets are always incomplete and uncertain and represent only a snapshot of the real system. Thus, model calibration involves professional judgment.

Table 3- 5. List of Model Frameworks Applied to Ecosystems

Ecosystem	Model Frameworks Applied
Brier Creek, GA	WASP, WCS
Eagle Butte, SD	SERAFM, WASP, BASS
Lake Barco, FL	SERAFM
Pawtuckaway Lake, NH	SERAFM, WASP
Lake Waccamaw, NC	SERAFM, WASP

To address model identification uncertainty, we conducted multiple calibrations of different model frameworks. Results of these model calibrations are described below. We applied the SERAFM model to all systems but Brier Creek. Because mercury dynamics in Brier Creek are dominated by a watershed that consists of 11 tributaries, a surface water body model like SERAFM is not suitable for application. At three sites (Eagle Butte, Lake Waccamaw and Pawtuckaway Lake) both the WASP and SERAFM models were applied as an internal quality assurance check.

All models were calibrated using the available empirical data and run to steady state before investigating any atmospheric loading scenarios. In some cases it was necessary to supplement the observed site-specific data with parameter values and ranges from the general scientific literature. The most reasonable set of parameters that best fit the observed data at each site were used as the base case. Details of these calibrations are contained in the Attachments 1-5 that describe each of the site-specific applications.

Although there was insufficient time to do a formal sensitivity analysis of these models, we investigated the effects of parameter values (summarized in Table 3-6) on ecosystem response times using several different model calibrations. For each ecosystem, we modeled the fast, medium and slow response scenarios by varying the depth of the active sediment layer and the macro-dispersion coefficient for the sediments (i.e., exchange rate between surface sediment and water column. We chose the same upper sediment depth modeled in the U.S. EPA *Mercury Study Report to Congress* (US EPA 1997), where a default value of 2 cm and a uniform distribution of one to three centimeters were justified (see the technical discussion in Volume III, Appendix B.2.27, p. B-50). The sediment-water dispersion coefficient is known to be greater than molecular diffusion coefficient, with a commonly accepted value of around 10^{-6} to10^{-5} cm^2 sec^{-1} (Bowie et al., 1985). Dispersion coefficients for sediments subject to bioturbation (disturbance by benthic organisms) are commonly accepted to fall between 10^{-5} to 10^{-4} cm^2sec^{-1} (Schnoor, 1987). We used these alternate calibrations in the scenario projection phase of this project to provide a semi-quantitative uncertainty envelope for the temporal responses of the various ecosystems. In addition, we investigated the uncertainty in watershed loading response times by conducting sensitivity analyses with the WCS model of the Brier Creek watershed. Given more time, a formal error propagation analysis would be a valuable addition to this study.

Table 3- 6. Summary of Mercury Parameters Used in the SERAFM Model

	H [atm-m3/mol]	$K_{d,abio}$ [L/kg]	$K_{d,bio}$ [L/kg]	$K_{d,DOC}$ [L/kg]
Hg0	$7.1x10^{-3}$	0	0	0
HgII	$7.10x10^{-10}$	250,000	399,052	251,188.6
MeHg	$4.7x10^{-7}$	100,000	516,313	100,000

Parameter	**Range**
Active Sediment Layer Thickness	1 – 3 cm
Macro-Dispersion in Sediments	$5x10^{-5} – 10^{-4}$ $cm^2 s^{-1}$

Runoff Coefficients: Ratio of Watershed Export to Deposition Loading			
	Hg0	**HgII**	**MeHg**
Impervious	1	1	1
Wetland/Forest	0.2	0.2	4.9
Riparian	0.2	0.2	2.0
Upland	0.2	0.2	0.2

H=Henry's Law Constant (used to describe portioning of mercury between air and water), K_d=partition coefficient between solids and water for abiotic solids (abio), biotic solids (bio) and dissolved organic carbon (DOC)

3.4.5 Baseline Fish Mercury Concentrations

To maintain consistency with the original IEM-2M model, the SERAFM model forecasts fish mercury concentrations as a function of an empirically derived bioaccumulation factor (BAF). This BAF was based on site specific fish mercury concentration data and MeHg concentrations in the water column. Site specific fish mercury concentration data were also used as the baseline value used in the MMaps model to forecast fish mercury levels with changing atmospheric deposition scenarios that are described in the next section.

Empirically derived BAFs for these systems fall within the range of those measured at other sites (Table 3-7). Bioaccumulation factors were calculated for all SERAFM applications by normalizing residues to the length of the dominant species of Level 4 (top) piscivoire for which age data were available (e.g., 2 yr old largemouth bass).

Table 3- 7. Empirically Derived BAFs for Each of the Ecosystem Case Studies

BAFs (L/kg)	MeHg (ng/L)	HgT (ng/L)	BAF-MeHg	BAF - HgT
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

We chose to base fish mercury responses in the SERAFM model on empirically derived BAFs to be consistent with the original IEM model, which was the basis for the MMaps derivation. However, there are some limitations to the BAF approach that deserve mention. Because MeHg concentrations in the water column are highly variable, the derived BAF is inherently underdetermined. One less variable approach might be to normalize between model sites using the fraction of organic carbon in sediments as the fraction of MeHg available for uptake by organisms appears to be directly related to the organic carbon content of the sediments (Lawrence and Mason, 2001; Lawrence et al., 1999; Mason and Lawrence, 1999). More important though, response times (i.e., rates to and from equilibrium) in aquatic communities to changes in loading rates are driven by food web dynamics and species metabolic rates responding to changes in concentrations of MeHg in the water column and sediments.

To cross check empirically calibrated BAFs and to improve the estimated response times of mercury residues in different species and age/size classes of fish, the BASS bioenergetics based trophic dynamics model was also applied to the Eagle Butte site (Figure 3-1).

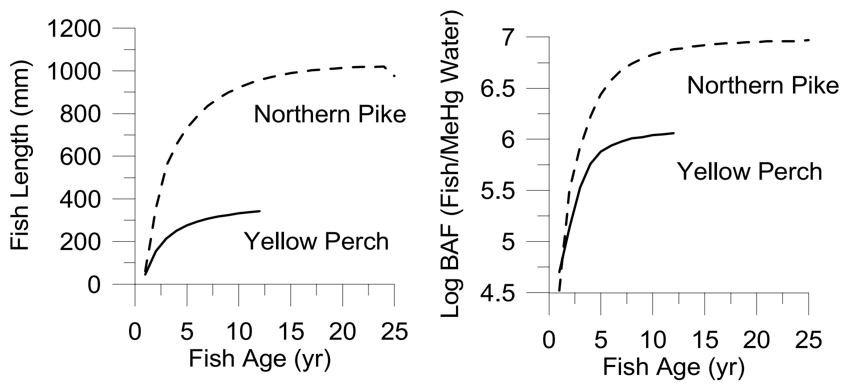
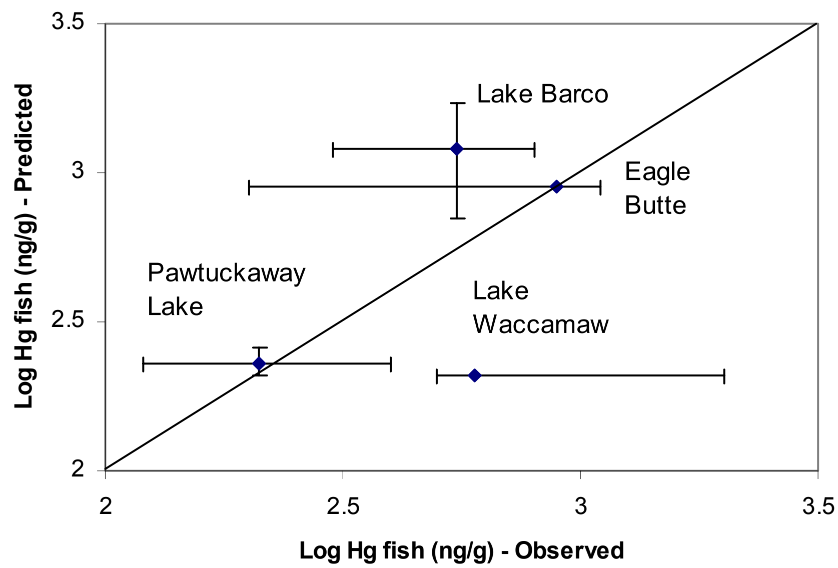


Figure 3- 1. BASS Predicted BAFs for Pike/Perch in Lee Dam, Eagle Butte

Annual size classes for fishes between ages 1 and 25 and ages 1 and 12 were simulated for northern pike and yellow perch in the BASS model, respectively. Results show that BAFs predicted by the BASS model are in general agreement with the empirically derived BAFs for Lee Dam, Eagle Butte (note that the SERAFM BAF was developed for four year-old northern pike). The $\log_{10}$ length normalized BAF used in SERAFM (i.e., $\log_{10}(8.90 \times 10^5) = 5.96$) differs slightly from the BASS prediction of 6.22 (based on dissolved water column MeHg). These BAFs would be equivalent when half of the water column MeHg is dissolved, as specified in the BASS model.

For the SERAFM model applications to each ecosystem, initial concentrations of mercury in water, sediment and fish are run to steady state before any atmospheric deposition scenario projections are investigated. Concentrations at steady state in each compartment vary depending on the parameterization of the SERAFM model (e.g., fast, medium, and slow parameter values described above).

Model forecasted fish mercury concentrations are compared to observed fish concentrations in Figure 3-2.



Straight line represents 1:1 relationship between observed and modeled results.

Figure 3- 2. Observed vs. Predicted Fish Mercury Concentrations in Model Ecosystems at Steady State with No Change in Atmospheric Loading

At steady state, SERAFM overpredicts fish mercury concentrations in Lake Barco and underpredicts concentrations in Lake Waccamaw. Underprediction of observed fish mercury levels in Lake Waccamaw is likely a function of the recent closure of the chlor-alkali facility in proximity to the lake. When the SERAFM model is run to steady state under the current loading scenario, expected fish mercury concentrations are somewhat lower than observed values. It is therefore likely that fish mercury concentrations in Lake Waccamaw are not presently at steady state and will continue to decline if the current level of atmospheric deposition remains constant. We do not have enough information the recent history of Lake Barco to speculate as to why we observe the difference between model-predicted and observed fish concentrations. Given additional time, a more comprehensive sensitivity analysis of factors affecting these results would be a useful starting point. On a purely speculative level, it is possible that fish mercury concentrations in Lake Barco are also not at steady state with respect to current deposition levels. If mercury loading to this ecosystem has increased recently but has not yet been reflected in observed fish tissue residues, than fish Hg levels would be overpredicted by the SERAFM model.

Analysis of the degree of corroboration between observed and model predicted fish mercury concentrations is a useful starting point for discussion of the MMaps model. Like SERAFM, the MMaps model uses a proportional relationship between declines in deposition forecasted by atmospheric fate and transport models and concentrations in fish. Unlike SERAFM, MMaps assumes the empirical data reflect steady state concentrations in the fish, thereby over and underestimating mercury concentrations in fish in ecosystems that have experienced recent changes in mercury inputs.

Initial mercury concentrations in fish used in the MMaps model are therefore critical for model outputs. As illustrated in Figure 3-2, the error bars around one species of piscivorous fish from a single water body are large. National fish tissue data coverages for even a five year period used by the MMaps approach are limited and span different trophic levels and ages of species. In addition to uncertainty in atmospheric models used to forecast changes in mercury deposition described above, uncertainty in MMaps forecasts as the result of variability in fish mercury residues will be significant and is not likely to capture the true variability in mercury concentrations among ecosystems in the United States.

3.4.6 Magnitude of Changes in Fish Tissue Residues

For all ecosystems but Brier Creek, we ran scenario projections for fish mercury concentrations after a 50% reduction in atmospheric deposition. Examples of SERAFM forecasted atte ake Barco are shown in Figu sediments mainly to inve y deposition, and the effects

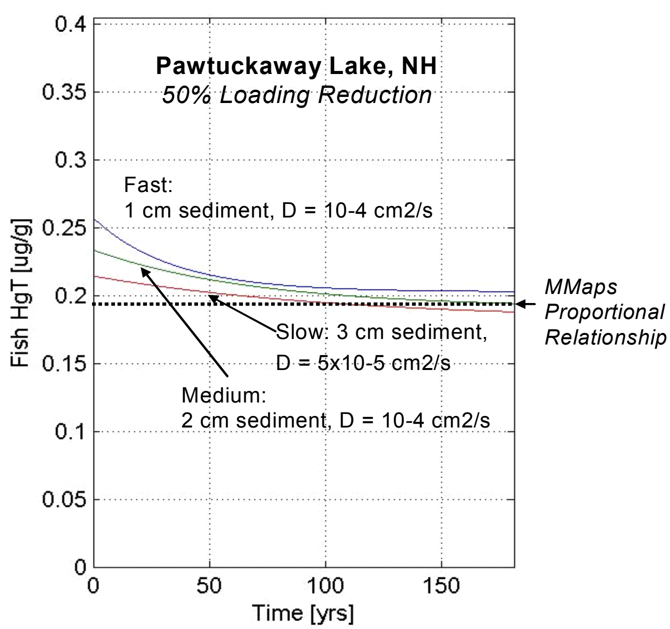


Figure 3- 3. T wtuckaway Lake, NH to a

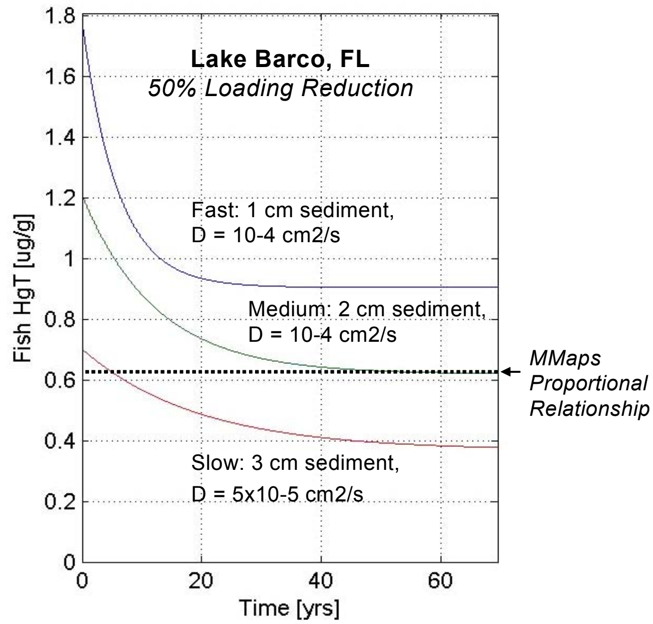


Figure 3- 4. T ke Barco, FL to a Decline in

3.4.6.1 Zero-O

For Paw ve simulated fish mercury concentrations using the forecasted decline in deposition in each ecosystem associated with the removal of U.S. utilities as a component of atmospheric deposition (Table 3-8).

Table 3- 8. MMaps and SERAFM Forecasted Fish Mercury Concentration at Steady State after Removal of Coal Fired Utilities as a Component of Deposition Using the REMSAD and CMAQ Models (Zero-out Scenario)

Ecosystem	% Difference in Atm. Hg Deposition Zero-Out Scenario	MMaps Forecasted Fish Hg Zero-Out Scenario (μg g^{-1})	SERAFM Forecasted Fish Hg Zero-Out Scenario (μg g^{-1})
Eagle Butte	4.3%	0.85	0.86
Lake Barco	6.8%	0.51	1.12 (0.65-1.58)
Pawtuckaway Lake	8.5%	0.19	0.21 (0.19-0.24)
Lake Waccamaw	14.8%	0.51	0.18

Fish mercury concentrations under the “zero-out” scenario for both MMaps and SERAFM are shown in the columns following the percent differences in deposition at each site calculated from the REMSAD and CMAQ models (Table 3-4). Although both models use the proportional relationship described above, comparing the magnitude of forecasted fish mercury levels illustrates some of the uncertainty in model-predicted values. Overall, it is clear that the magnitude of the uncertainty in the atmospheric and ecosystem models at this time is much greater than the signal derived from a change in loading following removal of the coal fired utilities at these sites. This may not be the case for highly impacted systems described above, where atmospheric deposition declines forecasted by the REMSAD and CMAQ models exceed 70% at some sites. Attenuation curves for fish mercury using the 50% decline in loading scenario may therefore better represent the response of mercury in fishes in similar types of ecosystems that are highly affected by mercury deposition from coal fired utilities (Figures 3-3, 3-4).

The major differences between the SERAFM model and the MMaps model across all sites are a function of the initial fish mercury concentrations at steady state that drive the response in both systems. Although both models assume fish mercury concentrations are at steady state with respect to baseline atmospheric deposition levels used to calculate the percent change in deposition (e.g., the same percent change is used in both models but overall loading rates and initital fish concentrations are different), only SERAFM calculates an expected concentration of mercury in fish at steady state for each ecosystem. Relative confidence in the two models is equivocal at this time, although the uncertainties in both approaches are highlighted by the observed differences in results.

3.4.7 Summary of Observed Temporal Responses to Declines in Loading

Table 3-9 compares the 50% decline scenario projections using the SERAFM and WASP models. The Brier Creek scenario incorporates response times in the watershed soils as well as in the water body sediments. The four lake scenarios account for the dynamic internal sediment response to instantaneous declines in total mercury loading. Accounting for watershed response times would add to the overall lake response time estimates.

Table 3- 9. Sediment Response Times in Years to Reach 90% of Steady-state Concentrations Following 50% Mercury Deposition Reductions

Site	Fast SERAFM	Medium SERAFM	Slow SERAFM	Fast WASP	Medium WASP	Slow WASP
Brier Creek – Upstream	n/a	n/a	n/a	40	74	113
Brier Creek – Downstream	n/a	n/a	n/a	58	89	132
Eagle Butte	3	4	6	6	11	16
Lake Barco	14	28	45	n/a	n/a	n/a
Pawtuckaway Lake	80	125	>180	24	44	69
Lake Waccamaw	3	6	12	10	15	17

Note: Fast = 1 cm active sediment layer, D (macro-dispersion coefficient) = 10^{-4} cm^2s^{-1}; Medium = 2 cm active sediment layer, D = 10^{-4} cm^2s^{-1}, Slow = 3 cm sediment, D = $5x10^{-5}$ cm^2s^{-1}.

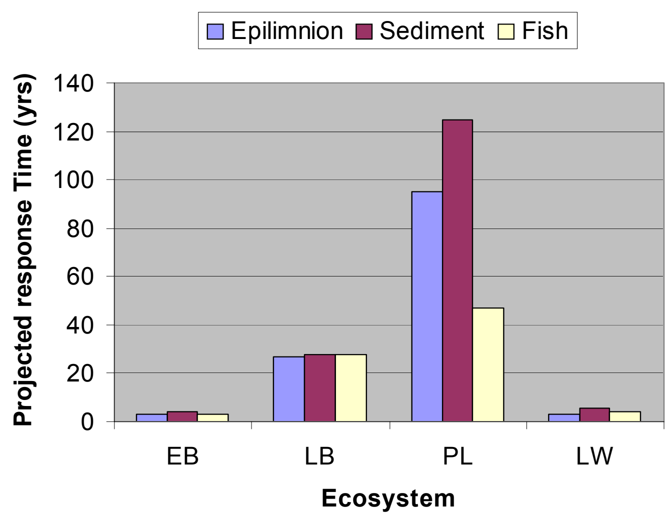
Overall, forecasted time lags were comparable between models, increasing confidence in the range of lag time obtained from the modeling results. The greatest difference between the two models was observed in Pawtuckaway Lake. Forecasted recovery of Pawtuckaway Lake was 2-3 times faster than SERAFM using the WASP model. This can likely be attributed to differences in the treatment of the sediment solids balance between the two models. We used the same parameters in SERAFM as the MCM model applied to Lake Barco by EPRI (EPRI, 2003). The EPRI study forecasts a slight longer temporal response (54-160 years) to achieve 90% of steady state concentrations in fish, compared to SERAFM estimates of 14-43 years for fish and 14-48 years for sediments.

The Brier Creek-WASP/WCS model was only used to forecast a response times in water and sediments with a 50% reduction in atmospheric deposition. Epilimnion mercury response times for Brier Creek under the fast, medium and slow scenarios were 8, 11, and 11 years respectively. These response times should be comparable to the response of fish in this system, depending on the predominance of benthic or pelagic food webs in this system.

The response times forecasted by the SERAFM model for fish mercury concentrations to reach 90% of steady state with a 50% decline in atmospheric deposition and the zero-out scenario are presented in Table 3-10 and Figure 3-5.

Table 3- 10. Fish Tissue Response Times in Years to Reach 90% of Steady-state Concentrations Following 50% Mercury Deposition Reductions

Site	Fast	Medium	Slow
Eagle Butte	2	3	4
Lake Barco	14	28	43
Pawtuckaway Lake	34	56	64
Lake Waccamaw	1	1	2



EB = Eagle Butte, LB = Lake Barco, PL = Pawtuckaway Lake, LW = Lake Waccamaw.

Figure 3- 5. Projected Ecosystem Response Times to Zero-out Deposition Scenario Using the SERAFM Model

Results presented in Figure 3-5 show SERAFM forecasted response times in the water, sediments and fish. One artifact of the formulation of the SERAFM model is that fish tissue residues decline faster than total mercury concentrations in the water and sediments in Figure 4-5. Fish mercury residues are a function of a BAF that is linked to the MeHg concentrations in the water column in the SERAFM model. Because MeHg concentrations in water respond more quickly than the total Hg pools in the water and sediments and the BAF multiplier is applied directly to the water column MeHg concentration, the model shows an instantaneous response in fish mercury.

A more realistic lag time will be obtained from a bioaccumulation model that takes into account trophic interactions and bioenergetics drive mercury dynamics. For example, the BASS model forecasts the comparable response for the same age/length class used to develop a BAF calibration in SERAFM (average concentration trophic level four species) to be roughly twice the water column response time (See Figure 3-1). Growth dilution and mortality affect the response of all fish cohorts to decreased atmospheric loading. The oldest, most heavily contaminated fish (also captured in the BASS model) take a longer period to show the same decline in these cohorts (i.e., length/size class). For example, the largest size class for Northern Pike at Eagle Butte indicated that it would take about 10 years for the most heavily contaminated fish to respond to a decline in loadings. Overall, the response time of mercury in fish for all sites is expected to be roughly twice that of the water column response. Simple averaging of residues for all fish of a species using the BAF approach obscures the fact that cohorts with lower body burdens respond quicker and the most long-lived cohort responds over a much longer period of time.

3.4.8 Effect of Land Uses Changes

As mentioned above, we investigated the uncertainty in watershed loading response times by conducting sensitivity analyses with the WCS model of the Brier Creek watershed. The depth of soil incorporation significantly influences soil response time. The default of 1 cm was varied plus and minus 50% to get a range of response times. The loading responses for Upper Brier Creek are given in Figure 3-6. An initial rapid drop-off in loading (due to instantaneous drop in deposition to water surfaces and impervious runoff) is followed by a slower drop-off in runoff and erosion fluxes, controlled by soil mercury concentrations. The 50% loading response varied between 8, 10, and 15 years for incorporation depths of 0.5, 1.0, and 1.5 cm. The 90% loading response times were much longer, varying between 35, 70, and 100 years, respectively.

Land use changes can also significantly affect future loading response from a watershed. Urbanization can increase the total impervious areas in a watershed, decrease the amount of wetlands, and alter the stream's hydrological response. The net effect of these changes on fish mercury concentrations is uncertain. Reducing wetlands and hydraulic residence time should reduce net methylation. On the other hand, covering pervious land with impervious surfaces should increase the delivery of atmospheric deposition fluxes to the water body. In particular, changing pervious land use areas to impervious areas will directly affect the delivery of atmospheric deposition fluxes to the water body.

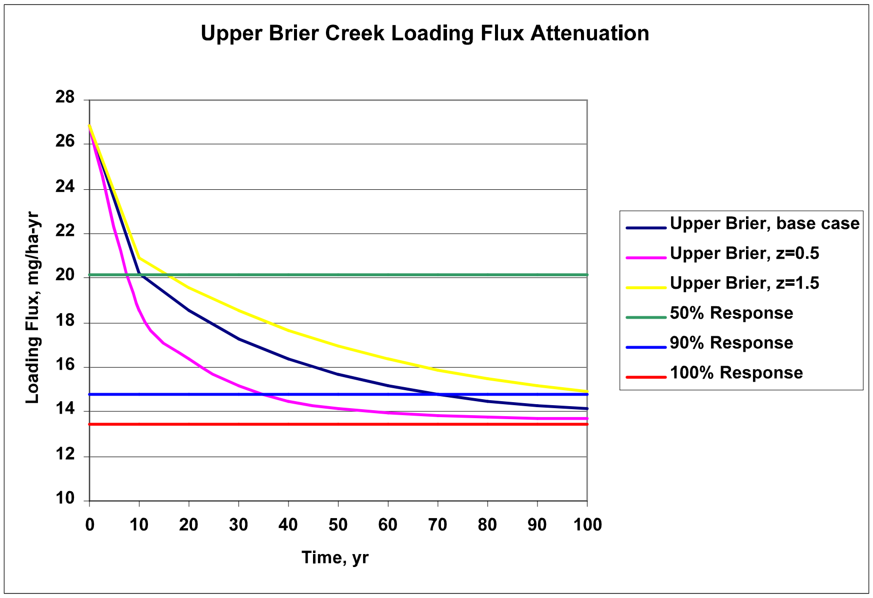


Figure 3- 6. Upper Brier Creek Loading Flux Attenuation

In the WCS model, impervious areas were assumed to deliver 100% of the deposition load, while pervious areas delivered a much smaller fraction through runoff and erosion. By comparison, the SWAT model (Neitsch et al., 2002) assigns a curve number that delivers virtually all rainfall and associated loads from *hydraulically-connected* impervious areas, which average between 73% and 97% of the total impervious area, depending on land use. Although the fraction of the Brier Creek watershed covered by impervious surfaces is small (about 3% of the upper watershed), even modest growth over many years could increase the total watershed delivery of deposited mercury, working against the overall reductions in atmospheric emissions.

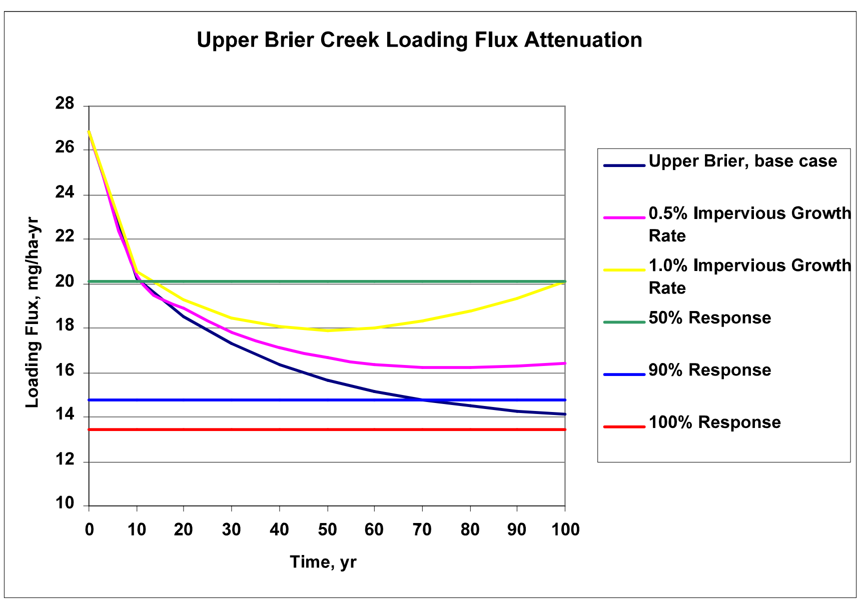


Figure 3- 7. Watershed Loading Flux Attenuation Considering Land-use Change

The loading response of Upper Brier Creek to 3 land use scenarios is shown in the next Figure 3-7. All scenarios assume an immediate 50% cut in atmospheric deposition. The base case assumes present land use patterns. The other two scenarios assume modest impervious surface growth rates of 0.5 % per year and 1 % per year, reaching a total of 4.7% and 7.9% impervious coverage in 100 years. For the 0.5% scenario, total watershed loads reach a minimum in 80 years, with watershed loadings stalling at 40% of present levels. For the 1% scenario, total watershed loads reach a reduction level of 33% in 50 years, and then increase significantly. After 100 years, the 50% cut in atmospheric deposition would translate into a 25% drop in ambient watershed loading (Figure 3-7). These simulations are meant to suggest possible responses to land use changes. If impervious areas deliver only 50% of deposited mercury, then the 1% growth scenario should follow the 0.5% trend line in Figure 3-7.

3.4.9 Summary

Among the five ecosystems investigated in this study, the range in temporal response to declines in mercury loading ranged from less than five years to an order of decades. Response is gauged by the time required for mercury to reach equilibrium at 90% of the initial concentrations. Fast and slow response scenarios for the five ecosystems ranged from 1 to over 180 years in sediments. The medium response scenarios also varied widely but were generally on the order of one to three decades for fish mercury concentrations. Forecasted response times to changes in mercury inputs were longest for Brier Creek, a system strongly influenced by the watershed mercury loading, and Pawtuckaway Lake, a stratified cold water lake. Shallow, well-mixed systems like Lake Waccamaw and Lee Dam are projected to respond to changes in atmospheric deposition in less than a decade.

Consistent with other investigations (e.g., (EPRI, 2003), this analysis showed that individual ecosystems are highly sensitive to uncertainty in model parameters. Long term mercury response is strongly influenced by the sediment balance, which varies greatly among water bodies. Key parameters include watershed erosion and sediment delivery, water body production, deposition, resuspension and burial rates, and surface sediment mixing depth (active sediment layer). The effect of epistemic uncertainty (i.e., lack of knowledge) about key mercury process variables, such as the functional form of equations used to quantify methylation rate constants, is a major contributor to overall uncertainty that cannot be quantified at this time. Although scientific understanding of these process variables is rapidly progressing, no modeling framework can be considered *a priori* predictive at this time. Accordingly, all models must be parameterized to specific ecosystems being investigated to develop a credible set of rate constants and process terms.

Our best available, practicable science suggests that over the long-term (i.e., at steady state), the change in mercury concentrations in freshwater fish will be proportional to changes in mercury inputs. Thus, in systems where atmospheric deposition of inorganic mercury is the major source of mercury to surface waters, long-term changes in fish mercury concentrations will be proportional to declines in atmospheric deposition as suggested by the MMaps approach. There is some discrepancy between results from our modeled scenarios based on the MMaps model that can be attributed to uncertainty in both the **models and data**. Where there are sources of mercury in the watershed other than atmospheric loading, the response of fish tissue concentrations will be affected in a manner that is not proportional to changes in atmospheric deposition. Preliminary findings of the METALLICUS study show negligible concentrations of deposited mercury in fish three years after the addition of labeled mercury isotopes to the watershed, supporting the supposition that mercury deposited on the watershed takes significantly longer to accumulate in fish than mercury deposited directly on the surface of a waterbody (Pers. Comm., R. Harris).

A preliminary assessment of the expected effect of land-use changes on fish mercury concentrations for a watershed dominated system illustrates changes like urbanization within a watershed can alter the magnitude and timing of fish mercury concentrations. A number of peer-reviewed studies have shown that other environmental changes such as enhanced nutrient loading in Long Island Sound (Hammerschmidt et al., 2004; Hammerschmidt and Fitzgerald, 2004), sulfate deposition in the Everglades (Krabbenhoft et al., presentation to EPA 2/7/05) and reservoir flooding can also dramatically change fish mercury concentrations independently of changes in total mercury additions to the ecosystem. With further advances in knowledge and new data on mercury fate and transport, EPA's long term goal is to provide tools that have the capability to provide *a priori* forecasts of ecosystem responses to changes in mercury deposition.

3.5 National Scale Ecosystem Variability

As mentioned in the introductory sections of this report, different ecosystems exhibit dramatically different responses to changes in mercury loading depending on their chemical and physical attributes. Results from individual case studies must therefore be qualified by their representativeness of ecosystem condition and variability across the United States. Using georeferenced empirical databases that describe some of the watershed and waterbody characteristics across the United States, we present the ecosystem case studies along the gradient of data known to be important for MeHg formation and bioaccumulation in fish. Although this analysis has not been completed, the concept is demonstrated by a *preliminary qualitative* assessment of how much of the ecosystem variability in MeHg formation and bioaccumulation has been captured by the modeling case studies. Developing broad categories of ecosystem types based on their propensity for MeHg formation and bioaccumation in fish and their frequency of occurrence is an iterative effort. By combining the frequency of each category of ecosystem type with the magnitude and time lag in fish tissue reductions modeled using dynamic, ecosystem scale models, such an analysis could ultimately provide a different methodology for a national scale assessment of expected changes in fish MeHg concentrations resulting from reductions in atmospheric mercury deposition.

3.5.1 United States Lakes Distribution

- We used the USGS National Hydrography Database (NHD)(1:100,000 scale) to obtain information on the frequency of different lake sizes across the country
- For each Electricity Generating Unit (EGU) we constructed a circular buffer area having a 50 km radius using GIS software (ArcMap, ArcView). This buffer zone was chosen to reflect the geographic extent of the majority of elevated deposition from a local source (Expert Panel on Mercury Atmospheric Processes, 1994). This distance is not meant to conclusively define the extent of local deposition, but is used to approximate near-field characteristics of the areas surrounding EGUs.
- The NHD lake coverage was then clipped using the 50km radii.

- The lake area was divided into three different classes and the total number in each class was calculated across all of the buffered areas (see Table 3-11).

Table 3- 11. Frequency of Different Lake Sizes Across the United States

Lake Type Frequency	All U.S.	50 km Radius Around EGUs
Small l (0.01-0.1 km^2)	328,564	119,492
Medium (0.1 – 10 km^2)	64,260	23,563
Large Large (>10 km^2)	1,200	442

From this analysis, it is apparent that there are a large number of lakes (<0.1 km^2) surrounding the EGUs across the country. Small, well-mixed shallow systems (like Eagle Butte and Lake Barco), tend to respond more rapidly to changes in mercury deposition than larger, deeper lakes. The temporal response of these systems is also driven by the catchment to lake area ratio of these systems, which was beyond the scope of our initial analysis. Generally, systems where mercury dynamics are dominated by the response of the watershed to changes in loading (like Brier Creek), will respond more slowly to changes in atmospheric deposition.

Ecosystems differ dramatically in their propensity to convert incoming inorganic mercury to methylmercury. Other environmental factors that are generally accepted as reasonable ancillary variables for MeHg formation in waterbodies include: dissolved organic carbon content of surface waters, pH, sulfate, % wetland area of the catchment, sulfide concentrations and sediment organic matter. National scale coverages of these data are currently under development by a number of groups. In this assessment, we assembled the available data on sulfate deposition, % wetland coverage, and total mercury deposition projected by the CMAQ model for 2001. Over the long term, by combining these data layers, a statistical model of MeHg production will be developed to characterize ecosystems affected by atmospheric inputs of mercury.

Data layers used to generate Figures 3-8 through 3-13 are described below:

- Model ecosystems relative to EGUs across the country. We used EGRID 2002 Version 2.01 Plant File (Year 2000 data) to generate the data points representing the EGUs. We used lat/long to create point coverage for the U.S.
- WETLANDS: We used the NLCD 1993 land cover data.
- Mean wet sulfate deposition 1987-1999 (kg/ha/yr). Sulfur deposition aggregated by HUC based on data from "Enhanced wet deposition estimates using modeled precipitation inputs", Grimm, Jeffrey, W. , James A. Lynch. 2004. Environmental Monitoring and Assessment Vol. 90 P. 243-268.
- CMAQ deposition 2001 scenario. Source: Russ Bullock, NOAA-ARL and EPA/ORD/NERL
- FISH tissue data for all US. Figure 3-12 is based on raw data from the USGS EMMA database (http://emmma.usgs.gov/fishHgAbout.aspx and http://emmma.usgs.gov/datasets.aspx). Figure 3-13 shows the normalized fish tissue data from NLFA and NLFTS databases used in the Regulatory Impact Assessment.

3.5.2 Summary

Extrapolating the results of the case studies to other systems is an ongoing challenge. Qualitatively, the ecosystem case studies fall in the mid-range of sulfate deposition and do not capture much of the variability across the country. Sites capture a wide range of % wetland coverage and appear to be in regions of predominantly moderate mercury deposition rates, but they do not capture the tails of this distribution. Based on this preliminary assessment it is likely that the case studies represent ecosystem types of moderate methylation potential across the United States. This is supported by fish tissue concentrations measured in these regions, which generally fall within the impacted category that is above EPA's 0.3 ppm tissue residue guideline (US EPA, 2001) but do not represent the extremes in fish mercury concentrations observed on a national scale. Therefore, while these ecosystem case studies cover the bulk of the distributions of the key environmental characteristics that will affect MeHg production, they may miss the tails of the distributions for some characteristics.

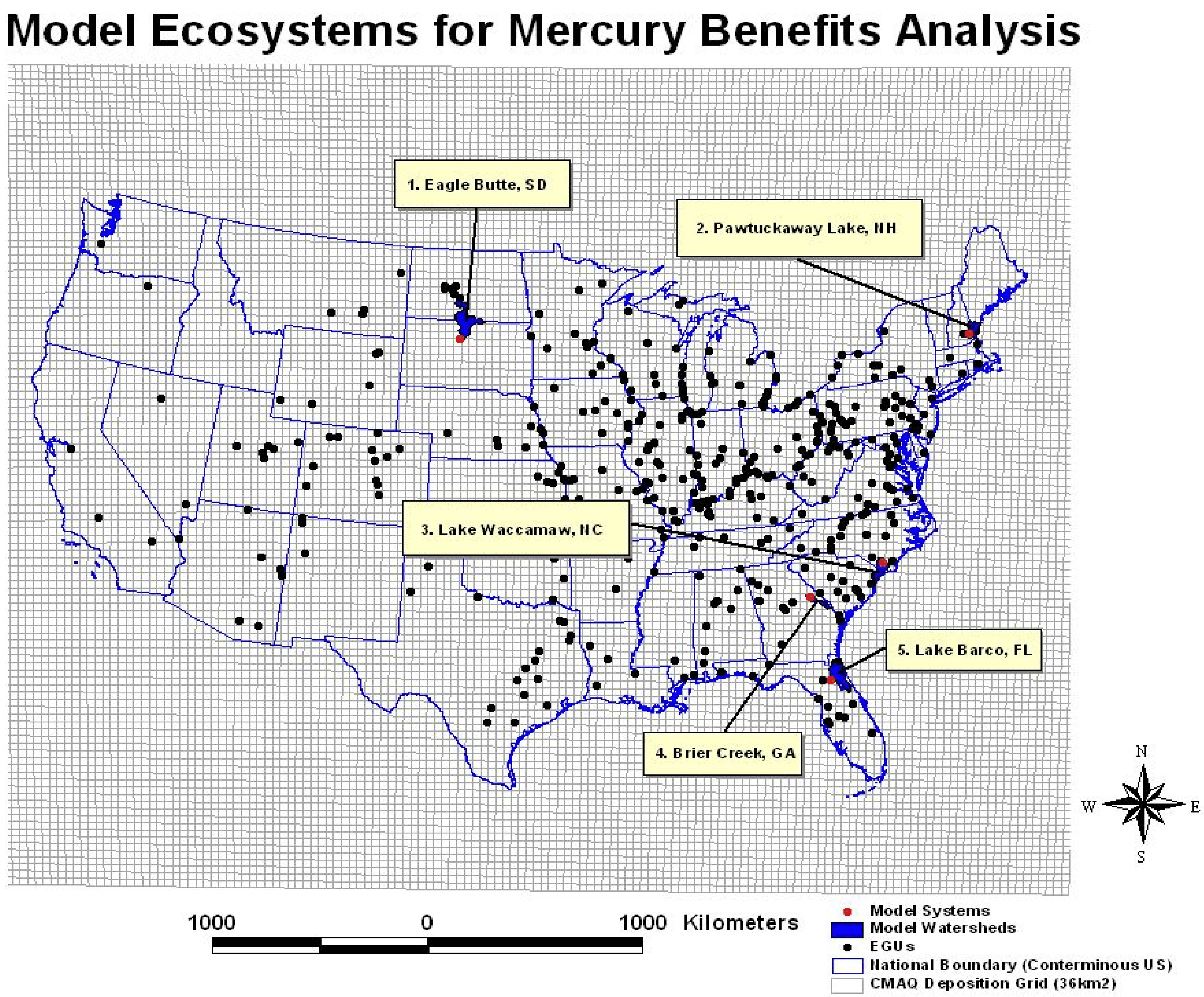


Figure 3- 8. Model Ecosystem Locations with CMAQ Grid Cells and Locations of Electricity Generating Units (EGUs)

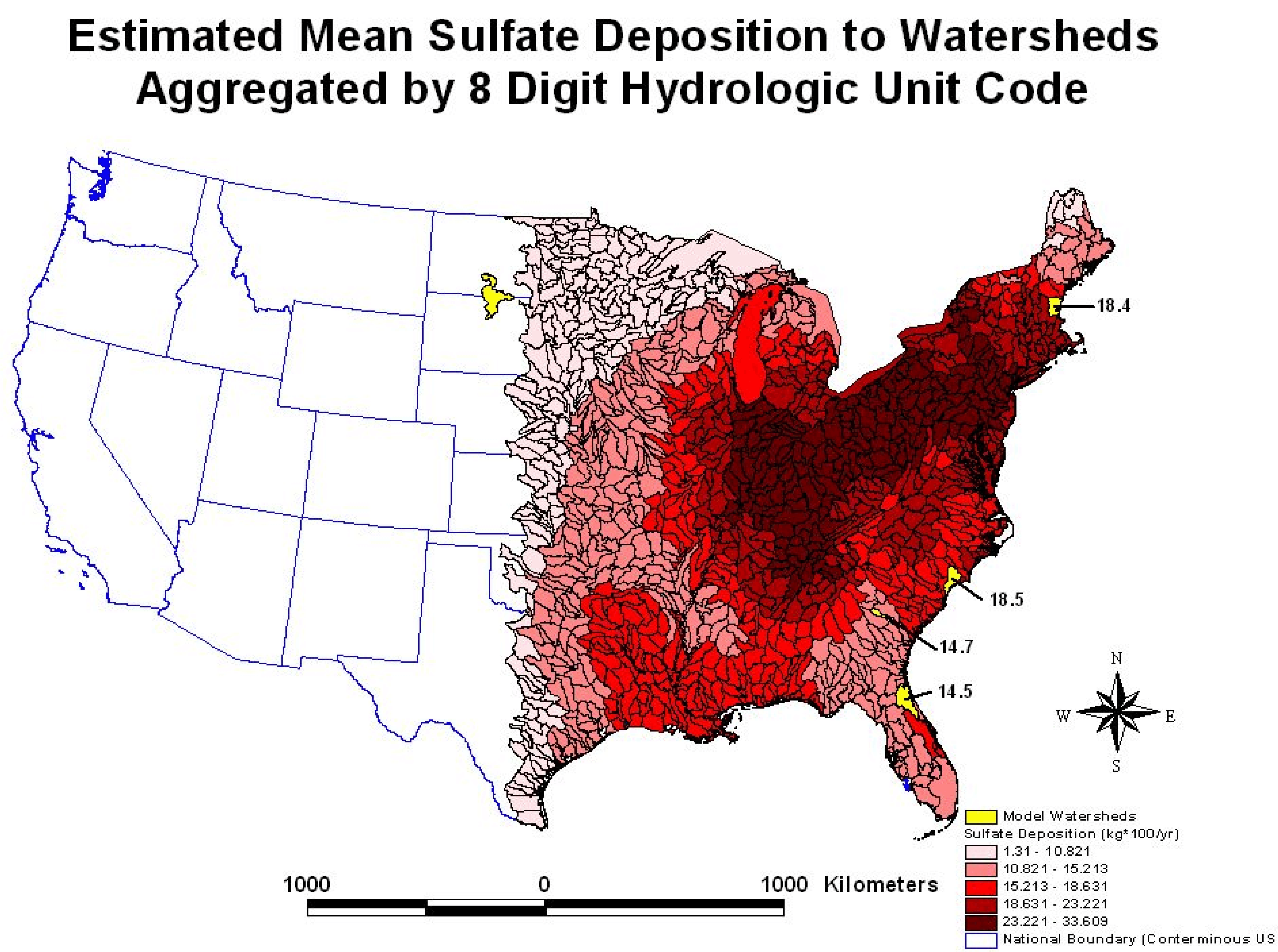


Figure 3- 9. Model Ecosystem Locations with Gradient in Sulfate Deposition Across the Eastern US

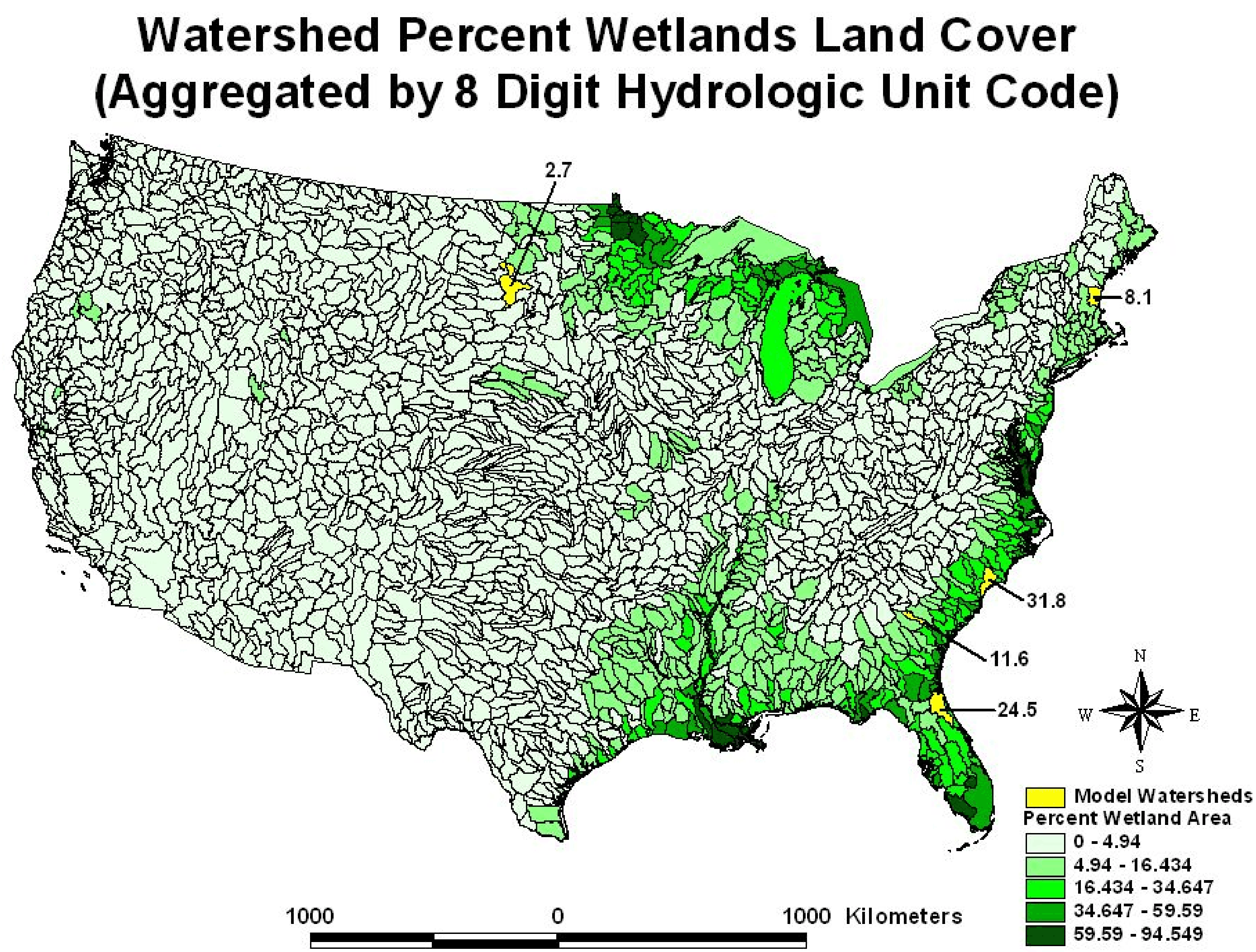


Figure 3- 10. Model Ecosystem Locations with Percent Wetland Area Aggregated for Each HUC

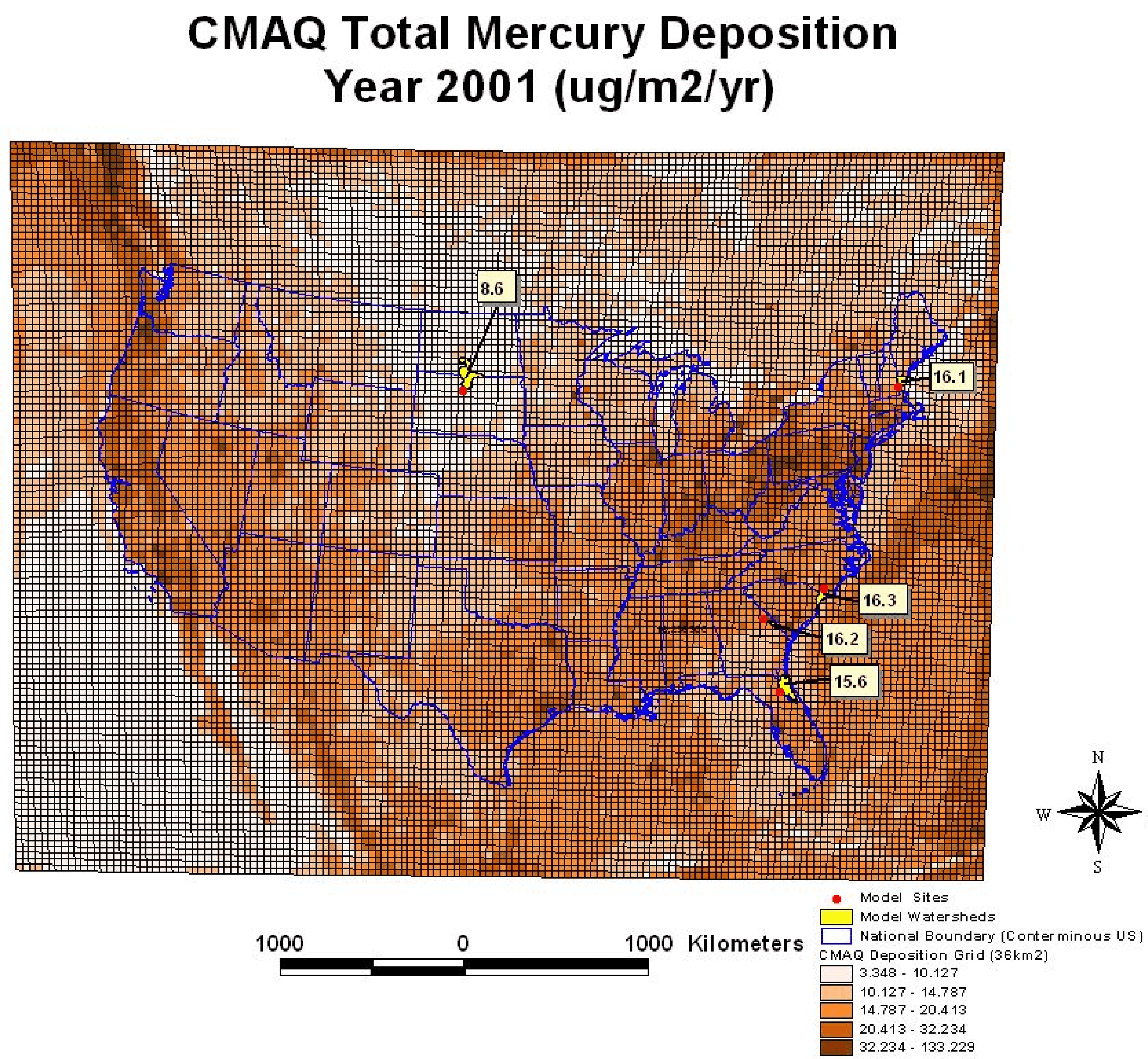


Figure 3- 11. Model Ecosystem Locations with CMAQ 2001 Total Mercury Deposition

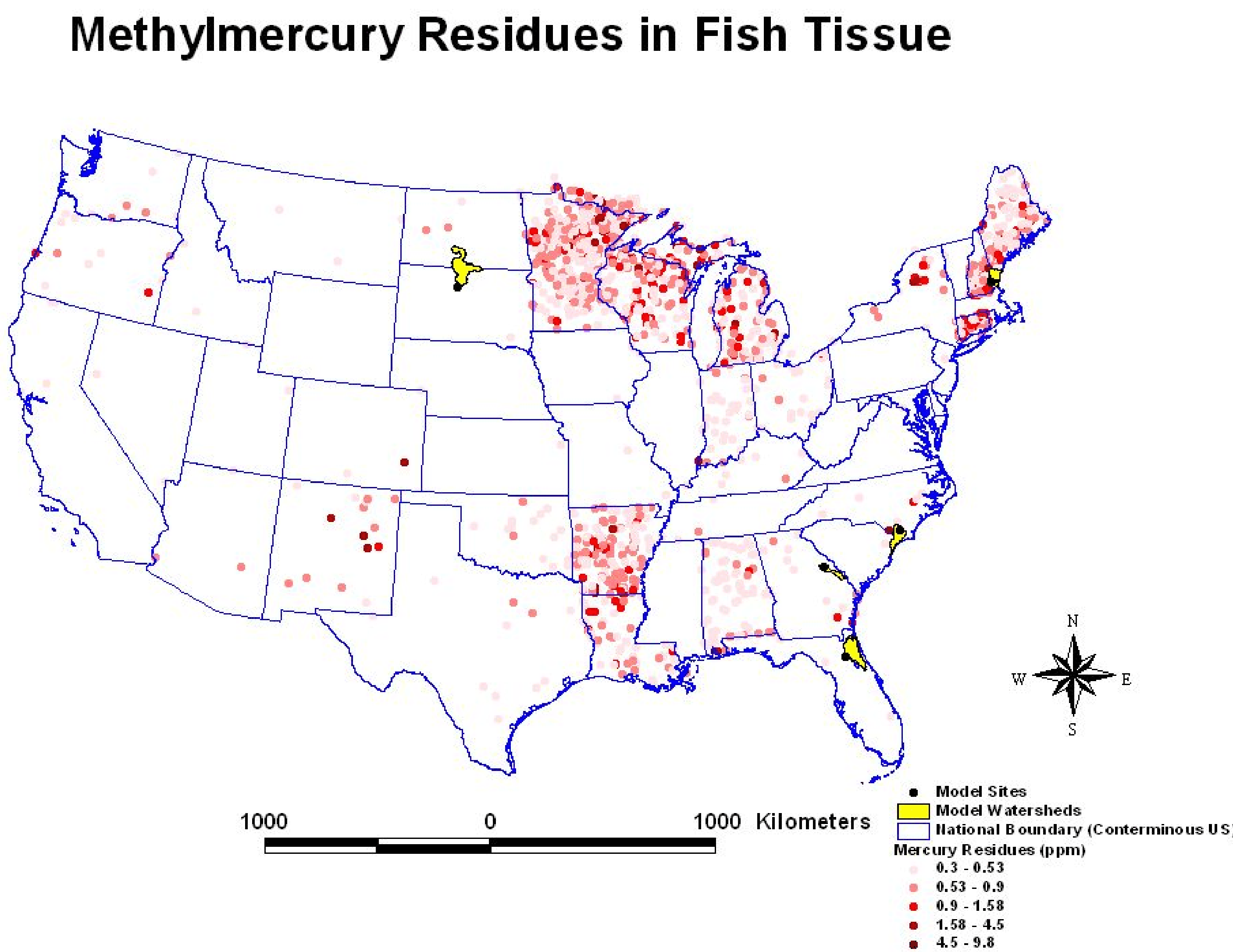
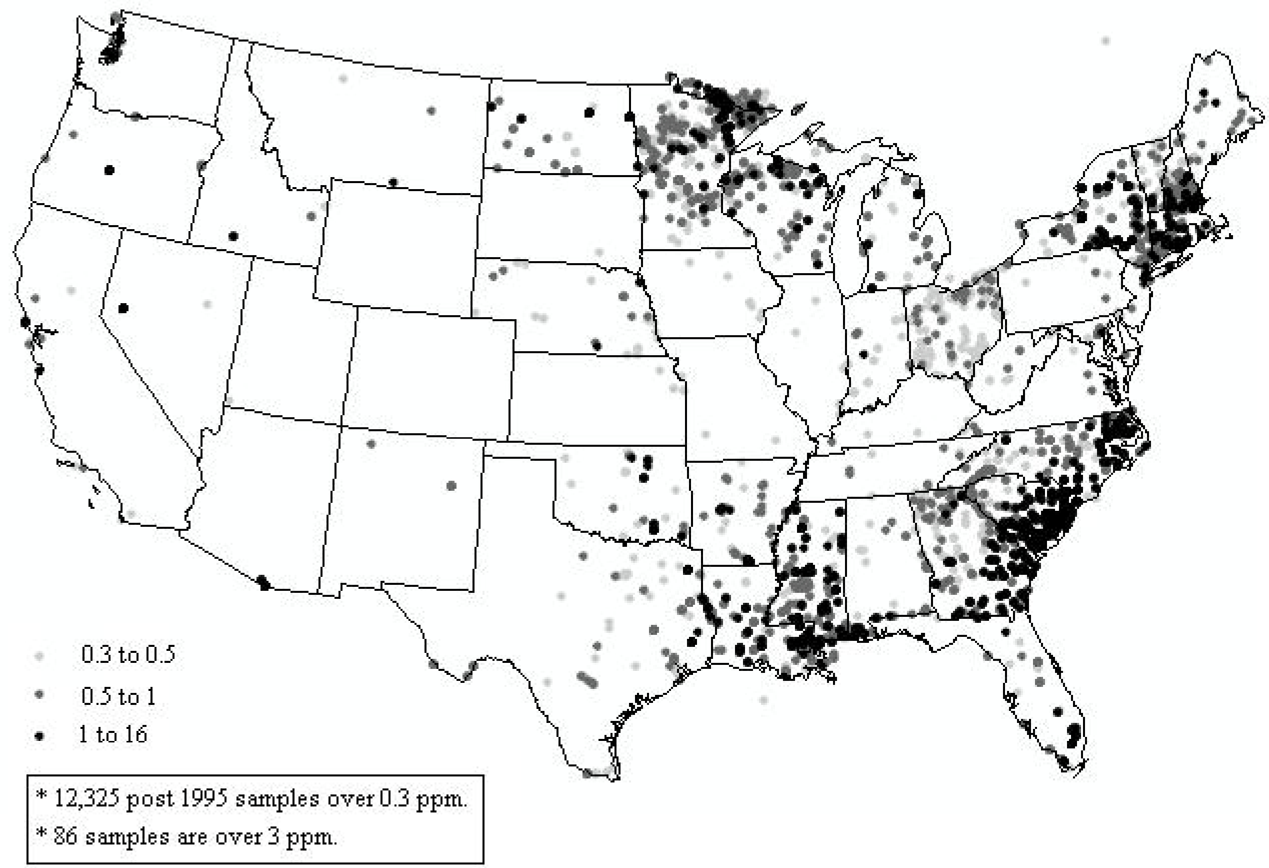


Figure 3- 12. Model Ecosystem Locations with Measured Fish Tissue Concentrations



Source: EPA NLFA and NLFTS

Figure 3- 13. Measured 1995-2001 Fish Hg Concentrations > 0.3 ppm

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Tables

Figures

SECTION 4

PROFILE OF FISHING ACTIVITY IN THE U.S.

4.1 Industry Characterization

Because fish consumption is the primary pathway for exposure to methylmercury, this section provides background information through a profile of the fishing industry in the United States. We provide a characterization of the commercial and recreational fishing industries, followed by information on U.S. commercial and recreational fish production, U.S. commercial and recreational fishing demand, data on the pricing of affected fish products, and an industry outlook. We also provide a characterization of populations that consume fish, and the potential magnitude of quantities consumed in each category of fish consumption pathways. Based on data availability, most of the information in this profile is based on calendar year 2002. However, in some sections, historic and projected industry data are provided.

This section demonstrates that the recreational freshwater fisher population (28 million) is significantly larger than the recreational saltwater population (9 million), while both of these populations are significantly smaller than the general population of fish consumers in the U.S. (184 million) which includes many individuals receiving a significant fraction of their fish from commercially-produced stocks.

Based on information provided in this section, we see that commercial fish consumption constitutes a large portion of exposure to methylmercury. However, a large majority of the commercial fish consumed are imported from foreign sources, or 3-200 miles offshore by domestic commericial fishermen (with a majority of domestic landings occuring off the Pacific coast). These sources of exposure are not likely to be impacted by the control of utilities from the CAMR rule. However, methylmercury concentrations from freshwater sources are likely to be affected by control domestic electric utilities. Therefore, the quantified benefit analysis in Section 11 evaluates the benefits of improved health from reduced exposure to methylmercury from recreational freshwater fishing activities.

4.1.1 Introduction and Overview of the Fishing Industry

The most common mechanism of exposure to mercury for humans and wildlife is through the consumption of mercury containted in predatory fish. These include saltwater fish such as tuna, shark, and swordfish, which are most often caught commercially. They also include freshwater fish such as bass, perch, and walleye, which are often caught recreationally.

The fish that Americans eat come from a variety of sources. Figure 4-1 shows that total fish consumption is composed of commercial fish and shellfish, aquaculture (or farm raised fish for commercial sale), fish caught from recreational activities, fish caught for cultural or traditional practices, and imports from international waters. The types of fish consumed vary greatly and may come from saltwater or freshwater sources. The figure also shows that the benefits analysis focuses on the consumption of recreationally-caught freshwater fish.

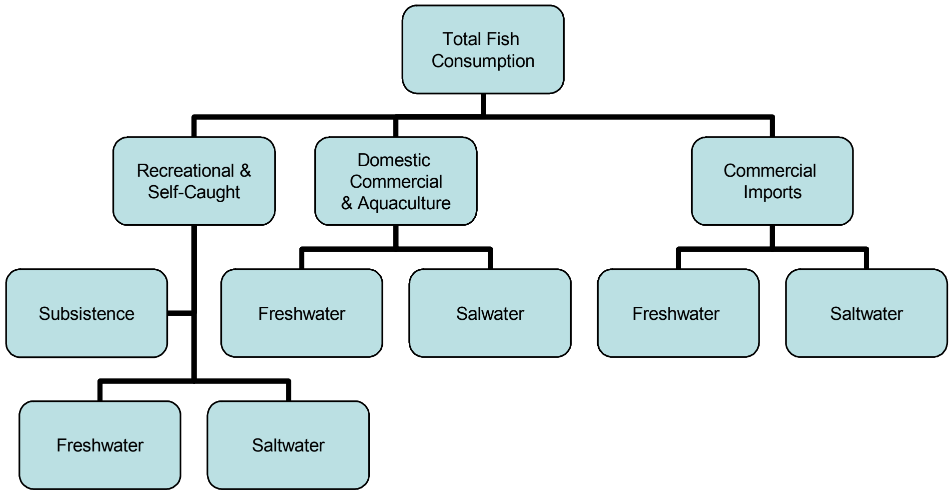


Figure 4-1. Sources of U.S. Fish Consumption

The focus of this profile is a characterization of the commercial fishing industry and recreational fishing activity. However, some information on industries associated with commercial and recreational fishing is also included. The industrial categories with information in this profile are provided below with their North American Industrial Classification System (NAICS) code:

Description	NAICS	Description	NAICS
Finfish Fishing	114111	Seafood Canning	311711
Shellfish Fishing	114112	Fresh and Frozen Seafood Processing	311712
Finfish Farming and Fish Hatcheries	112511	Fish and Seafood Wholesalers	422460
Shellfish Farming	112512	Fish and Seafood Retail Markets	445220
Shipbuilding and Repair	336611	Seafood Restaurants	7221, 7222
Boat Building	336612	Fishing Supply Stores	451110
Boat Dealers	441222	Charter Fishing Boat Services	4872102

The rest of this section provides an industry characterization for the commercial and recreational fishing industries. Section 4.1.2 covers the commercial fishing industry. Section 4.1.3 provides information on the recreational fishing industry.

4.1.2 Commercial Fishing

The commercial fishing industry grossed over $3 billion in revenures in 2003 and is dominated by saltwater species of fish (i.e., tuna, shrimp, clams, shark). Over 60% of domestic commercial saltwater fish are caught between 3 and 200 miles from shore (most of the remaining commercial saltwater catch is made within 3 miles from shore. Approximately, two-thirds of the U.S. commercial landings occur along the Pacific coast. Imports of commercial seafood account for 35% of U.S. demand for commercial fishery products.

4.1.2.1 Industry Characterization

The commercial fishing industry is classified under the standard industrial classification system (SIC) as 0912 for finfish firms and 0913 for shellfish firms, and the NAICS under codes 114111 and 114112, respectively. Although detailed industry gross domestic product (GDP) data were not available at the five digit NAICS code level, the gross industrial output for NAICS code 114100 was $3.19 billion in 2002 (BEA, 2004). Gross industrial output for the years 1998 through 2003 were:

Year	Gross Output (Billion $)
1998	3.28
1999	3.56
2000	3.65
2001	3.39
2002	3.19
2003	3.45

The estimated value of year 2002 U.S. edible and nonedible fish products was $7.67 billion, which was 0.07% of U.S. GDP in 2002. Production and marketing for the commercial fishing industry generated an estimated $28.4 billion in value added to the U.S. Gross National Product. In addition to primary, secondary, and imports processing, value added activities include retail trade from food service and retail trade from stores. Total consumer expenditures for commercial fishing products were estimated to be $55.1 billion in 2002 (NMFS, 2003a).

The commercial fishing industry includes products from finfish fishing, shellfish fishing, and aquaculture. Each sub-industry is discussed further below. As expected, the majority of U.S. commercial fish landings occur off the coast of the United States. Some commercial fishing products are caught within three miles of the U.S. coast, while the majority of commercial products are caught further from shore (i.e., between 3-200 miles off the U.S. coast). In fact, most commercial fishing occurs in saltwater from the Pacific and Atlantic Oceans, and in the Gulf of Mexico, with the largest portion of landings from the Pacific Ocean. Section 4.2 provides more detail on U.S. commercial and recreational fish landings.

4.1.2.1.1 Finfish Fishing

In 2002, there were 1,113 establishments with 4,301 employees for NAICS 114111, finfish fishing. Table 4-1 shows the number of firms, number of establishments, employment and payroll for 1998-2002 from the Bureau of Census (BOC) publication *County Business Patterns* and *Statistics of U.S. Businesses* (BOC, 2004a; 2004b). *Statistics of U.S. Businesses* from the Bureau of Census 2002 have not been released yet, so the number of firms for 2002 is not available (BOC, 2004b). The data demonstrate that the finfish industry is dominated by one-facility firms rather than firms that have multiple facilities nationwide and the average firm employs approximately 3.5 people.

Table 4-1. Finfish Fishing Industry (NAICS code 114111)

Year	Number of Firms	Number of Establishments	Number of Employees	Annual Payroll ($1,000)
1998	1,269	1,275	4,630	181,872
1999	1,307	1,312	5,088	191,521
2000	1,350	1,357	4,833	190,500
2001	1,293	1,302	4,586	183,490
2002	NA	1,113	4,301	190,657

Source: BOC, 2004a; BOC, 2004b.

4.1.2.1.2 Shellfish Fishing

The Census Bureau reports that there were 793 establishments with 2,195 employees in 2002 for NAICS 114112, shellfish fishing. Table 4-2 shows the number of firms, number of establishments, employment and payroll for 1998-2002 from the *County Business Patterns* and *Statistics of U.S. Businesses* from the Bureau of Census (BOC, 2004a, 2004b). Statistics of U.S. Businesses from the Bureau of Census 2002 have not been released yet, so the number of firms for 2002 is not available. Like the finfish industry, the shellfish industry is also dominated by firms with only one establishment with average employment of 3 people per firm.

Table 4-2. Shellfish Fishing Industry (NAICS code 114112)

Year	Number of Firms	Number of Establishments	Number of Employees	Annual Payroll ($1,000)
1998	831	832	2,535	68,723
1999	858	859	(G)	(D)
2000	940	941	2,628	75,414
2001	935	936	(H)	(D)
2002	NA	793	2,195	60,944

(D) -- Withheld to avoid disclosing data for individual companies; data are included in broader industry totals
(A)-(C), (E)-(M) -- Employment-size classes are indicated as follows:
A--0 to 19 B--20 to 99 C--100 to 249 E--250 to 499
F--500 to 999 G--1,000 to 2,499 H--2,500 to 4,999
I--5,000 to 9,999 J--10,000 to 24,999
K--25,000 to 49,999 L--50,000 to 99,999
M--100,000 or more

Source: BOC, 2004a; BOC, 2004b.

4.1.2.1.3 Aquaculture

Although firm-level statistics on aquaculture were not available from the BOC for NAICS codes 112511 (finfish) or 112512 (shellfish), the U.S. Department of Agriculture (USDA), National Agricultural Statistics Service (NASS) reports that in 2002 aquacultural sales totaled $1.13 billion with 6,653 farms (NASS, 2002). Detailed aquacultural data are available from the 1998 Census of Aquaculture (NASS, 1998). According to USDA, there were 4,028 aquacultural farms in 1998 with 3,252 used for freshwater fish and 815 used for saltwater fish farming. Total sales in 1998 were $978 million with food fish accounting for over 70% of sales. These data indicate a large increase in the number of farms from 1998 to 2002, but a relatively small increase in sales.

An estimated 68% of these farms were in the southern states of the U.S. Mississippi was the top State in aquacultural sales, capturing nearly 30 percent of the $978 million dollar total in 1998. Arkansas, Florida, Maine, and Alabama came in second through fifth, respectively, in sales. Food fish (e.g., catfish, trout, salmon, tilapia, hybrid striped bass, etc.) accounted for about two-thirds of the aquacultural sales. Other aquaculture activities include raising fish to stock waterbodies with particular species. Figure 4-2 provides a map of aquacultural farming operations in the U.S. based on total freshwater and saltwater acreage from the U.S. Department of Agriculture's 1998 Census of Aquaculture.

Table 4-3. Value of Aquacultural Products Sold, 1998

Product	Farms	Sales ($1,000)
Catfish	1,370	450,710
Trout	561	72,473
Food fish, other than catfish and trout	435	168,532
Baitfish	275	168,532
Ornamental fish	345	68,982
Sport or game fish	204	7,390
Other fish	11	267
Crustaceans	837	36,318
Mollusks	535	89,128
Other animal aquaculture, algae, and sea vegetables	216	46,734
Total	**4,028**	**978,012**

Source: NASS, 1998.

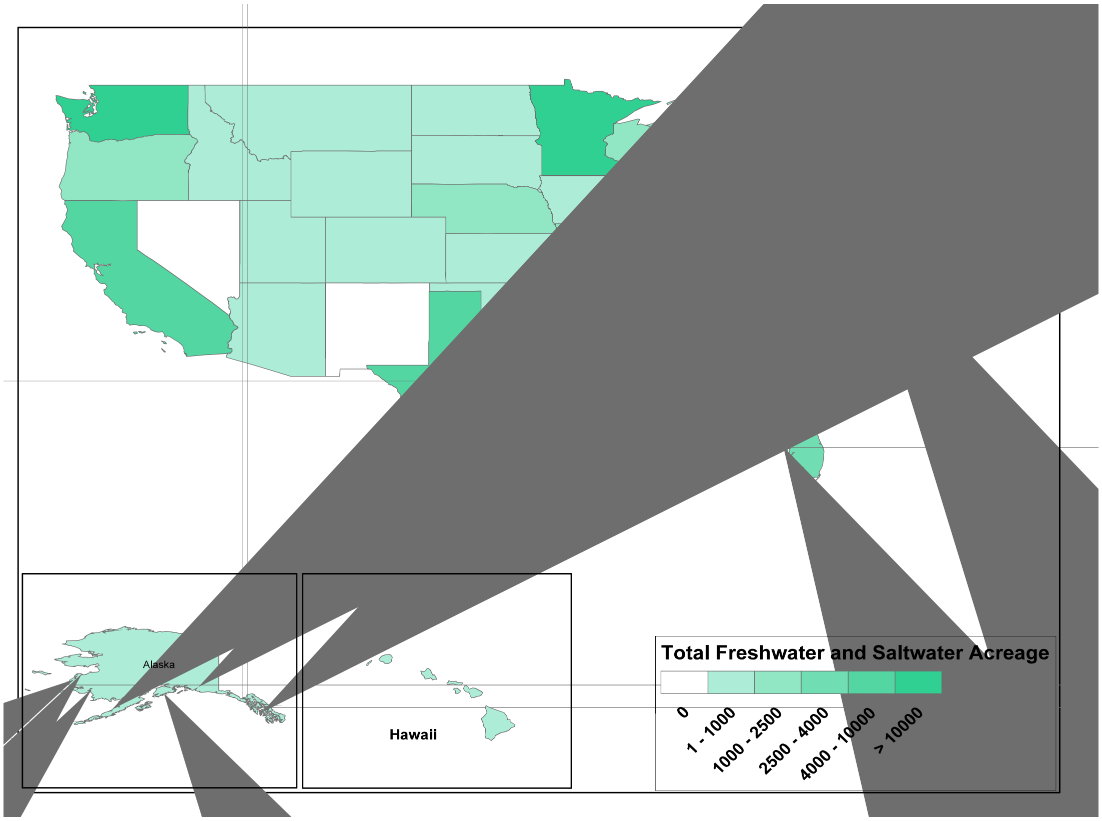


Figure 4-2. Location of U.S. Aquacultural Operations, 1998

4.1.3 Recreational Fishing

Recreational fishing is characterized by individuals fishing for sport/recreational purposes, and/or for subsistence.

4.1.3.1 Industry Characterization

4.1.3.1.1 Information on Recreational Anglers in the United States

In 2001, 29.5 million fishing licenses were sold in the United States, while the number of anglers (aged 16 and over) is estimated to be 34.1 million (USFWS, 2002). Of these 34.1 million anglers, 28.4 million participated in freshwater fishing and 9.1 million participated in saltwater fishing. Table 4-4 shows the number of fishing licenses sold, the number of anglers, and the participation rate (percent of population) for each region of the United States. Figure 4-3 shows the number of anglers per capita (aged 16 and over) by state. Data on recreational fishing demographics are provided in Section 4.3.3.2 below.

Approximately 16% of the U.S. population participates in recreational fishing activities. The states with the greatest numbers of anglers are California and Texas. However, when taking population into account, these states have relatively low participation rates (9% and 15%, respectively). As shown in Figure 4-3, the states with the highest levels of fishing participation are in the Northern Plains region: Minnesota (36%), Montana (32%), and Wyoming (32%) and Alaska (41%).

Table 4-4. Number of Anglers and Fishing Licenses in the United States

Region	Fishing Licenses (thousands)	Number of Anglers* (thousands)	Population* (thousands)	Participation Rate
United States Total	29,452	34,071	212,298	16%
Great Lakes	5,395	6,182	36,294	17%
Northeast	4,578	6,433	51,975	12%
Northern Plains	4,355	3,722	13,858	27%
South Central	4,556	5,572	28,880	19%
Southeast	5,463	7,192	40,241	18%
West	5,106	4,947	40,624	12%

*Ages 16 and over only

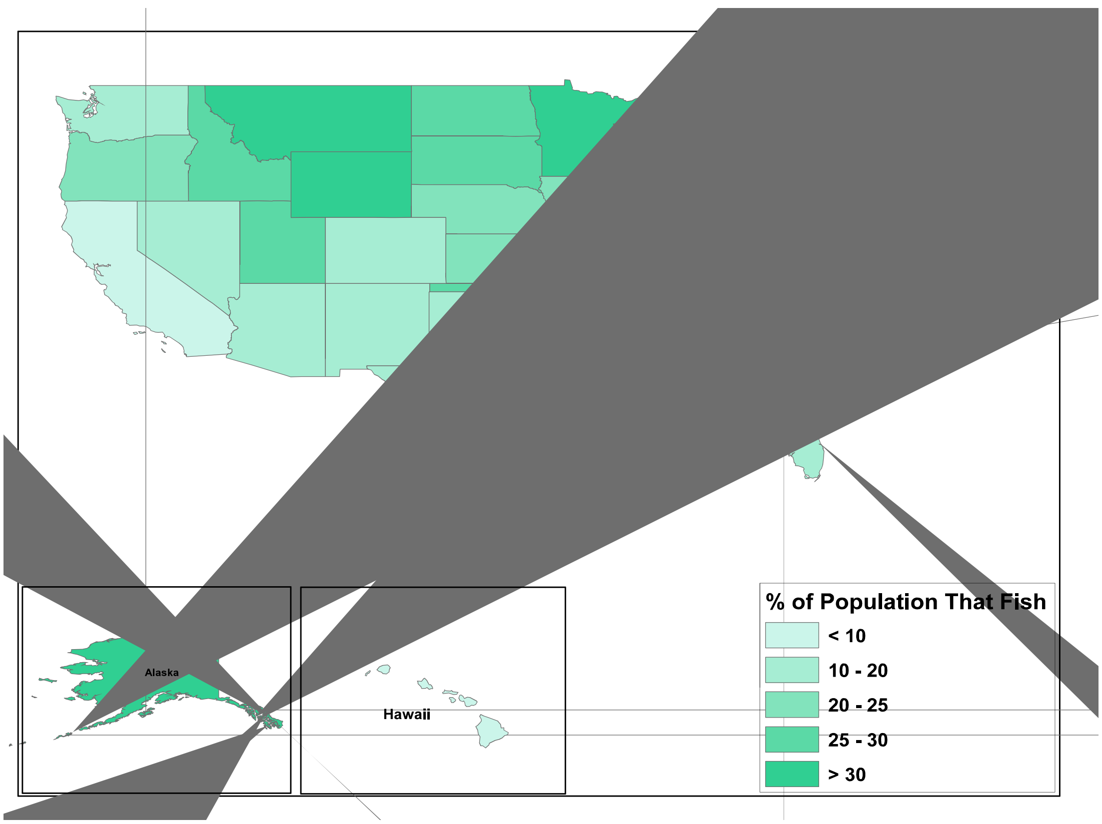


Figure 4-3. Recreational Fishing Participation Rates by State

4.2 U.S. Production Statistics for Commercial and Recreational Fishing

Data are provided in this section on production estimates for the commercial and recreational fishing sectors. Primarily, these data are estimates of the amount of fish landed by either commercial or recreational anglers. Available data are often aggregated as either finfish or shellfish or by groups of individual fish species. Production data for recreational freshwater fishing are lacking. More detailed production data are provided in the report titled "Profile of the Fishing Industry in the United States" available in the docket for this rulemaking (Pechan, 2005).

4.2.1 Commercial Fishing

This section summarizes available U.S. commercial production information (fish landings), including landings of finfish or shellfish, fish types, locations of fish landings, edible and non-edible uses of fish products, and seasonal production information. These data were taken from the National Marine Fisheries Service (NMFS) of the National Oceanic and Atmospheric Administration (NOAA).

The NMFS provides both monthly and annual summaries of commercial fishery landings that are updated weekly. Domestic fishery landings are those fish and shellfish that are landed and sold in the 50 states by U.S. fishermen and do not include landings made in U.S. territories or by foreign fishermen. Landings are provided in round (whole) weight even though many fish may be processed while at sea. Landings do not include aquaculture products except for clams, mussels and oysters.

4.2.1.1 Domestic Commercial Fishery Landings

Table 4-5 provides a summary of total domestic commercial landings of finfish and shellfish for the years 1993 – 2003 (landed by U.S. fishermen and sold in the U.S.). Although production data for 2003 are available, the year 2002 was selected for use in this profile, since the NMFS indicates that data for 2003 may be incomplete (NMFS, 2004). Pechan (2005) provides detailed 2002 landings data by fish species for finfish and shellfish, respectively.

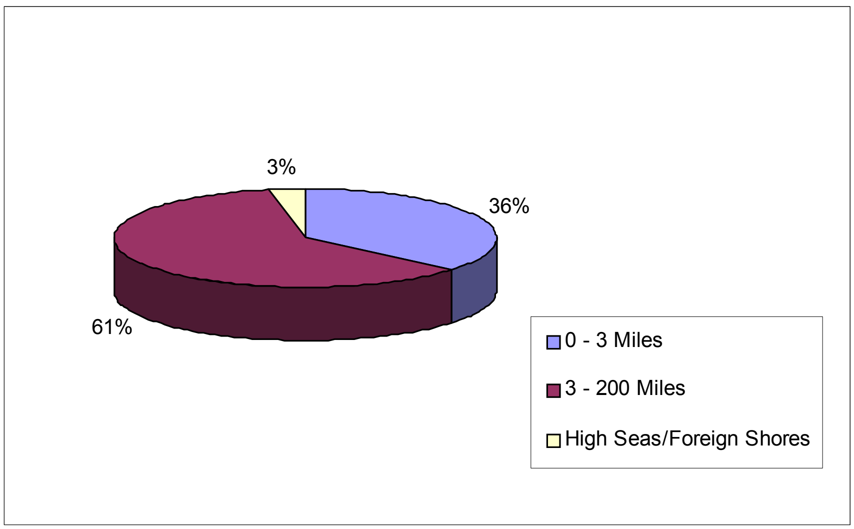
Table 4-5. Annual Domestic Landings for Commercial Fishing (Finfish and Shellfish)

Year	Metric Tons
1993	4,630,853
1994	4,762,637
1995	4,496,420
1996	4,374,409
1997	4,514,152
1998	4,233,291
1999	4,267,982
2000	4,147,069
2001	4,314,492
2002	4,270,030
2003	4,311,663

Source: NMFS, 2004.

NMFS categorizes commercial landings by distance to shore and identifies these distances as inland, state territorial sea, and federal exclusive economic zone with Florida and Puerto Rico having unique location identifiers. Inland landings refer to commercial landings in bays, estuaries, and sounds within the United States. State territorial sea represents the area that extends out three nautical miles from shore for all states with the exception of Florida and Puerto Rico whose territorial sea extends to ten nautical miles. Federal exclusive economic zone refers to the ocean area that extends to 200 nautical miles beyond the state territorial sea boundary (NMFS, 2003b).

Figure 4-4 is a chart showing the amount of commercial fish landed in inland waters and state and territorial seas (0 - 3 miles from shore) to be 36% of the total, while 61% of commercial fish were landed in the Federal exclusive economic zone (3 - 200 miles offshore). Only 3% of domestic commercial fish landings occurred in high seas locations or off foreign shores. Pechan (2005) provides additional details on the commercial fish species caught by distance from shore.



Source: NMFS, 2004.

Figure 4-4. 2002 Commercial Landings by Distance from Shore

A review of the commercial fish landings by region demonstrated that the Pacific coast accounted for 65% of total landings (see Table 4-6) with Alaska and California accounting for the largest portion. Alaska alone accounted for over 26% of total U.S. landings. The next highest landing region in percent, the Gulf Coast (18%), had significantly less landings than the Pacific Coast. Among the Gulf Coast landings, Louisiana was the state with the highest landing totals. New England and the Chesapeake Bay contributed 6% and 5% to U.S. landings, respectively, with total commercial landings in Virginia and Massachusetts (686 million pounds) having the highest landing numbers among their groups. The Mid-Atlantic, South Atlantic, and the Great Lakes regions combined for the remaining 4.5% of total commercial landings. Within the United States, the ports registering the highest number of fish landings in 2002 were Dutch Harbor-Unalaska, AK, Empire-Venice, LA, and Reedville, VA.

Table 4-6. Commercial Fish Landings by Region

Region	Metric Tons	Percent
New England	264,859	6.2
Middle Atlantic	93,756	2.2
Chesapeake Bay	224,834	5.3
South Atlantic	97,431	2.3
Gulf Coast	778,428	18.3
Pacific Coast	2,784,260	65.5
Great Lakes	8,096	0.2
TOTAL	**4,251,664**	**100**

Source: NMFS, 2004.

Fish landings can be disaggregated into edible fish (fish used for human consumption) and non-edible products (often used for animal consumption, fish meal, fish oil, or bait). Fish landings ready to be used for human food consumption are generally either fresh or frozen, while fish that are canned, cured, or processed for both human and non-human consumption are categorized as landings for industrial purposes.

Of the 4.3 million tons landed by U.S. fishermen in 2002, an estimated 3.4 million metric tons were to be used for human food while the remaining 0.9 million metric tons of fish landings were designated to be used for other purposes (fish meal, oil, or other including bait and animal food). These data are shown in Table 4-7 below. Among the canned and cured products, about 320,000 metric tons were to be used for human food. About 4% of total landings were used for bait and other animal consumption. About 19% of landings were used for reduction to fish meal, oil, or other fish products. The amount of this last end use destined for human consumption (i.e. fish oil) was not available.

Table 4-7. 2002 U.S. Commercial Fish Landings by End Use

End Use	Metric Tons	Percent
Fresh and Frozen:		
For human food	2,940,000	69.3
For bait and animal food	150,000	3.5
Subtotal	**3,090,000**	**72.8**
Canned:		
For human food	260,053	6.1
For bait and animal food	24,926	0.6
Subtotal	**284,979**	**6.7**
Cured:		
Cured for human food	53,024	1.2
Reduction to meal, oil, other	816,666	19.2
Subtotal	**869,690**	**20.4**
Total	**4,244,669**	**100**

Source: NMFS, 2003b.
Totals do not add exactly or match between tables due to rounding.

In 2002, almost one-third of all commercial landings for human consumption occurred during the months of July and August (see Table 4-8). An additional 25% of total commercial landings occurred during the months of February and March while roughly 11% of fish were landed during September.

Table 4-8. 2002 U.S. Commercial Landings by Month

Month	Landings for Human Food Metric Tons	Landings for Human Food Percent	Landings for Industrial Purposes Metric Tons	Landings for Industrial Purposes Percent	Total Metric Tons	Total Percent
January	149,234	4.6	24,494	2.5	173,729	4.1
February	432,734	13.2	20,412	2.1	453,146	10.6
March	400,982	12.3	16,783	1.7	417,766	9.8
April	107,050	3.3	54,432	5.5	161,482	3.8
May	154,224	4.7	90,266	9.1	244,490	5.7
June	216,367	6.6	136,534	13.7	352,901	8.3
July	484,445	14.8	203,213	20.4	687,658	16.1
August	597,845	18.3	168,739	17.0	766,584	18.0
September	358,344	11.0	117,936	11.9	476,280	11.2
October	218,635	6.7	107,503	10.8	326,138	7.7
November	90,720	2.8	28,123	2.8	118,843	2.8
December	57,607	1.8	25,855	2.6	83,462	2.0
Total	**3,268,188**	**100**	**994,291**	**100**	**4,262,479**	**100**

Totals do not match between tables due to rounding.
Source: NMFS, 2003b.

Table 4-9 provides a summary of the species of finfish with the highest landings by U.S. commercial fishermen during 2002 (tuna were added for comparison; most tuna consumed in the U.S. are imported). Pechan (2005) provides additional detail in 2002 finfish landings by species. Pollock and Menhaden were by far the finfish types with the highest commercial landings by U.S. craft in 2002. These two finfish types are almost exclusively caught by commercial fishermen. Pollock are often used in making frozen fish products (e.g., fish sticks), surimi (see Section 4.2.1.3), and other minced fish products. Menhaden are often used to produce cut or live bait, fishmeal and fish oil.

Table 4-9. Finfish Species with the Highest 2002 U.S. Commercial Landings

Finfish Type	Metric Tons	% of Total Finfish Landed
Pollock	1,519,101	41.4
Menhaden	793,486	21.6
Salmon	254,613	6.94
Cod	245,705	6.70
Hake	141,642	3.86
Sole	116,779	3.18
Herring	99,173	2.70
Sardine	97,527	2.66
Tuna*	22,513	0.61
Other Finfish	377,904	10.3
Total 2002 Finfish	**3,668,443**	**100**

Source: NMFS, 2004.
Note: These estimates are for domestic landings and do not include quantities of imports.
* There are a number of species that rank higher than tuna, but are not displaed in this table.
Tuna is added for comparison purposes only to other species.

Table 4-10 provides a summary of the shellfish types with the highest commercial harvests in 2002 by the domestic fleet. Nearly half of the 2002 landings were for shrimp and crab. Of the total finfish and shellfish landings in 2002, shellfish represent about 14% of the total. Details on the commercial landings of shellfish in 2002 are provided in a separate report (Pechan, 2005).

Table 4-10. Shellfish Types with the Highest 2002 Commercial Landings

Shellfish Type	Metric Tons	% of Total Shellfish Landed
Shrimp	156,059	25.9
Crab	139,840	23.2
Squid	93,217	15.5
Clam	59,059	9.82
Lobster	39,454	6.56
Scallop	24,047	4.00
Oyster	16,701	2.78
Other Shellfish[a]	73,238	12.2
Total 2002 Shellfish	**601,615**	**100**

Source: NMFS, 2004.
[a] NMFS includes some non-shellfish species in the "Other shellfish" group including seaweed (7.8%), sponges (0.05%) and turtles (0.006%).

4.2.1.2 Aquaculture

Total aquacultural production in 2002 was 393 thousand metric tons with catfish accounting for 73% or 286 thousand metric tons (NMFS, 2004). Aquaculture provides most of the commercial catfish production in the U.S. Data for both finfish and shellfish are provided in Table 4-11. Trout production levels were 6% of the total, while salmon accounted for 3%. An additional 7% of aquacultural production in 2002 was attributed to raising crawfish.

4.2.1.3 Exports of Finfish and Shellfish

Total exports include both "exports" (exports of fishery products of domestic origin) and "re-exports" (exports of fishery products of foreign origin). Total edible finfish and shellfish exports for 2002 were about 970,000 metric tons (product weight), with finfish accounting for about 85% of this amount (NMFS, 2003b). Table 4-12 provides 2002 summary export data for finfish. These export data include products of both domestic and foreign origin. The largest export of a particular finfish product was surimi, which made up 23% of the exported finfish products. Surimi is a Japanese word meaning "minced fish". It is typically produced from skinless Alaskan pollack and used in the subsequent manufacture of imitation fish products (e.g., imitation crabmeat). Over 91% of the finfish products exported were either fresh or frozen. (Pechan, 2005) provides additional details of fresh and frozen fish exports in 2002.

Table 4-13 provides 2002 export data for shellfish. Among U.S. 2002 shellfish exports, the largest export products were as follows: squid (39%), lobster (20%), crabs (10%), and shrimp (10%). Nearly 89% of shellfish products were exported either fresh or frozen. Pechan (2005) provides more detailed information on shellfish exports.

Table 4-11. Total 2002 U.S. Aquacultural Production

Fish Species	Metric Tons	Percent
Finfish:		
Baitfish[a]	6,329	1.6
Catfish	286,039	72.7
Salmon[a]	12,734	3.2
Striped bass[a]	4,758	1.2
Tilapia	9,000	2.3
Trout	24,699	6.2
Finfish Subtotal	**343,559**	**87.3**
Shellfish:		
Clams[a]	4,473	1.1
Crawfish	27,825	7.1
Mussels[a]	627	0.2
Oysters[a]	8,413	2.1
Shrimp[a]	4,080	1.0
Miscellaneous[a]	4,425	1.1
Shellfish Subtotal	**49,843**	**12.7**
Total	**393,402**	**100**

Source: NMFS, 2003b.

[a] Saltwater species. The baitfish and and all shellfish species (except crawfish) are believed to be primarily saltwater species, although definitive data were not available from the source material.

Table 4-12. 2002 Exports of Edible Finfish Products

Product	Metric Tons	% of Total
Fresh or Frozen Finfish:		
Freshwater (whether or not whole):		
Eels	3,063	0.36
Tilapia	2,331	0.28
Trout	600	0.07
Total Freshwater (whether or not whole)	**5,994**	**0.73**
Saltwater (whether or not whole):		
Flatfish[a]	67,519	8.01
Groundfish[b]	103,900	12.3
Salmon	78,539	9.32
Tuna	15,302	1.82
Other[c]	159,613	18.9
Total Saltwater Finfish (whether or not whole)	**424,873**	**51.5**
Freshwater (fillets and steaks):		
Catfish	73	0.01
Tilapia	2,065	0.25
Total Freshwater (fillets and steaks)	2,138	0.06
Saltwater (fillets and steaks):		
Flatfish[a]	760	0.10
Groundfish[b]	80,514	9.56
Other	16,397	1.95
Total Saltwater (fillets and steaks)	97,671	11.4
Blocks, Regular And Minced	26,372	2.93
Surimi	190,911	23.1
Fish Sticks And Similar Products	21,332	2.55
Total Finfish: Fresh And Frozen	**769,291**	**91.3**
Canned Finfish		
Anchovy	333	0.04
Herring	3,313	0.39
Sardine	16,190	1.92
Mackerel	1,049	0.12
Salmon	44,708	5.31
Tuna	1,628	0.19
Total Finfish: Canned	**67,221**	**7.98**
Cured Finfish		
Dried	843	0.10
Pickled or Salted	4,554	0.54
Smoked or Kippered	503	0.06
Total Finfish: Cured	**5,900**	**0.70**
Total Finfish Exports[d]	**842,412**	**100**

Source: NMFS, 2003b. Includes exports of edible products of both domestic and foreign origin.

[a] Flatfish includes halibut.

[b] Groundfish include cod and pollock.

[c] Other include atka mackerel, butterfish, herring, lingcod, mackerel, monkfish, mullet, sablefish, sardine, scorpionfish, sea bass, shark and dogfish, toothfish, and unclassified.

[d] Excludes other fish products such as fish balls, cakes, and puddings, fish/shellfish juice, soups and broths, caviar and roe, prepared fish meals, and other.

Table 4-13. 2002 Exports of Edible Shellfish Products

Product	Metric Tons	% of Total
Fresh and Frozen Shellfish Products		
Crabs and Crabmeat	16,089	10.3
Crawfish (freshwater)	199	0.13
Lobsters and Lobster Meat	30,748	19.7
Shrimp	15,060	9.67
Conch	444	0.29
Clams	834	0.54
Cuttlefish	145	0.09
Mussels	645	0.41
Oysters	1,788	1.15
Scallops	4,589	2.95
Octopus	528	0.34
Squid	60,151	38.6
Snails	36	0.02
Sea Urchins	1,505	0.97
Unclassified	5,472	3.51
Frog Legs and Meat[a]	79	0.05
Total Fresh and Frozen Shellfish Products	**138,312**	**88.8**
Canned Shellfish Products		
Crabmeat	538	0.35
Lobsters and Lobster Meat	31	0.02
Shrimp	1,507	0.97
Clams	1,762	1.13
Squid	13,575	8.72
Total Canned Shellfish Products	**17,413**	**11.2**
Total Shellfish	**155,725**	**100**

Source: NMFS, 2003b.
[a] NMFS includes frog legs and meat in the shellfish category.

4.2.2 Recreational Fishing

Data on recreational freshwater fish harvest are not available for this profile. Information on the freshwater species most often targeted by recreational anglers are provided later in Section 4.3.3.1 (see Table 4-24).

NMFS provided data on estimated recreational marine (saltwater) finfish landings based on the Marine Recreational Fishing Statistical Survey (MRFSS; NMFS, 2003b). Similar data for recreational shellfish landings are not available. The purpose of the MRFSS is to establish a reliable data base for estimating the impact of marine recreational fishing on marine resources.

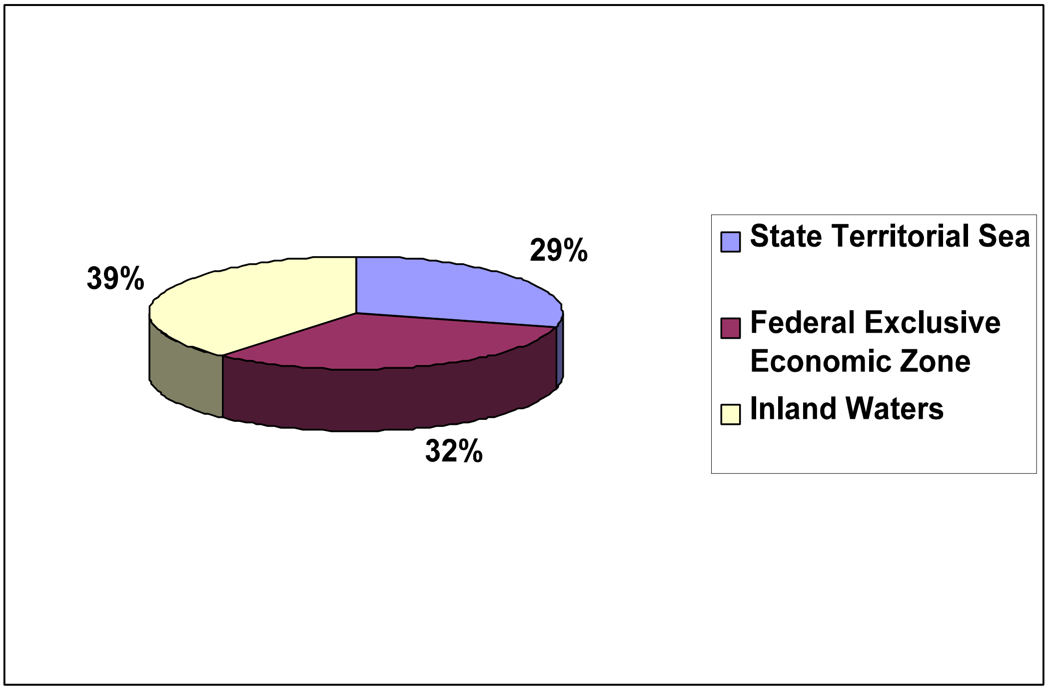
This annual survey has been conducted since 1979. The MRFSS covers all coastal states except Texas and Alaska. The survey provides coverage of saltwater sport fishing (including estuarine and brackish water) from private/rental boats, charter and head boats, and the shore on the Atlantic Coast (Maine-East Florida), Gulf Coast (Louisiana-West Florida), and Pacific coast (Washington through California).

A summary of recreational marine finfish landed for 2002 is provided in Table 4-14 below. This summary excludes the fish that were caught and subsequently released alive. More detailed data are provided in a separate report (Pechan, 2005). It should be noted that recreational marine finfish landings in 2002 were less than 20% of the average annual landings estimated by NMFS from 1999 - 2003 (Pechan, 2005). NMFS indicates that recreational boat fishing trips were not included for 2002 for Washington and Oregon, but other reasons for the large discrepancy were not identified.

Table 4-14. 2002 Recreational Marine Landings for Selected Finfish Types

Finfish Type	2002 Landings (Metric Tons)	% of Total	1999-2003 Average Landings (Metric Tons)
Dolphinfish	6,712	6.5	35,826
Drums	20,366	20	116,503
Flounders	5,997	5.8	37,533
Pacific Barracuda	9,292	9.0	3,393
Sea Basses	6,896	6.7	29,866
Temperate Basses	8,903	8.6	44,561
Tunas and Mackerels	14,103	14	90,921
Other Finfish	31,262	30	200,714
Total	**103,531**	**100**	**559,317**

Figure 4-5 shows that roughly 32% of the 103,529 metric tons of marine finfish landed in U.S. waters in 2002 occurred in the Federal Exclusive Economic Zone (distances of 3-200 miles from shore) while 29% of total finfish landings were within 3 miles from shore (State Territorial Sea). The remaining 39% of finfish were landed in Inland Waters. For shellfish, 65% of the 534,608 metric tons landed were made within Inland Waters and State and Territorial Sea (< 3 miles from shore). A separate report provides more details on the types of marine recreational fish caught by distance from shore (Pechan, 2005).



Source: NMFS, 2004.

Figure 4-5. 2002 Recreational Marine Finfish Landings by Distance from Shore

Table 4-15 provides 2002 recreational marine catch data by region and top harvested species group. This summary data was compiled based on the MRFSS and summarized by U.S. region. The regions were constructed by the American Sportfishing Association (ASA). The data are summarized this way to be consistent with recreational fishing data provided later in this report.

The northeast region includes the coastal New England States, Delaware, District of Columbia, Maryland, New Jersey, New York, and Virginia. The south central region includes data only for Louisiana, since data for Texas were not included in the MRFSS. The southeast region includes Alabama, Florida, Georgia, Mississippi, North Carolina, and South Carolina. The west region includes California, Oregon, Washington, and Hawaii (Alaska was not included in the MRFSS). Harvested weight was not available for all state-species groups. Therefore, average weights per fish for each region were estimated using the available data. These average weights per fish were used to estimate harvested weight where unavailable.

The greatest amount of recreational marine catch was harvested in the Southeast region followed by the Northeast. Harvest values for the West region would likely be much higher if the harvest for Alaska were included, and values in the south central region would be higher if data for Texas were included in the MRFSS.

Table 4-15. 2002 Recreational Marine Catch and Harvest by Region and Top Species Group

Species Group	# of Fish Caught	# of Fish Harvested[a]	Harvested Weight[b] (metric tons)
Northeast			
Total for Region	**128,019**	**41,934**	**37,013**
Drums	28,675	14,361	5,568
Tunas and Mackerels	4,356	3,821	3,711
Bluefish	10,411	3,684	4,326
Porgies	7,641	3,661	1,682
Flounders	17,326	3,550	3,802
South Central[c]			
Total for Region	**24,565**	**10,457**	**9,974**
Drums	18,724	8,953	8,143
Porgies	2	652	679
Flounders	320	272	144
Catfishes	3,123	194	127
Snappers	190	130	307
Southeast			
Total for Region	**225,303**	**111,486**	**40,882**
Herrings	53,400	46,693	211
Drums	42,101	15,566	6,671
Porgies	22,793	10,448	2,593
Mullets	10,240	8,442	1,103
Jacks	13,796	6,839	2,813
West[c]			
Total for Region	**42,015**	**24,067**	**15,944**
Rockfishes	5,435	4,270	2,797
Smelts	4,186	4,174	142
Flounders	5,464	3,875	1,378
Sea Basses	6,931	2,403	1,538
Herrings	2,399	2,216	151

[a]Harvest equals catch minus fish released alive.
[b]Weight estimated for some state/species groups.
[c]Does not include Alaska (west) and Texas (south central).

4.3 U.S. Demand for Commercial and Recreational Fishing

This section presents information on the U.S. demand for commercial and recreational fishing and related industries.

4.3.1 Commercial Imports

This section provides information on commercial imports of finfish and shellfish products. Included are data on the type of products and supplying countries. These products include those produced through commercial catches, as well as foreign aquaculture. Total edible imports for 2002 were 4.0 million metric tons with finfish contributing 65% of the total. A detailed breakdown, by product weight, of edible fish imports is provided in a separate report (Pechan, 2005). The following sections provide a summary of edible finfish and shellfish imports. In comparison to domestic production (net of exports), commercial imports represent a large component of total U.S. demand for fish products. It should be noted that the CAMR is not likely to substantially affect the level of methylmercury in fish that is imported to the U.S., because of the small contribution of U.S. emissions to the global pool that impacts MeHg in imported fish.

Figures 4-6 and 4-7 provide a breakdown of commercial fishery products by region and country, respectively (NMFS, 2003b). An estimated 46% of US imports were from Asia while 26% were from other regions in North America. South America could be credited with landing 17% of U.S. imports while Europe, Africa, and Oceania combined to total 11% (Figure 4-6). Although the largest portions of U.S. fish imports were from Asian landings, Canada represents the largest share at 18% when compared on a per country basis (Figure 4-7).

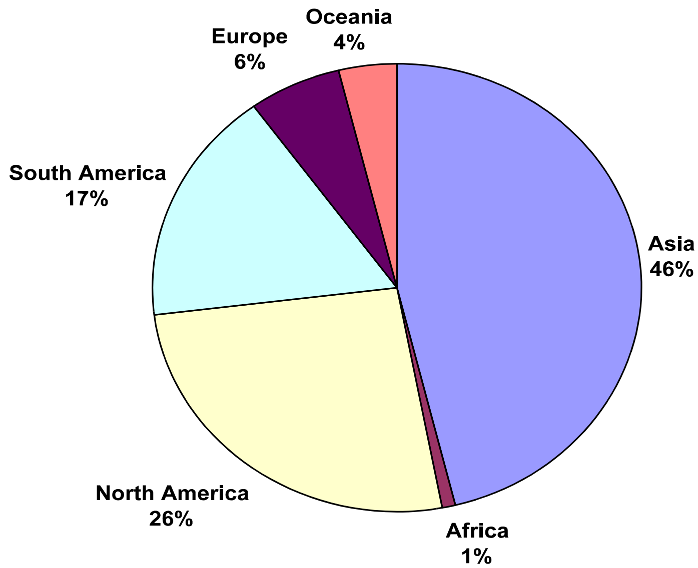
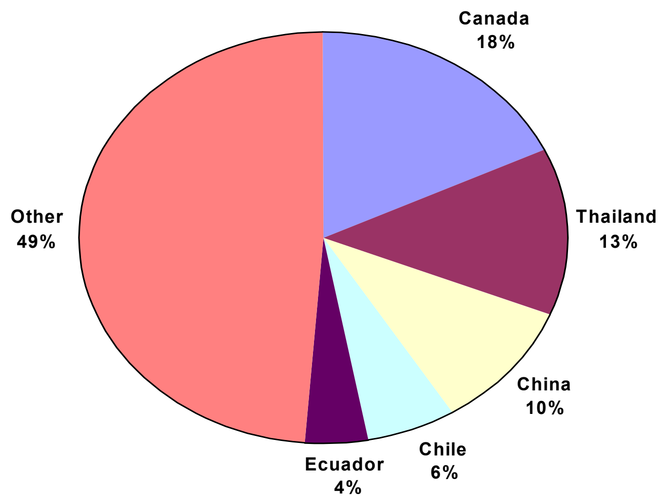


Figure 4-6. U.S. Commercial Fish Imports by Area, 2002



Source: NMFS, 2003b.

Figure 4-7. U.S. Commercial Fish Imports by Country, 2002

4.3.1.1 Finfish

Table 4-16 provides a breakdown of edible fresh and frozen finfish products imported to the United States in 2002 (see detailed breakdown in Pechan (2005)). Fresh and frozen finfish are categorized into the following products: whether or not whole; regular blocks; minced blocks; fillets and steaks; surimi; and fish sticks and similar products. The total imported fresh or frozen finfish products in 2002 was almost 1 million metric tons. Of the freshwater species imported, tilapia was the most common and accounted for over 6% of the total finfish product imports. Tuna was the largest group of saltwater fish species imported at nearly 17% of the fresh or frozen imports.

Table 4-17 provides information on imported canned and cured finfish products imported in 2002 (see details in Pechan, 2005). In 2002, about 288,000 metric tons of canned and cured finfish products were imported into the United States for consumption. Nearly 60% of this amount was canned tuna.

Table 4-16. 2002 Imports of Edible Fresh or Frozen Finfish Products

Finfish Product	Metric Tons	% of Total Finfish by Weight
Whether or Not Whole		
Freshwater:		
Tilapia	40,748	4.19
Unclassified	17,642	1.81
Total Freshwater Fish	**58,390**	**6.00**
Saltwater:		
Flatfish[a]	21,111	2.17
Groundfish[b]	24,615	2.53
Salmon	82,665	8.50
Tuna	162,252	16.68
Other	116,840	12.01
Total Saltwater Fish	**407,483**	**41.9**
Total Whether Or Not Whole	**465,873**	**47.9**
Blocks, Regular		
Total Freshwater Fish	127	0.01
Saltwater:		
Flatfish[a]	1,460	0.15
Groundfish[b]	50,572	5.20
Other	2,494	0.26
Total Saltwater Fish	**54,256**	**5.61**
Total Regular Blocks	**54,653**	**5.62**
Blocks, Minced		
Total Freshwater Fish	1	0.00
Saltwater:		
Flatfish[a]	13	0.00
Groundfish[b]	1,934	0.20
Other	10,091	1.04
Total Saltwater Fish	**12,038**	**1.24**
Total Minced Blocks	**12,039**	**1.24**
Fillets And Steaks		
Freshwater:		
Tilapia	26,440	2.72
Other	24,134	2.48
Total Freshwater Fillets	50,574	5.20
Saltwater:		
Flatfish[a]	23,369	2.40
Groundfish[b]	104,985	10.8
Other	239,535	24.6
Total Saltwater Fillets	**367,889**	**37.8**
Total Fillets And Steaks	**418,463**	**43.0**
Surimi	**3,559**	**0.37**
Fish Sticks And Similar Products	**18,271**	**1.88**
Total Fresh And Frozen Finfish	**972,858**	**100**

Source: NMFS, 2003b.

[a] Flatfish include halibut, Greenland turbot, plaice, sole, and other.

[b] Groundfish include cod, haddock, hake, Pollock and ocean perch.

Table 4-17. 2002 Imports of Edible Canned and Cured Finfish Products

Finfish Product	Metric Tons
Canned	
Anchovy	3,298
Herring	3,814
Mackerel	9,928
Salmon	4,542
Sardines	22,220
Tuna	171,523
Balls, Cakes, and Puddings	9,014
Other (includes Finfish and Shellfish)	28,880
Total, Canned	**253,219**
Cured	
Dried	7,468
Pickled or Salted	20,952
Smoked or Kippered	6,498
Total, Cured	**34,918**
Total Canned and Cured Imports	**288,137**

Source: NMFS, 2003b.

4.3.1.2 Shellfish

Table 4-18 provides data on 2002 edible fresh and frozen shellfish imported into the United States. Total shellfish imports were about 692,000 metric tons in 2002. By far, shrimp was the largest fresh and frozen shellfish product imported (over 60% of total shellfish). Crab and crabmeat was the next highest product imported at over 11% of total shellfish. Pechan (2005) contains details of the fresh and frozen shellfish products imported in 2002.

Table 4-19 provides data on 2002 edible canned shellfish product imports (for this table, it was assumed that all cured edible fish products are finfish products and they were placed in Table 4-17 above). Over 33,000 metric tons of canned shellfish products were imported for consumption in 2002. About 60% of this amount was canned crabmeat.

Table 4-18. 2002 Imports of Edible Fresh and Frozen Shellfish Products

Shellfish Product	Metric Tons	% of Total Shellfish
Total Crab and Crabmeat	79,805	11.5
Total Lobster and Lobster Meat	50,586	7.30
Shrimp	427,454	61.8
Clam	8,800	1.27
Mussels	20,727	3.00
Scallops	21,868	3.16
Octopus	14,164	2.05
Squid	46,759	6.76
Unclassified or Other[a]	21,774	3.15
Total Shellfish	**691,937**	**100**

Source: NMFS, 2003b.

[a] Other includes Abalone, krill, crawfish, conch, cuttlefish, oysters, snails, sea urchins, and frog legs and meat.

Table 4-19. 2002 Imports of Edible Canned Shellfish Products

Shellfish Product	Metric Tons	% of Total Shellfish
Clams	5,330	15.9
Crabmeat	20,545	61.2
Lobsters	47	0.14
Oysters	5,825	17.3
Shrimp	1,849	5.50
Total	**33,596**	**100**

Source: NMFS, 2003b.

4.3.2 U.S. Demand for Commercial Fishery Products

Table 4-20 provides a summary of commercial finfish demand per capita for several important finfish types. These estimates were derived from the production data provided in Section 4.2, the import data summarized above, and a 2002 U.S. population estimate of 288 million (BOC, 2005). These estimates should not be considered per capita consumption estimates for several reasons. First, not all fish species landed commercially are intended for human consumption (about 23% are for industrial purposes). Most of the non-edible landings are for menhaden (about 80%). This is noted at the bottom of Table 4-20. In addition, existing U.S. inventory of fishery products are not included in these estimates. Conversions have not been made to estimate the edible portion of each type of edible finfish or shellfish. Finally, landings are provided in units of live weight, whereas product import and export data are provided as product weights. Table 4-21 provides similar estimates of per capita demand for shellfish products.

Table 4-20. 2002 U.S. Demand for Commercial Finfish (metric tons)

Finfish Type	2002 Production[a]	2002 Exports	2002 Imports	2002 U.S. Demand	Demand per capita (lb)
Pollock	1,519,101	331,976[b]	80,889	1,268,014	9.71
Menhaden	793,486	0	0	793,486	6.08
Salmon	275,382	133,380	210,735	352,737	2.70
Cod	245,705	103,934	78,777	220,548	1.69
Hake	141,642	8,886	10,622	143,378	1.10
Sole	116,779	48,117	12,353	81,015	0.62
Herring	99,173	18,565	7,846	88,454	0.68
Sardine	97,527	61,288	24,794	61,033	0.47
Tuna	22,513	16,930	333,775	339,358	2.60
Other Finfish	693,821	168,460	435,722	961,083	7.36
Total Finfish	**4,005,129**	**891,536**	**1,195,513**	**4,309,106**	**32.99**

Source: NMFS, 2004.
[a] Production = Commercial Landings + Aquacultural Production.
[b] Assumes that surimi and regular/minced fish blocks are all made up of pollock.

Table 4-21. 2002 U.S. Demand for Commercial Shellfish (metric tons)

Shellfish Type	2002 Production[a]	2002 Exports	2002 Imports	2002 U.S. Demand	Demand per capita (lb)
Shrimp	159,666	16,567	429,303	572,402	4.38
Crab	139,840	16,627	85,135	208,348	1.60
Squid	93,217	73,726	46,759	66,250	0.51
Clam	63,584	2,596	14,130	75,118	0.58
Lobster	39,454	30,779	50,633	59,308	0.45
Scallop	24,047	4,589	21,868	41,326	0.32
Oyster	24,330	1,788	5,825	28,367	0.22
Other Shellfish	92,260	15,289	56,665	133,636	1.02
Total Shellfish	**636,398**	**161,961**	**710,318**	**1,184,755**	**9.07**

Source: NMFS, 2004.
[a] Production = Commercial Landings + Aquacultural Production.

The data in Table 4-20 show that most of the U.S. tuna demand is met through imports. In 2002, less than 7% of the tuna demand is met through commercial landings by the domestic fleet.

Total 2002 per capita demand for commercial fishery products is estimated at 42.1 lb/person. This estimate can be compared to an NMFS estimate of "per capita use" of 66.0 lb/person in 2002, which includes both edible and industrial uses but does not include exports, beginning/ending year inventory, or defense purchases (NMFS, 2003b).

NMFS developed estimates of 2002 U.S. per capita consumption for edible commercial fishery products (NMFS, 2003b). To do this, a "disappearance model" was developed to account for the edible portion of commercial landings and imports, exports of edible products, and inventories of edible products. NMFS caveats their estimates by noting that the data sources for the model are not always reported completely, and that incorrect model assumptions can lead to significant changes in estimated consumption. NMFS estimated a total 2002 U.S. per capita consumption of 15.6 lb edible meat/person. Additional consumption data are provided in Table 4-22 below.

Table 4-22. 2002 U.S. Consumption of Commercial Fishery Products

Commercial Product	2002 Consumption (lb/person)
Fresh and Frozen Finfish and Shellfish	11.0
Canned Finfish and Shellfish	4.3
Cured Finfish and Shellfish	0.3
Total	**15.6**
Consumption of Specific Products	
Canned:	
Salmon	0.5
Sardines	0.1
Tuna	3.1
Shellfish	0.3
Other	0.3
Fresh and Frozen:	
Fillets and Steaks	4.1
Sticks and Portions	0.8
Shrimp (all preparations)	3.7

Source: NMFS, 2003b.

4.3.3 Recreational Fishing

4.3.3.1 U.S. Consumption of Recreationally-Caught Fish

Data on recreational saltwater fishing catch is available from the Marine Recreational Fishing Statistics Survey (MRFSS) from the NOAA Fisheries Statistics Division. These data were provided in Section 4.2.2. This survey provides data on catch (all fish caught) and harvest (all fish not released alive).

Data for freshwater recreational harvest are not available; however, the number of anglers and number of days of fishing is available from the 2001 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (USFWS, 2002). This survey is a partnership effort between the USFWS, States and national conservation organizations. The purpose of the survey is to quantify the economic impact of wildlife-based recreation. The 2001 survey was the tenth in a series that began in 1955. Information from a total of 25,070 sportspersons (anglers and hunters) were gathered for the 2001 survey. Data from the survey were summarized by the American Sportfishing Association (ASA, 2003). Table 4-23 shows the number of anglers for the top freshwater, Great Lakes, and saltwater species.

Table 4-23. Number of Anglers and Days of Fishing for 2001

Type of Fish Targeted	Anglers (millions)	Days of Fishing (millions)	Average # Days/Angler
Freshwater except Great Lakes			
Black bass	10.7	160	15.0
Panfish	7.9	103	13.0
Trout	7.8	83	10.6
Catfish/bullhead	7.5	104	13.9
Crappie	6.7	95	14.2
White bass, striped bass, and striped bass hybrids	4.9	62	12.7
Total Freshwater except Great Lakes	**45.5**	**607**	**13.3**
Great Lakes			
Perch	0.7	7	10.0
Walleye, sauger	0.6	6	10.0
Black bass	0.6	6	10.0
Salmon	0.5	4	8.0
Lake trout	0.3	4	13.3
Steelhead	0.3	4	13.3
Total Great Lakes	**3.0**	**31**	**10.3**
Saltwater			
Flatfish (flounder, halibut)	2.3	21	9.1
Striped bass	1.7	17	10.0
Sea trout	1.5	17	11.3
Bluefish	1.1	12	10.9
Salmon	0.7	5	7.1
Mackerel	0.6	6	10.0
Total Saltwater	**7.9**	**78**	**58.4**
Total - All Fish	**56.4**	**716**	**12.7**

Source: USFWS, 2002.

Table 4-24 shows the most targeted freshwater species, measured by angler participation, for each region of the country. Bass (large and small mouth), pan fish, trout, catfish, and crappie were the most popular target species for freshwater anglers nationwide. The region with the largest number of anglers for all targeted species was the southeast followed by the south central United States. In the western part of the United States, trout were by far the most targeted freshwater species.

Table 4-24. 2001 U.S. Recreational Freshwater Fishing: Targeted Species by Region

Targeted Species	Number of Anglers (thousands)						
	West	South East	South Central	North East	Great Lakes	Northern Plains	All Regions
Crappie	186	2,027	1,822	477	1,478	880	6,648
Panfish	230	1,907	1,177	876	2,893	1,062	7,894
Bass (white, striped)	526	1,387	1,217	718	904	327	4,925
Bass (large & smallmouth)	873	2,897	2,320	2,139	2,144	796	10,694
Catfish	612	2,275	2,330	651	1,295	517	7,494
Walleye	36	108	99	333	1,142	1,601	3,214
Sauger	[a]	[a]	[a]	[a]	63	63	174
Pike	[a]	[a]	[a]	336	763	914	2,060
Trout	2,645	583	749	1,926	458	1,768	7,797
Salmon	932	[a]	31	189	99	111	1,368
Steelhead	370	[a]	[a]	59	96	[a]	536
Other	644	1,176	409	440	296	211	3,176
Anything	392	1,413	856	963	812	445	4,689
Total[b]	**4,127**	**6,107**	**5,208**	**4,587**	**5,388**	**4,344**	**27,913**

[a] Sample size is too small to report results with any reliability, (N = 30-39); or the species is not present in this region.
[b]Angler numbers reported here are the number of anglers who fished in the given region. Summing across the regions to derive a U.S. total will result in an overestimate as some anglers fished in more than one region. The All Regions totals reported in this table eliminates all double-counting and reports the actual number of anglers in the United States.

A separate report provides information on the number of days spent fishing by anglers in each State (Pechan, 2005). Figures 4-8 through 4-10 are maps showing the total number of days spent fishing in 2001 by all anglers, resident anglers, and non-resident anglers, respectively. High levels of recreational fishing are shown to occur in the Great Lakes States, Florida, New York, Texas, and California. Pechan (2005) provides additional information on the expenditures on fishing equipment and related recreational angling activities across the U.S.

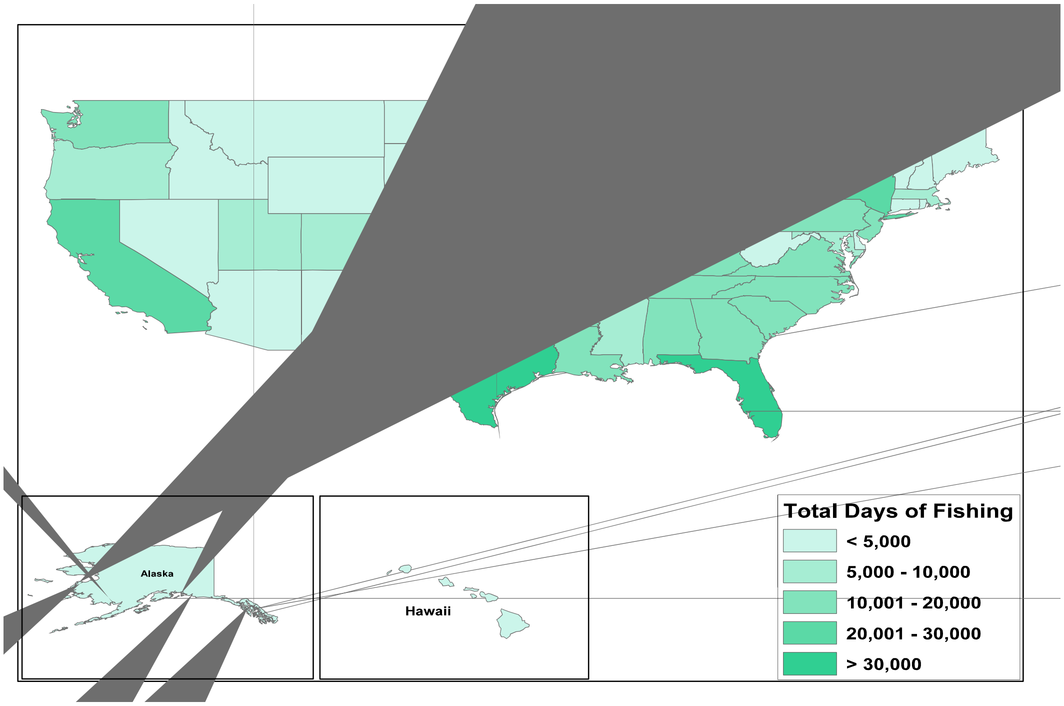


Figure 4-8. 2001 Total Recreational Fishing Days

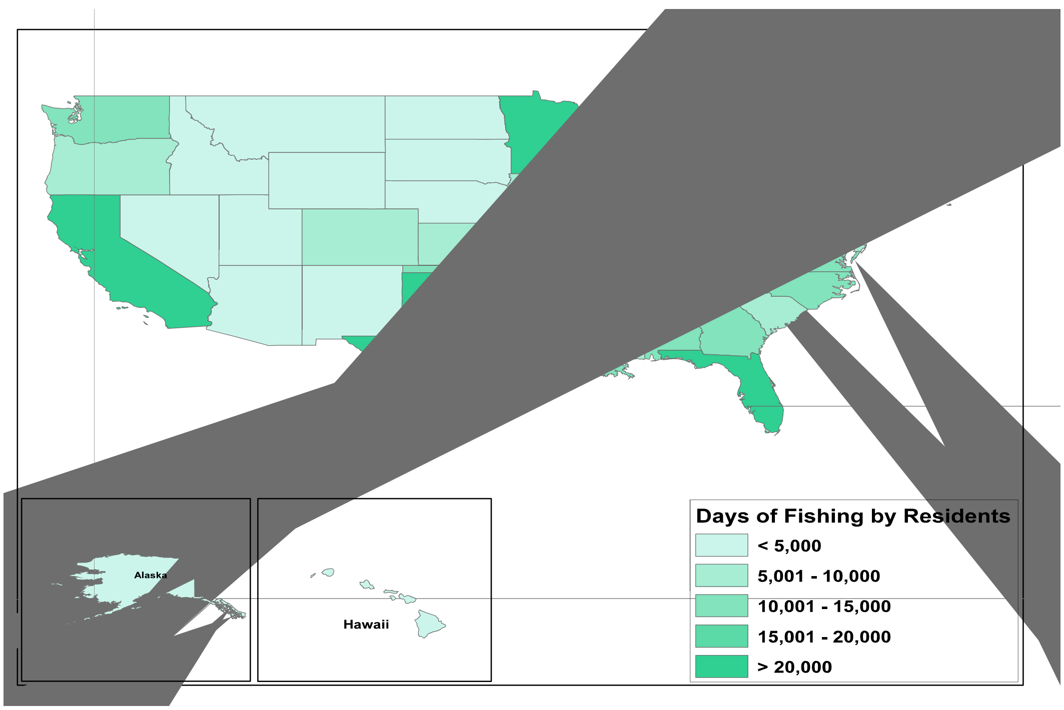


Figure 4-9. 2001 Recreational Fishing Days, State Residents

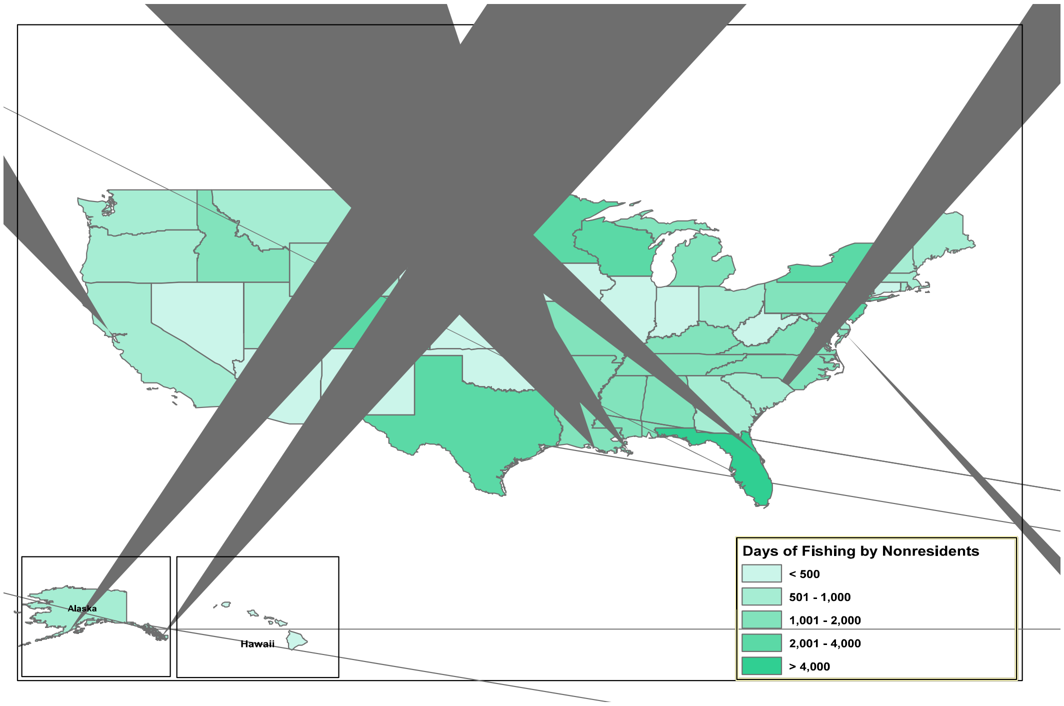


Figure 4-10. 2001 Recreational Fishing Days, Non-State Residents

A number of surveys have been conducted to determine the amount of sport fish consumed by anglers. A discussion of consumption rates for freshwater recreational anglers are provided in Section 4.5 and in Section 11 of this report. Tables 4-25 and 4-26 provide information on bag limits and size limits for recreational fishing.

Table 4-25. Freshwater Fishing Bag and Size Limits for Selected States and Species

Species	California		Texas		Minnesota		Florida	
	Bag Limit	Min. Size (inches)	Bag Limit	Min. Size (inches)	Bag Limit	Min. Size (inches)	Bag Limit	Min. Size (inches)
Black Bass	5-10[a]	12-no limit[a]	5[b]	14[b]	6[b]	none[b]	only 1 fish may be over 22” [b]	12-24[a]
White bass	none	none	25[b]	10[b]	30	none	only 6 fish may be over 24”	none
Striped and Hybrid Bass	2[b]	18[b]	5[b]	18[b]	none	none	only 6 fish may be over 24” [b]	none
Panfish	none	none	none	none	none	none	none[b]	none
Trout	0-5[a]	8-no limit[a]	5[b]	none[b]	Lake: 2[b] Stream: 0-5[a]	none[b]	none	none
Catfish	none	none	channel/blue: 25[b] flathead: 5[b]	channel/blue: 12[b] flathead: 18[b]	5, 1 fish over 24’	none	none	none
Bullhead	none	none	none	none	100	none	none	none
Crappie	25[b]	none[b]	25[b]	10[b]	10[b]	none[b]	none[b]	none[b]

[a]Limit depends on location.
[b]Special limits apply in certain areas.

Table 4-26. Saltwater Fishing Bag and Size Limits for Selected States and Species

	Florida		California		Texas		North Carolina	
Species	**Bag Limit**	**Min. Size (inches)**	**Bag Limit**	**Min. Size (inches)**	**Bag Limit**	**Min. Size (inches)**	**Bag Limit**	**Min. Size (inches)**
Flounder	10	12	none	none	10	14	none	13-14 (TL)[a]
Halibut	none	none	California: 3-5[a] Pacific: 1	California: 22 Pacific: 32	none	none	none	none
Striped Bass	none	none	2	18-no limit[a]	5	18	0-3[a]	18-28 (TL)[a]
Spotted Seatrout	4-5[a], 1 fish over 20”	15	3 (all trout)	none	10, 1 over 25”	15	10	12 (TL)
Bluefish	10	12 (FL)	none	none	none	none	15; only 6 over 24” (coastal waters)	none
Salmon	none	none	2	20-24[a]	none	none	none	none
Mackerel	none	none	none	none	King: 2 Spanish: 15	King: 27 Spanish: 14	King: 3 Spanish: 15 (FL)	King: 24 Spanish: 12 (FL)

[a]Limit depends on location.
[b]Special limits apply in certain areas.
Total length (TL) is measured from tip of snout with mouth closed to tip of compressed tail.
Fork length (FL) is measured from tip of snout to middle of fork in tail.

4.3.3.2 Recreational Fishing Demographics

The USFWS currently maintain statistics on U.S. freshwater angler participation rates in addition to demographic characteristics of U.S. anglers. Angler participation rates are usually measured in number of anglers and the number of angler fishing days for each state (Pechan, 2005). Additional survey data are available that provide demographic information for U.S. anglers. Below are some of the highlights of the demographic statistics followed by a break down of angler participation by region.

Based on USFWS survey data, 67% of recreational fishermen were between the ages of 25 to 54. Male anglers outnumbered women by a ratio of 3 to 1. Freshwater recreational anglers fished in ponds, lakes, and reservoirs over rivers and streams at a ratio of 2 to 1. Table 4-27 shows the percentage of freshwater anglers by age group. Table 4-28 shows the percentage of anglers by sex and age. These data show that women of childbearing age (16-44) make up 60% of female anglers. Female anglers make up 26% of all anglers. Thus, women of childbearing age make up 15.6% of all anglers (i.e. 60% of the 26%).

Table 4-27. Percent of Freshwater Anglers by Age Group

Age Group	Percent of Total Anglers
16-17	4
18-24	9
24-34	19
35-44	27
45-54	20
55-64	12
65-Older	9

Source: USFWS, 2002.

Table 4-28. Percent of Anglers by Sex and Age Group

Age Group	Male	Female
16 to 17 years	4	3
18 to 24 years	7	8
25 to 34 years	19	21
35 to 44 years	26	28
45 to 54 years	20	21
55 to 64 years	13	11
65 years and older	10	8
All Ages	74	26

Source: USFWS, 2002.

USFWS statistics showed that non-white fishermen encompassed a greater portion of saltwater recreational fishermen with an average of 12% compared to comprising only 7% of total freshwater recreational fishermen. From Table 4-29, one can see that the western part of the

United States had the highest participation rate per capita of non-white anglers followed by the Southeast. The lowest participation rates among non-white anglers were in the Great Lakes region and the Northern Plains.

Table 4-29. 2001 Demographic Summary, Angler Race (% Non-White)

Region	Freshwater	Saltwater	Great Lakes
West	12	22	N/A
Southeast	11	9	N/A
South Central	7	7	N/A
Northeast	5	10	3
Great Lakes	4	N/A	11
Northern Plains	3	N/A	0
Average	7	12	5

Source: USFWS, 2002.

According to a *Recreational Boating and Fishing Foundation 2003 Boating and Fishing Attitude, Segmentation Study* (RBFF, 2003) on minorities, Hispanics demonstrated to be more active participants than African Americans, among a group of respondents that were categorized to be active or prospective fishing participants, with 74% showing some incidence of fishing and 13% considered avid fishermen compared to 55% and 7%, respectively, for African Americans. Only 17% of Hispanics had never participated in fishing while 25% of African Americans had not. However, compared to whites, both ethnic groups showed a lower active participation rate (see Table 4-30). It should be noted that Asian fishing participation statistics were not available.

Table 4-30. Incidence of Fishing Among White, African American, and Hispanic Recreational Boaters

Participant Description	White (%)	African American (%)	Hispanic (%)
Active Participant	80	55	74
Avid	18	7	13
Semi-Avid	19	17	14
Occasional	43	31	47
Lapsed (prospective)	11	20	10
Never Participated (prospective)	10	25	17

Source: RBFF, 2003.

USFWS survey data showed fishing participation rate to be positively correlated with household income (see Table 4-31). Household income groups of $40 thousand per year or more had the highest participation rate with over 22% of each income group participating in fishing. Only 8% of households with annual income less than $10 thousand participated in some kind of fishing activity.

Table 4-31. Percent of U.S. Population Who Fished (By Household Income)

Income Group	Percent of Income Group that Fished
Less Than $10,000	8
$10,000 - 19,999	11
$20,000 - 24,999	14
$25,000 - 29,999	16
$30,000 - 34,999	18
$35,000 - 39,999	20
$40,000 - 49,999	22
$50,000 - 74,999	23
$75,000 - 99,999	23
$100,000 or Greater	22

Source: USFWS, 2002.

4.3.4 Total U.S. Demand

NMFS estimated that the total 2002 U.S. commercial finfish and shellfish demand was 66.0 lb/capita (NMFS, 2003b). Of this total, 15.6 lb/capita represented edible fish and shellfish meat. For comparison, Jacobs et al (1998) obtained an estimate of total fish and shellfish consumption of 15.65 grams/person/day (12.6 lb/capita) from the Continuing Survey of Food Intake by Individuals (CSFII) Study. This survey was conducted during the years of 1989-1991. During those years, NMFS estimated an average U.S. consumption rate of 15.2 lb/capita. Therefore, these two sources show reasonably good agreement, although the NMFS estimate does not include recreational demand.

4.3.4.1 Finfish

From Table 4-14, the total 2002 U.S. recreational marine demand was 103,531 metric tons (all finfish). From this value, the total demand for recreationally-harvested finfish is 0.79 lb/capita. From Table 4-20, the total 2002 U.S. commercial finfish demand was estimated to be 33.0 lb/capita. As mentioned earlier in this section, there are no similar data for recreational freshwater finfish demand.

4.3.4.2 Shellfish

From Table 4-21, the total 2002 U.S. commercial shellfish demand was estimated to be 9.07 lb/capita. No data were identified to estimate recreational shellfish demand.

4.4 Economic Value of Key Species

This section presents available information on the economic value of the commercial and recreational fishing industries. Available information on the wholesale and retail pricing of products from the commercial fishing industry are also provided. In other sections, we have discussed the level of production of commercial fish. In this section, we combine production levels with economic value and see that in the commercial fishing industry, shellfish sales total $1.8 million while finfish sales total $1.3 million. Shrimp, crab, and lobster are the top shellfish species sold, while pollock (used in fish sticks) is the top finfish species sold. Domestic tuna sales are lower in sales than several other species (however, the U.S. imports large quantities of tuna).

4.4.1 Finfish

Table 4-32 provides a summary of the economic value of commercial landings for several finfish types. Note that these data exclude aquacultural products, and that for catfish, most of the production of these fish comes from aquaculture. The economic value data are also shown graphically in Figure 4-11 as a percentage of the total 2002 commercial finfish landings ($1,368,877,000). Additional details on the economic value of 2002 commercial landings are provided in a separate report (Pechan, 2005). Table 4-33 provides information on 2001 wholesale pricing of several fresh finfish (NMFS, 2005). Data for 2002 were not available. The NMFS gathers wholesale pricing data from the Fulton Fish Market in New York City. Additional data are provided in Pechan (2005).

Table 4-32. 2002 Economic Value of Commercial Landings for Important Finfish Types

Finfish Type	Economic Value (thousand dollars)	% of Total Finfish Landed by weight
Pollock	209,890	41.4
Menhaden	105,172	21.6
Salmon	156,082	6.94
Cod	126,844	6.70
Hake	26,215	3.86
Sole	22,437	3.18
Herring	21,310	2.70
Sardine	10,824	2.66
Tuna	85,478	0.61
Other Finfish	604,625	10.3
Total 2002 Finfish	**1,368,877**	**100**

Source: NMFS, 2003b.

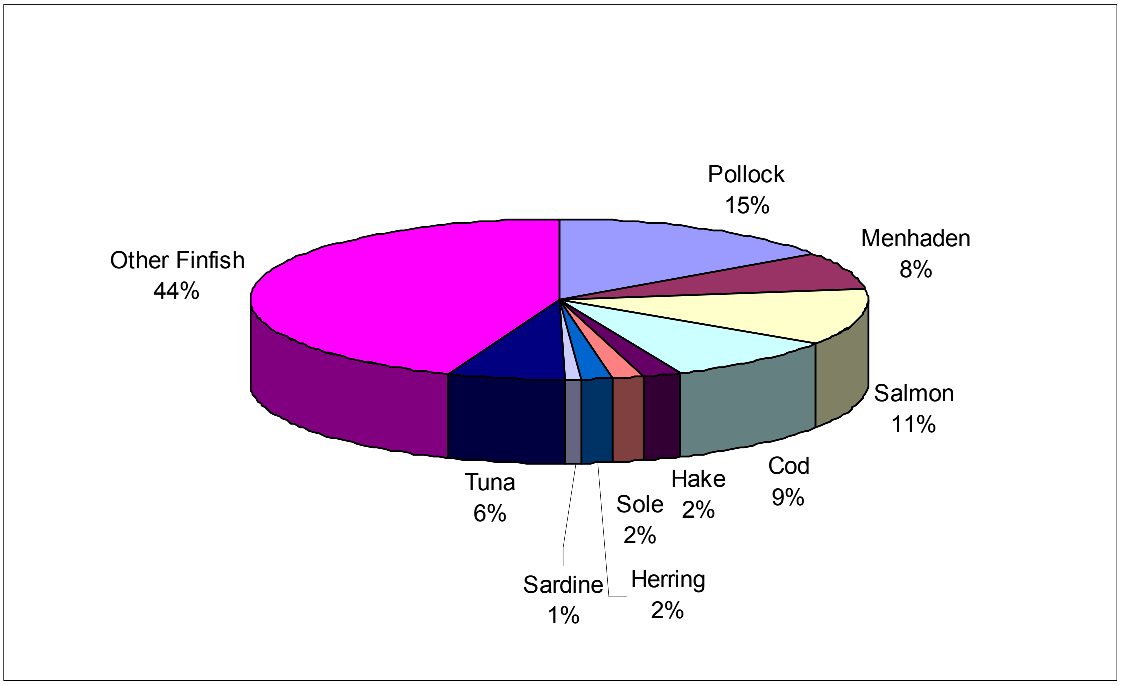


Figure 4-11. 2002 Market Share of Commercial Finfish (% of Total Economic Value)

Table 4-33. Average 2001 Wholesale Prices for Several Fresh Finfish Types

Finfish Type	Price ($/lb)
Pollock	1.45
Flounder	1.95
Swordfish	3.91
Cod	1.77
Whiting	0.59
Croaker	0.28

Source: NMFS, 2005 (Fulton Fish Market).

4.4.2 Shellfish

Table 4-34 provides a summary of the economic value of commercial landings for several shellfish types. The economic value data are also shown in Figure 4-12 as a percentage of the total 2002 commercial shellfish landings ($1,808,167,000). Additional details on the economic value of 2002 commercial landings are provided in Pechan (2005). Table 4-35 provides 2001 wholesale pricing data for several shellfish types as reported by NMFS (NMFS, 2005). Data for 2002 were not available.

Table 4-34. 2002 Economic Value of Commercial Landings of Several Shellfish Types

Shellfish Type	Economic Value (thousand dollars)	% of Total Shellfish Landed by Weight
Shrimp	522,399	25.9
Crab	397,349	23.2
Squid	43,540	15.5
Clam	171,134	9.82
Lobster	316,085	6.56
Scallop	203,494	4.00
Oyster	93,449	2.78
Other Shellfish[a]	60,717	12.2
Total 2002 Shellfish	**1,808,167**	**100**

Source: NMFS, 2003b.

[a] NMFS includes some non-shellfish species in the "Other shellfish" group including seaweed, sponges and turtles (Pechan, 2005).

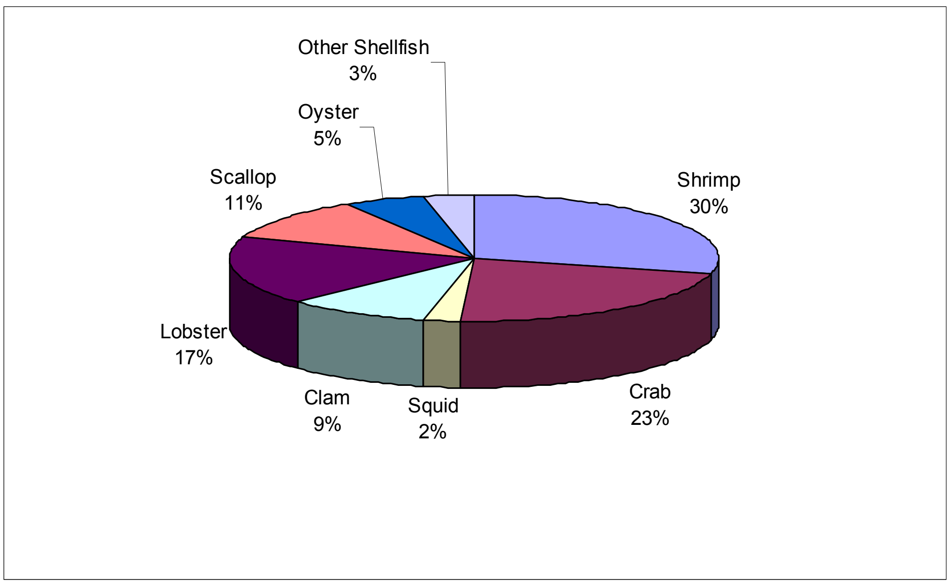


Figure 4-12. 2002 Market Share of Commercial Shellfish (% of Total Economic Value)

Table 4-35. Average 2001 Wholesale Prices for Several Fresh Shellfish Types

Shellfish Type	Price ($/lb)
Crab[a]	48.07
Clam[a]	106.82
Lobster	6.85
Oyster[a]	60.87
Squid	0.92

Source: NMFS, 2005 (Fulton Fish Market).
[a] Pricing is in $/bushel.

4.4.3 Fish Products

Fishery products are processed fish products from commercially caught fish. These products are categorized as: fresh and frozen (fish fillets and steaks, fish sticks and portion, and breaded shrimp); canned products (e.g. tuna, etc.); and industrial fishery products (fish meal and oil). A summary of the 2002 economic value of these products is provided in Table 4-36 below (NMFS, 2003b). In November of 2004, the wholesale price of fish meal was $589/metric ton (NMFS, 2005). The wholesale price of fish oil was $617/metric ton. Wholesale pricing information for other fishery products was not identified.

Table 4-36. 2002 Economic Value of Commercial Fishery Products

Fishery Product	Economic Value (thousand dollars)	Weight (metric tons)
Fresh and Frozen		
Fillets and Steaks	983,900	235,464
Fish Sticks and Portions	288,600	106,777
Breaded Shrimp	475,500	67,360
Canned		
Salmon	295,600	101,470
Sardines	n/a	n/a
Tuna	675,300	248,119
Clams	117,400	63,005
Other	139,600	165,337
Industrial Fishery Products		
Fish Meal	139,700	289,351
Fish Oils	41,400	95,664
Other	78,900	n/a
Total	**3,235,900**	

Source: NMFS, 2003b.

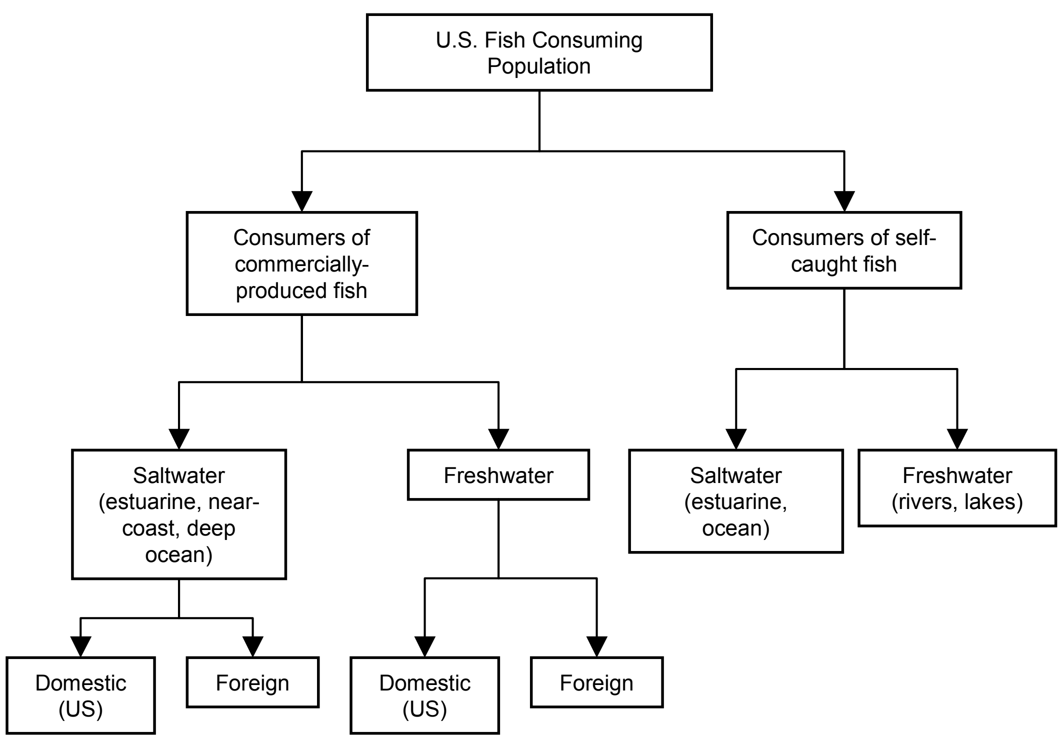
4.5 Characterization of Fish Consuming Populations

This section provides background information on fish consumption in the U.S. including (a) identification of specific fish consumption pathways (e.g., commercial saltwater fish consumption, self-caught freshwater fish consumption), (b) the numbers of individuals potentially exposed through those pathways (i.e., specific fish consuming populations such as recreational freshwater anglers) and (c) a characterization of the general levels of fish consumption associated with those fish consuming populations. In Section 4.5.1, we begin by identifying a variety of fish consumption pathways through which the U.S. population can be exposed to mercury that has bioaccumulated in fish. This section also clarifies the linkage between fish consumption pathways and specific consuming populations (e.g., consumption of self-caught freshwater fish by the freshwater recreational angler population). Section 4.5.2 presents some basic demographic information for the fish consuming populations. Section 4.5.3 presents general fish consumption rate data for the three key populations covered in Section 4.5.2. Note, that Section 4.5.4 is not intended to provide an exhaustive review of published fish consumption data, but rather to provide the reader with a perspective for how consumption rates may differ across pathways and populations. Section 4.5.4 combines the demographic (count) and consumption information to provide a characterization of the populations potential affected by mercury exposures and to help identify those populations that are to be included in the benefits analysis in Section 11 of this report.

4.5.1 Fish Consumption Pathways and Associated Fish-Consuming Populations

Mercury exposure through fish consumption can be considered in terms of specific fish consumption pathways such as self-caught freshwater fish consumption and commercial saltwater fish consumption. A list of fish consumption pathways is presented in Figure 4-13. The commercial fish consumption pathways in Figure 4-13 can be further differentiated as domestic- versus foreign-sourced fish. Note, it is also possible to further differentiate both commercial and self-caught pathways by fish species (e.g., foreign-sourced commercial yellow-fin tuna).

Figure 4-13. Fish Consumption Pathways



Each of the pathways listed in Figure 4-13 can be associated with a specific fish consuming population. For example, commercial saltwater fish is purchased and consumed by a subset of the U.S. population including those individuals that buy this type of fish in foodstores and restaurants. However, this delineation of fish consuming populations according to fish consumption pathway is somewhat artificial since, in reality, most individuals consume a mixture of fish over time reflecting several of these pathways. For example, recreational anglers may fish in both freshwater and saltwater waterbodies and consume commercially-produced freshwater and saltwater fish (e.g., store-bought or restaurant-bought). Therefore, these recreational anglers are exposed to multiple fish consumption pathways simultaneously.

4.5.2 Fish Consuming Populations

In the previous sections, we present total fish produced and U.S. demand for various fish species. This information gives the reader a general characterization of the magnitude of fish consumed in the U.S. (and in relation to the various consumption pathways discussed in this section). In addition, Table 4-37 provides a characterization of the number of people in the population who eat fish from the different consumption pathways. Table 4-37 provides population counts for three of the key fish consuming populations of concern from a benefits standpoint including: (a) the general population who consumes both commercial and (to a lesser extent) self-caught fish, in both cases including a mix of freshwater, saltwater and estuarine species (b) recreational freshwater anglers who consume fish obtained from inland lakes, creeks and rivers and (c) recreational saltwater anglers who fish in the estuarine, near-coastal and open ocean areas.

The *freshwater angler* and *saltwater angler* numbers presented in Table 4-37 include both "total anglers" (number of consuming and non-consuming anglers) and "total consumers"(the number of total "consuming" individuals linked to the recreational self-caught fishing activity, i.e., the total number of individuals including family and friends with which a fisher shares their catch). These two sets of counts (total anglers and total consumers) were calculated separately for freshwater and saltwater recreational categories. The "total anglers" estimates are obtained directly from the National Survey of Fishing, Hunting and Wildlife-Associated Recreation (USFWS, 2002). However, the "total consumers" numbers are calculated by multiplying the total recreational fisher number by (a) the fraction of recreational fishers who consumer their catch (0.84 or 84%) and (b) a factor reflecting the number of family/friends with which the average recreational fisher shares their catch (2.5).[1] For purposes of comparing the number of recreational fishers (freshwater and saltwater) to the number of fish consumers in the general population, it is most appropriate to use the "total consumers" numbers for the recreational groups.

Note that there is overlap in the three populations presented in Table 4-37. For example, the general population includes both the recreational freshwater and saltwater anglers. In addition, there is some degree of overlap between freshwater and saltwater anglers. Each of the

[1] The "fraction of recreational fishers who consume" factor (0.84) was derived by taking the average of "consuming fraction" values presented in three key freshwater studies presented in EPA, 1997 including West et al., 1989, Chemrisk 1991, and West et al., 1993. No comparable "consuming fraction" values were readily available for saltwater anglers, so this freshwater value was applied to the saltwater angler category. The "sharing" factor of 2.5 was obtained from EPA, 1997 and is based on values presented in a number of different fishing activity surveys as documented in EPA, 1997.

three populations in Table 4-37 is further differentiated into: (a) adult consumers and (b) prenatal infants likely exposed to mercury through maternal consumption of mercury-contaminated fish (the target group considered in the benefits analysis).

The purpose, in presenting the data in Table 4-37, is to provide the reader with perspective on the sizes of key fish consuming populations in the U.S.. These data demonstrate that the recreational freshwater fisher population (28 million) is significantly larger than the recreational saltwater population (9 million), while both of these populations are significantly smaller than the general population of fish consumers in the U.S. (184 million) which includes many individuals receiving a significant fraction of their fish from commercially-produced stocks. This suggests that, while the benefits analysis has captured a key fish consuming population in modeling recreational freshwater anglers, a potentially large group of individuals (general consumers and recreational saltwater anglers) are not included in the primary benefits analysis (this issue is discussed in greater detail in Section 4.5.4.1).

4.5.3 General Fish Consumption Rates for Key Fish Consuming Populations

In presenting perspective on the degree of potential exposure to mercury for different fish consuming populations in the U.S., in additional to considering the total number of individuals within each population (information presented in Section 4.5.4), it is also important to consider fish consumption rates for those populations. Table 4-38 presents general fish consumption rates (including mean and high-end estimates where available) for key fish consuming populations including: (a) the general population exposed through commercial and self-caught fish, (b) recreational freshwater anglers and (c) recreational saltwater anglers. Due to limitations in available data, it was not possible to identify consumption rates for the full set of fish consumption pathways identified in Figure 4-13. However, the values presented in Table 4-38 do provide coverage for key populations. All of the values presented Table 4-38 represent long-term dietary consumption (i.e., annual-averaged daily intake rates), thereby allowing comparison across pathways/sub-populations (additional characteristics relevant to this discussion are also included in Table 4-38 as part of the Comments section).

Table 4-37. Demographic (count) Data for Key Fish Consuming Populations in the U.S.

Population	Population Count	Comments	References
General population (including consumption of commercial and self-caught fish including saltwater, freshwater and estuarine species)			
Adult fish consumers (>18yrs)	184,000,000	88% of adults surveyed reported consuming fish or shellfish at least once within the last month. Fish consumption likely includes commercial saltwater and freshwater as well as self-caught freshwater and saltwater for some fraction of respondents.	Percent fish consumption obtained from NHANES III as summarized in EPA, 1997. Demographic count for adults obtained from US Census 2000.
Female adult (15-44 yrs) fish consumers	53,000,000	86% of females 15-44 yrs (general fertility range) surveyed reported consuming fish or shellfish within the last month. Fish consumption likely includes commercial saltwater and freshwater as well as self-caught freshwater and saltwater for some fraction of respondents.	Percent fish consumption obtained from NHANES III as summarized in EPA, 1997. Demographic count for females 15-44 yrs obtained from US Census 2000.
Infants born to mothers who consume fish in the general population	3,430,000	Developed by applying the general U.S. fertility rate for 2000 (64.8 births per 1000 females 15-44 yrs old) to the number of adult females aged 15-44yrs.	Fertility rate data from US Census 2000.
Freshwater anglers (self-caught freshwater fish consumption only)			
Total anglers (>15 yrs)	27,900,000	Includes total number of anglers fishing in freshwater water bodies including streams, rivers and lakes (excludes Great Lakes). Includes consumers and non-consumers (i.e., catch and release)	National Survey of Fishing, Hunting and Wildlife-Associated Recreation (USFWS, 2002)
Total consumers of recreationally -caught freshwater fish (all ages)	58,590,000	Based on application of "consuming" factor and "sharing" factor to the total anglers estimate above (see text for details)	
Infants born to mothers who consume recreationally-caught freshwater fish	420,000 to 580,000	Modeled as part of the mercury benefits analysis conducted for recreational freshwater anglers	Estimate generated using USFWS, 2002 data, combined with US Census 2000 data (see Section 11 for additional details).
Saltwater anglers (self-caught saltwater and estuarine fish consumption only)			
Saltwater angler (self-caught saltwater and estuarine fish consumption)	9,100,000	Includes consumers and non-consumers (i.e., catch and release)	National Survey of Fishing, Hunting and Wildlife-Associated Recreation (USFWS, 2002)
Total consumers of recreationally-caught saltwater fish (all ages)	19,110,000	Based on application of "consuming" factor and "sharing" factor to the total anglers estimate above (see text for details)	
Infants born to mothers who consume recreationally-caught saltwater fish	135,000 to 186,000	Estimated using the applicable ratio (between prenatally-exposed infants and anglers) developed for freshwater recreational anglers (see above)	

Table 4-38 Fish Consumption Rates for Key Fish Consuming Populations in the U.S.

Population	Fish Consumption (long-term annual-average equivalent in g/day)	Comments	References
General population (including consumption of commercial and self-caught fish including saltwater, freshwater and estuarine species)			
General population fish consumers (all ages combined)	- freshwater/ estuarine: 6.0 (mean) - saltwater: 14.1 (mean) - total fish: 20.1 (mean); 60.3 (95^{th}%)	- uncooked (but does seem to represent edible portion to some extent) - total population (consumers and non-consumers in survey)	EPA, 1996a (CSF II, 1989-1991) (as reported in EPA, 1997)
General population adults (>18 yrs)	- freshwater/ estuarine: 5.6 (mean) - saltwater: 12.4 (mean) - total fish: 18.0 (mean)	- as consumed - total population (consumers and non-consumers in survey)	EPA, 1996a (CSF II, 1989-1991) (as reported in EPA, 1997)
General population females (15-44 yrs)	- freshwater/estuarine: 4.3 (mean) - saltwater: 10.0 (mean) - total fish: 14.3 (mean)	- as consumed - total population (consumers and non-consumers in survey)	EPA, 1996a (CSF II, 1989-1991) (as reported in EPA, 1997)
Freshwater anglers (self-caught freshwater fish consumption only)			
Anglers (all ages combined)	- Maine 5 (mean); 13 (95th%) - New York 5 (mean); 18 (95th%) - Michigan 12 (mean); 39 (96th%) - Michigan 17 (mean) - EPA "recommended" freshwater fish consumption rate: 8 (mean); 25 (95th%)	- consumers plus non-consumers (catch and release) - uncooked (but does seem to represent edible portions)	- Ebert et al., 1992 - Connelly et al., 1987 - West et al., 1989 - West et al., 1993 (all as reported in EPA, 1997)
Saltwater anglers (self-caught saltwater and estuarine fish consumption only)			
Anglers and consumers of fish caught by anglers (e.g., family members) (all ages combined)	- Atlantic: 5.6 (mean); 18.0 (95th%) - Pacific: 2.0 (mean); 6.8 (95th%) - Gulf: 7.2 (mean); 26.0 (95th%)	- edible fraction (uncooked) - consumers and non-consumers	NMFS, 1993 (as reported in EPA, 1997)

Review of the values presented in Table 4-38 reveals some interesting comparisons and contrasts between consumption rates for different populations:

- The general population has greater average consumption rates for saltwater fish 14.1 g/day) than do the saltwater anglers (2.0 to 7.2 g/day depending on region). It is important to point out that the saltwater angler values refer to self-caught fish only. It is likely that they also consume some amount of commercially-produced saltwater fish, which would mean that in terms of total saltwater fish, the saltwater anglers might have higher consumption than the general population. However, in comparing self-caught saltwater fish consumption to general population saltwater fish consumption, the latter is a larger value on average.

- Self-caught freshwater fish consumption (5 to17 g/day depending on region) is larger, on average, than self-caught saltwater fish consumption (2 to 7 g/day depending on region). These data suggest that freshwater anglers may, on average, have twice the intake of self-caught fish than saltwater anglers.

- Regional data on consumption rates suggests that populations can differ significantly in their fish consumption depending on where they are located. This can have important implications for modeling distributional benefits for fish consumption as part of an equity analysis. If fish consumption rates differ significantly across regions, then this may suggest that exposure through fish consumption may also differ regionally (of course this will also depend on the regional variability of mercury concentrations in fish consumed). In considering distributional equity, it may be necessary to conduct more refined exposure modeling that tracks these different patterns of fish consumption regionally (and links them to spatial distribution of fishers and mercury fish tissue concentrations). However, in reality, the patchiness of data characterizing regional variability in fish consumption rates tends to prevent a comprehensive treatment of this issue in the context of a national-scale benefits analysis. Instead, individual case studies focusing on specific regions and assessing the potential importance of regional variability in factors such as fish consumption rates and the spatial distribution of fishing populations can be conducted.

4.5.4 Discussion of Population and Fish Consumption Data in the Context of the Mercury Benefits Analysis

Information presented in Section 4.5.4 can be used to gain a perspective on the degree to which the benefits analysis presented in this RIA has covered key fish consuming populations in the U.S. from the standpoint of potential benefits linked to mercury emissions reductions. Specifically, the recreational freshwater angler population modeled for the benefit analysis can be compared, in terms of total fish consumption, to the other key populations (i.e., recreational saltwater anglers and the general U.S. fish consuming population).

Table 4-39 presents total fish consumption estimates for the three key populations covered in Sections 4.5.2 and 4.5.3 (total consumers associated with recreational saltwater angler activity, total consumers associated with recreational freshwater angler activity and the general fish consuming population in the U.S.). Note, for the recreational angling scenarios, the total consumer categories were used in conducting this comparison rather than the total angler categories, since the former focus on the total number of individuals consuming fish caught by the anglers. These estimates were generated by multiplying average (mean) fish consumption values presented in Table 4-38 by the total demographic counts for these populations presented in Table 4-37. While this approach is relatively simplistic and is subject to uncertainty, it is considered sufficient to provide a general perspective on the magnitude of differences in total fish consumption by the three populations.

Table 4-39. Total Fish Consumption for Recreational Saltwater Anglers, Recreational Freshwater Anglers and General U.S. Fish Consuming Population

Population (adults)	Population count	Consumption Rate (g/day)	Total fish consumption (kg/year)
General fish consuming population	184,000,000	20.1	1,349,916,000
Recreational freshwater anglers	58,590,000	8	171,082,800 (13% of general population consumption value)
Recreational saltwater anglers	19,110,000	4.9*	34,178,235 (3% of general population consumption value)

*The consumption rate presented for recreational saltwater anglers was derived by taking the average of the regional values.

Total fish consumption values presented in Table 4-39 highlight the fact that the primary benefits analysis for this RIA is capturing a relatively small fraction of overall fish consumption. Specifically, the recreational freshwater angler population modeled for the primary benefits estimates represents only 13% of total fish consumption in the U.S. (comparing self-caught freshwater fish consumption by recreational anglers to consumption of all fish types by the general population). It is important to note, that the actual magnitude of IQ benefits for a given population is a function of (a) that population's fish consumption rate, (b) the baseline fish tissue concentrations for fish that the population consumes and (c) the magnitude of changes in mercury deposition to the waterbodies containing fish that a given population catches and the relationship between those deposition changes and mercury fish tissue concentrations. In short, the likely difference in IQ benefits between the three populations presented in Table 4-39 is dependent on several key factors related to mercury concentration in fish in addition to total fish consumption rates for these populations. While it is still informative to consider that approximately 86% of fish consumption by the U.S. population is not being covered in the benefit analysis, Section 8 of this RIA shows that deposition to U.S. waterbodies from coal-fired power plants will predominently occur in freshwater waterbodies in the Eastern-half of the U.S. Thus, the benfeit analysis of freshwater recreational anglers captures the primary segment of the affected population. To the extent that CAMR reductions will impact fish in coastal regions and the ocean, there remains a potential for a small amount of additional IQ benefits related to the general population of fish consumers (including foreign and domestically-caught commercial fish and coastal recreationally-caught fish).

4.5.4.1 Potentially High-Exposure Subpopulations

The primary benefits analysis includes consideration for several potentially high-exposure subpopulations including:

- A high fish consumption rate study population that is defined as "subsistence" fishers for the purposes of this study;

- A low-income high fish consumption study population (an alternative approach to model subsistence fishers);

- A Southeast Asian ethnic group with high freshwater fish consumption due to cultural practices; and

- A Native American population with high freshwater fish consumption due to cultural practices.

While these special subpopulations provide important insights into the issue of distributional (equity) benefits (i.e., the potential for IQ benefits to be disproportionately distributed across the U.S. population), they do not represent a significant (net) fraction of overall benefits due to the size of the subpopulations relative to overall U.S. fish consumption. In the case of the high fish consumption (subsistence) population (first bullet above), this group is a subset of the larger recreational freshwater fisher population and therefore, is incorporated as part of the recreational (self-caught) freshwater angler analysis in Section 10.

Because these subpopulations are included primarily to provide insights into potential distributional (equity) issues, and do not contribute a significant fraction to net benefits, they are not discussed in detail in this section, which is primary concerned with provide the reader with perspective on the magnitude of total U.S. population consumption through various fish consumption pathways.

4.6 Summary

Because fish consumption is the primary pathway for exposure to methylmercury, this section provides background information on fishing activity through a profile of the fishing industry in the United States. Methylmercury exposure through fish consumption is considered in terms of specific fish consumption pathways such as commercial and self-caught (recreational) fish consumption, which is composed of fish species from freshwater and saltwater sources.

4.6.1 Commercial Fish Production, Demand, and Consumption

Fish products from commercial fishing activities include: fresh and frozen (fish fillets and steaks, fish sticks and portion, and breaded shrimp); canned products (e.g. tuna, etc.); and industrial fishery products (fish meal and oil).

The amount of commercial fish landed in inland waters and state and territorial seas (0 - 3 miles from shore) is estimated to be 36% of the total commercial landings, while 61% of commercial fish were landed in the Federal exclusive economic zone (3 - 200 miles offshore). The Pacific coast region accounted for 65% of total commercial landings with Alaska and California accounting for the largest portion. Alaska alone accounted for over 26% of total U.S. landings. The Gulf Coast accounts had significantly less landings than the Pacific Coast at 18%, followed by New England and the Chesapeake Bay contributing 6% and 5% to U.S. landings, respectively.

Total commercial finfish landings equal 3.6 million metric tons. Commercial shellfish landings total approximately 600,000 metric tons. Pollock and Menhaden were by far the finfish types with the highest commercial landings by U.S. craft in 2002. These two finfish types are almost exclusively caught by commercial fishermen. Pollock are often used in making frozen fish products (e.g., fish sticks), surimi (see Section 4.2.1.3), and other minced fish products. Menhaden are often used to produce cut or live bait, fishmeal and fish oil. Shrimp and crab are the top species of shellfish caught commercially.

Total aquacultural production from fish farms in 2002 was 393 thousand metric tons with catfish accounting for 73% or 286 thousand metric tons, followed by trout at 6% of total aquaculture production.

The U.S. exports total 1.09 million metric tons of commercial fish, with 56% of exports going to Asia, followed by 20% to Europe and 18% to North America. Total edible imports for 2002 were 4.0 million metric tons with finfish contributing 65% of the total. An estimated 46% of U.S. imports were from Asia while 26% were from other regions in North America.

NMFS estimated that the total 2002 U.S. commercial finfish and shellfish per capita demand was 66.0 lb/capita (NMFS, 2003b). Of this total, 15.6 lb/capita represented edible fish and shellfish meat. Most of the U.S. tuna demand is met through imports (less than 7% of the tuna demand is met through commercial landings by the domestic fleet).

4.6.2 Recreational Fishing Activity, and Consumption

Recreational fishing activity occurs in saltwater (estuaries, coastal regions, and open ocean) and freshwater locations (lakes, rivers, and streams). Roughly 32% of the 103,529 metric tons of marine finfish landed from recreational angling in U.S. waters in 2002 occurred at distances of 3-200 miles from shore, while 29% of total finfish landings were within 3 miles from shore. The remaining 39% of finfish were landed in Inland Waters. For shellfish, 65% of the 534,608 metric tons landed were made within Inland Waters and State and Territorial Sea (< 3 miles from shore).

Data for freshwater recreational harvest are not available; however, the number of anglers and number of days of fishing is available from the 2001 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (USFWS, 2002). There are approximately 28 million freshwater recreational anglers and 9 million saltwater recreational anglers. The region with the largest number of anglers was the southeast followed by the south central United States. Bass (large and small mouth), pan fish, trout, catfish, and crappie were the most popular target species for freshwater anglers nationwide. Considering the number of days spent fishing, high levels of recreational fishing activity are shown to occur in the Great Lakes States, Florida, New York, Texas, and California.

The total demand for recreationally-harvested finfish is 0.79 lb/capita. From Table 4-31, the total 2002 U.S. commercial finfish demand was estimated to be 33.0 lb/capita. The total 2002 U.S. commercial shellfish demand was estimated to be 9.07 lb/capita. No data were identified to estimate recreational shellfish demand.

4.6.3 Overall Conclusions

This section demonstrates that the recreational freshwater fisher population (28 million) is significantly larger than the recreational saltwater population (9 million), while both of these populations are significantly smaller than the general population of fish consumers in the U.S. (184 million) which includes many individuals receiving a significant fraction of their fish from commercially-produced stocks.

Based on information provided in this section, we see that commercial fish consumption constitutes a large portion of exposure to methylmercury. However, a large majority of the commercial fish consumed are imported from foreign sources, or 3-200 miles offshore by domestic commericial fishermen (with a majority of domestic landings occuring off the Pacific coast). These sources of exposure are not likely to be impacted by the control of utilities from the CAMR rule. However, methylmercury concentrations from freshwater sources are likely to be affected by control domestic electric utilities. Therefore, the quantified benefit analysis in Section 10 evaluates the benefits of improved health from reduced exposure to methylmercury from recreational freshwater fishing activities.

4.7 References

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Tables

Figures

SECTION 5

MERCURY CONCENTRATIONS IN FISH

Because fish consumption is a major pathway for human exposure to methylmercury, it is important to determine the methylmercury concentrations of consumable fish. A baseline concentration of consumable fish will enable EPA to determine the reduction in exposure related to a reduction in Hg in fish tissue resulting from reduction in air deposition of Hg.

As is discussed in Section 4, the fish that Americans consume come from a variety of sources including saltwater species from the ocean and estuaries, and freshwater species from lakes, rivers, and streams. Fish may be purchased from commercial sources, or caught recreationally and consumed. In this section, we describe the level of methylmercury contamination in shellfish (saltwater species) and finfish (both saltwater and freshwater species).

5.1 Methylmercury Concentrations in Saltwater Fish Species

The EPA's Office of Water has developed the Mercury in Marine Life database, which provides information on the level of methylmercury contamination in estuarine and marine species (i.e., commercial and non-commercial seafood)[1]. The Mercury in Marine Life database contains over 15,000 records on methylmercury tissue concentrations in approximately 250 different fish and shellfish species. The geographic coverage includes data from all 24 coastal states, the District of Columbia and Puerto Rico. Data was not evenly distributed by coast. More tissue samples were taken from the Gulf of Mexico than either the Atlantic or Pacific Ocean.

The average methylmercury concentration (i.e., mean in ppm) for several species in the database are presented in Table 5-1. Not surprisingly, many of the species groups with high methylmercury concentrations are top-level predators. This was expected considering methyl mercury's tendency to bioaccumulate up the food chain. Twenty-six percent of the samples contained total mercury concentrations above 0.3 ppm. Five percent of the samples contained total methylmercury tissue concentrations above 1.0 ppm, the FDA action level for issuance of a fish advisory. King mackerel and a number of shark species contain the highest means. Other species with relatively high concentrations include barracuda, jack crevalle, Spanish mackerel, ladyfish, and seatrout.

[1] As discussed in ch. 3, samples are Hg, which are used as a proxy for MeHg.

Table 5-1. Concentrations of Mercury in Marine Life

Species Group	Count	Mean (ppm)	Range (ppm)	St Dev
Marlins	11	2.424	0.270 - 6.80	1.838
Mackerels	849	0.791	0.013 - 4.47	0.703
Sharks	620	0.776	0.020 - 6.90	0.729
Barracudas	54	0.757	0.076 - 3.10	0.642
Tunas	34	0.647	0.071 - 1.57	0.421
Jacks	486	0.509	0.017 - 3.90	0.453
Ladyfish	194	0.479	0.020 - 2.60	0.408
Groupers	229	0.473	0.045 - 3.30	0.418
Bluefish	289	0.445	0.020 - 2.00	0.331
Snook	496	0.398	0.030 - 2.08	0.315
Drums	2544	0.382	0.001 - 6.62	0.499
Sea basses	57	0.352	0.020 - 1.32	0.343
Rockfish	315	0.294	0.004 - 1.44	0.221
Catfishes	385	0.245	0.000 - 1.80	0.305
Wrasses	54	0.234	0.070 - 0.75	0.149
Temperate basses	625	0.226	0.000 - 1.25	0.211
Crabs	385	0.224	0.001 - 3.68	0.427
Eels	133	0.223	0.000 - 0.80	0.154
Snappers	380	0.218	0.004 - 2.80	0.182
Sturgeons	10	0.216	0.120 - 0.35	0.07
Rays	104	0.197	0.013 - 0.91	0.151
Tripletails	110	0.192	0.014 - 1.28	0.187
Porgies	292	0.178	0.001 - 1.73	0.191
Grunts	67	0.177	0.020 - 0.66	0.116
Kingfish	86	0.167	0.017 - 0.78	0.147
Lobsters	18	0.127	0.050 - 0.25	0.067
Flounders	1006	0.117	0.001 - 1.70	0.152
Dolphin Fish	44	0.107	0.031 - 0.49	0.106
Croakers	425	0.098	0.000 - 1.10	0.124
Oysters	2212	0.073	0.002 - 3.91	0.121
Mullets	203	0.065	0.001 - 1.14	0.121
Salmon	300	0.065	0.015 - 0.61	0.051
Mussels	1157	0.059	0.002 - 0.93	0.077
Shrimp	171	0.051	0.000 - 1.02	0.094
Clams	51	0.049	0.008 - 0.12	0.022
Cods	150	0.045	0.007 - 0.26	0.045
Herring	186	0.043	0.010 - 0.24	0.038
Surfperches	391	0.041	0.002 - 0.26	0.039

Source: U.S. EPA, Office of Water; Mercury in Marine Life Database.

5.2 Methylmercury in Freshwater Fish Species

The most extensive source of national-level monitored mercury data for freshwater fish is the National Listing of Fish and Wildlife Advisories (NLFA) maintained by EPA's Office of Water. The NLFA includes more than 91,500 samples of fish tissue contaminant data collected by states, Native American tribal governments, territories, and Canada (and submitted to EPA) from over 10,700 locations nationwide from 1967 through 2003. Figure 5-1 shows the continental locations of Hg fish tissue samples recorded in the NLFA to date. In some watersheds[2], freshwater fish were sampled hundreds or even thousands of times, but most areas have a handful of samples. Watersheds without samples are typically found in the West. Table 5-2 shows the frequency at which watersheds were sampled.

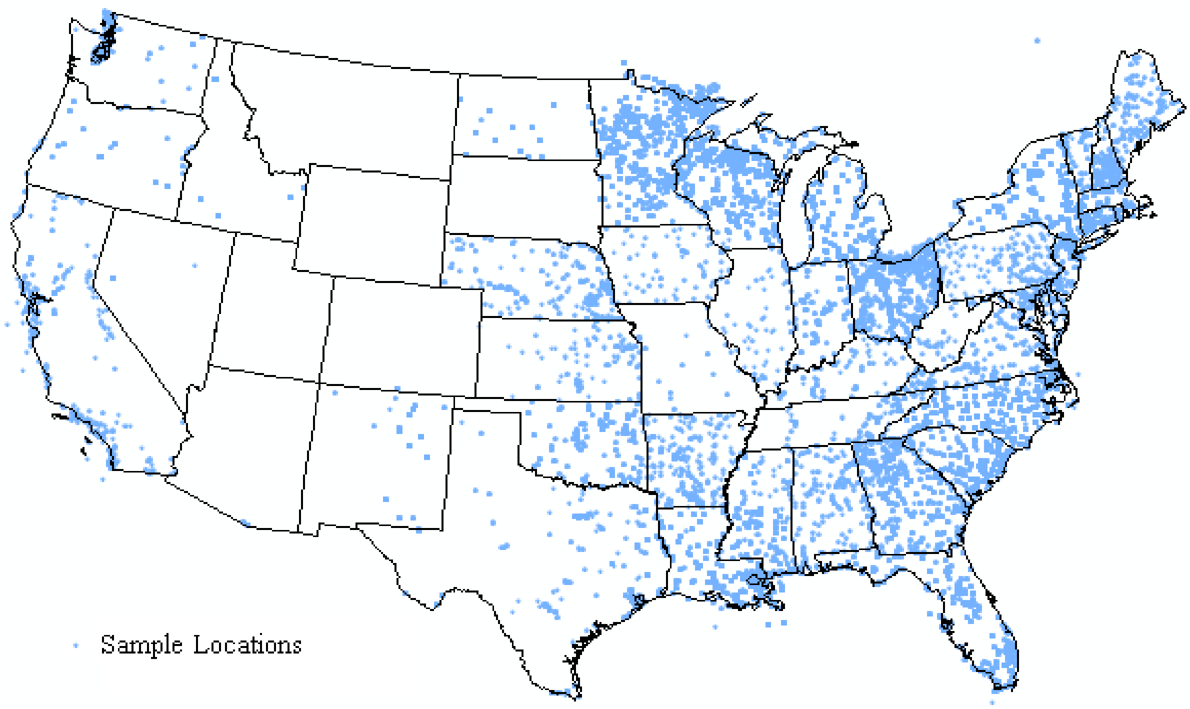

Figure 5-1. NLFA Sample Locations

Table 5-2. Number of Fish Tissue Samples / Watershed From the National Listing of Fish Advisories

Number of Samples	Number of Watersheds
No samples	1069
1 - 4 samples	237
5 - 49 samples	547
50 - 99 samples	126
100 - 499 samples	157
500 - 999 samples	13
1,000 or more samples	3

The average watershed Hg fish tissue concentration is .29 ppm., and samples within a watershed are typically within .81 ppm. of each other. Between watersheds, average watershed concentrations range from .001 ppm. to over 4 ppm. The frequency distribution shown in Figure 5-2 illustrates the average concentration found in watersheds.

[2] A watershed is defined as a USGS 8-digit HUC. HUCs are discussed later in this chapter.

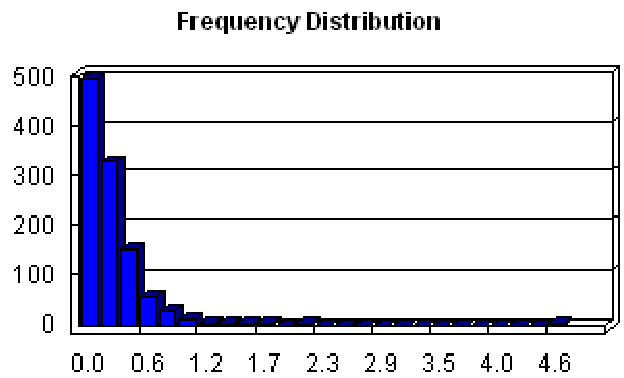

Figure 5-2. Frequency Distribution of Average Watershed Fish Tissue Concentrations (ppm)

In the NLFA dataset, each sample is described according to the sample location, sample date, measured methylmercury concentration, species and size of fish, and the part of the fish sampled. Each of these elements in the data result in large variation across the NLFA samples.

5.2.1 Sources of Variability in Hg within the NLFA

Variation Across Sample Methods

Different states have used various sampling methods (cuts of fish) over the years to determine fish tissue concentrations. Figure 5-3 shows the frequency various sampling methods were employed in the NLFA, and the average fish tissue concentration of Hg associated with each sampling method.

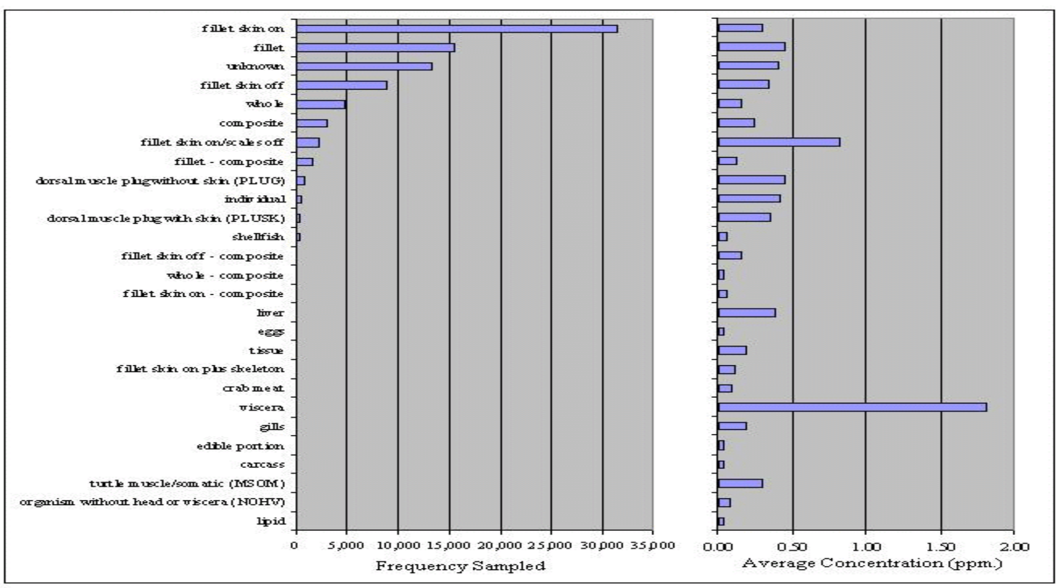

Figure 5-3. Frequency and Average Concentrations of Various NLFA Sample Methods (Cuts of Fish)

Variation Across Species

There are close to 400 different species of fish sampled in the NLFA. Average concentrations for freshwater species, where samples were geocoded, range from .007 to 1.85 ppm The mean average concentration for a given species (freshwater or saltwater) is .25, with a standard deviation of .43.

*Variation Across Lake/River/Other Types of Ecosystem*s

About 70% of the samples have geographic coordinates associated with them. Where samples were geocoded (i.e. a latitude and longitude were recorded based on a written site description), it was possible to map the sample location. Mapping sample locations allowed the EPA to classify locations as either lake or river. Locations closer to known non-flowing waterbodies were assigned as type "lake", and locations closer to known flowing waterbodies were assigned as type "river". Locations that did not have geographic coordinates associated with them were mostly saltwater fish species, but are considered unclassified. On average, river environments were .017 ppm higher in fish tissue concentrations than lake environments, however river environments typically contained more variability in concentrations. Table 5-3 provides details related to fish tissue samples from unclassified, river and lake environments.

Table 5-3. Hg Fish Tissue Concentrations From Various Environments (ppm)

Type	Number of Samples	Minimum	Maximum	Average	Standard Deviation	Variance
unclassified	40,965	0.00	29.00	0.3455	0.4827	0.2330
lakes	17,623	0.00	7.59	0.3438	0.3519	0.1238
rivers	33,048	0.00	8.94	0.3614	0.4386	0.1924

Other Potential Sources of Variability in NLFA Hg samples

Sometimes, the same location was sampled multiple times over the span of a few months or even years. On average, most locations were sampled about 8 times, and same-location repeat samples typically range within .4 ppm of each other.

Sampling efforts over time are not evenly spatially distributed. There is fairly evenly distributed sampling efforts in the first half of the 1990's, but the late 1990's and early 2000's show much heavier sampling in the north and southeast. There are some states that submitted samples to the NLFA for some years, and not for others.

The NLFA Hg fish tissue concentrations represent total Hg concentration from all sources. The state of South Carolina is of particular interest because concentrations are relatively high. This could be due to the historic use of mercury in gold mining that took place in the middle part of the state.

It is possible that Hg fish tissue concentrations are changing over time, but due to these other sources of variability, and inconsistent spatial patterns of sampling, it is not possible to determine if indeed these temporal trends are present, or the strength of their influence.

5.2.2 National Lake Fish Tissue Study

In addition to the NLFA, EPA's National Lake Fish Tissue Survey (NLFTS) also provides useful data on concentrations of Hg in fish tissue. Conducted in 1999-2003, this study sampled fish tissue from 500 randomly selected lakes and reservoirs across the U.S. (from the estimated 270,000 lakes and reservoirs in the lower 48 States). Figure 5-4 shows locations of NLFTS sample sites.

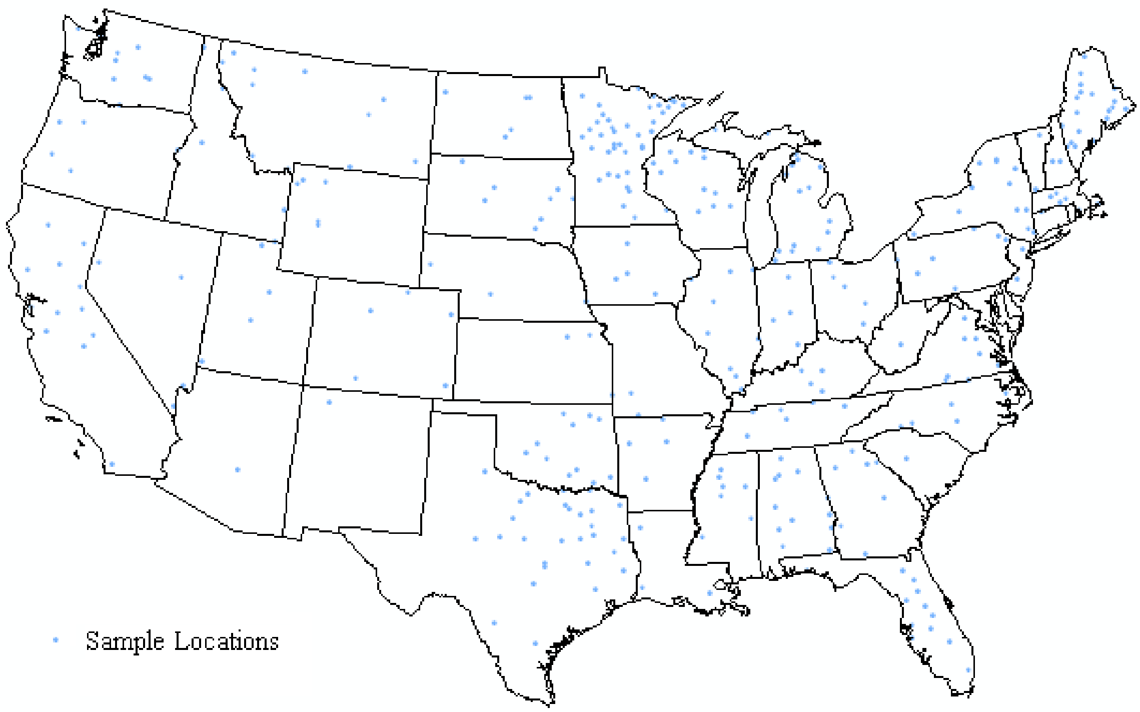

Figure 5-4. Sample Locations from the NLFTS

In comparison to the NLFA, the design of the NLFTS provides more evenly spatially distributed sampling sites across the United States, but are much fewer in number than the NLFA. The NLFTS was statistically designed to be representative and sampled 54 different species of fish using same-species composite sample methods[3]. For each lake sampled, one composite sample of a predatory species, and one of a bottom-dwelling species were taken.

Some of the more commonly sampled species in the NLFTS are Largemouth Bass, Carp, Catfish, Trout, and Walleye. The average length of all the species used in each same-species composite sample is also recorded. Samples were collected from mid-1999 through 2003. Hg concentrations in the NLFTS range from .004 to 1.46 ppm The average measured NLFTS Hg fish tissue concentration is .22 ppm

[3] A same-species composite sample is where several fish of the same species are used together to determine the methyl mercury concentration.

5.3 Comparison of the Differences in the NLFA and the NLFTS

Of the two major sources of Hg fish tissue concentration data available (the NLFA and NLFTS), each has it's strengths and weaknesses. The NLFA contains a large number of samples, but could be biased given the purpose for which it was developed (fishing advisories). The potential for bias arises because NLFA samples typically collected either at sites that are known to be popular fishing locations, or because sites are suspected of having elevated levels of Hg. The NLFTS is known to be unbiased (because sample locations were selected based on a stratified random sample), but has comparatively fewer samples from fewer locations. Therefore, it is important to compare the two datasets to determine the presence or absence and significance of the suspected upward NLFA bias

Our initial data investigations of the NLFA and NLFTS indicated that fish tissue concentrations varied according to the species sampled, length of the fish sampled, and the sample method. A straight comparison of the NLFA and NLFTS would be inappropriate given these other sources of variability in fish tissue concentration. To control for these sources of variability for a comparison of the data sets, both of the data sets were normalized using the National Descriptive Model of Mercury and Fish (NDMMF), which is discussed in detail in section 5.5. The NDMMF is a statistical model that normalizes Hg fish tissue concentration data to control for species/size/sample method variability[4]. Once the data has been normalized to account for these factors, the outputs of the model are then compared by source of data input (NLFA vs. NLFTS)

The EPA's Office of Water conducted an analysis to determine if the NLFA and NLFTS provide substantially different estimates of fish tissue concentrations. The purpose of the comparison study was to determine if there is a visible and/or statistically significant bias in the NLFA data relative to that in the NLFTS.

Since concentrations of methyl mercury in fish may be different in rivers than in lakes, and the NLFTS study is just of lakes, the lake subset of NLFA normalized data was selected for use in the comparison. Where multiple regression estimates were available for the same lake, the normalized concentrations were averaged. Cumulative Distribution Function (CDF) estimates were generated using the statistical software R v. 1.9.1 (Venables and Smith, 2004) with Probability Survey Data Analysis Functions (psurvey.analysis v. 2.2) (Kincaid, 2004) module. CDF estimates were generated for both the NLFA and NLFTS. Stratified random sample weight adjustments were incorporated in the CDF estimates for the NLFTS data.

[4] The NDMMF is a statistical normalization technique that is specifically designed for Hg fish tissue concentrations. In this sense, the NDMMF is a non-traditional statistical normalization technique.

Figure 5-5 shows the CDF comparison plot for the NLFA and NLFTS normalized lake data sets as continuous functions by fish tissue methyl mercury concentrations, with 95th percentile limits for the NLFTS[5]. A surprising result that is apparent in the figure, is that there is little difference between the two distributions for lake fish. However, the upper end of the distribution (e.g. 95th percentile) shows an upward bias (of 0.14 ppm) in NLFA with respect to NLFTS. Differences at other points in the distribution (5th, 25th, 50th, and 75th percentiles) are quite small (range from -0.017 ppm to 0.025 ppm). Four statistical CDF comparison tests were performed on the two data sets. All four tests indicate the two CDFs can not be considered statistically different at the 95% level (p values > 0.17). A z-test for difference in the means of the two data sets indicated they are not different, at the 95% level.

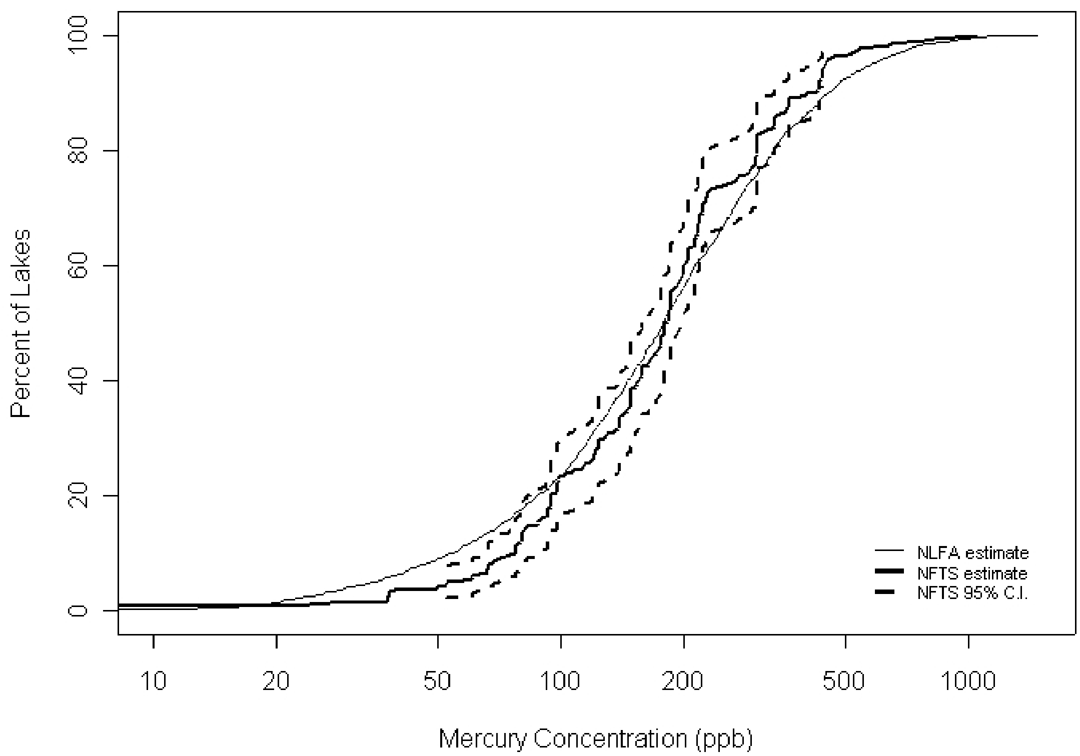

Figure 5-5. Cumulative Distribution Functions (CDFs) for Normalized NLFTS and NLFA Lake Data.

The overall conclusion is that the two data sets, for NLFA and NLFTS lakes, are not statistically different though there is a clear upward bias at the very upper end of the NLFA concentration distribution.

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[5] Since calculations of the 95th percentile confidence limits on the CDF estimates are dependent on an assumption of simple random sampling, which does not hold for the NLFA data, the accuracy of the confidence limits are affected and thus are not presented for the NLFA data.

5.4 Combining the NLFA and NLFTS Data

The lack of statistical difference in the distributions of fish tissue concentrations (except in the highest 95th percentile) in the NLFA supports its use along with the NLFTS data for a benefits analysis for this rule.

When the two data sets (NLFA and NLFTS) are combined, on average, there is one sample location / 1,000 sq. km. The sampling network is more-dense in the East and on the West Coast, and less dense in the Midwest. Figure 5-6. illustrates the variable spatial density of the combined NLFA and NLFTS sample locations.

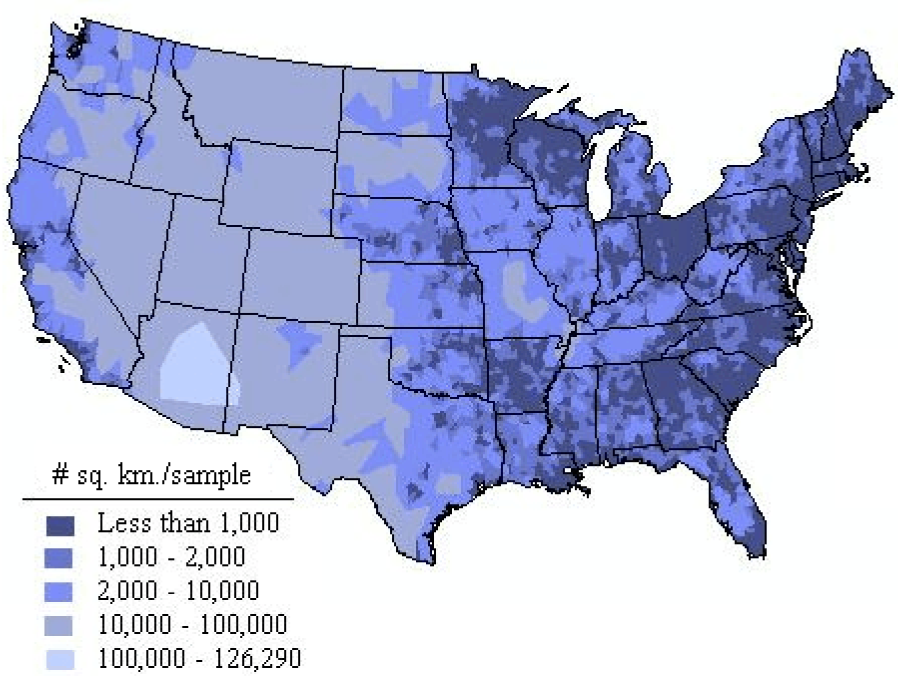

Figure 5-6. Total Area/Sample in the Combined NLFA and NLFTS Data Set

An examination of the combined NLFA and NLFTS fish tissue concentrations reveals further data problems that need to be addressed. In the combined data set, there are over 92,224 records of concentrations sampled in the late 1960's through 2003, and 35 different sampling methods were used to obtain Hg concentrations from over 450 different species of fish that range in length from .01 to 379 in. (over 30 ft., and an obvious data entry error). Samples were obtained from over 11,000 different locations across the U.S., and concentrations range from below detection limits to over 10 ppm

To eliminate errors within the data due to data entry and focus benefit analysis efforts, the following criteria were applied to the data that were selected for further analysis:

- Sampling date must be later than 1990.
- The sample must be geo-referenced.

- The sample length must have a recorded length[67] that is reasonable when compared to world record lengths.[8]
- All Hg concentration units that were recorded in .ppb were converted to .ppm to maintain consistency with the majority of the samples.
- Only freshwater species samples are used.

5.5 Normalization of Hg Fish Tissue Concentration Data

Hg fish tissue concentration variability due to species/size/sample method can be seen at a global database scale in averages reported previously in this report, and on an individual sample level. For example, at Abbots Creek in North Carolina, at least 20 largemouth bass were sampled using the same sample method on the same day in 1992. The lengths of these fish ranged from 9 to 19 inches, and fish tissue concentrations ranged from .24 ppm (9 inch fish), to 1.4 ppm (19 inch fish). In addition, a 12.41 in black crappie and a 12.41 in. largemouth bass were sampled at Abbots Creek. The concentration of the black crappie was .34 ppm, while the concentration of the largemouth bass was .19 ppm An example of variability potentially introduced by sample method can also be found from the Abbots Creek samples where two 8 in. goldfish were sampled four months apart. One was sampled using the whole fish, and one was sampled using a fillet. The whole fish sample was .32 ppm and the fillet sample was .96 ppm

Due to these sources of variability, a computation of the simple average of existing fish tissue concentrations at a location would not provide a representative estimate of Hg in fish for use in a benefits analysis. Averaging would include many species that are not typically targeted by anglers, and very small fish which would misrepresent Hg fish tissue concentrations consumed by the public. Because of this, we considered focusing on key fish species that are frequently targeted by anglers and consumable sizes representative of sizes typically found in lakes and rivers.

We considered using a subset of the combined NLFA and NLFTS consisting of these species larger that a minimum length, but we found that this would reduce the already sparse data set. Based on size alone, the removal of sampled fish less than 6 inches long would reduce the data set by about 20%. Controlling for species of fish that are typically targeted by anglers would reduce the data set even further.

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[6] Where a fish weight was recorded, the length was predicted using a regression for each sampled fish species of the log of the length as a dependent variable, and the weight of a fish as the independent variable. Typical residual error is approximately 10% across all species.

[7] It is of interest to note that almost all of the samples from the states of IA, NE, OH, TN, VA, and PA, were removed because the length of the fish was not recorded.

[8] Must not be more than 10% longer than a recorded world record length for that fish species. An additional 10% over current world records was allowed so that only obvious data entry problems are filtered out of the data set. A 10% threshold over the world record represents, for this study, fish lengths considered outside the realm of possibility. The recorded length must also be less than 108 inches (108 in. is 10% greater than the length of the largest world record of any freshwater species). Background research on world record lengths for each species was extremely time consuming. To focus QA/QC efforts, only species that were sampled at least 30 times in the U.S. were investigated.

To take full advantage of the Hg fish tissue concentration samples within the NLFA and NLFTS, it is important to control for variability due to species/fish size/sample method. Using information on species, size, sample method, location, and date of sampling event enables the use of statistical procedures to estimate what Hg fish tissue concentration would have been, had a different species/size/sample method been used to collect samples and detect the Hg concentrations.

The USGS developed a procedure called the National Descriptive Model of Mercury and Fish (NDMMF) (Wente 2004). The NDMMF is a statistical model related to covariance. It is calibrated using the NLFA and NLFTS data. The model is designed to allow the prediction of different species, cuts, and lengths of fish for sampling events, even when those species/lengths/cuts of fish were not sampled during those sampling events.

The idea is to model methyl mercury concentration as a power function of fish length, i.e., $y = ax^b$, where y = methyl mercury concentration, x = length, and a,b are parameters. Potentially, that could be done for each species at each site, but the data are far too sparse for that, and, further, we wanted to devise some way to predict mercury concentrations for a species that was not even collected at some given site. So, the assumptions are made that: (i) a universal, national value of "b" for each tissue type from each fish species, and (ii) the value of "a" at a given site at a given time is the same for all species. Thus a prediction of mercury concentration for a given tissue from a fish of an arbitrary species of length x could then be predicted by "looking up" the right value of "b" for that species and tissue and the value of "a" for that site and time.

To implement the NDMMF, the NLFA and NLFTS fish tissue concentrations are further prepared for analysis. If the method of sample collection was "fillet skin on scales off" or "composite fillet skin on", these were assigned to the "fillet skin on" sample method type. Various methods of "Fillet skin off" sample types were treated in a similar manner. "Composite" types were treated as "whole" sample types. Sample types that were "tissue carcass, crab, meat, dorsal muscle plug with skin, dorsal muscle plug without skin ,edible portion, eggs, gills, lipid, liver, organism without head or viscera, shellfish, tissue, turtle muscle/somatic , unknown, or viscera" were deleted.

Those species sampled 29 or fewer times were excluded from the analysis due to the lack of quality assurance background information collected for these samples (see section 5.4 description of data criteria). In total, the steps discussed in section 5.4 to remove data entry errors and to select relevant data for benefits analysis leave a total of 42,756 remaining samples from the original population of 92,224 samples.

The remaining samples are log-transformed to model the relationship as $\log y = b * \log x + \log a$, 1 is added to x and to y to escape the problem of taking the log of zero. The SAS program LIFEREG is then used to estimate the parameters. More detail about the NDMMF can be found at http://pubs.water.usgs.gov/sir20045199/.

The NDMMF was used to generate estimates of fish tissue concentrations for six species that are both commonly targeted by freshwater fish anglers and were frequently sampled in the NLFA and NLFTS data bases (largemouth bass, catfish, brown trout, white crappie, white perch, and walleye). The target sizes selected for each species is representative of the typical adult size of that fish species found in the wild (Schultz 2004).

For sampling events where samples were removed during data preparation, and the target species were sampled, the sampled target species Hg concentrations were averaged for every unique sampling event.

This means that for some sample locations, we cannot control for variability resulting from the size of the sample fish through the normalization process. In these cases, we retain the raw data from the NLFA and NLFTS. Figure 5-7 shows locations where it was possible to compute NDMMF estimates, and where raw data was used. Table 5-4 gives the average, max, min, and standard deviation of Hg fish tissue concentrations of the normalized data using the NDMMF. Table 5-5 below gives the average, max., min., and standard deviation of the non-normalized fish tissue concentrations used for this analysis.

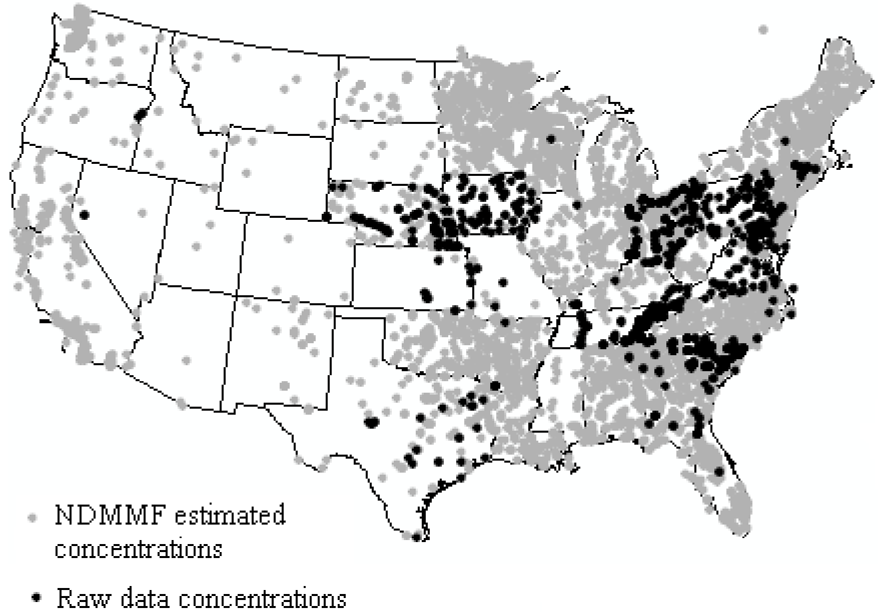


Figure 5-7. Locations Where Normalized Fish Tissue Concentrations are Utilized, and Where Non-Normalized Data are Utilized

Table 5-4. Statistical Distribution of Normalized Hg Fish Tissue Concentrations

SPECIES	Average	Maximum	Minimum	Standard Deviation
Largemouth Bass	.31	4.08	0	.36
Catfish	.23	2.98	0	.26
White Crappie	.14	1.89	0	.17
White Perch	.27	3.45	0	.31
Brown Trout	.11	1.47	0	.13
Walleye	.42	5.47	0	.48

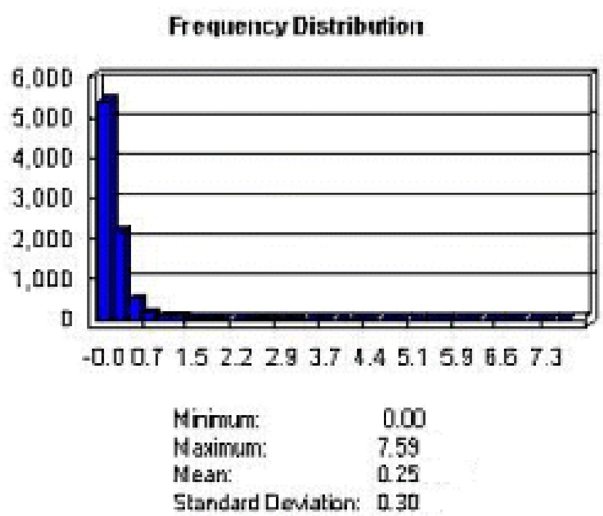
Table 5-5. Statistical Distribution of Non-Normalized Hg Fish Tissue Concentrations Shown in Figure 5-5

SPECIES	Average	Maximum	Minimum	Standard Deviation	Number of Samples
Largemouth Bass	.57	4.22	0	.68	558
Catfish	.14	1.30	0	.13	556
White Crappie	.13	1.11	0	.18	60
White Perch	.10	.84	0	.15	50
Brown Trout	.09	.56	0	.09	56
Walleye	.66	7.59	0	1.21	55

5.6 Resulting Fish Tissue Concentrations

To develop a baseline of Hg fish tissue concentrations for the benefits analysis of this rule, both the normalized and non-normalized data values shown in Tables 5-4 and 5-5 were merged to form a single data set, and the average of all the fish tissue concentrations (either normalized or non-normalized), by sampling event was computed. To do this we specify a sampling location and date as an "event", and then calculate an average for each event. For example, at event X, estimates were computed for largemouth bass, catfish, white crappie, white perch, brown trout, and walleye. All six species estimates were then averaged at event X to compute a single representative Hg fish tissue concentration for that event. For the raw data, where multiple target species were sampled in a particular event, these were averaged. Figure 5-8 shows the statistical distribution of averages. Figure 5-9 is a map of sample locations and representative Hg fish tissue concentrations in ppm

Figure 5-8. Baseline Average Hg Fish Tissue Concentrations



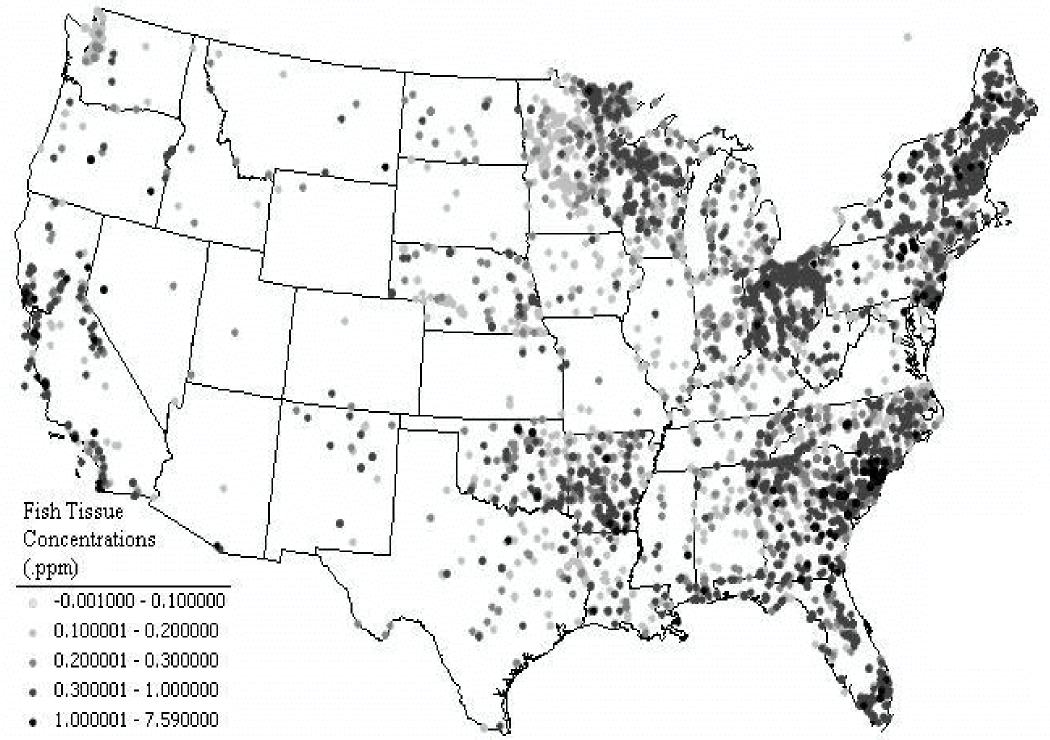


Figure 5-9. Statistical Distribution of Averages of Hg. Fish Tissue Concentrations in ppm

The species - averaged concentrations at each event were then averaged for each unique location. These averages were then averaged by US Geological Survey (USGS) 8-digit watershed units called Hydrologic Unit Code (HUC) Classification Areas. 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 HUCs. The 8-digit HUC is the smallest USGS Hydrologic Unit Code (HUC) Classification. Figure 5-10 geographically shows the USGS 8-digit HUCs and the frequency they were sampled Approximately 35% of HUCs in the states east and south of North Dakota (excluding North Dakota) were not sampled.

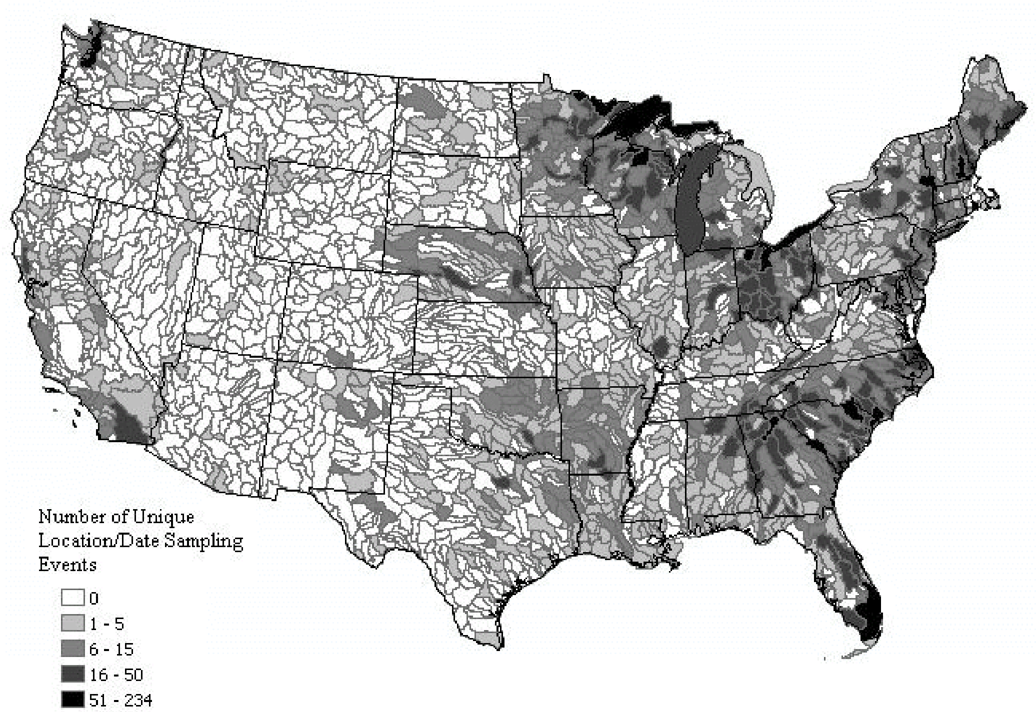


Figure 5-10. Number of Unique Sampling Events Within Each HUC

For the most part, higher concentrations averaged by HUC can be found in the southeastern coastal plains of North Carolina, south and west in the coastal plains around to Mississippi. Maine, New Hampshire, and some areas of New York, Pennsylvania and Ohio show some higher concentrations. Mercury Maps has recorded the presence of gold mines in South Carolina, chlor-alkali plants in Ohio, and mercury mines in Arkansas that may, in part, explain their higher Hg fish tissue concentrations. Figure 5-11 shows a map of mercury fish tissue concentrations averaged by HUC.

Average HUC concentration is .25 ppm The overwhelming majority of HUCs have concentrations below 1 ppm Figure 5-12 shows a frequency distribution of average HUC concentrations.

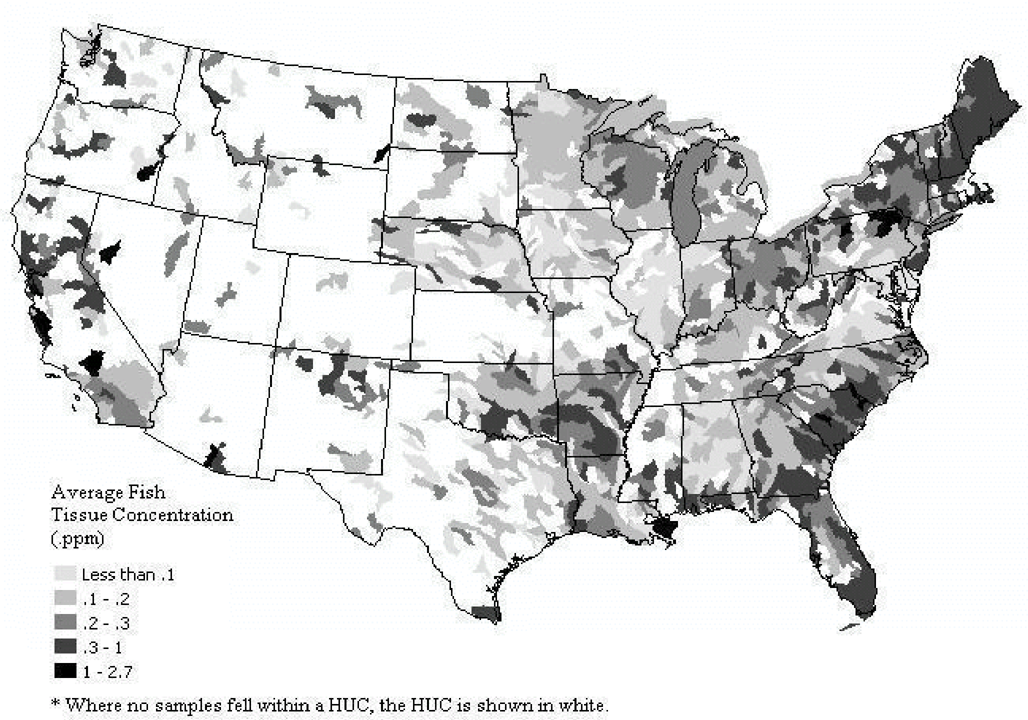


Figure 5-11. Average Fish Tissue Concentrations By HUC[9] [10]

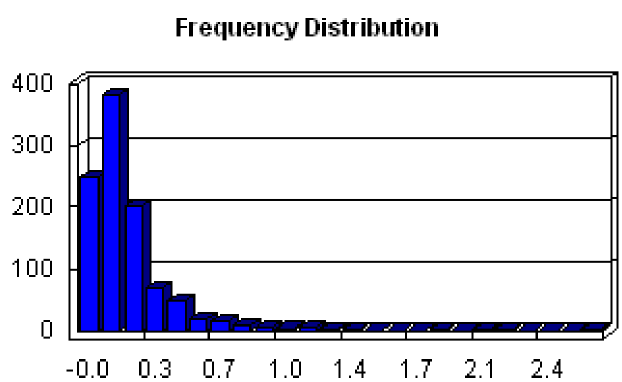


Figure 5-12. Frequency Distribution of HUC Averaged Concentrations (ppm)

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[9] 226 8-digit HUC watersheds have average concentrations higher than the Office of Water Methylmercury water quality criterion of .3 ppm

[10] Where no samples fall within a HUC, the HUC is shown in white. This does not indicate that Hg is not in the fish, but simply that data do not exist to estimate Hg concentration at this location.

5.7 Summary

The data of mercury concentrations in finfish and shellfish from freshwater and saltwater sources indicate a wide variety of contamination levels in fish species. The data demonstrate the finding in Section 3 of this report that larger predatory fish in the higher tropic levels tend to have higher levels of methyl mercury contamination in fish tissue.

We obtained data for a variety of fish species from the Mercury in Marine Life Database, the National Listing of Fish Advisories (NLFA), and the National Lake Fish Tissue Study (NLFTS). There are considerably more fish tissue samples from freshwater sources than for saltwater sources. The mean concentrations for the saltwater fish species obtained from the Mercury in Marine Life Database ranges from 0.04 ppm to 2.4 ppm, but some samples from predatory fish such as marlin, mackerel, shark, and barracuda indicate levels as high as 6.9 ppm Data for the freshwater fish species from the NLFA and NLFTS are used in the quantified benefit analysis provided in Section 11 of this report. Because the concentration of mercury in fish tissue can vary by species, the length of the fish, the sampling method (i.e., fillet, whole fish, skin on/off), and by location, EPA normalized the data from the NLFA and the NLFTS to control for variability from factors other than location of the fish tissue sample. The resulting normalized fish tissue samples were then average to the 8-digit HUC to provide a characterization of fish tissue concentrations in a watershed. The mean mercury concentrations in the HUCs ranges from 0 ppm to 7.59 ppm, with an overall mean of 0.25 ppm

5.8 References

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SECTION 6

PROFILE OF THE UTILITY SECTOR

This section discusses important aspects of the power sector as they relate to CAMR, including the types of power-sector sources affected by CAMR, and provides background on the power sector and EGUs. In addition, this section provides some historical background on EPA regulation of and future projections for the power sector.

6.1 Power-Sector Overview

The functions of the power sector can be separated into three distinct operating activities: generation, transmission, and distribution.

6.1.1 Generation

Electricity generation is the first process in the delivery of electricity to consumers. The process of generating electricity, in most cases, involves creating heat to rotate turbines which, in turn, create electricity. The power sector is comprised of over 16,000 generating units, consisting of fossil-fuel fired units, nuclear units, and hydroelectric and renewable sources dispersed throughout the country (see Table 6-1).

Table 6-1. Existing Electricity Generating Capacity by Energy Source, 2002

Energy Source	Number of Generators	Generator Nameplate Capacity (MW)
Coal	1,566	338,199
Petroleum	3,076	43,206
Natural Gas	2,890	194,968
Dual Fired	2,974	180,174
Other Gases	104	2,210
Nuclear	104	104,933
Hydroelectric	4,157	96,343
Other Renewables	1,501	18,797
Other	41	756
Total	**16,413**	**979,585**

Source: EIA

These electric-generating sources provide electricity for commercial, industrial, and residential uses, each of which consumes roughly one-third of the total electricity produced (see Table 6-2).

Table 6-2. Total U.S. Electric Power Industry Retail Sales in 2003 (Billion kWh)

	N	%
Residential	1,280	37%
Commercial	1,119	32%
Industrial	991	28%
Other	109	3%
All Sectors	**3,500**	**100%**

Source: EIA

In 2003, electric-generating sources produced 3,848 billion kWh to meet electricity demand. Roughly 70 percent of this electricity was produced through the combustion of fossil fuels, primarily coal and natural gas, with coal accounting for more than half of the total (see Table 6-3).

Table 6-3. Electricity Net Generation in 2003 (billion kWh)

	N	%
Coal	1,970	51%
Petroleum	118	3%
Natural Gas	629	16%
Other Gases	11	0.3%
Nuclear	764	20%
Hydroelectric	275	7%
Other	81	2%
Total	**3,848**	**100%**

Source: EIA

Note: Retail sales and net generation may not correspond exactly because net generation data may include net exported electricity and loss of electricity.

Coal-fired generating units typically supply "base-load" electricity, which means these units operate continuously throughout the day. Coal-fired generation, along with nuclear generation, meet the part of electricity demand that is relatively constant. Gas-fired generation, however, typically supplies "peak" power, when there is increased demand for electricity (e.g., when businesses operate throughout the day or when people return home from work and run appliances and heating/air-conditioning, versus late at night or very early morning when demand for electricity is reduced).

6.1.2 Transmission

Transmission is the term used to describe the movement of electricity, through use of high voltage lines, from electric generators to substations where power is stepped down for local distribution. Transmission systems have been traditionally characterized as a collection of independently operated networks or grids interconnected by bulk transmission interfaces.

Within a well-defined service territory, the regulated utility has historically had responsibility for all aspects of developing, maintaining, and operating transmission of electricity. These responsibilities typically included system planning and expanding, maintaining power quality and stability, and responding to failures.

6.1.3 Distribution

Distribution of electricity involves networks of smaller wires and substations that take the higher voltage from the transmission system and step it down to lower levels to match the needs of customers. The transmission and distribution system is the classic example of a natural monopoly because it is not practical to have more than one set of lines running from the electricity-generating sources to neighborhoods or from the curb to the house.

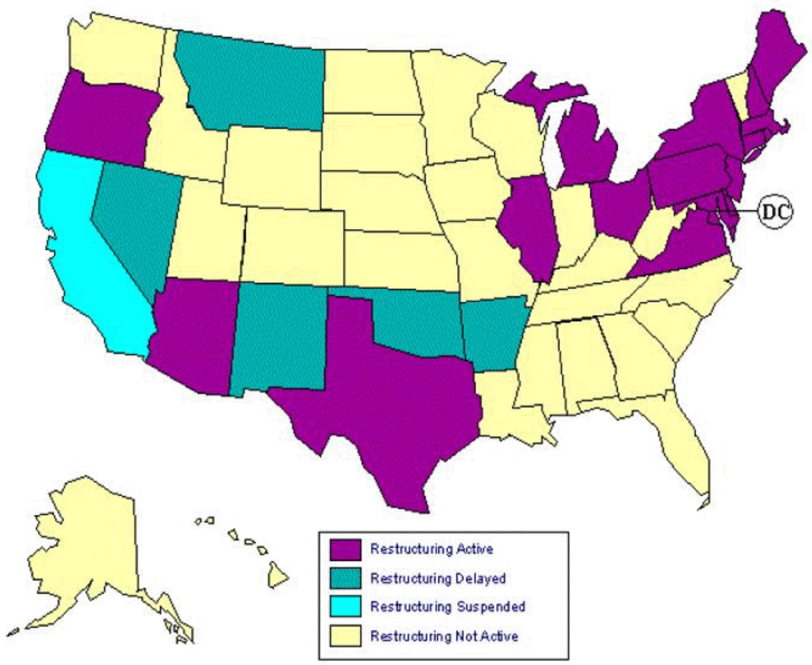
Transmission and distribution have been considered differently than generation in current efforts to restructure the industry. Transmission has generally been developed by the larger vertically integrated utilities that typically operate generation and distribution networks. Distribution is handled by a large number of utilities that often only sell electricity. Electricity restructuring has focused primarily on converting the industry to fully compete the sale of electricity production or generation and not the transmission or distribution of electricity. The restructuring of the industry is, in large part, the separating of generation assets from the transmission and distribution assets into separate economic entities in many state efforts. Transmissions and distribution remain price regulated throughout the country based on the cost of service.

6.2 Deregulation and Restructuring

The ongoing process of deregulation of wholesale and retail electric markets is changing the structure of the electric power industry. In addition to reorganizing asset management between companies, deregulation is aimed at the functional unbundling of generation, transmission, distribution, and ancillary services the power sector has historically provided to competitors in the generation segment of the industry.

Beginning in the 1970s, government policy shifted against traditional regulatory approaches and in favor of deregulation for many important industries, including transportation, communications, and energy, which were all thought to be natural monopolies (prior to 1970) that warranted governmental control of pricing. Some of the primary drivers for deregulation of electric power included the desire for more efficient investment choices, the possibility of lower electric rates, reduced costs of combustion turbine technology that opened the door for more companies to sell power, and complexity of monitoring utilities' cost of service and establishing cost-based rates for various customer classes (see Figure 6-1). The pace of restructuring in the electric power industry slowed significantly in response to market volatility and financial turmoil associated with bankruptcy filings of key energy companies in California. By the end of 2001, restructuring had either been delayed or suspended in eight states that previously enacted legislation or issued regulatory orders for its implementation. Another 18 other states that had seriously explored the possibility of deregulation in 2000 reported no legislative or regulatory activity in 2001 (DOE, EIA, 2003a). Currently, there are 17 states where price deregulation of generation (restructuring) has occurred. The effort is more or less at a standstill; however, at the federal level, there are efforts in the form of proposed legislation and proposed Federal Energy Regulatory Commission (FERC) actions aimed at reviving restructuring. For states that have not begun restructuring efforts, it is unclear when and at what pace these efforts will proceed.

Figure 6-1. Status of State Electricity Industry Restructuring Activities (as of February



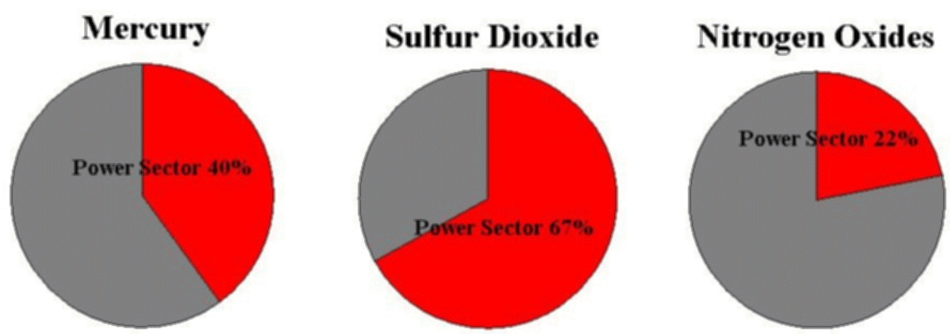
2003)

6.3 Pollution and EPA Regulation of Emissions

The burning of fossil fuels, which generates about 70 percent of our electricity nationwide, results in air emissions of Hg, SO_2 and NO_x, important precursors in the formation of fine particles and ozone (NO_x only). The power sector is a major contributor of these three pollutants, and reductions of SO_2 and NO_x emissions are critical to EPA's efforts to bring about attainment with the fine particle and ozone NAAQS through programs like CAIR, and critical to

EPA's effort under CAMR because control of SO_2 and NO_x also can result in Hg reduction. In 2003, the power sector accounted for 40% of total nationwide Hg emissions, 67 percent of total nationwide SO_2 emissions and 22 percent of total nationwide NO_x emissions (see Figure 6-2).

Figure 6-2. Emissions of Hg, SO_2, and NO_x from the Power Sector (2003)



Mercury emissions from the power sector come mainly from coal-fired units, and CAMR represents the first time Hg emissions from these units will be regulated. Mercury emissions can vary by coal type, especially for units with existing PM, NO_x, and SO_2 controls. In general, given the different properties of coal, these existing controls are best able to capture Hg from bituminous coals, with less capture from subbituminous and lignite coals.

6.4 Pollution Control Technologies

There are two primary options for reducing SO_2 emissions from coal-burning power plants. Units may switch from higher to lower sulfur coal, or they may use flue gas desulfurization (FGD, commonly referred to as scrubbers). According to data submitted to EPA for compliance with the Title IV Acid Rain Program, the SO_2 emission rates for coal-fired units varied from under 0.4 lbs/mmBtu to over 5 lbs/mmBtu depending on the type of coal combusted.

It is generally easier to switch to a coal within the same rank (e.g., bituminous or sub-bituminous) because these coals will have similar heat contents and other characteristics. Switching completely to sub-bituminous coal (which typically has a lower sulfur content) from bituminous coal is likely to require some modifications to the unit. Limited blending of sub-bituminous coal with bituminous coal can often be done with much more limited modifications.

The two most commonly used scrubber types include wet scrubbers and spray dryers. Wet scrubbers can use a variety of sorbents to capture SO_2 including limestone and magnesium enhanced lime. The choice of sorbent can affect the performance, size, and capital and operating costs of the scrubber. New wet scrubbers typically achieve at least 95 percent SO_2 removal. Spray dryers can achieve over 90 percent SO_2 removal.

One method of reducing NO_x emissions is through the use of combustion controls (such as low NO_x burners and over-fired air). Combustion controls reduce NO_x by ensuring that the combustion of coal occurs under conditions that form less NO_x. Post-combustion controls reduce NO_x by removing the NO_x after it has been formed. The most common post-combustion control is SCR. SCR systems inject ammonia (NH_3), which combines with the NO_x in the flue gas, to form nitrogen and water, and uses a catalyst to enhance the reaction. These systems can reduce NO_x by 90 percent and achieve emission rates of around 0.06 lbs NO_x/mmBtu. Selective noncatalytic reduction also removes NO_x by injecting ammonia, but no catalyst is used. These systems can reduce NO_x by up to 40 percent.

Hg capture can occur through existing controls and through Hg-specific control technologies. Many power plants have existing mercury capture as a co-benefit of air pollution control technologies for NO_x, SO_2 and particulate matter (PM). This includes capture of particulate-bound mercury in PM control equipment and capture of soluble ionic Hg in wet flue gas desulfurization (FGD) systems. Additional data have also shown that the use of SCR for NO_x control enhances oxidation of Hg to the soluble ionic form, resulting in increased removal in the wet FGD system for units burning bituminous coal. The range of Hg removal depends on the control configuration and the coal type burned, and can vary between 0 and 98 percent. (For further discussion see *Control of Emissions from Coal-Fired Electric Utility Boilers: An Update*, EPA/Office of Research and Development, March 2005, in docket.)

Mercury-specific controls, most notably activated carbon injection (ACI), are used on municipal waste combustor (MWC) and medical waste incinerator (MWI) facilities in the U.S. and Europe. At present, ACI is the most widely studied of the mercury-specific control technologies for coal-fired power plants and shows the potential to achieve moderate-to-high levels of mercury control. EPA's modeling provides ACI as a compliance choice and assumes a 90% removal with the addition of a fabric filter PM control device.

For more detail on the cost and performance assumptions of pollution controls, see the documentation for the Integrated Planning Model (IPM), a dynamic linear programming model that EPA uses to examine air pollution control policies for Hg, SO_2, and NO_x throughout the contiguous United States for the entire power system. Documentation for IPM can be found at www.epa.gov/airmarkets/epa-ipm.

6.5 Regulation of the Power Sector

At the federal level, efforts to reduce emissions of SO_2 and NO_x have been occurring since 1970. Policy makers have recognized the need to address these harmful emissions, and incremental steps have been taken to ensure that the country meets air quality standards.

Federal regulation of SO_2 and NO_x emissions at power plants began with the 1970 Clean Air Act. The Act required the Agency to develop performance standards for a number of source categories including coal-fired power plants. The first New Source Performance Standards (NSPS) for power plants (subpart D) required new units to limit SO_2 emissions either by using scrubbers or by using low sulfur coal. NO_x was required to be limited through the use of low NO_x burners. A new NSPS (subpart Da), promulgated in 1978, tightened the standards for SO_2 requiring scrubbers on all new units.

The 1990 Clean Air Act Amendments (CAAA) placed a number of new requirements on power plants. The Acid Rain Program, established under Title IV of the 1990 CAAA, requires major reductions of SO_2 and NO_x emissions. The SO_2 program sets a permanent cap on the total amount of SO_2 that can be emitted by electric power plants in the contiguous United States at about one-half of the amount of SO_2 these sources emitted in 1980. Using a market-based cap-and-trade mechanism allows flexibility for individual combustion units to select their own methods of compliance. The program uses a more traditional approach to NO_x emission limitations for certain coal-fired electric utility boilers, with the objective of achieving a 2 million ton reduction from projected NO_x emission levels that would have been emitted in 2000 without implementation of Title IV.

The Acid Rain Program comprises two phases for SO_2 and NO_x. Phase I applied primarily to the largest coal-fired electric generation sources from 1995 through 1999 for SO_2 and from 1996 through 1999 for NO_x. Phase II for both pollutants began in 2000. For SO_2, the Acid Rain Program applies to thousands of combustion units generating electricity nationwide; for NO_x it generally applies to affected units that burned coal during 1990 through 1995. The Acid Rain Program has led to the installation of a number of scrubbers on existing coal-fired units as well as significant fuel switching to lower sulfur coals. Under the NO_x provisions of Title IV, most existing coal-fired units were required to install low NO_x burners.

The CAAA also placed much greater emphasis on control of NO_x to reduce ozone nonattainment. This has led to the formation of several regional NO_x trading programs as well as an intrastate NO_x trading program in Texas. The Ozone Transport Commission (a group of northeast states) created an interstate NO_x trading program that began in 1999. In 1998, EPA promulgated regulations (the NO_x SIP Call) that required 21 states in the eastern United States and the District of Columbia to reduce NO_x emissions that contributed to nonattainment in downwind states using the cap-and-trade approach. This program began in the summer of 2004 and has resulted in the installation of significant amounts of selective catalytic reduction.

In addition to federal programs to reduce emissions of SO_2 and NO_x, several states have also taken action. Several states, like North Carolina, New York, Connecticut, and Massachusetts, have moved to control these emissions to address nonattainment. To date, there have not been any regulations on the utility sector to control mercury emissions.

6.6 Cap and Trade

The cap-and-trade system under CAMR, which is largely based on the Acid Rain Trading Program and the NO_x SIP Call, provides the power sector with considerable flexibility in meeting the emission reduction requirements. Cap-and-trade regulation is an extremely efficient tool that allows for environmental goals to be met in the most cost-effective manner, because firms have economic incentives to achieve emissions reductions where they are cheapest. The system allows for various compliance options, with each firm determining what option works best given certain costs, such as fuel costs or costs of pollution controls.

In addition to the pollution control options discussed above, companies can comply with cap-and-trade programs through more efficient use of the generating fleet to take advantage of generating sources that emit less and run more efficiently, commonly referred to as dispatch changes. By shifting generation to these more efficient units, the power sector is reducing the cost of compliance because there is a cost to pollute under a cap. Another option is purchasing additional allowances to cover emissions.

6.7 Clean Air Interstate Rule

The CAIR is a new regulatory action that addresses air quality problems and improves public health and the environment by substantially reducing emissions of SO_2, NO_x, and Hg. The final CAIR requires annual SO_2 and NO_x reductions in 23 States and the District of Columbia, and also requires ozone season NO_x reductions in 25 States and the District of Columbia. Many of the CAIR States are affected by both the annual SO_2 and NO_x reduction requirements and the ozone season NO_x requirements. CAIR allows affected states to adopt a two-phased cap-and-trade program to meet emissions reduction requirements of roughly 73 percent for SO_2 and 61 percent for NO_x from 2003 levels.

The rule would affect roughly 3,000 fossil fuel-fired units with a nameplate capacity greater than 25 MW. These sources accounted for roughly 90 percent of nationwide SO_2 emissions and 78 percent of nationwide NO_x emissions in 2003 (see Table 6-4).

Table 6-4. Emissions of SO_2 and NO_x in 2003 and Percentage of Emissions in the CAIR Affected Region (tons)

	SO_2	NO_x
CAIR Region	9,501,201	3,251,980
Nationwide	10,595,069	4,165,026
CAIR Emissions as % of Nationwide Emissions	90%	78%

Source: EPA.
Note: Region includes the States of Alabama, Connecticut, District of Columbia, Florida, Georgia, Illinois, Indiana, Iowa, Kentucky, Louisiana, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, New York, North Carolina, Ohio, Pennsylvania, South Carolina, Tennessee, Texas, Virginia, West Virginia, and Wisconsin

EPA modeling[1] shows that coal-fired and oil/gas-fired generation will continue to play an important part of the electricity generating portfolio in the United States. Electricity demand is anticipated to grow by 1.6 percent a year, and total electricity demand is projected to be 4,198 billion kWh by 2010. Table 6-5 shows current electricity generation and projected levels in 2010 and 2015 using EPA modeling.

Table 6-5. Current Electricity Net Generation and EPA Projections for 2010 and 2015 (billion kWh)

	2003	2010	2015
Coal	1,970	2,198	2,242
Oil/Gas	758	777	1,026
Other	1,119	1,223	1,235
Total	**3,848**	**4,198**	**4,503**

Source: 2003 data is from EIA. Projections are from the Integrated Planning Model run by EPA.

[1]EPA uses the IPM to make power-sector forecasts about emissions, costs, and other key factors of the power sector. Industry projections presented here are from EPA's base case scenario. For more information about IPM, see http://www.epa.gov/airmarkets/epa-ipm/index.html.

Tables

Figures

SECTION 7

COST AND ENERGY IMPACTS

This chapter reports the cost, economic, and energy impact analysis performed for CAMR. EPA used the IPM, developed by ICF Consulting, to conduct its analysis. IPM is a dynamic linear programming model that can be used to examine air pollution control policies for Hg, SO_2, and NO_x throughout the contiguous United States for the entire power system. Documentation for IPM can be found at www.epa.gov/airmarkets/epa-ipm.

7.1 Modeling Background

The analysis presented here covers the electric power sector, a major source of Hg, SO_2, and NO_x emissions nationwide. CAMR requires that states control electric generation units fueled by coal through state Hg emissions reduction requirements. EPA has assumed that states implement those reductions through a cap-and-trade program. This analysis also assumes that electric generating units will also comply with CAIR requirements through a cap-and-trade program. For mercury, the analysis examines three control options, all implemented in multiple phases. See Table 7-1 for total annual Hg emissions caps for CAMR options examined. For SO_2 and NO_x, EPA modeled the requirements of the final CAIR. This modeling includes regionwide annual SO2 and NO_x caps on the 23 States and the District and Columbia that are required to make annual reductions, and includes a regionwide ozone season NO_x cap on the 25 States and the District of Columbia required to make ozone season reductions. See Table 7-2 for total annual emissions caps under CAIR used in EPA modeling.

Table 7-1. CAMR Options Annual Emissions Caps (Tons)

	2010–2014	2015-2017	2018–Thereafter
Option 1 (38/15)	38	38	15
Option 2 (15/15)	38	15	15
Option 3 (24/15)	38	24	15

Table 7-2. CAIR Emissions Caps (Million Tons)

	2010–2014	2015–Thereafter
SO_2	3.6	2.5
NO_x (Annual)	1.5	1.3
NO_x (Summer)	0.6	0.5

The final CAMR requires annual Hg reductions in 50 States and the District of Columbia. The final CAMR will require a 38 ton cap in 2010 and a 15 ton cap in 2018. Using IPM, EPA modeled the cost and emissions impacts of three Hg control options to aid in its decision for the final CAMR. This chapter will provide the analysis conducted for all three options. IPM output files for the model runs used in CAMR analyses are available in the CAMR docket.

The modeling conducted for this analysis assumes that sources are complying with the final CAIR control strategy along with a CAMR control strategy. To provide incremental comparison, the CAIR modeling results are also presented. The CAIR IPM modeling includes regionwide annual SO_2 and NO_x caps on the 23 States and the District of Columbia for States required to make annual reductions, and includes a regionwide ozone season NO_x cap on the 25 States and the District of Columbia required to make ozone season reductions. EPA modeled the final CAIR NO_x strategy as an annual NO_x cap with a nested, separate ozone season NO_x cap.

CAMR was designed to achieve significant Hg emissions reductions from the power sector in a highly cost-effective manner. EPA analysis has found that the most efficient method to achieve the emissions reduction targets is through a cap-and-trade system that States have the option of adopting. States, in fact, can choose not to participate in the optional cap-and-trade program. However, EPA believes that a cap-and-trade system for the power sector is the best approach for reducing Hg emissions. As a result, EPA modeling has focused on the cap-and-trade approach for meeting the CAMR requirements. The modeling done with IPM assumes a nation-wide Hg cap and trade system on the power sector for the 48 contiguous states. However, EPA recognizes that states may use a different approach for reducing emissions, given that CAMR allows States to choose how they will meet their Hg emissions budget through reductions from utility units. States can elect not to participate in the federal trading program, and pursue reductions through other means including facility limits and trading limited to inside the state borders. This would likely impact the cost estimate of the program.

IPM has been used for evaluating the economic and emission impacts of environmental policies for over a decade. The model's base case incorporates title IV of the Clean Air Act (the Acid Rain Program), the NO_x SIP Call, various New Source Review (NSR) settlements, and several state rules affecting emissions of SO_2 and NO_x that were finalized prior to April of 2004. The NSR settlements include agreements between EPA and Southern Indiana Gas and Electric Company (Vectren), Public Service Electric & Gas, Tampa Electric Company, We Energies (WEPCO), Virginia Electric Power Company (Dominion), and Santee Cooper. IPM also includes various current and future state programs in Connecticut, Illinois, Maine, Massachusetts, Minnesota, New Hampshire, North Carolina, New York, Oregon, Texas, and Wisconsin. IPM includes state rules that have been finalized and/or approved by a state's legislature or environmental agency. The base case is used to provide a reference point to compare environmental policies and assess their impacts and does not reflect a future scenario that EPA predicts will occur.

The economic modeling presented in this chapter has been developed for specific analyses of the power sector. Thus, the model has been designed to reflect the industry as accurately as possible. As a result, EPA has used discount rates in IPM that are appropriate for the various types of investments and other costs that the power sector incurs. The discount rates used in IPM may differ from discount rates used in other EPA analyses done for CAMR, particularly the discount rates used in the benefits analysis that are assumed to be social discount rates. EPA uses the best available information from utilities, financial institutions, debt rating agencies, and government statistics as the basis for the discount rates used for power sector modeling. These discount rates have undergone review by the power sector and the Energy Information Administration. EPA's discount rate approach has not been challenged in court.

EPA's modeling is based on its best judgment for various input assumptions that are uncertain, particularly assumptions for Hg control technology, future fuel prices and electricity demand growth. To some degree, EPA addresses the uncertainty surrounding these assumptions through its sensitivity analysis provided in the chapter. Other uncertainties, like states choosing not to participate in the trading program, would also impact the cost estimate.

More detail on IPM can be found in the model documentation, which provides additional information on the assumptions discussed here as well as all other assumptions and inputs to the model (www.epa.gov/airmarkets/epa-ipm).

7.2 Projected Hg Emissions

Because excess emission reductions are projected to be banked under the first phase of the Hg program, emissions in the second (or third phase) will be initially higher than the cap that are required for CAMR. As shown in Figure 7-1, the results of EPA modeling of CAMR show state-by-state emissions in 2020 for some states do change significantly among CAMR options. However, for some states, the emissions projections among options follow the same profile as the national emission projections in 2020.

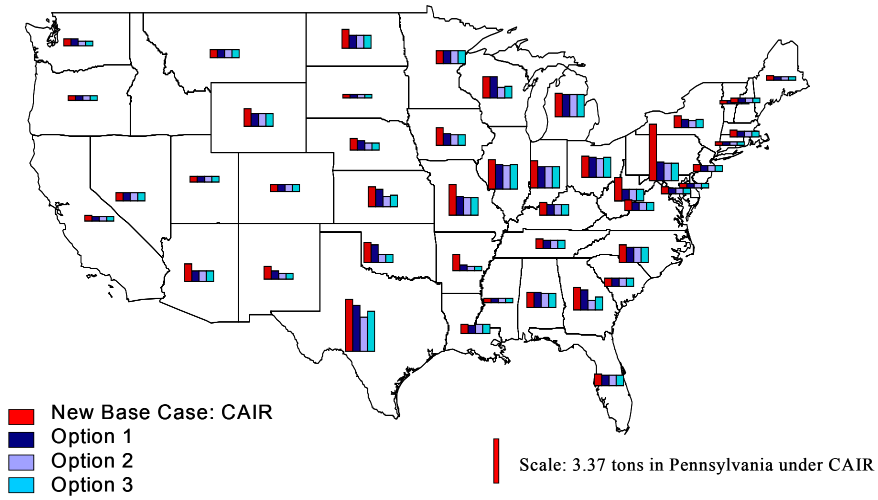


Figure 7-1. Projected Mercury Emissions in 2020 by State

Table 7-3 provides projected total Hg emissions levels and Table 7-4 provides projected speciated Hg emissions levels in 2020. EPA projections of Hg emissions are based on 1999 Hg ICR emission test data and other more recent testing conducted by EPA, DOE, and industry participants (for further discussion see *Control of Emissions from Coal-Fired Electric Utility Boilers: An Update*, EPA/Office of Research and Development, March 2005, in docket). That emissions testing has provided a better understanding of Hg emissions and their capture in pollution control devices. Mercury speciates into three basic forms, ionic, elemental, and particulate. In general, ionic Hg compounds are more readily adsorbed than elemental Hg. The presence of chlorine compounds (which tend to be higher for bituminous coals) results in increased ionic mercury.

Overall the 1999 Hg ICR data revealed higher levels of Hg capture for bituminous coal-fired plants as compared to low-rank coal-fired plants, large ranges of Hg capture in existing plants, higher levels of Hg capture in fabric filters (FF) compared to electrostatic precipitators (ESPs), and a significant capture of ionic Hg in wet SO_2 scrubbers. Additional Hg testing indicates that for bituminous coals SCR has the ability to convert elemental Hg to ionic Hg and thus allow easier capture in a wet scrubber. This understanding of Hg capture was incorporated into EPA modeling assumptions (see IPM documentation, Hg EMFs) and is the basis for projections of Hg co-benefits from installation of scrubbers and SCR under CAIR.

Table 7-3. Projected Emissions of Hg with the Old Base Case[a], New Base Case, and with CAMR Options (Tons)

	2010	2015	2020
Old Base Case	46.6	45.0	46.2
New Base Case: CAIR	38.0	34.4	34.0
Option 1 (38/15)	31.3	27.9	24.3
Option 2 (15/15)	30.9	25.7	20.1
Option 3 (24/15)	31.1	27.4	21.1

Note: The emissions projections are for coal-fired electric power units greater than 25 MW.
[a] Base case includes Title IV Acid Rain Program, NO_x SIP Call, and State rules finalized before March 2004.
Source: Integrated Planning Model run by EPA.

Table 7-4. Projected Speciated Emissions of Hg in 2020 with New Base Case (CAIR) and CAMR Options (Tons)

	Elemental Hg	Ionic Hg	Particulate Hg	Total
1999	26.2	20.6	1.7	48.6
New Base Case: CAIR	25.8	7.9	0.8	34.4
Option 1 (38/15)	17.6	6.6	0.8	25.0
Option 2 (15/15)	14.3	5.7	0.8	20.9
Option 3 (24/15)	15.1	5.9	0.8	21.8

Note: Numbers may not add due to rounding and include un affected units. The emissions data presented here are EPA modeling results and include some unaffected units. 1999 emissions from 1999 Hg ICR estimate.

7.3 Projected SO_2 and NO_x Emissions

The addition of Hg cap does not significantly affect SO_2 and NOx emissions when compared to CAIR alone. National SO_2 emissions are somewhat lower in 2020 under the CAMR scenarios because sources are projected to install more scrubbers to achieve compliance for both CAIR and CAMR. Because of excess emission reductions are projected to be banked under the title IV Acid Rain Program that sources will be allowed to use under the requirements of CAIR, emissions in 2010 and 2015 will be higher than the caps that are required for CAIR. Tables 7-5 and 7-6 provide projected emissions levels for SO_2 and NOx.

Table 7-5. Projected Emissions of SO_2 with the Old Base Case[a], New Base Case (CAIR), and with CAMR Options (Million Tons)

	2010		2015		2020	
	Nationwide	CAIR Region	Nationwide	CAIR Region	Nationwide	CAIR Region
Old Base Case	9.7	8.8	8.9	8.0	8.6	7.7
New Base Case: CAIR	6.1	5.1	5.0	4.0	4.3	3.3
Option 1 (38/15)	6.1	5.1	4.9	4.0	4.2	3.3
Option 2 (15/15)	6.1	5.1	4.9	3.9	4.2	3.3
Option 3 (24/15)	6.1	5.1	4.9	4.0	4.2	3.3

Note: Emissions projections are for fossil-fired electric power sector.
[a] Base case includes Title IV Acid Rain Program, NO_x SIP Call, and State rules finalized before March 2004.
Source: Integrated Planning Model run by EPA.

Table 7-6. Projected Emissions of NOx with the Old Base Case[a], New Base Case (CAIR), and with CAMR Options (Million Tons)

	2010		2015		2020	
	Nationwide	CAIR Region	Nationwide	CAIR Region	Nationwide	CAIR Region
Old Base Case	3.6	2.8	3.7	2.8	3.7	2.8
New Base Case: CAIR	2.5	1.5	2.2	1.3	2.2	1.3
Option 1 (38/15)	2.4	1.5	2.2	1.3	2.2	1.3
Option 2 (15/15)	2.4	1.5	2.2	1.3	2.2	1.3
Option 3 (24/15)	2.4	1.5	2.2	1.3	2.2	1.3

Note: Emissions projections are for fossil-fired electric power sector.
[a] Base case includes Title IV Acid Rain Program, NO_x SIP Call, and State rules finalized before March 2004.
Source: Integrated Planning Model run by EPA.

7.4 Projected Costs

Table 7-7 provides EPA's projections of annual and present value costs incremental to CAIR. The cost of electricity generation represents roughly one-third to one-half of total electricity costs, with transmission and distribution costs representing the remaining portion. A better impact measure of the cost to the consumer is the impact on electricity pricing, which is shown in a later table.

The presence of an earlier cap under CAMR Option 2 (an the reduction of years of banking excess emissions) results in higher projected costs than Option 1. CAMR Option 2 costs are projected to be the highest of the options and is reflected by the lowest projected Hg emissions in 2020. The intermediate cap of 24 tons under Option 3 also reduces the amount of banking of excess emissions and results in higher projected costs than Option 1. However, because the final cap goes into place in 2018, the projected costs are lower than Option 2 and is reflected by the projected Hg emission in 2020 being higher than Option 2.

The marginal costs for Hg, SO_2 and NO_x can be found in Table 7-8. EPA projects a reduction in the SO_2 allowance price and changes in the NOx allowance price under CAMR when compared to CAIR alone. The changes in SO_2 and NOx allowance prices are due to the different set of costs faced by sources under CAMR. In th case of SO_2, the ability to control for both Hg and SO_2 effectively through scrubbers results in marginal cost of SO_2 being reflected in th Hg allowance price such that SO_2 allowance price falls. In th case of NOx, because SCR is an effective Hg control when combined with a scrubber, facilities choose to control different units than they would in th absence of a cap on Hg emissions. Sources will choose to control unit where they can install a combination of scrubbers and SCR to achieve both mercury and NOx reductions.

Table 7-7. Annualized National Private Compliance Cost and Present Value Cost ($1999)

Cost (billions)	2010	2015	2020	Present value (2007-2025)
Option 1 (38/15)	$0.16	$0.10	$0.75	$3.9
Option 2 (15/15)	$0.16	$0.36	$1.04	$6.0
Option 3 (24/15)	$0.16	$0.18	$1.04	$5.2

Note: Annual incremental costs of CAIR are $2.4 billion in 2010, $3.6 billion in 2015, and $4.4 billion in 2020, present value (2007-2025) is $41.1 billion.
Note: Numbers rounded to the nearest ten million for annualized cost.
Source: Integrated Planning Model run by EPA.

Table 7-8. Marginal Cost of Hg, SO_2, and NOx Reductions with CAMR Options ($1999)

		2010	2015	2020
New Base Case: CAIR	SO_2 ($/ton)	$800	$1,000	$1,300
	NOx ($/ton)	$1,300	$1,600	$1,600
Option 1 (38/15)	SO_2 ($/ton)	$700	$900	$1,200
	NOx ($/ton)	$1,200	$1,500	$1,300
	Hg ($/lb)	$23,200	$30,100	$39,000
Option 2 (15/15)	SO_2 ($/ton)	$700	$900	$1,100
	NOx ($/ton)	$1,200	$1,500	$1,200
	Hg ($/lb)	$29,000	$37,600	$48,700
Option 3 (24/15)	SO_2 ($/ton)	$700	$900	$1,100
	NOx ($/ton)	$1,200	$1,500	$1,300
	Hg ($/lb)	$26,400	$34,200	$44,400

Note: Numbers rounded to the nearest hundred for marginal cost.
Source: Integrated Planning Model run by EPA.

Actual costs may be lower than those presented since modeling assumes no improvements in the cost of mercury control technology. Given that this is the first time mercury emission will be regulated at the federal level[1] for the coal-fired power sector and given the current level of research and demonstration of mercury control technologies, control cost are expected to improve over time. For purposes of options comparisons, EPA has conservatively assumed no cost improvements in Hg control technologies. Later, in this Chapter, EPA will present a sensitivity analysis in which we examine impact of mercury technology improvements by providing a lower cost mercury control option in future years.

7.5 Projected Control Technology Retrofits

Under the modeled Hg options, Hg reduction is projected to result from the installation of additional flue gas desulfurization (FGD or scrubbers) on existing coal-fired generation capacity for SO_2 control, additional selective catalytic reduction technology (SCR) on existing coal-fired generation capacity for NO_x control, and activated carbon injection (ACI) on existing coal-fired capacity for Hg-specific control (see Table 7-9). In addition, during the first phase of the Hg program, some Hg banking of emissions is projected to be attributed to coal switching and dispatch changes. Most of the NO_x reductions achieved in the first phase of the rule can be attributed to the large pool of existing SCR that are used during the ozone season in the NO_x SIP call region that, for relatively little additional cost, run the SCRs year-round. Due to earlier second phase cap (Option 2) and the addition of a third phase (Option 3), less banking is projected in 2010 to 2015 timeframe and results in more ACI in 2020 as emissions approach the 15 ton cap.

Table 7-9. Pollution Controls by Technology with the Old Base Case, New Base Case (CAIR), and with CAMR Options (GW)

	2010			2015			2020		
	FGD	SCR	ACI	FGD	SCR	ACI	FGD	SCR	ACI
Old Base Case	110	111	--	116	119	--	117	121	0.3
New Base Case: CAIR	146	125	--	177	151	--	198	153	0.5
Option 1 (38/15)	146	126	2	179	153	3	199	156	13
Option 2 (15/15)	146	127	3	179	153	12	198	156	38
Option 3 (24/15)	147	127	3	179	153	5	199	156	30

Note: Numbers may not add due to rounding. Base case retrofits include existing scrubbers and SCR as well as additional retrofits for the Title IV Acid Rain Program, the NO_x SIP call, NSR settlements, and various state rules.
Source: Integrated Planning Model run by EPA.

[1] Some states have enacted Hg reduction requirements for the coal-fired power sector. See IPM documentation for modeled State Hg regulations.

7.6 Projected Generation Mix

Table 7-10 show the generation mix with CAMR. Coal-fired generation and natural gas-fired generation are projected to remain relatively unchanged because of the phased-in nature of CAMR, which allows industry the appropriate amount of time to install the necessary pollution controls.

Table 7-10. Generation Mix with the Old Base Case, with New Base Case (CAIR), and with CAMR Options (Thousand GWhs)

		2010	2015	2020	Change From New Base Case in 2020
Old Base Case	Coal	2,198	2,195	2,410	
	Oil/Natural Gas	777	1,072	1,221	
	Other	1,223	1,233	1,218	
New Base Case: CAIR	Coal	2,165	2,197	2,384	
	Oil/Natural Gas	807	1,069	1,247	
	Other	1,217	1,232	1,217	
Option 1 (38/15)	Coal	2,160	2,194	2,365	-0.8%
	Oil/Natural Gas	812	1,072	1,265	1.5%
	Other	1,216	1,233	1,217	0.0%
Option 2 (15/15)	Coal	2,158	2,191	2,365	-0.8%
	Oil/Natural Gas	813	1,075	1,266	1.5%
	Other	1,216	1,233	1,217	0.0%
Option 3 (24/15)	Coal	2,159	2,193	2,367	-0.7%
	Oil/Natural Gas	812	1,074	1,263	1.3%
	Other	1,216	1,232	1,217	0.0%

Note: Numbers may not add due to rounding.
Source: 2003 data are from EIA: Coal - 1,970; Oil/Natural Gas - 758; Other - 1,120. Projections are from the Integrated Planning Model run by EPA.

Under all three Hg control options modeled and relative to the new base case, no coal-fired generation is projected to be uneconomic to maintain under CAMR.

7.7 Projected Capacity Additions

In addition, EPA projects that future growth in electric demand will be met with a combination of new natural gas- and coal-fired capacity (see Table 7-11).

Table 7-11. Total Coal and Natural Oil/Gas-Fired Capacity by 2020 (GW)

	Current	Old Base Case	New Base Case: CAIR	Option 1 (38/15)	Option 2 (15/15)	Option 3 (24/15)
Pulverized Coal	305	318	315	314	314	314
IGCC	0.6	8	9	8	8	8
Oil/Gas	395	467	469	471	471	471

Source: Current data are from EPA's NEEDS 2004; projections are from the Integrated Planning Model run by EPA.

7.8 Projected Coal Production for the Electric Power Sector

Coal production for electricity generation is expected to increase relative to current levels, with or without CAIR (see Table 7-12). The reductions in emissions from the power sector will be met through the installation of pollution controls for Hg, SO_2 and NO_x removal. The pollution controls can achieve up to a 95 percent SO_2 removal rate, which allows industry to rely more heavily on local bituminous coal in the eastern and central parts of the country that has a higher sulfur content and is less expensive to transport than western subbituminous coal.

Table 7-12. Coal Production for the Electric Power Sector with the Old Base Case, New Base Case (CAIR) , and with CAMR Options (Million Tons)

			Old Base Case			New Base Case: CAIR		
Supply Area	2000	2003	2010	2015	2020	2010	2015	2020
Appalachia	299	275	325	315	301	306	306	331
Interior	131	135	161	162	173	165	191	218
West	475	526	603	631	714	607	586	609
National	905	936	1,089	1,109	1,188	1,078	1,083	1,158

Supply Area	Option 1 (38/15)			Option 2 (15/15)			Option 3 (38/15)		
	2010	2015	2020	2010	2015	2020	2010	2015	2020
Appalachia	303	310	330	303	309	322	304	309	325
Interior	169	194	224	170	195	231	171	194	232
West	589	568	572	587	565	574	587	567	570
National	1,061	1,071	1,127	1,060	1,069	1,127	1,061	1,070	1,127

Source: 2000 and 2003 data are derived from EIA data. All projections are from the Integrated Planning Model run by EPA.

7.9 Projected Retail Electricity Prices

Retail electricity prices for the U.S. are projected to increase a small amount with CAMR (see Table 7-13). The cap-and-trade approach allows industry to meet the requirements of CAMR in the most cost-effective manner, thereby minimizing the costs passed on to consumers. Retail electricity prices by NERC region (see Figure 7-2) are provided in Table 7-14 and show small increases in retail prices for the NERC regions in the eastern part of the country. By 2020, national retail electricity prices are projected to be roughly 0.3 percent higher with CAMR when compared to CAIR.

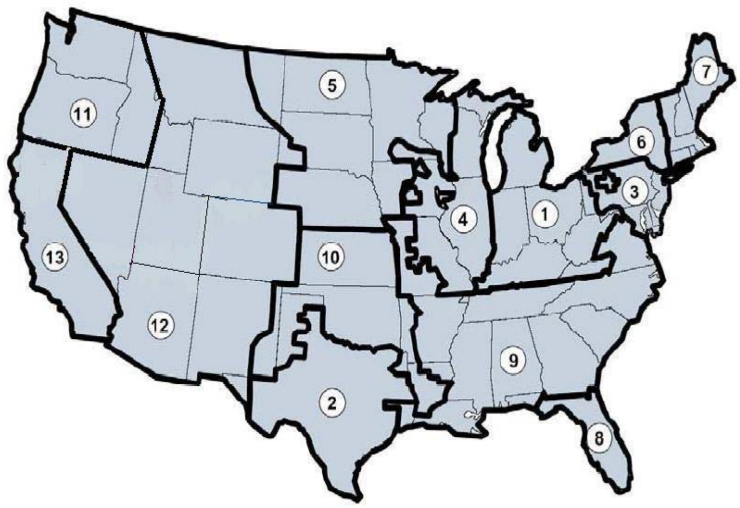


Figure 7-2. NERC Power Regions

Table 7-13. Projected National Retail Electricity Prices with the Old Base Case, New Base Case (CAIR), and CAMR Options (Mills/kWh) ($1999)

Year	Old Base Case	New Base Case: CAIR	Option 1 (38/15)	Option 2 (15/15)	Option 3 (24/15)
2010	58	61	61	61	61
2015	61	64	65	65	65
2020	61	64	65	65	65

Source: Retail Electricity Price Model run by EPA. 2000 national electric price is 66 mills/kWh from EIA's AEO 2003.

Table 7-14. Retail Electricity Prices by NERC Region with the Old Base Case, New Base Case (CAIR), and with CAMR Options (Mills/kWh) ($1999)

Option 1

			Base Case			CAIR			Option 1			Change from CAIR
Power Region	**Primary States Included**	**2000**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2020**
ECAR (1)	OH, MI, IN, KY, WV, PA	57.4	51.7	55.2	56.1	53.7	58.6	58.0	53.9	58.7	58.1	0.2%
ERCOT (2)	TX	65.1	57.9	64.4	62.6	59.4	64.5	63.3	59.1	64.9	63.4	0.1%
MAAC (3)	PA, NJ, MD, DC, DE	80.4	59.3	69.4	72.2	61.0	72.0	72.7	61.3	72.1	72.9	0.2%
MAIN (4)	IL, MO, WI	61.2	52.6	57.8	61.0	53.9	60.4	62.0	54.1	60.5	62.3	0.5%
MAPP (5)	MN, IA, SD, ND, NE	57.4	52.8	49.3	47.6	52.9	49.6	48.0	53.0	49.6	48.2	0.5%
NY (6)	NY	104.3	82.8	87.9	88.1	83.3	88.9	88.5	83.3	89.1	88.7	0.4%
NE (7)	VT, NH, ME, MA, CT, RI	89.9	77.4	83.9	82.8	77.4	84.7	83.1	77.5	84.8	83.1	0.0%
FRCC (8)	FL	67.9	71.2	71.3	69.5	71.7	72.3	70.5	71.8	72.3	70.7	0.3%
STV (9)	VA, NC, SC, GA, AL, MS, TN, AR, LA	59.3	56.2	55.1	55.3	57.0	56.2	56.6	57.1	56.2	56.8	0.3%
SPP (10)	KS, OK, MO	59.3	54.2	57.0	56.7	54.6	57.5	57.0	54.6	57.5	57.2	0.3%
PNW (11)	WA, OR, ID	45.9	49.6	47.4	46.9	49.8	47.5	46.9	49.8	47.5	47.1	0.5%
RM (12)	MT, WY, CO, UT, NM, AZ, NV, ID	64.1	63.9	65.2	64.7	64.1	65.6	65.4	64.2	65.6	65.1	-0.4%
CALI (13)	CA	94.7	97.1	98.9	99.3	97.3	99.1	99.5	97.3	99.1	99.7	0.3%
National	Contiguous Lower 48 States	66.0	60.3	63.1	63.4	61.3	64.5	64.3	61.3	64.5	64.5	0.2%

Option 2

			Base Case			CAIR			Option 2			Change from CAIR
Power Region	**Primary States Included**	**2000**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2020**
ECAR (1)	OH, MI, IN, KY, WV, PA	57.4	51.7	55.2	56.1	53.7	58.6	58.0	53.9	58.8	58.1	0.2%
ERCOT (2)	TX	65.1	57.9	64.4	62.6	59.4	64.5	63.3	59.1	64.9	63.4	0.1%
MAAC (3)	PA, NJ, MD, DC, DE	80.4	59.3	69.4	72.2	61.0	72.0	72.7	61.3	72.1	72.9	0.2%
MAIN (4)	IL, MO, WI	61.2	52.6	57.8	61.0	53.9	60.4	62.0	54.1	60.6	62.4	0.7%
MAPP (5)	MN, IA, SD, ND, NE	57.4	52.8	49.3	47.6	52.9	49.6	48.0	53.0	49.7	48.3	0.8%
NY (6)	NY	104.3	82.8	87.9	88.1	83.3	88.9	88.5	83.3	89.2	88.7	0.3%
NE (7)	VT, NH, ME, MA, CT, RI	89.9	77.4	83.9	82.8	77.4	84.7	83.1	77.5	84.7	83.2	0.2%
FRCC (8)	FL	67.9	71.2	71.3	69.5	71.7	72.3	70.5	71.7	72.3	70.7	0.3%
STV (9)	VA, NC, SC, GA, AL, MS, TN, AR, LA	59.3	56.2	55.1	55.3	57.0	56.2	56.6	57.1	56.4	56.8	0.4%
SPP (10)	KS, OK, MO	59.3	54.2	57.0	56.7	54.6	57.5	57.0	54.6	57.6	57.6	1.0%
PNW (11)	WA, OR, ID	45.9	49.6	47.4	46.9	49.8	47.5	46.9	49.8	47.4	47.2	0.6%
RM (12)	MT, WY, CO, UT, NM, AZ, NV, ID	64.1	63.9	65.2	64.7	64.1	65.6	65.4	64.2	65.6	65.0	-0.6%
CALI (13)	CA	94.7	97.1	98.9	99.3	97.3	99.1	99.5	97.3	99.1	99.7	0.3%
National	Contiguous Lower 48 States	66.0	60.3	63.1	63.4	61.3	64.5	64.3	61.3	64.6	64.5	0.3%

Option 3

			Base Case			CAIR			Option 3			Change from CAIR
Power Region	**Primary States Included**	**2000**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2020**
ECAR (1)	OH, MI, IN, KY, WV, PA	57.4	51.7	55.2	56.1	53.7	58.6	58.0	53.9	58.8	58.1	0.2%
ERCOT (2)	TX	65.1	57.9	64.4	62.6	59.4	64.5	63.3	59.1	64.9	63.4	0.1%
MAAC (3)	PA, NJ, MD, DC, DE	80.4	59.3	69.4	72.2	61.0	72.0	72.7	61.3	72.1	72.9	0.2%
MAIN (4)	IL, MO, WI	61.2	52.6	57.8	61.0	53.9	60.4	62.0	54.0	60.6	62.5	0.7%
MAPP (5)	MN, IA, SD, ND, NE	57.4	52.8	49.3	47.6	52.9	49.6	48.0	53.0	49.7	48.3	0.7%
NY (6)	NY	104.3	82.8	87.9	88.1	83.3	88.9	88.5	83.3	89.2	88.7	0.3%
NE (7)	VT, NH, ME, MA, CT, RI	89.9	77.4	83.9	82.8	77.4	84.7	83.1	77.5	84.8	83.1	0.0%
FRCC (8)	FL	67.9	71.2	71.3	69.5	71.7	72.3	70.5	71.8	72.3	70.7	0.3%
STV (9)	VA, NC, SC, GA, AL, MS, TN, AR, LA	59.3	56.2	55.1	55.3	57.0	56.2	56.6	57.1	56.3	56.8	0.4%
SPP (10)	KS, OK, MO	59.3	54.2	57.0	56.7	54.6	57.5	57.0	54.6	57.5	57.5	0.9%
PNW (11)	WA, OR, ID	45.9	49.6	47.4	46.9	49.8	47.5	46.9	49.8	47.4	47.2	0.6%
RM (12)	MT, WY, CO, UT, NM, AZ, NV, ID	64.1	63.9	65.2	64.7	64.1	65.6	65.4	64.2	65.6	65.1	-0.5%
CALI (13)	CA	94.7	97.1	98.9	99.3	97.3	99.1	99.5	97.3	99.1	99.8	0.3%
National	Contiguous Lower 48 States	66.0	60.3	63.1	63.4	61.3	64.5	64.3	61.3	64.6	64.5	0.3%

Source: Retail Electricity Price Model run by EPA. 2000 prices from EIA's AEO 2003.

7.10 Projected Fuel Price Impacts

The impacts of CAMR on coal prices and natural gas prices before shipment are shown in Table 7-15 .

Table 7-15. Henry Hub Natural Gas Prices and Average Delivered Coal Prices with the Old Base Case, New Base Case (CAIR), and with CAMR Options (1999$/mmBtu)

	2010		2015		2020	
	Delivered Coal	**Henry Hub Gas**	**Delivered Coal**	**Henry Hub Gas**	**Delivered Coal**	**Henry Hub Gas**
Old Base Case	1.05	3.20	1.01	3.25	0.96	3.16
New Base Case: CAIR	1.05	3.25	0.98	3.30	0.93	3.20
Option 1 (38/15)	1.05	3.25	0.98	3.30	0.93	3.25
Option 2 (15/15)	1.05	3.25	0.98	3.30	0.93	3.25
Option 3 (24/15)	1.05	3.25	0.98	3.30	0.94	3.25

Source: Integrated Planning Model run by EPA. 2000 natural gas data are from Platts GASdata is $4.15/mmBtu. 2000 coal price from EIA is $1.25/mmBtu.
Note: Coal price changes largely result from changes in the mix of coal types used. Delivered coal prices vary widely, but large changes in the cost of each type of coal are not projected.

7.11 Social Cost Calculations

The annualization factor used for pure social cost calculations (for annualized costs) normally includes the life of capital and the social discount rate. For purposes of benefit-cost analysis of this rule, EPA has calculated the annualized social costs using the discount rates from the benefits analysis for CAMR (3% and 7% and a 30 year life of capital. The costs of added insurance was included in the calculations, but local taxes were not included because they are considered to be transfer payments, and not a social cost). Using these discount rates, the social costs of CAMR incremental to CAIR are $151 million in 2010 and $848 million in 2020 using a discount rate of 3%, and are $157 million in 2010 and $896 million in 2020 using a discount rate of 7%.

Recent research suggests that the total social costs of a new regulation may be affected by interactions between the new regulation and pre-existing distortions in the economy, such as taxes. In particular, if cost increases due to a regulation are reflected in a general increase in the price level, the real wage received by workers may be reduced, leading to a small fall in the total amount of labor supplied. This "tax interaction effect" may result in an increase in deadweight loss in the labor market and an increase in total social costs. The limited empirical data available to support quantification of any such effect leads to this qualitative identification of the costs.

7.12 Limitations of Analysis

EPA's modeling is based on its best judgment for various input assumptions that are uncertain, particularly assumptions for Hg control technologies and future fuel prices and electricity demand growth. To some degree, EPA addresses the uncertainty surrounding these three assumptions through its sensitivity analysis. Sensitivity analysis on future fuel prices and electricity demand growth are provided in section 7.15. A discussion on Hg technology cost uncertainty and sensitivity analysis are provided below in section 7.14. As a general matter, the Agency selects the best available information from available engineering studies of air pollution controls and has set up what it believes is the most reasonable modeling framework for analyzing the cost, emission changes, and other impacts of regulatory controls.

The annualized cost estimates of the private compliance costs that are provided in this analysis are meant to show the increase in production (engineering) costs of CAMR to the power sector. In simple terms, the private compliance costs that are presented are the annual increase in revenues required for the industry to be as well off after CAMR is implemented as before. To estimate these annualized costs, EPA uses a conventional and widely-accepted approach that is commonplace in economic analysis of power sector costs for estimating engineering costs in annual terms. For estimating annualized costs, EPA has applied a capital recovery factor (CRF) multiplier to capital investments and added that to the annual incremental operating expenses. The CRF is derived from estimates of the cost of capital (private discount rate), the amount of insurance coverage required, local property taxes, and the life of capital. The private compliance costs presented earlier are EPA's best estimate of the direct private compliance costs of CAMR.

The annualized cost of CAMR, as quantified here, is EPA's best assessment of the cost of implementing CAMR, assuming that States adopt the model cap and trade program. Under CAMR, States are required to meet Hg emission budget based on reductions from coal-fired utility units. States have the discretion to participate in the federal cap-and-trade program or to meet their budget through other options (including facility limits and trading restricted inside state boarder). These costs are generated from rigorous economic modeling of changes in the power sector due to CAMR. This type of analysis using IPM has undergone peer review and federal courts have upheld regulations covering the power sector that have relied on IPM's cost analysis.

The direct private compliance cost includes, but is not limited to, capital investments in pollution controls, operating expenses of the pollution controls, investments in new generating sources, and additional fuel expenditures. EPA believes that the cost assumptions used for CAMR reflect, as closely as possible, the best information available to the Agency today. The cost associated with monitoring emissions, reporting, and record keeping for affected sources is not included in these annualized cost estimates, but EPA has done a separate analysis and estimated the cost to be about $76 million (see final CAMR preamble Section VI.B. Paperwork Reduction Act).

Furthermore, there are some unquantified costs that EPA wants to identify as limits to its analysis. These costs include the costs of federal and State administration of the program, which we believe are modest given our experience with the Acid Rain Program and the NOx Budget Trading Program and likely to be less than the alternative of States developing approvable State

Plans, securing EPA approval of those State Plans, and federal/state enforcement. There also may be unquantified costs of transitioning to CAMR, such as the costs associated with the retirement of smaller or less efficient electricity generating units, and employment shifts as workers are retrained at the same company or re-employed elsewhere in the economy. There are certain relatively small permitting costs associated with Title IV that new program entrants face (we believe there are far less than 1,000 new entrants who may require one day of additional work for trading permits). In a separate analysis for the CAIR RIA, EPA estimated the indirect cost and impacts of higher electricity prices on the entire economy for the CAIR scenario (see Regulatory Impact Analysis for the Final Clean Air Interstate Rule, Appendix E (March 2005)). Given the small difference in electricity prices between CAMR and CAIR, analysis for CAMR would project similar results.

Cost estimates for CAMR are based on results from ICF's Integrated Planning Model. The model minimizes the costs of producing electricity (including abatement costs) while meeting load demand and other constraints (full documentation for IPM can be found at www.epa.gov/airmarkets/epa-ipm). The structure of the model assumes that the electric utility industry will be able to meet the environmental emission caps at least cost. Montgomery (1972) has shown that this least cost solution corresponds to the equilibrium of an emission permit system.[2] See also Atkinson and Tietenburg (1982), Krupnick et al. (1980), and McGartland and Oates (1985).[3] [4] [5] However, to the extent that transaction and/or search costs, combined with institutional barriers, restrict the ability of utilities to exhaust all the gains from emissions trading, costs are underestimated by the model. Utilities in the IPM model also have "perfect foresight." To the extent that utilities misjudge future conditions affecting the economics of pollution control, costs may be understated as well.

This modeling analysis does not take into account the potential for advancements in the capabilities of pollution control technologies for SO_2 and NO_x removal as well as reductions in their costs over time. Market-based cap-and-trade regulation serves to promote innovation and the development of new and cheaper technologies. As an example, recent cost estimates of the Acid Rain SO_2 trading program by Resources for the Future (RFF) and MIT's Center for Energy and Environmental Policy Research (CEEPR) have been as much as 83% lower than originally projected by the EPA.[6] It is important to note that the original analysis for the Acid Rain

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[2] Montgomery, W. David 1972. "Markets in Licenses and Efficient Pollution Control Programs." *Journal of Economic Theory* 5(3): 395-418.

[3] S. Atkinson and T. Tietenberg 1982. "The empirical properties of two classes of design for transferable discharge permit markets," *Journal of Environmental Economics and Management* 9:101-121

[4] Krupnick, A., W. Oates and E. Van De Verg. 1980. "On Marketable Air Pollution Permits: The Case for a System of Pollution Offsets." *Journal of Environmental Economics and Management* 10: 233-47.

[5] McGartland, A and W. Oates. 1985. "Marketable Permits for the Prevention of Environmental Deterioration," *Journal of Environmental Economics and Management* 12: 207-228.

[6] See (1) Carlson, Curtis.; Burtraw, Dallas R.; Cropper, Maureen and Palmer, Karen L. 2000. Sulfur Dioxide Control by Electric Utilities: What Are the Gain from Trade? *Journal of Political Economy* 108 (#6): 1292-1326, and (2) Ellerman, Denny. January, 2003. Ex Post Evaluation of Tradable Permits: The U.S. SO2 Cap-and-Trade Program.

Program done by EPA also relied on an optimization model like IPM. Ex ante, EPA costs estimates of roughly $2.7 to $6.2 billion[7] in 1989 were an overestimate of the costs of the program in part because of the limitation of economic modeling to predict technological improvement of pollution controls and other compliance options such as fuel switching. Ex post estimates of the annual cost of the Acid Rain SO_2 trading program range from $1.0 to $1.4 billion. Harrington et al. have compared estimates of actual costs of many large EPA regulatory programs to predictions of those costs made while programs were under development and found a tendency for predicted costs to overstate actual implementation costs for market-based programs.[8] EPA's mobile source programs use adjusted engineering cost estimates to account for this fact, which EPA has not done in this case.[9]

As configured in this application, the IPM model does not take into account demand response (i.e., consumer reaction to electricity prices). The increased retail electricity prices shown in Table 7-14 would prompt end users to curtail (to some extent) their use of electricity and encourage them to use substitutes.[10] The response would lessen the demand for electricity, lowering electricity prices and reducing generation and emissions. Because of demand response, certain unquantified negative costs (i.e., savings) result from the reduced resource costs of producing less electricity because of lower demand. To some degree, these saved resource costs will offset the additional costs of pollution controls and fuel switching that we would anticipate with CAMR. Although the reduction in electricity use is likely to be small, the cost savings from such a large industry ($250 billion in revenues in 2003) is likely to be substantial. EIA analysis examining multi-pollutant legislation under consideration in 2003 indicates that the annualized costs of CAMR may be overstated substantially by not considering demand response.

It is also important to note that the capital cost assumptions for scrubbers used in EPA modeling applications are highly conservative. These are a substantial part of the compliance costs. Data available from recent published sources show the reported FGD costs from recent installations to be below the levels projected by the IPM. In addition, EPA conducted a survey of recent FGD installations and compared the costs of these installations to the costs used in IPM. This survey included small, mid-size, and large units. Examples of the comparison of these referenced published data with the FGD capital cost estimates obtained from IPM are provided in the Final CAMR docket. There is also evidence that scrubber costs will decrease in the future because of the learning-by-doing phenomenon, as more scrubbers are installed[11].

Massachusetts Institute of Technology Center for Energy and Environmental Policy Research.

[7] 2010 Phase II cost estimate in $1995.

[8] Harrington, W. R.D. Morgenstern, and P. Nelson, 2000. "On the Accuracy of Regulatory Cost Estimates," Journal of Policy Analysis and Management 19(2): 297-322.

[9] See recent regulatory impact analysis for the Tier 2 Regulations for passenger vehicles (1999) and Heavy-Duty Diesel Vehicle Rules (2000).

[10] The degree of substitution/curtailment depends on the price elasticity of electricity.

[11] Manson, Nelson, and Neumann, 2002. "Assessing the Impact of Progress and Learning Curves on Clean Air Act Compliance Costs," Industrial Economics Incorporated.

Another area of uncertainty is the performance of mercury control removal systems, like the one assumed in the modeling, activated carbon injection with added pulse-jet fabric filters. ACI systems have shown great promise in demonstrated tests. However, there is uncertainty about the availability and effectiveness of ACI across all coal types in the 2010 timeframe, since these systems have not been fully deployed on coal-fired generating plants. EPA's assumption of 90% removal for ACI is based on EPA's Office of Research and Development (ORD) assessment (for further discussion see *Control of Emissions from Coal-Fired Electric Utility Boilers: An Update*, EPA/Office of Research and Development, March 2005, in CAMR docket). Although modeled in IPM to be available immediately for all coal-fired generation as a simplification of modeling, ORD assessment concluded that ACI could not be fully deployed on all plants by 2010 timeframe. EPA's modeling projects only a small amount of ACI use in the 2010 timeframe which is consistent with ORD's conclusion about the availability of ACI.

An additional limitation of Hg control assumptions is that we are assuming no development in control technologies even though we recognize that this is a fast moving area with new developments nearly monthly. Actual costs may be lower than those presented since modeling assumes no improvements in the cost of mercury control technology. Given that this is the first time mercury is regulated for the coal-fired power sector and the current level of research and demonstration of mercury control technologies, control costs are expected to improve over time. For purposes of modeling, EPA has conservatively based its cost assumptions for mercury control on today's knowledge and not included cost improvement assumptions in the modeling. Later, in this Chapter (section 7.14), EPA presents a sensitivity analysis in which we examine impact of mercury technology improvements by providing a lower cost mercury control option in future years. It is important to note that CAMR's cap-and-trade approach will encourage technological innovation in Hg emissions control and allow sources to exploit currently unforeseen emissions control technologies.

Further, while there are many choices of technology for mercury control in existence or under development, several are not offered to model plants in IPM. Plants in IPM cannot retrofit with a fabric filter or make improvements to existing controls to capture mercury, such as improving the cloth to air ratio of the fabric filter, up-grading their ESP or injecting carbon. In addition, research and development continues on other Hg control technologies, including the use of pre-combustion controls (e.g. K-fuels), or multi-pollutant controls (i.e., one control removing SO_2, NOx, and Hg). Given a cap-and-trade approach, we would expect further development and innovation of technology.

EPA's latest update of IPM incorporates State rules or regulations adopted before March 2004 and various NSR settlements. Documentation for IPM can be found at www.epa.gov/airmarkets/epa-ipm. Any State or settlement action since that time has not been accounted for in our analysis in this chapter.

On balance, after consideration of various unquantified costs (and savings that are possible), EPA believes that the annual private compliance costs that we have estimated are more likely to overstate the future annual compliance costs that industry will incur, rather than understate those costs.

7.13 Significant Energy Impact

According to *E.O. 13211: Actions that Significantly Affect Energy Supply, Distribution, or Use*, this rule is not significant, measured incrementally to CAIR, because it does not have a greater than a 1 percent impact on the cost of electricity production and it does not result in the retirement of greater than 500 MW of coal-fired generation.

Several aspects of CAMR are designed to minimize the impact on energy production. First, EPA recommends a trading program rather than the use of command-and-control regulations. Second, compliance deadlines are set cognizant of the impact that those deadlines have on electricity production. Both of these aspects of CAMR reduce the impact of the proposal on the electricity sector.

7.14 Sensitivity Analysis on Assumptions for Hg Control Costs

This section presents results of cost sensitivity analysis using the IPM. As discussed earlier in this Chapter, actual costs may be lower than those presented since modeling assumes no improvements in the cost of mercury control technology. Given that this is the first time mercury is federally regulated for the coal-fired power sector and given the current level of research and demonstration of mercury control technologies, control cost are expected to improve over time. The sensitivity analysis presented examines the impacts of possible improvements in Hg control costs over time. EPA selected Option 1 as the policy option for the final CAMR. For that reason, EPA proceeded with sensitivity analyses for that option.

The sensitivity analysis presented includes examination of the impact of mercury technology improvements by providing a lower cost mercury control option in future years. Specifically, the sensitivity analysis examines the impact of providing a second ACI option in 2013 with brominated sorbents and lower capital costs. The assumptions of costs and performance for the sensitivity analysis is based on recent testing sponsored by EPA, DOE, and industry and more information on these advanced sorbents can be found in white paper by EPA's Office of Research and Development, available in the docket. For purposes of modeling, EPA has assumed the availability of two ACI options: (1) ACI using conventional sorbents and achieving 90% removal with the addition of a fabric filter; and (2) ACI using advanced sorbents and achieving 80 to 90% removal without the addition of a fabric filter (see memorandum to the docket entitled "Assumptions used in sensitivity analysis for the Clean Air Mercury Rule"). The first ACI option is available at the start of the model and the second ACI is available in 2013. For comparison of impacts, the sorbent sensitivity was modeled based on the reduction levels for CAMR Option 1 (Hg trading scenario plus CAIR of 38 tons in 2010, 15 tons in 2018).

Tables 7-16 and 7-17 provide Hg, SO_2, and NOx emission projections for the sorbent sensitivity option. Because the banking of excess emission under the first phase of the Hg program, emissions are projected to be higher than the cap that is required for CAMR in 2020. However, with lower future Hg technology costs, less banking and higher emissions are projected in 2010 and 2015 under the sorbent sensitivity option.

Table 7-19 provides annual and present value costs incremental to CAIR and Table 7-20 provides marginal costs. Under the sorbent sensitivity, the second ACI option has higher O&M

costs, but lower capital costs resulting in overall lower cost projections. Compared with CAMR option 1, annual costs and present value cost are projected to be lower for the sorbent sensitivity option and Hg marginal cost are projected to be about 50 percent lower.

The lower costs for ACI technology also results in higher projections of ACI retrofits in 2020 for the sorbent sensitivity option (see Table 7-21). When compared to Option 1, Coal generation and production are projected to increase under the sorbent sensitivity option (see Tables 7-22, 7-23, and 7-24). Retail electricity prices are not projected to changes significantly when comparing the sorbent sensitivity option to Option 1 (see Table 7-25).

Table 7-16. Projected Emissions of Hg with New Base Case (CAIR) and CAMR, without and with Selected Technological Advances (Tons)

	2010	2015	2020
New Base Case: CAIR	38.0	34.4	34.0
Option 1 – Current Technology	31.3	27.9	24.3
Option 1 – Sorbent Sensitivity	32.6	29.3	23.1

Note: The emissions data presented here are EPA modeling results.

Table 7-17. Projected Emissions of SO_2 with New Base Case (CAIR) and CAMR, without and with Selected Technological Advances (Million Tons)

	2010		2015		2020	
	Nationwide	CAIR Region	Nationwide	CAIR Region	Nationwide	CAIR Region
New Base Case: CAIR	6.1	5.1	5.0	4.0	4.3	3.3
Option 1 – Current Technology	6.1	5.1	4.9	4.0	4.2	3.3
Option 1 – Sorbent Sensitivity	6.1	5.1	4.9	4.0	4.3	3.3

Source: Integrated Planning Model run by EPA.

Table 7-18. Projected Emissions of NOx with the New Base Case (CAIR) and CAMR without and with Selected Technological Advances (Million Tons)

	2010		2015		2020	
	Nationwide	CAIR Region	Nationwide	CAIR Region	Nationwide	CAIR Region
New Base Case: CAIR	2.5	1.5	2.2	1.3	2.2	1.3
Option 1 – Current Technology	2.4	1.5	2.2	1.3	2.2	1.3
Option 1 – Sorbent Sensitivity	2.4	1.5	2.2	1.3	2.2	1.3

Source: Integrated Planning Model run by EPA.

Table 7-19. Annualized Private Compliance Cost and Present Value Cost Incremental to the New Base Case (CAIR) ($1999)

Cost (billions)	2010	2015	2020	Present value (2007-2025)
Option 1 – Current Technology	$0.16	$0.10	$0.75	$3.9
Option 1 – Sorbent Sensitivity	$0.10	$0.04	$0.56	$2.2

Note: Annual incremental costs of CAIR are $2.4 billion in 2010, $3.6 billion in 2015, and $4.4 billion in 2020, present value (2007-2025) is $41.1 billion.
Note: Numbers rounded to the nearest hundred million for annualized cost.
Source: Integrated Planning Model run by EPA.

Table 7-20. Marginal Cost of Hg, SO_2, and NO_x Reductions with CAMR without and with Selected Technological Advances ($1999)

		2010	2015	2020
New Base Case: CAIR	SO_2 ($/ton)	$800	$1,000	$1,300
	NOx ($/ton)	$1,300	$1,600	$1,600
Option 1 – Current Technology	SO_2 ($/ton)	$700	$900	$1,200
	NOx ($/ton)	$1,200	$1,500	$1,300
	Hg ($/lb)	$23,200	$30,100	$39,000
Option 1 – Sorbent Sensitivity	SO_2 ($/ton)	$800	$1,000	$1,300
	NOx ($/ton)	$1,200	$1,500	$1,400
	Hg ($/lb)	$11,800	$15,300	$19,900

Note: Numbers rounded to the nearest hundred for marginal cost.
Source: Integrated Planning Model run by EPA.

Table 7-21. Pollution Controls by Technology with the New Base Case (CAIR), and CAMR without and with Selected Technological Advances (GW)

	2010			2015			2020		
	FGD	SCR	ACI	FGD	SCR	ACI	FGD	SCR	ACI
New Base Case: CAIR	146	125	--	177	151	--	198	153	0.5
Option 1 – Current Technology	146	126	2	179	153	3	199	156	13
Option 1 – Sorbent Sensitivity	146	126	1	179	153	3	197	155	25

Note: Retrofits include existing scrubbers and SCR as well as additional retrofits for the Title IV Acid Rain Program, the NO_x SIP call, NSR settlements, and various state rules.
Source: Integrated Planning Model run by EPA.

Table 7-22. Generation Mix with the Old Base Case, the New Base Case (CAIR), and with CAMR without and with Selected Technological Advances (Thousand GWhs)

		2010	2015	2020	Change From New Base Case in 2020
Old Base Case	Coal	2,198	2,195	2,410	
	Oil/Natural Gas	777	1,072	1,221	
	Other	1,223	1,233	1,218	
New Base Case: CAIR	Coal	2,165	2,197	2,384	
	Oil/Natural Gas	807	1,069	1,247	
	Other	1,217	1,232	1,217	
Option 1 – Current Technology	Coal	2,160	2,194	2,365	-0.8%
	Oil/Natural Gas	812	1,072	1,265	1.5%
	Other	1,216	1,233	1,217	0.0%
Option 1– Sorbent Sensitivity	Coal	2,161	2,196	2,372	-0.5%
	Oil/Natural Gas	811	1,070	1,258	0.9%
	Other	1,217	1,233	1,217	0.0%

Note: Numbers may not add due to rounding.
Source: 2003 data are from EIA: Coal - 1,970; Oil/Natural Gas - 758; Other - 1,120. Projections are from the Integrated Planning Model run by EPA.

Table 7-23. Total Coal and Natural Oil/Gas-Fired Capacity by 2020 (GW)

	Current	Old Base Case	New Base Case: CAIR	Option 1 – Current Technology	Option 1 – Sorbent Sensitivity
Pulverized Coal	305	318	315	314	314
IGCC	0.6	8	9	8	9
Oil/Gas	395	467	469	471	470

Source: Current data are from EPA's NEEDS 2004; projections are from the Integrated Planning Model run by EPA.

Table 7-24. Coal Production for the Electric Power Sector with the Old Base Case, New Base Case (CAIR) , and with CAMR without and with Selected Technological Advances (Million Tons)

			Old Base Case			New Base Case: CAIR		
Supply Area	2000	2003	2010	2015	2020	2010	2015	2020
Appalachia	299	275	325	315	301	306	306	331
Interior	131	135	161	162	173	165	191	218
West	475	526	603	631	714	607	586	609
National	905	936	1,089	1,109	1,188	1,078	1,083	1,158

Supply Area	Option 1 – Current Technology			Option 1 – Sorbent Sensitivity		
	2010	2015	2020	2010	2015	2020
Appalachia	303	310	330	305	312	333
Interior	169	194	224	168	191	220
West	589	568	572	592	578	579
National	1,061	1,071	1,127	1,065	1,081	1,132

Source: 2000 and 2003 data are derived from EIA data. All projections are from the Integrated Planning Model run by EPA.

Table 7-25. Retail Electricity Prices by NERC Region with the Old Base Case, New Base Case (CAIR), and with CAMR without and with Selected Technological Advances (Mills/kWh) ($1999)

Option 1

			Base Case			CAIR			Option 1			Change from CAIR
Power Region	**Primary States Included**	**2000**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2020**
ECAR	OH, MI, IN, KY, WV, PA	57.4	51.7	55.2	56.1	53.7	58.6	58.0	53.9	58.7	58.1	0.2%
ERCOT	TX	65.1	57.9	64.4	62.6	59.4	64.5	63.3	59.1	64.9	63.4	0.1%
MAAC	PA, NJ, MD, DC, DE	80.4	59.3	69.4	72.2	61.0	72.0	72.7	61.3	72.1	72.9	0.2%
MAIN	IL, MO, WI	61.2	52.6	57.8	61.0	53.9	60.4	62.0	54.1	60.5	62.3	0.5%
MAPP	MN, IA, SD, ND, NE	57.4	52.8	49.3	47.6	52.9	49.6	48.0	53.0	49.6	48.2	0.5%
NY	NY	104.3	82.8	87.9	88.1	83.3	88.9	88.5	83.3	89.1	88.7	0.4%
NE	VT, NH, ME, MA, CT, RI	89.9	77.4	83.9	82.8	77.4	84.7	83.1	77.5	84.8	83.1	0.0%
FRCC	FL	67.9	71.2	71.3	69.5	71.7	72.3	70.5	71.8	72.3	70.7	0.3%
STV	VA, NC, SC, GA, AL, MS, TN, AR, LA	59.3	56.2	55.1	55.3	57.0	56.2	56.6	57.1	56.2	56.8	0.3%
SPP	KS, OK, MO	59.3	54.2	57.0	56.7	54.6	57.5	57.0	54.6	57.5	57.2	0.3%
PNW	WA, OR, ID	45.9	49.6	47.4	46.9	49.8	47.5	46.9	49.8	47.5	47.1	0.5%
RM	MT, WY, CO, UT, NM, AZ, NV, ID	64.1	63.9	65.2	64.7	64.1	65.6	65.4	64.2	65.6	65.1	-0.4%
CALI	CA	94.7	97.1	98.9	99.3	97.3	99.1	99.5	97.3	99.1	99.7	0.3%
National	Contiguous Lower 48 States	66.0	60.3	63.1	63.4	61.3	64.5	64.3	61.3	64.5	64.5	0.2%

Sorbent Sensitivity

			Base Case			CAIR			Sensitivity			Change from CAIR
Power Region	**Primary States Included**	**2000**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2020**
ECAR (1)	OH, MI, IN, KY, WV, PA	57.4	51.7	55.2	56.1	53.7	58.6	58.0	53.8	58.6	58.0	0.1%
ERCOT (2)	TX	65.1	57.9	64.4	62.6	59.4	64.5	63.3	59.2	64.9	63.4	0.2%
MAAC (3)	PA, NJ, MD, DC, DE	80.4	59.3	69.4	72.2	61.0	72.0	72.7	61.1	72.0	72.9	0.2%
MAIN (4)	IL, MO, WI	61.2	52.6	57.8	61.0	53.9	60.4	62.0	54.0	60.5	62.3	0.4%
MAPP (5)	MN, IA, SD, ND, NE	57.4	52.8	49.3	47.6	52.9	49.6	48.0	52.9	49.6	48.0	0.1%
NY (6)	NY	104.3	82.8	87.9	88.1	83.3	88.9	88.5	83.3	89.1	88.8	0.3%
NE (7)	VT, NH, ME, MA, CT, RI	89.9	77.4	83.9	82.8	77.4	84.7	83.1	77.5	85.0	83.0	-0.1%
FRCC (8)	FL	67.9	71.2	71.3	69.5	71.7	72.3	70.5	71.8	72.3	70.7	0.2%
STV (9)	VA, NC, SC, GA, AL, MS, TN, AR, LA	59.3	56.2	55.1	55.3	57.0	56.2	56.6	57.1	56.2	56.7	0.3%
SPP (10)	KS, OK, MO	59.3	54.2	57.0	56.7	54.6	57.5	57.0	54.6	57.5	57.2	0.3%
PNW (11)	WA, OR, ID	45.9	49.6	47.4	46.9	49.8	47.5	46.9	49.8	47.5	47.2	0.5%
RM (12)	MT, WY, CO, UT, NM, AZ, NV, ID	64.1	63.9	65.2	64.7	64.1	65.6	65.4	64.1	65.6	65.3	-0.2%
CALI (13)	CA	94.7	97.1	98.9	99.3	97.3	99.1	99.5	97.2	99.1	99.7	0.22%
National	Contiguous Lower 48 States	66.0	60.3	63.1	63.4	61.3	64.5	64.3	61.3	64.5	64.4	0.2%

Source: Retail Electricity Price Model run by EPA. 2000 prices from EIA's AEO 2003.

7.15 Sensitivity Analysis on Assumptions for Natural Gas Prices and Electricity Growth

Sensitivity analyses were performed using projections from the 2004 Annual Energy Outlook produced by the Energy Information Administration (EIA). EPA used EIA estimates for the difference between natural gas prices and coal prices, which we have short-handed as "EIA natural gas prices," as well as EIA's projection of electricity growth. These particular assumptions involve considering the higher differential between minemouth coal and wellhead natural gas prices. For the years 2010, 2015, and 2020, there was a higher differential of $0.25 mmBtu, $0.42 mmBtu, and $0.38 mmBtu, respectively. The electricity growth was changed to match EIA's growth of 1.8 percent a year rather than EPA's growth of 1.6 percent.

Nationwide emissions of Hg, SO_2, and NOx using EIA assumptions are presented in Tables 7-26 and 7-27. Mercury emissions profiles with EIA assumptions are similar and lower than emissions with EPA assumptions. Lower Hg emissions for EIA assumptions can be attributed to the building of new and cleaner coal-fired capacity.

Total annual costs and present value costs of CAMR incremental to CAIR with EIA assumptions are in Table 7-28. The costs of CAMR with EIA assumptions for natural gas prices and electricity growth in 2010 and 2015 are only slightly different from costs of CAIR without those assumptions and can be attributed to the building of new and cleaner coal-fired capacity that leads to lower overall costs (see Tables 7-28 and 7-29). As demand continues to grow, coal-fired generation continues to increase and requires the use of additional scrubbers. Although more pollution controls are installed using EIA assumptions, dispatch changes lead to the use of more efficient generation. The power sector is less inclined to use gas as a compliance option in the region because of the higher operating cost. Once the power sector passes the point where there is no longer excess gas capacity in the marketplace (as currently exists), new coal-fired capacity is the logical choice to meet demand.

Coal-fired generation under CAMR increases using EIA assumptions for natural gas prices and electricity growth. Table 7-31 shows the generation mix with EIA assumptions. Coal production patterns change slightly and production for all three major coal-producing regions is higher, because coal-fired generation is a cheaper source of electricity than natural gas in most parts of the country with the higher EIA prices, even as more pollution controls are added to coal-fired generation and used to meet the additional electricity demand (see Table 7-32).

Electricity prices are not greatly altered with EIA assumptions for natural gas and electricity growth (see Table 7-33). Average electricity prices are projected to be lower than current levels (2000) using both EPA and EIA assumptions for natural gas and electricity growth.

Table 7-26. Projected Emissions of Hg for the New Base Case (CAIR) and CAMR with EPA and EIA Assumptions for Natural Gas Prices and Electric Growth (Tons)

		2010	2015	2020
Old Base Case	EPA Assumptions	46.6	45.0	46.2
	EIA Assumptions	47.5	47.0	47.8
New Base Case: CAIR	EPA Assumptions	38.0	34.4	34.0
	EIA Assumptions	38.3	35.2	35.4
Option 1	EPA Assumptions	31.3	27.9	24.3
	EIA Assumptions	31.5	28.5	23.5

Note: The emissions data presented here are EPA modeling results.

Table 7-27. Projected Nationwide Emissions of SO_2 and NO_x under the New Base Case (CAIR) and CAMR with EPA and EIA Assumptions for Natural Gas and Electric Growth (Million Tons)

	SO_2			NOx		
	2010	**2015**	**2020**	**2010**	**2015**	**2020**
Old Base Case with EPA Assumptions	9.7	8.9	8.6	3.6	3.7	3.7
New Base Case (CAIR) with EPA Assumptions	6.1	5.0	4.3	2.5	2.2	2.2
Option 1 with EPA Assumptions	6.1	4.9	4.2	2.4	2.2	2.2
Old Base Case with EIA Assumptions	9.7	8.8	8.6	3.7	3.8	3.8
New Base Case (CAIR) with EIA Assumptions	6.1	5.0	4.0	2.4	2.1	2.2
Option 1 with EIA Assumptions	6.1	4.9	4.3	2.4	2.2	2.2

Source: Integrated Planning Model run by EPA.

Table 7-28. Annualized Cost and Present Value Cost Incremental to the New Base Case (CAIR) with EPA and EIA Assumptions for Natural Gas Prices and Electric Growth (Billion $1999)

	2010	**2015**	**2020**	**Present value (2007-2025)**
Option 1- EPA Assumptions	$0.16	$0.10	$0.75	$3.9
Option 1 - EIA Assumptions	$0.16	$0.21	$0.53	$3.1

Note: Annual incremental costs of CAIR with EPA assumptions are $2.4 billion in 2010, $3.6 billion in 2015, and $4.4 billion in 2020, present value (2007-2025) is $41.1 billion. Annual incremental costs of CAIR with EIA assumptions are $2.6 billion in 2010, $3.4 billion in 2015, and $4.1 billion in 2020, present value (2007-2025) is $42.9 billion.
Note: Numbers rounded to the nearest tenth million for annualized cost.
Source: Integrated Planning Model run by EPA.

Table 7-29. Marginal Cost of SO_2 and NO_x Reductions under the New Base Case (CAIR) and CAMR with EPA and EIA Assumptions for Natural Gas Prices and Electric Growth ($/ton, in $1999)

			2010	**2015**	**2020**
New Base Case: CAIR	SO_2	EPA Assumptions	$800	$1,000	$1,300
		EIA Assumptions	$800	$1,200	$1,500
	NOx	EPA Assumptions	$1,300	$1,600	$1,600
		EIA Assumptions	$1,400	$1,700	$1,700
Option 1	SO_2	EPA Assumptions	$700	$900	$1,200
		EIA Assumptions	$800	$1,000	$1,300
	NOx	EPA Assumptions	$1,200	$1,500	$1,200
		EIA Assumptions	$1,200	$1,600	$1,300
	Hg	EPA Assumptions	$23,200	$30,100	$39,000
		EIA Assumptions	$26,400	$34,200	$44,400

Source: Integrated Planning Model run by EPA.

Table 7-30. Pollution Controls under the New Base Case (CAIR) with EPA and EIA Assumptions for Natural Gas and Electricity Growth (GWs)

		EPA Assumptions			EIA Assumptions		
	Technology	2010	2015	2020	2010	2015	2020
New Base Case: CAIR	FGD	146	177	198	157	185	209
	SCR	125	151	153	134	161	162
Option 1	FGD	146	179	199	155	187	203
	SCR	126	153	156	137	160	162
	ACI	2	3	13	3	4	26

Note: Retrofits include existing scrubbers and SCR as well as additional retrofits for the Title IV Acid Rain Program, the NO_x SIP call, NSR settlements, and various state rules.
Source: Integrated Planning Model run by EPA.

Table 7-31. Generation Mix under the New Base Case (CAIR) and CAMR with EPA and EIA Assumptions for Natural Gas and Electric Growth (Thousand GWhs)

		EPA Assumptions			EIA Assumptions		
	Fuel	2010	2015	2020	2010	2015	2020
Old Base Case	Coal	2,198	2,242	2,410	2,243	2,638	3,048
	Oil/Natural Gas	777	1,026	1,221	902	867	873
	Other	1,223	1,235	1,218	1,224	1,235	1,224
	Total	4,198	4,503	4,850	4,369	4,739	5,145
New Base Case: CAIR	Coal	2,165	2,197	2,384	2,228	2,632	3,045
	Oil/Natural Gas	807	1,069	1,247	916	871	874
	Other	1,217	1,232	1,217	1,223	1,234	1,221
	Total	4,190	4,498	4,848	4,367	4,738	5,141
Option 1	Coal	2,160	2,194	2,365	2,221	2,616	3,014
	Oil/Natural Gas	812	1,072	1,265	922	887	904
	Other	1,216	1,233	1,217	1,222	1,235	1,219
	Total	4,188	4,499	4,847	4,366	4,738	5,138

Note: Numbers may not add due to rounding.
Source: Integrated Planning Model run by EPA.

Table 7-32. Coal Production for the Electric Power Sector under the New Base Case (CAIR) and CAMR with EPA and EIA Assumptions for Natural Gas and Electricity Growth (Million Tons)

			EPA Assumptions			EIA Assumptions		
Supply Area	2000	2003	2010	2015	2020	2010	2015	2020

Old Base Case	Appalachia	299	275	325	315	301	328	341	340
	Interior	131	135	161	162	173	161	182	247
	West	475	526	603	631	714	626	748	840
	National	905	936	1,089	1,109	1,188	1,115	1,271	1,428
New Base Case: CAIR	Appalachia	299	275	306	310	331	320	367	390
	Interior	131	135	164	193	219	174	207	260
	West	475	526	607	579	607	614	676	765
	National	905	936	1,077	1,082	1,156	1,109	1,250	1,415
Option 1	Appalachia	299	275	303	310	330	317	377	396
	Interior	131	135	169	194	224	179	209	269
	West	475	526	589	568	572	595	639	706
	National	905	936	1,061	1,071	1,127	1,091	1,225	1,371

Source: 2000 and 2003 data are from EIA. All projections are from the Integrated Planning Model run by EPA.

Table 7-33. Retail Electricity Prices by NERC Region for the Base Case (No Further Controls), CAIR, and CAMR with EPA and EIA Assumptions for Natural Gas and Electricity Growth (Mills/kWh) ($1999)

EPA Assumptions for Natural Gas and Electricity Growth

			Base Case			CAIR			Option 1			Change from CAIR
Power Region	**Primary States Included**	**2000**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2020**
ECAR (1)	OH, MI, IN, KY, WV, PA	57.4	51.7	55.2	56.1	53.8	58.5	58.0	53.9	58.7	58.1	0.2%
ERCOT (2)	TX	65.1	57.9	64.4	62.6	59.3	64.6	63.3	59.1	64.9	63.4	0.1%
MAAC (3)	PA, NJ, MD, DC, DE	80.4	59.3	69.4	72.2	61.2	71.7	72.8	61.3	72.1	72.9	0.2%
MAIN (4)	IL, MO, WI	61.2	52.6	57.8	61.0	54.0	60.3	62.0	54.1	60.5	62.3	0.5%
MAPP (5)	MN, IA, SD, ND, NE	57.4	52.8	49.3	47.6	52.9	49.6	48.0	53.0	49.6	48.2	0.5%
NY (6)	NY	104.3	82.8	87.9	88.1	83.3	88.8	88.4	83.3	89.1	88.7	0.4%
NE (7)	VT, NH, ME, MA, CT, RI	89.9	77.4	83.9	82.8	77.5	84.7	83.0	77.5	84.8	83.1	0.2%
FRCC (8)	FL	67.9	71.2	71.3	69.5	71.7	72.3	70.5	71.8	72.3	70.7	0.3%
STV (9)	VA, NC, SC, GA, AL, MS, TN, AR, LA	59.3	56.2	55.1	55.3	57.0	56.2	56.6	57.1	56.2	56.8	0.3%
SPP (10)	KS, OK, MO	59.3	54.2	57.0	56.7	54.6	57.5	57.0	54.6	57.5	57.2	0.3%
PNW (11)	WA, OR, ID	45.9	49.6	47.4	46.9	49.8	47.5	46.9	49.8	47.5	47.1	0.5%
RM (12)	MT, WY, CO, UT, NM, AZ, NV, ID	64.1	63.9	65.2	64.7	64.1	65.6	65.4	64.2	65.6	65.1	-0.4%
CALI (13)	CA	94.7	97.1	98.9	99.3	97.3	99.1	99.5	97.3	99.1	99.7	0.3%
National	Contiguous Lower 48 States	66.0	60.3	63.1	63.4	61.3	64.4	64.3	61.3	64.5	64.5	0.2%

EIA **Assumptions for Natural Gas and Electricity Growth**

			Base Case			CAIR			Option 1			Change from CAIR
Power Region	**Primary States Included**	**2000**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2010**	**2015**	**2020**	**2020**
ECAR (1)	OH, MI, IN, KY, WV, PA	57.4	53.5	59.8	57.1	55.3	61.5	58.8	55.5	61.7	59.2	0.6%
ERCOT (2)	TX	65.1	63.3	66.0	64.4	63.6	66.6	65.0	63.5	66.9	65.2	0.3%
MAAC (3)	PA, NJ, MD, DC, DE	80.4	63.1	74.7	72.8	64.0	75.4	73.7	64.0	75.6	73.7	0.0%
MAIN (4)	IL, MO, WI	61.2	54.9	63.8	62.4	55.9	65.2	63.3	56.0	65.2	63.5	0.3%
MAPP (5)	MN, IA, SD, ND, NE	57.4	52.9	49.6	48.1	53.1	49.9	48.6	53.1	50.0	48.9	0.6%
NY (6)	NY	104.3	89.0	91.3	87.8	89.1	91.9	88.8	89.1	91.7	89.0	0.2%
NE (7)	VT, NH, ME, MA, CT, RI	89.9	85.1	85.5	81.2	84.7	85.9	81.8	84.6	86.0	82.5	0.8%
FRCC (8)	FL	67.9	72.5	74.6	73.7	73.3	75.3	74.3	73.4	75.5	74.4	0.0%
STV (9)	VA, NC, SC, GA, AL, MS, TN, AR, LA	59.3	57.1	57.1	57.1	57.8	58.3	58.6	57.8	58.2	58.8	0.4%
SPP (10)	KS, OK, MO	59.3	56.2	59.5	57.9	56.7	59.7	58.1	56.7	59.9	58.7	1.0%
PNW (11)	WA, OR, ID	45.9	50.4	50.0	49.9	50.7	50.2	49.9	50.3	49.9	49.5	-0.7%
RM (12)	MT, WY, CO, UT, NM, AZ, NV, ID	64.1	66.0	67.9	66.6	66.3	68.0	66.4	66.3	68.2	66.9	0.8%
CALI (13)	CA	94.7	99.5	101.4	101.8	99.6	101.5	101.8	99.9	101.8	102.0	0.2%

National	Contiguous Lower 48 States	66.0	62.8	66.1	64.9	63.5	67.0	65.8	63.6	67.1	66.1	0.5%

Source: Retail Electricity Price Model run by EPA. 2000 prices from EIA's AEO 2003.

7.16 Small Entity Impacts

The Regulatory Flexibility Act (5 U.S.C. § 601 et seq.), as amended by the Small Business Regulatory Enforcement Fairness Act (Public Law No. 104-121), provides that whenever an agency is required to publish a general notice of proposed rulemaking, it must prepare and make available an initial regulatory flexibility analysis, unless it certifies that the proposed rule, if promulgated, will not have "a significant economic impact on a substantial number of small entities" (5 U.S.C. § 605[b]). Small entities include small businesses, small organizations, and small governmental jurisdictions.

For the purposes of assessing the impacts of CAMR on small entities, a small entity is defined as:

(1) A small business according to the Small Business Administration size standards by the North American Industry Classification System (NAICS) category of the owning entity. The range of small business size standards for electric utilities is 4 billion kilowatt-hours of production or less;
(2) a small government jurisdiction that is a government of a city, county, town, district, or special district with a population of less than 50,000; and
(3) a small organization that is any not-for-profit enterprise that is independently owned and operated and is not dominant in its field.

Table 7-34 lists entities potentially affected by this proposed rule with applicable NAICS code.

Table 7-34. Potentially Regulated Categories and Entities[a]

Category	NAICS Code[b]	Examples of Potentially Regulated Entities
Industry	221112	Coal-fired electric utility steam generating units.
Federal Government	221112[c]	Coal-fired electric utility steam generating units owned by the federal government.
State/Local/ Tribal Government	221112[c]	Coal-fired electric utility steam generating units owned by municipalities.
	921150	Coal-fired electric utility steam generating units in Indian Country.

[a] Include NAICS categories for source categories that own and operate electric generating units only.
[b] North American Industry Classification System.
[c] Federal, state, or local government-owned and operated establishments are classified according to the activity in which they are engaged.

Courts have interpreted the RFA to require a regulatory flexibility analysis only when small entities will be subject to the requirements of the rule.[12] In the January 30, 2004 Notice of Proposed Rulemaking (NPR) EPA determined that the proposed rule would not have a significant impact on a substantial number of small entities. However, to provide additional information to States and affected sources, EPA conduct a general analysis of the potential economic impact of CAMR on small entities.

EPA examined the potential economic impacts to small entities associated with this rulemaking based on assumptions of how the affected states will implement control measures to meet their NO_x and SO_2 budgets under the Clean Air Interstate Rule (CAIR) and their Hg budgets for EGUs under CAMR. Under CAMR, States have the option of either participating in an EPA-run trading program, or implementing their Hg budget as a strict cap on Hg emissions from EGUs. This analysis assumes that all affected States in the CAIR region choose to meet their CAIR budgets by controlling EGUs only, and that all States participate in the nationwide Hg cap-and-trade program. This analysis does not examine potential indirect economic impacts associated with CAIR or CAMR, such as employment effects in industries providing fuel and pollution control equipment, or the potential effects of electricity price increases on industries and households. Because CAMR is implemented in conjunction with CAIR, the costs of CAMR are measured incrementally to the costs of CAIR alone.

This analysis presents the annualize cost of CAMR for the year 2020, which is two years into the second phase of the Hg cap-and-trade program, and for which the Hg emission cap is 15 tons. An important caveat to note in considering the results presented in this section is (as discussed earlier in this chapter) that EPA assumes no development in control technologies over the course of the Hg cap-and-trade program. In reality, Hg emissions control is a fast moving area with new developments nearly monthly. Actual costs may be lower than those presented since modeling assumes no improvements in the cost of mercury control technology, while in reality, control costs are expected to improve over time. As a result, this the projected costs of the Hg cap-and-trade program for 2020 presented in this analysis most certainly overstate the impact of the rule on small entities during the second phase of the program. At the same time, however, the marginal cost projected for mercury control in 2020 may also overstate the cost-savings that entities selling allowances may experience under the rule. Finally, it should be noted that during the first phase of the program, the fact that the cap is equal to co-benefits under CAIR should limit the impact of CAMR on small entities.

7.16.1 Identification of Small Entities

EPA used EGRID data as a basis for compiling the list of potentially affected small entities. EGRID is EPA's Emissions & Generation Resource Integrated Database, which contains emissions and resource mix data for virtually every power plant and company that generates electricity in the United States.[13] The data set contains detailed ownership and

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[12] See Michigan v. EPA, 213 F.3d 663, 668-69 (D.C. Cir. 2000), cert. den. 121 S.Ct. 225, 149 L.Ed.2d 135 (2001). An agency's certification need consider the rule's impact only on entities subject to the rule.

[13] eGRID is available at http://www.epa.gov/cleanenergy/egrid/download.htm.

corporate affiliation information. For plants burning coal as the primary fuel, plant-level boiler and generator capacity, heat input, generation, and emissions data were aggregated by owner and then parent company. Entities with more than 4 billion kWh of annual electricity generation were removed from the list, as were municipal-owned entities serving a population greater than 50,000. Finally, for cooperatives, investor-owned utilities, and subdivisions that generate less than 4 billion kWh of electricity annually but may be part of a large entity, additional research on power sales, operating revenues, and other business activities was performed to make a final determination regarding size. Because the rule does not affect units with a generating capacity of 25 MW or less, small entities that do not own at least one coal-fired generating unit with a capacity greater than 25 MW were dropped from the data set. According to EPA's analysis, approximately 35 small entities were exempted by this provision. EPA identified a total of 81 potentially affected small entities, out of a possible 116. The number of potentially affected small entities by ownership type, and summary of projected impacts, is listed in Table 7-35.

Table 7-35. Projected Impact of CAMR on Small Entities

EGU Ownership Type	Number of Potentially Affected Entities	Total Net Compliance Cost in 2020 Incremental to CAIR ($1999 millions)	Number of Small Entities with Compliance Costs >1% of Generation Revenues in 2020	Number of Small Entities with Compliance Costs >3% of Generation Revenues in 2020
Cooperative	21	8.5	7	1
Investor-Owned Utility	2	6.4	2	0
Municipal	48	15.2	28	11
Subdivision	8	6.3	5	2
Other	1	-0.003	0	0
Total	80	36.5	42	14

Note: The total number of potentially affected entities in this table excludes the 35 entities that have been dropped because they will not be affected by CAMR. Also, the total number of entities with costs greater than 1 percent or 3 percent of revenues includes only entities experiencing positive costs.

Source: IPM and TRUM analysis

7.16.2 Overview of Analysis and Results

This section presents the methodology and results for estimating the impact of CAMR to small entities in 2020 based on the following endpoints:

- annual economic impacts of CAMR on small entities and
- ratio of small entity impacts to revenues from electricity generation.

7.16.2.1 Methodology for Estimating Impacts of CAMR on Small Entities

An entity can comply with CAMR through some combination of the following: installing retrofit technologies, purchasing allowances, switching to a lower Hg fuel, or reducing emissions through a reduction in generation. Additionally, units with more allowances than needed can sell these allowances on the market. The chosen compliance strategy will be primarily a function of the unit's marginal control costs and its position relative to the marginal control costs of other units. Because CAMR will be implemented in conjunction with CAIR, units affected by both rules will attempt to minimize their cost of compliance over both rules, by considering Hg, SO_2, and NO_x control strategies simultaneously.

To attempt to account for each potential control strategy over the combined rules, EPA estimates compliance costs as follows:

$$C_{Compliance} = \Delta\, C_{Operating+Retrofit} + \Delta\, C_{Fuel} + \Delta\, C_{Allowances} + \Delta\, C_{Transaction} - \Delta\, R \qquad (8.1)$$

where C represents a component of cost as labeled, and Δ R represents the retail value of foregone electricity generation.

In reality, compliance choices and market conditions can combine such that an entity may actually experience a savings in any of the individual components of cost. Under CAIR and CAMR, for example, EPA projects that the price of low-sulfur coal will fall as many units install scrubbers and switch away from low-sulfur coal to cheaper bituminous coal, such that many entities actually experience a reduction in fuel costs relative to the base caes as a result of lower prices due to the demand shift. Similarly, although some units will forgo some level of electricity generation (and thus revenues) to comply, this impact will be lessened on these entities by the projected increase in electricity prices under CAIR and CAMR as well as reductions in fuel costs, and those not reducing generation levels will see an increase in electricity revenues. Elsewhere, unscrubbed units burning low-sulfur coal might find it most economical to install mercury-specific controls such as ACI, and sell their surplus of Hg allowances on the market. Because this analysis evaluates the total costs along each of the four compliance strategies laid out above for each entity, it inevitably captures savings or gains such as those described. As a result, what we describe as cost is really more of a measure of the net economic impact of the rule on small entities.

For this analysis, EPA used IPM-parsed output to estimate net compliance costs at the unit level. These impacts were then summed for each small entity, adjusting for ownership share. Net impact estimates were based on the following: operating and retrofit costs, sale or purchase of allowances, and the change in fuel costs or electricity generation revenues under CAMR relative to CAIR. These individual components of compliance cost were estimated as follows:

(1) **Operating and retrofit costs:** Using the IPM-parsed output for the base case, CAIR, and CAMR (available in the docket), EPA identified units that install control technology under CAIR and CAMR and the technology installed. The equations for calculating retrofit costs were adopted from EPA's Technology Retrofit and Updating Model (TRUM). The model calculates the capital cost (in \$/MW); the fixed operation and maintenance (O&M) cost (in \$/MW-year); the

variable O&M cost (in $/MWh); and the total annualized retrofit cost for units projected to install FGD, SCR, SNCR, or ACI.

(2) **Sale or purchase of allowances:** EPA estimated the value of initial SO_2, NO_x, and Hg allowance holdings. For SO_2, units were assumed to retain their Phase II allowance allocations as determined under EPA's 1998 reallocation of Acid Rain allowances, adjusted to reflect the 50 percent reduction in 2010 and 65 percent reduction in 2015 under CAIR. Because of the resources involved in compiling allowance-holding data, the value of banked SO_2 allowances was not considered in this analysis. The implication of this is that the annual net purchase of allowances may be overstated for some units. For NO_x, the state emission budgets were assumed to be apportioned to units on a heat-input basis. Each unit was assumed to receive a share of the state NO_x emission budget equal to its share of the total state heat input for that year in the base case. This is a simplification of what is included in the model rule, which proposes allocating NO_x allowances based on heat input from 1999-2002.[14] However, states can ultimately decide how to allocate NO_x allowances. For Hg, unit allocations were the same as those listed in the March 16, 2004 Supplemental Notice of Proposed Rulemaking.

To estimate the value of allowances holdings, allocated NOx and SO_2 allowances were subtracted from projected emissions, and the difference was then multiplied by the allowance prices projected by IPM for 2020. Units were assumed to purchase or sell allowances to exactly cover their projected NO_x and SO_2 emissions under CAIR + CAMR. For Hg, units that did not have allowances sufficient to cover projected 2020 emissions were projected to withdraw allowances from their respective Hg allowance banks if available, or else purchase the required amount of allowances. Units holding 2020 allowances in excess of projected 2020 emissions were projected to sell these excess allowances. The estimation of the size of a unit's mercury allowance bank is discussed further below.

(3) **Fuel costs:** Fuel costs were estimated by multiplying fuel input (MMBtu) by region and fuel-type-adjusted fuel prices ($/MMBtu) from TRUM. The change in fuel expenditures under CAMR was then estimated by taking the difference in fuel costs between CAMR and CAIR.

(4) **Value of electricity generated:** EPA estimated electricity generation by first estimating unit capacity factor and maximum fuel capacity. Unit capacity factor is estimated by dividing fuel input (MMBtu) by maximum fuel capacity (MMBtu). The maximum fuel capacity was estimated by multiplying capacity (MW) * 8,760 operating hours * heat rate (MMBtu/MWh). The value of electricity generated is then estimated by multiplying capacity (MW)*capacity factor*8,760*regional-adjusted retail electricity price ($/MWh).

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[14] A similar approach was used in regulatory impact analyses for the 126 FIP and NO_x SIP Call.

As discussed later in this analysis, many small entities projected to be affected by CAMR do not have to operate in a competitive market environment and thus should be able to pass compliance costs on to consumers. To somewhat account for this, we incorporated the projected regional-adjusted retail electricity price calculated under CAMR in our estimation of generation revenue under CAMR.

(5) **Administrative costs:** Because most affected units are already monitored as a result of other regulatory requirements, EPA considered the primary administrative cost to be transaction costs related to purchasing or selling allowances. EPA assumed that transaction costs were equal to 1.5 percent of the total absolute value of a unit's allowances. This assumption is based on market research by ICF Consulting.

(6) **Value of the Mercury Bank:** EPA's economic analysis of CAMR suggests that a significant bank of approximately 70 tons of Hg allowances will be built up during the first phase of the cap-and-trade program. Sources will be relying heavily on this bank for compliance during the second phase of the program. While not all sources will have banked allowances during the first phase of the program, many sources will be able to draw from this bank during the second phase and avoid or limit Hg allowance purchases. EPA estimated the size of the bank by comparing projected emissions for the years 2010-2019 with allocations for those years. This estimate assumed that small entity sources with surplus allowances in those years would bank those allowances rather than sell them on the market, and would draw from this bank in any year that they were short allowances. EPA estimated the cost of using banked allowances by taking the average cost of Hg control in the first phase of the program discounted to 2020, multiplied by the number of banked allowances used. Finally, any surplus allowances remaining in the small entity banks in 2020 were valued at the 2020 Hg allowance price.

7.16.2.2 Results

The potential impacts of CAMR on small entities are summarized in Table 7-35. All costs are presented in \$1999. EPA estimated the incremental annualized net compliance cost to small entities to relative to CAIR to be approximately \$37 million in 2020. This cost is driven largely by mercury allowance purchases and additional retrofits relative to CAIR. The costs to small entities in 2020 are limited, however, by the ability of approximately 30 of the 81 small entities to sell surplus 2020 and/or banked allowances in 2020.

EPA does not project that any coal-fired generation would be uneconomic to maintain relative to CAIR. This finding suggests that the extent of CAMR's adverse economic impacts beyond CAIR on small entities is limited.

EPA further assessed the economic and financial impacts of the rule using the ratio of compliance costs to the value of revenues from electricity generation, focusing in particular on entities for which this measure is greater than 1 percent. Although this metric is commonly used in EPA impact analyses, it makes the most sense when as a general matter an analysis is looking at small businesses that operate in competitive environments. However, small businesses in the electric power industry often operate in a price-regulated environment where they are able to recover expenses through rate increases. Given this, EPA considers the 1 percent measure in this case a crude measure of the price increases these small entities will be asking of rate commissions or making at publicly owned companies.

Of the 80 small entities considered in this analysis, and 116 total small entities in the affected region 42 were projected to have compliance costs greater than 1% of revenues, while 14 were projected to have compliance costs greater than 3% of revenues. As was emphasized earlier, this result is largely due to the magnitude of the projected marginal Hg control cost in 2020. A marginal cost similar to what was projected in the sensitivity analysis discussed earlier in this chapter would eliminate significant impacts. Furthermore, the majority of small entities in this analysis operate in a competitive market and thus should be able to recover their costs of complying with CAMR. It should also be emphasized that under CAMR, states, through their choice of Hg allowance allocation methodologies, can potentially mitigate adverse affects of CAIR on small entities.

The distribution across entities of economic impacts as a share of base case revenue is summarized in Table 7-36. Although the distributions of economic impacts on each ownership type are in general fairly tight. Entities with the lowest negative net impacts are those that have complied with the Hg rule without additional retrofits, and have a number of surplus Hg allowances for sale. On average, the impact of the rule on small entities is less than 1% of electricity generation revenues.

Table 7-36. Summary of Distribution of Economic Impacts of CAIR on Small Entities

EGU Ownership Type	Capacity-Weighted Average Economic Impacts as a % of Generation Revenues	Min	Max
Cooperative	0.52 %	-6.30 %	4.7 %
Investor-owned utility	1.94 %	1.48 %	2.22 %
Municipal	1.21 %	-5.30 %	6.39 %
Subdivision	1.54 %	-0.52 %	3.31 %
Other	-0.09 %	-0.09 %	-0.09 %
All	**0.96 %**	**-6.30 %**	**6.4%**

Source: IPM and TRUM analysis

In the cases where entities are projected to experience positive net impacts that are a high percentage of revenues, these entities generally have a shortage of Hg allowances and must purchase them on the market at the projected 2020 price. Many of these entities also reduce generation slightly, and thus generation revenues, relative to CAIR alone.

The separate components of annualized costs to small entities under CAIR and CAIR + CAMR are summarized in Table 7-37. Under CAMR, allowance purchases, driven largely by the marginal cost projected for Hg in 2020, as well as additional retrofits, are the most significant components of compliance cost for small entities in 2020. Also, fuel costs under for all groups with the exception of IOUs increase relative to CAIR, largely because of an increased demand for bituminous coal and the resulting higher bituminous coal price relative to CAIR. Retrofit and operating costs for subdivisions, municipals, and cooperatives increase significantly, largely because of the installation of FGD, SCR and ACI. Finally, all groups with the exception of IOUs experience an increase in electricity revenues relative to CAIR alone. This increase is largely driven by increases in the retail price of electricity relative to CAIR alone, although a few units are projected to increase generation under CAIR + CAMR. The two IOUs in this analysis experience an increased revenue loss that results largely from generation reductions relative to CAIR in 2020.

Table 7-37. Incremental Annualized Costs under CAMR relative to CAIR, Summarized by Ownership Group and Cost Category ($1,000,000)

EGU Ownership Type	Retrofit + Operating Cost	Net Purchase of Allowances	Fuel Cost	Lost Electricity Revenue	Administrative Cost
Cooperative	4.9	2.9	2	-1.3	0.1
IOU	-0.1	4.1	-0.6	3	0.1
Municipal	6.2	10.3	6.3	-7.6	0.1
Subdivision	5.3	0.8	0.4	-0.2	0.1
Other	0	0	0	-0.1	0.001

Note: Numbers may not add to totals in Table 7-35 due to rounding.

Source: IPM and TRUM analysis.

7.16.3 Summary of Small Entity Impacts

While EPA has certified, based on earlier analysis that was summarized in the January 30, 2004 NPR, that CAMR will not have a significant impact on a substantial number of small entities, this analysis has been conducted to provide additional understanding of the nature of potential impacts, and additional information to the states as they propose plants to meet the emissions budgets set by this rulemaking.

EPA projects an incremental impact on small entities relative to CAIR of approximately $37 million relative to CAIR. EPA also projects that no additional small entity coal capacity will be uneconomic to maintain under CAMR relative to what was projected to be uneconomic to maintain under CAIR, which is the new base case. This finding suggests that the incremental impact of CAMR on small entities is limited.

Furthermore, of the 81small entities potentially affected, and the 116 small entities with in the country with coal units included in EPA's modeling, 42 may experience compliance costs in excess of 1 percent of revenues, while 14 are projected to experience compliance costs in excess of 3 percent of revenues, based on our assumptions of how the affected states implement control measures to meet their emissions budgets as set forth in this rulemaking. As is discussed earlier in this analysis, the finding of a significant impact to some entities during the second phase of the program is largely a product of the marginal cost projected for Hg control in 2020. In reality, control costs of Hg are expected to be lower by 2020, such that allowance prices would be reduced, and significant impacts unlikely. Further, the majority of these small entities operate in cost-of-service markets where they should be able to pass on their costs of compliance to rate-payers.

Two other points should be considered when evaluating the impact of CAMR, specifically, and cap-and-trade programs more generally, on small entities. First, under CAIR, the cap-and-trade program is designed such that states determine how Hg allowances are to be allocated across units. States electing to participate in the Hg cap-and-trade program could allocate allowances in a manner that would mitigate any potential disadvantage faced by small entities. Further, States that chose to implement their State budget as a strict cap could provide some level of exemption to sources owned by small entities, and require greater reductions from other sources. Finally, it should be noted that, the use of a cap-and-trade program in general will limit impacts on small entities relative to a less flexible command-and-control program.

7.17 Unfunded Mandates Reform Act (UMRA) Analysis

Title II of the UMRA of 1995 (Public Law 104-4)(UMRA) establishes requirements for federal agencies to assess the effects of their regulatory actions on state, local, and Tribal governments and the private sector. Under Section 202 of the UMRA, 2 U.S.C. 1532, EPA generally must prepare a written statement, including a cost-benefit analysis, for any proposed or final rule that "includes any Federal mandate that may result in the expenditure by State, local, and Tribal governments, in the aggregate, or by the private sector, of $100,000,000 or more ... in any one year." A "Federal mandate" is defined under Section 421(6), 2 U.S.C. 658(6), to include a "Federal intergovernmental mandate" and a "Federal private sector mandate." A "Federal intergovernmental mandate," in turn, is defined to include a regulation that "would impose an enforceable duty upon State, Local, or Tribal governments," Section 421(5)(A)(i), 2 U.S.C. 658(5)(A)(i), except for, among other things, a duty that is "a condition of Federal assistance," Section 421(5)(A)(i)(I). A "Federal private sector mandate" includes a regulation that "would impose an enforceable duty upon the private sector," with certain exceptions, Section 421(7)(A), 2 U.S.C. 658(7)(A).

Before promulgating an EPA rule for which a written statement is needed under Section 202 of the UMRA, Section 205, 2 U.S.C. 1535, of the UMRA generally requires EPA to identify and consider a reasonable number of regulatory alternatives and adopt the least costly, most cost-effective, or least burdensome alternative that achieves the objectives of the rule.

In the NPR, EPA concluded that the proposed Hg MACT contained a Federal Mandate that may result in expenditures of $100 million or more for State, local, and Tribal governments in aggregate, or the private sector in any one year. For that reason, EPA prepared a written statement for the NPR consistent with the requirements of Section 202 of the UMRA. In today's final rule, EPA is not directly establishing any regulatory requirements that may significantly or uniquely affect small governments, including Tribal governments. Thus, under CAMR, EPA is not obligated to develop under Section 203 of the UMRA a small government agency plan. Furthermore, in a manner consistent with the intergovernmental consultation provisions of Section 204 of the UMRA, EPA carried out consultations with the governmental entities affected by this rule.

EPA analyzed the economic impacts of the final CAMR. This analysis does not examine potential indirect economic impacts associated with CAIR, such as employment effects in industries providing fuel and pollution control equipment, or the potential effects of electricity price increases on industries and households.

This analysis presents the annualize cost of CAMR for the year 2020, which is two years into the second phase of the Hg cap-and-trade program, and for which the Hg emission cap is 15 tons. An important caveat to note in considering the results presented in this section is (as discussed earlier in this chapter) that EPA assumes no development in control technologies over the course of the Hg cap-and-trade program. In reality, Hg emissions control is a fast moving area with new developments nearly monthly. Actual costs may be lower than those presented since modeling assumes no improvements in the cost of mercury control technology, while in reality, control costs are expected to improve over time. As a result, this the projected costs of the Hg cap-and-trade program for 2020 presented in this analysis most certainly overstate the impact of the rule on government-owned entities during the second phase of the program. At the same time, however, the marginal cost projected for mercury control in 2020 may also overstate the cost-savings that entities selling allowances may experience under the rule. Finally, it should be noted that during the first phase of the program, the fact that the cap is equal to co-benefits under CAIR should limit the impact of CAMR on government entities.

7.17.1 Identification of Government-Owned Entities

Using eGRID data, EPA identified state- and municipality-owned utilities and subdivisions. EPA then used IPM-parsed output to associate these plants with individual generating units. Entities that did not own at least one unit with a generating capacity of greater than 25 MW were omitted from the analysis because of their exemption from the rule. This exempts 37 entities owned by state or local governments. Thus, EPA identified 88 state and municipality-owned utilities that are potentially affected by CAIR, out of a possible 125, which are summarized in Table 7-38.

Table 7-38. Summary of Potential Impacts on Government Entities under CAIR

EGU Ownership Type	Potentially Affected Entities	Net Compliance Cost in 2020 Incremental to CAIR ($1999 millions	Number of Government Entities with Compliance Costs >1% of Generation Revenues	Number of Government Entities with Compliance Costs >3% of Generation Revenues
Subdivision	8	6.5	5	2
State	10	9.2	4	0
Municipal	70	32.2	35	12
Total	88	47.9	44	14

Note: The total number of potentially affected entities in this table excludes the 37 entities that have been dropped because they will not be affected by CAMR. Also, the total number of entities with costs greater than 1 percent or 3 percent of revenues includes only entities experiencing positive costs.
Source: IPM and TRUM analysis

7.17.2 Overview of Analysis and Results

After identifying potentially affected government entities, EPA estimated the impact of CAMR + CAIR, relative to CAIR alone, in 2020 based on the following:

- total impacts of compliance on government entities and
- ratio of small entity impacts to revenues from electricity generation.

The financial burden to owners of EGUs under CAMR is composed of compliance and administrative costs. This section outlines the compliance and administrative costs for the 88 potentially affected government-owned units identified in EPA modeling.

7.17.2.1 Methodology for Estimating Impacts of CAMR on Government Entities

The primary burden on state and municipal governments that operate utilities under CAMR is the cost of installing control technology on units to meet their Hg emission budget or the cost of purchasing allowances. An entity can comply with CAMR through some combination of the following: installing retrofit technologies, purchasing allowances, switching to a lower Hg fuel, or reducing emissions through a reduction in generation. Additionally, units with more allowances than needed can sell these allowances on the market. The chosen compliance strategy will be primarily a function of the unit's marginal control costs and its position relative to the marginal control costs of other units. Because CAMR will be implemented in conjunction with CAIR, units affected by both rules will attempt to minimize their cost of compliance over both rules, by considering Hg, SO_2, and NO_x control strategies simultaneously.

To attempt to account for each potential control strategy over the combined rules, EPA estimates compliance costs as follows:

$$C_{Compliance} = \Delta\, C_{Operating+Retrofit} + \Delta\, C_{Fuel} + \Delta\, C_{Allowances} + \Delta\, C_{Transaction} - \Delta\, R \qquad (8.2)$$

where C represents a component of cost as labeled, and ΔR represents the retail value of foregone electricity generation.

In reality, compliance choices and market conditions can combine such that an entity may actually experience a savings in any of the individual components of cost. Under CAIR and CAMR, for example, EPA projects that the price of low-sulfur coal will fall as many units install scrubbers and switch away from low-sulfur coal to cheaper bituminous coal, such that many entities actually experience a reduction in fuel costs relative to the base caes as a result of lower prices due to the demand shift. Similarly, although some units will forgo some level of electricity generation (and thus revenues) to comply, this impact will be lessened on these entities by the projected increase in electricity prices under CAIR and CAMR as well as reductions in fuel costs, and those not reducing generation levels will see an increase in electricity revenues. Elsewhere, unscrubbed units burning low-sulfur coal might find it most economical to install mercury-specific controls such as ACI, and sell their surplus of Hg allowances on the market. Because this analysis evaluates the total costs along each of the four compliance strategies laid out above for each entity, it inevitably captures savings or gains such as those described. As a result, what we describe as cost is really more of a measure of the net economic impact of the rule on small entities.

In this analysis, EPA used IPM-parsed output for the base case, CAIR, and CAMR to estimate net compliance cost at the unit level. These costs were then summed for each government entity, adjusting for ownership share. Compliance cost estimates were based on the following: operating and retrofit costs, sale or purchase of allowances, and the change in fuel costs or electricity generation revenues under CAMR relative to CAIR. These components of compliance cost were estimated as follows:

(1) **Operating and retrofit costs:** Using the IPM-parsed output for the base case, CAIR, and CAMR (available in the docket), EPA identified units that install control technology under CAIR and CAMR and the technology installed. The equations for calculating retrofit costs were adopted from EPA's Technology Retrofit and Updating Model (TRUM). The model calculates the capital cost (in \$/MW); the fixed operation and maintenance (O&M) cost (in \$/MW-year); the variable O&M cost (in \$/MWh); and the total annualized retrofit cost for units projected to install FGD, SCR, SNCR, or ACI.

(2) **Sale or purchase of allowances:** EPA estimated the value of initial SO_2, NO_x, and Hg allowance holdings. For SO_2, units were assumed to retain their Phase II allowance allocations as determined under EPA's 1998 reallocation of Acid Rain allowances, adjusted to reflect the 50 percent reduction in 2010 and 65 percent reduction in 2015 under CAIR. Because of the resources involved in compiling allowance-holding data, the value of banked SO_2 allowances was not considered in this analysis. The implication of this is that the annual net purchase of allowances may be overstated for some units. For NO_x, the state emission budgets were assumed to be apportioned to units on a heat-input basis. Each unit was assumed to receive a share of the state NO_x emission budget equal to its share of the total state heat input for that year in the base case. This is a simplification of what is included in the model rule, which proposes allocating NO_x allowances

based on heat input from 1999-2002.[15] However, states can ultimately decide how to allocate NO_x allowances. For Hg, unit allocations were the same as those listed in the March 16, 2004 Supplemental Notice of Proposed Rulemaking.

To estimate the value of allowances holdings, allocated NOx and SO_2 allowances were subtracted from projected emissions, and the difference was then multiplied by the allowance prices projected by IPM for 2020. Units were assumed to purchase or sell allowances to exactly cover their projected NO_x and SO_2 emissions under CAIR + CAMR. For Hg, units that did not have allowances sufficient to cover projected 2020 emissions were projected to withdraw allowances from their respective Hg allowance banks if available, or else purchase the required amount of allowances. Units holding 2020 allowances in excess of projected 2020 emissions were projected to sell these excess allowances. The estimation of the size of a unit's mercury allowance bank is discussed further below.

(3) **Fuel costs:** Fuel costs were estimated by multiplying fuel input (MMBtu) by region and fuel-type-adjusted fuel prices ($/MMBtu) from TRUM. The change in fuel expenditures under CAMR was then estimated by taking the difference in fuel costs between CAMR and CAIR.

(4) **Value of electricity generated:** EPA estimated electricity generation by first estimating unit capacity factor and maximum fuel capacity. Unit capacity factor is estimated by dividing fuel input (MMBtu) by maximum fuel capacity (MMBtu). The maximum fuel capacity was estimated by multiplying capacity (MW) * 8,760 operating hours * heat rate (MMBtu/MWh). The value of electricity generated is then estimated by multiplying capacity (MW)*capacity factor*8,760*regional-adjusted retail electricity price ($/MWh).

As discussed later in this analysis, most government entities projected to be affected by CAMR do not have to operate in a competitive market environment and thus should be able to pass compliance costs on to consumers. To somewhat account for this, we incorporated the projected regional-adjusted retail electricity price calculated under CAMR in our estimation of generation revenue under CAMR.

(5) **Administrative costs:** Because most affected units are already monitored as a result of other regulatory requirements, EPA considered the primary administrative cost to be transaction costs related to purchasing or selling allowances. EPA assumed that transaction costs were equal to 1.5 percent of the total absolute value of a unit's allowances. This assumption is based on market research by ICF Consulting.

[15] A similar approach was used in regulatory impact analyses for the 126 FIP and NO_x SIP Call.

(6) **Value of the Mercury Bank:** EPA's economic analysis of CAMR suggests that a significant bank of approximately 70 tons of Hg allowances will be built up during the first phase of the cap-and-trade program. Sources will be relying heavily on this bank for compliance during the second phase of the program. While not all sources will have banked allowances during the first phase of the program, many sources will be able to draw from this bank during the second phase and avoid or limit Hg allowance purchases. EPA estimated the size of the bank by comparing projected emissions for the years 2010-2019 with allocations for those years. This estimate assumed that state and local government-owned sources with surplus allowances in those years would bank those allowances rather than sell them on the market, and would draw from this bank in any year that they were short allowances. EPA estimated the cost of using banked allowances by taking the average cost of Hg control in the first phase of the program discounted to 2020, multiplied by the number of banked allowances used. Finally, any surplus allowances remaining in the government entity banks in 2020 were valued at the 2020 Hg allowance price.

7.17.2.2 Results

A summary of economic impacts on government-owned entities is presented in Table 7-38. According to EPA's analysis, the total net economic impact on government-owned entities (state- and municipality-owned utilities and subdivisions) is expected to be approximately $48 million in 2020. This cost is driven largely by mercury allowance purchases and additional retrofits relative to CAIR. The costs to government entities in 2020 are limited, however, by the projection that 33 of the 88 entities sell surplus and/or banked allowances in 2020. In the absence of banked allowances, costs to these entities in 2020 would be greater.

EPA does not project that any coal-fired generation would be uneconomic to maintain relative to CAIR. This finding suggests that the extent of CAMR's adverse economic impacts beyond CAIR on small entities is limited.

As was done for the small entities analysis, EPA further assessed the economic and financial impacts of the rule using the ratio of compliance costs to the value of revenues from electricity generation in the base case, also focusing specifically on entities for which this measure is greater than 1 percent. EPA projects that 44 government entities will have compliance costs greater than 1 percent of revenues from electricity generation in 2020, and 12 entities are projected to have compliance costs greater than 3 percent of revenues. Entities that are projected to experience negative compliance costs under CAMR are not included in those totals. This approach is more indicative of a significant impact when an analysis is looking at entities operating in a competitive market environment. Government-owned entities do not operate in a competitive market environment and therefore will be able to recover expenses under CAIR and CAMR through rate increases. Given this, EPA considers the 1 percent measure in this case a crude measure of the extent to which rate increases will be made at publicly owned companies.

The distribution across entities of economic impacts as a share of base case revenue is summarized in Table 7-39. For state-owned entities and subdivisions, the maximum economic impact as a share of base case revenues is approximately 3 percent. A few municipality-owned entities experience economic impacts that are significantly higher than the capacity-weighted average for this group. In the cases where entities are projected to experience positive net costs that are a high percentage of revenues, these entities do not find it economic to retrofit and are unable to switch to a lower-sulfur coal. Thus, these entities comply primarily through the purchase of allowances and reductions in generation. Overall, the capacity-weighted average impact of the rule as a share of revenues is well under 1%.

Table 7-39. Distribution of Economic Impacts on Government Entities under CAMR

EGU Ownership Type	Capacity-Weighted Average Economic Impacts as a % of Generation Revenues	Min	Max
Sub-division	1.50 %	-0.52 %	3.31 %
State	0.30 %	-0.96 %	2.88 %
Municipal	0.38 %	-16.55 %	6.39 %
All	0.40 %	-16.55 %	6.39 %

Source: IPM and TRUM analysis

Additionally, a few municipal entities are projected to experience negative net costs that are a high percentage of base case generation revenues. These entities have units that are able to switch to a cheaper, lower-sulfur coal to comply with CAIR and are able to maintain or increase generation levels, thus increasing revenues. Further, entities in regions for which we project large electricity price increases relative to other regions tend to be among those at the lower end of the distribution.

The various components of annualized incremental cost under CAIR to each group of government entities are summarized in Table 7-40. Under CAMR, the most significant components of control costs for these entities are allowance purchases, driven largely by the marginal cost projected for Hg in 2020, as well as additional retrofits. Also, the increased demand for bituminous coal and the resulting higher bituminous coal price relative to CAIR leads to an increase in fuel costs for all groups. Retrofit and operating costs for all groups increase relative to CAIR alone, because of the installation of ACI, as well as some additional FGD and SCR. Finally, both states and municipals are projected to experience an increase in electricity generation revenues relative to CAIR alone, while subdivisions are projected to experience a slight additional drop in revenues relative to CAIR alone. Increased generation revenues are largely a result of slight increases in the retail price of electricity in most regions under CAMR, although some facilities are projected to increase generation. Subdivisions experience a loss in generation revenues because of a net decrease in electricity generation relative to CAIR that is not offset by the increase in electricity prices.

Table 7-40. Incremental Annualized Costs under CAMR Relative to CAIR Summarized by Ownership Group and Cost Category ($1,000,000)

EGU Ownership Type	Retrofit + Operating Cost	Net Purchase of Allowances	Fuel Cost	Lost Electricity Revenue	Administrative Cost
Subdivision	5.3	1.0	0.4	0.2	0.1
State	8.5	7.2	2.0	-8.6	0.2
Municipal	7.6	17.5	9.0	-2.1	0.3

Note: Numbers may not add to totals in Table 7-38 due to rounding.
Source: IPM and TRUM analysis.

7.17.3 Summary of Government Entity Impacts

EPA examined the potential economic impacts on state and municipality-owned entities associated with this rulemaking based on assumptions of how the affected states will implement control measures to meet their emissions. These impacts have been calculated to provide additional understanding of the nature of potential impacts and additional information to the states as they create State plans to meet the Hg emission budgets set by this rulemaking.

According to EPA's analysis, the total net economic impact on government-owned entities is expected to be approximately $48 million in 2020. These costs are driven largely by the purchase of Hg allowances and the cost of additional retrofits under the combination of CAIR and CAMR. EPA projects that no additional government entity capacity will be uneconomic to maintain under CAMR relative to what was projected to be uneconomic to maintain under CAIR. This suggests that the incremental impact of CAMR on small entities relative to CAIR alone is limited.

Of the 88 government entities considered in this analysis and the 125 government entities that are included in EPA's modeling, 44 are projected to experience compliance costs in excess of 1 percent of electricity generation revenues in 2020, and 14 of these are projected to experience compliance costs in excess of 3% of generation revenues. may in 2015, based on our assumptions of how the affected states implement control measures to meet their emissions budgets as set forth in this rulemaking. As is discussed earlier in this analysis, the finding of a significant impact to some entities during the second phase of the program is largely a product of the marginal cost projected for Hg control in 2020. In reality, control costs of Hg are expected to be lower by 2020, such that allowance prices would be reduced, and significant impacts unlikely. Further, government entities operate in cost-of-service markets where they should be able to pass on their costs of compliance to rate-payers. The above points aside, potential adverse impacts of CAMR on state- and municipality-owned entities could be limited by the fact that the cap-and-trade program is designed such that states determine how Hg allowances are to be allocated across units. A state that wishes to mitigate the impact of the rule on state- or municipality-owned entities might choose to allocate Hg allowances in a manner that is favorable to these entities. Finally, in general, the use of cap-and-trade programs in general will limit impacts on entities owned by small governments relative to a less flexible command-and-control program.

EPA has determined that this rule may result in expenditures of more than $100 million to the private sector in any single year. EPA believes that the final rule represents the least costly, most cost-effective approach to achieve the air quality goals of this rule. The costs and benefits associated with the final rule are discussed throughout this RIA.

7.18 List of IPM Runs in Support of CAMR

A list of the IPM runs that were used in the various analyses done in support of the final CAMR is provided. Model output from each of the IPM runs listed in this memo is available in the CAMR docket and also on EPA's Web site at www.epa.gov/airmarkets/epa-ipm.

Table 7-41. Listing of Runs from the Integrated Planning Model Used in Analyses Done in Support of the CAMR Final Rule Analyses

Run Name	Run Description
Base Case 2004	Base case model run, which includes the national Title IV SO_2 cap-and-trade program; NO_x SIP Call regional ozone season cap-and-trade program; and state-specific programs in Connecticut, Illinois, Maine, Massachusetts, Minnesota, Missouri, New Hampshire, New York, North Carolina, Oregon, Texas, and Wisconsin. This run represents conditions without the proposed CAIR.
CAIR 2004_Analysis	CAIR control strategy used for much of the analytical work for the final CAIR (includes AR/DE/NJ for annual controls and no ozone season cap and is the IPM run used for air quality modeling)
CAIR 2004_Final	Final CAIR policy (includes annual and ozone season caps for the States who contribute to PM2.5 and/or ozone nonattainment), used in Hg cost modeling
CAMR_Option 1	Final CAMR control strategy
CAMR_Option 2	CAMR option with Hg caps of 38 tons in 2010 and 15 tons in 2015
CAMR_Option 3	CAMR option with Hg caps of 38 tons in 2010, 24 tons in 2015, and 15 tons in 2018
CAMR_Sorbent Sensitivity_Option 1	CAMR run with second ACI control option in 2013 using advanced sorbents
Base Case 2004_EIA	Base Case run with EIA assumptions for the difference between natural gas prices and coal prices, as well as EIA's projection of electricity growth
CAIR 2004_EIA	CAIR run with EIA assumptions for the difference between natural gas prices and coal prices, as well as EIA's projection of electricity growth
CAMR 2004_EIA	CAMR run with EIA assumptions for the difference between natural gas prices and coal prices, as well as EIA's projection of electricity growth
Parsed Files	
EPA base case parsed for year 2010	
EPA base case parsed for year 2015	
EPA base case parsed for year 2020	
EPA CAIR parsed for year 2020	
EPA CAMR_Option 1 parsed for year 2020	
EPA CAMR_Option 2 parsed for year 2020	
EPA CAMR_Option 3 parsed for year 2020	

Tables

Figures

SECTION 8

AIR QUALITY MODELING: CHANGES IN HG DEPOSITION TO U.S. WATERBODIES

This section summarizes the emissions inventories and air quality modeling that serve as the inputs to the benefits analysis for the Clean Air Mercury Rule (CAMR). EPA used a sophisticated photochemical air quality model to predict the levels of mercury deposition for a 2001 base year and a 2020 baseline reflecting co-control of mercury from implementation of the Clean Air Interstate Rule (CAIR) as well as two control options for CAMR. The estimated changes in mercury deposition associated with the control options were then combined with fish tissue data for use in estimating health and welfare effects. In addition, utility attributable deposition of mercury was estimated based on zero-out modeling for both the 2001 and 2020 baselines.

The 1997 Mercury Study Report to Congress noted that "a single air quality model which was capable of model both the local as well as regional fate of mercury was not identified." In fact, at that time such a model did not exist. Thus, the modeling approach for this report 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). This approach also required assumptions to be made about the background concentrations of mercury that were uniformly added to the regional component and the use of "model plants" to represent typical sources for the local-scale transport. At this time, the Agency would have significant concerns about using the ISC3 model for assessments of Hg deposition associated with CAMR. The Agency will later this year promulgate the American Meteorological Society/Environmental Protection Agency Regulatory MODel (AERMOD) that will replace ISC3 as the recommended and preferred model for use in regulatory permit modeling assessments. This model contains the Argonne National Laboratory (ANL) versions of the wet and dry deposition algorithm which contain refinements beyond the ISC3 model and are considered more robust through extensive testing and evaluation. The ISC3 outputs for wet and dry deposition were never fully tested and verified for use in regulatory applications.

The Agency views the application of a more robust and sophisticated modeling approach as critical and required for assessing the Hg deposition associated with CAMR because of the density and properties of Hg and its complex transport and reactions in the atmosphere. The Community Multiscale Air Quality (CMAQ) modeling system best meets our requirements and the recommendations of the Report to Congress for a 'single air quality model" to address Hg deposition. 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 Hg emissions, CMAQ is the best available model for evaluating the impacts of the CAMR on U.S. mercury depositions. This model appropriately accounts for the atmospheric reactions of specific Hg emissions and their significance in 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 are performed based on plant-specific emissions of Hg by species as provided by the Integrated Planning Model (IPM).

Section 8.1 provides a summary of the emissions inventories that were modeled for this rule. Section 8.2 summarizes the model, domain, configuration, inputs, and application. Section 8.3 summarizes the model performance. Section 8.4 summarizes the results of estimating mercury depositions for the 2001 and 2020 scenarios modeled. Section 8.5 summarizes the findings at water bodies for the scenarios modeled and Section 8.6 provides the key references for this analysis.

8.1 Emissions Inventories and Estimated Emissions Reductions

The CAMR Emissions Inventory Technical Support Document (TSD) discusses the development of the 2001 and 2020 emissions inventories for input to the air quality modeling of this final rule in greater detail. Table 8-1 provides the emission sources and the basis for current and future-year inventories, while Table 8-2 summarizes the mercury emissions by species from utilities, also known as Electric Generating Units (EGUs), and other sources that were used in modeling of mercury deposition.

As Table 8-2 demonstrates, a total of almost 115 tons of mercury were emitted across all sources in 2001. EGUs emitted a total of 48.6 tons, or 42.3 percent of mercury emissions across all sources during this base year. Almost 21 tons of the most readily deposited form of mercury, i.e., reactive gaseous mercury (RGM), were emitted by these utilities and therefore comprised 42.4 percent of their mercury emissions.

The 2020 baseline emissions shown in Table 8-2 accounts for increases in economic activity and population growth between 2001 and 2020 that lead to increased production in the utility and manufacturing sectors and hence increases in emissions over time, as well as the implementation of regulatory policies from MACT standards (primarily on non-EGU sources) and the CAIR controls (as applied to EGUs in the eastern U.S.) which decreases emissions over this time period. Total mercury emissions in 2020 are roughly 87 tons, reflecting a net reduction of almost 28 tons (or 24 percent) from 2001 levels. As shown, the 2020 baseline with CAIR shows net reductions in mercury emissions for EGUs of 14.2 tons or a 29.1 percent reduction from 2001 levels. Utility emissions are expected to account for 39.5 percent of total mercury emissions in 2020, which is only slightly lower than their share in 2001. However, the reductions associated with CAIR co-control show a large reduction of 61.8 percent in their emissions of reactive gaseous mercury relative to their 2001 level of emissions, i.e., 20.58 tons in 2001 to only 7.87 tons in 2020.

Table 8-1. Summary of Emissions Sources for 2001 and 2020 Mercury Emissions Inventories

Sector	Emissions Source	2001 Base Year	2020 Base Case Projections
Utilities - Electric Generating Units (EGU)	Power industry electric generating units (EGUs)	1999 National Emission Inventory (NEI) data	Integrated Planning Model (IPM) reflecting growth in Btu demand as well as regulatory policies implemented through 2020, such as the Clean Air Interstate Rule
Non-EGU point sources	Non-Utility Point	1999 NEI, with medical waste incinerator sources replaced with draft 2002 NEI	(1) Department of Energy (DOE) fuel use projections, (2) Regional Economic Models, Inc. (REMI) Policy Insight® model, (3) decreases to REMI results based on trade associations, Bureau of Labor Statistics (BLS) projections and Bureau of Economic Analysis (BEA) historical growth from 1987 to 2002, (4) Maximum Achievable Control Technology category growth and control assumptions
Non-point sources	All other stationary sources inventoried at the county level	1999 NEI, with medical waste incinerator sources replaced with draft 2002 NEI	same as above

[a]This table documents only the sources of data for the U.S. inventory. The sources of data used for Canada and Mexico are explained in the technical support memorandum and were held constant from the base year to the future years.

Table 8-2. Summary of Mercury Emissions by Species: 2001 and 2020 (with CAIR) Baselines

Emissions Source	Mercury Emissions Species (tons)			Total Mercury Emissions (tons)
	Elemental	Reactive Gaseous	Particulate	
2001 Base Year				
EGUs	26.26	20.58	1.73	48.57
Non-EGU Point	37.85	13.33	7.60	58.78
Non-point	5.05	1.53	0.96	7.54
Total, All Sources	69.16	35.44	10.29	114.89
2020 (with CAIR) Baseline				
EGUs	25.72	7.87	0.83	34.42
Non-EGU Point	28.03	10.37	6.61	45.01
Non-point	5.69	1.30	0.77	7.76
Total, All Sources	59.44	19.54	8.21	87.19

Table 8-3 shows the reductions in mercury emissions associated with the CAMR Control Option 1 in 2020. The 2020 EGU emissions are reduced by approximately 10 tons to a total of 25 tons, representing a 11 percent reduction from total baseline emissions in 2020 (with CAIR), or a 27 percent reduction from the EGU sector alone. Under CAMR Control Option 2

(Table 8-4), EGU emissions are further reduced by an additional 4 tons to a total of roughly 21 tons. This represents a 16 percent reduction from total emissions from the 2020 baseline (with CAIR), or a 39 percent reduction from the EGU sector alone.

Table 8-3. Summary of Changes in Mercury Emissions Associated with CAMR Control Option 1: 2020

Emissions Source	Change in Mercury Emissions Species (tons)			Total Change in Mercury Emissions (tons)
	Elemental	Reactive Gaseous	Particulate	
EGUs	8.07 (31.4%)	1.30 (16.5%)	0.00 (0.0%)	9.37 (27.2%)
Non-EGU Point	n/a	n/a	n/a	n/a
Non-point	n/a	n/a	n/a	n/a
Total, All Sources	8.07 (13.6%)	1.30 (6.7%)	0.00 (0.0%)	9.37 (10.7%)

Note: n/a is not applicable.

Table 8-4. Summary of Changes in Mercury Emissions Associated with CAMR Control Option 2: 2020

Emissions Source	Change in Mercury Emissions Species (tons)			Total Change in Mercury Emissions (tons)
	Elemental	Reactive Gaseous	Particulate	
EGUs	11.39 (44.3%)	2.16 (27.4%)	0.04 (4.8%)	13.59 (39.5%)
Non-EGU Point	n/a	n/a	n/a	n/a
Non-point	n/a	n/a	n/a	n/a
Total, All Sources	11.39 (19.2%)	2.16 (11.1%)	0.04 (0.5%)	13.59 (15.6%)

Note: n/a is not applicable.

In comparison to current mercury emissions (i.e., the 2001 base year scenario), the CAIR and CAMR Option 1 achieve a total reduction in EGU emissions of approximately 24 tons (48 percent), while CAIR and CAMR Option 2 achieve a total reduction in EGU emissions of approximately 28 tons (57 percent).

8.2 Model, Domain, Configuration, Inputs, Application

This section summarizes the methods for and results of estimating mercury depositions for 2001 and 2020 base cases and control scenarios for the purposes of the benefits analysis. The mercury deposition changes were estimated using national-scale applications of the Community Multi-Scale Air Quality (CMAQ) model. In Section 8.2.1, we describe the estimation of mercury depositions using CMAQ.

8.2.1 Air Quality Model

We use the emissions inputs summarized above with a national-scale application of the Community Multi-scale Air Quality (CMAQ) modeling system to estimate mercury depositions in the contiguous United States. 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 emissions, CMAQ is useful for evaluating the impacts of the CAMR on U.S. mercury depositions. Our analysis applies the modeling system to the entire United States for six emissions scenarios: a 2001 base year, a 2001 base year with utility mercury emissions zeroed-out, a 2020 projection with CAIR incorporated, a 2020 projection with CAIR incorporated and utility mercury emissions zeroed-out, a 2020 projection with CAIR and control option 1 incorporated, a 2020 projection with CAIR and control option 2 incorporated.

The CMAQ version 4.3 was employed for this CAMR modeling analysis (Byun and Schere, 2004, Bullock and Brehme 2002). This version reflects updates in a number of areas to improve performance and address comments from its peer review. The updates in mercury chemistry used for CAMR from that described in (Bullock and Brehme 2002) are as follows: (1) the elemental mercury (Hg0) reaction with H_2O_2 assumes the formation of 100 percent reactive gaseous mercury (RGM) rather than 100 percent particulate mercury (HgP), (2) the Hg0 reaction with ozone assumes the formation of 50 percent RGM and 50 percent HgP rather than 100 percent HgP, (3) the Hg0 reaction with OH assumes the formation of 50 percent RGM and 50 percent HgP rather than 100 percent HgP, and (4) the rate constant for the Hg0 + OH reaction was lowered from 8.7 to 7.7 $\times 10^{-14} cm^3 molecules^{-1} s^{-1}$. CMAQ simulates every hour of every day of the year and, thus, requires a variety of input files that contain information pertaining to the modeling domain and simulation period. These include hourly emissions estimates and meteorological data in every grid cell, as well as a set of pollutant concentrations to initialize the model and to specify concentrations along the modeling domain boundaries. These initial and boundary concentrations were obtained from output of a global chemistry model. We use the model predictions in a relative sense by first determining the ratio of mercury deposition predictions. The calculated relative change is then combined with the corresponding fish tissue concentration data to project fish tissue concentrations for the future case scenarios. The following sections provide a more detailed discussion of the modeling and a summary of the results.

8.2.2 Modeling Domain

As shown in Figure 8-1, the modeling domain encompasses the lower 48 states and extends from 126 degrees west longitude to 66 degrees west longitude and from 24 degrees north latitude to 52 degrees north latitude. The modeling domain is segmented into rectangular blocks referred to as grid cells. The model actually predicts pollutant concentrations for each of these grid cells. For this application the horizontal domain consisted of 16,576 grid cells that are roughly 36 km by 36 km. In addition, the modeling domain contains 14 vertical layers with the top of the modeling domain at about 16,200 meters, or 100 millibar. The height of the surface layer is 38 meters.

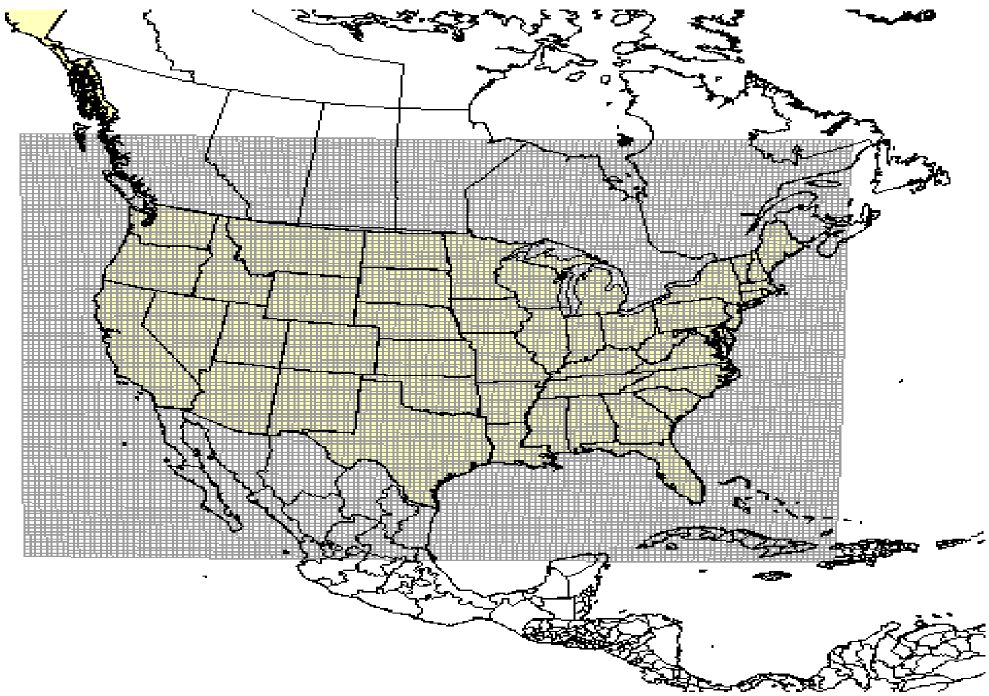

Figure 8-1. CMAQ Modeling Domain

8.2.3 Time Periods Modeled for Mercury Deposition

CMAQ was run for a full year for each of the six CAMR emissions scenarios modeled. The overall model run time for completing an annual simulation was reduced by dividing the year into two six-month periods which were run in parallel on different computer processors. That is, the annual simulation was performed as two separate six month model runs. One run was for January through June and the other run was for July through December. Each six-month run included a 10-day ramp-up (i.e., "spin-up") period designed to minimize the influence of the initial concentration fields (i.e., initial conditions) used at the start of the model run. The development of initial condition concentrations is described in Section 8.2.4 below. The ramp-up periods used for the CAMR CMAQ applications are as follows:

- First six-month ramp-up period is December 22 - 31, 2000
- Second six-month ramp-up period is June 21 - 30, 2001

Model predictions from these ramp-up periods were discarded and not used in analyses of the modeling results. The meteorological conditions, initial conditions and boundary conditions were held constant for each of the emissions scenarios modeled and are described below in section 8.2.4.

8.2.4 Model Inputs

CMAQ requires a variety of input files that contain information pertaining to the modeling domain and simulation period. These include gridded, hourly emissions estimates and meteorological data as well as initial and boundary conditions. Separate emissions inventories were prepared for the 2001 base year and each of the future-year base cases and control scenarios. All other inputs were specified for the 2001 base year model application and remained unchanged for each future-year modeling scenario.

CMAQ requires detailed emissions inventories containing temporally allocated emissions for each grid cell in the modeling domain for each species being simulated. The previously described annual emission inventories were processed into model-ready inputs through the emissions processing system. Details of the processing of emissions are provided in the Clean Air Mercury Rule Emissions Inventory Technical Support Document (EPA, 2005).

Meteorological data, such as temperature, wind, stability parameters, and atmospheric moisture contents influence the formation, transport, and removal of air pollution. The CMAQ model requires a specific suite of meteorological input files in order to simulate these physical and chemical processes. For the CAMR CMAQ modeling, meteorological input files were derived from a simulation of the Pennsylvania State University / National Center for Atmospheric Research Mesoscale Model (Grell *et al.*, 1994) for the entire year of 2001. This model, commonly referred to as MM5, is a limited-area, nonhydrostatic, terrain-following system that solves for the full set of physical and thermodynamic equations which govern atmospheric motions. For this analysis, version 3.6.1 of MM5 was used.

National modeling, such as the CAMR annual mercury modeling, requires the prescription of boundary conditions (BC's) to account for the influx of pollutants and precursors from the upwind source areas outside the modeling domain. A scientifically sound approach to estimate incoming pollutant concentration associated with intercontinental transport is to use a global chemistry model to provide the dynamic BC's for the national model simulation. For the CAMR mercury modeling, we used the predictions from a three-dimensional global atmospheric chemistry and transport model, the GEOS-CHEM model (Yamatosca B., 2004) developed at Harvard University to provide the lateral boundary and initial species concentrations. The lateral boundary species concentrations varied with height and time (every 3 hours). Terrain elevations and land use information were obtained from the U.S. Geological Survey database at 10 km resolution and aggregated to the roughly 36 km horizontal resolution used for this CMAQ application.

8.3 CMAQ Model Performance Evaluation

At this point in time, it is difficult to assess model performance for total mercury deposition. Scientist currently believe through analysis of very limited measurements that wet and dry deposition are approximately equal in magnitude. There currently is no measurement network to evaluate the performance of models in estimating dry deposition of mercury. Thus, we are not able to evaluate the performance of air quality models in predicting dry deposition, which is thought to be roughly half of total mercury deposition. There is a network of mercury wet deposition monitors, which are scattered throughout remote locations in the United States and Canada, mostly in the east. Thus, model predictions of wet deposition can be evaluated by a monitoring network.

An operational model performance evaluation for mercury wet deposition for 2001 was performed to estimate the ability of the CMAQ modeling system to replicate base-year wet depositions of mercury. The wet deposition evaluation principally comprises statistical assessments of model versus observed pairs that were matched in time and space on a seasonal and annual basis. The statistics are presented separately for the entire domain, the East, and the West (using the 100th meridian to divide the eastern and western United States). These statistics on model performance along with an annual observed versus predicted performance scatter plot can be found in the Clean Air Mercury Rule Emissions Inventory and Air Quality Modeling Technical Support Document.

For mercury wet deposition, this evaluation includes comparisons of model predictions to the corresponding measurements from the Mercury Deposition Network (MDN). The principal evaluation statistics used to evaluate CMAQ performance are the fractional bias and fractional error. Fractional bias is defined as:

$$FBIAS = \frac{2}{N} \sum_{i=1}^{N} \frac{(Pred_{x,t}^{i} - Obs_{x,t}^{i})}{(Pred_{x,t}^{i} + Obs_{x,t}^{i})} * 100$$

where: N = the number of measurement sites
Pred = model predicted deposition at site x over time t (i.e. Annual)
Obs = observed deposition at site x over time t

Fractional bias is a useful model performance indicator because it has the advantage of equally weighting positive and negative bias estimates. Fractional error is similar to fractional bias except the absolute value of the difference is used so that the error is always positive. Fractional error is defined as:

$$FERROR = \frac{2}{N} \sum_{i=1}^{N} \frac{\left|Pred_{x,t}^{i} - Obs_{x,t}^{i}\right|}{Pred_{x,t}^{i} + Obs_{x,t}^{i}} * 100$$

The fractional bias and fractional error statistics were calculated using the predicted-observed pairs for the full year of 2001 and for each season, separately. These metrics were calculated annually and seasonally for all available MDN sites in 2001. Only sites where data was available more than half the weeks in a season were utilized for the seasonal performance evaluation and only sites that had four seasons meeting this data completeness requirement were utilized for the annual performance evaluation. There were 52 MDN sites in 2001 that meet the annual data completeness requirements, of those sites 48 were located in the east and 4 were located in the west. Fractional bias for cases where the model underpredicts by a factor of 2 would be -67 and for cases where the model overpredicts by a factor of 2 would be + 67 percent. The results in Table 8-5 shows that averaged annually over all MDN monitoring sites, CMAQ underestimates mercury wet deposition with a fractional bias of approximately -23 percent. This underprediction bias is well within a factor of 2. The 4 MDN sites in the west do not provide an adequate or representative basis for inferring model performance.

Table 8-5. CMAQ Performance Statistics for Mercury Wet Deposition: 2001

Area	#MDN Sites	Fractional Bias (%)	Fractional Error (%)
Entire Domain	52	-23.2	30.2
East	48	-27.0	30.2
West	4	21.7	30.5

8.4 Mercury Deposition Results

Maps showing the mercury deposition results are provided below. The annual total modeled mercury deposition for the 2001 base case is shown in Figure 8-2. The reduction in total mercury deposition that would result if all US power plant mercury emissions were zeroed-out in 2001 is shown in Figure 8-3. The change in 2001 total mercury deposition in 2020 with CAIR is shown in figure 8-4. The total mercury deposition for 2020 with CAIR is shown in Figure 8-5. The decrease in 2020 with CAIR when all US power plant emissions are zeroed-out is shown in Figure 8-6. The change in 2020 CAIR total mercury depositions with CAMR Option 1 is shown in Figure 8-7. The change in 2020 CAIR total mercury depositions with CAMR Option 2 is shown in Figure 8-8. It can be seen in Figures 8.3 and 8.4 that the implementation of CAIR and other minor non-utility mercury emissions decreases in 2020 result in a similar reduction in total mercury deposition as completely eliminating power plant mercury emissions. The main cause of this result is that CAIR results in a very large decrease in reactive gaseous mercury (RGM) emissions from Power Plants through the implementation of scrubber control technology (see Table 8-2). RGM is the most readily deposited form of mercury. It can be seen in Figures 8-7 and 8-8 that the implementation of CAMR Option 1 and CAMR Option 2 results in some scattered total mercury deposition reductions beyond CAIR in 2020, but for the most part these reductions are not very significant compared to those obtained by CAIR. Most of the mercury emissions reductions from CAMR are in the form of elemental mercury (Hg0). This form of mercury is not readily deposited, but enters the global pool of mercury. Thus, CAMR will result in a reduction of the transport of mercury to other places in the world.

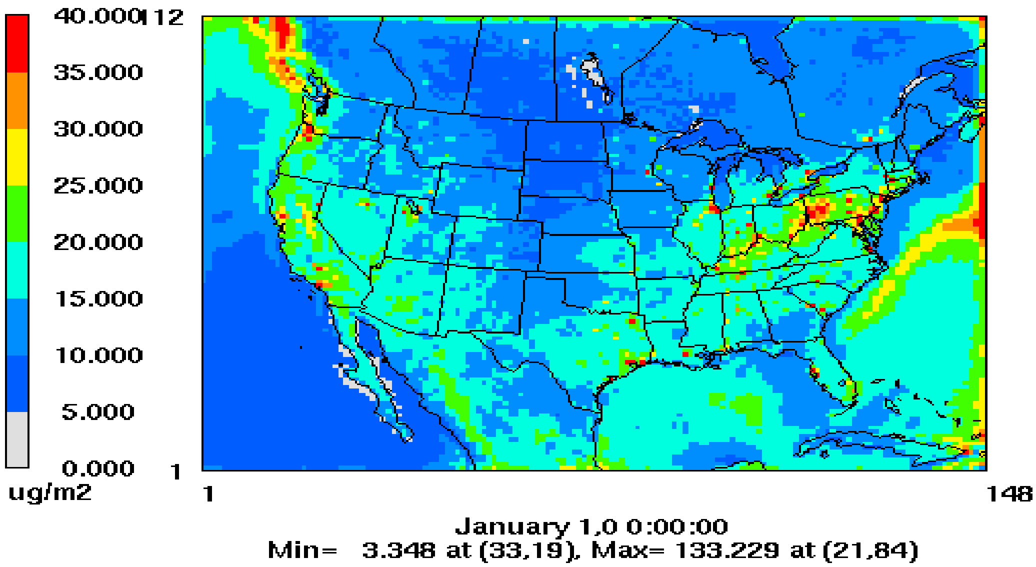

Figure 8-2. Base Case Total Mercury Deposition: 2001

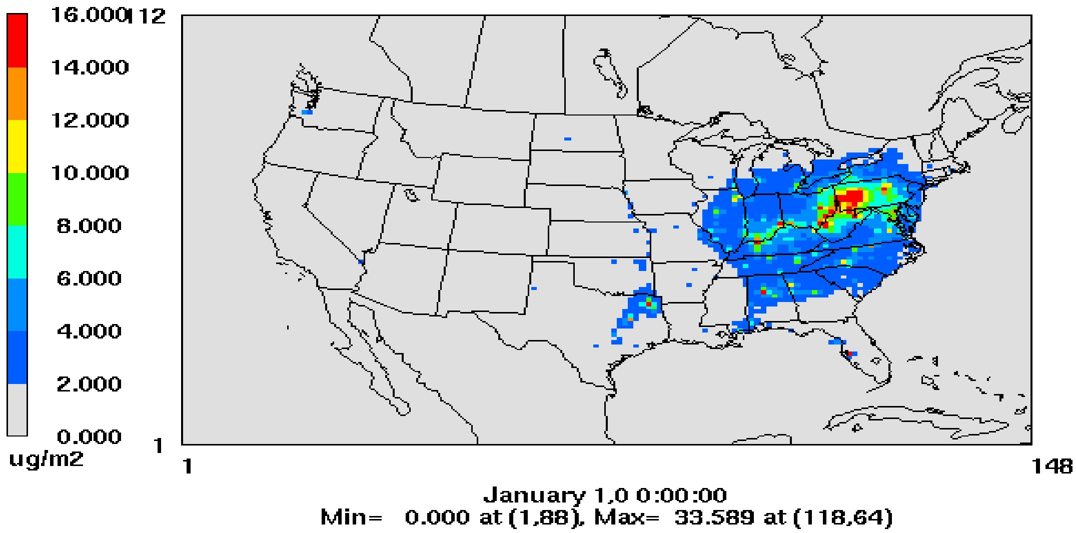

Figure 8-3. Decrease in Total Mercury Deposition with Power Plant Zero-Out Simulation: 2001

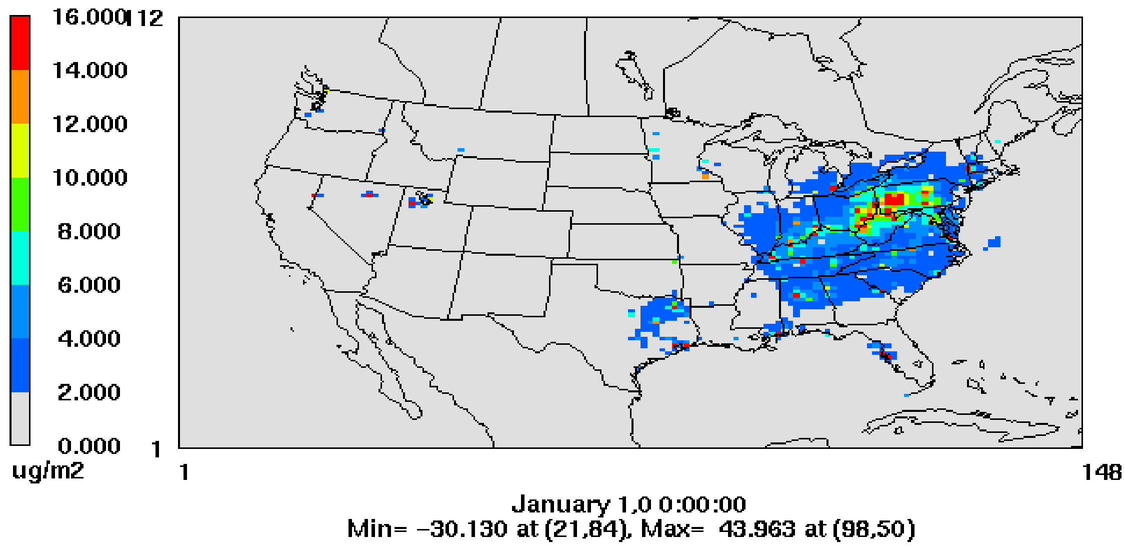

Figure 8-4. Change in Total Mercury Deposition for All Sources: 2020 (with CAIR) Relative to 2001

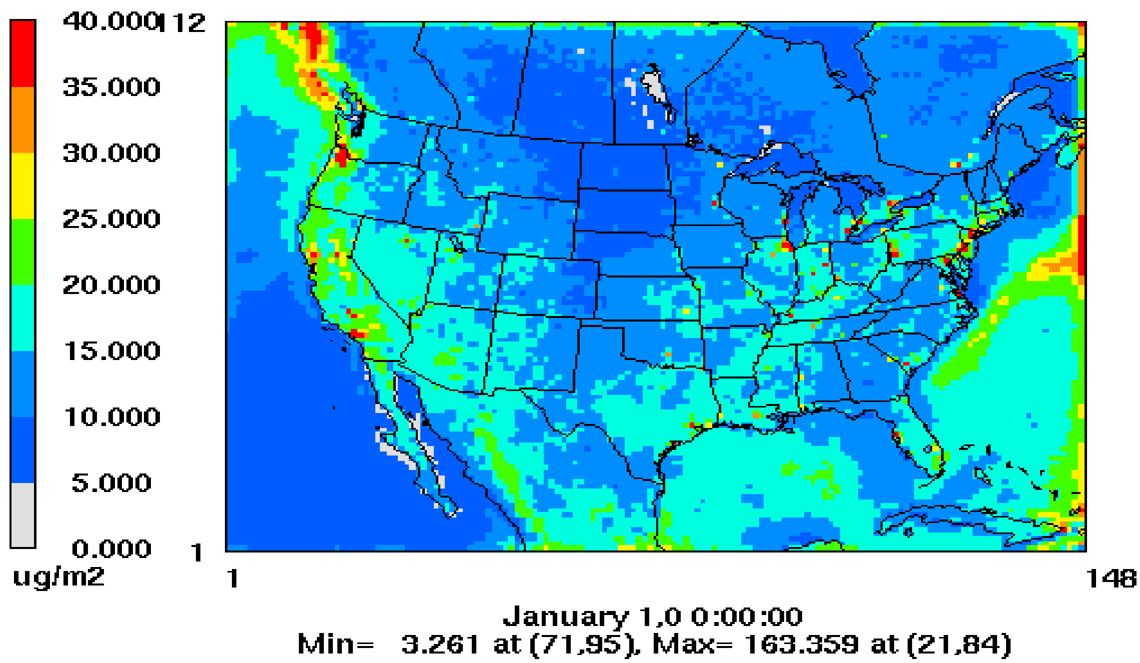

Figure 8-5. Total Mercury Deposition: 2020 (with CAIR)

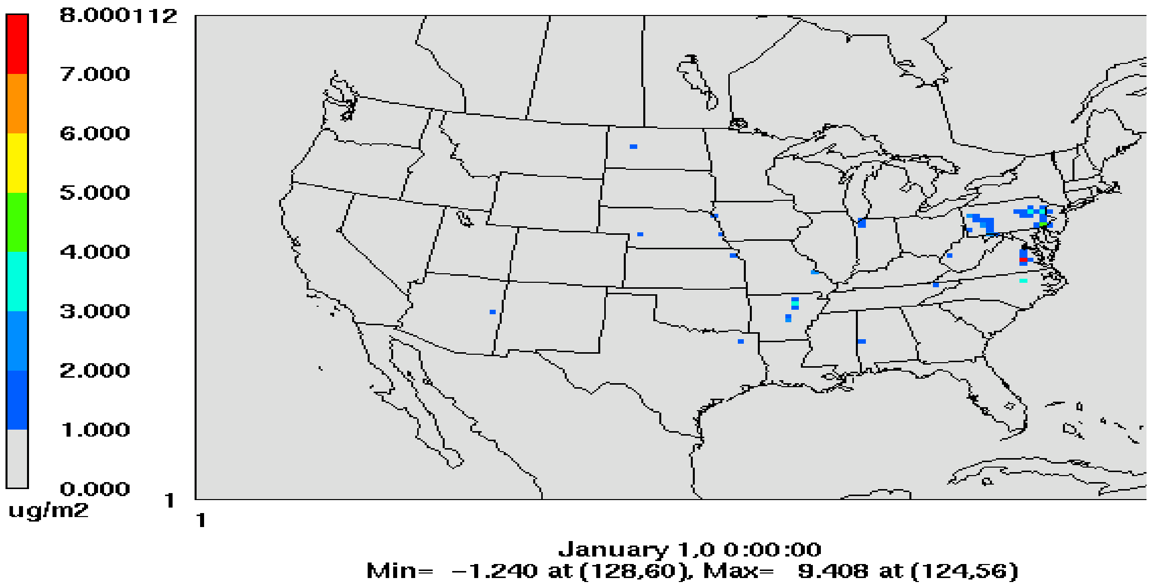

Figure 8-6. Change in Mercury Depositions from Power Plants Due to CAMR Option 1: 2020

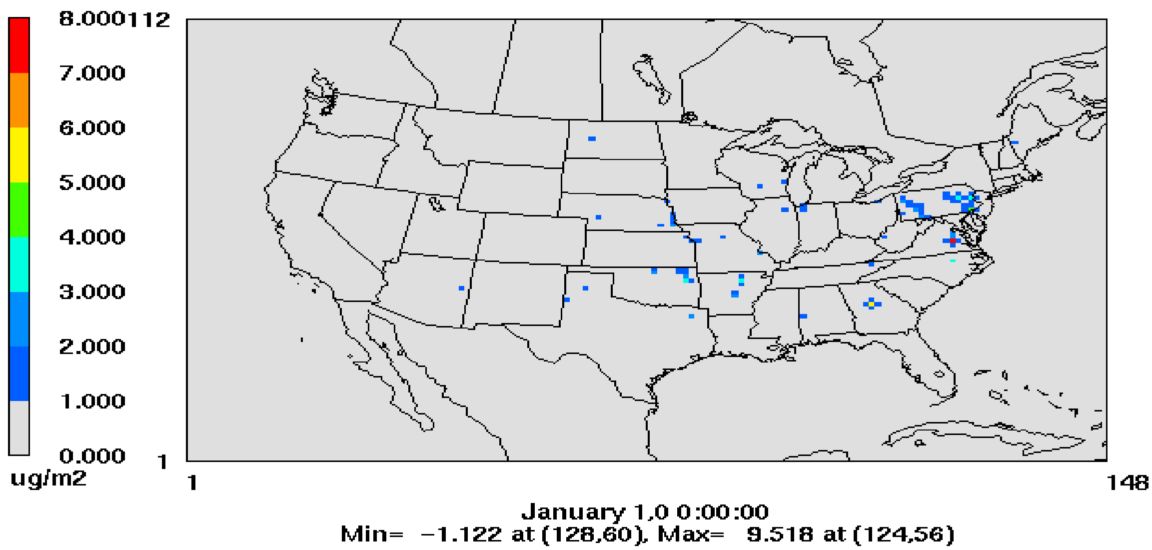

Figure 8-7. Change in Mercury Deposition from Power Plants Due to CAMR Option 2: 2020

8.5 Summary of Findings: HUC Level Deposition Analysis

The cumulative distribution of Hydrologic Unit Code (HUC) level depositions across watersheds are provided in Table 8-6 and Figure 8-8. The cumulative percentage of HUCs that have deposition less than the value on the x-axis for each of the six modeled scenarios are shown in Figure 8-8. For example, 90 percent of the HUCs have depositions below 22.16 ug/m^2 in the 2001 base case. For the 2020 CAIR plus CAMR Option 1 scenario, 90 percent of the HUCs have depositions below 19.48 ug/m^2.

Table 8-6. Summary Statistics of Total Mercury Depositions (ug/m^2) by Modeling Scenario

Statistics	2001 Base Case	2001 Utility Hg Zero-Out	2020 CAIR	2020 Utility Hg Zero-Out	2020 CAIR & CAMR Option 1	2020 CAIR & CAMR Option 2
Minimum	6.994	6.942	6.078	5.898	6.075	6.075
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

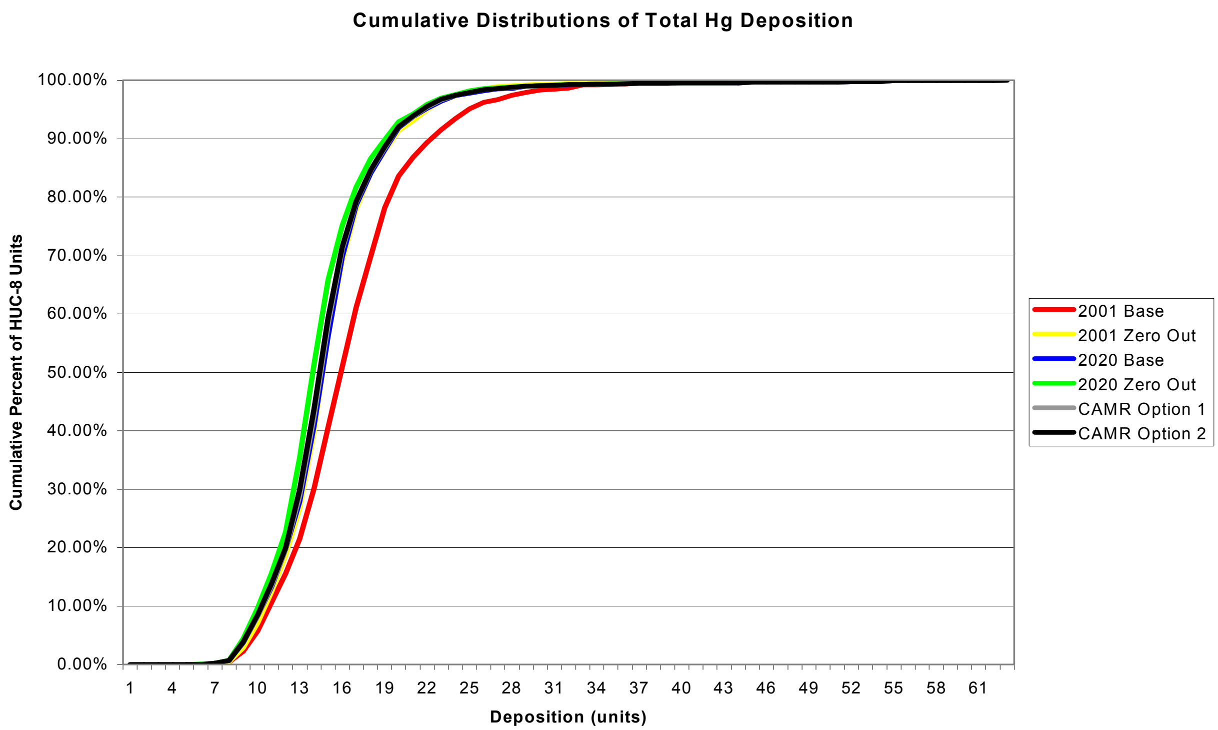

Figure 8-8. Cumulative Distribution of Total Mercury Deposition (ug/m^2) Fat HUC-8 Level by Modeling Scenario

The cumulative distribution of Hydrologic Unit Code (HUC) level depositions attributable to utilities are provided in Table 8-7 and Figure 8-9. The cumulative percentage of HUCs that have deposition less than the value on the x-axis for 4 of the modeled scenarios are shown in Figure 8-9. For example, 90 percent of the HUCs have depositions attributable to utilities below 4.08 ug/m^2 in the 2001 base case. For the 2020 CAIR plus CAMR Option 1 scenario, 90 percent of the HUCs have depositions attributable to utilities below 1.16 ug/m^2**.** CAIR shifts the distribution of utility 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 Option 1 and Option 2 results in an additional reduction in 2020 utility attributable deposition in the 99th percentile of 15 and 20 percent , respectively. At the 50th percentile, CAMR Option 1 and Option 2 result in an additional reduction of 2020 utility attributable deposition of 16 and 29 percent, respectively. As can be seen in Figure 8-10, CAIR also shifts the distribution of percentage of HUCs with deposition attributable to utilities. In the 2001 base case, 10 percent of HUCs had greater than 20 percent of deposition attributable to utilities. In the 2020 with CAIR scenario, 10 percent of HUCs had greater than 10 percent of deposition attributable to utilities. In the 2020 CAIR plus CAMR Option 1 scenario, 10 percent of HUCs had greater than 7 percent of deposition attributable to utilities.

Table 8-7. Summary Statistics of Utility Attributable Deposition (ug/m^2) by Modeling Scenario

Statistics	2001 Base Case	2020 CAIR	2020 CAIR & CAMR Option 1	2020 CAIR & CAMR Option 2
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

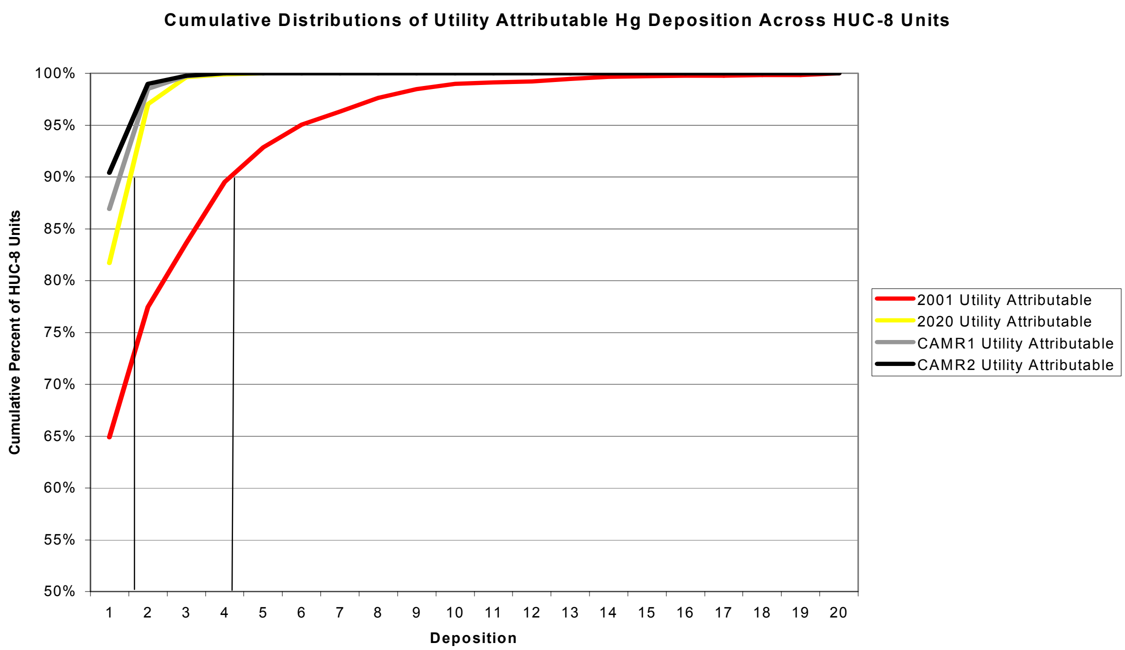

Figure 8-9. Cumulative Distribution of Utility Attributable Mercury Deposition at HUC-8 Level by Model Scenario

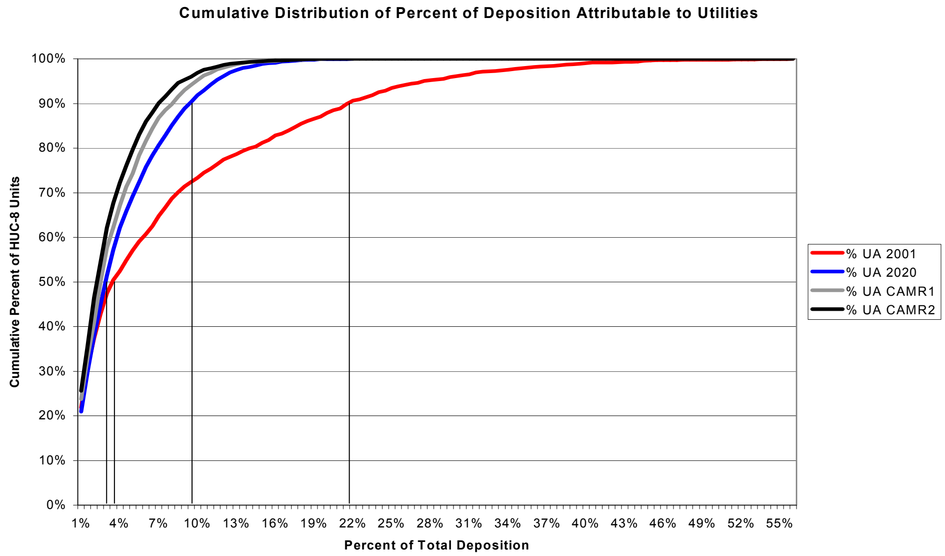

Figure 8-10. Cumulative Distribution of Percent Deposition (ug/m^2) Attributable to Utilities at HUC-8 Level by Modeling Scenario

8.6 References

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Tables

Figures

SECTION 9

ANALYSIS OF THE DOSE-RESPONSE RELATIONSHIP BETWEEN MATERNAL MERCURY BODY BURDEN AND CHILDHOOD IQ

9.1 Introduction

In considering possible health endpoints for quantification and monetization in this analysis, EPA reviewed the scientific literature on the health effects of mercury, including the *Toxicological Effects of Methylmercury*, published by the National Research Council (NRC) in 2000 (NRC 2000).

Epidemiological studies of prenatal mercury exposure conducted in the Faroe Islands (Grandjean et al. 1997), New Zealand (Kjellstrom et al. 1989, Crump et al. 1998), and the Seychelles Islands (Davidson et al. 1998, Myers et al. 2003) have examined neurodevelopmental outcomes through the administration of tests of cognitive functioning. Each of these studies included some but not all of the following tests: full-scale IQ, performance IQ, problem solving, social and adaptive behavior, language functions, motor skills, attention, memory and other functions. The NRC reviewed the studies and determined that "Each of the studies was well designed and carefully conducted, and each examined prenatal MeHg [methylmercury] exposures within the range of the general U.S. population exposures" (NRC 2000).

EPA held a neurotoxicology workshop with several of the NRC panel members in November 2002. Participants were asked about which studies should be considered in generating dose-response functions for developmental neurotoxicity. Participants were also asked about endpoints to consider for monetization and they suggested looking at neurological tests that might lead to changes in IQ or other neurodevelopmental impacts.

EPA has chosen to focus on quantification of intelligence quotient (IQ) decrements associated with prenatal mercury exposure as the initial endpoint for quantification and valuation of mercury health benefits. Reasons for this initial focus on IQ include the availability of thoroughly-reviewed, high-quality epidemiological studies assessing IQ or related cognitive outcomes suitable for IQ estimation, and the availability of well-established methods and data for economic valuation of avoided IQ deficits, as applied in EPA's previous benefits analyses for childhood lead exposure. Bellinger (2005) provides a more detailed discussion of the use of IQ as the focus of benefits analysis.

The focus of this analysis was to identify the appropriate dose-response coefficients from the Faroe Islands, New Zealand, and Seychelles studies, and to devise a statistical approach for combining those coefficients to provide an integrated estimate of the IQ dose-response coefficient.

EPA is using a linear model that goes through the origin to fit population-level dose-response relationships to the pooled data from the three studies. The application of a linear model should not be interpreted to suggest that any of the three studies used have data showing health effects from methylmercury exposure at or below the 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 2002). EPA believes that exposures at or below the RfD are 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 near 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. The regulation is focused on the reduction of exposures and the associated health benefits that would accrue to people currently exposed at levels above the RfD due solely to power plants.

Use of a linear model that goes through the origin, rather than one that reflects a threshold effect is technically more simple and practical. It associates an increment of IQ benefit with a given reduction in exposure. A linear model allows us to estimate the benefits of reductions in exposure due to power plants without a complete assessment of other sources of exposure. Other models would require information on the joint distribution of exposure from power plants and other sources to estimate the benefits of reducing the exposure due to power plants, which would require much more precise information about consumption patterns.

9.2 Epidemiological Studies of Mercury and Neurodevelopmental Effects

The IQ dose-response analysis uses data from three major prospective studies investigating potential neurotoxicity of low-level, chronic mercury exposure: the New Zealand study, the Seychelles Child Development Study, and the Faroe Islands study.

In assembling the New Zealand sample, Kjellstrom et al. (1989) ascertained the fish consumption of 10,930 of 16,293 pregnant women in the study area. They identified 935 women who reportedly consumed fish at least 3 times per week. Hair samples were obtained from these women, and 73 were found to have a hair mercury level of 6 parts per million (ppm) or greater. In this group, the mean was 8.3 ppm, with a range of 6 to 86 ppm, although only one woman had a level greater than 20 ppm. Each woman with 6 ppm hair mercury or greater was matched to 3 controls - one with hair mercury between 3-6 ppm, one with hair mercury less than 3 ppm and high fish consumption, and one with hair mercury less than 3 ppm and low fish consumption. Ethnic group, age, smoking, residence time in New Zealand, and child sex were also used to select controls. The final study group included 237 children, including 57 fully-matched sets of 4 children. Although children were assessed at 4 and 6 years of age, only the data collected at the older age is considered in this analysis, as the reliability and validity of neurodevelopmental testing generally increases with child age. Table 9-1 lists the tests administered at the 6 year evaluation, and indicates the general functional domain each is considered to assess.

Table 9-1. Neurobehavioral Tests Administered at the 6-Year Evaluations in the New Zealand Study

Test	Primary Domain Assessed
Wechsler Intelligence Scale for Children-Revised	General intelligence
McCarthy Scales of Children's Abilities	General development
Test of Language Development	General verbal skills
Peabody Picture Vocabulary Test	Receptive language
Clay Reading Diagnostic Survey	Reading
Burt Word Recognition Test	Single word reading
Key Math Diagnostic Arithmetic Test	General math skills
Everts Behavioral Rating Scale	Behavior disorders

The Faroe Islands investigators assembled a birth cohort of 1,353 newborns recruited from 3 hospitals over a 21 month period in 1986-1987. In 1,022 women, two biomarkers of prenatal mercury exposure were collected: cord-blood mercury, and maternal hair mercury at delivery. Neurodevelopmental assessments of 917 children were conducted at age 7 (Grandjean et al. 1997). For these 917 children, the geometric mean concentration of mercury in cord-blood was 22.6 parts per billion (ppb) (interquartile range 13.1 – 40.5 ppb, full range 0.9 – 351 ppb). The geometric mean concentration of mercury in maternal hair was 4.2 ppm (interquartile range: 2.5-7.7 ppm, full range 0.2 – 39.1 ppm) (Budtz-Jorgensen et al. 2004a). Neurodevelopmental assessments of the children were conducted at age 7 years (Grandjean et al. 1997). Table 9-2 lists each test administered and the corresponding general functional domain.

Table 9-2. Neurobehavioral Tests Administered at the 7-Year Evaluations in the Faroe Islands Study

Test	Primary Domain Assessed
Wechsler Intelligence Scale for Children-Revised (selected subtests)	
Digit span	Short-term memory
Similarities	Abstract verbal reasoning
Block Design	Constructional praxis
Bender-Gestalt Test	Visual-motor integration
California Verbal Learning Test-Children	Verbal learning and memory
Boston Naming Test	Confrontational naming
Tactual Performance Test	Nonverbal memory
Neurobehavioral Evaluation System (NES) (selected tests)	
Finger tapping	Motor speed
Hand-eye coordination	Hand-eye coordination
Continuous performance test	Vigilance
Profile of Mood States	Mood
Child Behavior Checklist (selected items)	Behavior disorders

In assembling the Seychelles Child Development Study sample, investigators obtained hair samples from 779 pregnant women and ultimately enrolled a study sample consisting of 740 newborns. The mean maternal hair mercury level was 6.8 ppm (range 0.9-25.8 ppm) (Davidson et al. 1998). Neurodevelopmental assessments were conducted when the children were 6.5, 19, 29, and 66 months, and at 9 years. The mean maternal hair mercury level for the 643 children who participated in the assessment at age 9 years was 6.9 ppm (standard deviation 4.5 ppm) (Myers et al. 2003). Table 9-3 lists the tests administered at this age and the corresponding general functional domain.

Table 9-3. Neurobehavioral Tests Administered at the 9-Year Evaluations in the Seychelles Islands Study

Test	Primary Domain Assessed
Wechsler Intelligence Scale for Children-Third Edition	General intelligence
California Verbal Learning Test-Children	Verbal learning and memory
Boston Naming Test	Confrontational naming
Finger tapping	Motor speed
Continuous performance test	Vigilance
Developmental Test of Visual-Motor Integration	Visual-motor integration
Bruininks-Oseretsky Test of Motor Proficiency (selected subtests)	Gross and fine motor skills
Grooved Pegboard	Manual dexterity
Trail-Making Test	Visual tracking and executive function
Woodcock-Johnson Tests of Achievement (selected subtests) Letter-Word Identification Applied Math	 Single word reading Quantitative problem-solving
Wide Range Assessment of Memory and Learning Design Memory subtest	 Visual memory
Haptic Discrimination Test	Cross-modal integration
Child Behavior Checklist	Behavioral disorders
Connors' Hyperactivity Index	ADHD screener

9.3 Statistical Analysis

A statistical analysis was conducted to integrate data from the three studies to produce a single estimate of the IQ dose-response relationship. Details of the analysis, including statistical model formulation, selection of input values, results and sensitivity analysis are reported in Ryan (2005) and are summarized below.

Data available for this analysis consisted of dose-response coefficients estimated by the investigators for each of the three studies. These coefficients express a central estimate of the average reduction in children's scores in tests of IQ (or other tests of cognitive performance) for a one unit change in the mercury body burden of the mother during pregnancy.

A Bayesian hierarchical statistical model was used to estimate the integrated dose-response coefficient. This is similar to the approach used by the NRC panel to calculate a benchmark dose value integrating data from all three studies (NRC 2000). A more technical description of these same methods has been provided by Coull et al. (2004). The model makes use of dose-response coefficients for IQ, and also incorporates coefficients for other cognitive tests conducted in the studies, in an effort to obtain more robust estimates of the IQ relationship that account for within-study (endpoint-to-endpoint) variability as well as variability across studies. As compared with a model that uses only the IQ dose-response coefficients and their variances, this approach makes use of more data from the studies to better characterize the variability in estimation of the integrated IQ coefficient.

The key parameter inputs to the statistical model are the estimated IQ dose-response coefficients for each study. The Wechsler Intelligence Scales for Children (WISC) is a standard test of childhood IQ that was used in each of the three studies. The version of the test administered in the Seychelles Islands (WISC-III) was different from that used in New Zealand and the Faroe Islands (both WISC-R). As part of the standardization of the WISC-III, however, both versions were administered to approximately 200 children. The correlation between the Full-Scale IQ scores for the two versions of the WISC was 0.89; thus the WISC-R and WISC-III appear to measure the same constructs and generate scores with similar dispersion (Wechsler 1991).

For the New Zealand study, full-scale IQ dose-response coefficients were reported in Crump et al. (1998). In Table III of this paper, two coefficients for full scale IQ are reported: one with the complete cohort, and the other for which one very influential observation (with unusually high maternal hair mercury) was excluded. The NRC Committee on the Toxicological Effects of Methylmercury reviewed the influence of the one outlier on the model outcome in comparison to a model without this outlier, and determined that exclusion of the outlier was reasonable and appropriate (NRC, 2000). In keeping with the conclusions by the NRC committee, this analysis uses the coefficient from the regression in which the outlier child was excluded: an IQ change of -0.53 IQ points (95% confidence interval -1.1, 0.069) for each ppm of mercury in maternal hair.

For the Seychelles study, a 2003 paper reports results for IQ tests administered at age 9 (Myers et al. 2003). This analysis uses the coefficient from Table 9-2 of this study: an IQ change of -0.13 IQ points (95% confidence interval -0.33, 0.07) for each ppm of mercury in maternal hair.

The WISC-R includes 10 core subtests and 3 supplementary subtests. For the Faroes study, the investigators did not administer the complete version of the WISC-R because of their conclusion that a methylmercury-associated deficit in a broad measure such as Full-Scale IQ provides relatively little insight into the specific nature of methylmercury's neuropsychological effects on children. The Faroes investigators did administer three of the WISC-R subtests to the children in their study (Similarities, Block Design, and Digit Span). Thus, to include data from the Faroe Islands in this integrated assessment of prenatal mercury exposure on childhood IQ, it was necessary to estimate a Faroe Islands dose-response coefficient for full-scale IQ from the three available subtests.

Information on correlations between WISC-R subtest scores and WISC-R full-scale IQ scores is available to assess the validity of a full-scale IQ estimated from the subtests.

Similarities and Block Design are core subtests of the WISC-R, and Digit Span is a supplementary subtest. The WISC-R was standardized on a nationally-representative sample of U.S. children ages 6 to 16 years. Based on subtest scores, Sattler (1988) identified the pair, triad, quartet, etc. of subtests that provides the most valid estimate of full-scale IQ. Only the 10 core subtests were considered in this exercise. Of the 45 possible combinations of 2 core subtests (i.e., 10 subtests taken 2 at a time), the combination of Similarities and Block Design, the two core subtests administered in the Faroe Islands study, ranked 3rd in the magnitude of the validity coefficient (0.885). The top-ranked combination was Vocabulary and Block Design (0.906). The combination ranked 2nd was Information and Block Design (0.888). It is reasonable to expect that taking into account Digit Span scores, the supplementary subtest administered, will increase the validity coefficient. The results of this exercise indicate that combining the scores of the Faroese children on Similarities, Block Design and Digit Span will provide valid estimates of their full-scale IQ scores.

In support of this integrated IQ dose-response analysis, the Faroes research team conducted further analysis to estimate a full-scale IQ dose-response coefficient based on the data for the three subtests. This new Faroes analysis makes use of a structural equation model similar to that described in Budtz-Jorgensen et al. (2002), and is reported in Budtz-Jorgensen et al. (2005). As with other reports on results of the Faroe Islands study, the full-scale IQ coefficient for the Faroes data is reported using cord blood mercury as the marker for exposure; results from New Zealand and the Seychelles are presented in terms of maternal hair mercury. The Faroes coefficient was converted to terms of hair mercury, using the reported median maternal hair:cord blood mercury ratio for the Faroes cohort of approximately 200 (Budtz-Jorgensen et al. 2004a). After conversion of the Faroes estimate from terms of cord blood mercury to hair mercury, the estimated Faroes coefficient is an IQ change of -0.12 IQ points (95% confidence interval -0.24, -0.01) for each ppm of mercury in maternal hair (Ryan 2005).

The IQ dose-response estimates for each of the three studies, along with 95% confidence intervals, are shown in Table 9-4 and Figure 9-1.

Table 9-4. Relationship Between Maternal Mercury Body Burden and IQ in Three Studies: IQ Decrement per ppm of Maternal Hair Mercury

Study	Regression Coefficient (95% Confidence Interval)	Notes
New Zealand	-0.53 (-1.1, 0.069)	Reported in Table III of Crump (1998); outlier child omitted.
Seychelles	-0.13 (-0.33, 0.07)	Reported in Table 2 of Myers (2003).
Faroe Islands	-0.12 (-0.24, -0.01)	Reported in Ryan (2005), based on structural equation modeling of three IQ subtests by Budtz-Jorgensen et al. (2005).
Ryan (2005) Integrative Analysis - Main Case	-0.13 (-0.28, -0.03)	see text and Ryan (2005)

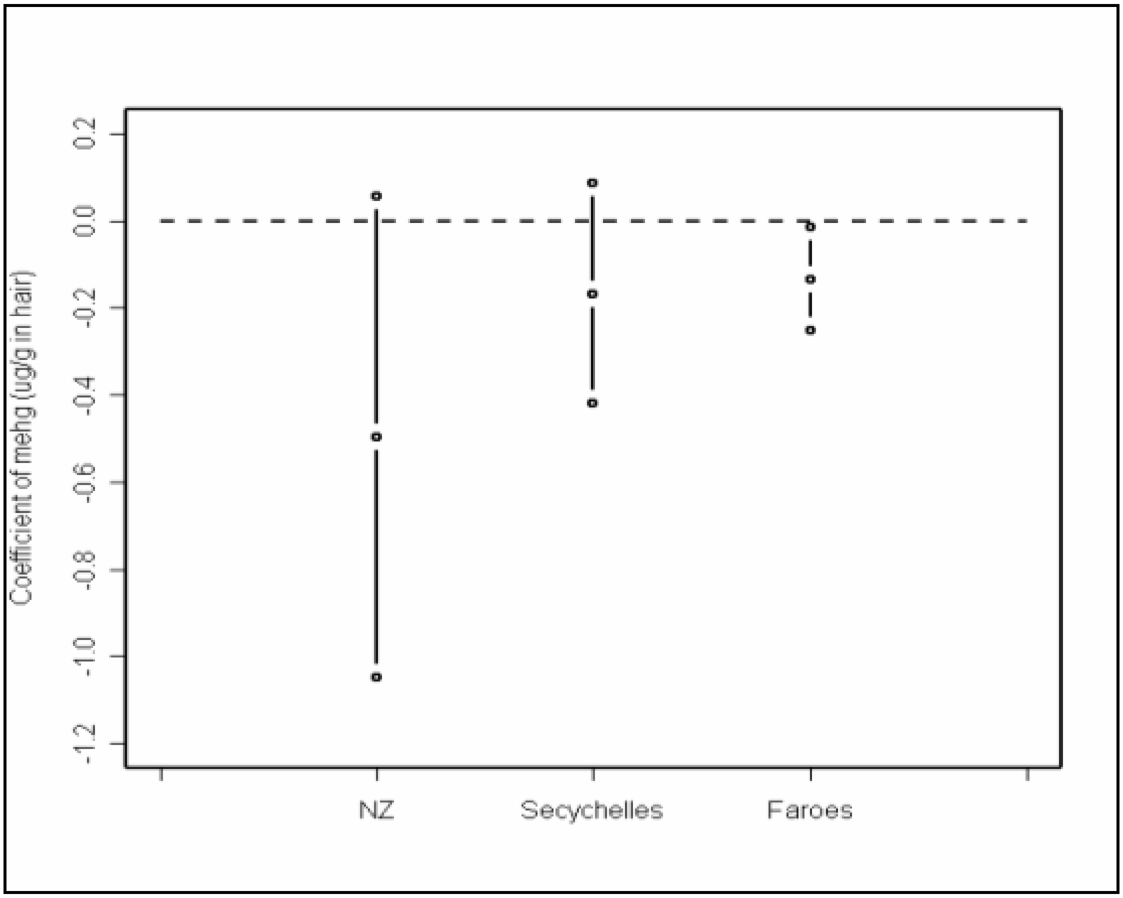

Figure 9-1. 95% Confidence Intervals for Full Scale IQ from the New Zealand, Seychelles and Faroes Studies

Coefficients for other cognitive outcomes used in the model were obtained from the same sources as the IQ dose-response coefficients (Crump et al. 1998, Myers et al. 2003, and Budtz-Jorgensen et al. 2005). Tests included in the model included: California Verbal Learning Test (Faroes and Seychelles); Boston Naming Test (Faroes and Seychelles); the Wide-Range Assessment of Memory and Learning (WRAML) and Visual-Motor Integration (VMI) (Seychelles only); Bender Visual Motor Gestalt Test (Faroes only); and the Test of Language Development - Spoken Language (TOLD-SL), WISC performance IQ, and McCarthy Scales of Children's Abilities perceptual performance scale (New Zealand only). Criteria for selection of these outcomes and details on how they were used in the model are described in the report on the statistical analysis (Ryan 2005).

The statistical analysis produced a dose-response relationship, integrating data from all three studies, with a central estimate of an IQ change of -0.13 IQ points (95% confidence interval -0.28, -0.03) for every ppm of mercury in maternal hair. This central estimate is close to the values for the Faroes and Seychelles studies, suggesting relatively little influence on the integrated value from the larger coefficient estimated in the New Zealand study. The smaller influence of the New Zealand coefficient is due to the smaller size of the cohort in this study, as well as the greater uncertainty in the central estimate of the dose-response coefficient in this study, as depicted in Figure 9-1.

Several sensitivity analyses were conducted, reflecting the following variations in the inputs to the model:

- Use of only the IQ dose-response coefficients, without the coefficients for the additional cognitive endpoints;
- Use of an alternate hair:blood mercury ratio for calculation of the Faroes coefficients in hair mercury terms;
- Use of the New Zealand coefficients that include one highly influential observation; and
- Use of an alternate interpretation of the Faroes structural equation model outputs for the IQ dose-response coefficient.

These sensitivity analyses found very consistent results, with central estimates all in the approximate range of -0.10 to -0.25 IQ points for each ppm of mercury in maternal hair. The results consistently suggested a significant association between mercury and IQ, with lower confidence limits ranging from about -0.2 to -0.5, and upper confidence limits between -0.02 and -0.04. Details of the sensitivity analyses are presented in the statistical analysis report (Ryan 2005).

9.4 Strengths and Limitations of the IQ Dose-Response Analysis

This analysis has produced, for the first time, an estimate of the relationship between maternal mercury body burdens during pregnancy and childhood IQs that incorporates data from all three epidemiologic studies judged by the NRC to be of high quality and suitable for risk assessment. The statistical approach makes use of all the available data (including information on results for related tests of cognitive function), and can be used to produce population-based estimates of a health outcome that can be readily monetized for use in benefit-cost analysis[1].

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[1] There is limited evidence directly linking IQ and methylmercury exposure in the three large epidemiological studies that were evaluated by the NAS and EPA. Based on its evaluation of the three studies, EPA believes that children who are prenatally exposed to low concentrations of methylmercury 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. For this analysis, EPA is adopting IQ as a surrogate for the neurobehavioral endpoints that NAS and EPA relied upon for the RfD.

In the Faroes Island Study, a full scale IQ evaluation was not conducted. However, two core subtests were evaluated (Similarities and Block design) and one supplementary test was conducted (Digit Span). The Similarities and Block Design tests are reported to be well correlated with the full WISC-R battery (0.885, see Bellinger (2005)), but how the Digit Span test relates is not reported. In the EPA analysis, we assume that it relates similarly. In the Faroes study, performance scores on the Similarities and Block Design tests were not shown to be statistically related to

There are several aspects of IQ as a metric for neurodevelopmental effects in this benefit-cost analysis that are important to recognize.

Full-Scale IQ is a composite index that averages a child's performance across many functional domains, providing a good overall picture of cognitive health. An extensive body of data documents the predictive validity of full-scale IQ, as measured at school-age, and late outcomes such as academic and occupational success (Neisser et al. 1996). In addition, methods are readily available for valuing shifts in IQ and thus conducting a benefits analysis of interventions that shift the IQ distribution in a population. Methods for monetization of the other tests administered in the three studies have not been developed.

It is important to recognize, however, that full-scale IQ might not be the cognitive endpoint that is most sensitive to prenatal mercury exposure. Significant inverse associations were found, in both the New Zealand and Faroe Islands studies, between prenatal mercury levels and neurobehavioral endpoints other than IQ. If the effects of mercury are highly focal, affecting only specific cognitive functions, taking full-scale IQ as the primary endpoint for a benefits analysis might underestimate the impacts. In averaging performance over diverse functions in order to compute full-scale IQ, the specific effects of mercury on only certain of these functions would be "diluted," and the estimated magnitude of the change in performance per unit change in the mercury biomarker would be underestimated.

Moreover, it is well-known that there may be substantial deficits in cognitive well-being even in individuals with normal or above average IQ. The criterion most frequently used to identify children with learning disabilities for the purposes of assignment to special education services is a discrepancy between IQ and achievement. Specifically, the child's achievement in reading, math, or other academic areas is significantly lower than what would be expected, given his or her full-scale IQ. Thus, there are deficits in cognitive functioning that are not captured by IQ scores. For example, two of the most sensitive endpoints in the Faroe Islands study were the Boston Naming Test, which assesses word retrieval, and the California Verbal Learning Test-Children, which assesses the acquisition and retention of information presented verbally. Depending on the severity of the deficits, a child who has deficits in either of these skills could be at a considerable disadvantage in the classroom setting and at substantial educational risk. Neither of these abilities is directly assessed by the WISC-R or WISC-III, however, and so do not explicitly contribute to a child's IQ score.. Therefore, benefits calculations relying solely on IQ decrements are likely to underestimate the benefits to cognitive functioning of reduced mercury exposures. In additions, impacts on other neurological domains (such as motor skills

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cord blood or maternal mercury levels; the Digit Span test did show a statistical relationship with cord blood mercury.

Both the New Zealand and Seychelles study administered the WISC IQ test (WISC III in Seychelles, WISC R in New Zealand). A reanalysis of the New Zealand data found a positive association, but it was not statistically significant. No significant associations were seen in the Seychelles study. As displayed in Figure 5 of Ryan (2005), the confidence intervals for full scale IQ in both these studies include zero. However, Ryan conducted an integrative analysis, combining results from all three studies. When combined, the statistical power of the analysis increases. While the size of the dose-response relationship declined relative to past studies with a statistically significant finding, Ryan found a statistically significant relationship between IQ and mercury. The confidence interval did not include zero.

and attention/behavior) are not represented by IQ scores and thus are also excluded from the benefits analysis.

As discussed above, the Faroe Islands study did not include testing for full-scale IQ. For this analysis, an estimate of a dose-response coefficient for full-scale IQ was estimated using the three subtests. While this extrapolation introduces some uncertainty, information has been presented that demonstrates a high correlation between the subtests and full-scale IQ scores.

While the Seychelles and New Zealand studies use maternal hair mercury as the exposure biomarker, the Faroe Islands study uses cord blood mercury. For purposes of the integrated analysis, it was necessary to express results from all three studies in the same terms. Several studies have examined the relationship between hair mercury and blood mercury, and have reported hair:blood ratios typically in the range of 200 to 300 (see ATSDR 1999, pages 249-252 for a review). However, these studies generally do not use cord blood mercury, which is the exposure metric in the Faroes study. A recent analysis found that mercury concentrations in cord blood are, on average, 70 percent higher than those in maternal blood (Stern and Smith 2003). For conversion of Faroes data from cord blood mercury to maternal hair mercury, it was most appropriate to use data specific to this population, indicating a median maternal hair:cord blood mercury ratio of 200 (Budtz-Jorgensen et al. 2004a). An alternate ratio of 250 was examined in sensitivity analysis, and resulted in an integrated dose-response coefficient that is reduced by about 12 percent (central estimate of -0.131 vs. -0.115).

This analysis relies on use of summary statistics, i.e. dose-response coefficients and associated variability statistics, for each of the three studies. Original data were not available for this analysis. While a lack of original data is often cited as a problem for cross-study analyses, its impact is lessened in this application for several reasons. All three studies had careful epidemiological designs that measured a variety of important potential confounders such as maternal age and education. All estimated dose-response coefficients were derived from well documented regression models that adjusted for age, maternal education and other important factors. Also, the National Research Council had asked the individual study investigators for very specific details about the way in which their analyses had been done and had also asked for additional analyses where necessary. Further, work by Dominici et al. (2000) took a similar approach for hierarchical modeling of estimated dose-response coefficients extracted from separate studies.

The major uncertainty concerning the New Zealand study is the strong influence of one child in the study population with a particularly high maternal hair mercury level. Published analyses of the New Zealand study presented results with data for this child both included and excluded (Crump et al. 1998). In keeping with the conclusions of the NRC (2000), the integrated dose-response analysis presented in this section made use of the dose-response coefficients calculated with this child omitted. A sensitivity analysis using the New Zealand coefficient with this child included results in an integrated dose-response coefficient that is reduced by about 17 percent (central estimate of -0.131 vs. -0.108).

Some uncertainty is also associated with the Seychelles study due to the exclusion of some members of the cohort from the data reported by Myers et al. (2003) and used as input to this integrated dose-response analysis. The Seychelles researchers did not include a small number of outliers (defined as observations with model residuals exceeding 3 standard deviation units), and no results are available for the full cohort. However, the authors report that "In all cases, the association between prenatal MeHg exposure and the endpoint was the same, irrespective of whether outliers were included" (Myers et al. 2003).

Finally, the integrated dose-response analysis assumes the exposures assigned to each study subject are accurate representations of true exposure. In reality, there is likely to be some discrepancy between measured and actual exposures, for example, due to variation in hair length. Alternatively, the true exposure of interest may have been during the first trimester of pregnancy, whereas exposures in maternal hair and cord blood measured at birth reflect exposures later in pregnancy. Presence of exposure measurement error could introduce a bias in the results, most likely towards the null (Budtz-Jorgensen et al. 2004b).

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Tables

Figures

SECTION 10

EXPOSURE MODELLING AND BENEFIT METHODOLOGY WITH AN APPLICATION TO A NO-THRESHOLD MODEL

10.1 Introduction

In this section, we describe two exposure modeling approaches designed to provide an estimate the underlying benefits analysis of reducing mercury emissions that may result from the Clean Air Mercury Rule. Sections 10 and 11 together form the basis for our benefits methodology and calculations. In this Section, we construct a scenario that reflects an upper-bound on the number of people affected by mercury. In particular, the scenario incorporates an assumption of no theshold. We estimate benefits with this assumption by deploying a very disaggregated, spatially-rich model. This exercise provides very detailed results. In Section 11, the model is simplified a bit by aggregating recreational fishers into descrete bins or categories, making the analysis much more manageable. The analysis in Section 11 simulates exposure scenarios under the assumption of thresholds. Two different thresholds are explored. The threshold analysis gives "scaling factors" or benefits as a percent of the no threshold case developed in this Section. Benefit estimates are then estimated by multiplying the scaling factors by the benefits calculated in this Section. Hence, this Section forms the core analytic underpinnings for the final benefit numbers that are derived and presented in Section 11.

In this Section, we quantify and monetize, to the extent feasible, benefits associated with modelled avoided IQ deficits due to reduced exposure from the consumption of recreationally-caught freshwater fish assuming there is no threshold in effects at low doses of mercury. The analysis focuses on estimating changes in exposures to women of childbearing age because adverse health effects in children have been linked to prenatal mercury exposures. In addition, because mercury emissions in the U.S. predominantly affect the eastern-half of the country, the analysis is also focused on affected populations in that part of the country. While the geographic coverage and the exposed population largely reflect the areas impacted by the CAMR, it should be noted that, as Section 4 discusses, this analysis focuses on freshwater exposures in the eastern-half of the U.S., which will reduce the size of the exposed population considered for analysis. This focus for the analysis is necessary because of limitations in modeling how changes in mercury deposition will effect fish tissue concentrations for the other fish consumption pathways discussed in Section 4 of this report and there is relatively little fish tissue information for the Western-half of the U.S. As discussed in Section 8 the largest change in power plant deposition associated with the recently finalized CAIR and CAMR program will occur in the eastern-half of the U.S., so the unquantified benefits for the western-hald of the U.S. is expected to be quite small. As is discussed in previous sections, we are unable to quantify several categories of potential benefits, such as benefits from other health and ecological effects, as well as commercial and recreationally-caught saltwater species. As metioned throughout the report, power plant emission reductions under CAIR and CAMR will have a minimal effect on exposure levels associated with these other consumption pathways. Our benefit assessment has several known uncertainties and biases, which are discussed further in Section 10.7. While some of these are downward biases and some are upward biases, taken together, the Agency believes that the benefits presented in this section likely underestimate the total benefits of reducing mercury emissions from power plants due to the potential health effects and potentially exposed populations that are not quantified in this analysis.

10.1.1 Summary

The basic methodology used in this Section is to project the change in IQ of a population of children due to mercury exposure *in utero*. The exposure is based on consumption of fish by pregnant women. The mercury in the fish is due in part to atmospheric deposition of mercury from power plants. A monetary value is placed on incremental loss of IQ by these children[1]. The incremental reduction in exposure due to mercury emission reductions from power plants is then applied to this methodology to calculate the improvement in IQ and the monetary value of that improvement, attributable to the emissions reduction. The study examines only consumption of freshwater fish, because our analysis indicates that these are the only fish significantly impacted by U.S. power plants.

The analysis first examines impacts on the general population of children of freshwater fishers. It then considers much smaller populations that consume greater amounts of fish than the general population, including subsistence fishers, certain Native Americans, and Asian Americans.

With respect to impacts on the general population, two methods of approaching the problem were used, a "Population Centroid" and an "Angler Destination" approach. These approaches reflect different ways of estimating where freshwater fishers fish. Table 1 shows the relative impacts on IQ deriving from different mercury emission rates and the two different analytical approaches, for the average child in the general population. More detailed results are presented in the body of this Section.

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[1] There is limited evidence directly linking IQ and methylmercury exposure in the three large epidemiological studies that were evaluated by the NAS and EPA. Based on its evaluation of the three studies, EPA believes that children who are prenatally exposed to low concentrations of methylmercury 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. For this analysis, EPA is adopting IQ as a surrogate for the neurobehavioral endpoints that NAS and EPA relied upon for the RfD.

In the Faroes Island Study, a full scale IQ evaluation was not conducted. However, two core subtests were evaluated (similarities and block design) and one supplementary test was conducted (Digit Span). The similarities and block design tests are reported to be well correlated with the full WISC-R battery (0.885 see Bellinger paper), but how the Digit Span test relates is not reported. In the EPA analysis, we assume that it relates similarly. In the Faroes study, performance scores on the similarities and block design tests were not shown to be statistically related to cord blood or maternal mercury levels; the digit span test did show a statistical relationship with cord blood mercury.

Both the New Zealand and Seychelles study administered the WISC IQ test (WISC III in Seychelles, WISC R in New Zealand). A reanalysis of the New Zealand data found a positive association, but it was not statistically significant. No significant associations were seen in the Seychelles study. In the EPA analysis, the the confidence intervals for full scale IQ in both these studies include zero. However, Ryan (2005) conducted an integrative analysis, combining results from all three studies. When combined, the statistical power of the analysis increases. While the size of the dose-reponse relationship declined relative to past studies with a statistically significant finding, Ryan found a statistically significant relationship between IQ and mercury. The confidence interval did not include zero.

The data in Table 10-1a show that both analytical approaches yield similar results. A typical child of freshwater fishers lost approximately 0.06 - 0.07 IQ points due to mercury exposure in 2001, depending on the analytical approach. Average IQ is, by definition, 100 points. Implementing CAIR would reduce this IQ loss by a little less than 0.007 to 0.009 IQ points in 2020. Under Options 1 and 2 of the CAMR, this reduction would be increased by 0.0006 to 0.0009 IQ points. Total elimination of power plant emissions would have about the same effect as CAIR Option 2. Focusing on the Population Centroid approach, it is seen that CAIR reduces the 2001 mercury impact on IQ by 11.8%; CAIR plus CAMR Option 1 reduces the impact by 12.7% (an additional 0.9%); and totally eliminating power plant mercury emissions reduces the 2001 impact by 13.2% (another 0.5% beyond CAIR plus CAMR Option 1.

Table 10-1(a). Summary of Per Capita Changes in IQ Due to Mercury Exposure

Measurement of IQ impact per capita (average impact over study population)	Approach	
	Population Centroid	Angler Destination
IQ loss due to mercury exposure in 2001	0.0621	0.069
IQ loss in 2020 under CAIR emission reductions	0.0548	0.060
Avoided IQ loss due to CAIR in 2020, relative to 2001	0.0073	0.0089
Avoided IQ loss with no power plant emissions in 2020, relative to 2001	0.0082	0.0090
Avoided IQ loss w/ no power plant emissions in 2020, vs CAIR in 2020	0.0009	0.0001
IQ loss in 2020 w/ CAIR & Option 1 of CAMR	0.0542	0.0594
Avoided IQ loss due to CAIR & Option 1, vs CAIR alone	0.0006	0.0006
IQ loss in 2020 w/ CAIR & Option 2 of CAMR	0.0539	0.0591
Avoided IQ loss due to CAIR & Option 2, vs CAIR alone	0.0009	0.0009

In short, the overall impact of mercury on the IQ of children in the general population is relatively small, less than one-thousanth of a normal IQ (Normal = 100). Implementing CAIR dramatically reduces the contribution of power plants to this small projected mercury impact on children, and CAMR Option 1 eliminates the majority of the remaining impact associated with power plants.

We apply a value of about $8,800 (net present value) per IQ point improvement per capita. Thus, the value of CAMR Option 1 is equal to the number of exposed children x the mean improvement in IQ (0.0006 points) x $8,800[2], or $2.6 million. In the body of this Section, this number is adjusted to reflect various "lag" periods, to reflect the amount of time required for emission reductions to result in changes in fish mercury concentrations.

In addition to the analysis of the general US population, this Section also assesses the benefit of CAMR on subsistence anglers, Native Americans, and Asian Americans, who consume more fish than the general population. Table 2 presents results similar to Table 1, for subsistence fishers and two Native American tribes.

As expected, a larger impact on IQ due to mercury was found for these smaller groups, with an average IQ impact in 2001 from all mercury sources of 0.331 IQ points on children of subsistence fishers, for example. Implementation of CAIR reduced this impact to 0.290 IQ points in 2020. Application of CAMR Option 1 reduced impacts by an additional 0.0033 points and complete elimination of emissions from US power plants reduced impacts by another 0.012 points, or down to 0.275 points. Hence, for this more sensitive group, CAIR again provides the bulk of the reduction possible by controlling power plants, but in this case totally eliminating power plant emissions can provide about a one-hundredth of a point of improvement in average IQ, compared to CAMR Option 1.

For the Native American tribes (the Hmong and the Chippewa), current impacts on IQ are estimated to be about 0.1 IQ point. CAIR provided no benefit to the Hmong, where power plants contribute only about 6% of the impact associated with mercury consumption. For the Chippewa, CAIR reduced impacts about 11%. The CAMR options reduced impacts another 0.8% or 1.5%, and total elimination of power plant emissions would contribute another 8.5% reduction. In absolute terms, the effect of total elimination of power plant mercury emissions, beyond CAMR Option 1, was projected to be about one-hundredth of an IQ point.

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[2] This value is based on foregone earnings over a lifetime discounted at 3 percent. The value per IQ point when calculated at a 7 percent discount rate is $1580 per IQ point (1999$).

Table 10-1(b). Impacts of Mercury on High Fish Consuming Groups

Measurement of IQ impact per capita	Subsistence Fishers	Hmong (MN, WI)	Chippewa (MI, MN, WI)
IQ loss due to Hg Exposure in 2001, mean per capita	0.3310	0.1140	0.1340
IQ loss in 2020 under CAIR	0.2900	0.1140	0.1220
Avoided IQ loss due to CAIR 2020, v 2001	0.0410	(0.0007)	0.0150
Avoided IQ loss w/ no PP Hg 2020, v 2001			
Avoided IQ loss w/ no PP Hg 2020, v CAIR 2020	0.0154	0.0069	0.0130
IQ loss in 2020 w/ CAIR & CAMR Opt1	0.2867	0.1136	0.1210
Avoided IQ loss due to CAIR & CAMR Opt 1, v CAIR	0.0033	0.0004	0.0010
IQ loss in 2020 w/CAIR & CAMR Opt2	0.2852	0.1126	0.1200
Avoided IQ loss due to CAIR & CAMR Opt2, v CAIR	0.0048	0.0014	0.0020
Number of children in group	22,302	553	1,094

10.1.2 Modeling Overview

The mercury benefits model developed to support this analysis estimates the IQ decrement for children of recreational freshwater fishers exposed prenatally to methylmercury through maternal fish consumption. The model is designed to provide two types of benefits results:

1. Total reductions in IQ decrement (and associated dollar values) for the entire modeled population of prenatally exposed children; and

2. Distributional results in the form of per-capita reductions in IQ decrements for each of the modeled children in the analysis population.

The first category of results (total IQ benefits) can be used to support a traditional cost-benefit analysis comparing the total monetized benefits against total monetized costs. The second category of results (distribution of per-capita IQ losses) can be used to examine the distributional equity of IQ impacts across the study population of prenatally-exposed children (e.g., what is the range of individual IQ changes across the study population and how many children are projected to have IQ changes above specific levels of interest?).

In addition to generating benefits estimates for the children of recreational freshwater fishers, the mercury benefits model also provides benefits estimates for several high-exposure sub-populations including:

1. A high fish consumption rate study population that is defined as “subsistence” fishers for the purposes of this study;

2. A low-income high fish consumption study population (an alternative approach to model subsistence fishers);

3. A Southeast Asian ethnic group with high freshwater fish consumption due to cultural practices; and

4. A Native American population with high freshwater fish consumption due to cultural practices.

These high-exposure scenarios are intended to provide coverage for special populations potentially experiencing health impacts from the consumption of self-caught freshwater fish due to increased consumption rates.

Because key factors in modeling freshwater fisher exposure (e.g., methylmercury fish tissue concentrations, mercury deposition rates from power plants, the distribution of fishers and fishing activity) can display significant spatial variability, the mercury benefits model has been developed to provide adequate spatial resolution for a regulatory analysis of modeling fishing activity and subsequent mercury exposure associated with CAMR. The mercury benefits model has been developed to provide coverage for local- to regional-scale trends in fisher exposure linked to more generalized spatial patterns of fishing activity, mercury fish tissue concentrations and mercury deposition. The model also considers variability in the consumption rate of self-caught freshwater fish by fishers, which is not necessary for a total (best estimate) prediction of IQ impacts, but which is critical in modeling the distribution of per-capita IQ impacts across the study population. By considering local- to regional-scale trends in patterns of recreational fisher exposure as well as variability in fish consumption rates, this model provides a quantitative assessment of the distribution and magnitude of exposures and IQ decrements across the fisher study population (prenatally-exposed children).

In addition, because of the complexity in modeling fishing activity, two models of freshwater fishing behavior have been developed for this analysis. One model (the "population centroid" approach) represents a "push" model in that it focuses first on identifying where recreational fishers live and then models their fishing behavior in the form of fishing trips out to different distance rings (10, 20, 50 and 100 miles) from their home residences. This model is applied at the US Census block group level, which results in exposure estimates being generated for a relatively large number of polygons (165,000 block groups in the study area). The second model (the "angler destination" approach), represents a "pull" model in that it focuses on identifying where anglers fish and does not consider their residential location. This model is applied at the watershed-level as identified by USGS 8-digit hydrologic unit code (HUC) and assesses the distribution of recreational fishing activity across HUCs in the study area. Because fishing activity (behavioral) data used in the HUC model does not include coverage for special subpopulations evaluated in this analysis (e.g., Native Americans, Southeast Asian subpopulations and economically disadvantaged subsistence subpopulations), these specialized analyses were implemented using the population centroid approach. In addition, due to the greater spatial precision of the population centroid model, relative to the HUC model, the population centroid model was also used as the basis for generating distributional (per-capita IQ impact) results for the fishers.

The mercury benefits model generates health impact and valuation results by first estimating the change in total mercury deposition over waterbodies within the 37 state study area (mercury deposition is modeled using CMAQ).[3] These changes in mercury deposition are generated by comparing two air modeling scenarios (e.g., a control scenario versus a baseline scenario for a particular simulation year). These changes in mercury deposition are then translated into changes in methylmercury fish tissue concentrations based on the proportionality assumption advanced in Mercury Maps (i.e., a incremental percent change in deposition produces a matching percentage change in mercury fish tissue concentrations)[4]. Modeled changes in methylmercury fish tissue concentrations can, in turn, be used together with the fishing behavioral models described above, to predict changes in population-level mercury exposure. These exposure changes can be translated through modeling into IQ reductions, which can then be monetized using valuation functions based primarily on foregone (lost) earnings resulting from reductions in IQ. Case studies of individual ecosystems (as presented in Section 3) 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. Thus, benefits results generated for this analysis are reported using a range of lag periods following regulatory implementation (e.g., 5, 10, 20, and 50 years). Based on the response times from the case studies discussed above, we present a range of benefits based on the 10 and 20 year lag as central estimates.[5] We also provide results for the 5 and 50 years to demonstrate how benefits would differ under potential shorter and longer lag periods. Modeling of benefits for these different lags reflects the effects of economic discounting as well as demographic growth in the exposed population.

10.1.3 Monetized Benefits: Results in Brief

The mercury benefits analysis generated two categories of results including total IQ decrements and associated monetary (dollar) values for modeled populations and distributional results in the form of per-capita IQ reduction estimates for the group of modeled individuals. Total IQ decrement and valuation results were generated for the recreational fisher population as well as for the four potentially high-risk sub-populations described above. Distributional results (per-capita IQ decrements) were generated only for the recreational fisher population.

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[3] The 37 states included in the analysis are the following (plus the District of Columbia): Connecticut, Maine, Massachusetts, New Hampshire, Rhode Island, Vermont, Delaware, Maryland, New Jersey, New York, Pennsylvania, Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, Missouri, North Carolina, South Carolina, Tennessee, Virginia, West Virginia, Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Nebraska, North Dakota, Ohio, South Dakota, Wisconsin, Oklahoma, and Texas

[4] There are several limitations with the Mercury Maps approach that are discussed fully in Section 3 of this report. In particular, it applies only to waterbodies where air deposition is the primary source of mercury load on a system. In Section 10.7.1.1, we estimate the number of areas (HUCs) that have non-air deposition to the waterbodies and remove them from the analysis to determine the affect on total benefits.

[5] A 30 year lag is also indicated by the case studies in Section 3, but are not provided in the benefit analysis in this Section. EPA expects that results of the 30 year lag would not significantly differ from the 20 year lag presented in this Section.

The following conditions and emissions control scenarios were modeled for this RIA:

- 2001 Base Case
- 2001 Utility Emissions Zero-Out
- 2020 Base Case with CAIR
- 2020 Utility Emissions Zero-Out
- 2020 CAMR Control Option 1
- 2020 CAMR Control Option 2

Table 10-1c provides a summary of the total benefits estimated for each of these emissions control scenarios.

Table 10-1(c). Summary of Total Benefits Associated with Modelled Avoided IQ Decrements in Prenatally Exposed Children Due to Reduced Mercury Exposure from Freshwater Recreational Angling

Emission Control Scenario	Range of Estimated Benefits Associated with Modelled Avoided IQ Decrements Due to Mercury Exposure (millions of 1999 dollars)[a]			
	Recreational Freshwater Angler[b]	Subsistence Anglers[c]	Native American Case Study Population[d]	Asian American Case Study Population[e]
2001 Zero Out of EGU Emissions (Relative to 2001 Baseline Emissions)				
-Using a 3% discount rate	$19.0 - $37.0	$ 4.9 - $ 6.7	$0.10 - $0.12	$0.047-$0.050
-Using a 7% discount rate	$ 8.9 - $20.2	$ 2.3 - $4.6	$0.05 - $0.08	$0.021-$0.034
2020 Base Case (with CAIR) (Relative to 2001 Baseline Emissions)				
-Using a 3% discount rate	$20.5 - $43.8	$ 4.9 - $7.0	$0.12 - 0.13	$0.005-$0.007
-Using a 7% discount rate	$ 9.6 - $30.0	$ 2.3 - $4.8	$0.6 - $0.9	approx. $0.003
2020 Zero Out of EGU Emissions (Relative to 2020 Base Case with CAIR)				
-Using a 3% discount rate	$ 8.1 - $16.1	$ 2.2 - $2.7	approx. $0.10	$0.060-$0.064
-Using a 7% discount rate	$ 3.8 - $11.0	$ 1.0 - $ 1.8	$0.05 - $0.08	$0.028-$0.043
CAMR Option 1 (Relative to 2020 Base Case with CAIR)				
-Using a 3% discount rate	$ 1.7 - $ 3.0	$ 0.5 - $ 0.6	approx. $0.007	approx. $0.003
-Using a 7% discount rate	$ 0.8 - $ 2.0	$ 0.2 - $ 0.4	$0.005-$0.003	$0.001-$0.002
CAMR Option 2 (Relative to 2020 Base Case with CAIR)				
-Using a 3% discount rate	$ 2.5 - $ 4.6	$ 0.7 - $ 0.9	approx. $0.015	approx. $0.012
-Using a 7% discount rate	$ 1.2 - $ 3.1	$ 0.3 - $ 0.6	$0.007-$0.010	$0.006-$0.009
Combined Benefits of CAIR and CAMR (CAMR Option 1 + 2020 Base with CAIR Relative to 2001 Base Case)				
-Using a 3% discount rate	$22.2 - $46.8	$ 5.5 - $7.6	$0.011-$0.012	$0.008-$0.010
-Using a 7% discount rate	$10.4 - $32.0	$ 2.5 - $5.2	$0.050-$0.080	$0.023-$0.036

a The value per IQ point used to calculcate total benefits presented in this table is $8800/IQ point and is based on a 3 percent discount rate of net earnings over a lifetime. The value per IQ point at a 7 percent discount rate is $1580/ IQ point (1999$) according to EPA(1997c).

b The range of results presented for the Recreational Freshwater Angler of recreational anglers are based on potential outcomes from the Angler Destination exposure modeling and the Population Centroid exposure modeling discussed in this chapter, as well as results ranging in value from a 10 to 20 year lag. See Table 10-21 and Table 10-29 for the full matrix of potential results from the benefit modeling of this population.

c The range of results presented for Subsistence Anglers are based on potential outcomes from the Income-based modeling and the Consumption-based modeling discussed in this chapter, as well as results ranging in value from a 10 to 20 year lag. See Table 10-33 for the full matrix of potential results from the benefit modeling of this population.

d The range of results presented for the Native American case study of the Chippewa in Minnesota, Wisconsin, and Michigan are based on potential outcomes from the Population Centroid exposure modeling discussed in this chapter, as well as results ranging in value from a 10 to 20 year lag. See Table 10-42 for the full matris of potential results frm the benefit modeling of this population.

e The range of results presented for the Asian American case study of the Hmong in Minnesota and Wisconsin are based on potential outcomes from the Population Centroid exposure modeling discussed in this chapter, as well as results ranging in value from a 10 to 20 year lag. See Table 10-41.

10.1.4 Key Steps

The process used for estimating mercury exposures is divided into three general steps:

1. Estimate mercury levels in freshwater fish across the eastern half of the United States.

2. Estimate the size of the exposed populations of interest (i.e., prenatally exposed children of freshwater anglers) and their spatial relation to mercury levels in freshwater fish (i.e., location and methylmercury concentration in fish consumed by mothers).

3. Estimate fish consumption and mercury ingestion rates for the mothers of prenatally exposed children of freshwater anglers.

Based on these exposure estimates, it is then possible to estimate associated health decrement and monetary losses. The process for quantifying these losses can be summarized as:

1. Estimate reductions in expected IQ levels for the exposed population

2. Estimate the expected value of foregone future earnings associated with the IQ decrements.

To estimate the benefits of mercury emissions reductions, the preceding exposure assessment and IQ valuation steps were conducted under two baseline (i.e., "base case") scenarios—one for 2001 and the other for 2020 (with CAIR)—and four emissions reduction scenarios. The benefits associated with each of the emissions reduction scenarios, in particular the CAMR control options, were then estimated as the difference (reduction) in the total value of IQ losses, going from the relevant baseline scenario to conditions with the emissions reductions in place. These steps and the results are described in detail in the following sections of this report.

Section 10.2 describes the data sources and methods used to estimate the spatial distribution of mercury levels in freshwater fish across the eastern United States. It also summarizes the estimates of mercury concentrations based on these methods.

Section 10.3 describes data sources and two discrete methods for estimating the number of modelled prenatally exposed children and average levels of mercury in freshwater fish consumed by their mothers. Results from applying the two methods are also reported and compared.

Section 10.4 describes the data, modeling approach, and results for estimating levels of mercury ingestion through consumption of noncommercial freshwater fish in the study area. The modeling approach was applied with both of the methods described in Section 10.3 to estimate distributions of mercury ingestion rates across the exposed population. This section also describes methods for estimating IQ decrements and foregone future earnings resulting from the estimated exposures.

Section 10.5 summarizes results and provides quantitative estimates of the benefits associated with the zero out scenario. Estimates of exposures, IQ decrements, and foregone future earnings are reported and compared for both baseline conditions and for the zero out scenario.

Section 10.6 examines the distributional implications of the regulation by describing how the estimated changes in IQ effects are distributed across the exposed population.

Section 10.6 also adapts the methods described in Sections 10.3 and 10.4 to assess mercury exposures, IQ effects, and benefits for potentially highly exposed subpopulations. Three methods are explored for evaluating high mercury exposures, each of which focuses on subpopulations with high expected rates of freshwater fish consumption. The first method focuses on exposures among individuals in the top fifth percentile of freshwater fish consumption rates. The second method implements an approach for examining exposures among low-income individuals for whom self-caught freshwater fish may be an integral part of their diet. The third method focuses on two specifically studied ethnic groups in the Great Lakes area who have been found to consume relatively high rates of noncommercial freshwater fish—the Hmong and the Chippewa Indians.

Section 10.7 discusses the main assumptions, limitations, and uncertainties associated with the methods described in the report. It also describes the unquantified benefits associated with reductions in mercury emissions.

10.2 Estimation of Mercury Levels in Freshwater Fish

To estimate mercury levels in consumed freshwater fish across the eastern half of the United States, this analysis relied primarily on monitoring data (i.e., fish tissue samples drawn from freshwater sites across the study area). A potential alternative to monitored data would be to estimate mercury levels based on dynamic and localized fate and transport modeling; however, models of this type present significant technical challenges when applied at regional/national spatial scale.[6]

Data from the NLFA and NLFTS were used as inputs into the NDMMF[7] model to generate MeHg fish tissue concentrations normalized to the typical sizes of frequently targeted fish species (largemouth bass, walleye, crappie, catfish, trout, and perch). All six species concentrations were generated for each waterbody, then averaged to develop a representative MeHg concentration for that waterbody at a given sample date.[8] Where a single location was

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[6] An independent peer review of the benefits methodology indicated that ecosystem based fate and transport modeling of bioaccumulation of MeHg in fish tissue for a national scale analysis would not be practical or even feasible at this time.

[7] A comprehensive evaluation of the performance of the NDMMF, and detailed description of the model are available in Appendix E.

[8] Details related to the selection of MeHg sample data, variability within the data, how the NDMMF was applied, and overall concentrations is provided in ch. 5.

sampled on multiple dates, the representative MeHg concentrations were averaged at that location. The resulting estimates are summarized by state in Table 10-2.

After MeHg concentrations are computed for every sample location, the concentrations are then linked to exposed populations using two exposure modeling methods. The angler destination approach, described in section 10.1.2, evaluates exposure based on fishing pressure at the HUC, thus, we estimate the average fish tissue concentration for lakes and rivers located in each HUC in the study area. In the second approach, the population centroid approach discussed in Section 10.1.2, concentrations are linked to populations based on trip travel distance rings. More detail about these two approaches is provided later in this section.

The values reported in Table 10-2 are based on these estimated averages per sampling location. Across all sampling locations the average (median) estimated mercury concentration in fish tissue was 0.23 ppm (0.18 ppm) for lake sites and 0.25 ppm (0.19 ppm) for river sites.

Table 10-3 describes how the lake and river mercury concentration data (summarized in Table 10-2) are distributed across the 1,362 HUCs in the study area. Over 60 percent of HUCs are without lake estimates of mercury concentrations in fish, and almost 50 percent are without river estimates. For the 512 HUCs with fish tissue concentration estimates for lakes, the average HUC level concentrations range from 0.004 ppm to 1.5 ppm. Roughly 15 percent have average mercury concentrations below 0.1 ppm.

The minimum and maximum mercury concentration in the lake fish are 0.004 ppm in Illinois and 2.64 ppm in Pennsylvania. In the case of 707 HUCs with river estimates, the mean mercury concentration in HUCs range from 0.004 ppm to 2.2 ppm. Roughly 25 percent HUCs have average mercury concentrations below 0.1 ppm. The minimum and maximum mercury concentration in the river fish are 0.0003 ppm in Louisiana and 3.3 ppm in Pennsylvania.

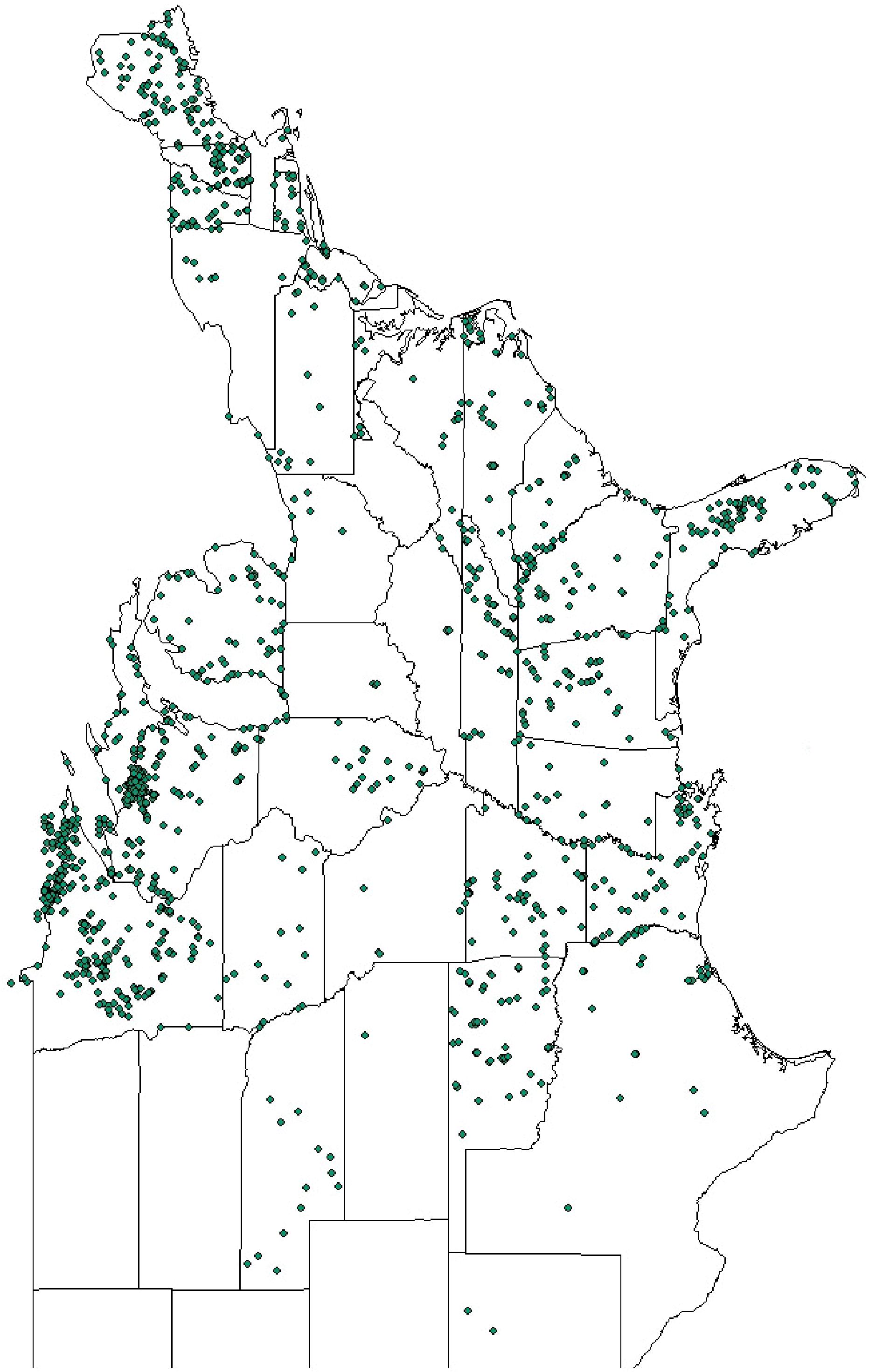

Figure 10-1. Locations of Lake Fish Tissue Mercury Sampling Sites Used in the Analysis

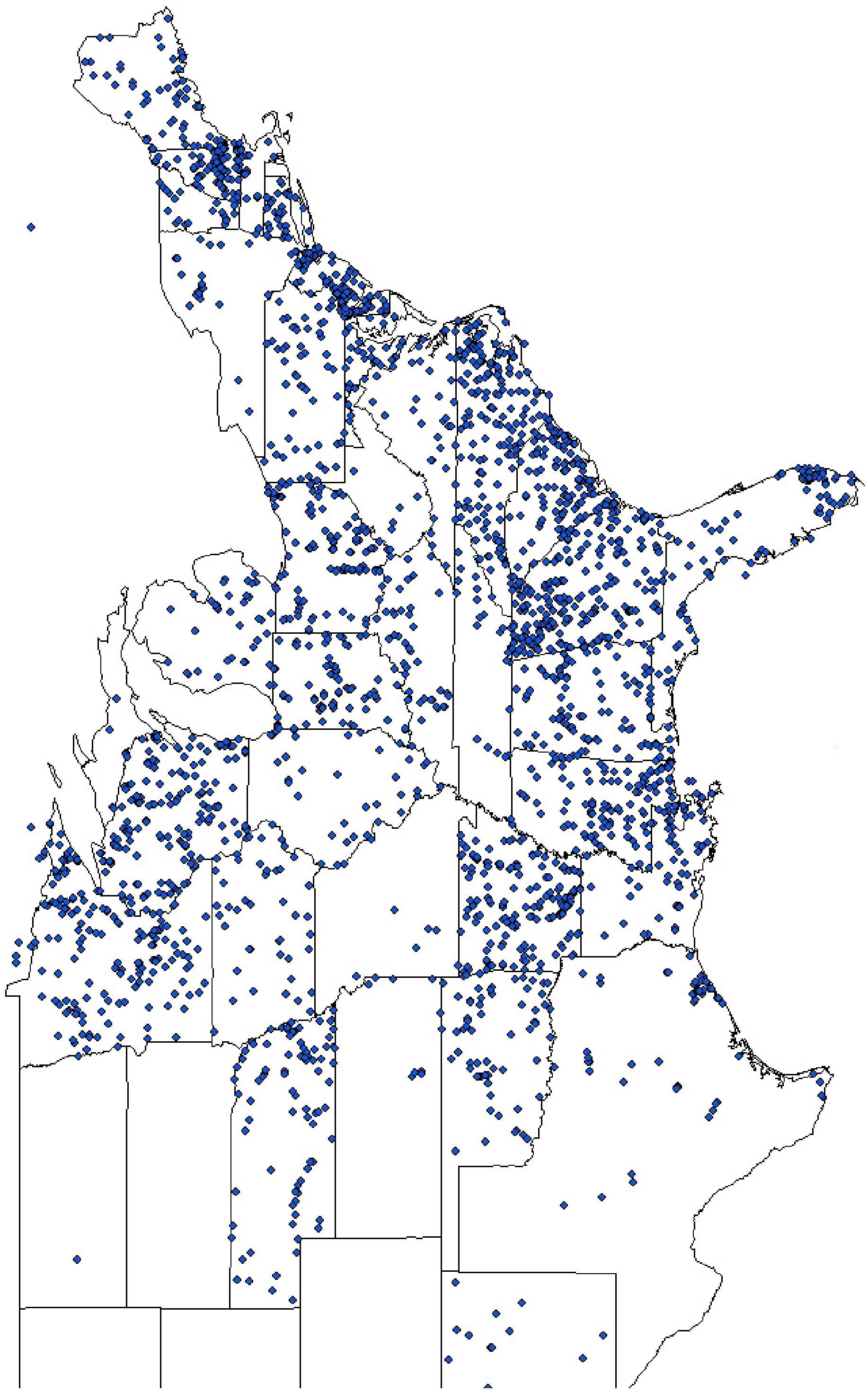

Figure 10-2. Locations of River Fish Tissue Mercury Sampling Sites Used in the Analysis

Table 10-2. Summary Statistics for Estimated Fish Tissue Mercury Concentrations (ppm) by State: 2001 Base Case[a]

	Lake Sampling Sites					River Sampling Sites				
State	Number	Min	Mean	Max	Median	Number	Min	Mean	Max	Median
AL	56	0.011	0.113	0.575	0.076	104	0.012	0.157	1.003	0.094
AR	50	0.016	0.261	0.943	0.221	174	0.009	0.340	1.311	0.295
CT	20	0.072	0.311	1.511	0.246	38	0.030	0.215	0.581	0.194
DC					0.000	5	0.070	0.109	0.165	0.090
DE	3	0.036	0.070	0.102	0.102	80	0.002	0.102	1.563	0.072
FL	91	0.022	0.299	1.105	0.282	98	0.027	0.374	0.983	0.353
GA	65	0.013	0.195	0.674	0.174	258	0.002	0.282	3.223	0.156
IA	18	0.030	0.069	0.168	0.060	45	0.030	0.131	0.480	0.110
IL	31	0.004	0.081	0.358	0.071	41	0.005	0.097	0.552	0.076
IN	15	0.030	0.124	0.311	0.088	109	0.030	0.186	0.502	0.173
KS	3	0.103	0.361	0.649	0.649	9	0.023	0.150	0.380	0.140
KY	9	0.099	0.153	0.268	0.154	51	0.039	0.170	0.548	0.159
LA	52	0.013	0.175	1.489	0.126	44	0.000	0.238	3.190	0.152
MA	6	0.124	0.266	0.394	0.278	1	0.578	0.578	0.578	0.578
MD	7	0.015	0.061	0.177	0.064	35	0.007	0.046	0.173	0.030
ME	92	0.026	0.480	1.020	0.479	52	0.134	0.496	1.787	0.453
MI	179	0.048	0.190	0.624	0.165	67	0.066	0.181	0.477	0.172
MN	309	0.018	0.173	1.022	0.145	216	0.014	0.164	0.734	0.143
MO	8	0.050	0.152	0.350	0.120	7	0.044	0.162	0.287	0.171
MS	7	0.080	0.215	0.299	0.244	18	0.082	0.387	0.727	0.343
NC	84	0.065	0.304	1.210	0.228	341	0.028	0.249	0.985	0.203
ND	16	0.096	0.204	0.377	0.185	4	0.168	0.325	0.517	0.307
NE	17	0.029	0.136	0.391	0.129	127	0.001	0.135	0.647	0.111
NH	56	0.037	0.390	1.425	0.350	76	0.059	0.348	0.804	0.298
NJ	10	0.081	0.254	0.753	0.162	64	0.068	0.337	2.129	0.207
NY	66	0.033	0.310	0.947	0.266	118	0.037	0.320	1.350	0.251
OH	100	0.013	0.205	1.037	0.149	504	0.010	0.252	0.980	0.240
OK	68	0.017	0.214	1.164	0.179	66	0.016	0.223	0.648	0.179
PA	19	0.061	0.546	2.640	0.255	60	0.020	0.403	3.300	0.170
SC	39	0.109	0.221	0.497	0.208	162	0.123	0.493	2.479	0.375
TN	44	0.090	0.184	0.505	0.158	21	0.100	0.194	0.414	0.170
TX	41	0.006	0.141	0.363	0.137	57	0.004	0.182	1.228	0.147

(continued)

Table 10-2. Summary Statistics for Estimated Fish Tissue Mercury Concentrations (ppm) by State: 2001 Base Case (continued)

	Lake Sampling Sites					River Sampling Sites				
State	Number	Min	Mean	Max	Median	Number	Min	Mean	Max	Median
VA	18	0.018	0.146	0.470	0.092	38	0.011	0.118	1.900	0.050
VT	31	0.054	0.291	0.641	0.270	22	0.069	0.345	0.945	0.279
WI	187	0.029	0.201	0.900	0.182	204	0.029	0.240	0.897	0.212
WV	6	0.170	0.296	0.400	0.306	45	0.034	0.195	0.848	0.117
All	**1,823**	**0.004**	**0.227**	**2.640**	**0.175**	**3,361**	**0.000**	**0.254**	**3.300**	**0.185**

[a] For summary purposes, in this table the data are summarized at a state level; however, the data used in the analysis were analyzed at a Census block group level or at a HUC level.

Table 10-3. HUC-Level Distribution of Mercury Sampling Sites and Estimated Fish Tissue Concentrations: 2001 Base Case [a]

	Lake		River	
	Number[b]	Percent	Number[b]	Percent
Number of Samples in HUC				
0	850	62.41	655	48.09
1–3	366	26.87	429	31.50
4–10	125	9.18	205	15.05
11–25	16	1.17	60	4.41
26+	5	0.37	13	0.95
Average Concentration (in ppm) in HUC				
No data	850	62.41	655	48.09
0.00–0.05	47	3.45	59	4.33
0.06–0.10	81	5.95	107	7.86
0.11–0.19	168	12.33	218	16.01
0.20–0.29	115	8.44	159	11.67
0.30 and over	101	7.42	164	12.04

[a] Based on average concentration across samples and species-specific estimates at each sampling station.

[b] Number of sampling stations.

10.3 Estimation of Exposed Populations and Fishing Behaviors

Based on the spatial distribution of estimated mercury levels in fish, the next step in the analysis is to estimate the average daily ingestion of mercury (g/day) through noncommercial freshwater fish consumption (HgI) for selected populations of interest. Because the primary measurable health effect of concern—developmental neurological effects in children—occurs as a result of in-utero exposures to mercury, the specific population of interest in this case is prenatally exposed children. To identify and estimate the size of this exposed population, the analysis focuses on women of childbearing age in freshwater angler households in the 37-state study area. More specifically, it estimates the number of pregnant women in these households in 2001 and in other selected years.

Generally speaking, mercury ingestion for a population i in a given year can be calculated as the product of three estimates, as shown in Eq. (10.1):

$$HgI_i = N_i * CHg_i * C_i \qquad \text{(Eq. 10.1)}$$

where

N_i	=	size of the exposed population of interest i (annual number of pregnant women in freshwater angler households during the year),
CHg_i	=	average concentration (µg/g) of methyl mercury in noncommercial freshwater fish filets consumed by population i, and
C_i	=	average daily consumption rate (g/day) of noncommercial freshwater fish by population i.

As described in more detail below, EPA applied two alternative approaches for defining and estimating N_i and CHg_i for the 37-state study area. Both approaches rely primarily on two national-level data sources for characterizing freshwater angler populations and their fishing behaviors. The discussion below begins by describing these two data sources, and then describes how they are used in the two alternative approaches (in combination with data from the Census and other information sources) to estimate N_i and CHg_i. Consumption rate estimates for recreationally caught freshwater (C_i) are based primarily on recommendations in EPA's *Exposure Factors Handbook* (EPA, 1997a) although several other sources were also considered (see Section 10.1.3). These estimates are described in more detail in Section 10.1.4.

10.3.1 Primary Data Sources on Fishing Activity in the United States

Two main sources of national-level activity data are available and suitable for estimating the size and spatial distribution of freshwater angler populations and activities:

- The National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (NSFHWR), maintained by the Department of the Interior (DOI) (DOI and DOC, 1992, 1997, 2002); and
- The National Survey of Recreation and the Environment (NSRE 1994).

NSFHWR Angler Data. The NSFHWR, conducted by the U.S. Census Bureau about every 5 years since 1955, includes data on the number and characteristics of participants as well as time and money spent on hunting, fishing, and wildlife watching. The most recent version, the 2001[9] NSFHWR (DOI and DOC, 2002), collected data on 25,070 respondents in the sportspersons (hunters and anglers) sample and 15,303 in the wildlife watchers sample, all over the age of 15.

The freshwater fishing subset of the sportspersons survey is the most relevant to the mercury analysis. Of the 25,070 respondents included in the 2001 sportspersons sample, 11,280 respondents participated in freshwater fishing. Using the weights provided in the NSFHWR data, it is estimated that 28.4 million Americans participated in freshwater fishing in 2001, spending approximately 467 million total days freshwater fishing.

The survey distinguishes between several types of freshwater fishing. Respondents were asked the number of days spent fishing in the Great Lakes; in other ponds, lakes, or reservoirs; and in rivers or streams. Table 10-4 summarizes the NSFHWR data on freshwater fishing by state. It reports number of anglers and fishing days separately for lakes (including the Great Lakes, ponds, reservoirs, etc.) and rivers (including rivers and streams), and it distinguishes between states as the residence of the anglers and states as the destination for anglers (residents and nonresidents).

Over 80 percent of freshwater anglers in the United States reside in our study area, and they accounted for roughly 88 percent of lake-fishing days and 77 percent of river-fishing days in the country in 2001. The states with the largest numbers of resident freshwater anglers in 2001 were Texas with 1.9 million anglers and Illinois and Minnesota, both with 1.3 million anglers. Minnesota had the highest number of lake-fishing days (including both state residents and nonresidents) in 2001 (25.1 million) followed by Texas and Wisconsin, with 22.6 million and 18.2 million days, respectively. Pennsylvania, New York, and Florida had the largest number of river-fishing days, each with over 6 million days in 2001.

The NSFHWR data also contain information on demographic characteristics of the anglers, including gender, age, and marital status. For example, in the states in our study area, roughly 17 percent of freshwater anglers were women of childbearing age. About 32 percent were married male adults less than 45 years old. These two subpopulations are of interest for our analysis because it is among these groups that we most expect to find either pregnant women who consume their freshwater catch or men who share their catch with pregnant women.

[9] The screening survey for the 2001 NSFHWR covered activities in 2000, but the follow-up and more in-depth surveys focused on 2001.

Table 10-4. Summary of Fishing Activity Levels by State in 2001 from NSFHWR

	Fishing by State Residents			Fishing in Each State (by Residents and Nonresidents)		
State	Number of Freshwater Anglers[a]	Number of Lake-Fishing Days[a,b]	Number of River-Fishing Days[b]	Number of Freshwater Anglers	Number of Lake-Fishing Days[a]	Number of River-Fishing Days
AL	572,604	6,237,724	4,168,160	732,204	6,838,700	4,078,525
AR	546,099	9,805,445	2,279,229	781,772	10,576,329	2,716,241
CT	236,880	2,413,792	1,699,745	254,482	2,320,467	1,404,884
DE	47,496	318,592	284,984	73,147	315,203	329,185
DC	13,921	0	41,764	6,961	0	6,961
FL	1,153,514	12,036,042	5,831,107	1,315,528	12,332,411	6,386,844
GA	953,119	10,286,143	4,332,960	1,016,703	9,761,303	3,525,712
IL	1,302,368	16,091,265	4,976,135	1,125,760	11,866,587	4,105,517
IN	729,603	12,043,601	3,433,917	754,408	10,985,097	2,942,734
IA	511,674	4,891,453	3,978,169	541,613	4,183,232	3,539,145
KS	420,418	5,405,143	978,403	403,691	4,910,572	795,508
KY	609,959	9,774,419	2,439,232	779,677	10,288,090	2,553,575
LA	552,769	5,961,407	2,414,115	659,237	6,117,510	2,523,430
ME	180,475	2,318,947	828,351	271,840	2,774,182	1,005,643
MD	327,679	1,640,229	2,390,203	366,585	1,461,502	2,912,591
MA	335,587	4,496,619	1,458,347	324,740	4,028,656	1,098,457
MI	970,174	15,373,711	2,594,901	1,275,200	16,562,948	2,933,216
MN	1,294,333	23,908,920	2,966,955	1,565,228	25,139,287	2,859,056
MS	409,808	6,832,990	1,820,004	494,165	7,441,168	1,538,864
MO	976,151	9,840,152	2,427,720	1,214,950	10,848,890	2,697,573
NE	264,223	2,726,954	654,359	296,090	2,586,388	656,657
NH	133,489	1,824,024	861,481	220,552	2,004,541	1,036,094
NJ	345,726	4,729,331	2,133,589	330,957	4,290,623	1,476,570
NY	864,957	11,839,200	5,768,677	1,051,982	12,560,010	6,417,990
NC	710,251	9,470,233	4,315,601	847,994	9,163,218	4,346,851
ND	137,839	2,291,206	335,634	178,621	1,936,036	316,964
OH	1,246,262	16,197,914	4,348,070	1,260,043	15,098,754	3,979,234
OK	679,571	11,384,571	2,312,773	774,255	11,193,434	2,225,586
PA	1,008,107	8,856,695	11,274,947	1,182,356	9,124,678	10,664,975
RI	46,446	509,566	184,055	50,733	429,931	222,018
SC	511,862	6,951,092	2,247,640	591,069	6,997,418	2,249,877
SD	144,382	1,807,212	785,679	214,429	2,200,846	1,044,079
TN	763,484	11,159,872	5,100,470	903,385	11,118,719	5,375,402
TX	1,882,755	24,014,324	6,275,434	1,841,749	22,573,278	5,419,444

(continued)

Table 10-4. Summary of Fishing Activity Levels by State in 2001 from NSFHWR (continued)

	Fishing by State Residents			Fishing in Each State (by Residents and Nonresidents)		
State	Number of Freshwater Anglers[a]	Number of Lake-Fishing Days[a,b]	Number of River-Fishing Days[b]	Number of Freshwater Anglers	Number of Lake-Fishing Days[a]	Number of River-Fishing Days
VT	99,564	1,360,820	552,364	171,420	1,409,024	900,343
VA	652,561	8,348,688	3,514,615	721,301	8,237,651	3,545,041
WV	260,343	2,177,141	2,172,734	317,632	1,917,931	2,316,617
WI	948,912	15,763,077	4,077,048	1,349,553	18,295,238	4,506,934
All 50 States plus DC	28,438,814	340,972,513	141,048,761	28,438,814	340,972,513	141,048,761

[a] Includes days fished in other states.
[b] Includes Great Lakes, lakes, ponds, and reservoirs.
Source: U.S. Department of the Interior (DOI), Fish and Wildlife Service and U.S. Department of Commerce, Bureau of the Census. 2002. *2001 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*. Washington, DC: U.S. Government Printing Office.

NSRE Angler Data. The NSRE, formerly known as the National Recreation Survey (NRS), is a nationally administered survey designed to assess outdoor recreation participation in the United States and elicit information regarding people's opinions about their natural environment. The NSRE sample of freshwater anglers is smaller than the NSFHWR sample, but it is nonetheless a useful resource because it provides a wide variety of information about fishing activities. Importantly, it includes relatively detailed information about the nature and location of recent freshwater trips. Because the sampling procedure is designed to be representative, inferences may be drawn as to the relative popularity of particular types of freshwater bodies (e.g., lakes, rivers) among the general public and the average distance traveled to reach these sites. Although most recently conducted in 2000, data from 1994 survey (NSRE 1994) are currently best suited to support this analysis.

NSRE was conducted by the Survey Research Center at the University of Georgia. Surveyors asked 16,000 individuals by telephone about their water-based recreation activities—specifically boating, fishing, swimming, or viewing—during the previous year. The survey elicited information from respondents about *the most recent* trip taken in each of the four categories. Of particular interest for the mercury analysis is the information regarding fishing trip destination for all respondents who fish.

In addition to information about the location of the last fishing trip destinations, the NSRE contains information on the type of waterbody visited on the last trip. Waterbodies are broken down into four categories: lakes, rivers and streams, wetlands, and coastal areas. Of the 3,257 respondents who had fished in the previous year, 1,698 indicated that their last trip was to a lake, 694 to a river or stream, 5 to a wetland, and 720 to a coastal area. (Type of waterbody visited was not available for 141 responses.)

One of the main advantages of NSRE is that it includes geocoded data for reported fishing destinations. To specify the location of the last fishing trip, respondents were asked to provide the name of the waterbody, the nearest town to the waterbody, and an estimate of the distance and direction from their home to the waterbody. Of the 2,520 freshwater destinations reported in NSRE, HUC codes have been identified for 1,768 respondents/trips.

10.1.2.2 Estimation Approaches and Results for Exposed Populations and Fishing Behaviors

To define and estimate exposed populations and their corresponding mercury exposures through freshwater fishing in our study area, EPA has developed two distinct but related approaches:

- The population centroid approach; and
- The angler destination approach.

Using two separate approaches, with respect to specific modeling assumptions, provides useful insights into model uncertainty. These issues are discussed in more detail in Section 10.7.

The two approaches are similar in several respects. Most importantly, both approaches define the main population of interest as prenatally exposed children in freshwater angler households in the 37-state study area. They also both use estimates of mercury fish tissue concentration estimates based on NLFA and NLFTS sampling data (as described in Section 10.2). In addition, the two approaches use state-level data from the NSFHWR as an essential input for estimating the size, characteristics, and behaviors of angler populations in the study area.

However, there are also key differences in the way the two approaches use the mercury sampling data, the NSFHWR, and other supporting data to estimate populations and exposures. The population centroid approach focuses on the residential location of freshwater anglers, the typical distances they travel to fish, and the distribution of mercury concentrations within these travel distances. It uses Census information to define the location and demographic characteristics of potentially exposed populations and uses data from the NSFHWR to estimate the fraction of freshwater anglers with respect to each population and location. The second approach—the angler destination approach—focuses on the fishing destination of anglers and the distribution of mercury concentrations across these destinations. It defines destinations according to standard watershed codes (eight-digit HUC) and uses information from both the NSFHWR and NSRE to estimate levels of angler activity and fish consumption on a watershed-by-watershed basis.

Below we describe each of the two approaches in more detail. We also summarize and compare results from the two approaches. Key elements associated with each modeling approach are presented in Table 10-5.

Table 10-5. Overview of Key Attributes of the Population Centroid and Angler Destination Models

Population Centroid Model	Angler Destination Model
• Push" model based on first estimating the number of recreational fisher within each US Census block group and then predicting fishing activity in the form of trip travel distances out to different fishing trip travel rings (10, 20, 50 and 100 mile rings). • Fish tissue samples are averaged within each ring to provide exposure levels for fishers assigned to a particular ring (averaged tissue samples are also differentiated for rivers versus lakes, and fishing activity within each ring is separated into river- versus lake-activity). • Fishing activity (i.e., trip travel distances) are also differentiated according to income level and urban/rural status of modeled recreational fishers. • Model generates estimates of recreational fisher exposure generated for 165,000 block groups (significantly more spatial differentiation compared with the angler destination approach) • Total modeled population: 434,000 prenatally-exposed infants in 2001. • Model used to support the per-capita distributional IQ impact analysis, due to its greater spatial differentiation and use of more detailed demographic data relative to the angler destination model.	• "Pull" model based on first determining the relative attractiveness of individual watersheds for recreational fishing activity (this model does not consider the residential location of fishers but instead, focuses on modeling their fishing locations). Fishing activity modeled at the 8-digit hydrologic unit code (HUC) watershed level (these HUCs are 1,600 square miles on average, which is significantly larger than the block groups used in the population centroid model). • Predictive model for fishing activity in individual HUCs consider range of factors related to fishing activity (e.g., population density in vicinity of HUC, HUC surface area, total number of lake boundary and river miles within the HUC). • Model generates separate recreational fisher exposure estimates 1,360 HUCs (less than 1/10th of the number of block groups modeled). • Total modeled population: 587,000 prenatally-exposed infants in 2001. Note, this estimate is larger than the population centroid's modeled population because the angler destination model estimates exposures for pregnant women as the combined effect of (1) their own fishing activity and (2) adult males bringing home and sharing fish they catch.

10.3.2 Population Centroid Approach.

This approach uses Census block groups to spatially separate and characterize potentially affected populations in the study area. In the Census, block groups are generally defined to contain between 600 and 3,000 people. A total of 503 block groups were identified in the study area, and five distance intervals were mapped from the centroid of each block group. Then, freshwater fishing trips and fish consumption are allocated to anglers in each group according to data on typical travel distances for freshwater fishing in the United States. The flow diagram in Figure 10-3 illustrates the main components of the approach, spatial scale of the data used to estimate these components, and how these components are interrelated. For each selected block group, the following steps were applied.

First, Census data were used to define the size, age, gender distribution, and income of the population within each of the roughly 165,000 block groups in the study area.

Second, the size of the exposed population of interest (annual number of prenatally exposed children in freshwater angler households) in each block group was estimated by combining Census, Vital Statistics, and NSFHWR data. For each block group, this estimation required:

1. Estimating the number of pregnant women (NP) living in the block group as

$$NP = NF * f_s \qquad \text{(Eq. 10.2)}$$

where

NF = number of females aged 15 to 44 in (Census 2000) and
f_s = state-level general fertility rate (average number of live births in an year per 1,000 women aged 15 to 44) (Hamilton, Sutton, and Ventura, 2003).

2. Estimating the annual number of prenatally exposed children in angler households (NPA) as

$$NPA = NP*(NA_s/N_s) \qquad \text{(Eq. 10.3)}$$

where

NA_s = state-level number of 15 years and older angler residents (NSFHWR) and
N_s = 15 years and older adult population of states (Census).

Eq. (10.3) reflects an estimate of the number of pregnant women in each state who reside in freshwater angler households (not necessarily the number who are freshwater anglers themselves). To estimate this value (NPA), Eq. (10.3) implies that (1) the fraction of pregnant women in a state who are in freshwater angler households is equal to the fraction of households in the state that include freshwater anglers (i.e., pregnant women are no more or less likely than the rest of the state population to live in households with freshwater anglers) and (2) the fraction of *households* in the state that include freshwater anglers is equal to the fraction of adult *residents* in the state who are freshwater anglers. The implications of these assumptions for the results of the analysis are discussed in more detail in Section 10.7.

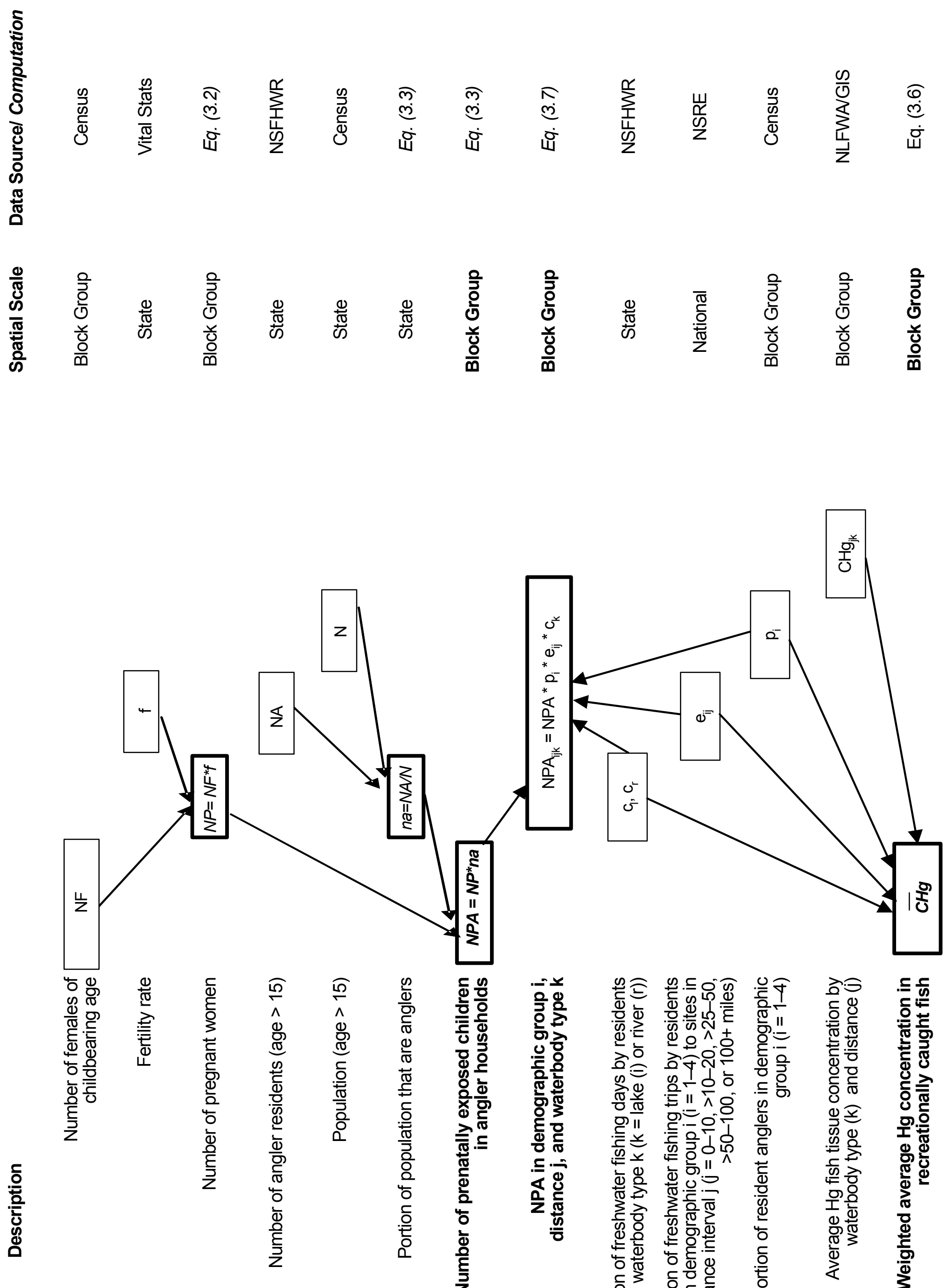

Figure 10-3. Flow Diagram for Population Centroid Approach

To estimate NPA for years after 2001, it was assumed that state-level fertility rates (f_s) and angler participation rates (NA_s/N_s) would remain constant; however, the number of women of childbearing age in each block (NF) was increased based on county-level population growth projections (Woods and Poole, 2001). In other words, for the period 2001-2025, the estimated NPA for each block group was assumed to increase at the same rate as the projected annual population growth rates for females 15-44 in their corresponding counties. Woods and Poole (2001) provide population projections only for years up to 2025. For years beyond 2025, it was assumed that NPA for each block group increase linearly at the same rate as the annual population growth rate for NF in corresponding counties from 2024 to 2025. As discussed in more detail in Section 10.1.4, estimates of exposed populations were developed for future years to account for lagged effects between reductions in mercury emissions and reductions in mercury concentrations in fish. Uncertainties resulting from these assumptions are discussed in more detail in Section 10.7.

Third, average mercury concentrations in freshwater fish for angler households were separately estimated for each block group. To estimate these averages, the available mercury concentration data were separated according to (1) distance from each block group centroid (separating them into five distance categories) and (2) waterbody type (lake or river). As shown in Figure 10-4, a separate average mercury concentration was estimated for each waterbody type and distance interval. These estimates were then applied to calculate a weighted average of these mercury concentration estimates for each block group. To weight these estimates it was assumed that the percentage of fish consumed from each distance and waterbody category is equivalent to the percentage of fishing trips to each distance and waterbody category. The procedure used to estimate the distribution of trips across waterbody types and distance categories for each block group is described below.

To approximate the percentage freshwater fishing trips from each block group to each waterbody type (c_l or c_r), state-level averages were used. These averages were calculated for each state, based on the portion of residents' freshwater angler days that are to each waterbody type. In other words, these portions were calculated as

$$c_l = D_{ls}/(D_{ls} + D_{rs}) \qquad \text{(Eq. 10.4)}$$

$$c_r = (1 - c_l) \qquad \text{(Eq. 10.5)}$$

where

D_{ls} = number of lake-fishing days by state resident anglers (DOI, 2002) and

D_{rs} = number of river-fishing days by state resident anglers (DOI, 2002).

These lake- and river-fishing day estimates from the NSFHWR are also summarized in Table 10-4.

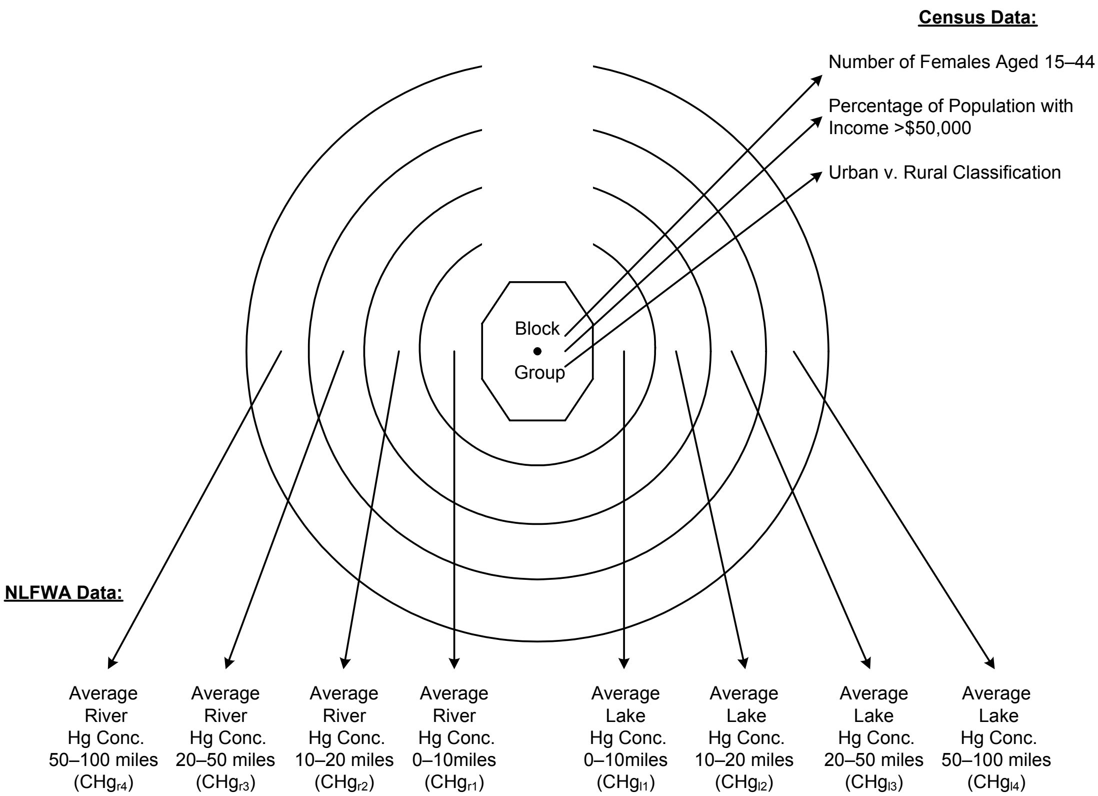

Figure 10-4. Population Centroid Approach: Linking Census Block Groups to Demographic Data and Mercury Fish Tissue Samples

Data from NSRE 1994 were used to approximate the percentage of freshwater fishing trips to different distances from anglers' residential location. Five distance intervals were defined as 0–10 miles, >10–20 miles, >20–50 miles, >50–100 miles, and 100+ miles. Based on self-reported trip distance information from nearly 2,000 respondents (see Appendix E-1 for details), each of these distance categories was associated with roughly 20 percent of the reported trips in the NSRE sample. Four distinct demographic groups were also found to have significantly different average travel distances for freshwater fishing in the NSRE sample: urban with above $50,000 income, rural with above $50,000 income, urban with below $50,000 income, and rural with below $50,000 income.[10] The portion of trips for each demographic group ($i = 1 – 4$) to each distance interval ($j = 1 – 5$) are defined as e_{ij}. The estimated values for e_{ij} are reported in Appendix E-3. Uncertainties associated with this trip travel distance analysis are discussed in Section 10.7.

To estimate the portion of households in each demographic group (p_i for $i = 1 – 4$), each block group was categorized as either urban (including suburban) or rural. Census data on the percentage of the population in each block group with household income less than $50,000 were used to define the portion in the below $50,000 income groups.

[10] An annual household income of $50,000 (in 2000 dollars) is close to the median value for both the NSRE sample and the U.S. population.

To estimate a weighted average mercury concentration for each block group (CHg), the average mercury concentration in freshwater fish for each combination of waterbody (i.e, lake or river) and distance interval (i.e., 10 = 2 x 5 combinations) was first estimated. This average concentration is defined as CHg_{kj} for k = lake or river and j = 1 – 5 distance interval.[11] For each block group, this approach assumes that the sample averages of CHg (from available NLFA and NLFTS sampling data) for each distance–waterbody combination provide an reasonable estimate of the true average mercury concentrations in the distance–waterbody combination.

The weighted average mercury concentration in freshwater fish for each block group was then calculated as

$$\overline{CH}g = \Sigma_i p_i * \left[\Sigma_{jk} (e_{ij}) * (c_k) * (CHg_{jk}) \right] \qquad \text{(Eq. 10.6)}$$

for

i	=	1 – 4 demographic group,
j	=	1 – 5 distance interval,
k	=	lake or river, and
p_i	=	percentage of block group households in demographic group i (Census).

To match exposed populations with corresponding average mercury concentrations in fish, one method is to match the total NPA (annual number of prenatally exposed children in angler households) from each block group with the corresponding weighted average mercury concentration (CHg) for that block group. This method is equivalent to assuming that all exposed individuals in the same block group and demographic group (i) are exposed to the same average levels of mercury. This would occur, for example, if they all allocate trips to (or fish consumption from) the different distance intervals and waterbody types according to the same proportions, e_{ij} and c_k.

An alternative method for matching populations with mercury concentrations using the population centroid approach is to assume that:

- Each exposed individual in a block group is associated with freshwater fishing in a single distance interval and a single waterbody type (i.e., all the fish they consume comes from the same distance and type of waterbody);

- The exposed populations in each block group (rather than just the fishing trips) are distributed across the distance intervals and waterbody types according to the estimated proportions e_{ij} and c_k.

In this case, as many as 20 separate exposed subpopulations can be defined for each block group:

$$NPA_{ijk} = NPA * p_i * e_{ij} * c_k \text{ (for all i, j, and k.)} \qquad \text{(Eq. 10.7)}$$

[11] For the 100+ miles category (j = 5) the mean mercury concentration for lake and river samples in the entire 37-state study area (see Table 10-4) was used to define CHg_{5l} and CHg_{5r} respectively.

Each subpopulation NPA_{ijk} can then be separately matched with the block group's average mercury concentration for the corresponding distance and waterbody category (CHg_{jk}).

Using the model described in Section 10.1.4 (below) to estimate mercury ingestion rates, these two methods will produce the same estimates of total mercury ingestion for a block group NPA. However, they will differ according to how individual ingestion rates are distributed within the exposed population. The first method assumes constant ingestion rates across a block group NPA, whereas the second method estimates separate ingestion rates for each subpopulation (NPA_{ijk}) in the block group.

Modeling separate exposure levels for each subpopulation (NPA_{ijk}) within each block group is critical to characterizing the per-capita distribution of IQ changes within the study population and does provide the basis for that assessment (along with consideration for variability in fish consumption rates). However, providing this level of differentiation in modeling individual exposure is not necessary for generating a mean (best estimate) IQ loss and associated monetary value for the entire study population.

Summary of Results. This section summarizes some of the key results of applying the steps outlined above. Table 10-6 reports state-level summaries of the Census block group data used in the analysis. Nearly 165,000 block groups were identified in the 37-state study area. The average number of women of childbearing age (16 to 44) per block group in 2000 was 281. On average, there were approximately 60 percent of households with incomes above $50,000 and approximately 75 percent of block groups that were predominantly urban (or suburban).

Table 10-7 reports estimates of the annual number of prenatally exposed children in selected years from 2001 to 2051 aggregated to the state level.[12] For reasons discussed in more detail below, the selected years correspond to lag periods of 5, 10, 20, and 50 years. We also provide the estimate of prenatally exposed children in a base year of analysis (2001) for comparison purposes. These estimates are derived from the Census demographic data reported in Table 10-6 combined with state-level fertility rates, freshwater fishing participation rates, and county-level population growth projections. The size of the exposed population of interest in 2001 was estimated to average over 11,000 individuals per state in the study area, with a total of over 434,000 in the entire area.

Table 10-8 estimates the annual number of prenatally exposed children in selected years and lag periods starting in 2020. We also provide the estimate of prenatally exposed children in the base year (2020) for comparison purposes. In 2020, the size of the exposed population of interest was estimated to be 482,000.

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[12] Although the results are reported at study area and the state levels, the analysis is conducted at the individual block group level.

Table 10-6. Block Group Demographic Characteristics by State (in 2000): Data Used in Population Centroid Approach

State	Number of Block Groups	Percentage Urban Block Groups[a]	Female Population, Aged 16–44, in Block Group			Percentage of Block Group Households in Above $50,000 Income Category		
			Min	Mean	Max	Min	Mean	Max
AL	3,329	55	0	283	2,591	0	71	100
AR	2,135	54	0	255	1,752	0	73	100
CT	2,620	88	0	269	3,718	0	46	100
DC	433	99	0	327	2,243	0	57	98
DE	502	83	0	337	2,899	0	53	94
FL	9,112	87	0	344	6,355	0	64	100
GA	4,788	69	0	386	4,239	0	63	100
IA	2,634	58	0	224	2,618	0	64	100
IL	9,843	85	0	271	3,960	0	55	100
IN	4,798	69	0	268	3,559	0	62	100
KS	2,299	69	0	242	2,773	0	63	100
KY	3,157	53	0	273	3,207	0	70	100
LA	3,509	73	0	278	2,962	0	70	100
MA	5,053	92	0	274	2,678	0	50	100
MD	3,678	84	0	315	4,212	0	48	100
ME	1,143	38	0	226	960	0	65	100
MI	8,450	74	0	247	3,452	0	56	100
MN	4,082	68	0	257	1,882	0	54	100
MO	4,540	67	0	257	2,004	0	67	100
MS	2,148	51	0	286	2866	0	73	100
NC	5,271	58	0	329	5,413	0	65	100
ND	630	45	0	207	2,114	0	70	100
NE	1,591	69	0	222	2,111	0	62	100
NH	874	59	0	297	2,000	0	52	100
NJ	6,510	94	0	271	3,473	0	45	100
NY	15,079	85	0	272	4,840	0	55	100
OH	9,354	78	0	253	3,941	0	62	100
OK	2,901	66	0	245	2,154	0	71	100
PA	10,387	78	0	239	2,659	0	63	100
RI	821	92	0	276	2,348	0	58	100
SC	2,859	61	0	301	3,764	0	67	100
SD	688	45	0	221	1,622	0	70	100
TN	4,014	64	0	303	3,181	0	68	100
TX	14,463	81	0	318	4,936	0	63	100
VA	4,749	70	0	326	3,535	0	55	100
VT	530	36	28	238	1,534	15.05	62	98.13
WI	4,388	69	0	255	2,758	0	58	100
WV	1,588	46	0	228	1,954	0	75	100
Study Area	164,950	75	0	281	6,355	0	60	100

[a] Urban designation was assigned to a block group if 50 percent or greater of its population was categorized as urban.

Source: 2000 Census.

Table 10-7. Estimated Annual Number of Prenatally Exposed Children for Selected Lag Periods from 2001: Population Centroid Approach

	Base Year of Comparison (2001)			Central Estimate of Fish Tissue Response Times						Alternative Estimate of Fish Tissue Response Times					
				10 Year Lag (2011)			20 Year Lag (2021)			5 Year Lag (2006)			50 Year Lag (2051)		
	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population
	Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.	
Study Area	2.63	2.51	434,059	2.75	2.80	452,575	2.95	3.22	486,487	2.69	2.64	442,938	3.84	4.83	632,017
State															
AL	3.0	2.3	9,948	3.0	2.4	9,869	3.0	2.5	9,906	3.0	2.3	9,907	3.4	3.5	11,230
AR	4.6	2.9	9,832	4.8	3.2	10,242	5.1	3.7	10,968	4.7	3.0	10,033	6.8	5.9	14,557
CT	1.6	1.0	4,134	1.6	1.0	4,176	1.7	1.1	4,491	1.6	1.0	4,152	2.4	1.5	6,229
DC	0.5	0.5	236	0.5	0.4	224	0.5	0.4	211	0.5	0.4	228	0.7	0.5	286
DE	1.7	1.3	836	1.7	1.3	854	1.8	1.3	901	1.7	1.3	850	2.2	1.6	1,109
FL	2.3	2.5	21,218	2.6	2.9	23,903	3.0	3.4	27,786	2.5	2.7	22,532	4.6	5.4	41,941
GA	4.2	3.9	20,255	4.6	4.7	22,252	5.3	5.8	25,175	4.4	4.3	21,306	7.8	10.6	37,558
IA	3.2	2.5	8,448	3.1	2.6	8,249	3.2	2.8	8,339	3.2	2.6	8,326	3.4	3.6	9,054
IL	2.8	2.5	27,670	2.9	2.7	28,470	3.1	2.9	30,205	2.8	2.6	27,963	3.8	4.1	37,881
IN	2.8	2.4	13,640	2.9	2.6	14,027	3.1	2.8	14,783	2.9	2.5	13,799	3.8	3.9	18,037
KS	3.6	2.5	8,246	3.8	2.8	8,606	4.1	3.2	9,294	3.7	2.7	8,410	5.4	5.1	12,289
KY	3.3	2.3	10,322	3.2	2.3	10,139	3.3	2.5	10,347	3.2	2.3	10,211	3.8	3.4	12,041
LA	3.1	2.1	10,709	3.0	2.1	10,582	3.1	2.3	10,892	3.0	2.1	10,580	3.3	2.9	11,677
MA	1.1	0.7	5,663	1.1	0.7	5,671	1.2	0.8	5,928	1.1	0.7	5,686	1.6	1.0	7,846
MD	1.6	1.2	5,881	1.7	1.3	6,084	1.8	1.5	6,610	1.6	1.3	6,017	2.4	2.2	8,742
ME	2.1	1.2	2,378	1.9	1.1	2,223	1.9	1.1	2,139	2.0	1.2	2,324	1.9	1.1	2,124
MI	2.0	1.4	17,028	2.0	1.5	17,185	2.1	1.6	17,672	2.0	1.4	17,186	2.5	2.1	20,816
MN	5.6	3.8	22,806	5.8	4.3	23,777	6.4	5.2	25,952	5.7	4.1	23,286	8.1	7.6	32,953
MO	3.7	2.7	16,804	3.7	2.8	17,015	3.9	3.2	17,924	3.7	2.8	16,906	4.9	4.5	22,289

(continued)

Table 10-7. Estimated Annual Number of Prenatally Exposed Children for Selected Lag Periods from 2001: Population Centroid Approach (continued)

				Central Estimate of Fish Tissue Response Times						Alternative Estimate of Fish Tissue Response Times					
	Base Year of Comparison (2001)			10 Year Lag (2011)			20 Year Lag (2021)			5 Year Lag (2006)			50 Year Lag (2051)		
	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population
	Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.	
State															
MO	3.7	2.7	16,804	3.7	2.8	17,015	3.9	3.2	17,924	3.7	2.8	16,906	4.9	4.5	22,289
MS	3.7	2.3	7,909	3.7	2.5	7,920	3.7	2.7	8,008	3.7	2.4	7,886	4.3	3.8	9,208
NC	2.6	2.1	13,483	2.8	2.5	14,702	3.1	3.0	16,534	2.7	2.3	14,066	4.8	4.9	25,075
ND	3.4	3.1	2,101	3.2	3.0	2,016	3.2	3.1	2,004	3.3	3.0	2,050	3.4	3.2	2,138
NE	3.2	1.9	5,138	3.4	2.1	5,393	3.7	2.4	5,849	3.3	2.0	5,256	4.8	3.2	7,635
NH	2.2	1.6	1,966	2.3	1.6	1,991	2.3	1.7	2,045	2.3	1.6	2,004	3.0	2.3	2,625
NJ	1.1	0.7	6,838	1.1	0.8	7,024	1.2	0.9	7,636	1.1	0.8	6,927	1.6	1.2	10,130
NY	1.1	0.9	16,877	1.1	1.0	16,994	1.2	1.0	17,663	1.1	1.0	16,937	1.4	1.3	21,400
OH	2.3	1.7	21,457	2.3	1.7	21,317	2.3	1.8	21,659	2.3	1.7	21,356	2.7	2.4	25,190
OK	4.5	3.0	13,117	4.6	3.1	13,298	4.9	3.4	14,281	4.5	3.1	13,126	6.1	4.5	17,713
PA	1.5	0.9	15,146	1.4	1.0	14,756	1.4	1.0	14,702	1.4	1.0	14,965	1.6	1.4	16,862
RI	0.9	0.6	755	1.0	0.6	785	1.0	0.6	815	0.9	0.6	773	1.3	0.9	1,066
SC	3.2	2.6	9,010	3.3	2.7	9,327	3.4	3.0	9,716	3.2	2.7	9,222	4.2	3.9	12,031
SD	3.7	3.1	2,508	3.7	3.2	2,502	3.9	3.5	2,590	3.7	3.1	2,500	4.4	4.3	2,941
TN	3.3	2.6	13,317	3.4	2.8	13,663	3.6	3.2	14,511	3.4	2.7	13,510	4.8	4.8	19,133
TX	3.9	3.4	55,802	4.5	4.1	64,381	5.2	4.9	74,645	4.2	3.8	59,828	7.6	7.6	108,932
VA	2.5	2.0	11,793	2.6	2.3	12,440	2.8	2.6	13,471	2.6	2.1	12,158	3.8	4.1	17,946
VT	2.4	1.5	1,288	2.3	1.5	1,223	2.2	1.5	1,161	2.4	1.5	1,270	2.3	1.7	1,242
WI	3.6	2.4	15,848	3.6	2.5	15,839	3.7	2.6	16,427	3.6	2.5	15,884	4.3	3.3	18,932
WV	2.3	1.3	3,651	2.2	1.3	3,460	2.0	1.3	3,248	2.2	1.3	3,518	2.0	1.6	3,160

Table 10-8. Estimated Annual Number of Prenatally Exposed Children for Selected Lag Periods from 2020: Population Centroid Approach

	Base Year of Comparison (2020)			Central Estimate of Fish Tissue Response Times: 10 Year Lag (2030)			Central Estimate of Fish Tissue Response Times: 20 Year Lag (2040)			Alternative Estimate of Fish Tissue Response Times: 5 Year Lag (2025)			Alternative Estimate of Fish Tissue Response Times: 50 Year Lag (2070)		
	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population
	Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.	
Study Area	2.92	3.17	481,987	3.21	3.67	528,721	3.51	4.22	577,910	3.06	3.41	504,127	4.40	5.93	725,474
State															
AL	3.0	2.5	9,871	3.1	2.8	10,275	3.2	3.1	10,730	3.0	2.6	10,048	3.6	4.2	12,093
AR	5.1	3.6	10,877	5.6	4.3	11,978	6.2	5.1	13,206	5.3	4.0	11,364	7.9	7.5	16,891
CT	1.7	1.1	4,437	1.9	1.2	5,005	2.1	1.4	5,588	1.8	1.2	4,714	2.8	1.8	7,335
DC	0.5	0.4	211	0.5	0.4	227	0.6	0.5	255	0.5	0.4	213	0.8	0.7	339
DE	1.8	1.3	894	1.9	1.4	966	2.1	1.5	1,034	1.9	1.4	932	2.5	1.8	1,239
FL	3.0	3.3	27,339	3.5	4.0	31,950	4.0	4.6	36,708	3.2	3.7	29,572	5.6	6.7	50,980
GA	5.2	5.7	24,796	6.0	7.2	28,805	6.9	8.8	32,973	5.6	6.4	26,721	9.5	13.7	45,477
IA	3.2	2.8	8,325	3.2	3.0	8,551	3.3	3.3	8,791	3.2	2.9	8,431	3.6	4.1	9,510
IL	3.0	2.9	29,953	3.3	3.3	32,470	3.6	3.7	35,047	3.2	3.1	31,181	4.3	4.8	42,778
IN	3.1	2.8	14,665	3.3	3.2	15,769	3.5	3.5	16,849	3.2	3.0	15,229	4.2	4.7	20,089
KS	4.1	3.2	9,222	4.4	3.7	10,073	4.9	4.4	11,128	4.2	3.4	9,545	6.3	6.5	14,294
KY	3.3	2.5	10,303	3.4	2.7	10,808	3.6	3.0	11,395	3.3	2.6	10,515	4.2	4.0	13,156
LA	3.1	2.3	10,857	3.2	2.5	11,120	3.2	2.7	11,385	3.1	2.4	10,987	3.5	3.3	12,182
MA	1.2	0.7	5,867	1.3	0.8	6,499	1.4	0.9	7,140	1.2	0.8	6,178	1.8	1.2	9,064
MD	1.8	1.5	6,533	2.0	1.7	7,253	2.2	2.0	7,962	1.9	1.6	6,898	2.7	2.7	10,089
ME	1.9	1.1	2,137	1.9	1.1	2,133	1.9	1.1	2,129	1.9	1.1	2,136	1.9	1.1	2,115
MI	2.1	1.6	17,539	2.2	1.7	18,610	2.3	1.9	19,660	2.1	1.6	18,085	2.7	2.4	22,811
MN	6.3	5.1	25,696	6.9	5.9	28,010	7.4	6.7	30,364	6.6	5.5	26,833	9.2	9.2	37,425
MO	3.9	3.1	17,805	4.2	3.5	19,123	4.5	4.0	20,630	4.0	3.3	18,369	5.5	5.4	25,154

(continued)

Table 10-8. Estimated Annual Number of Prenatally Exposed Children for Selected Lag Periods from 2020: Population Centroid Approach (continued)

				Central Estimate of Fish Tissue Response Times						Alternative Estimate of Fish Tissue Response Times					
	Base Year of Comparison (2020)			10 Year Lag (2030)			20 Year Lag (2040)			5 Year Lag (2025)			50 Year Lag (2070)		
	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population	Per Block Group		Total Exposed Population
	Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.		Mean	S.D.	
State															
MS	3.7	2.7	7,974	3.9	3.0	8,341	4.1	3.4	8,754	3.8	2.9	8,135	4.7	4.6	9,992
NC	3.1	2.9	16,264	3.6	3.5	19,020	4.2	4.2	21,903	3.3	3.2	17,578	5.8	6.2	30,553
ND	3.2	3.1	2,005	3.2	3.1	2,022	3.3	3.1	2,077	3.2	3.1	1,994	3.6	3.5	2,244
NE	3.7	2.4	5,829	4.0	2.6	6,347	4.4	2.9	6,960	3.8	2.5	6,041	5.5	3.9	8,800
NH	2.3	1.7	2,023	2.5	1.8	2,211	2.8	2.0	2,408	2.4	1.7	2,112	3.4	2.6	3,000
NJ	1.2	0.9	7,554	1.3	1.0	8,378	1.4	1.1	9,212	1.2	0.9	7,961	1.8	1.5	11,714
NY	1.2	1.0	17,537	1.2	1.1	18,769	1.3	1.2	20,022	1.2	1.1	18,143	1.6	1.5	23,779
OH	2.3	1.8	21,660	2.4	2.0	22,715	2.6	2.2	23,893	2.4	1.9	22,126	2.9	2.7	27,429
OK	4.9	3.4	14,132	5.2	3.7	15,182	5.6	4.1	16,387	5.0	3.5	14,579	6.9	5.2	20,003
PA	1.4	1.0	14,621	1.5	1.1	15,340	1.5	1.3	16,065	1.4	1.1	14,978	1.8	1.7	18,239
RI	1.0	0.6	807	1.1	0.7	889	1.2	0.8	974	1.0	0.7	847	1.5	1.0	1,227
SC	3.4	2.9	9,634	3.6	3.2	10,399	3.9	3.5	11,176	3.5	3.1	10,010	4.7	4.5	13,508
SD	3.9	3.5	2,587	4.0	3.7	2,669	4.2	4.0	2,799	3.9	3.6	2,605	4.8	4.9	3,187
TN	3.6	3.2	14,372	3.9	3.7	15,848	4.3	4.2	17,412	3.8	3.4	15,065	5.5	5.9	22,105
TX	5.1	4.8	73,600	5.9	5.7	84,608	6.7	6.6	96,191	5.5	5.2	78,817	9.1	9.3	130,939
VA	2.8	2.6	13,318	3.1	3.1	14,795	3.4	3.6	16,295	3.0	2.8	14,044	4.4	5.1	20,798
VT	2.2	1.5	1,156	2.2	1.5	1,185	2.3	1.6	1,212	2.2	1.5	1,172	2.4	1.9	1,293
WI	3.7	2.6	16,324	3.9	2.8	17,182	4.1	3.0	18,015	3.8	2.7	16,766	4.7	3.7	20,515
WV	2.1	1.3	3,261	2.0	1.4	3,194	2.0	1.5	3,178	2.0	1.3	3,202	2.0	1.8	3,129

Table 10-9 summarizes the mercury concentration data for freshwater fish in each of the distance and waterbody type categories.[13] Some of the key results from this table are the following:

- A relatively small percentage of block groups have lake samples within 0–10 miles (26 percent) and >10–20 miles (49 percent);
- A larger percentage of block groups have river samples within 0–10 miles (52 percent) and >10–20 miles (69 percent);
- Almost 94 percent of block groups have at least one lake sample within 100 miles;
- Over 99 percent of block groups have at least one river sample within 100 miles; and
- For all samples within 100 miles, the mean (median) concentration in lake samples is 0.21 ppm (0.18 ppm) and in river samples is 0.23 ppm (0.2 ppm).

Table 10-9. Average Estimated Mercury Concentrations (ppm) in Freshwater Fish by Distance Interval from Block Group Centroids: Base Case 2001

Distance from Centroid	N[a]	Min	Mean	Max	Median
Lake Sampling Sites					
0–10 miles	44,327	0.0036	0.209	2.640	0.150
>10–20 miles	80,524	0.0036	0.204	2.640	0.164
>20–50 miles	147,696	0.0063	0.201	2.225	0.177
>50–100 miles	161,837	0.0063	0.229	1.054	0.193
0–100 miles	163,402	0.0036	0.213	2.640	0.180
River Sampling Sites					
0–10 miles	86,269	0.0004	0.217	3.300	0.165
>10–20 miles	114,505	0.0004	0.221	3.300	0.184
>20–50 miles	157,663	0.0004	0.224	3.190	0.203
>50–100 miles	163,311	0.0335	0.232	1.022	0.225
0–100 miles	164,084	0.0004	0.225	3.300	0.202

[a]Number of block groups (out of 164,950) with at least one sample in the distance interval.

[13] Note that each of the sampling site mercury concentration estimates reported in Table 10-4 is included multiple times in the estimates reported in Table 10-9, because they are all located within 100 miles of multiple block groups.

10.3.3 Angler Destination Approach

Rather than focusing on the point of origin (i.e., residential location) of freshwater anglers, as is done in the population centroid approach, the angler destination approach focuses on recreational fishing behavior and determines the areas that are more likely to be fished, using information on inland watersheds (defined by HUCs).[14] The advantage of the NSFHWR, as described above, is that it is based on a relatively large number of observations and, therefore, can provide reliable estimates of angler activities and demographics on a state level. However, it does not provide information at a finer level of spatial resolution than the state. In contrast, the NSRE provides a much smaller sample on angler activities and demographics, but it also provides more detailed information on the destination (i.e., HUC) of fishing trips.

The flow diagram in Figure 10-5 illustrates the main components of this approach and how they are interrelated including the spatial scale of the data used for various model components. To implement the angler destination approach, the following steps were applied.

First, for each state, data from the NSFHWR were used to provide the total annual number of lake- and river-fishing days in the state (D_{ls} and D_{rs}) by both residents and nonresidents in 2001 (see Table 10-4).

Second, data from the NSRE were used to estimate the portion of these state-level fishing days that occur in each HUC within the state. The details of this analysis of the NSRE data are provided in Appendix E-2. To summarize, the following process was applied:

1. Counted the number of lake and river trips in NSRE 1994 to each HUC in the United States (T_{lh}, T_{rh}), and used these counts as indicators of the "level of use" for each HUC.

2. Used a negative binomial regression model to estimate the HUC-level determinants of the level-of-use counts (determinants include the number of lake or river miles in the HUC, population within 50 miles of the HUC, size of the HUC, etc). In a simplified form, the regression model assumes that the probability of observing the value t_h for T_{lh} (or T_{rh}) can be expressed as:

$$\Pr(T_{ih} = t_h) = \frac{\exp(-\exp(\beta_i X_h))(\exp(\beta_i X_h))^{t_h}}{t_h!} \quad \text{for i = l,r} \qquad \text{(Eq. 10.8)}$$

[14] A site-choice random utility model (RUM) approach was also considered for modeling fishing behavior, but the data and analytical requirements of this alternative approach were determined to be beyond the scope of this analysis.

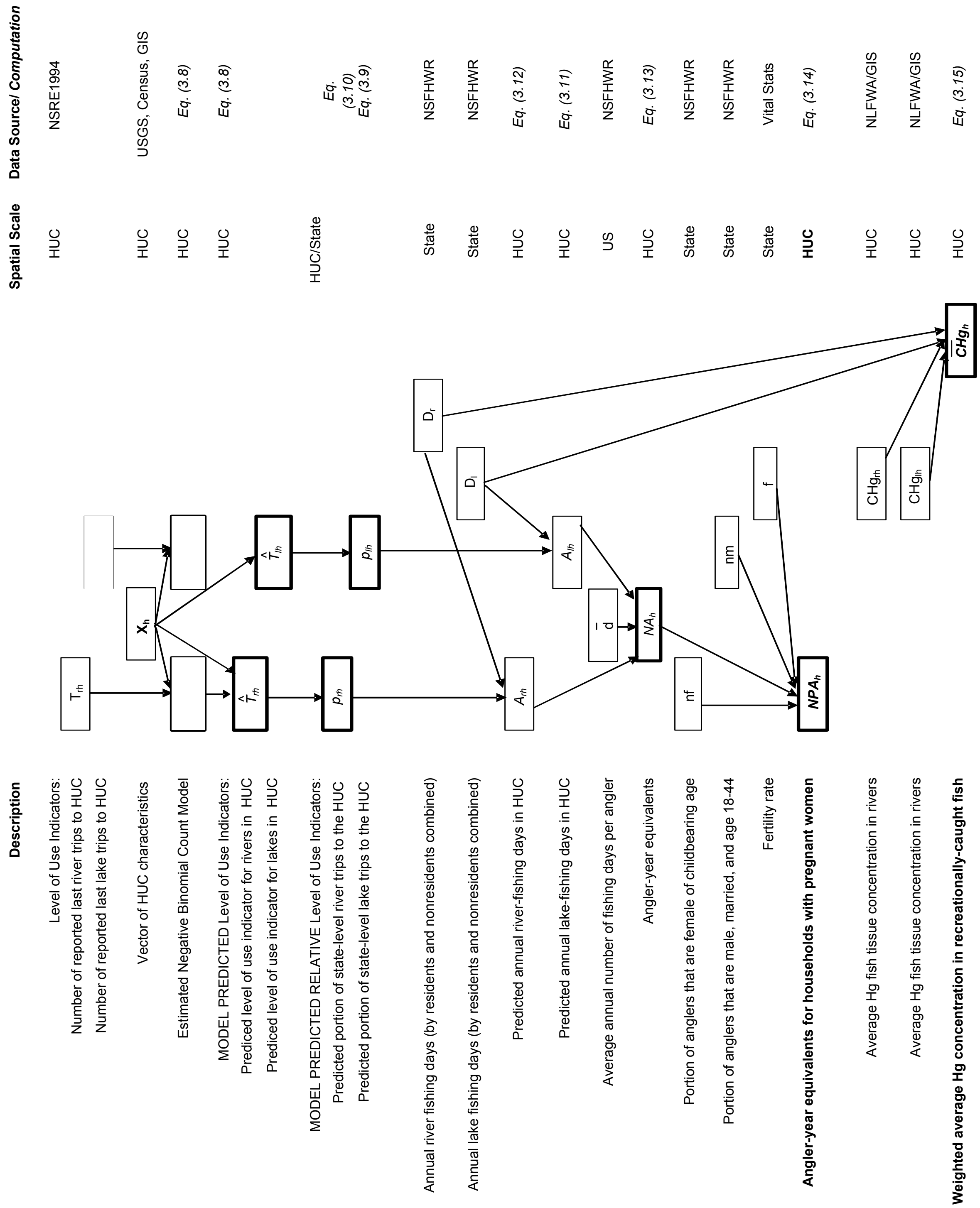

Figure 10-5. Flow Diagram for Angler Destination Approach

where

X_h represents a vector of HUC-level characteristics and β_i represents a corresponding vector of coefficients. These coefficient vectors were estimated using maximum likelihood methods and are represented as $\hat{\beta}_l$ and $\hat{\beta}_r$.

3. Used the results of the estimated econometric model to predict a level-of-use indicator for each HUC ($\hat{T}_{rh}$, $\hat{T}_{rh}$), based on observed HUC-level characteristics.
4. For each state, approximate the percentage of state-level lake- and river-fishing days that occur in each HUC as:

$$p_{lh} = \frac{\hat{T}_{lh}}{\sum_h \hat{T}_{lh}} \quad \text{(Eq. 10.9)}$$

$$p_{rh} = \frac{\hat{T}_{rh}}{\sum_h \hat{T}_{rh}} \quad \text{(Eq. 10.10)}$$

Third, fishing days in each HUC (h) were estimated for each waterbody type (A_{lh} and A_{rh}) by combining the state-level fishing day totals from the NSFHWR (Step 1) with the HUC-level portions estimates (Step 2):

$$A_{lh} = p_{lhs} * D_{ls} \quad \text{(Eq. 10.11)}$$

$$A_{rh} = p_{rhs} * D_{rs} \quad \text{(Eq. 10.12)}$$

Fourth, "angler-year equivalents" were estimated for each HUC (N_h), using the nationwide estimate from the NSFWHR of the average number of fishing days per year per angler ($\bar{d}$=16.42 days/yr). In other words, each 16.42 estimated angler days in HUC h in 2001 were treated as the equivalent of one angler-year to the HUC in 2001.

$$NA_h = (A_{lh} + A_{rh}) / \bar{d} \quad \text{(Eq. 10.13)}$$

Fifth, the number of these angler-year equivalents that are specifically for pregnant women were approximated as

$$NPA_h = NA_h * (nf_s + nm_s) * f_s \quad \text{(Eq. 10.14)}$$

where

nf_s = percentage of anglers fishing in state s that are female and of childbearing age (NSFHWR) and

nm_s = percentage of anglers fishing in state s that are male and married and between the ages of 18 and 44 (NSFHWR).

f_s = state-level fertility rate (see Eq.[10.2])

The base year for the NPA_h estimates is 2001, which is the most recent year for NSFHWR data. To estimate NPA_h for subsequent years, it was assumed that this exposed population associated with each HUC would grow at the same rate as the projected annual population growth for females 15-44 in the entire 37-state study area.[15] Population growth rates based on Woods and Poole (2001) population projections for females 15-44 in the study area were used to estimate NPAh up to 2025. For years beyond 2025, it was assumed that NPA_h in the study area increase linearly at the same rate as the annual population growth rate for females 15-44 from 2024 to 2025.

Sixth, the georeferenced mercury concentration estimates (summarized in Tables 10-2 and 10-3) were used to estimate average mercury concentrations in each HUC for each waterbody type (CHg_{lh}, CHg_{rh}). This approach assumes that the sample averages of CHg (from available NLFA and NLFTS sampling data) for each waterbody type in each HUC provide an unbiased estimate of the true average mercury concentrations in the corresponding waterbody type and HUC. The weighted (by number of fishing days in each waterbody type) average mercury concentration in freshwater fish in each HUC was estimated as follows:

$$CHg_h = \frac{(A_{lh} * CHg_{lh}) + (A_{rh} * CHg_{rh})}{A_{lh} + A_{rh}}. \qquad \text{(Eq. 10.15)}$$

As in the case of the population centroid approach, an alternative method is to apportion NP_h to different water bodies (j = river or lake). Each subpopulation (NP_{jh}) can then be matched with the average mercury concentration for the corresponding waterbody j (CHg_{jh}).

Summary of Results. This section summarizes some of the key results from applying the angler destination approach outlined above. Figures 10-6 and 10-7 display the estimated distributions of lake- and river-fishing days across HUCs (A_{lh} and A_{rh}) in 2001. The HUCs with the highest estimated density of lake fishing (i.e., 200 or more annual lake-fishing days per square mile) are widely distributed across the study area. As expected, many are located in areas with extensive lake shoreline miles such as the Great Lakes region and in Minnesota, and several are also located in Tennessee, Kentucky, and Florida. The HUCs with the highest density of river fishing (i.e., 200 or more annual river fishing days per square mile) are most heavily concentrated in Pennsylvania, New York, Connecticut, and Massachusetts, as well as in North Carolina, Tennessee and Florida.

Table 10-10 reports the estimated annual number of prenatally exposed children across states and HUCs in 2001. It is important to note that, in contrast to the estimates reported in Tables 10-7 and 10-8 for the population centroid approach, the states in this table refer to where fishing took place and where the mercury exposure originated from rather than state of residence. Using the angler destination approach, the size of the exposed population of interest

[15] Woods and Poole county level population projections were not used to predict future exposed populations in the angler destination approach because the exposed population are based on fishing destination rather than residential location.

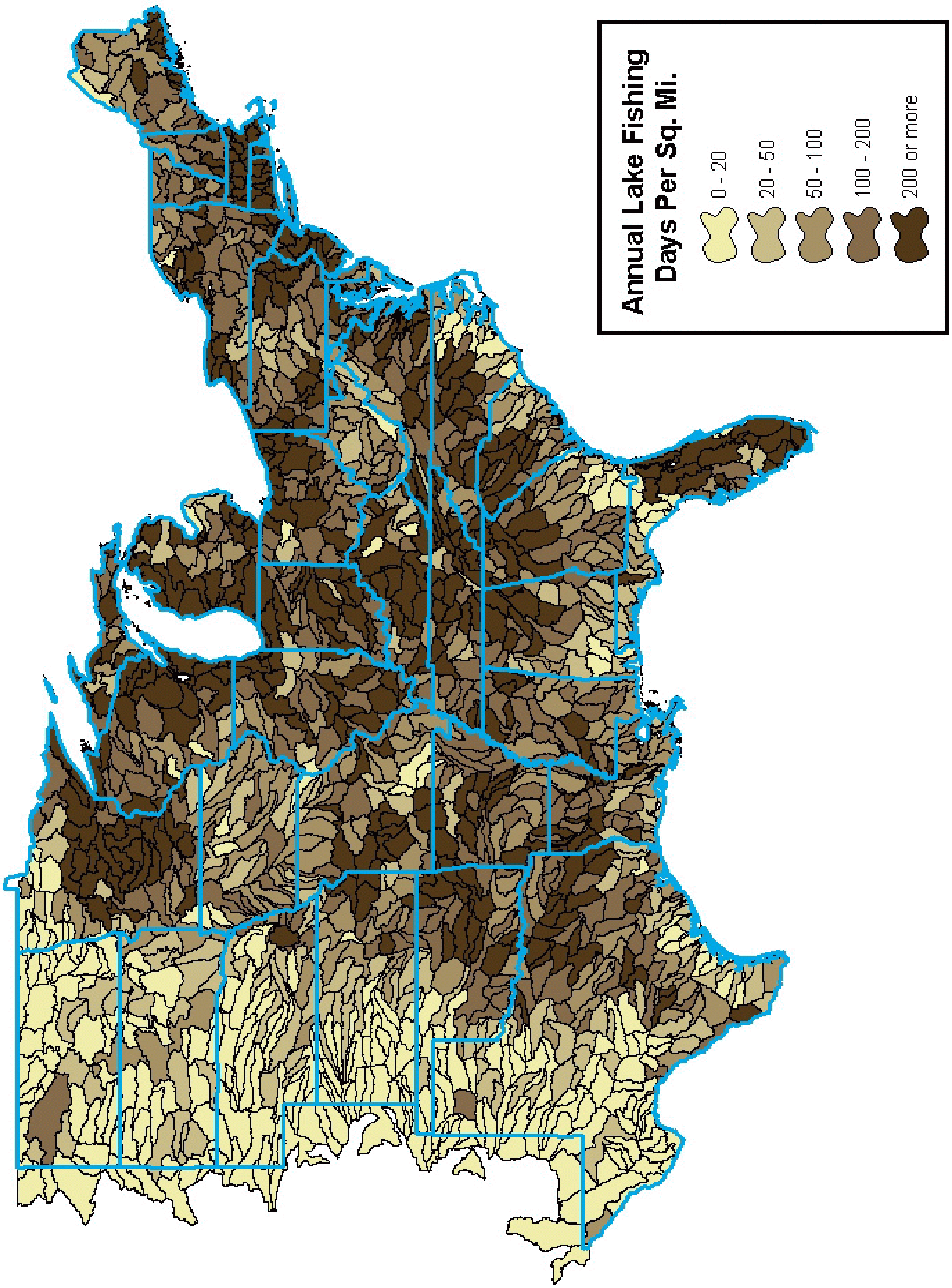

Figure 10-6. Estimated Distribution of Lake-Fishing Days Across HUCs in 2001

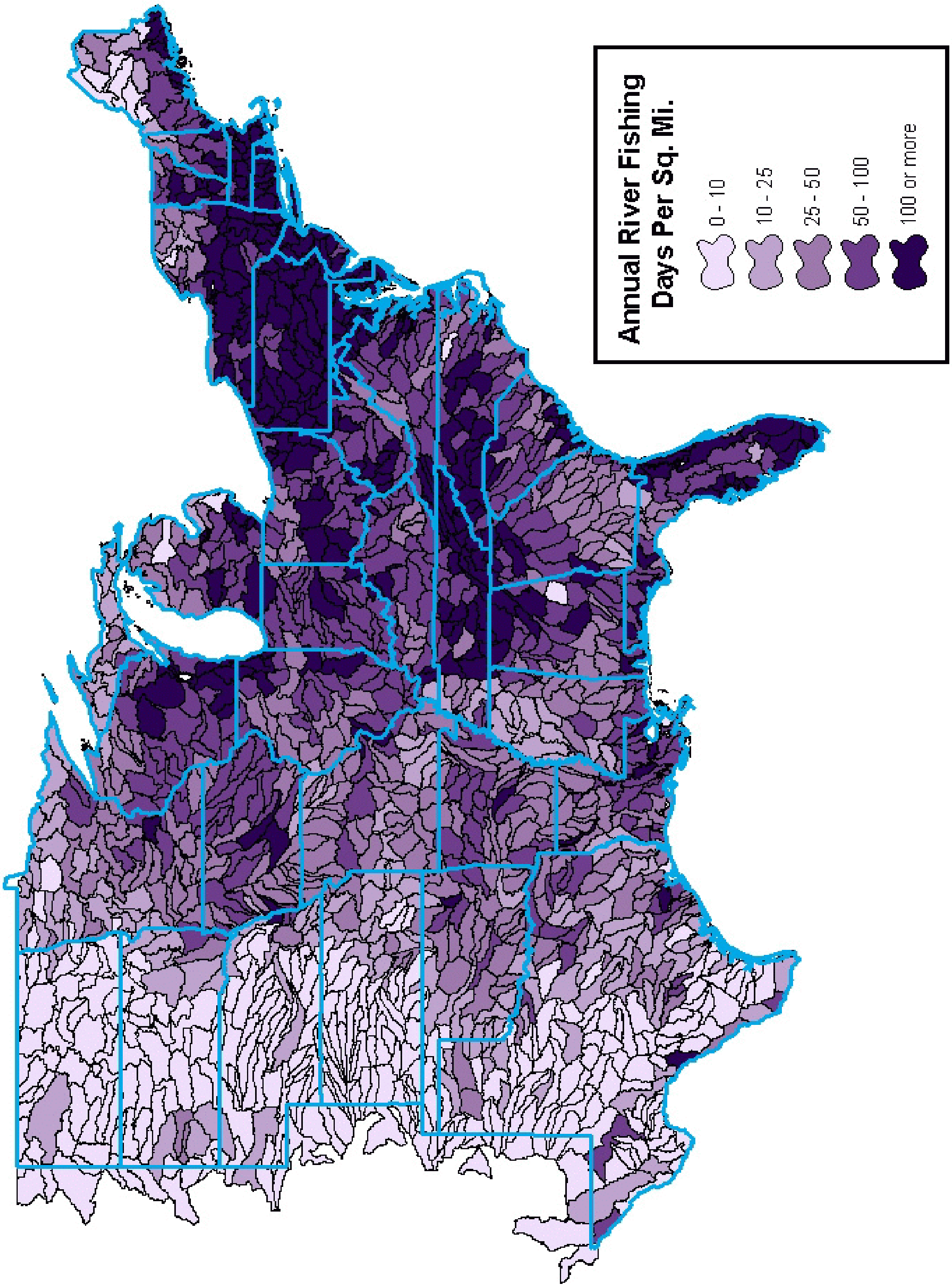

Figure 10-7. Estimated Distribution of River-Fishing Days Across HUCs in 2001

Table 10-10. State-Level Summary of Exposed Population Estimates: Angler Destination Approach

	Estimated Annual Number of Prenatally Exposed Children in 2001				
State	Average per HUC	Total	State	Average per HUC	Total
AL	251	13,077	NC	335	19,451
AR	253	14,669	ND	54	2,684
CT	397	4,762	NE	77	5,477
DC	4	8	NH	165	2,805
DE	59	592	NJ	809	10,520
FL	421	22,748	NY	412	21,401
GA	406	21,530	OH	679	30,556
IA	208	12,069	OK	350	23,791
IL	636	33,696	PA	433	24,704
IN	603	23,517	RI	167	835
KS	133	11,929	SC	329	11,837
KY	330	15,840	SD	65	3,798
LA	212	12,491	TN	340	20,723
MA	340	6,808	TX	309	64,537
MD	333	7,318	VA	357	18,224
ME	135	2,977	VT	102	1,726
MI	344	20,998	WI	470	24,429
MN	448	36,700	WV	139	5,149
MO	299	19,720	**Study Area**	**313**	**586,516**
MS	230	12,417			

(i.e., prenatally exposed children) was estimated to average more than 15,000 per state in the study area, with a total of almost 590,000 in the entire area. The state with the highest estimated number of prenatally exposed children is Texas, with almost 65,000, followed by Minnesota and Ohio, both with over 30,000. This total exposed population estimate is roughly 35 percent greater than the estimate based on the population centroid approach. For reasons discussed in more detail in Section 10.7, the assumptions underlying the angler destination approach are likely to overestimate the exposed population of interest, whereas those underlying the population centroid approach are likely to underestimate the exposed population.

10.4 Estimation of Mercury Exposures, IQ Decrements, and Lost Future Earnings

This section describes the methods and results for estimating mercury exposures through consumption of noncommercial freshwater fish. It describes how the estimates derived and summarized in the previous section—in particular, distributions of (1) the annual number of prenatally exposed children and (2) the average mercury concentrations in freshwater fish consumed by their mothers—were combined to calculate mercury ingestion levels from recreationally caught freshwater fish in this population. In addition it describes methods for translating these exposure estimates into corresponding estimates of (1) expected reductions in IQ levels achieved by the prenatally exposed children and (2) expected reductions in the value of future earnings resulting from the IQ decrements.

10.4.1 Modeling Approach for Estimating Individual Exposures

Both the population centroid approach and the angler destination approach described above define distinct subpopulations of women of childbearing age in angler households. Women in each subpopulation i are estimated to be exposed to the same average levels of mercury in recreationally caught freshwater fish. In the population centroid approach these subpopulations are grouped by Census block group (and still further by income and urban-rural area categories), whereas in the angler destination approach they are grouped by HUC.

To estimate average daily mercury ingestion rates for each subpopulation i, a similar method was applied using results from both the population centroid and the angler destination approach. In both cases, the ingestion rates were calculated based on the following equation:

$$HgI_i = CHgFC_i * C = (CHgFU_i * CCF) * C \qquad (10.16)$$

where

HgI	=	average daily mercury ingestion rate (µg/day);
CHgFU	=	average mercury concentration in uncooked freshwater fish (µg/g = ppm);
CCF	=	cooking conversion factor: ratio of mercury concentration in cooked fish to mercury concentration in uncooked fish (= 1.5);
CHgFC	=	average mercury concentration in cooked freshwater fish (ppm); and
C	=	average daily self-caught freshwater cooked fish consumption rate (g/day) = 8 g/day.

To determine an appropriate daily fish consumption rate (C) for the analysis, EPA conducted an extensive review of existing literature characterizing self-caught freshwater fish consumption including: (a) EPA's 1997 Exposure Factors Handbook (EPA, 1997b), (b) EPA Office of Water's documentation presenting methodologies for deriving Ambient Water Quality Criteria (USEPA, 2000c), (c) studies recommended by peer reviewers of the mercury benefits model, and (d) studies identified in the open literature, in order to insure that the best available data were used in modeling ingestion for the recreational fisher population. In conducting this literature review, the modeling team applied several key criteria in assessing the applicability of consumption data presented in studies:

1. *Self-caught freshwater consumption data for regions relevant to the study area*: The ingestion rate data had to apply to the consumption of freshwater fish caught by fishers at locations representative of the 37 state study area. Many of the studies that were reviewed included saltwater fish species, or some component of commercially caught (i.e., store-purchased) fish, either of which resulted in exclusion of the study from consideration. In addition, some of the studies, provided estimates for populations located in areas of the United States outside of the 37 state study area, where fishing behavior could differ systematically (in terms of types of fish harvested and seasonality) from the eastern half of the country.

2. *Consumption rates (including meal event frequency data) that supported generation of an annual-averaged daily consumption rate for fish*: The concentration-response function for IQ decrements is based on maternal hair mercury concentrations, which, in turn, is based on the estimated annual-averaged daily exposure level for methylmercury in ug/kg bodyweight/day. This exposure level is itself based on an annual-averaged daily consumption rate for fish (for the study population, or modeled individuals within that population) combined with the methylmercury concentration in ingested fish. Because the analysis required annual-averaged daily fish consumption estimates, the underlying consumption data had to include information on meal event frequency over longer periods. This is especially true for the high-end percentile consumption estimates required to generate the lognormal distribution of recreational fisher consumption rates. Many of the fish consumption studies that were reviewed, provided data for fish consumption over a relatively short survey period (days) and did not include information on the long-term frequency of self-caught fish meals. Only those studies providing long-term frequency data were ultimately selected for this analysis.

3. *Coverage for the entire recreational fisher population including consumers and non-consumers (catch and release)*: Modeling of recreational fisher exposure can be conducted either focusing on consumers only (i.e., excluding catch-and-release fishers), or by modeling "all fishers" including those who consume their catch and those who engage in catch-and-release activity. Because we are not able to identify in the data the portion of recreational anglers that are consumers only, we have selected a consumption rate that applies to "all fishers" `(i.e., one that incorporates a 0 g/d consumption rate for non-consumers). [16]

Based on application of the above criteria to available studies characterizing recreational freshwater fish consumption, it was decided that the ingestion rates for recreational freshwater fishers specified as "recommended" in the EPA's Exposure Factors Handbook (mean of 8 g/day and 95th percent of 25 g/day), represented the most appropriate values to use in this analysis. These recommended values were derived based on ingestion rates from four studies conducted in Maine, Michigan and Lake Ontario (Ebert et al., 1992; Connelly et al., 1996; West et al., 1989; West et al., 1993) that matched the suitability criteria presented above (i.e., annual-averaged daily intake rates for self-caught freshwater fish by all recreational fishers including consumers and non-consumers). The mean values presented in these four studies ranged from 5 to 17 g/day, while the 95th percent values ranged from 13 to 39 g/day (Note: the 39 g/day value actually represents a 96th percent value). The EPA "recommended values" were developed by

[16] Note: exposure modeling for recreational fishers could have focused on consumers and excluded those engaging in catch-and-release (i.e., non-consumers). However this would require representative information on "percent consumers" within the recreational fisher population to allow the overall recreational fisher population to be parsed appropriately. While this information is available from a number of fish consumption surveys, the values can differ significantly across those surveys, suggesting that there is uncertainty associated with our ability to differentiate (especially at the national level) recreational fishers based on consumption versus non-consumption status. Consequently, EPA decided to model recreational fisher exposure focusing on all fishers and did not attempt to differentiate consumers from non-consumers.

considering the range and spread of means and 95th percent values presented in the four studies. The average daily fish consumption rate of 8 g/day correspond to approximately (8 x 365 / 117 =) 25 fish meals per year using the average fish meal size of 117 g/meal from Pao et al. (1982). EPA recognizes that use of mean and 95th percent consumption rates based on these four studies may not be representative of fishing behavior across the entire 37 state study area and that there may be regional trends in consumption that differ from the values used in this analysis. However, EPA believes that these four studies do represent the best available data for developing recreational fisher ingestion rates for the 37 state study area that meet all of the study suitability criteria presented above. It should be noted that there are a large number of local-scale studies, including Creel surveys, that characterize fishing practices for specific waterbodies or groups of waterbodies in particular geographic areas. However, these studies often do not meet one or more of the suitability criteria presented above and are not readily generalizable to larger regional areas. Consequently, these local-scale studies were not used in deriving ingestion rates for this analysis.

Because the consumption rate estimate C is for cooked fish and the mercury concentrations are estimated for uncooked filet, a conversion factor (CCF) was applied to estimate mercury concentrations in cooked fish. Cooking fish tends to reduce the overall weight of fish by approximately one-third (Great Lakes Sport Fish Advisory Task Force, 1993). Because volatilization of mercury is unlikely to occur during cooking, the overall amount of mercury will stay unchanged during cooking, and the concentration of mercury will increase by a factor of roughly 1.5 (Morgan, Berry, and Graves, 1997).

10.4.2 Modeling Approach for Estimating IQ Effects and Lost Earnings

Estimating the IQ decrements in children that result from mothers' ingestion of mercury required two steps. First, based on the estimated average daily maternal ingestion rate, the expected mercury concentration in the hair of exposed pregnant women was estimated as follows:

$$CHgH_i = (0.08)^{-1} * (HgI_i/W) \qquad \text{(Eq. 10.17)}$$

where

CHgH = average mercury concentration in maternal hair (ppm)
W = average body weight for female adults ages 15-44 (= 64 kg)

This conversion rate between average daily ingestion rate and maternal hair concentration is based on the one compartment toxicokinetic model used for deriving EPA's reference dose for MeHg by Swartout and Rice (2000). Uncertainty and variability in various input parameters was analyzed using Monte Carlo simulation to establish a relationship between the mercury ingestion dose and hair mercury concentration. The simulation results indicated that the median conversion factor was 0.08 μg/Kg-day of mercury ingestion for 1 ppm of mercury concentration in hair. The 2002 EPA Workshop on Methylmercury Neurotoxicity recommended that this one compartment model might be better suited than PBPK model in modeling dose-response (EPA, 2002c). The average body weight estimate (W) was based on EPA's Exposure Factor Handbook (EPA, 1997a).

Second, to estimate the expected IQ decrement in offspring resulting from in utero exposure to mercury through mothers' fish consumption, the following dose-response relationship was applied:

$$dIQ_i = 0.131 * CHgH_i \qquad \text{(Eq. 10.18)}$$

where

dIQ = IQ decrement in exposed mother/child (IQ pts)

This dose response relationship is based on the statistical analysis in Section 9 that integrates the results of the available epidemiological studies of mercury and neurobehavior effects (see Section 9 for additional details).

The monetary value of losses resulting from IQ decrements were then assessed in terms of foregone future earnings for the affected individuals. These losses were estimated using to the following equation:

$$V_i = VIQ_i * dIQ_i \qquad \text{(Eq. 10.19)}$$

where

V = present value of net earnings losses per IQ point loss per prenatally exposed child (1999 dollars)

VIQ = value per change in IQ point (= $8,807)

This valuation approach for assessing losses associated with IQ decrements is based on an approach used by EPA to assess benefits for reductions in lead exposures (EPA, 2000a). For that analysis, EPA used results from a study by Salkever (1995) to estimate the effects of IQ loss on expected future earnings and years of education.

Salkever (1995) analyzes data from the National Longitudinal Study of Youth (NLSY) and uses a three equation regression model to estimate the relationships between IQ levels, educational attainment, and expected future earnings. The results of this study indicate that the average effect (for men and women combined) of a one point decrease in IQ is:

1. A 2.379 percent decrease in future earnings; and
2. A 0.1007 decrease in years of schooling.

To estimate the expected monetary value these effects, EPA first estimated the average present value of future earnings at the time of birth for a person born in the U.S. Using earnings data from the 1992 Current Population Survey (CPS) and discounting at a 3 percent annual rate, this present value was estimated to be $366,021 in 1992 dollars.

EPA then estimated the average direct and indirect costs associated with one additional year of schooling. Based on Department of Education data, the average annual expenditure per student was estimated to be $5,500, and the average annual opportunity cost (lost income from being in school) was estimated to be $10,925. Assuming that these costs were incurred at age 19 (based on an average of 12.9 years of education among those over age 25 in the U.S.) the combined present value of these two costs at time of birth (discounted at 3 percent) were estimated to be $9,367 per additional year of schooling in 1992 dollars.

Combining these estimates with the results from the Salkever (1995) study summarized above implies that the average present value of net earnings losses associated with a one point decrease in IQ is $7,765 in 1992 dollars. This value is calculated as the average present value of lost earnings per IQ point loss ($8,708 = $366,021 * 0.2379) minus the partially offsetting change in average education costs per IQ point loss ($943 = $9,367 * 0.1007). Corrected for inflation using the GDP deflator, the average present value of net earnings losses per IQ point loss is $8,807 in 1999 dollars[17].

It is important to note that this value per IQ point lost is considered a "cost of illness" measure rather than a measure of willingness-to-pay (WTP) to prevent a loss of an IQ point. The cost-of-illness approach simply measures ex post costs and does not attempt to measure the loss in utility due to pain and suffering or the costs of any averting behaviors that individuals have taken to avoid the illness altogether (EPA, 2000b). However, the cost-of-illness estimate may be considered a lower bound estimate of WTP (Harrington and Portney, 1987; Berger et al., 1987). The main reason that the cost of illness understates total WTP is the failure to account for many effects of disease beyond those associated solely with net earnings.

10.5 Model Results: Estimated Benefits of Utility Mercury Emission Controls

Based on the modeling approach described above, mercury ingestion levels, IQ decrements, and lost future earnings were estimated for each modeled subpopulation and then aggregated across the study area. These estimates were calculated for the following conditions and emissions control scenarios:

- 2001 Base Case
- 2001 Utility Emissions Zero-Out
- 2020 Base Case with CAIR
- 2020 Utility Emissions Zero-Out
- 2020 CAMR Control Option 1
- 2020 CAMR Control Option 2

For the 2001 Base Case, it was assumed the mercury emissions, and therefore mercury concentration levels in fish, would remain constant at their currently observed levels (summarized in Table 10-2).

For each of the other five emissions control scenarios, mercury concentration levels were re-estimated for the entire study area. To estimate the effect of emissions controls, EPA first

[17] The value per IQ estimate using a 7 percent discount rate is $1,580 per IQ point.

conducted air quality modeling runs and estimated mercury deposition levels across the study area for baseline (i.e., 2001 Base Case) emissions conditions. Based on the format of the air quality modeling results, the study area was segmented into roughly 10,000 36x36 km grids, and separate mercury deposition estimates were generated for each grid. EPA then repeated this process for each of the five emissions control scenarios. Comparing each set of model results to baseline results, EPA estimated percent reductions in mercury deposition relative to baseline for each grid and for each emissions control scenario. Based on results from EPA's Mercury MAPs (MMaps) approach discussed in Section 3, reductions in mercury fish tissue concentrations in each grid were assumed to be directly proportional to the estimated mercury deposition reductions for the grid. As Section 3 discusses, this proportionality assumption is limited to areas where air deposition is the primary source of mercury loading to a waterbody, and is applicable at steady-state (i.e., when the waterbody comes to equilibrium after a given change in mercury loading).

Therefore, to re-estimate fish tissue mercury concentrations at each mercury sampling point for each emissions control scenario, the following steps were applied:

1. Each lake and river sampling point in the study area was mapped to its corresponding air quality grid using GIS.

2. The baseline (Base Case 2001) mercury fish tissue concentration estimates for each sampling point (as summarized in Table 10-2) were reduced by the same percentage as the estimated percent reduction in mercury deposition for its corresponding grid.

The resulting estimates of changes in mercury fish tissue concentrations are summarized in Tables 10-11 and 10-12. Table 10-11 reports results relative to the 2001 baseline levels for two emissions control scenarios: 2001 Utility Emissions Zero-Out and 2020 Base Case with CAIR.

Under the 2001 Utility Emissions Zero-Out scenario, the fish tissue mercury concentrations in lakes and rivers were estimated to decline by an average of 11 and 15 percent respectively across the study area, with the largest percentage reductions occurring in Pennsylvania, Ohio, and West Virginia. The percent reductions across sampling points varied from less than one percent to over 67 percent relative to 2001 baseline levels.

Under the 2020 Base Case with CAIR, mercury concentrations were estimated to decline on average by similar amount—11 percent for lakes and 14 percent for rivers. Again the largest percentage reductions were estimated to occur in Pennsylvania, Ohio, and West Virginia; however, the range of estimated reductions across sampling points was much larger for this scenario, going from over 10 percent *increases* in mercury levels at some points to over 60 percent reductions at others.

Table 10-11. Effects of Emission Control Scenarios—Percent Reduction in Estimated Fish Tissue Mercury Concentrations from 2001 Base Case[a]

	Mean (Min–Max) Percent Reduction Across Sampling Sites			
	Utility Mercury Emissions Zero-Out in 2001		Base Case with CAIR in 2020	
	Lake Sites	River Sites	Lake Sites	River Sites
Study Area	10.6 (0.6 – 67.7)	15.3 (0.5 – 67.7)	10.7 (–24.5 – 61.8)	14.4 (–27.1 – 61.8)
State				
AL	19 (8 – 41.3)	16.8 (7.6 – 53.4)	19.1 (–1.7 – 41.2)	15.5 (–3.3 – 43.4)
AR	7.3 (4.7 – 18.3)	7.2 (4.4 – 18.3)	6 (–0.5 – 10.2)	6.3 (–3.6 – 13.2)
CT	7.7 (4.7 – 10.1)	8.3 (4.7 – 10.1)	15.1 (5.4 – 46.1)	13.1 (5.1 – 46.1)
DC	—	28.5 (28.5 – 28.5)	—	39.2 (39.2 – 39.2)
DE	19 (16.7 – 22.5)	19.6 (12.3 – 29.5)	15.6 (11.7 – 19.7)	17.6 (2.5 – 25.7)
FL	4.1 (0.6 – 15.9)	4.3 (0.5 – 45.5)	8.2 (–3.1 – 19.3)	7 (–3.1 – 40.5)
GA	15.3 (5.4 – 33.6)	15.2 (4.8 – 42.3)	15.5 (7 – 32.7)	16 (6.8 – 41.5)
IA	7.8 (4.1 – 25.3)	7.2 (4.1 – 25.3)	4.5 (–2.7 – 6.1)	2.9 (–12.5 – 6.1)
IL	23.4 (9.9 – 40.3)	16.4 (6.2 – 30)	15.5 (–0.9 – 31.4)	8.1 (–15.2 – 27.1)
IN	22.7 (15.9 – 32.4)	19.7 (7.5 – 51.9)	12.1 (–0.9 – 33.6)	16 (–9.3 – 44.4)
KS	5.3 (3.6 – 8.6)	7.6 (2.9 – 13.2)	4.2 (2.8 – 4.9)	3.8 (–0.9 – 9.8)
KY	17.1 (2.7 – 21.5)	23.9 (2.7 – 38.1)	22.1 (15.6 – 41.4)	23.1 (2.9 – 41.4)
LA	5.6 (3.1 – 11.3)	4.4 (3 – 7)	6.3 (2.7 – 12.4)	5.4 (3.2 – 15.8)
MA	9.3 (5.9 – 21.4)	3.6 (3.6 – 3.6)	11 (4.6 – 18.1)	55.9 (55.9 – 55.9)
MD	33.6 (22.8 – 50.5)	32.4 (13.9 – 58.5)	32 (21 – 48.4)	30.6 (5.1 – 56.9)
ME	3.1 (2.1 – 5)	3.4 (2.1 – 6)	5.6 (–2.9 – 37.2)	4.6 (–13.3 – 37.2)
MI	14 (2.6 – 42.8)	14.5 (2.8 – 27.8)	9.8 (0.5 – 30.7)	16.8 (3.7 – 40.1)
MN	2.3 (0.9 – 9.5)	2.9 (0.9 – 8.2)	4.5 (–24.5 – 41.9)	2.4 (–24.5 – 41.9)
MO	8.9 (5.9 – 15.5)	8.9 (6.2 – 11.2)	5.9 (–5.9 – 21.7)	2.9 (–0.9 – 5.3)
MS	6.9 (4.8 – 9.4)	10.1 (5.2 – 15)	8.5 (6.4 – 9.6)	17 (4.2 – 50.3)
NC	18.5 (10.1 – 30.8)	16.4 (8.5 – 51.1)	18.3 (12 – 30.9)	17.4 (4.9 – 49.8)
ND	2.9 (1.3 – 5)	6.1 (1.2 – 16.1)	4.3 (3 – 5.1)	6.3 (2.6 – 10.4)
NE	5 (1.7 – 25.3)	5.4 (1.4 – 25.3)	2.4 (–15.1 – 6.1)	3.7 (–15.1 – 8.7)
NH	5.9 (4.3 – 6.8)	6 (4.9 – 9)	8.4 (3.8 – 15.6)	9.4 (2.2 – 30.1)
NJ	16.9 (10 – 28)	13.1 (10 – 28)	9.4 (0.4 – 16)	2.6 (–7.6 – 16)
NY	13.9 (6.1 – 30.5)	12.5 (5.5 – 30.4)	14.3 (–7.6 – 26.7)	11.2 (–7.6 – 35.9)
OH	27.2 (15.4 – 45.6)	29.8 (15.4 – 55.8)	25.6 (16.1 – 45.3)	27.1 (9.7 – 53.7)
OK	7.3 (1.8 – 18.9)	8.3 (1.8 – 18.9)	5.4 (–3.7 – 13.8)	6.6 (–3.7 – 35.8)

(continued)

Table 10-11. Effects of Emission Control Scenarios—Percent Reduction in Estimated Fish Tissue Mercury Concentrations from 2001 Base Case (continued)

	Mean (Min–Max) Percent Reduction Across Sampling Sites			
	Utility Mercury Emissions Zero-Out in 2001		Base Case with CAIR in 2020	
	Lake Sites	River Sites	Lake Sites	River Sites
State				
PA	31.7 (10.3 – 67.7)	34.8 (10.3 – 67.7)	27.5 (4.6 – 61.8)	33 (–11.9 – 61.8)
SC	15.3 (13.4 – 20.2)	14.8 (4.3 – 28.9)	14.8 (12.5 – 20.7)	13.5 (–21.5 – 44.3)
TN	20.3 (9.9 – 37.4)	21.7 (11.5 – 37.4)	21.3 (–7.4 – 38.9)	22.4 (12.7 – 38.9)
TX	9 (1.1 – 64.4)	6.8 (0.5 – 30.7)	10.3 (3.3 – 55.6)	6.8 (–14.5 – 26.2)
VA	25.6 (16.6 – 41.1)	23.8 (9.2 – 35.7)	23 (–4.8 – 31.3)	26 (17.2 – 39.2)
VT	6.8 (5.7 – 8.3)	6.9 (2 – 8.3)	10.8 (7.5 – 22.6)	11.9 (4.7 – 30.1)
WI	5.6 (2.3 – 40.3)	8 (2.8 – 40.3)	5.9 (–4.8 – 24.1)	5.8 (–27.1 – 35.6)
WV	33.2 (32.8 – 34.3)	37.4 (20.3 – 60.6)	33.5 (32.6 – 34.5)	36.8 (21.4 – 60.2)

[a] For summary purposes data are reported at a state level in this table. The analysis accounted for within state variations in estimated percent reductions in mercury concentrations.

Using the resulting mercury concentration estimates from the 2020 Base Case with CAIR scenario as a new reference point, Table 10-12 reports estimated reductions in mercury levels for the other three emissions control scenarios. Under the 2020 Utility Emissions Zero-Out scenario, the fish tissue mercury concentrations in lakes and rivers were estimated to decline by an average of 5 and 6 percent respectively across the study area, with over 10 percent reductions in both lake and river levels estimated to occur in Illinois, Michigan, and Missouri.

The 2020 CAMR Control Option 1 was estimated to reduce fish concentrations estimates relative to the 2020 baseline by an average of less than 1 percent, whereas the 2020 CAMR Control Option 2 was estimated to reduce these concentrations by an average of between 1 and 2 percent. In both cases, states with relatively high estimated reductions (3 percent or more reductions in both lakes and rivers) included Pennsylvania and New Jersey.

Although Mercury Maps assumes that reductions in mercury deposition levels result in roughly proportionate reductions in steady state fish tissue concentrations (everything else being equal), there is uncertainty regarding how long it takes for these effects to occur (i.e., for fish tissue concentrations to reach a new steady state), which is discussed in detail in Section 3 of this report. Based on the response times from the case studies discussed in Section 3[18], to reflect the possibility that reductions in mercury emissions in 2001 and in 2020 would have lagged effects on fish tissue concentrations, we present a range of benefits based on the 10 and 20 year lags as

[18] Case studies of individual ecosystems (as presented in Section 3) 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.

Table 10-12. Effects of Emission Control Scenarios—Percent Reduction in Estimated Fish Tissue Mercury Concentrations from 2020 Base Case with CAIR[a]

	Mean (Min–Max) Percent Reduction Across Sampling Sites					
	Utility Mercury Emissions Zero-Out in 2020		CAMR Control Option 1		CAMR Control Option 2	
	Lake Sites	River Sites	Lake Sites	River Sites	Lake Sites	River Sites
Study Area	5.41 (0.32 – 45.8)	6.38 (0.27 – 33.19)	0.7 (–4.68 – 36.5)	0.99 (–8.96 – 19.49)	1.27 (–3.55 – 36.92)	1.65 (–8.83 – 28.13)
State						
AL	4.82 (2.58 – 13.16)	5.31 (2.39 – 24.74)	0.37 (–4.34 – 1.91)	0.8 (–4.34 – 5.93)	1.19 (0.24 – 2.38)	1.37 (–0.33 – 6.11)
AR	5.55 (2.85 – 22.33)	5.43 (2.6 – 23.08)	2.46 (0.68 – 17.95)	2.24 (0.6 – 18.38)	2.97 (0.92 – 18.28)	2.85 (0.82 – 18.72)
CT	3.03 (2.4 – 3.86)	2.95 (2.4 – 3.86)	0.67 (0.56 – 1.08)	0.76 (0.54 – 1.08)	0.77 (0.65 – 1.19)	0.87 (0.65 – 1.19)
DC	—	16.32 (16.32 – 16.32)	—	1.57 (1.57 – 1.57)	—	2.37 (2.37 – 2.37)
DE	8.83 (7.18 – 9.88)	10.51 (7.05 – 18.64)	2.38 (0.55 – 3.85)	3.56 (–7.46 – 10.12)	2.84 (0.96 – 4.47)	4.01 (–6.75 – 10.12)
FL	1.5 (0.32 – 4.16)	1.24 (0.27 – 5.67)	0.1 (–1.39 – 2.34)	-0.01 (–8.96 – 2.34)	0.23 (–1.26 – 2.57)	0.12 (–8.83 – 2.57)
GA	5.43 (1.95 – 16.36)	4.77 (1.79 – 33.19)	0.39 (–0.44 – 1.25)	0.41 (–0.44 – 1.68)	2.58 (–0.09 – 12.63)	1.95 (–0.09 – 28.13)
IA	9.02 (4.25 – 25.26)	8.84 (3.89 – 25.26)	1.71 (0.45 – 7.56)	1.31 (0.43 – 7.56)	3.09 (1.31 – 11.87)	2.41 (1.01 – 11.87)
IL	13.2 (7.99 – 18.79)	11.56 (4.51 – 24.5)	1.38 (0.52 – 2.43)	1.19 (–0.42 – 4.35)	2.05 (1.07 – 3.96)	1.78 (–0.11 – 4.76)
IN	14.08 (7.03 – 25.12)	9.28 (2.85 – 26.27)	1.95 (–0.13 – 6.72)	0.63 (–3.78 – 6.66)	2.37 (0.14 – 7.31)	0.97 (–3.62 – 7.23)
KS	6.51 (4.24 – 11.06)	9.78 (3.07 – 16.52)	1.22 (0.63 – 2.41)	0.89 (0.34 – 3.02)	2.89 (2.13 – 4.42)	3.95 (1.73 – 6.14)
KY	5.72 (1.84 – 12.06)	7.21 (1.84 – 14.98)	0.2 (0.06 – 0.35)	0.36 (–0.41 – 1.14)	0.53 (0.29 – 0.65)	0.66 (–0.21 – 1.4)
LA	3.24 (1.57 – 6.35)	2.51 (1.57 – 6.17)	0.69 (0.24 – 4.02)	0.37 (0.16 – 0.7)	0.92 (0.35 – 4.19)	0.52 (0.24 – 0.89)
MA	2.92 (2.1 – 5.59)	2.16 (2.16 – 2.16)	0.69 (0.49 – 1.38)	–1.33 (–1.33 – –1.33)	0.8 (0.58 – 1.5)	–1.24 (–1.24 – –1.24)
MD	8.53 (6.03 – 11)	8.02 (3.04 – 12.7)	2.63 (1.52 – 4.36)	2.31 (–0.74 – 4.54)	2.88 (1.76 – 4.59)	2.79 (–0.39 – 5.77)
ME	2.3 (0.95 – 10.41)	2.46 (0.95 – 10.41)	0.98 (0.26 – 6.05)	1.04 (0.26 – 6.05)	1.18 (0.33 – 8.59)	1.29 (0.33 – 8.59)
MI	10.21 (2.41 – 29.78)	10.88 (2.94 – 22.25)	0.61 (–3.63 – 6.72)	0.6 (–0.97 – 1.98)	1.15 (–3.35 – 7.31)	1.04 (–0.57 – 2.61)
MN	2.49 (0.96 – 9.29)	3.01 (1.06 – 12.15)	0.2 (0.05 – 0.83)	0.22 (0.05 – 0.66)	0.43 (0.18 – 2.14)	0.57 (0.18 – 6.13)
MO	10.34 (4.96 – 23.21)	11.51 (6.59 – 16.52)	1.73 (0.86 – 4.59)	1.22 (0.59 – 2.15)	2.95 (1.8 – 6.09)	3.39 (1.49 – 5.9)

(continued)

Table 10-12. Effects of Emission Control Scenarios—Percent Reduction in Estimated Fish Tissue Mercury Concentrations from 2020 Base Case with CAIR (continued)

	Mean (Min–Max) Percent Reduction Across Sampling Sites					
	Utility Mercury Emissions Zero-Out in 2020		CAMR Control Option 1		CAMR Control Option 2	
	Lake Sites	River Sites	Lake Sites	River Sites	Lake Sites	River Sites
State						
MS	3.14 (2.33 – 3.64)	3.61 (2.35 – 10.17)	0.68 (0.48 – 1.11)	0.74 (0.34 – 3.3)	0.98 (0.71 – 1.43)	0.97 (0.53 – 3.51)
NC	6.34 (2.79 – 11.45)	5.23 (2.29 – 24.07)	0.48 (–1.92 – 1.85)	1.13 (–0.11 – 19.49)	0.83 (–1.47 – 2.16)	1.43 (0.23 – 19.73)
ND	2.67 (1.07 – 4.51)	5.85 (1.79 – 12.78)	0.71 (0.26 – 2.29)	1.43 (0.19 – 3.57)	0.78 (0.32 – 2.32)	1.54 (0.28 – 3.61)
NE	6.79 (1.59 – 27.99)	7.2 (1.52 – 27.99)	1.5 (0.21 – 8.14)	1.37 (0.2 – 8.14)	2.27 (0.32 – 9.81)	3.32 (0.32 – 16.8)
NH	3.72 (2.12 – 7.25)	3.12 (2.12 – 7.42)	1.07 (0.46 – 3.07)	0.92 (0.46 – 5.85)	1.19 (0.56 – 3.46)	1.04 (0.56 – 6.08)
NJ	8.62 (3.17 – 21.19)	6.79 (3.17 – 21.19)	4.74 (1.26 – 15.88)	2.97 (1.13 – 15.88)	4.94 (1.36 – 16.09)	3.09 (1.2 – 16.09)
NY	4.4 (1.97 – 11.34)	4.23 (1.62 – 13.11)	1.1 (0.48 – 2.37)	1.2 (–0.73 – 2.71)	1.34 (0.59 – 2.83)	1.44 (0.51 – 3.17)
OH	10.21 (6.05 – 16.69)	9.42 (4.53 – 16.69)	0.13 (–4.68 – 1.55)	0.56 (–4.68 – 2.54)	0.78 (–3.55 – 4.61)	1 (–3.55 – 4.61)
OK	6.29 (1.84 – 25.38)	6.58 (1.82 – 25.38)	0.6 (0.19 – 2.15)	0.72 (0.19 – 2.15)	3.21 (0.7 – 20.16)	3.16 (0.7 – 20.16)
PA	9.65 (4.53 – 18.4)	10.38 (3.96 – 25.64)	3.66 (0.63 – 10.58)	4.47 (0.36 – 16.73)	3.97 (0.89 – 10.94)	4.75 (0.84 – 16.95)
SC	5.8 (3.6 – 9.08)	5.98 (1.33 – 19.71)	0.49 (0.16 – 0.92)	0.29 (–0.23 – 1.57)	1.45 (0.71 – 2.57)	1.04 (0.27 – 2.3)
TN	5.12 (2.18 – 14.35)	5.58 (2.87 – 14.35)	1.03 (–0.28 – 11.02)	1.17 (0.22 – 11.02)	1.49 (0.41 – 11.33)	1.65 (0.72 – 11.33)
TX	3.5 (0.87 – 23.63)	3.35 (0.4 – 24.33)	0.54 (–0.05 – 7.28)	0.37 (0.06 – 2.11)	1 (0.2 – 11.4)	0.7 (0.12 – 3.9)
VA	8.88 (3.96 – 45.8)	7.72 (2.95 – 18.66)	3.86 (0.91 – 36.5)	2.09 (0.29 – 11.25)	4.22 (1.17 – 36.92)	2.46 (0.74 – 12.33)
VT	2.35 (1.82 – 2.76)	2.39 (0.88 – 2.74)	0.59 (0.08 – 1.04)	0.57 (0.15 – 1.04)	0.71 (0.2 – 1.14)	0.69 (0.22 – 1.14)
WI	5.1 (2.8 – 18.79)	7.4 (2.96 – 18.79)	0.34 (–0.05 – 2.43)	0.54 (–0.05 – 2.54)	1.08 (0.4 – 3.96)	2.02 (0.53 – 11.58)
WV	6.71 (6.07 – 7.52)	7.9 (4.38 – 14.88)	2.14 (1.56 – 2.99)	2.58 (0.69 – 9.4)	2.39 (1.81 – 3.24)	2.86 (0.89 – 9.67)

[a] For summary purposes data are reported at a state level in this table. The analysis accounted for within state variations in estimated percent reductions in mercury concentrations.

central estimates. We also provide results for the 5 and 50 year lags to demonstrate how benefits would differ under shorter and longer lag periods.

Results based on the modeling framework described above were estimated using exposure estimates from both the population centroid approach and the angler destination approaches. These results are summarized and discussed separately below.

The monetized benefits of the two CAMR control options—2020 CAMR Control Option 1 and 2020 CAMR Control Option 2—were estimated as the reduction in total IQ related losses in the affected population relative to 2020 baseline conditions (2020 Base Case with CAIR). In other words, benefits were calculated as the difference between (1) the estimated aggregate present value of earnings losses in the 2020 baseline scenario and (2) the estimated aggregate present value of earnings losses in the 2020 control scenario.

10.5.1 Results for the Population Centroid Approach

Applying the population centroid approach and equations 10.16 and 10.17 to estimate mercury ingestion levels requires matching block group subpopulations (N) with corresponding average mercury levels. As discussed in Section 10.1.2 one method for matching is to first subdivide the exposed population in each block group (NPA) according to demographic group i, distance interval j, and waterbody type k (lake or river) and then estimate the size of each subpopulation, NPA_{ijk}, using equation 10.7. This method assumes that each individual in the subpopulation receives all of his/her noncommercial freshwater fish from the same distance interval j and waterbody type k; therefore, each individual can be matched with the corresponding average mercury concentration from that distance interval and waterbody type (CHg_{jk}). The results reported in this section were generated using this method to match subpopulations and mercury concentrations.

One of the limitations of the available mercury concentration estimates, is that they do not provide lake and river estimates for each possible location in the study area. As reported in Table 10-9, for many block groups in the study area, mercury sampling locations did not exist for each of the specified distance categories. To address these data limitations, a simple spatial extrapolation method was used to provide estimates of exposures for all the distance intervals within and beyond 100 miles of each block group centroid. For the four distance intervals within 100 miles of each centroid, if they did not contain a lake or river sampling location for mercury, they were assigned the average lake or river concentration estimate (respectively) from the other distance categories within 100 miles of the centroid. For exposures beyond 100 miles from each centroid, the mean lake and river concentrations from the entire study area (0.23 ppm for lakes and 0.25 ppm for rivers, as reported in Table 10-2) were used.

Table 10-13 reports model results for mercury ingestion rates under baseline conditions in 2001. It disaggregates results by separating the estimated exposed subpopulations according to their assumed distance to the freshwater fish they consume. It reports the distribution of estimated HgI levels for subpopulations in the four distance intervals below 100 miles. The distributions are roughly similar for subpopulations in each distance interval, with an average (and 75th percentile) HgI of between 2.5 and 3 μg/day.

Table 10-13. Estimated Distribution of Mercury Ingestion by Distance Traveled to Fish: Population Centroid Approach—2001 Base Case

	Average Daily Maternal Ingestion of Mercury (µg/day) by Distance Interval					
	0–10 miles	>10–20 miles	>20–50 miles	>50–100 miles	>100 miles	Average
1st percentile	0.48	0.42	0.53	0.80	2.72	1.25
5th percentile	0.92	0.80	0.91	1.12	2.72	1.47
25th percentile	1.48	1.45	1.49	1.70	2.76	1.81
50th percentile	2.03	2.01	2.13	2.23	2.76	2.26
75th percentile	2.74	2.74	2.75	2.93	2.79	2.82
95th percentile	4.56	4.61	4.66	4.67	2.86	4.16
99th percentile	6.51	6.73	6.86	6.46	2.90	5.06
Mean	2.50	2.47	2.46	2.71	2.77	2.58
Std Dev	1.65	1.79	1.37	1.23	0.23	1.01

Table 10-14 summarizes model results for the 2001 Base Case based on exposure estimates from the population centroid approach in 2001.[19] The estimated annual number of prenatally exposed children in freshwater angler households is just over 434,000. The states with the largest estimated exposed populations are Texas and Illinois. The average daily mercury ingestion rate for pregnant women in freshwater angler households (HgI) was estimated to be 2.44 µg/day for the entire study area. The states with the highest average rates were primarily in New England, which is also where the highest estimates of average mercury concentrations in freshwater fish were located. For example, the average rates for Maine and New Hampshire were both estimated to be above 4 µg/day under baseline conditions. Average IQ decrements in prenatally exposed children were estimated to be over 0.06 points for the study area, with the highest average (per capita) reductions estimated again in New England states. Under baseline conditions in 2001, the total IQ losses associated with self-caught freshwater fish consumption were estimated to be 27,100 and the present value in 2001 of foregone net earnings associated with these IQ decrements was estimated to sum to almost $240 million (in 1999 dollars).

Table 10-15 summarizes model results for the 2020 Base Case with CAIR[16], based on the exposure estimates from the population centroid approach for 2020, as reported in Table 10-8. To provide a direct comparison with the 2001 Base Case results, the results reported in Table 10-15 assume that all mercury emissions reductions associated with CAIR have been implemented by 2020 and that there is no lag between emissions reductions and fish tissue mercury concentrations. Due to population growth between 2001 and 2020, the estimated annual number of prenatally exposed children in freshwater angler households is almost 482,000. The average daily mercury ingestion rate for pregnant women in freshwater angler households (HgI) was estimated to be roughly 12 percent below the 2001 Base Case level, at 2.14 µg/day for the

[19] For comparison purposes between the 2001 Base Case and the 2020 Base Case with CAIR, the benefits presented in Tables 10-14 and 10-15 do not reflect potential lags in fish tissue response times for a change in mercury deposition.

Table 10-14. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2001 Base Case[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (μg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	434,059	2.44	1.01	0.062	0.026	27,094	$550	$227	$238,626,988
State									
AL	9,948	1.99	0.72	0.051	0.018	507	$449	$161	$4,464,277
AR	9,832	2.56	0.57	0.066	0.015	645	$578	$128	$5,680,355
CT	4,134	3.37	0.90	0.086	0.023	357	$760	$202	$3,141,787
DC	236	1.80	0.20	0.046	0.005	11	$406	$45	$95,779
DE	836	2.26	0.28	0.058	0.007	48	$509	$63	$425,082
FL	21,218	3.74	0.88	0.096	0.022	2,029	$842	$198	$17,872,632
GA	20,255	2.59	0.68	0.066	0.017	1,340	$583	$153	$11,800,171
IA	8,448	1.54	0.19	0.039	0.005	334	$348	$43	$2,937,846
IL	27,670	1.69	0.28	0.043	0.007	1,193	$380	$63	$10,507,395
IN	13,640	2.04	0.33	0.052	0.008	713	$460	$75	$6,280,533
KS	8,246	3.29	1.00	0.084	0.026	694	$741	$225	$6,112,146
KY	10,322	2.18	0.47	0.056	0.012	574	$490	$106	$5,059,688
LA	10,709	2.51	0.69	0.064	0.018	689	$567	$155	$6,066,865
MA	5,663	3.85	0.35	0.098	0.009	557	$867	$80	$4,909,319
MD	5,881	1.60	0.28	0.041	0.007	240	$360	$62	$2,117,268
ME	2,378	4.86	0.73	0.124	0.019	296	$1,095	$165	$2,604,794
MI	17,028	2.40	0.34	0.061	0.009	1,044	$540	$76	$9,198,072
MN	22,806	1.74	0.30	0.045	0.008	1,017	$393	$68	$8,955,172
MO	16,804	2.28	0.97	0.058	0.025	982	$515	$218	$8,648,998
MS	7,909	2.63	0.52	0.067	0.013	532	$592	$118	$4,681,160
NC	13,483	2.73	0.66	0.070	0.017	943	$616	$150	$8,305,285

(continued)

Table 10-14. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2001 Base Case (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	2,101	2.22	0.55	0.057	0.014	119	$500	$124	$1,050,953
NE	5,138	1.73	0.33	0.044	0.008	228	$391	$73	$2,008,393
NH	1,966	4.25	0.41	0.109	0.011	214	$957	$93	$1,881,273
NJ	6,838	3.08	0.48	0.079	0.012	538	$693	$107	$4,739,979
NY	16,877	3.06	0.76	0.078	0.020	1,320	$689	$172	$11,628,981
OH	21,457	2.54	0.55	0.065	0.014	1,395	$573	$125	$12,288,120
OK	13,117	2.49	0.43	0.064	0.011	837	$562	$97	$7,369,896
PA	15,146	3.34	1.98	0.086	0.051	1,295	$753	$447	$11,405,684
RI	755	3.66	0.30	0.094	0.008	71	$825	$68	$623,444
SC	9,010	3.10	0.71	0.079	0.018	715	$699	$161	$6,301,059
SD	2,508	2.01	0.91	0.051	0.023	129	$453	$205	$1,135,391
TN	13,317	2.28	0.40	0.058	0.010	776	$513	$90	$6,834,282
TX	55,802	2.01	0.56	0.052	0.014	2,875	$454	$127	$25,324,716
VA	11,793	1.95	0.54	0.050	0.014	588	$439	$121	$5,174,750
VT	1,288	3.58	0.65	0.092	0.017	118	$806	$146	$1,038,103
WI	15,848	2.18	0.38	0.056	0.010	882	$490	$85	$7,770,204
WV	3,651	2.66	0.40	0.068	0.010	248	$599	$90	$2,187,136

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table. For comparison purposes with the 2020 Base Case with CAIR, benefits presented in this table do not reflect potential lags in fish tissue response to a change in mercury deposition.

Table 10-15. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2020 Base Case with CAIR[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (μg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	481,987	2.14	0.80	0.0548	0.0204	26,413	$1,412	$1,598	$232,623,719
State									
AL	9,871	1.69	0.72	0.0432	0.0185	427	$1,129	$1,205	$3,759,332
AR	10,877	2.34	0.52	0.0599	0.0133	651	$2,687	$1,771	$5,736,031
CT	4,437	2.93	0.75	0.0750	0.0192	333	$1,119	$809	$2,931,055
DC	211	1.47	0.06	0.0375	0.0016	8	$161	$133	$69,675
DE	894	1.98	0.23	0.0507	0.0060	45	$796	$617	$399,568
FL	27,339	3.46	0.88	0.0885	0.0226	2,420	$2,339	$2,854	$21,317,014
GA	24,796	2.17	0.55	0.0556	0.0140	1,378	$2,534	$2,760	$12,132,565
IA	8,325	1.44	0.15	0.0368	0.0038	306	$1,024	$956	$2,696,242
IL	29,953	1.50	0.18	0.0384	0.0045	1,149	$1,028	$1,014	$10,121,039
IN	14,665	1.74	0.26	0.0445	0.0067	652	$1,198	$1,138	$5,746,041
KS	9,222	3.16	0.97	0.0809	0.0249	746	$2,896	$2,729	$6,573,870
KY	10,303	1.72	0.32	0.0441	0.0082	454	$1,267	$915	$4,001,455
LA	10,857	2.30	0.62	0.0589	0.0158	640	$1,606	$1,218	$5,634,355
MA	5,867	3.44	0.25	0.0879	0.0064	516	$899	$581	$4,544,090
MD	6,533	1.32	0.21	0.0337	0.0053	220	$527	$424	$1,937,548
ME	2,137	4.47	0.54	0.1144	0.0137	245	$1,885	$1,049	$2,154,061
MI	17,539	2.04	0.26	0.0522	0.0067	915	$954	$749	$8,061,103
MN	25,696	1.67	0.24	0.0427	0.0060	1,096	$2,366	$2,064	$9,656,733
MO	17,805	2.15	0.96	0.0551	0.0245	980	$1,902	$1,838	$8,634,048
MS	7,974	2.38	0.47	0.0609	0.0120	486	$1,992	$1,501	$4,278,349
NC	16,264	2.22	0.59	0.0569	0.0150	925	$1,546	$1,474	$8,149,444

(continued)

Table 10-15. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2020 with CAIR (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	2,005	2.02	0.50	0.0516	0.0128	103	$1,452	$1,261	$910,706
NE	5,829	1.61	0.28	0.0413	0.0072	241	$1,332	$875	$2,118,974
NH	2,023	3.83	0.26	0.0980	0.0067	198	$1,998	$1,444	$1,745,904
NJ	7,554	2.82	0.28	0.0721	0.0072	545	$737	$556	$4,797,708
NY	17,537	2.71	0.46	0.0694	0.0117	1,216	$710	$653	$10,713,538
OH	21,660	1.94	0.32	0.0496	0.0083	1,074	$1,011	$786	$9,458,550
OK	14,132	2.32	0.37	0.0595	0.0095	841	$2,552	$1,842	$7,402,873
PA	14,621	2.64	1.43	0.0675	0.0366	986	$836	$754	$8,687,193
RI	807	3.25	0.09	0.0830	0.0022	67	$719	$455	$590,322
SC	9,634	2.66	0.60	0.0681	0.0155	657	$2,023	$1,812	$5,782,498
SD	2,587	1.85	0.99	0.0475	0.0252	123	$1,616	$1,918	$1,081,144
TN	14,372	1.89	0.26	0.0483	0.0066	694	$1,523	$1,302	$6,113,673
TX	73,600	1.83	0.43	0.0469	0.0111	3,450	$2,117	$2,118	$30,388,891
VA	13,318	1.60	0.47	0.0410	0.0120	546	$1,012	$917	$4,805,964
VT	1,156	3.15	0.52	0.0806	0.0133	93	$1,549	$936	$820,772
WI	16,324	1.99	0.33	0.0509	0.0085	831	$1,667	$1,186	$7,316,669
WV	3,261	1.84	0.26	0.0472	0.0068	154	$853	$515	$1,354,724

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table. For comparison purposes with the Base Cases in 2001, benefits presented in this table do not incorporate potential lags in fish tissue response to a change in mercury deposition.

entire study area.[20] The states with the highest estimated average rates continued to be primarily in New England. Average IQ decrements in prenatally exposed children were estimated to be less than 0.06 points for the study area. Under the 2020 conditions with CAIR, the total IQ losses associated with prenatal exposures to self-caught freshwater fish consumption in 2020 were estimated to be 26,400 IQ points and the present value in 2020 of foregone net earnings associated with these IQ decrements was estimated to sum to almost $233 million (in 1999 dollars).

Tables 10-16, 10-17, and 10-18 report estimated *reductions* in exposures, IQ decrements, and net earnings losses associated with CAIR and the two utility emissions zero-out scenarios. Based on the response times from the case studies discussed in Section 3[21], to reflect the possibility that reductions in mercury emissions in 2020 would have lagged effects on fish tissue concentrations, we present a range of benefits based on the 10 and 20 year lags as central estimates. We also provide results for the 5 and 50 year lags to demonstrate how benefits would differ under shorter and longer lag periods. Differences in annual benefit estimates across the selected lag periods are attributable to two factors. The first factor is differences in the size of the exposed population, due to the inclusion of projected population growth factors across time. For example, from 2001 to 2021, the exposed population was estimated to grow by 12 percent to over 486,000. The second factor is differences in the present value of IQ-related losses. To make the estimates of lost future earnings comparable across the four lag periods, they were all expressed as present values in 2001 or 2020; therefore, losses to children born in future years (2006, 2011, 2021, and 2051) were discounted assuming a 3 percent discount rate[22].

Table 10-16 reports results for the 2020 with CAIR scenario, by comparing them to conditions with (1) 2001 Base Case mercury levels in fish and (2) exposed population levels estimated to occur in the year of the selected lag periods. The per capita IQ decrements avoided were estimated to decrease by 0.007 points (12 percent). Total IQ decrements avoided are estimated to decrease by approximately 3,900 to 4,200 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with CAIR are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately $20.5 to $25.3 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 3,500 IQ points at a value of $28.1 million under a 5 year lag and 5,200 IQ points at a value of $10.5 million under a 50 year lag.

[20] The average daily mercury ingestion rate given here of 2.14 ug/day from freshwater fish is below the EPA's Reference Dose (RfD) for mercury of 5.8 ug/day. This estimate does not account for total exposure from consumption from other rifhs sources. See Section 11 of this report for a detailed discussion of the implication of the RfD on this analysis.

[21] Case studies of individual ecosystems (as presented in Section 3) 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.

[22] Due to the large number of tables and calculations provided in this section, the implications on results of using a 7 percent discount rate are presented in the summary tables for each approach analyzed.

Table 10-17 reports results for the 2001 Zero-Out relative to the 2001 Base Case. Under this scenario, the average daily mercury ingestion rate for pregnant women in freshwater angler households was estimated to decrease by roughly 13 percent for the entire study area. The average per capita IQ decrements in prenatally exposed children were also estimated to decline by roughly 13 percent under the zero out scenario, decreasing by an average of 0.008 points over the entire study area. As shown in Table 10-17, the states that were estimated to benefit from the largest per capita reductions in IQ losses are Pennsylvania and West Virginia (0.02 points in each state). Total IQ decrements avoided are estimated to decrease by approximately 3,700 to 3,900 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with the 2001 zero out of utility emissions are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately \$19.0 to \$24.0 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 3,600 IQ points at a value of \$27.5 million under a 5 year lag and 5,000 IQ points at a value of \$10.0 million under a 50 year lag.

Table 10-18 reports comparable results for the 2020 Utility Emissions Zero-Out relative to the 2020 Base Case with CAIR. The average daily mercury ingestion rate for pregnant women in freshwater angler households and average per capita IQ decrements in prenatally exposed children were estimated to decrease by roughly 5 percent for the entire study area. Total IQ decrements avoided are estimated to decrease by approximately 1,500 to 1,600 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with the zero out of emission remaining after 2020 with CAIR are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately \$8.0 to \$10.0 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 1,500 IQ points at a value of \$11.0 million under a 5 year lag and 2,000 IQ points at a value of \$4.1 million under a 50 year lag.

Table 10-16. 2020 Base Case with CAIR: Modelled Avoided Losses Relative to 2001 Base Case (Applied to 2020 Demographics)—Population Centroid Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0073	3,866	4,209	3,695	5,238	$25,335,445	$20,524,068	$28,068,173	$10,522,314
State									
AL	0.0078	80	83	78	94	$522,582	$406,589	$591,997	$189,434
AR	0.0047	57	62	54	79	$372,468	$304,097	$410,802	$158,587
CT	0.0114	57	64	54	84	$373,838	$310,480	$408,213	$167,847
DC	0.0085	2	2	2	3	$12,639	$10,569	$13,744	$5,794
DE	0.0069	7	7	6	9	$43,569	$34,696	$48,735	$17,108
FL	0.0071	226	260	209	361	$1,481,167	$1,266,474	$1,589,056	$724,896
GA	0.0104	299	344	276	481	$1,959,059	$1,679,796	$2,098,096	$966,526
IA	0.0028	24	25	24	27	$157,468	$120,449	$179,995	$53,675
IL	0.0049	158	170	151	207	$1,032,355	$828,493	$1,149,790	$415,887
IN	0.0080	127	136	122	164	$830,840	$663,683	$927,759	$329,616
KS	0.0041	41	45	39	58	$269,820	$221,159	$296,914	$116,312
KY	0.0112	121	127	118	145	$793,923	$619,390	$898,068	$290,574
LA	0.0049	54	55	54	59	$355,039	$269,529	$407,422	$117,651
MA	0.0105	68	75	65	95	$447,789	$365,837	$493,684	$191,062
MD	0.0068	49	54	47	68	$322,312	$262,666	$355,859	$136,439
ME	0.0095	20	20	20	20	$132,502	$98,398	$153,759	$40,296
MI	0.0093	173	182	168	209	$1,130,786	$886,044	$1,276,121	$420,204
MN	0.0020	55	60	53	74	$363,410	$292,938	$403,744	$148,529
MO	0.0038	73	79	70	96	$480,351	$385,610	$534,904	$193,700
MS	0.0063	53	56	51	64	$346,536	$271,555	$391,058	$128,809
NC	0.0125	238	274	220	380	$1,558,876	$1,333,737	$1,671,787	$764,276

(continued)

Table 10-16. 2020 Base Case with CAIR: Modelled Avoided Losses Relative to 2001 Base Case Applied to 2020 Demographics—Population Centroid Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0049	10	10	10	11	$64,360	$49,733	$73,175	$22,767
NE	0.0028	18	20	17	25	$118,046	$96,254	$130,292	$50,057
NH	0.0105	23	25	22	32	$152,746	$123,654	$169,288	$63,299
NJ	0.0067	56	62	53	80	$369,285	$303,096	$406,047	$159,866
NY	0.0086	162	171	157	199	$1,062,160	$835,940	$1,195,819	$400,747
OH	0.0152	344	361	336	413	$2,256,841	$1,762,498	$2,551,488	$828,953
OK	0.0042	64	69	61	84	$419,008	$336,105	$466,796	$168,535
PA	0.0168	258	267	253	296	$1,689,870	$1,304,303	$1,922,502	$595,299
RI	0.0107	10	10	9	13	$62,288	$50,737	$68,790	$26,328
SC	0.0110	115	123	111	148	$752,632	$600,633	$840,877	$297,638
SD	0.0037	10	10	10	12	$65,022	$50,419	$73,792	$23,289
TN	0.0097	154	169	146	214	$1,007,148	$823,356	$1,109,960	$430,603
TX	0.0045	385	439	358	599	$2,522,650	$2,138,657	$2,720,686	$1,204,386
VA	0.0080	118	129	112	163	$774,228	$631,324	$854,523	$328,354
VT	0.0102	12	12	12	13	$79,133	$59,908	$90,938	$25,949
WI	0.0047	81	85	79	97	$533,092	$415,898	$603,022	$195,109
WV	0.0200	64	63	64	62	$419,609	$309,364	$488,673	$123,914

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Per capita IQ decrements and mercury ingestion rate vary only very slightly across different lag periods. Therefore, for brevity sake, we report the results for the 10 year lag period case.

Table 10-17. 2001 Utility Mercury Emissions Zero Out: Modelled Avoided Losses Relative to 2001 Base Case—Population Centroid Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2001; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0081	3,674	3,891	3,623	4,960	$24,080,244	$18,973,876	$27,522,949	$9,963,920
State									
AL	0.0077	76	77	76	87	$498,871	$373,165	$580,312	$174,977
AR	0.0056	57	61	56	80	$374,531	$296,529	$426,616	$159,757
CT	0.0076	32	34	31	47	$207,370	$166,027	$238,981	$94,933
DC	0.0088	2	2	2	3	$12,861	$9,006	$15,185	$5,024
DE	0.0091	8	8	8	10	$51,035	$40,007	$58,919	$20,280
FL	0.0044	105	122	100	184	$690,798	$595,815	$756,618	$370,192
GA	0.0097	216	247	206	376	$1,417,902	$1,205,174	$1,566,735	$754,952
IA	0.0038	31	32	32	34	$205,618	$154,912	$240,401	$68,796
IL	0.0081	232	247	228	312	$1,520,561	$1,204,058	$1,728,824	$626,223
IN	0.0092	129	137	127	167	$848,492	$666,643	$966,985	$335,634
KS	0.0066	57	62	55	84	$372,660	$302,207	$420,109	$169,416
KY	0.0109	111	113	112	129	$724,909	$548,625	$847,860	$259,550
LA	0.0043	45	46	45	49	$296,048	$225,716	$344,048	$98,620
MA	0.0078	44	46	44	61	$290,190	$225,540	$337,552	$122,917
MD	0.0077	47	51	47	67	$308,041	$248,152	$353,594	$134,345
ME	0.0057	13	12	13	12	$82,477	$59,203	$99,791	$24,305
MI	0.0114	196	202	196	236	$1,287,068	$983,452	$1,492,774	$474,548
MN	0.0029	70	77	68	98	$457,894	$373,857	$518,260	$196,272
MO	0.0063	106	112	106	141	$697,528	$546,921	$803,148	$282,823
MS	0.0058	46	47	46	54	$302,039	$227,761	$348,550	$108,785
NC	0.0125	183	206	176	310	$1,201,286	$1,002,879	$1,333,984	$623,774

(continued)

Table 10-17. 2001 Utility Mercury Emissions Zero Out: Modelled Avoided Losses Relative to 2001 Base Case—Population Centroid Approach (continued)

	Total Avoided IQ Decrements					Total Avoided Net Earnings Losses (Present Value in 2001; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0031	6	6	6	6	$40,482	$29,829	$47,850	$12,925
NE	0.0042	23	25	22	32	$149,297	$121,372	$168,120	$64,976
NH	0.0073	15	15	15	19	$95,231	$72,725	$111,107	$38,443
NJ	0.0111	78	85	77	114	$512,870	$415,666	$586,162	$228,680
NY	0.0108	184	190	184	226	$1,207,790	$926,866	$1,399,430	$454,284
OH	0.0159	338	342	339	395	$2,214,893	$1,666,293	$2,577,917	$793,664
OK	0.0046	61	66	60	82	$398,767	$319,584	$455,579	$164,099
PA	0.0203	300	294	306	328	$1,965,224	$1,435,975	$2,326,920	$658,018
RI	0.0086	7	7	7	9	$44,208	$34,111	$50,576	$18,333
SC	0.0117	109	113	108	139	$712,105	$550,761	$817,218	$279,334
SD	0.0037	9	9	9	11	$59,949	$46,251	$69,514	$21,198
TN	0.0095	130	138	129	182	$852,552	$672,804	$978,646	$366,004
TX	0.0043	279	324	259	473	$1,828,979	$1,577,641	$1,968,963	$950,221
VA	0.0113	140	151	137	200	$918,053	$736,190	$1,043,174	$401,475
VT	0.0069	8	8	9	8	$55,273	$38,890	$66,685	$16,971
WI	0.0069	109	113	109	130	$712,244	$550,450	$827,480	$260,986
WV	0.0205	71	66	72	64	$464,152	$322,820	$548,360	$128,187

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Per capita IQ decrements and mercury ingestion rate vary only very slightly across different lag periods. Therefore, for brevity sake, we report the results for the 10 year lag period case.

Table 10-18. 2020 with CAIR Emissions Zero Out: Modelled Avoided Losses Relative to 2020 with CAIR Base Case—Population Centroid Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0029	1,520	1,651	1,455	2,043	$9,962,464	$8,050,132	$11,052,917	$4,104,007
State									
AL	0.0018	18	19	18	22	$120,628	$93,754	$136,729	$43,564
AR	0.0032	38	41	36	52	$247,726	$202,009	$273,412	$105,073
CT	0.0023	11	13	11	17	$74,045	$61,506	$80,846	$33,261
DC	0.0025	1	1	1	1	$3,735	$3,123	$4,062	$1,712
DE	0.0039	4	4	4	5	$24,599	$19,586	$27,519	$9,653
FL	0.0012	39	45	36	62	$254,611	$217,598	$273,240	$124,432
GA	0.0024	68	78	63	108	$445,245	$379,786	$478,394	$216,388
IA	0.0026	22	23	22	24	$144,142	$109,892	$165,047	$48,533
IL	0.0039	125	135	120	166	$820,256	$660,566	$911,783	$334,213
IN	0.0035	55	58	53	68	$357,823	$282,024	$402,531	$135,681
KS	0.0064	65	72	61	96	$422,799	$353,482	$459,854	$193,683
KY	0.0023	25	26	24	30	$163,530	$127,946	$184,696	$60,455
LA	0.0016	17	18	17	19	$113,423	$86,041	$130,207	$37,480
MA	0.0027	17	19	16	24	$113,281	$92,592	$124,858	$48,405
MD	0.0023	17	19	16	23	$110,453	$90,289	$121,735	$47,211
ME	0.0033	7	7	7	7	$46,197	$34,351	$53,574	$14,122
MI	0.0054	101	107	98	124	$664,343	$522,489	$748,223	$250,057
MN	0.0015	43	47	41	58	$282,405	$228,469	$313,104	$116,785
MO	0.0050	95	104	91	130	$621,907	$505,203	$687,897	$260,597
MS	0.0019	16	17	15	19	$103,267	$80,568	$116,811	$37,801
NC	0.0030	58	66	53	92	$378,000	$322,827	$405,830	$184,367

(continued)

Table 10-18. 2020 with CAIR Emissions Zero Out: Modelled Avoided Losses Relative to 2020 with CAIR Base Case—Population Centroid Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0015	3	3	3	3	$20,371	$15,544	$23,314	$6,882
NE	0.0032	20	22	19	28	$133,935	$108,936	$148,042	$56,345
NH	0.0031	7	7	7	9	$44,812	$36,300	$49,647	$18,608
NJ	0.0041	34	37	32	48	$222,831	$182,732	$245,137	$96,203
NY	0.0032	60	64	58	75	$391,969	$309,697	$440,351	$149,881
OH	0.0037	84	88	82	101	$549,932	$430,087	$621,252	$203,004
OK	0.0027	41	44	39	55	$266,732	$215,406	$296,026	$109,671
PA	0.0053	81	85	80	96	$533,357	$414,507	$604,567	$192,572
RI	0.0026	2	3	2	3	$15,007	$12,219	$16,577	$6,335
SC	0.0036	37	40	36	48	$243,906	$194,560	$272,572	$96,312
SD	0.0020	5	6	5	6	$34,762	$26,989	$39,424	$12,507
TN	0.0021	34	37	32	47	$222,943	$181,794	$246,064	$94,553
TX	0.0014	119	135	111	184	$782,514	$660,448	$846,244	$368,728
VA	0.0051	75	83	71	106	$491,642	$403,406	$540,675	$212,643
VT	0.0021	2	2	2	3	$16,030	$12,151	$18,410	$5,282
WI	0.0037	64	67	62	76	$417,403	$325,491	$472,274	$152,519
WV	0.0030	9	9	9	9	$61,899	$45,764	$71,987	$18,490

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Per capita IQ decrements and mercury ingestion rate vary only very slightly across different lag periods. Therefore, for brevity sake, we report the results for the 10 year lag period case.

Table 10-19 report similar calculations for the 2020 CAMR Option 1. Relative to the 2020 Base Case with CAIR, this option is estimated to reduce per capita IQ decrements across the study area by 0.0006 points. Total IQ decrements avoided under Option 1 are estimated to decrease by approximately 318 to 346 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with CAMR Option 1 are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately $1.7 to $2.0 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 300 IQ points at a value of $2.3 million under a 5 year lag and 350 IQ points at a value of $0.8 million under a 50 year lag.

Table 10-20 report results of the 2020 CAMR Option 2. Relative to the 2020 Base Case with CAIR, this option is estimated to reduce per capita IQ decrements across the study area by 0.0009 points. Total IQ decrements avoided under Option 2 are estimated to decrease by approximately 475 to 520 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with CAMR Option 2 are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately $2.5 to $3.1 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 450 IQ points at a value of $3.4 million under a 5 year lag and 650 IQ points at a value of $1.3 million under a 50 year lag.

Table 10-21 summarizes the annual benefit estimates for CAMR Control Options 1 and 2, and it compares them to aggregate estimates associated with CAIR emissions reductions and the two Utility Emissions Zero-Out scenarios (in 2001 and 2020). To assess the sensitivity of these results to the assumed rate at which future gains/losses are discounted, the benefit estimates are also reported assuming 3 and 7 percent discount rates. In addition, we present a 1 percent discount rate for the 50 year lag period to reflect the discount rate for inter-generational effects as is recommended in the EPA Guidelines for Economic Analysis.

In comparison to the other mercury emissions reductions, the estimated benefits of Option 1 are roughly 21 percent of the estimated benefits that would be achieved by eliminating utility mercury emissions (i.e., 2020 Utility Emissions Zero-Out). The estimated benefits of Option 2 are approximately 31 percent as large as those for the 2020 Utility Emissions Zero-Out scenario.

10.5.2 Results for the Angler Destination Approach

This section summarizes results from applying the angler destination approach. As reported in Table 10-3, many of the HUCs in the study area do not contain lake and/or river mercury sampling locations. As was done in the population centroid approach, a simple spatial extrapolation method was used to address these data limitations. For HUCs that do not contain a lake (river) sample, they were assigned the lake (river) mean from the entire study area (e.g., 0.23 ppm for lakes and 0.25 ppm for rivers in the 2001 Base Case).

Table 10-19. Estimated Benefits of 2020 CAMR Control Option 1: Relative to 2020 with CAIR—Population Centroid Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0006	318	346	304	430	$2,086,359	$1,687,988	$2,313,079	$862,951
State									
AL	0.0002	3	3	3	3	$16,779	$13,010	$19,042	$6,009
AR	0.0013	15	16	14	20	$98,522	$79,453	$109,429	$40,326
CT	0.0006	3	3	3	4	$19,128	$15,904	$20,872	$8,617
DC	0.0005	0	0	0	0	$692	$579	$752	$317
DE	0.0014	1	1	1	2	$8,852	$7,046	$9,904	$3,470
FL	0.0002	6	7	6	9	$39,199	$33,373	$42,166	$18,946
GA	0.0003	8	9	8	13	$53,256	$45,606	$57,081	$26,179
IA	0.0004	3	3	3	4	$21,968	$16,694	$25,196	$7,307
IL	0.0005	18	19	17	24	$115,700	$93,378	$128,451	$47,477
IN	0.0004	7	7	6	8	$43,038	$33,644	$48,632	$15,862
KS	0.0012	12	14	12	19	$81,222	$68,174	$88,131	$37,650
KY	0.0003	3	3	3	3	$18,766	$14,509	$21,330	$6,652
LA	0.0003	4	4	4	4	$23,896	$18,081	$27,468	$7,820
MA	0.0006	4	5	4	6	$27,173	$22,211	$29,949	$11,612
MD	0.0008	6	6	5	8	$37,122	$30,494	$40,798	$16,113
ME	0.0013	3	3	3	3	$18,016	$13,390	$20,898	$5,497
MI	0.0004	8	8	8	10	$52,581	$41,328	$59,240	$19,750
MN	0.0002	4	5	4	6	$29,069	$23,439	$32,289	$11,893
MO	0.0009	18	19	17	25	$116,271	$94,896	$128,264	$49,451
MS	0.0004	4	4	3	4	$23,119	$17,953	$26,217	$8,323
NC	0.0005	9	11	9	15	$60,245	$51,447	$64,684	$29,376

(continued)

Table 10-19. Estimated Benefits of 2020 CAMR Control Option 1: Relative to 2020 with CAIR—Population Centroid Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0004	1	1	1	1	$5,528	$4,219	$6,326	$1,869
NE	0.0007	5	5	4	6	$29,969	$24,408	$33,099	$12,662
NH	0.0008	2	2	2	2	$11,734	$9,488	$13,013	$4,844
NJ	0.0016	14	15	13	19	$89,843	$73,692	$98,824	$38,814
NY	0.0010	19	21	19	25	$127,759	$101,285	$143,262	$49,417
OH	0.0003	6	6	6	7	$38,067	$29,742	$43,026	$14,004
OK	0.0004	5	6	5	7	$35,259	$28,314	$39,256	$14,233
PA	0.0021	33	34	32	39	$213,167	$166,443	$241,023	$78,245
RI	0.0006	1	1	0	1	$3,351	$2,728	$3,701	$1,414
SC	0.0003	3	3	3	4	$20,552	$16,486	$22,896	$8,267
SD	0.0004	1	1	1	1	$7,649	$5,945	$8,669	$2,763
TN	0.0004	7	7	6	9	$43,676	$35,203	$48,527	$17,845
TX	0.0003	29	33	27	43	$190,955	$159,347	$207,925	$86,980
VA	0.0031	46	51	44	66	$304,403	$250,091	$334,512	$132,187
VT	0.0005	1	1	1	1	$4,145	$3,146	$4,757	$1,373
WI	0.0003	6	6	6	7	$38,029	$29,668	$43,019	$13,917
WV	0.0008	3	3	3	3	$17,661	$13,174	$20,448	$5,468

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Per capita IQ decrements and mercury ingestion rate vary only very slightly across different lag periods. Therefore, for brevity sake, we report the results for the 10 year lag period case.

Table 10-20. Estimated Benefits of 2020 CAMR Control Option 2: Relative to 2020 with CAIR—Population Centroid Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0009	475	518	453	648	$3,112,816	$2,527,403	$3,444,104	$1,302,261
State									
AL	0.0005	5	5	5	6	$33,043	$25,636	$37,489	$11,857
AR	0.0017	20	22	19	28	$133,134	$108,432	$147,041	$56,252
CT	0.0007	4	4	3	5	$23,456	$19,500	$25,598	$10,564
DC	0.0006	0	0	0	0	$927	$775	$1,008	$425
DE	0.0016	2	2	1	2	$9,979	$7,943	$11,165	$3,913
FL	0.0003	11	12	10	17	$68,824	$58,661	$73,983	$33,374
GA	0.0009	27	31	25	42	$176,240	$150,021	$189,602	$85,143
IA	0.0007	6	6	6	7	$40,034	$30,470	$45,881	$13,394
IL	0.0008	27	30	26	37	$179,218	$144,755	$198,881	$73,728
IN	0.0006	10	10	9	12	$62,282	$48,840	$70,258	$23,206
KS	0.0027	27	30	26	40	$178,850	$148,340	$195,450	$79,973
KY	0.0004	5	5	5	5	$30,227	$23,483	$34,270	$10,899
LA	0.0005	5	5	5	6	$34,511	$26,132	$39,654	$11,327
MA	0.0008	5	6	5	7	$33,380	$27,285	$36,790	$14,266
MD	0.0009	7	7	6	9	$43,528	$35,723	$47,864	$18,838
ME	0.0016	3	3	3	3	$22,357	$16,613	$25,936	$6,816
MI	0.0007	13	13	12	15	$82,012	$64,560	$92,320	$30,968
MN	0.0003	9	10	9	13	$62,124	$50,114	$68,989	$25,453
MO	0.0016	30	33	29	42	$196,787	$161,225	$216,604	$84,711
MS	0.0006	5	5	5	6	$34,120	$26,506	$38,683	$12,301
NC	0.0007	13	15	12	20	$83,460	$71,474	$89,452	$41,030

(continued)

Table 10-20. Estimated Benefits of 2020 With CAIR Control Option 2: Relative to 2020 with CAIR—Population Centroid Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0005	1	1	1	1	$6,901	$5,270	$7,896	$2,337
NE	0.0015	10	10	9	13	$62,393	$50,688	$69,011	$26,150
NH	0.0010	2	2	2	3	$14,166	$11,457	$15,710	$5,852
NJ	0.0018	15	17	14	21	$99,911	$81,948	$109,901	$43,160
NY	0.0012	23	24	22	29	$149,088	$118,099	$167,255	$57,508
OH	0.0001	1	1	1	2	$8,284	$6,667	$9,212	$3,368
OK	0.0014	21	23	20	28	$136,825	$110,719	$151,679	$56,625
PA	0.0023	35	37	34	42	$228,368	$178,561	$258,016	$84,236
RI	0.0007	1	1	1	1	$4,147	$3,377	$4,581	$1,751
SC	0.0008	8	9	8	10	$52,469	$41,957	$58,555	$20,888
SD	0.0006	2	2	2	2	$10,937	$8,498	$12,399	$3,946
TN	0.0006	10	11	10	14	$67,360	$54,560	$74,632	$27,962
TX	0.0005	44	49	41	66	$285,177	$239,171	$309,587	$131,872
VA	0.0033	49	54	46	69	$321,251	$263,882	$353,067	$139,419
VT	0.0007	1	1	1	1	$5,233	$3,973	$6,005	$1,734
WI	0.0010	17	18	17	20	$110,883	$86,510	$125,426	$40,588
WV	0.0010	3	3	3	3	$20,928	$15,579	$24,255	$6,427

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Per capita IQ decrements and mercury ingestion rate vary only very slightly across different lag periods. Therefore, for brevity sake, we report the results for the 10 year lag period case.

Table 10-21. Summary of Annual Benefit Estimates: Population Centroid Approach[a]

	Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	10-Yr Lag	**20-Yr Lag**	**5-Yr Lag**	**50-Yr Lag**
Annual Number of Prenatally Exposed Children				
2001 Base Case	452,575	486,487	442,938	632,017
2020 Base Case (with CAIR)	528,721	577,910	504,127	725,474
Total Value of Benefits (1999$s)				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)				
1% Discount Rate; Present Value in 2001	N/A	N/A	N/A	$26,559,677
3% Discount Rate; Present Value in 2001	$24,080,244	$18,973,876	$27,522,949	$9,963,920
7% Discount Rate; Present Value in 2001	$16,451,115	$8,855,743	$22,748,995	$1,482,868
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$28,048,124
3% Discount Rate; Present Value in 2020	$25,335,445	$20,524,068	$28,068,173	$10,522,314
7% Discount Rate; Present Value in 2020	$17,308,643	$9,579,269	$23,199,647	$1,565,971
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$10,939,580
3% Discount Rate; Present Value in 2020	$9,962,464	$8,050,132	$11,052,917	$4,104,007
7% Discount Rate; Present Value in 2020	$6,806,145	$3,757,266	$9,135,749	$610,774
BENEFITS OF CONTROL OPTIONS				
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$2,300,269
3% Discount Rate; Present Value in 2020	$2,086,359	$1,687,988	$2,313,079	$862,951
7% Discount Rate; Present Value in 2020	$1,425,357	$787,840	$1,911,867	$128,428
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$3,471,289
3% Discount Rate; Present Value in 2020	$3,112,816	$2,527,403	$3,444,104	$1,302,261
7% Discount Rate; Present Value in 2020	$2,126,610	$1,179,624	$2,846,712	$193,807

[a] Case studies of individual ecosystems (as presented in Section 3) 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.

Figure 10-8 displays the estimated spatial distribution of 2001 baseline average mercury ingestion rates across HUCs in the study area. Maine, Pennsylvania, Massachusetts, and New Hampshire are the main states with high levels for average mercury ingestion rates. Given that the fish consumption rate is constant across the study area, the higher mercury ingestion rate in these states is primarily due to the relatively high average fish tissue mercury concentrations in these states (as evident from Table 10-2). Figure 10-9 displays the 2001 baseline distribution of total IQ point decrements across HUCs in the study area. States like New York, Texas, Florida, Pennsylvania, and Ohio have higher total IQ decrements because of higher mercury concentration and higher number of anglers together. The distribution of estimated percentage reductions in per capita IQ decrements associated with the 2001 Utility Emissions Zero-Out scenario are displayed in Figure 10-10. Higher percentage reductions from the baseline risk levels are mostly in Pennsylvania, Ohio, Virginia and West Virginia.

Table 10-22 summarizes model results for the 2001 Base Case, based on exposure estimates from the angler destination approach in 2001. The estimated annual number of prenatally exposed children in freshwater angler households is close to 587,000. The states with the largest estimated exposed populations are Texas, Minnesota and Illinois. The average daily mercury ingestion rate for pregnant women in freshwater angler households (HgI) was estimated to be 2.7 μg/day for the entire study area. The states with the highest average rates were primarily in the Northeast (except for Florida). For example, the average rates for Maine, Pennsylvania and New Hampshire were all estimated to be above 4 μg/day under baseline conditions.[23] Average IQ decrements in prenatally exposed children were estimated to be 0.07 points for the study area.

Under baseline conditions in 2001, the total IQ losses associated with self-caught freshwater fish consumption were estimated to be 40,200 IQ points and the present value in 2001 of foregone earnings associated with these IQ decrements was estimated to sum to $354 million (in 1999 dollars).

[23] The average daily mercury ingestion rate given here of 2.7 ug/day from freshwater fish is below the EPA's Reference Dose (RfD) for mercury of 5.8 ug/day. This estimate does not account for total exposure from consumption from other rifhs sources. See Section 11 of this report for a detailed discussion of the implication of the RfD on this analysis.

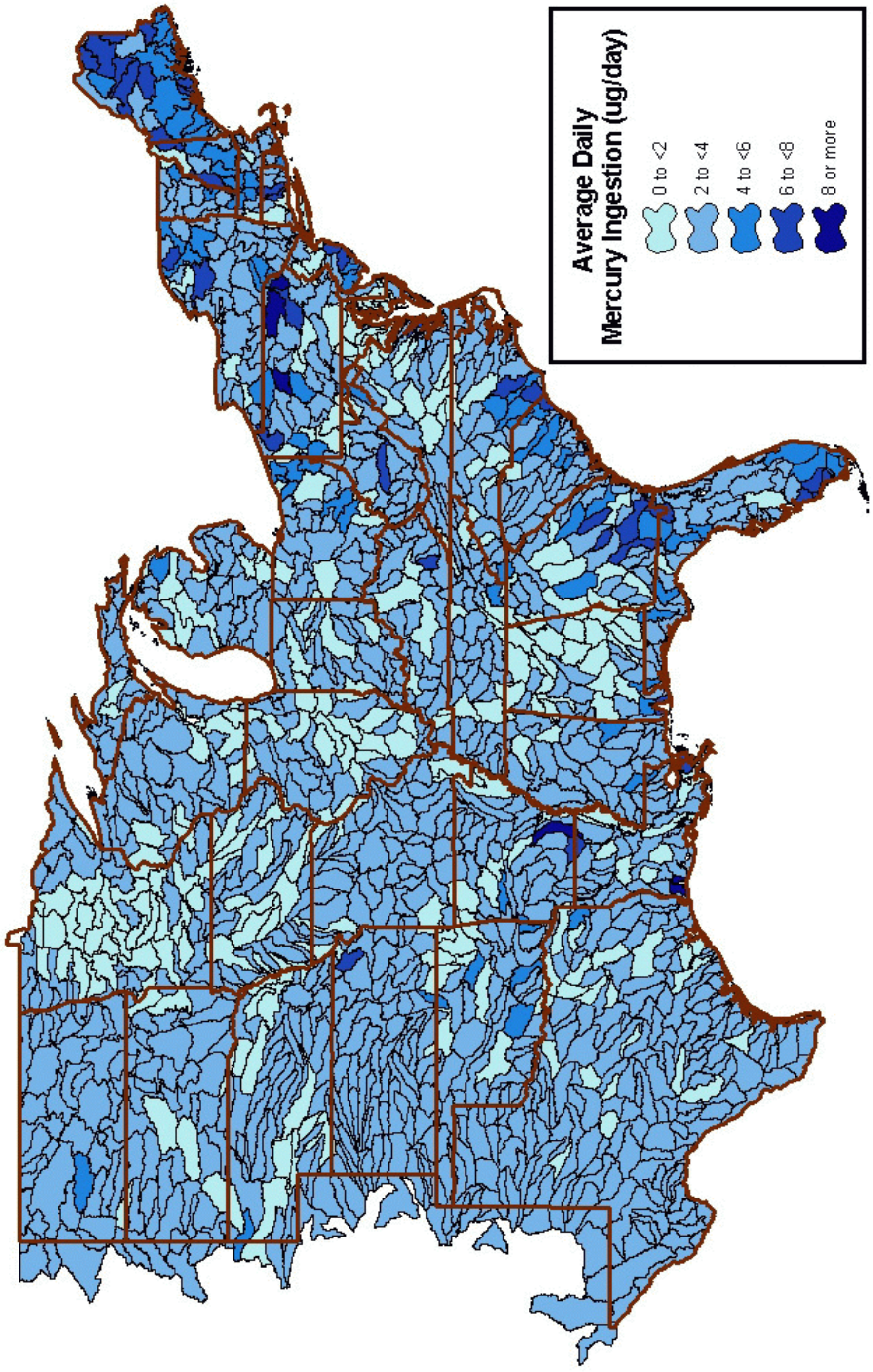

Figure 10-8. Spatial Distribution of Estimated Average Daily Maternal Mercury Ingestion Rates: Angler Destination Approach—2001 Base Case

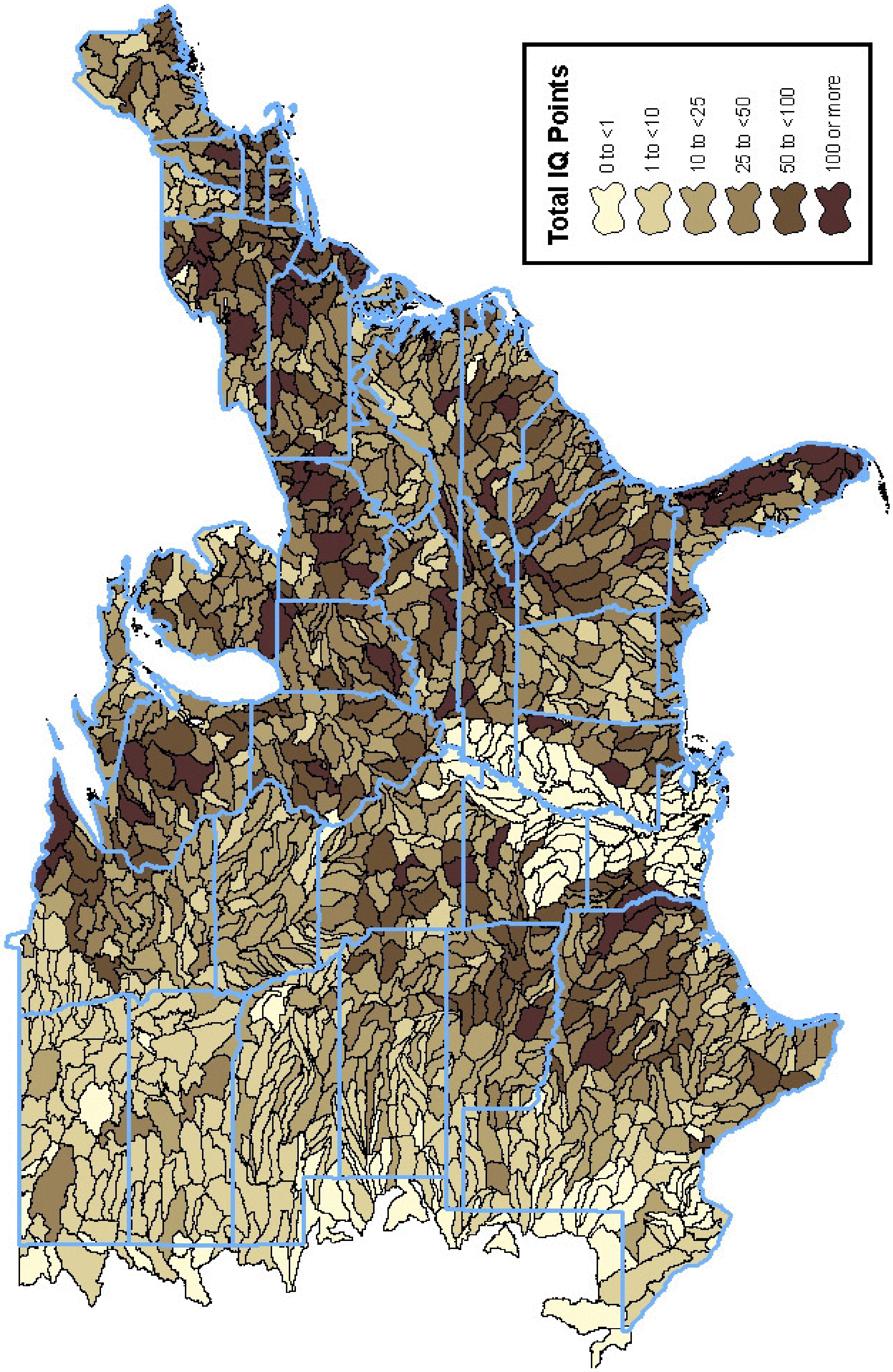

Figure 10-9. Spatial Distribution of Estimated IQ Decrements per HUC: Angler Destination Approach—2001 Base Case

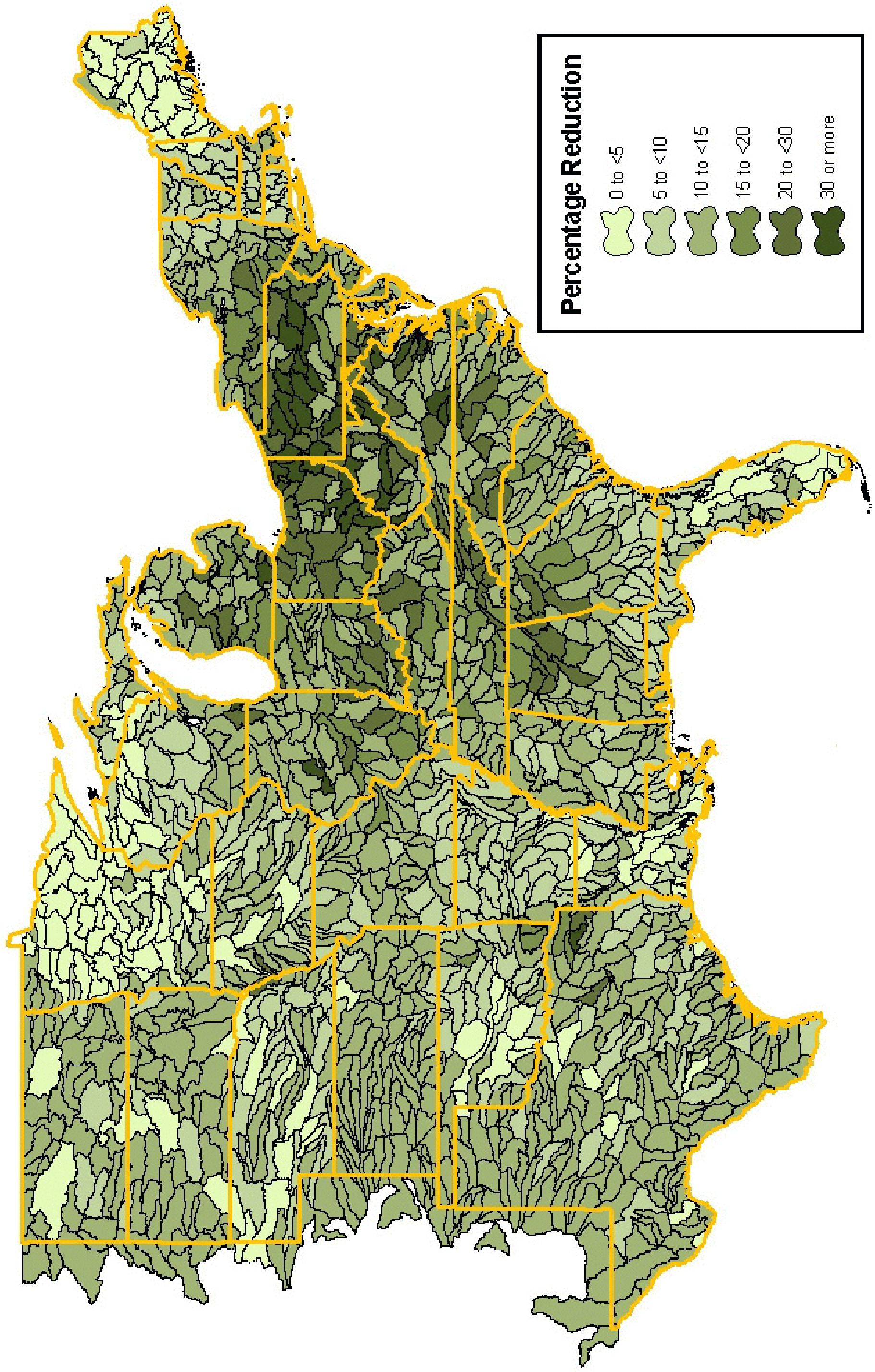

Figure 10-10. Spatial Distribution of Estimated Percent Reduction in IQ Losses: Improvement with 2001 Utility Emissions Zero Out Scenario (Zero Lag)

Table 10-22. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Angler Destination Approach—2001 Base Case[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	586,516	2.68	1.55	0.069	0.040	40,207	$604	$350	$354,113,271
State									
AL	13,077	1.70	1.31	0.043	0.034	569	$383	$296	$5,009,131
AR	14,669	2.32	1.01	0.059	0.026	872	$524	$229	$7,680,796
CT	4,762	2.95	1.34	0.075	0.034	359	$665	$302	$3,165,411
DC	8	1.08	0.06	0.028	0.002	0	$242	$13	$1,978
DE	592	2.06	0.60	0.053	0.015	31	$463	$136	$274,454
FL	22,748	3.56	1.41	0.091	0.036	2,074	$803	$318	$18,269,235
GA	21,530	2.91	1.58	0.074	0.040	1,603	$656	$355	$14,115,893
IA	12,069	1.95	0.72	0.050	0.018	601	$438	$162	$5,291,262
IL	33,696	1.84	0.96	0.047	0.025	1,586	$415	$217	$13,971,988
IN	23,517	2.38	0.78	0.061	0.020	1,430	$536	$177	$12,597,628
KS	11,929	2.90	1.03	0.074	0.026	885	$653	$233	$7,793,163
KY	15,840	2.39	0.60	0.061	0.015	969	$539	$134	$8,536,621
LA	12,491	2.47	0.93	0.063	0.024	790	$557	$210	$6,961,559
MA	6,808	3.45	1.08	0.088	0.028	602	$778	$243	$5,299,186
MD	7,318	1.32	1.00	0.034	0.026	247	$297	$225	$2,175,909
ME	2,977	5.54	1.39	0.142	0.035	422	$1,249	$312	$3,717,653
MI	20,998	2.72	1.16	0.070	0.030	1,462	$613	$262	$12,875,688
MN	36,700	2.03	0.71	0.052	0.018	1,903	$457	$161	$16,756,418
MO	19,720	2.64	0.53	0.068	0.013	1,332	$595	$119	$11,729,835
MS	12,417	2.84	0.57	0.073	0.014	901	$639	$128	$7,937,478
NC	19,451	2.79	1.02	0.071	0.026	1,389	$629	$229	$12,237,634

(continued)

Table 10-22. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Angler Destination Approach—2001 Base Case (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	2,684	2.78	0.37	0.071	0.009	191	$626	$83	$1,680,883
NE	5,477	2.26	0.79	0.058	0.020	317	$510	$178	$2,795,383
NH	2,805	4.45	1.33	0.114	0.034	319	$1,002	$301	$2,809,423
NJ	10,520	3.35	1.47	0.086	0.037	903	$756	$330	$7,952,715
NY	21,401	3.67	1.46	0.094	0.037	2,011	$828	$328	$17,712,372
OH	30,556	2.88	1.05	0.074	0.027	2,255	$650	$236	$19,859,741
OK	23,791	2.42	1.18	0.062	0.030	1,470	$544	$265	$12,950,097
PA	24,704	4.75	5.05	0.121	0.129	3,000	$1,069	$1,138	$26,420,063
RI	835	3.28	0.60	0.084	0.015	70	$740	$135	$618,220
SC	11,837	3.06	1.01	0.078	0.026	925	$689	$228	$8,150,271
SD	3,798	2.49	0.66	0.064	0.017	242	$561	$149	$2,131,921
TN	20,723	2.44	0.74	0.063	0.019	1,295	$550	$167	$11,407,980
TX	64,537	2.47	0.65	0.063	0.017	4,084	$557	$147	$35,973,218
VA	18,224	2.11	1.06	0.054	0.027	982	$474	$239	$8,647,351
VT	1,726	3.68	1.69	0.094	0.043	163	$830	$381	$1,431,637
WI	24,429	2.48	0.62	0.063	0.016	1,551	$559	$139	$13,660,658
WV	5,149	3.03	1.00	0.077	0.026	399	$682	$225	$3,512,420

[a] Benefits analyses using the angler destination approach were conducted at a HUC level, but the results are aggregated and reported at a state level in this table.

Table 10-23 summarizes model results for the 2020 Base Case with CAIR based on the exposure estimates from the angler destination approach. Due to population growth between 2001 and 2020, the estimated annual number of prenatally exposed children in freshwater angler households is approximately 675,000. The average daily mercury ingestion rate for pregnant women in freshwater angler households (HgI) was estimated to be roughly 13 percent below levels from the 2001 Base Case, at 2.33 μg/day for the entire study area. The states with the highest estimated average rates continued to be primarily in Northeast. Average IQ decrements in prenatally exposed children were estimated to be less than 0.06 points for the study area. Under the 2020 conditions with CAIR, the total IQ losses associated with self-caught freshwater fish consumption were estimated to be 40,200 and the present value in 2020 of foregone net earnings associated with these IQ decrements was estimated at $354 million (in 1999 dollars). These aggregate estimates are very similar to the 2001 Base Case results – the increase in exposed population from 2001 to 2020 and the reduction in mercury levels due to CAIR have roughly offsetting effects on the estimates of total IQ decrements.

Tables 10-24, 10-25, and 10-26 report estimated *reductions* in exposures, IQ decrements, and net earnings losses associated with CAIR and the two utility emissions zero-out scenarios. Based on the response times from the case studies discussed in Section 3[24], to reflect the possibility that reductions in mercury emissions in 2020 would have lagged effects on fish tissue concentrations, we present a range of benefits based on the 10 and 20 year lags as central estimates. We also provide results for the 5 and 50 year lags to demonstrate how benefits would differ under shorter and longer lag periods.

Table 10-24 reports results for the 2020 with CAIR scenario, by comparing them to conditions with (1) 2001 Base Case mercury levels in fish and (2) exposed population levels in 2020 to 2070. The per capita IQ decrements were estimated to decrease by 0.0089 points (13 percent). Total IQ decrements were estimated to decrease by 6,700 to 7,400 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with CAIR are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately $44.7 to $36 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 6,300 IQ points at a value of $48.1 million under a 5 year lag and 9,400 IQ points at a value of $19 million under a 50 year lag.

Table 10-25 reports results for the 2001 Zero-Out relative to the 2001 Base Case. The average per capita IQ decrements in prenatally exposed children were estimated to decline by roughly 13 percent under the zero out scenario, decreasing by an average of 0.009 points over the entire study area. As shown in Table 10-25 (and Figure 10-10), the states that were estimated to benefit from the largest per capita reductions in IQ losses are Pennsylvania, West Virginia, and Ohio (0.18 or more points in each state). Total IQ decrements avoided are estimated to decrease by approximately 5,600 to 6,200 points in 2020 under a 10 to 20 year lag

[24] Case studies of individual ecosystems (as presented in Section 3) 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.

period. The mercury emission reductions associated with the 2001 zero out of utility emissions are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately $30.0 to $37.0 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 5,500 IQ points at a value of $41.5 million under a 5 year lag and 8,200 IQ points at a value of $16.5 million under a 50 year lag.

Table 10-26 reports results for the 2020 Utility Emissions Zero-Out relative to the 2020 Base Case with CAIR. The average daily mercury ingestion rate for pregnant women in freshwater angler households and average per capita IQ decrements in prenatally exposed children were estimated to decrease by roughly 6 percent for the entire study area. Total IQ decrements avoided are estimated to decrease by approximately 2,400 to 2,700 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with CAIR are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately $13.1 to $16.0 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 2,300 IQ points at a value of $17.7 million under a 5 year lag and 3,500 IQ points at a value of $6.9 million under a 50 year lag.

Table 10-23. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Angler Destination Approach—2020 with CAIR[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (μg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2020; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	674,357	2.33	1.28	0.060	0.033	40,211	$525	$289	$354,146,357
State									
AL	15,035	1.44	1.19	0.037	0.030	554	$325	$268	$4,880,859
AR	16,866	2.17	0.92	0.055	0.024	935	$488	$208	$8,238,433
CT	5,475	2.52	1.28	0.064	0.033	353	$567	$288	$3,105,516
DC	9	0.68	0.07	0.017	0.002	0	$153	$15	$1,440
DE	681	1.80	0.54	0.046	0.014	31	$405	$121	$276,132
FL	26,155	3.25	1.40	0.083	0.036	2,178	$733	$316	$19,180,880
GA	24,755	2.52	1.46	0.064	0.037	1,595	$567	$329	$14,048,148
IA	13,876	1.77	0.62	0.045	0.016	629	$399	$139	$5,537,508
IL	38,743	1.59	0.81	0.041	0.021	1,573	$357	$182	$13,849,691
IN	27,039	2.03	0.70	0.052	0.018	1,407	$458	$157	$12,391,617
KS	13,716	2.62	1.02	0.067	0.026	920	$591	$229	$8,102,384
KY	18,213	2.03	0.55	0.052	0.014	947	$458	$125	$8,339,087
LA	14,362	2.28	0.84	0.058	0.021	838	$514	$189	$7,376,996
MA	7,828	3.00	0.94	0.077	0.024	602	$677	$211	$5,299,712
MD	8,414	1.06	0.79	0.027	0.020	227	$238	$177	$2,002,919
ME	3,423	5.23	1.36	0.134	0.035	458	$1,178	$306	$4,033,529
MI	24,143	2.46	1.11	0.063	0.028	1,520	$555	$250	$13,388,662
MN	42,196	1.92	0.70	0.049	0.018	2,069	$432	$158	$18,222,891
MO	22,673	2.39	0.47	0.061	0.012	1,384	$538	$107	$12,187,216
MS	14,277	2.52	0.52	0.065	0.013	921	$568	$116	$8,113,273
NC	22,364	2.30	0.92	0.059	0.023	1,316	$518	$207	$11,587,446

(continued)

Table 10-23. Summary of Estimated Mercury Exposures, with Associated IQ Decrements and Foregone Earnings: Angler Destination Approach—2020 with CAIR (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2020; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	3,086	2.56	0.33	0.066	0.008	202	$578	$75	$1,782,188
NE	6,297	2.07	0.70	0.053	0.018	334	$467	$159	$2,940,513
NH	3,225	4.07	1.24	0.104	0.032	336	$918	$278	$2,959,412
NJ	12,096	3.06	1.39	0.078	0.036	947	$689	$314	$8,337,532
NY	24,606	3.17	1.26	0.081	0.032	1,998	$715	$283	$17,595,195
OH	35,133	2.20	0.70	0.056	0.018	1,975	$495	$158	$17,390,082
OK	27,354	2.23	1.07	0.057	0.027	1,562	$503	$240	$13,758,988
PA	28,404	3.60	3.85	0.092	0.098	2,619	$812	$867	$23,070,030
RI	961	2.93	0.56	0.075	0.014	72	$660	$127	$634,231
SC	13,610	2.66	0.93	0.068	0.024	926	$599	$209	$8,154,050
SD	4,367	2.23	0.56	0.057	0.014	249	$502	$125	$2,191,439
TN	23,827	2.00	0.64	0.051	0.016	1,217	$450	$143	$10,719,756
TX	74,202	2.21	0.57	0.057	0.015	4,199	$498	$129	$36,978,242
VA	20,954	1.75	0.93	0.045	0.024	938	$394	$209	$8,261,322
VT	1,984	3.25	1.42	0.083	0.036	165	$731	$321	$1,451,289
WI	28,088	2.32	0.59	0.059	0.015	1,667	$523	$134	$14,680,994
WV	5,921	2.31	0.64	0.059	0.016	349	$520	$144	$3,076,753

[a] Benefits analyses using the angler destination approach were conducted at a HUC level, but the results are aggregated and reported at a state level in this table. For comparison purposes with the Base Cases in 2001, benefits presented in this table do not incorporate potential lags in fish tissue response to a change in mercury deposition.

Table 10-24. 2020 Base Case with CAIR: Modelled Avoided Losses Relative to 2001 Base Case Applied to 2020 Demographics—Angler Destination Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0089	6,679	7,363	6,337	9,415	$43,769,459	$35,903,739	$48,142,771	$18,913,979
State									
AL	0.0066	111	122	105	156	$725,459	$595,088	$797,945	$313,491
AR	0.0040	75	82	71	105	$489,454	$401,495	$538,359	$211,506
CT	0.0111	67	74	64	95	$440,961	$361,717	$485,021	$190,551
DC	0.0101	0.11	0.12	0.10	0.15	$689	$565	$758	$298
DE	0.0066	5	5	5	7	$32,559	$26,708	$35,812	$14,070
FL	0.0079	230	253	218	324	$1,506,686	$1,235,923	$1,657,230	$651,080
GA	0.0100	275	303	261	388	$1,801,801	$1,478,003	$1,981,831	$778,607
IA	0.0045	69	76	65	97	$451,069	$370,009	$496,139	$194,919
IL	0.0065	279	308	265	393	$1,829,053	$1,500,357	$2,011,806	$790,384
IN	0.0088	264	291	250	372	$1,728,205	$1,417,632	$1,900,882	$746,804
KS	0.0071	108	119	103	152	$708,499	$581,176	$779,290	$306,162
KY	0.0092	186	205	176	262	$1,218,938	$999,885	$1,340,731	$526,736
LA	0.0050	79	87	75	111	$517,930	$424,854	$569,681	$223,812
MA	0.0115	100	110	95	141	$654,970	$537,266	$720,412	$283,030
MD	0.0067	63	69	60	89	$411,975	$337,940	$453,138	$178,026
ME	0.0080	30	33	29	43	$198,943	$163,191	$218,821	$85,969
MI	0.0067	178	197	169	251	$1,168,843	$958,792	$1,285,630	$505,089
MN	0.0028	131	145	125	185	$861,393	$706,594	$947,461	$372,232
MO	0.0065	164	181	155	231	$1,073,033	$880,201	$1,180,247	$463,687
MS	0.0081	128	141	121	180	$836,533	$686,202	$920,117	$361,489
NC	0.0126	313	345	297	441	$2,050,491	$1,682,001	$2,255,369	$886,073

(continued)

Table 10-24. 2020 Base Case with CAIR: Modelled Avoided Losses Relative to 2001 Base Case Applied to 2020 Demographics—Angler Destination Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0055	19	21	18	27	$124,232	$101,907	$136,645	$53,684
NE	0.0049	34	38	33	49	$225,882	$185,289	$248,451	$97,610
NH	0.0095	34	38	32	48	$223,606	$183,422	$245,948	$96,626
NJ	0.0076	102	112	96	143	$665,803	$546,153	$732,328	$287,712
NY	0.0128	349	385	331	492	$2,287,436	$1,876,365	$2,515,989	$988,464
OH	0.0176	686	756	651	967	$4,495,750	$3,687,828	$4,944,952	$1,942,736
OK	0.0047	142	157	135	201	$933,671	$765,883	$1,026,961	$403,465
PA	0.0292	921	1,015	874	1,298	$6,034,162	$4,949,775	$6,637,078	$2,607,526
RI	0.0091	10	11	9	14	$63,239	$51,874	$69,557	$27,327
SC	0.0102	153	169	145	216	$1,004,906	$824,316	$1,105,313	$434,247
SD	0.0068	33	36	31	46	$214,525	$175,974	$235,960	$92,702
TN	0.0114	302	333	287	426	$1,979,290	$1,623,596	$2,177,054	$855,305
TX	0.0067	552	609	524	778	$3,619,204	$2,968,804	$3,980,824	$1,563,957
VA	0.0091	212	234	201	299	$1,388,296	$1,138,808	$1,527,011	$599,921
VT	0.0111	25	27	23	35	$160,835	$131,932	$176,906	$69,501
WI	0.0041	129	142	123	182	$846,939	$694,737	$931,562	$365,985
WV	0.0184	121	134	115	171	$794,199	$651,475	$873,553	$343,195

[a] Benefits analyses using the angler destination approach were conducted at a HUC level, but for summary purposes the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Estimated per capita IQ decrements and mercury ingestion rate do not vary across different lag periods with the angler destination approach.

Table 10-25. 2001 Utility Mercury Emissions Zero Out: Modelled Avoided Losses Relative to 2001 Base Case—Angler Destination Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2001; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0090	5,636	6,148	5,460	8,202	$36,934,050	$29,978,191	$41,479,456	$16,477,629
State									
AL	0.0066	91	100	89	133	$599,322	$486,451	$673,079	$267,379
AR	0.0051	79	86	77	115	$517,992	$420,437	$581,740	$231,095
CT	0.0060	30	33	29	44	$198,764	$161,331	$223,226	$88,676
DC	0.0081	0	0	0	0	$460	$373	$516	$205
DE	0.0066	4	5	4	6	$27,293	$22,153	$30,652	$12,176
FL	0.0055	134	146	130	195	$876,952	$711,794	$984,877	$391,240
GA	0.0093	213	233	207	310	$1,397,305	$1,134,148	$1,569,269	$623,389
IA	0.0051	65	71	63	95	$426,459	$346,143	$478,942	$190,259
IL	0.0076	273	298	264	397	$1,788,859	$1,451,960	$2,009,010	$798,075
IN	0.0095	237	258	229	345	$1,552,030	$1,259,734	$1,743,036	$692,417
KS	0.0074	94	102	91	136	$614,414	$498,700	$690,029	$274,113
KY	0.0087	147	160	142	214	$961,667	$780,554	$1,080,017	$429,035
LA	0.0050	66	73	64	97	$435,766	$353,698	$489,395	$194,411
MA	0.0087	63	69	61	92	$414,894	$336,756	$465,954	$185,099
MD	0.0069	53	58	52	78	$350,194	$284,241	$393,292	$156,234
ME	0.0051	16	18	16	24	$106,020	$86,053	$119,068	$47,299
MI	0.0081	181	197	175	263	$1,184,445	$961,376	$1,330,212	$528,424
MN	0.0018	69	75	67	100	$450,121	$365,349	$505,517	$200,815
MO	0.0067	141	153	136	205	$921,773	$748,173	$1,035,213	$411,236
MS	0.0075	99	108	96	143	$646,120	$524,435	$725,636	$288,258
NC	0.0122	252	275	244	367	$1,652,481	$1,341,266	$1,855,848	$737,232

(continued)

Table 10-25. 2001 Utility Mercury Emissions Zero Out: Modelled Avoided Losses Relative to 2001 Base Case—Angler Destination Approach (continued)

	Total Avoided IQ Decrements					Total Avoided Net Earnings Losses (Present Value in 2001; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0051	15	16	14	21	$95,044	$77,145	$106,741	$42,403
NE	0.0047	28	30	27	40	$181,100	$146,993	$203,388	$80,795
NH	0.0068	20	22	20	30	$133,480	$108,341	$149,907	$59,550
NJ	0.0125	140	153	135	203	$916,294	$743,727	$1,029,061	$408,792
NY	0.0125	284	310	275	414	$1,863,495	$1,512,540	$2,092,832	$831,373
OH	0.0180	586	639	568	853	$3,841,667	$3,118,159	$4,314,454	$1,713,908
OK	0.0052	133	145	128	193	$868,975	$705,320	$975,919	$387,682
PA	0.0323	851	928	824	1,238	$5,575,486	$4,525,445	$6,261,651	$2,487,428
RI	0.0091	8	9	8	12	$52,962	$42,987	$59,480	$23,628
SC	0.0106	134	146	129	194	$875,061	$710,259	$982,753	$390,397
SD	0.0065	26	29	25	38	$172,035	$139,635	$193,206	$76,751
TN	0.0109	241	263	234	351	$1,580,236	$1,282,627	$1,774,712	$705,001
TX	0.0064	438	478	425	638	$2,873,128	$2,332,026	$3,226,719	$1,281,807
VA	0.0103	200	218	193	291	$1,308,459	$1,062,035	$1,469,489	$583,751
VT	0.0070	13	14	13	19	$84,701	$68,749	$95,125	$37,788
WI	0.0042	109	119	106	159	$716,280	$581,382	$804,432	$319,559
WV	0.0187	103	112	99	149	$672,317	$545,698	$755,058	$299,945

[a] Benefits analyses using the angler destination approach were conducted at a HUC level, but for summary purposes the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Estimated per capita IQ decrements and mercury ingestion rate do not vary across different lag periods with the angler destination approach.

Table 10-26. 2020 with CAIR Emissions Zero Out: Modelled Avoided Losses Relative to 2020 with CAIR Base Case—Angler Destination Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0033	2,452	2,703	2,327	3,457	$16,071,187	$13,183,067	$17,676,972	$6,944,799
State									
AL	0.0018	30	33	28	42	$196,083	$160,845	$215,675	$84,733
AR	0.0035	65	72	62	92	$426,322	$349,709	$468,919	$184,226
CT	0.0019	12	13	11	17	$77,362	$63,459	$85,091	$33,430
DC	0.0024	0	0	0	0	$166	$136	$182	$72
DE	0.0025	2	2	2	3	$12,610	$10,344	$13,870	$5,449
FL	0.0019	54	60	52	77	$356,504	$292,437	$392,125	$154,055
GA	0.0028	77	85	73	109	$506,464	$415,448	$557,068	$218,857
IA	0.0029	45	50	43	63	$294,703	$241,743	$324,149	$127,349
IL	0.0031	135	149	128	190	$884,090	$725,212	$972,426	$382,040
IN	0.0037	111	123	106	157	$730,635	$599,334	$803,637	$315,727
KS	0.0038	59	65	56	83	$383,870	$314,886	$422,225	$165,881
KY	0.0027	55	61	52	77	$359,669	$295,034	$395,607	$155,423
LA	0.0024	38	42	36	54	$249,120	$204,351	$274,011	$107,651
MA	0.0029	25	28	24	35	$163,677	$134,263	$180,032	$70,729
MD	0.0017	16	17	15	22	$103,428	$84,841	$113,762	$44,694
ME	0.0031	12	13	11	16	$76,013	$62,353	$83,608	$32,847
MI	0.0047	126	139	120	178	$827,885	$679,107	$910,605	$357,752
MN	0.0014	65	71	61	91	$423,007	$346,989	$465,273	$182,793
MO	0.0035	87	96	83	123	$570,043	$467,602	$627,000	$246,331
MS	0.0030	47	52	45	67	$309,813	$254,137	$340,769	$133,879
NC	0.0032	79	87	75	111	$515,616	$422,956	$567,135	$222,812

(continued)

Table 10-26. 2020 with CAIR Emissions Zero Out: Modelled Avoided Losses Relative to 2020 with CAIR Base Case—Angler Destination Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0025	9	10	8	12	$57,208	$46,927	$62,924	$24,721
NE	0.0026	18	20	17	25	$117,725	$96,569	$129,488	$50,872
NH	0.0039	14	15	13	20	$91,363	$74,944	$100,492	$39,481
NJ	0.0057	77	85	73	109	$504,947	$414,204	$555,399	$218,201
NY	0.0038	105	115	99	147	$685,370	$562,204	$753,850	$296,167
OH	0.0043	166	183	158	234	$1,089,536	$893,738	$1,198,399	$470,818
OK	0.0031	94	104	90	133	$618,670	$507,490	$680,485	$267,344
PA	0.0081	254	280	241	358	$1,664,222	$1,365,148	$1,830,505	$719,156
RI	0.0034	4	4	3	5	$23,529	$19,301	$25,880	$10,168
SC	0.0036	55	61	52	78	$360,945	$296,080	$397,009	$155,974
SD	0.0027	13	14	12	18	$85,529	$70,159	$94,075	$36,959
TN	0.0026	70	77	66	99	$458,050	$375,735	$503,817	$197,936
TX	0.0026	211	232	200	297	$1,381,894	$1,133,556	$1,519,968	$597,154
VA	0.0037	86	95	82	121	$563,901	$462,563	$620,244	$243,677
VT	0.0023	5	5	5	7	$32,641	$26,775	$35,902	$14,105
WI	0.0035	108	119	102	152	$706,115	$579,220	$776,667	$305,131
WV	0.0038	25	27	24	35	$162,463	$133,267	$178,696	$70,205

[a] Benefits analyses using the angler destination approach were conducted at a HUC level, but the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Estimated per capita IQ decrements and mercury ingestion rate do not vary across different lag periods with the angler destination approach.

Table 10-27 report similar calculations for the 2020 CAMR Option 1. Relative to the 2020 Base Case with CAIR, this option is estimated to reduce per capita IQ decrements across the study area by 0.0006 points. Total IQ decrements avoided under Option 1 are estimated to decrease by approximately 460 to 500 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with CAMR Option 1 are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately \$2.5 to \$3.0 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 430 IQ points at a value of \$3.3 million under a 5 year lag and 650 IQ points at a value of \$1.3 million under a 50 year lag.

Table 10-28 report results of the 2020 CAMR Option 2. Relative to the 2020 Base Case with CAIR, this option is estimated to reduce per capita IQ decrements across the study area by 0.0009 points. Total IQ decrements avoided under Option 2 are estimated to decrease by approximately 700 to 775 points in 2020 under a 10 to 20 year lag period. The mercury emission reductions associated with CAMR Option 2 are estimated to reduce the present value of total net earnings losses due to prenatal exposures in 2020 by approximately \$3.8 to \$4.6 million under the 10 to 20 year lag periods. Under the alternative lag periods considered in the analysis, the total IQ decrements avoided and monetary value of benefits are 660 IQ points at a value of \$5.0 million under a 5 year lag and 990 IQ points at a value of \$2.0 million under a 50 year lag.

Table 10-29 summarizes the annual benefit estimates for CAMR Control Options 1 and 2, and it compares them to aggregate estimates associated with CAIR emissions reductions and the two Utility Emissions Zero-Out scenarios. To assess the sensitivity of these results to the assumed rate at which future gains/losses are discounted, the benefit estimates are also reported assuming 3 and 7 percent discount rates. In addition, we present a 1 percent discount rate for the 50 year lag period to reflect the discount rate for inter-generational effects as is recommended in the EPA Guidelines for Economic Analysis. Using a 3 percent discount rate, the aggregate present value in 2020 of avoided net earnings losses due to reductions in mercury exposures for selected years were estimated to range between \$2.5 to \$3.0 million for Option 1 and between \$3.8 to \$4.6 million for Option 2.

In comparison to the other mercury emissions reductions, the estimated benefits of Option 1 are roughly 19 percent of the estimated benefits that would be achieved by eliminating utility mercury emissions (i.e., 2020 Utility Emissions Zero-Out). The estimated benefits of Option 2 are almost 29 percent as large as those for the 2020 Utility Emissions Zero-Out scenario.

10.5.3 Comparison of Results from Two Approaches

Table 10-30 summarizes and compares results from the population centroid approach and the angler destination approach, for each of the selected years (i.e., lagged effects). Assuming a 3 percent annual discount rate, the estimated annual benefits associated with the CAMR Option 1 range from \$1.7 to \$2.0 million with the population centroid approach and \$2.5 to \$3.0 million with the angler destination approach, for an average range across both approaches under the 10 to 20 year lag period of \$1.7 to 3.0 million. For CAMR Option 2, the annual benefits range from

$2.5 to $2.1 million with the population centroid approach and $3.8 to 4.6 million with the angler

Table 10-27. Estimated Benefits of 2020 With CAIR Control Option 1: Relative to 2020 with CAIR—Angler Destination Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0006	457	504	434	644	$2,995,451	$2,457,145	$3,294,747	$1,294,416
State									
AL	0.0002	4	4	4	6	$25,793	$21,158	$28,370	$11,146
AR	0.0014	27	30	26	38	$176,298	$144,616	$193,913	$76,183
CT	0.0004	3	3	2	4	$17,025	$13,966	$18,726	$7,357
DC	0.0003	0	0	0	0	$20	$17	$22	$9
DE	0.0004	0	0	0	0	$1,997	$1,638	$2,197	$863
FL	0.0002	5	6	5	8	$35,495	$29,116	$39,041	$15,338
GA	0.0003	8	9	7	11	$51,482	$42,230	$56,626	$22,247
IA	0.0004	7	7	6	9	$43,839	$35,961	$48,219	$18,944
IL	0.0004	16	17	15	22	$102,905	$84,412	$113,187	$44,468
IN	0.0004	12	13	12	17	$79,906	$65,546	$87,890	$34,530
KS	0.0006	9	10	9	13	$60,012	$49,227	$66,008	$25,933
KY	0.0003	6	7	6	9	$40,766	$33,440	$44,840	$17,616
LA	0.0005	9	9	8	12	$56,228	$46,123	$61,846	$24,298
MA	0.0005	5	5	4	7	$30,738	$25,214	$33,810	$13,283
MD	0.0004	4	4	4	5	$24,727	$20,283	$27,197	$10,685
ME	0.0012	4	5	4	6	$29,092	$23,864	$31,998	$12,571
MI	0.0004	11	12	10	15	$68,912	$56,528	$75,797	$29,779
MN	0.0001	6	7	6	9	$40,853	$33,511	$44,935	$17,654
MO	0.0006	15	16	14	21	$96,596	$79,237	$106,248	$41,742
MS	0.0005	7	8	7	10	$48,514	$39,796	$53,361	$20,964
NC	0.0005	12	13	12	17	$79,502	$65,215	$87,446	$34,355

(continued)

Table 10-27. Estimated Benefits of 2020 With CAIR Control Option 1: Relative to 2020 with CAIR—Angler Destination Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0006	2	2	2	3	$12,868	$10,556	$14,154	$5,561
NE	0.0004	3	3	3	4	$20,484	$16,803	$22,530	$8,852
NH	0.0012	4	5	4	6	$28,501	$23,379	$31,349	$12,316
NJ	0.0023	31	34	29	43	$200,521	$164,486	$220,556	$86,650
NY	0.0008	22	24	21	31	$143,881	$118,024	$158,257	$62,175
OH	0.0003	14	15	13	19	$88,663	$72,730	$97,522	$38,314
OK	0.0004	11	13	11	16	$75,345	$61,805	$82,874	$32,559
PA	0.0030	93	103	89	132	$612,298	$502,263	$673,477	$264,591
RI	0.0006	1	1	1	1	$3,987	$3,271	$4,386	$1,723
SC	0.0003	5	6	5	7	$32,924	$27,008	$36,214	$14,228
SD	0.0004	2	2	2	3	$13,706	$11,243	$15,075	$5,923
TN	0.0005	14	15	13	20	$91,916	$75,398	$101,100	$39,720
TX	0.0004	32	35	30	45	$209,545	$171,888	$230,482	$90,550
VA	0.0016	37	41	35	52	$240,792	$197,519	$264,851	$104,053
VT	0.0006	1	1	1	2	$7,971	$6,538	$8,767	$3,444
WI	0.0003	9	9	8	12	$55,817	$45,787	$61,395	$24,120
WV	0.0011	7	8	7	10	$45,531	$37,349	$50,080	$19,675

[a] Benefits analyses using the angler destination approach were conducted at a HUC level, but the results are aggregated and reported at a state level in this table.
[b] Case studies of individual ecosystems (as presented in Section 3) 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.
[c] Estimated per capita IQ decrements and mercury ingestion rate do not vary across different lag periods with the angler destination approach.

Table 10-28. Estimated Benefits of 2020 With CAIR Control Option 2: Relative to 2020 with CAIR—Angler Destination Approach[a, b]

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
Study Area	0.0009	700	772	664	987	$4,586,570	$3,762,327	$5,044,846	$1,981,982
State									
AL	0.0004	7	8	7	10	$48,579	$39,849	$53,433	$20,992
AR	0.0018	35	38	33	49	$226,689	$185,951	$249,339	$97,959
CT	0.0005	3	3	3	4	$20,176	$16,551	$22,192	$8,719
DC	0.0004	0	0	0	0	$28	$23	$31	$12
DE	0.0006	0	1	0	1	$3,145	$2,580	$3,460	$1,359
FL	0.0004	11	12	11	16	$73,160	$60,013	$80,470	$31,614
GA	0.0010	28	31	27	40	$186,772	$153,207	$205,434	$80,709
IA	0.0008	12	13	11	17	$77,054	$63,207	$84,753	$33,297
IL	0.0006	25	28	24	35	$163,649	$134,240	$180,000	$70,717
IN	0.0006	19	21	18	27	$123,657	$101,435	$136,012	$53,435
KS	0.0012	18	20	17	26	$120,199	$98,598	$132,209	$51,941
KY	0.0005	11	12	10	15	$71,142	$58,357	$78,250	$30,742
LA	0.0007	12	13	11	16	$76,098	$62,423	$83,702	$32,884
MA	0.0007	6	7	6	9	$42,065	$34,506	$46,268	$18,177
MD	0.0005	5	5	5	7	$32,228	$26,436	$35,448	$13,926
ME	0.0014	5	6	5	8	$35,489	$29,112	$39,035	$15,336
MI	0.0007	19	21	18	27	$124,334	$101,990	$136,757	$53,728
MN	0.0003	12	14	12	18	$81,863	$67,151	$90,042	$35,375
MO	0.0010	25	27	23	35	$161,994	$132,882	$178,179	$70,002
MS	0.0008	12	14	12	17	$80,345	$65,907	$88,373	$34,719
NC	0.0007	18	19	17	25	$115,883	$95,058	$127,462	$50,076

(continued)

Table 10-28. Estimated Benefits of 2020 With CAIR Control Option 2: Relative to 2020 with CAIR—Angler Destination Approach (continued)

		Total Avoided IQ Decrements				Total Avoided Net Earnings Losses (Present Value in 2020; 3% discount rate; 1999$)			
		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times		Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	Per Capita[c]	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr. Lag
State									
ND	0.0008	3	3	2	4	$17,003	$13,947	$18,702	$7,347
NE	0.0008	6	6	5	8	$36,187	$29,684	$39,802	$15,637
NH	0.0014	5	5	5	7	$32,423	$26,597	$35,663	$14,011
NJ	0.0025	33	37	31	47	$217,220	$178,184	$238,924	$93,867
NY	0.0011	30	33	28	42	$194,435	$159,493	$213,862	$84,021
OH	0.0007	27	30	26	38	$178,335	$146,287	$196,154	$77,064
OK	0.0013	39	43	37	55	$256,552	$210,448	$282,186	$110,863
PA	0.0033	105	116	100	148	$687,982	$564,346	$756,723	$297,296
RI	0.0009	1	1	1	1	$5,986	$4,911	$6,585	$2,587
SC	0.0008	12	13	12	17	$79,822	$65,478	$87,798	$34,493
SD	0.0007	3	4	3	5	$21,853	$17,926	$24,036	$9,443
TN	0.0008	21	23	20	30	$137,697	$112,952	$151,456	$59,503
TX	0.0006	53	59	51	75	$349,061	$286,332	$383,938	$150,839
VA	0.0018	41	46	39	58	$271,563	$222,761	$298,697	$117,350
VT	0.0007	2	2	1	2	$9,959	$8,169	$10,954	$4,304
WI	0.0008	26	29	25	37	$170,194	$139,609	$187,199	$73,546
WV	0.0013	9	9	8	12	$55,746	$45,728	$61,316	$24,089

[a] Benefits analyses using the angler destination approach were conducted at a HUC level, but the results are aggregated and reported at a state level in this table.

[b] Case studies of individual ecosystems (as presented in Section 3) 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.

[c] Estimated per capita IQ decrements and mercury ingestion rate do not vary across different lag periods with the angler destination approach.

Table 10-29. Summary of Annual Benefit Estimates: Angler Destination Approach

	Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	10-Yr Lag	**20-Yr Lag**	**5-Yr Lag**	**50-Yr Lag**
Annual Number of Prenatally Exposed Children				
2001 Base Case	624,862	681,608	605,347	909,371
2020 Base Case (with CAIR)	748,424	825,065	710,103	1,054,990
Total Value of Benefits (1999$s)				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)				
1% Discount Rate; Present Value in 2001	N/A	N/A	N/A	$43,922,522
3% Discount Rate; Present Value in 2001	$36,934,050	$29,978,191	$41,479,456	$16,477,629
7% Discount Rate; Present Value in 2001	$25,232,565	$13,991,825	$34,284,695	$2,452,263
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$50,416,821
3% Discount Rate; Present Value in 2020	$43,769,459	$35,903,739	$48,142,771	$18,913,979
7% Discount Rate; Present Value in 2020	$29,902,373	$16,757,477	$39,792,234	$2,814,850
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$18,511,953
3% Discount Rate; Present Value in 2020	$16,071,187	$13,183,067	$17,676,972	$6,944,799
7% Discount Rate; Present Value in 2020	$10,979,497	$6,152,979	$14,610,837	$1,033,551
BENEFITS OF CONTROL OPTIONS				
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$3,450,377
3% Discount Rate; Present Value in 2020	$2,995,451	$2,457,145	$3,294,747	$1,294,416
7% Discount Rate; Present Value in 2020	$2,046,429	$1,146,832	$2,723,262	$192,640
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$5,283,142
3% Discount Rate; Present Value in 2020	$4,586,570	$3,762,327	$5,044,846	$1,981,982
7% Discount Rate; Present Value in 2020	$3,133,448	$1,756,004	$4,169,799	$294,966

Table 10-30. Summary and Comparison of Annual Benefit Estimates: Population Centroid Approach vs. Angler Destination Approach

	Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	10-Yr Lag	**20-Yr Lag**	**5-Yr Lag**	**50-Yr Lag**
Population Centroid Approach				
Annual Number of Prenatally Exposed Children				
2001 Base Case	452,575	486,487	442,938	632,017
2020 Base Case (with CAIR)	528,721	577,910	504,127	725,474
BENEFITS OF CONTROL OPTIONS				
2020 CAMR Control Option 1 (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$2,300,269
3% Discount Rate; Present Value in 2020	$2,086,359	$1,687,988	$2,313,079	$862,951
7% Discount Rate; Present Value in 2020	$1,425,357	$787,840	$1,911,867	$128,428
2020 CAMR Control Option 2 (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$3,471,289
3% Discount Rate; Present Value in 2020	$3,112,816	$2,527,403	$3,444,104	$1,302,261
7% Discount Rate; Present Value in 2020	$2,126,610	$1,179,624	$2,846,712	$193,807
Angler Destination Approach				
Annual Number of Prenatally Exposed Children				
2001 Base Case	624,862	681,608	605,347	909,371
2020 Base Case (with CAIR)	748,424	825,065	710,103	1,054,990
BENEFITS OF CONTROL OPTIONS				
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$3,450,377
3% Discount Rate; Present Value in 2020	$2,995,451	$2,457,145	$3,294,747	$1,294,416
7% Discount Rate; Present Value in 2020	$2,046,429	$1,146,832	$2,723,262	$192,640
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$5,283,142
3% Discount Rate; Present Value in 2020	$4,586,570	$3,762,327	$5,044,846	$1,981,982
7% Discount Rate; Present Value in 2020	$3,133,448	$1,756,004	$4,169,799	$294,966

destination approach, for an average range across both approaches under the 10 to 20 year lag period of $1.7 to 3.8 million. The main source of difference between the two approaches are the estimated sizes of the exposed populations. The estimates from the angler destination approach are generally between 35 percent and 40 percent higher than from the other approach. As described in Section 10.1.2 the two approaches use different modeling assumptions to identify numbers of pregnant women in angler households. As discussed in more detail in Section 10.7, the assumptions used in the population centroid approach are likely to underestimate these numbers, whereas the assumptions of the angler destination approach are likely to overestimate them. Therefore, the results of the two modeling approaches can be interpreted as providing lower and upper bound average estimates of exposed populations and benefits.

10.5.4 Sensitivity Analysis of Alternative Dose-Response Functions

EPA conducted a sensitivity analysis of alternative dose-response functions provided in Section 9 of this report. Specifically, we estimated the benefits with a lower dose-response function with a beta coeffiicent of -0.108, and a higher dose-response function with a beta coefficient of -0.233. The findings of the sensitivity analysis are present in Table 10-31 and show that with the lower dose-response function, the benefits would decrease by about 18 percent, while the higher dose-resopnse function would increase benefits by about 56 percent.

Table 10-31. Summary and Comparison of Annual Benefit Estimates Under Alternative IQ Dose-Response Assumptions: Population and Angler Destination Approach

	5-Yr Lag	20-Yr. Lag
Population Centroid Approach		
2020 Base Case (with CAIR)	504,127	577,910
BENEFITS OF CONTROL OPTION 1		
2020 CAMR Control Option 1 (Relative to 2020 Base Case with CAIR; 3% Discount Rate; Present Value in 2020)		
IQ Dose-Response Coefficient = -0.108	$ 1,906,966	$ 1,391,623
IQ Dose-Response Coefficient = -0.131	$ 2,313,079	$ 1,687,988
IQ Dose-Response Coefficient = -0.233	$ 4,114,102	$ 3,002,299
Angler Destination Approach		
Annual Number of Prenatally Exposed Children		
2020 Base Case (with CAIR)	710,103	825,065
BENEFITS OF CONTROL OPTION 1		
2020 CAMR Control Option 1 (Relative to 2020 Base Case with CAIR; 3% Discount Rate; Present Value in 2020)		
IQ Dose-Response Coefficient = -0.108	$ 2,716,280	$ 2,025,738
IQ Dose-Response Coefficient = -0.131	$ 3,294,747	$ 2,457,145
IQ Dose-Response Coefficient = -0.233	$ 5,860,123	$ 4,370,341

10.5.5 Distribution of Per-Capita IQ Changes for the Exposed Population (in support of distributional equity analysis)

In addition to considering the net benefits associated with reductions in mercury fish tissue concentrations, an additional factor to consider as part of a cost-benefit (as stipulated in Executive Order 12866) is the distributional equity of benefits in relation to the distribution of societal costs. To support an assessment of distributional equity in the context of this analysis, EPA has modeled per-capita IQ changes for the population of modeled recreational fishers. Consideration of the distribution of IQ changes across the modeled population using these results supports a determination regarding the degree of equity in benefits associated with this regulation.

Thus far, the analysis of mercury ingestion levels has assumed a constant rate of freshwater fish consumption (C) across the exposed population. However, the population centroid approach can also be adapted to allow for variation in C. As described above in Section 10.1.4, the constant consumption rate (C) is based on the recommended average rate for freshwater recreational anglers in EPA's *Exposure Factor Handbook (EFH)* (EPA, 1997b). However, the *EFH* also provides guidelines regarding the variability in average daily fish consumption rates across the recreational freshwater fishing population. It recommends mean *and 95th percentile* daily self-caught fish freshwater fish consumption rate for freshwater recreational anglers of 8 g/day and 25 g/day respectively. The distribution of consumption rate is skewed to the right because the 95th percentile is more than 3 times the mean. Therefore, the variability in the consumption rate can be represented by a distribution such as lognormal distribution. The *EFH* contains examples of lognormal distributions fitted to fish consumption data. Based on *EFH* recommendations, it is also possible to incorporate the variation in consumption rates by allowing it to vary randomly (within the defined distribution) across the exposed population.

To create a full distribution of consumption rates that is consistent with the EFH recommendations, the consumption rates was assumed to be log-normally distributed across the population with a mean of 8 g/day and a 95th percentile of 25 g/day; that is, the specified distribution for the consumption rate was log-normal (8, 10.45). Lognormal distribution was deemed to be appropriate because of its skewed shape and because it would not allow for fish consumption rates in the negative range. Drawing randomly from this distribution, a consumption rate was assigned to each of the roughly 165,000 block groups in the study area. The fish consumption in each block group was assumed to be constant for the modeling purposes. This approach was used to avoid the computational burden of assigning a random consumption rate to a fractional person in a block group (the *estimated* number of exposed persons in each block group is not an integer) and because the estimated number of exposed persons per block group is relatively small. Although some degree of randomness is lost by assuming a constant consumption rate in a block group, the inter-individual variability in consumption rate is adequately represented because the average number of exposed individuals in a block group is only 2.6 with standard deviation of 2.5. In effect, a sample of 165,000 people is used to represent the consumption rate variability in approximately 435,000 people, which is adequate for the modeling purposes. After randomly assigning consumption rates to the block groups, the population centroid approach was reapplied using Eq. (10.16) with a variable consumption rate to calculate average daily mercury ingestion rates for the estimated exposed population (NPA) in each block group.

Based on this adapted version of the population centroid approach, Figures 10-11 to 10-18 show how the avoided IQ decrements (i.e., benefits) are distributed in the exposed population under alternative emissions reductions scenarios. In all of these figures, the consumption rate is allowed to vary across the exposed population as described above.

Figures 10-11 and 10-12 report results for the 2001 Utility Emissions Zero-Out scenario, relative to the 2001 Base Case. Most of the prenatally exposed children avoid less than 0.025 IQ point decrements; however, a significant number have avoided IQ decrements between 0.025 and 0.1 IQ points. About 1,300 prenatally exposed children have avoided IQ loss greater than 0.1 IQ points. Figure 10-12 shows the cumulative distribution of avoided IQ decrements across the exposed population. Reduction in IQ decrements (i.e., benefits) due to the Zero-Out scenario are less than 0.02 IQ points from the baseline levels for more than 90 percent of the prenatally exposed children.

Figures 10-13 and 10-14 report results for the 2020 Utility Emissions Zero-Out scenario, relative to the 2020 Base Case with CAIR. Again, a large majority of the prenatally exposed children avoid less than 0.025 IQ point decrements. In this case, less than 100 prenatally exposed children have avoided IQ loss greater than 0.1 IQ points. Figure 10-14 shows the cumulative distribution of avoided IQ decrements across the exposed population. Reductions in IQ decrements (i.e., benefits) due to the 2020 Zero-Out scenario are less than 0.01 IQ points from the 2020 baseline levels for more than 90 percent of the prenatally exposed children.

Figures 10-15 through 10-18 report results for the CAMR Control Options relative to the 2020 Base Case with CAIR. In both cases 11 or less prenatally exposed children are estimated to avoid IQ losses of more than 0.1 IQ points. Figures 10-16 and 10-18 show the cumulative distributions of avoided IQ decrements for the two CAMR control options. In both cases, reductions in IQ decrements (i.e., benefits) are less than 0.003 IQ points from the 2020 baseline levels for more than 90 percent of the prenatally exposed children.

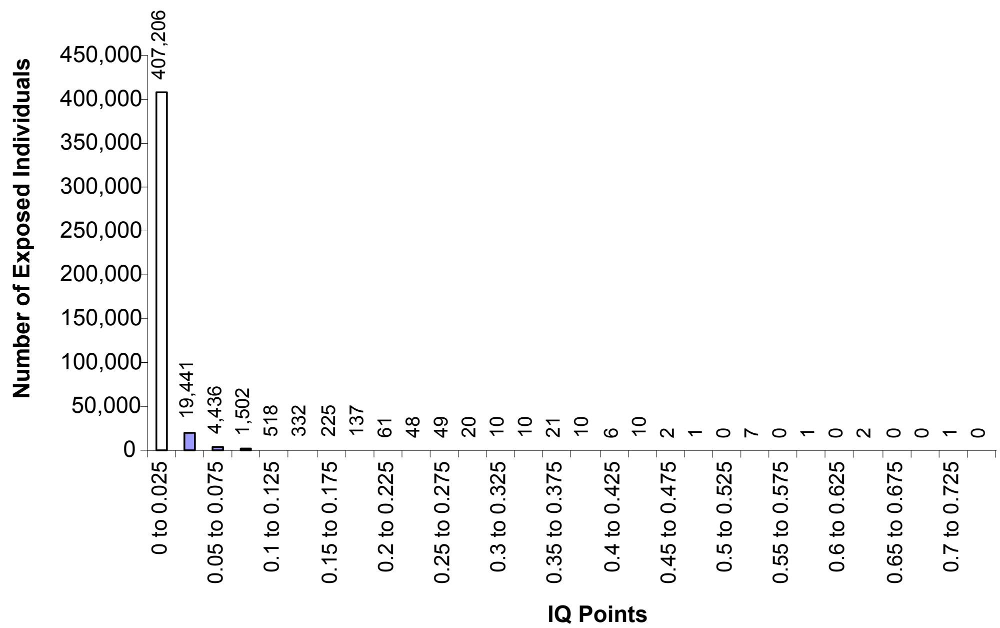

Figure 10-11. Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: 2001 Utility Emissions Zero-Out Relative to 2001 Base Case; Population Centroid Approach; Variable Consumption Rate

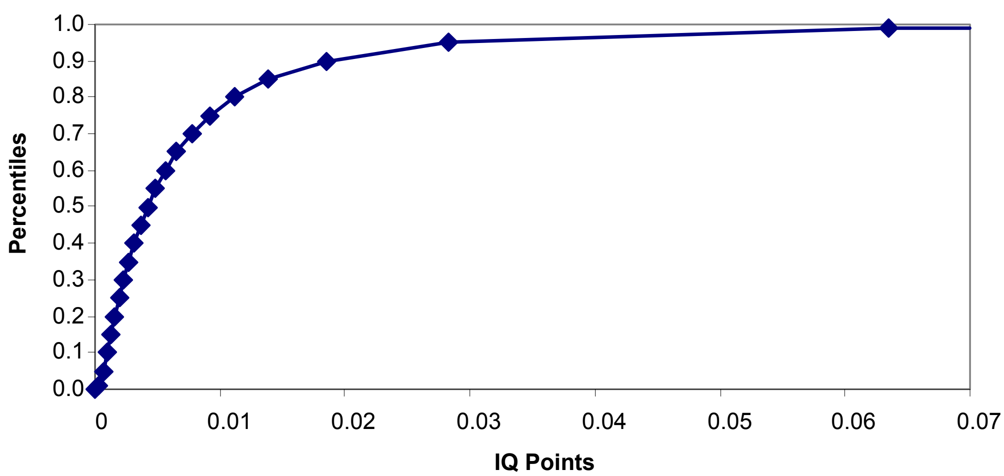

Figure 10-12. Cumulative Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: 2001 Utility Emissions Zero-Out Relative to 2001 Base Case; Population Centroid Approach; Variable Consumption Rate

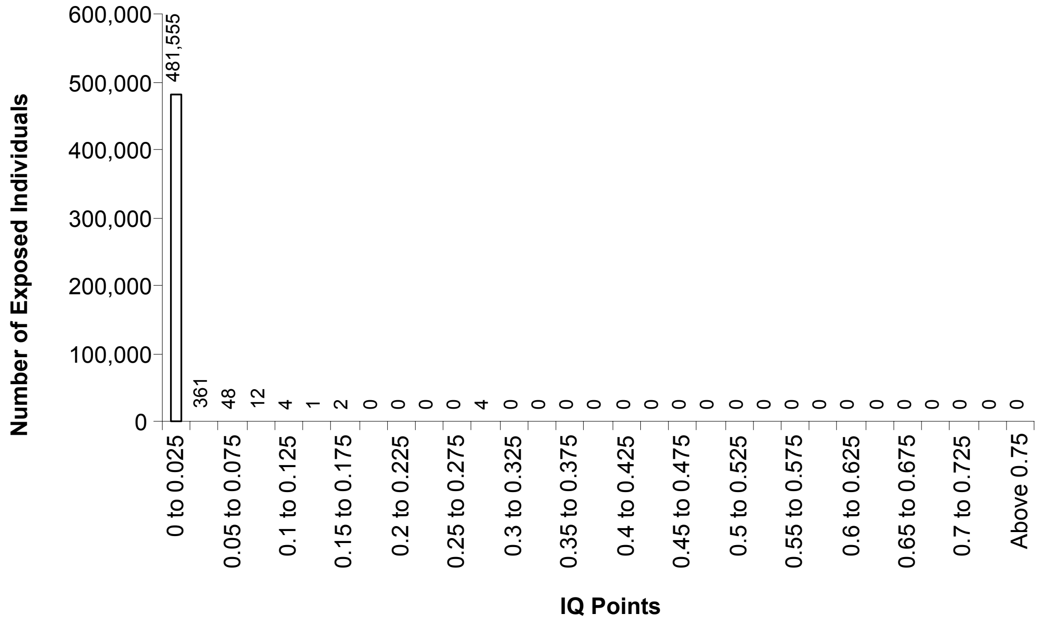

Figure 10-13. Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: CAMR Control Option 1 Relative to 2020 Base Case with CAIR; Population Centroid Approach; Variable Consumption Rate

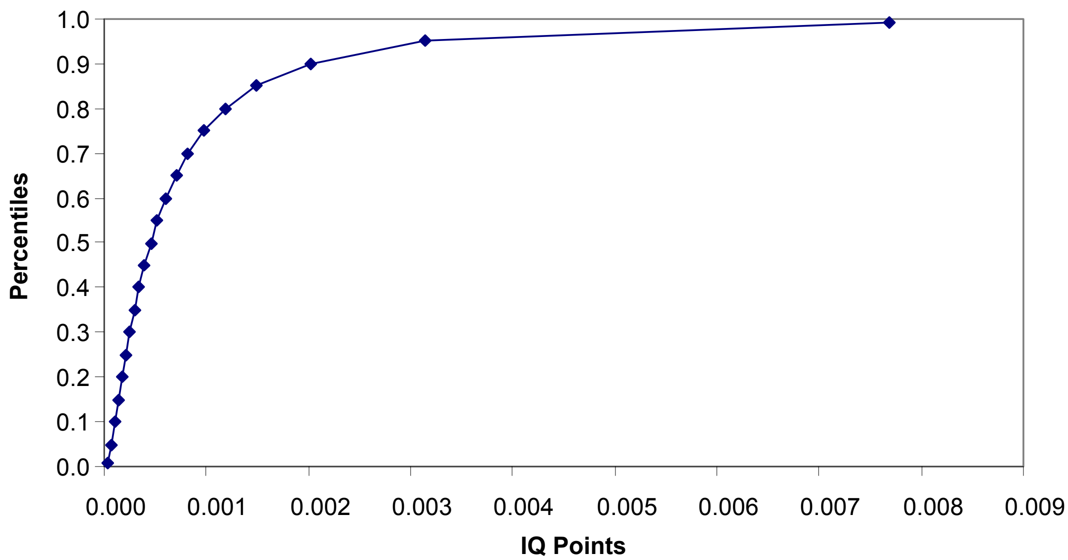

Figure 10-14. Cumulative Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: CAMR Control Option 1 Relative to 2020 Base Case with CAIR; Population Centroid Approach; Variable Consumption Rate

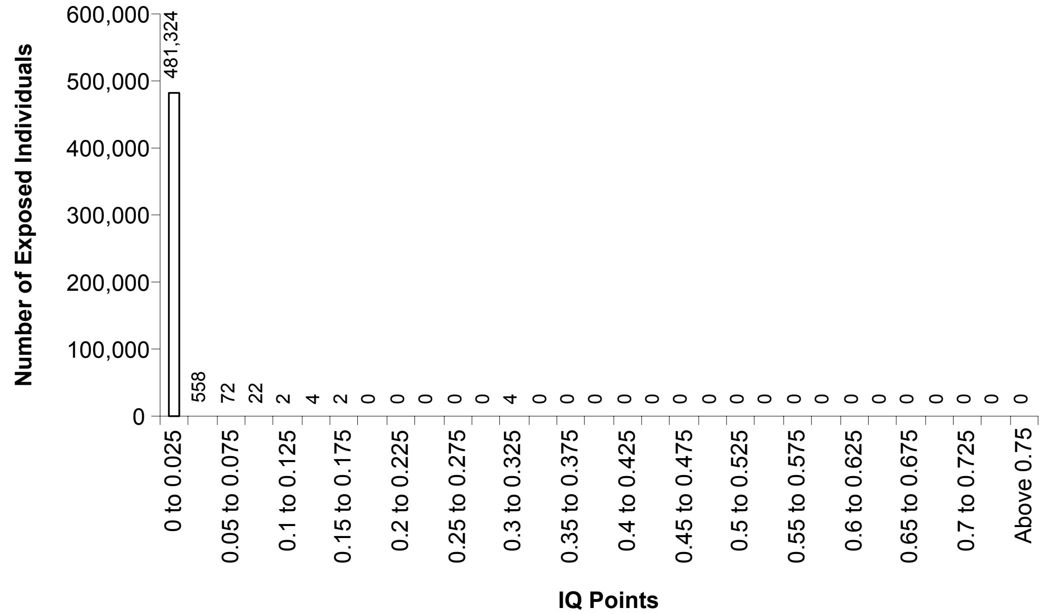

Figure 10-15. Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: CAMR Control Option 2 Relative to 2020 Base Case with CAIR; Population Centroid Approach; Variable Consumption Rate

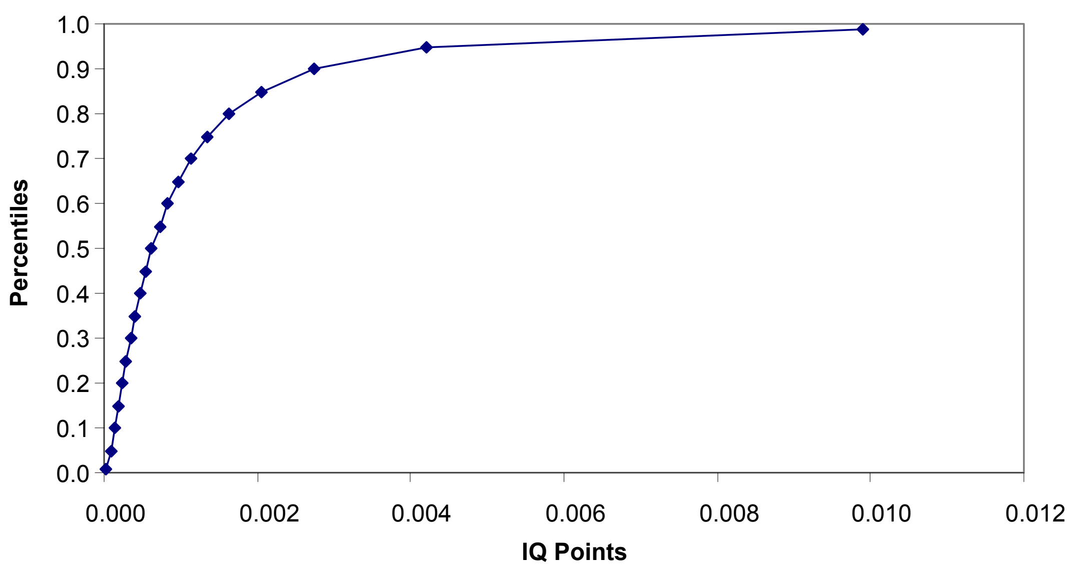

Figure 10-16. Cumulative Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: CAMR Control Option 2 Relative to 2020 Base Case with CAIR; Population Centroid Approach; Variable Consumption Rate

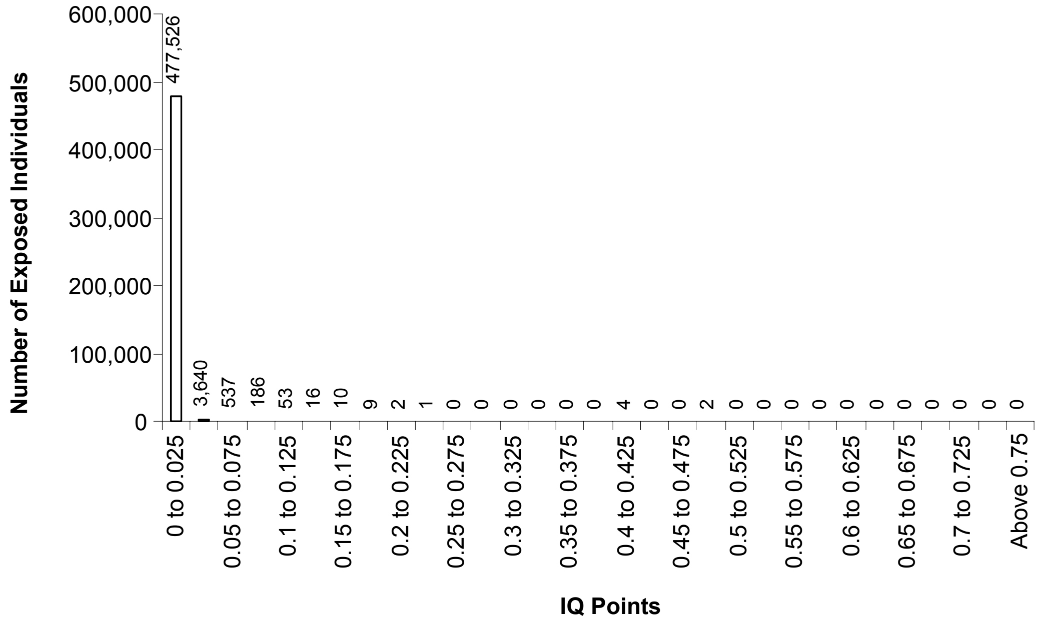

Figure 10-17. Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: 2020 Utility Emissions Zero-Out Relative to 2020 Base Case with CAIR; Population Centroid Approach; Variable Consumption Rate

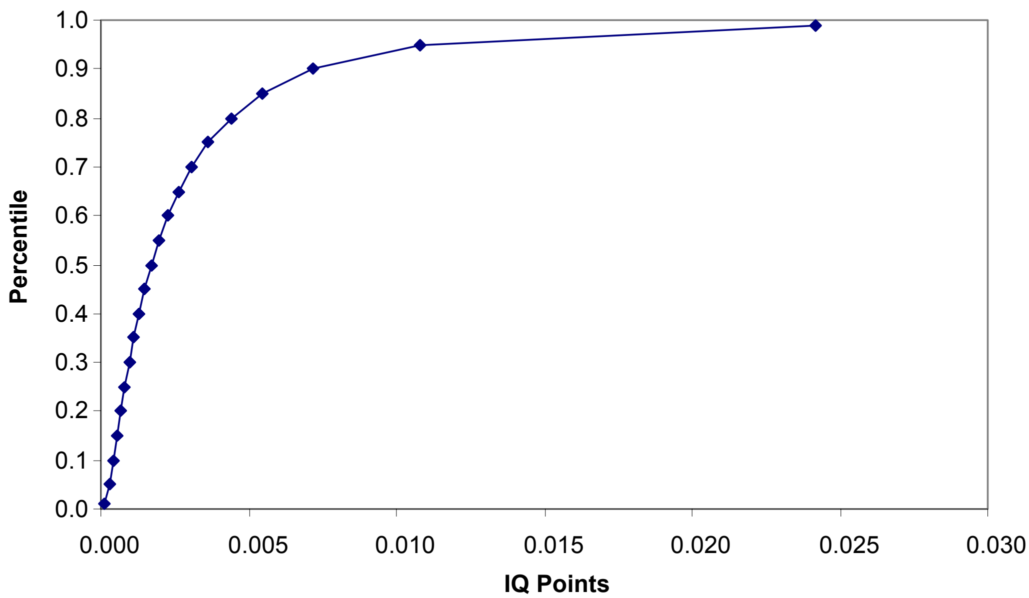

Figure 10-18. Cumulative Distribution of Modelled Avoided IQ Decrements (Benefits) due to Mercury Emissions Reductions: 2020 Utility Emissions Zero-Out Relative to 2020 Base Case with CAIR; Population Centroid Approach; Variable Consumption Rate

It is important to note that these results based on a variable consumption rate only represent one realization of the random distribution of fish consumption rates across the exposed populations. The results would be somewhat different if the randomization process were repeated; however, the summary results would most likely be very similar. In addition, the results presented at the tail of this distribution are based on a mathematical calculation of the results of the benefit modeling and are not based on any empirical evidence the Agency has that people consume large amounts of recreationally caught freshwater fish (i.e., 2-3 meals per day) such that they would incur this level of IQ loss.

10.6 Analysis of Potentially High-Risk Subpopulations

In addition to considering the full distribution of IQ benefits across the modeled study population of prenatally exposed children it is also possible to consider distributional equity by examining segments of the exposed population expected to experience disproportionately high impacts (and hence benefits) due to (a) proximity to high methylmercury fish concentrations and/or (b) relatively high fish consumption rates. In this analysis, to further examine this issue of distributional equity, EPA examined several special sub-populations believed to be at elevated risk of methylmercury fish exposure. The analyses conducted for these potential high-risk subpopulations are described in this section. Consideration of IQ change and benefits estimates generated for these special sub-populations can support a determination regarding the potential for distributional equity playing a key factor in benefits for this regulation.

In this section of the report the high-risk populations of interest are groups of individuals who are exposed to relatively high levels of mercury through consumption of self-caught freshwater fish. In particular, they are groups with high rates of fish consumption. The term "subsistence fishing"[25] may apply to the subpopulations analyzed in this section, depending on how this term is interpreted. These subpopulations are not necessarily from households for whom fishing is a primary activity for survival; however, their high rates of noncommercial freshwater fish consumption suggest that it is an integral part of their diet.

Below high-risk subpopulations are addressed in three ways, each of which is an adaptation of the population centroid approach. In the first case, fish consumption rates were allowed to vary systematically across the exposed population. The analysis focused on mercury ingestion by the those in the top fifth percentile of consumption rate distribution. In the second case, the analysis focused on the portion of the exposed population with incomes below the poverty threshold. Because we are lacking data on consumption rates by income categories, it was assumed that a portion of these populations have noncommercial freshwater fish consumption rates comparable to the population examined in the first case (top fifth percentile of the total consumption rate distribution). In the third case, the analysis focused on two ethnic subpopulations in the U.S. with high expected rates of freshwater fish consumption—the Hmong and the Chippewa in Minnesota, Wisconsin, and Michigan. Using existing studies of these populations' fishing and fish consumption behaviors, the population centroid approach was adapted and applied to estimate mercury exposures for prenatally exposed children in these ethnic groups.

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[25] "Subsistence fishing" is defined as the taking of fish for the sustenance of families, communities, and cultures.

10.6.1 Mercury Ingestion Estimates for Individuals in the Upper Range of the Fish Consumption Distribution

In this section of the report, mercury ingestion levels for potentially high risk subpopulations are analyzed by incorporating information about the variability in average daily fish consumption rates across the recreational freshwater fishing population. According to EPA's *Exposure Factors Handbook (EFH)* (EPA, 1997b), the recommended mean and 95th percentile daily self-caught fish freshwater fish consumption rate (C) for freshwater recreational anglers are 8 g/day and 25 g/day respectively. In this analysis of potentially high risk individuals, the consumption rate is allowed to vary systematically across the population, using the same method described in Section 10.4.

To create a full distribution of consumption rates that is consistent with the EFH recommendations, the consumption rate was assumed to be log-normally distributed across the population with a mean of 8 g/day and a 95th percentile of 25 g/day. Drawing randomly from this distribution, a different consumption rate was assigned to each of the roughly 165,000 block groups in the study area. The population centroid approach was then reapplied and combined with Eq. (10.16) using a variable consumption rate to calculate average daily mercury ingestion rates for the estimated exposed population (NPA) in each block group.

To specifically address potentially high risk groups, Tables 10-32 and 10-33 report results only for exposed populations that were randomly assigned consumption rates higher than 95th percentile consumption rate of the recreational freshwater angler (i.e., above 25 g/day). Table 10-32 reports detailed result for the 2001 Base Case. The annual number of prenatally exposed children in this high risk group was estimated to be 22,300 and the mean HgI across block groups was 12.95 μg/day. Average IQ decrements in this group were estimated to be 0.33 points and the total present value of foregone earnings was estimated to be $65 million. As with the estimates reported for the recreational freshwater angler in Section 10.1.4, these estimates are based on observed mercury levels in freshwater fish (as summarized in Table 10-2) and therefore include all sources of mercury freshwater fish.

Table 10-33 summarizes similar model results for the 2020 Base Case with CAIR. The annual number of prenatally exposed children in this high risk group in 2020 was estimated to be 24,700 and the mean HgI across block groups was 11.33 μg/day, which is 12 percent lower than in the 2001 Base Case. Average IQ decrements in this group were estimated to be 0.29 points and the total present value of foregone earnings was estimated to be $63 million.

Table 10-34 reports estimates of beneficial changes to this subsistence population under the five emissions control scenarios, including estimates for five lag periods and three assumed discount rates. The 2001 Utility Emissions Zero-Out and the 2020 Base Case with CAIR are both estimated to result in per capita IQ decrements that are on average between 0.042 and 0.047 less than under the 2001 Base Case. Compared to the 2020 Base Case with CAIR, the 2020 Utility Emissions Zero-Out is estimated to reduce per capita IQ decrements by an average of

Table 10-32. Summary of Estimated Mercury Exposures for Consumption-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2001 Base Case[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (μg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	22,302	12.95	10.19	0.331	0.261	7,390	$2,918	$2,297	$65,086,047
State									
AL	507	10.56	7.03	0.270	0.180	137	$2,379	$1,584	$1,206,122
AR	466	15.86	19.71	0.406	0.504	189	$3,573	$4,441	$1,666,437
CT	218	17.94	11.59	0.459	0.297	100	$4,042	$2,611	$882,544
DC	13	8.39	3.68	0.215	0.094	3	$1,890	$829	$24,954
DE	37	10.71	4.86	0.274	0.124	10	$2,413	$1,095	$90,402
FL	1,114	19.49	10.00	0.499	0.256	555	$4,391	$2,253	$4,891,366
GA	1,097	14.03	10.06	0.359	0.257	394	$3,161	$2,268	$3,469,580
IA	391	8.02	3.89	0.205	0.100	80	$1,808	$877	$707,055
IL	1,407	8.70	4.95	0.223	0.127	313	$1,961	$1,116	$2,759,572
IN	813	11.55	6.91	0.295	0.177	240	$2,602	$1,556	$2,116,091
KS	369	17.87	11.98	0.457	0.307	169	$4,027	$2,700	$1,487,071
KY	589	10.87	6.25	0.278	0.160	164	$2,449	$1,408	$1,441,147
LA	679	13.81	10.09	0.353	0.258	240	$3,111	$2,274	$2,111,880
MA	303	19.83	11.33	0.507	0.290	154	$4,468	$2,553	$1,352,521
MD	262	7.71	4.00	0.197	0.102	52	$1,736	$900	$455,573
ME	122	25.80	10.63	0.660	0.272	80	$5,813	$2,395	$707,079
MI	884	12.39	7.23	0.317	0.185	280	$2,791	$1,629	$2,467,205
MN	1,093	8.87	4.80	0.227	0.123	248	$1,998	$1,083	$2,184,445
MO	820	12.51	8.75	0.320	0.224	262	$2,819	$1,971	$2,311,088
MS	392	12.37	6.13	0.316	0.157	124	$2,787	$1,382	$1,090,960
NC	756	14.76	13.52	0.378	0.346	285	$3,327	$3,047	$2,514,047

(continued)

Table 10-32. Summary of Estimated Mercury Exposures for Consumption-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2001 Base Case (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (μg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	127	11.00	6.17	0.281	0.158	36	$2,478	$1,390	$314,521
NE	246	8.68	3.87	0.222	0.099	55	$1,956	$872	$480,417
NH	63	19.21	6.75	0.492	0.173	31	$4,330	$1,520	$272,212
NJ	365	15.70	13.68	0.402	0.350	146	$3,538	$3,082	$1,289,682
NY	874	15.92	9.85	0.407	0.252	356	$3,586	$2,219	$3,135,000
OH	1,150	14.63	13.53	0.374	0.346	430	$3,297	$3,049	$3,790,446
OK	759	12.98	7.69	0.332	0.197	252	$2,924	$1,732	$2,218,085
PA	775	18.10	13.21	0.463	0.338	359	$4,078	$2,976	$3,158,572
RI	30	18.00	11.51	0.460	0.295	14	$4,055	$2,594	$121,504
SC	465	16.90	11.82	0.432	0.303	201	$3,808	$2,664	$1,771,388
SD	124	9.60	5.29	0.246	0.135	31	$2,163	$1,192	$269,236
TN	686	12.95	8.44	0.331	0.216	227	$2,917	$1,901	$2,002,465
TX	2,586	10.53	5.69	0.269	0.146	696	$2,373	$1,282	$6,134,208
VA	668	10.02	6.66	0.256	0.170	171	$2,257	$1,501	$1,507,902
VT	50	17.22	8.05	0.441	0.206	22	$3,880	$1,814	$192,461
WI	812	10.50	5.20	0.269	0.133	218	$2,365	$1,171	$1,920,550
WV	192	13.21	8.78	0.338	0.225	65	$2,976	$1,977	$570,259

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table.

Table 10-33. Summary of Estimated Mercury Exposures for Consumption-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2020 with CAIR[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (μg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2020; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	24,724	11.33	8.34	0.290	0.213	7,165	$7,485	$10,476	$63,100,915
State									
AL	509	9.05	6.51	0.232	0.166	118	$6,141	$6,676	$1,037,813
AR	514	14.22	17.84	0.364	0.456	187	$14,079	$18,853	$1,647,238
CT	234	15.59	10.13	0.399	0.259	93	$6,079	$6,149	$820,718
DC	12	6.83	2.48	0.175	0.063	2	$824	$636	$18,136
DE	40	9.44	3.83	0.242	0.098	10	$4,025	$2,572	$84,529
FL	1,447	18.03	9.63	0.461	0.246	668	$13,271	$18,187	$5,878,893
GA	1,331	11.72	7.91	0.300	0.202	399	$14,174	$17,435	$3,515,077
IA	382	7.44	3.78	0.190	0.097	73	$4,741	$5,054	$639,995
IL	1,516	7.74	4.16	0.198	0.107	300	$5,374	$5,914	$2,644,042
IN	875	9.66	5.52	0.247	0.141	216	$6,852	$7,988	$1,904,965
KS	404	17.44	12.14	0.446	0.311	180	$15,136	$15,844	$1,589,252
KY	583	8.51	4.42	0.218	0.113	127	$6,283	$5,304	$1,118,388
LA	690	12.61	7.73	0.323	0.198	223	$9,201	$8,479	$1,959,919
MA	313	17.74	10.02	0.454	0.256	142	$4,807	$3,952	$1,249,821
MD	294	6.33	2.91	0.162	0.075	48	$2,268	$1,840	$419,500
ME	113	23.73	10.51	0.607	0.269	69	$11,602	$10,809	$603,321
MI	900	10.52	5.69	0.269	0.146	242	$4,774	$4,888	$2,134,093
MN	1,224	8.39	4.14	0.215	0.106	263	$11,582	$10,530	$2,316,415
MO	852	11.89	8.86	0.304	0.227	259	$10,289	$12,852	$2,284,126
MS	392	11.15	5.13	0.285	0.131	112	$9,472	$7,507	$985,087
NC	930	12.02	10.12	0.308	0.259	286	$9,228	$10,983	$2,519,356

(continued)

Table 10-33. Summary of Estimated Mercury Exposures for Consumption-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2020 with CAIR (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2020; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	123	9.85	5.41	0.252	0.138	31	$8,054	$5,855	$273,830
NE	282	8.07	3.61	0.206	0.092	58	$5,896	$4,237	$512,923
NH	64	17.30	5.13	0.443	0.131	28	$7,138	$3,622	$249,817
NJ	402	14.30	11.13	0.366	0.285	147	$3,713	$3,428	$1,295,668
NY	908	14.09	8.81	0.360	0.225	327	$3,723	$4,724	$2,881,885
OH	1,176	11.19	11.31	0.286	0.289	337	$6,163	$10,259	$2,964,188
OK	820	12.11	6.61	0.310	0.169	254	$13,318	$10,817	$2,237,462
PA	746	14.42	9.76	0.369	0.250	275	$4,615	$4,977	$2,422,759
RI	33	15.93	8.03	0.407	0.206	13	$3,679	$3,200	$117,744
SC	496	14.36	9.95	0.367	0.254	182	$10,696	$12,041	$1,604,395
SD	120	8.75	4.38	0.224	0.112	27	$6,765	$3,700	$236,769
TN	763	10.48	7.36	0.268	0.188	205	$9,441	$11,605	$1,803,311
TX	3,418	9.62	5.48	0.246	0.140	841	$10,962	$15,938	$7,410,196
VA	765	8.20	5.08	0.210	0.130	161	$5,238	$4,930	$1,414,187
VT	45	15.17	5.87	0.388	0.150	17	$5,454	$2,936	$152,718
WI	835	9.53	4.26	0.244	0.109	204	$8,155	$7,315	$1,794,118
WV	174	9.13	5.05	0.233	0.129	41	$4,714	$2,898	$358,264

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table. For comparison purposes with the Base Cases in 2001, benefits presented in this table do not incorporate potential lags in fish tissue response to a change in mercury deposition.

Table 10-34. Summary of Annual Benefit Estimates for Consumption-Based Subsistence Population: Population Centroid Approach

	Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	10-Yr Lag	**20-Yr Lag**	**5-Yr Lag**	**50-Yr Lag**
Annual Number of Prenatally Exposed Children				
2001 Base Case	23,232	24,955	22,747	32,450
2020 Base Case (with CAIR)	27,126	29,661	25,859	37,266
Per Capita Avoided IQ Decrements				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)	0.0439	0.0432	0.0442	0.0423
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)	0.0462	0.0469	0.0451	0.0449
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)	0.0152	0.0151	0.0153	0.0149
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)	0.0032	0.0032	0.0032	0.0032
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)	0.0048	0.0048	0.0048	0.0047
Total Value of Benefits (1999$s)				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)				
1% Discount Rate; Present Value in 2001	N/A	N/A	N/A	$7,358,513
3% Discount Rate; Present Value in 2001	$6,678,524	$5,255,162	$7,636,211	$2,760,562
7% Discount Rate; Present Value in 2001	$4,562,627	$2,452,760	$6,311,683	$410,837
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$7,794,954
3% Discount Rate; Present Value in 2020	$7,036,390	$5,701,309	$7,749,431	$2,924,924
7% Discount Rate; Present Value in 2020	$4,807,114	$2,660,991	$4,807,114	$435,204
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$2,967,472
3% Discount Rate; Present Value in 2020	$2,705,281	$2,185,277	$3,001,948	$1,113,254
7% Discount Rate; Present Value in 2020	$1,848,191	$1,019,942	$2,481,249	$165,679
BENEFITS OF CONTROL OPTIONS				
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$630,695
3% Discount Rate; Present Value in 2020	$573,373	$463,559	$635,939	$236,607
7% Discount Rate; Present Value in 2020	$391,716	$216,358	$525,633	$35,213
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$945,748
3% Discount Rate; Present Value in 2020	$850,046	$689,683	$940,902	$354,799
7% Discount Rate; Present Value in 2020	$580,733	$321,898	$777,699	$52,803

0.015 points. CAMR Options 1 and 2, by comparison, are estimated to reduce them by an average of 0.003 and 0.005 points respectively.

Under CAMR Option 1, the aggregate benefits for this subsistence population assuming 10 and 20 year lag periods are estimated to be \$0.6 million and \$0.5 million, respectively, for those prenatally exposed in 2020. These estimates are 21 percent as large as those for the 2020 Utility Emissions Zero-Out. For CAMR Option 2, these values are estimated to both be \$0.9 million and 31 percent as large as corresponding estimates for the 2020 Utility Emissions Zero-Out.

10.6.2 Mercury Ingestion Estimates for Individuals in Low Income, High Fish Consumption Households

Another potentially high-risk subpopulation in the U.S. is comprised of individuals in low income households who rely on self-caught freshwater fish as part of their diet (EPA, 2002b). Although studies have documented these types of behaviors among low income groups in specific locations in the U.S. (West, 1992; Belton, Roundy, and Weinstein, 1986), a broad definition of this behavior at the national level is not available.

To assess potentially high exposures among low income subpopulations, the population centroid approach was adapted to focus specifically on low-income populations. Mercury ingestion levels were estimated for this subpopulation under a defined set of assumptions. First, Census data were used to restrict the analysis to women of childbearing age in households with incomes below \$10,000.[26] That is, NF in Eq. (10.2) (i.e., number of females ages 15-44) was redefined to include only low income women of childbearing age from each block group. State-level fertility rate data were again used to estimate the number of low-income pregnant women in each block group.

Second, to estimate the portion of these women that reside in angler households, results from the NSFHWR (Pullis, 2000) were used, which found that angler participation rates among low income individuals are on average 35 percent lower than for the general population in the U.S. The state-level participation rate estimates were scaled down (NA_s/N_s in Eq. [10.3]) for the general population by 35 percent to estimate the annual number of prenatally exposed children in low-income angler households in each block group.

Third, it was assumed that 50 percent of these prenatally exposed children are in households that rely on self-caught freshwater fish. This assumption was based in part on data from NSRE 1994, which found that roughly half of the single-day freshwater fishing trips by low income anglers were to waterbodies within 20 miles of their residence. Therefore, it was assumed that low income individuals making single day trips to waterbodies within 20 miles represent subsistence-type (rather than strictly recreational) fishing behaviors.

Fourth, to match these individuals in low-income, high fish consumption households with mercury concentrations, average mercury concentrations within 20 miles of each block group

[26] Poverty level for a 2-3 person household in 2000 was \$11,239-\$13,738 (www.census.gov/hhes/poverty/threshld/thresh00.html)

were calculated. To weight lake and river concentrations in this calculation, the same proportion of lake and river fishing (c_l, c_k in Eqs. [10.4] and [10.5]) was assumed for low income households as for the general population in each block group.

Fifth, to estimate mercury ingestion rates for these subpopulations, the same model described in Eq. (10.1) was applied; however, the daily fish consumption rate (C) was allowed to vary randomly. To create a full distribution of consumption rates that is consistent with the assumption that subsistence fishers consume more than the 95th percentile amount of fish, consumption rates were randomly drawn from the tail beyond the 95th percentile of the log-normal distribution defined in Section 10.5. Then, a different sampled consumption rate was randomly assigned to each of the roughly 165,000 block groups in the study area. The population centroid approach was then reapplied.

Table 10-35 reports detailed result for the 2001 Base Case. The annual number of prenatally exposed children in this high risk group was estimated to be 22,400 and the mean HgI across block groups was 12.44 μg/day[27]. Average IQ decrements in this group were estimated to be 0.32 points and the total present value of foregone earnings was estimated to be $63 million.

Table 10-36 summarizes similar model results for the 2020 Base Case with CAIR. The annual number of prenatally exposed children in this high risk group in 2020 was estimated to be 24,100 and the mean HgI across block groups was 10.96 μg/day, which is 12 percent lower than in the 2001 Base Case. Average IQ decrements in this group were estimated to be 0.28 points and the total present value of foregone earnings was estimated to be $60 million.

Table 10-37 reports estimates of beneficial changes to this income-based subsistence population under the five emissions control scenarios, including estimates for five lag periods and three assumed discount rates. The 2001 Utility Emissions Zero-Out and the 2020 Base Case with CAIR are both estimated to result in per capita IQ decrements that are on average between 0.035 and 0.042 less than under the 2001 Base Case. Compared to the 2020 Base Case with CAIR, the 2020 Utility Emissions Zero-Out is estimated to reduce per capita IQ decrements by an average of 0.015 points. CAMR Options 1 and 2, by comparison, are estimated to reduce them by an average of 0.003 and 0.005 points respectively.

Under CAMR Option 1, the aggregate benefits for this subsistence population assuming 10 and 20 year lag periods are estimated to be $0.6 million and $0.5 million, respectively, for those prenatally exposed in 2020. These estimates are 21 percent as large as the corresponding estimates for the 2020 Utility Emissions Zero-Out. For CAMR Option 2, these values are estimated to be $0.8 million and $0.7 million, respectively, which are 31 percent as large as the estimates for the 2020 Utility Emissions Zero-Out.

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[27] The average daily mercury ingestion rate given here of 12.44 ug/day from freshwater fish is two times the EPA's Reference Dose (RfD) for mercury of 5.8 ug/day. This estimate does not account for total exposure from consumption from other fish sources. See Section 11 of this report for a detailed discussion of the implication of the RfD on this analysis.

10.6.3 Mercury Ingestion Estimates for Two Selected Ethnic Populations

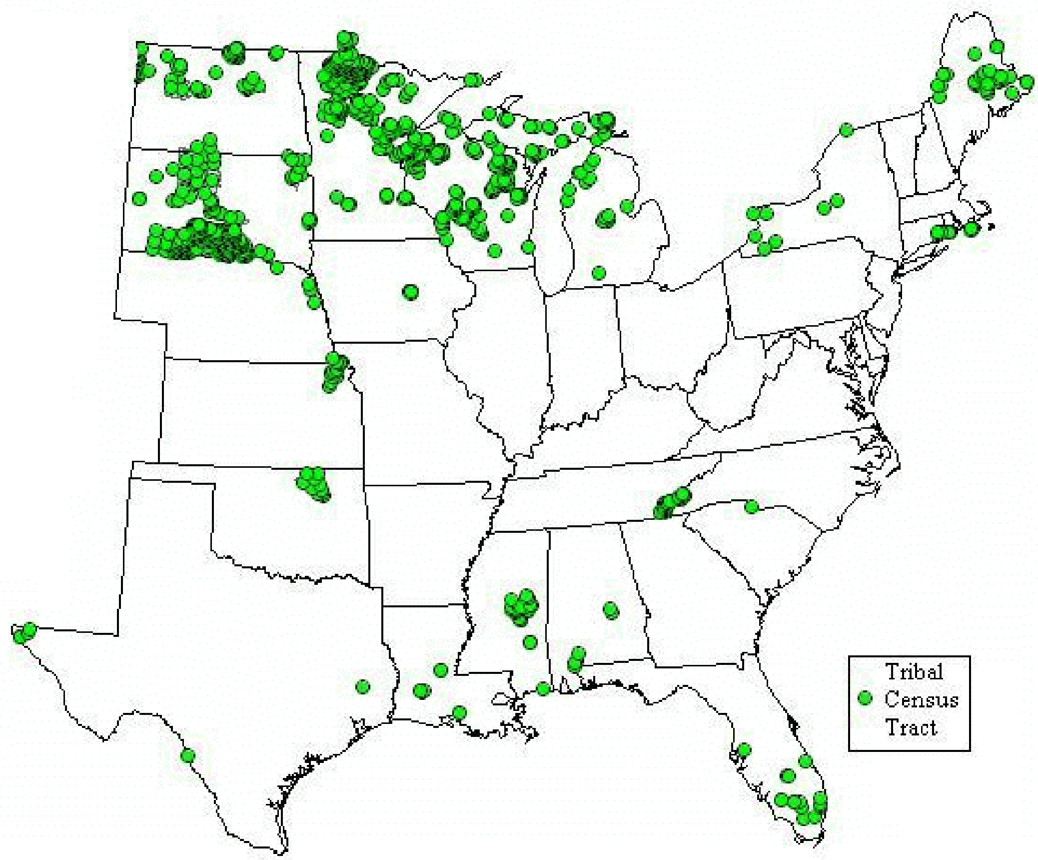
To further analyze mercury ingestion levels among potential high risk subpopulations, a separate analysis was conducted focusing on two selected special populations that are primarily located within a subregion of the study area:

1. Hmong in Minnesota and Wisconsin; and
2. Chippewa (Ojibwa) in Minnesota, Wisconsin, and Michigan.

These groups were selected for the following reasons. First, among ethnic groups in the United States, Southeast Asians and Native Americans have traditionally had relative high rates of fish consumption (EPA, 1997c). The Hmong are Southeast Asians of primarily Laotian origin, and the Chippewa are a Native American tribe from the Great Lakes area. As discussed in more detail below, studies investigating the fishing behaviors of these two groups have found relatively high rates of participation in freshwater fishing activities and of freshwater fish consumption.

Second, large portions of both the Hmong and Chippewa populations are located in three states within our 37-state study area. Other than California, the Hmong population in the United States has primarily settled in Minnesota and Wisconsin, where they now number over 75,000. The Chippewa are among the five most populous tribes in the United States, numbering over 100,000 in 2000 and are primarily located in Minnesota, Wisconsin, and Michigan. The Chippewa account for a large majority of the Native American population in the study areas of this benefit analysis, as is evident by Figure 10-19, which displays U.S. Bureau defined Native American tracts.

Third, due to the availability of Census data and fishing behavior studies for the Hmong and Chippewa, the exposure assessment methods developed for the recreational freshwater angler can be adapted for these two subpopulations. In particular, the population centroid approach described in the previous sections can be adapted to include some of the specific conditions of these special populations



Source: US Census Bureau 2004.

Figure 10-19. U.S. Census Tracts with Native American Populations

Table 10-35. Summary of Estimated Mercury Exposures for Income-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2001 Base Case[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (μg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	22,393	12.44	12.67	0.318	0.324	7,126	$2,803	$2,803	$62,756,741
State									
AL	732	10.00	9.15	0.256	0.234	187	$2,254	$2,061	$1,649,964
AR	663	12.96	9.75	0.332	0.249	220	$2,921	$2,197	$1,935,007
CT	158	22.21	18.23	0.568	0.466	90	$5,004	$4,108	$792,822
DC	20	5.04	2.77	0.129	0.071	3	$1,135	$625	$22,162
DE	31	8.84	5.53	0.226	0.142	7	$1,993	$1,247	$62,171
FL	1,059	19.64	13.16	0.502	0.337	532	$4,425	$2,965	$4,686,540
GA	1,027	13.86	10.76	0.355	0.275	364	$3,122	$2,424	$3,207,259
IA	348	6.21	3.52	0.159	0.090	55	$1,398	$794	$486,319
IL	1,222	6.33	4.78	0.162	0.122	198	$1,426	$1,076	$1,743,592
IN	565	9.04	6.42	0.231	0.164	131	$2,037	$1,448	$1,150,670
KS	360	16.98	14.12	0.434	0.361	156	$3,825	$3,183	$1,378,019
KY	716	11.46	7.86	0.293	0.201	210	$2,583	$1,771	$1,849,315
LA	855	13.07	13.05	0.334	0.334	286	$2,946	$2,941	$2,517,166
MA	259	21.59	14.11	0.552	0.361	143	$4,865	$3,179	$1,261,763
MD	199	6.53	4.72	0.167	0.121	33	$1,472	$1,063	$292,363
ME	119	28.78	15.65	0.736	0.401	88	$6,485	$3,528	$771,948
MI	750	12.09	8.07	0.309	0.207	232	$2,725	$1,819	$2,043,439
MN	742	7.60	5.36	0.194	0.137	144	$1,712	$1,207	$1,270,006
MO	849	10.68	9.73	0.273	0.249	232	$2,407	$2,194	$2,042,312
MS	651	13.29	8.60	0.340	0.220	221	$2,996	$1,938	$1,950,276
NC	733	14.32	9.29	0.366	0.238	269	$3,227	$2,093	$2,365,451

(continued)

Table 10-35. Summary of Estimated Mercury Exposures for Income-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2001 Base Case (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2001; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	119	12.52	12.12	0.320	0.310	38	$2,821	$2,731	$336,550
NE	222	7.33	5.16	0.188	0.132	42	$1,653	$1,162	$366,374
NH	58	24.69	17.20	0.632	0.440	37	$5,563	$3,876	$325,028
NJ	254	15.54	11.42	0.398	0.292	101	$3,501	$2,573	$888,373
NY	1,066	15.77	12.04	0.404	0.308	430	$3,554	$2,712	$3,790,552
OH	1,029	13.28	10.01	0.340	0.256	350	$2,993	$2,256	$3,079,702
OK	812	12.06	8.15	0.309	0.209	250	$2,717	$1,837	$2,205,353
PA	771	19.57	27.90	0.501	0.714	386	$4,409	$6,288	$3,398,897
RI	43	21.14	11.23	0.541	0.287	23	$4,764	$2,530	$206,753
SC	539	16.90	14.54	0.432	0.372	233	$3,807	$3,277	$2,052,104
SD	132	10.05	7.39	0.257	0.189	34	$2,264	$1,665	$298,184
TN	792	11.47	8.36	0.294	0.214	232	$2,585	$1,883	$2,047,437
TX	3,101	9.69	7.08	0.248	0.181	769	$2,184	$1,596	$6,771,389
VA	455	9.28	7.86	0.238	0.201	108	$2,092	$1,772	$951,503
VT	52	18.47	11.99	0.473	0.307	25	$4,162	$2,703	$218,402
WI	599	10.26	7.53	0.263	0.193	157	$2,313	$1,698	$1,384,873
WV	292	14.53	9.51	0.372	0.243	109	$3,275	$2,144	$956,703

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table. For comparison purposes with the Base Cases in 2020, benefits presented in this table do not incorporate potential lags in fish tissue response to a change in mercury deposition.

Table 10-36. Summary of Estimated Mercury Exposures for Income-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2020 with CAIR[a]

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2020; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
Study Area	24,132	10.96	9.58	0.280	0.245	6,769	$2,470	$2,159	$59,613,908
State									
AL	707	8.51	9.04	0.218	0.231	154	$1,918	$2,037	$1,356,146
AR	711	11.95	8.37	0.306	0.214	217	$2,693	$1,886	$1,914,546
CT	171	19.42	19.97	0.497	0.511	85	$4,376	$4,500	$747,241
DC	17	3.17	1.47	0.081	0.038	1	$715	$331	$12,454
DE	34	7.58	4.70	0.194	0.120	7	$1,708	$1,059	$57,258
FL	1,342	18.37	11.62	0.470	0.297	631	$4,139	$2,618	$5,553,735
GA	1,147	11.72	9.45	0.300	0.242	344	$2,642	$2,130	$3,029,686
IA	340	6.00	3.19	0.154	0.082	52	$1,352	$720	$460,380
IL	1,284	5.74	3.64	0.147	0.093	188	$1,292	$821	$1,659,781
IN	598	7.60	5.09	0.195	0.130	116	$1,714	$1,147	$1,024,603
KS	389	16.56	12.77	0.424	0.327	165	$3,733	$2,878	$1,451,555
KY	704	8.72	6.09	0.223	0.156	157	$1,964	$1,372	$1,382,855
LA	842	12.18	11.74	0.312	0.300	262	$2,745	$2,645	$2,311,469
MA	268	19.24	12.96	0.492	0.332	132	$4,335	$2,921	$1,163,445
MD	199	5.21	4.14	0.133	0.106	27	$1,174	$934	$233,479
ME	106	26.54	14.04	0.679	0.359	72	$5,980	$3,163	$631,624
MI	753	10.18	6.79	0.260	0.174	196	$2,294	$1,531	$1,727,063
MN	790	7.90	5.64	0.202	0.144	160	$1,780	$1,270	$1,405,268
MO	871	10.27	9.50	0.263	0.243	229	$2,313	$2,140	$2,015,424
MS	634	12.17	7.57	0.311	0.194	197	$2,743	$1,706	$1,739,298
NC	862	11.38	7.66	0.291	0.196	251	$2,563	$1,727	$2,209,022

(continued)

Table 10-36. Summary of Estimated Mercury Exposures for Income-Based Subsistence Population, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—2020 with CAIR (continued)

	Annual Number of Prenatally Exposed Children	Average Daily Maternal Ingestion of Mercury (µg/day/person)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (in 2020; 1999$)		
		Mean	S.D.	Mean	S.D.	Total	Mean	S.D.	Total
State									
ND	115	11.60	10.78	0.297	0.276	34	$2,613	$2,429	$300,095
NE	249	6.95	4.58	0.178	0.117	44	$1,566	$1,032	$389,257
NH	59	22.42	15.67	0.574	0.401	34	$5,052	$3,530	$300,514
NJ	278	15.33	10.79	0.392	0.276	109	$3,455	$2,432	$961,970
NY	1,116	15.09	10.47	0.386	0.268	431	$3,402	$2,359	$3,796,552
OH	1,026	9.57	6.77	0.245	0.173	251	$2,157	$1,525	$2,211,994
OK	869	11.44	7.66	0.293	0.196	254	$2,578	$1,726	$2,240,049
PA	727	14.99	19.86	0.383	0.508	279	$3,377	$4,474	$2,453,711
RI	47	18.67	9.39	0.478	0.240	23	$4,208	$2,115	$198,873
SC	563	14.35	11.49	0.367	0.294	207	$3,233	$2,589	$1,819,762
SD	135	9.53	8.28	0.244	0.212	33	$2,147	$1,865	$288,880
TN	836	9.25	6.58	0.237	0.168	198	$2,085	$1,484	$1,742,925
TX	3,968	8.98	6.45	0.230	0.165	912	$2,024	$1,453	$8,033,117
VA	462	7.59	6.91	0.194	0.177	90	$1,711	$1,557	$791,208
VT	47	16.09	9.65	0.412	0.247	19	$3,625	$2,174	$169,340
WI	609	9.38	6.03	0.240	0.154	146	$2,113	$1,359	$1,287,089
WV	258	9.32	9.88	0.239	0.253	62	$2,101	$2,227	$542,241

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table. For comparison purposes with the Base Cases in 2001, benefits presented in this table do not incorporate potential lags in fish tissue response to a change in mercury deposition.

Table 10-37. Summary of Annual Benefit Estimates for Income-Based Subsistence Population: Population Centroid Approach

	Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	10-Yr Lag	**20-Yr Lag**	**5-Yr Lag**	**50-Yr Lag**
Annual Number of Prenatally Exposed Children				
2001 Base Case	23,037	24,310	22,699	30,548
2020 Base Case (with CAIR)	27,126	29,661	25,859	37,266
Per Capita Avoided IQ Decrements				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)	0.042	0.041	0.043	0.040
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)	0.035	0.034	0.035	0.033
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)	0.015	0.015	0.015	0.014
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)	0.003	0.003	0.003	0.003
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)	0.005	0.005	0.005	0.004
Total Value of Benefits (1999$s)				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)				
1% Discount Rate; Present Value in 2001	N/A	N/A	N/A	$6,556,561
3% Discount Rate; Present Value in 2001	$6,379,897	$4,905,014	$7,362,516	$2,459,708
7% Discount Rate; Present Value in 2001	$4,358,611	$2,289,334	$6,085,461	$366,063
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$6,499,636
3% Discount Rate; Present Value in 2020	$6,139,097	$4,905,679	$6,853,900	$2,438,352
7% Discount Rate; Present Value in 2020	$4,194,102	$2,289,645	$5,665,066	$362,885
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$2,851,684
3% Discount Rate; Present Value in 2020	$2,717,338	$2,165,647	$3,038,206	$1,069,816
7% Discount Rate; Present Value in 2020	$1,856,428	$1,010,780	$2,511,218	$159,214
BENEFITS OF CONTROL OPTIONS				
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$593,633
3% Discount Rate; Present Value in 2020	$572,354	$454,554	$641,184	$222,703
7% Discount Rate; Present Value in 2020	$391,020	$212,155	$529,968	$33,143
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$885,766
3% Discount Rate; Present Value in 2020	$840,630	$670,774	$939,259	$332,297
7% Discount Rate; Present Value in 2020	$574,301	$313,073	$776,341	$49,454

10.6.4 Adaptation of the Population Centroid Approach to Estimate Exposed Hmong and Chippewa Population

The first step in applying the population centroid approach is to estimate the size and location of the relevant populations. The approach was adapted to address limitations in the Census data for analyzing specific ethnic subpopulations.

One of the limitations of Census data is that demographic data for specific racial groups, such as Hmong and Chippewa, are not available at a block group level. However, they are available at a Census tract level and at higher levels of spatial aggregation (e.g., county). To apply the population centroid model, tract-level data were first used to approximate the number of Hmong and Chippewa females aged 15 to 44 in each block group. First, for each tract, the percentage of its *total population* residing in each block group was calculated. These same percentages were then applied to divide the Hmong and Chippewa populations into each block group. In effect, it was assumed that distribution of the Hmong and Chippewa populations across block groups in a tract is the same as the distribution of the total population in the tract.

Another limitation of Census data is that, if a specific racial group numbers less than 100 in a tract, the population size for this group is not reported in Census tract-level data. Because this analysis does not include these unreported populations, it underestimate the total size of the relevant Hmong and Chippewa populations.

Table 10-38 summarizes the resulting block group population estimates in 2001 for females aged 15 to 44. State-level population growth rate projections to update Census 2000 data to 2001. Using tract-level data, Hmong populations could be estimated for roughly 7 percent of the Census block groups in Minnesota and about 9 percent in Wisconsin. Chippewas populations could be estimated in Michigan and Wisconsin for about 1 percent of block groups and in Minnesota for 4 percent. The estimated number of females of childbearing age in each block group ranged from 0 to 184 for the Hmong and from 0 to 662 for the Chippewa.

As shown in Table 10-38, the total number of Hmong women of childbearing age in the two states is estimated to be roughly 13,300, which is about 16 percent of the total Hmong population. Since women of childbearing age typically represent about 20 percent of the overall population, it is expected that the analysis based on tract level Census data underestimated the exposed Hmong population by about 20 percent. For Chippewa, a similar calculation suggests

Table 10-38. Block Group Demographics for Hmong and Chippewa Females, Aged 15 to 44 (in 2001)

State	Total Hmong Population	Total Chippewa Population	Total Number of Block Groups	Number of Block Groups		Hmong Female Population, Aged 15–44, per Block Group[a,b]			Chippewa Female Population, Aged 15–44, per Block Group[a,b]		
				With Hmong Populations[b]	With Chippewa Populations[b]	Mean	S.D.	Total	Mean	S.D.	Total
MI	—	32,267	8,450	—	103	—	—	—	23.33	29.84	2,403
MN	45,530	39,910	4,082	279	166	28.56	29.66	7,969	29.44	69.72	4,888
WI	36,809	16,560	4,388	398	48	13.29	13.45	5,290	37.73	56.48	1,811
Total	82,339	88,737	16,920	677	317	19.58	22.91	13,259	28.71	57.64	9,101

[a]For Census tracts with Hmong or Chippewa populations of less than 100, population is extrapolated from county-level data.
[b]Estimate extrapolated from Census tract-level data.

that the exposed Chippewa population was underestimated by about 50 percent. The uncounted populations are presumably in tracts with less than 100 Hmong or Chippewa.

The second step is to estimate the number of *pregnant* women by Census block group for each of the two special populations (denoted by subscript T). For the recreational freshwater angler analysis, state-level fertility rate data were used to estimate numbers of pregnant women. Fertility rates for broad ethnic groups are not available at a state level, but they are available at a national level (Hamilton, Sutton, and Ventura, 2003). Therefore, state-level fertility rates for the general population were adjusted by the ratio of (1) national-level fertility rates for each ethnic group to (2) national-level fertility rate for the general population. For the two special populations, the number of pregnant women in each Census block was estimated as follows.

$$\text{Number of pregnant women } (NP_T) = NF_T * f_s * (f_{NT} / f_N) \qquad \text{(Eq. 10.19)}$$

where

- NF_T = number of females aged 15 to 44 (Census) for the given special population,
- f_s = state-level fertility rate for the general population (live births per 1,000 women aged 15 to 44),
- f_N = national fertility rate for the general population (live births per 1,000 women aged 15 to 44), and
- f_{NT} = national fertility rate for Asians/Pacific Islander (including Hmong) or American Indians (including Chippewa) (live births per 1,000 women aged 15 to 44).

For Minnesota, Wisconsin, and Michigan, the state-level fertility rates for the general population in 2001 (f_s) were 61.8, 59.6, and 62 live births respectively per 1,000 women aged 15 to 44. The national fertility rate for the general population (f_N) was 65.3 births per 1,000. For Asians/Pacific Islanders and for American Indians, the national fertility rates were 64.2 and 58.1 respectively.

In the third step, the number of pregnant women in each block group who live in households with freshwater anglers was estimated. For both groups, the size of this exposed population of interest was estimated as:

$$NPA_T = NP_T * AH_T \qquad \text{(Eq. 10.20)}$$

where AH_T = percentage of households with freshwater anglers.

For the Hmong, results from a study by Hutchison and Kraft (1994) were used to estimate AH_T. This study examined a random sample of 125 Hmong households from Green Bay, Wisconsin, and collected data on fishing frequency, fish consumption frequency, fishing travel distances, types of fish caught and consumed, and other related behaviors. In their sample, 57.6 percent of the households were freshwater anglers. For this analysis, it was assumed that this same percentage applies to all Hmong households in Minnesota and Wisconsin.

For the Chippewa, this third step (estimation of AH_T and NPA_T) was not applied because, as discussed below, the freshwater fish consumption rate estimate available for the

Chippewa is not restricted to only angler households. Therefore, to maintain consistency with this consumption rate estimate, the exposed Chippewa population was not restricted to only include those in angler households.

After estimating the exposed populations of interest, the population centroid approach was applied to estimate the average mercury concentration in consumed freshwater fish and the rate of mercury ingestion for the exposed population so interest.

To estimate the percentage of trips to different distance categories for the Hmong, results from Hutchison and Kraft (1994) were again used. In their study, the maximum freshwater fishing trip distance for the Hmong was 48 miles; thus, it was assumed that all trips by Hmong anglers are within 50 miles of their residence. Within the 50-mile radius, trips were allocated to the three distance categories in the same proportion (23:17:23) as for the recreational freshwater angler. Therefore, the percentage of Hmong freshwater fishing trips in the three distance categories of 0–10 miles, >10–20 miles, and >20–50 miles were assumed to be 36.5 percent, 27 percent, and 36.5 percent, respectively.

No specific data on average distance traveled to fish are available for the Chippewa. However, information in Peterson et al. (1994) suggests that most fish is locally caught. Therefore, the same trip distance assumptions that were applied to the Hmong were also applied to the Chippewa. In other words, it was assumed that all fishing trips are within 50 miles and that, within this radius, they are distributed in the same proportion as for the recreational freshwater angler.

To further adapt the model for the two special populations, daily consumption rate (C) estimates based on specific data for these populations were included in the analysis. For the Hmong, a consumption rate of 21.2 g/day was included, which was calculated using the consumption frequency data reported by Hutchison and Kraft (1994). Using their summary data, the average number of fish meals per year (34.1) for anglers in the sample was estimated. This estimate was then multiplied by the assumed average fish meal size of 8 oz/meal (227 g/meal) reported in Hutchison and Kraft and divided by 365 to estimate the average daily fish consumption rate.

For the Chippewa, a daily consumption rate of 20 g/day was used, based on recommendations from EPA's *Exposure Factor Handbook* (EPA, 1997b). EPA's analysis used data from Peterson et al. (1994) to estimate a mean freshwater fish consumption rate (1.2 meals per week) and Pao et al. (1982) to estimate the average weight of a fish meal (117 g/meal). The Peterson et al. (1994) study specifically examined the fish consumption habits and blood mercury levels of Chippewa Indians from northern Wisconsin, using a random sample of 175 and a nonrandom sample of 152 tribal members. This daily consumption rate is based on a general sample of Chippewa adults, not only on a sample of those who fish or who consume freshwater fish. Therefore, this rate can reasonably be applied to the total estimated exposed population of Chippewa, rather than to a subset from angler households.

Summary and Discussion of Results. Table 10-39 summarizes estimates of the size of exposed population of interest. In 2001 the mean population of prenatally exposed children per block group was estimated to be 0.12 and 0.08, and the total exposed population estimate was

553 and 10,947 across the study area for the Hmong and Chippewa respectively. By 2021 (after a 20 year lag), the Hmong population of prenatally exposed children is expected almost double, and the exposed Chippewa population was projected to grow by over 60 percent.

The estimated average daily maternal ingestion rates (HgI) across the states in 2001 are reported in Table 10-40. For the Hmong, HgI was estimated to be 4.46 μg/day and per capita IQ decrements were estimated to be 0.11 points. For the Chippewa, the HgI estimate is 5.2 μg/day and the per capita IQ decrements were estimated to be 0.13 points. Under baseline conditions in 2001, the present value of total IQ related losses for the Hmong was estimated to be \$0.6 million. For the Chippewa, it was estimated to be \$1.3 million.

Table 10-41 reports comparable results for the 2020 Based with CAIR. For the Hmong, HgI was estimated to be 4.46 μg/day and per capita IQ decrements were estimated to be 0.11 points for the two states (MN and WI) combined. These estimates are the same as for the 2001 Base Case, but they reflect higher estimated average mercury concentrations for block groups in MN (relative to the 2001 baseline) and lower concentrations in WI. For the Chippewa, the HgI estimate is 4.7 μg/day and the per capita IQ decrements were estimated to be 0.12 points. Under baseline conditions in 2001, the present value of total IQ related losses for the Hmong was estimated to be \$1 million. For the Chippewa, it was estimated to be \$1.9 million. Both of these estimates are higher than for the 2001 Base Case due to the relatively large increase in exposed populations from 2001 to 2020 (as reported in Table 10-39)

Tables 10-42 and 10-43 report estimates of beneficial changes to the two subpopulations under the five emissions control scenarios, including estimates for five lag periods and three assumed discount rates. For the Hmong, the 2001 Utility Emissions Zero-Out is estimated to result in changes in per capita IQ decrements that range from 0.0007 less to 0.0012 more than under the 2001 Base Case, and the 2020 Base Case with CAIR is estimated to result in per capita IQ decrements that are between 0.0069 and 0.0078 less. Compared to the 2020 Base Case with CAIR, the 2020 Utility Emissions Zero-Out is estimated to reduce per capita IQ decrements by an average of 0.0075 points. CAMR Options 1 and 2, by comparison, are estimated to reduce them by an average of 0.0004 and 0.0014 points respectively.

For the Chippewa, the 2001 Utility Emissions Zero-Out is estimated to result in per capita IQ decrements that are approximately 0.023 less than under the 2001 Base Case, and the 2020 Base Case with CAIR is estimated to result in per capita IQ decrements that are 0.015 less. Compared to the 2020 Base Case with CAIR, the 2020 Utility Emissions Zero-Out is estimated to reduce per capita IQ decrements by an average of 0.013 points. CAMR Options 1 and 2, by comparison, are estimated to reduce them by an average of 0.001 and 0.002 points respectively.

Table 10-39. Estimated Annual Number of Prenatally Exposed Children from Special Populations for Selected Lag Periods: Population Centroid Approach

	LAG Periods from 2001														
	Base Year of Comparison (2001)			5 Year Lag (2006)			10 Year Lag (2011)			20 Year Lag (2021)			50 Year Lag (2051)		
Population	Per Block Group		Total Exposed	Per Block Group		Total Exposed	Per Block Group		Total Exposed	Per Block Group		Total Exposed	Per Block Group		Total Exposed
State	Mean	S.D.	Population	Mean	S.D.	Population	Mean	S.D.	Population	Mean	S.D.	Population	Mean	S.D.	Population
Hmong															
MN	0.136	0.449	321	0.165	0.544	390	0.192	0.636	452	0.238	0.796	561	0.422	1.357	995
WI	0.102	0.206	232	0.124	0.255	282	0.147	0.305	333	0.195	0.416	445	0.421	0.996	958
2-State Total	0.119	0.352	553	0.145	0.428	672	0.170	0.502	786	0.217	0.639	1,006	0.421	1.193	1,953
Chippewa															
MI	0.051	0.203	400	0.058	0.221	451	0.063	0.241	495	0.074	0.265	575	0.136	0.490	1,064
MN	0.172	0.912	506	0.208	1.099	613	0.235	1.226	693	0.293	1.474	862	0.607	3.060	1,788
WI	0.061	0.395	189	0.075	0.484	229	0.085	0.547	260	0.107	0.701	330	0.230	1.693	706
3-State Total	0.079	0.487	1,094	0.094	0.584	1,293	0.105	0.651	1,448	0.128	0.787	1,767	0.257	1.674	3,559

	LAG Periods from 2020														
	Base Year of Comparison (2020)			5 Year Lag (2025)			10 Year Lag (2030)			20 Year Lag (2040)			50 Year Lag (2070)		
Population	Per Block Group		Total Exposed	Per Block Group		Total Exposed	Per Block Group		Total Exposed	Per Block Group		Total Exposed	Per Block Group		Total Exposed
State	Mean	S.D.	Population	Mean	S.D.	Population	Mean	S.D.	Population	Mean	S.D.	Population	Mean	S.D.	Population
Hmong															
MN	0.269	0.899	634	0.264	0.882	621	0.297	0.993	701	0.364	1.215	859	0.565	1.816	1,333
WI	0.180	0.396	410	0.222	0.482	506	0.261	0.576	595	0.339	0.767	772	0.582	1.362	1,325
2-State Total	0.226	0.700	1,045	0.243	0.715	1,127	0.280	0.815	1,295	0.352	1.020	1,632	0.574	1.609	2,657
Chippewa															
MI	0.073	0.247	569	0.078	0.281	611	0.090	0.316	699	0.112	0.395	875	0.177	0.669	1,386
MN	0.310	1.794	914	0.322	1.589	949	0.375	1.836	1,104	0.480	2.344	1,413	0.826	4.157	2,432
WI	0.091	0.587	280	0.120	0.797	370	0.142	0.951	435	0.184	1.267	566	0.306	2.178	938
3-State Total	0.128	0.897	1,763	0.140	0.856	1,930	0.162	0.994	2,238	0.206	1.279	2,854	0.344	2.248	4,756

Table 10-40. Summary of Estimated Mercury Exposures for Special Populations in 2001, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—Base Case 2001

		Average Daily Maternal Ingestion of Mercury (ug/day)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (1999$s in 2001)		
Population	**State**	**Mean**	**S.D.**	**Mean**	**S.D.**	**Total**	**Mean**	**S.D.**	**Total**
Hmong									
	MN	3.783	0.364	0.097	0.009	31.106	$852	$82	$273,959
	WI	5.392	1.513	0.138	0.039	31.961	$1,215	$341	$281,492
	2-State Total	4.457	1.276	0.114	0.033	63.067	$1,004	$288	$555,451
Chippewa									
	MI	6.406	1.075	0.164	0.028	65.523	$1,444	$242	$577,082
	MN	4.057	0.976	0.104	0.025	52.526	$914	$220	$462,613
	WI	5.815	1.318	0.149	0.034	28.060	$1,310	$297	$247,128
	3-State Total	5.218	1.505	0.134	0.039	146.109	$1,176	$339	$1,286,823

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table.

Table 10-41. Summary of Estimated Mercury Exposures for Special Populations in 2020, with Associated IQ Decrements and Foregone Earnings: Population Centroid Approach—Base Case 2020 with CAIR[a]

		Average Daily Maternal Ingestion of Mercury (ug/day)		IQ Decrements in Prenatally Exposed Children			Present Value of Foregone Net Earnings due to IQ Decrements (1999$s in 2020)		
Population	**State**	**Mean**	**S.D.**	**Mean**	**S.D.**	**Total**	**Mean**	**S.D.**	**Total**
Hmong									
	MN	4.040	0.311	0.103	0.008	65.563	$910	$70	$577,428
	WI	5.101	1.570	0.131	0.040	53.566	$1,149	$354	$471,769
	2-State Total	4.456	1.112	0.114	0.028	119.128	$1,004	$251	$1,049,197
Chippewa									
	MI	5.733	1.311	0.147	0.034	83.480	$1,292	$295	$735,236
	MN	3.904	0.901	0.100	0.023	91.320	$880	$203	$804,278
	WI	5.506	1.445	0.141	0.037	39.476	$1,241	$326	$347,674
	3-State Total	4.749	1.325	0.122	0.034	214.276	$1,070	$299	$1,887,187

[a] Benefits analyses using the population centroid approach were conducted at a block group level, but for summary purposes the results are aggregated and reported at a state level in this table. For comparison purposes with the Base Cases in 2001, benefits presented in this table do not incorporate potential lags in fish tissue response to a change in mercury deposition.

Table 10-42. Summary of Annual Benefit Estimates for Hmong Special Population: Population Centroid Approach

	Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr Lag
Hmong (MN and WI)				
Annual Number of Prenatally Exposed Children				
2001 Base Case	786	1,006	672	1,953
2020 Base Case (with CAIR)	1,295	1,632	1,127	2,657
Per Capita Avoided IQ Decrements				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)	0.0096	0.0095	0.0097	0.0094
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)	0.0006	0.0008	0.0004	0.0012
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)	0.0075	0.0076	0.0074	0.0078
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)	0.0004	0.0004	0.0004	0.0004
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)	0.0015	0.0016	0.0015	0.0016
Total Value of Benefits (1999$s)				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)				
1% Discount Rate; Present Value in 2001	N/A	N/A	N/A	$98,779
3% Discount Rate; Present Value in 2001	$49,613	$46,758	$49,506	$37,057
7% Discount Rate; Present Value in 2001	$33,894	$21,824	$40,919	$5,515
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$16,402
3% Discount Rate; Present Value in 2020	$5,071	$6,539	$3,725	$6,153
7% Discount Rate; Present Value in 2020	$3,465	$3,052	$3,079	$916
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$110,358
3% Discount Rate; Present Value in 2020	$63,540	$60,308	$63,511	$41,401
7% Discount Rate; Present Value in 2020	$43,409	$28,148	$52,495	$6,161
BENEFITS OF CONTROL OPTIONS				
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$6,076
3% Discount Rate; Present Value in 2020	$3,477	$3,311	$3,466	$2,280
7% Discount Rate; Present Value in 2020	$2,375	$1,546	$2,865	$339
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$23,043
3% Discount Rate; Present Value in 2020	$12,968	$12,417	$12,877	$8,644
7% Discount Rate; Present Value in 2020	$8,859	$5,796	$10,643	$1,287

Table 10-43. Summary of Annual Benefit Estimates for Chippewa Special Population: Population Centroid Approach

	Central Estimate of Fish Tissue Response Times		Alternative Estimate of Fish Tissue Response Times	
	10-Yr Lag	20-Yr Lag	5-Yr Lag	50-Yr Lag
Chippewa (MI, MN, and WI)				
Annual Number of Prenatally Exposed Children				
2001 Base Case	1,448	1,767	1,293	3,559
2020 Base Case (with CAIR)	2,238	2,854	1,930	4,756
Per Capita Avoided IQ Decrements				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)	0.022	0.021	0.024	0.021
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)	0.015	0.016	0.015	0.015
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)	0.013	0.013	0.013	0.013
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)	0.001	0.001	0.001	0.001
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)	0.002	0.002	0.002	0.002
Total Value of Benefits (1999$s)				
2001 Utility Emissions Zero-Out (Relative to 2001 Base Case)				
1% Discount Rate; Present Value in 2001	N/A	N/A	N/A	$214,504
3% Discount Rate; Present Value in 2001	$115,066	$102,720	$120,309	$80,472
7% Discount Rate; Present Value in 2001	$78,611	$47,943	$99,441	$11,976
2020 Base Case with CAIR (Relative to 2001 Base Case applied to 2020 demographics)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$220,164
3% Discount Rate; Present Value in 2020	$130,050	$123,387	$130,029	$82,595
7% Discount Rate; Present Value in 2020	$88,847	$57,589	$107,475	$12,292
2020 Utility Emissions Zero-Out (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$190,562
3% Discount Rate; Present Value in 2020	$113,612	$106,942	$114,256	$71,490
7% Discount Rate; Present Value in 2020	$77,618	$49,913	$94,438	$10,639
BENEFITS OF CONTROL OPTIONS				
2020 CAMR **Control Option 1** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$11,266
3% Discount Rate; Present Value in 2020	$6,698	$6,331	$6,716	$4,227
7% Discount Rate; Present Value in 2020	$4,576	$2,955	$5,551	$629
2020 CAMR **Control Option 2** (Relative to 2020 Base Case with CAIR)				
1% Discount Rate; Present Value in 2020	N/A	N/A	N/A	$26,212
3% Discount Rate; Present Value in 2020	$15,331	$14,543	$15,331	$9,834
7% Discount Rate; Present Value in 2020	$10,474	$6,788	$12,671	$1,463

For the Hmong subpopulation, the aggregate benefits of CAMR Option 1 assuming 10 and 20 year lag periods are estimated to be $3,500 and $3,300 respectively. These estimates are 5 percent as large as the corresponding estimates for the 2020 Utility Emissions Zero-Out. For CAMR Option 2, these estimates are $14,000 and $12,400 respectively, which are 20 percent as large as the corresponding estimates for the 2020 Utility Emissions Zero-Out.

For the Chippewa subpopulation, the aggregate benefits of CAMR Option 1 assuming 10 and 20 year lag periods are estimated to be $6,700 and $6,300 respectively. These estimates are 6 percent as large as the corresponding estimates for the 2020 Utility Emissions Zero-Out. For CAMR Option 2, these estimates are $15,300 and $14,500 respectively, which are 14 percent as large as the corresponding estimates for the 2020 Utility Emissions Zero-Out.

The analyses presented in this section examining distributional equity in the context of the Hmong and Chippewa were constrained significantly by not having considered variability in fish consumption rates for these two high fish consuming populations. Challenges in identifying peer-reviewed data characterizing high-end percentile self-caught freshwater fish consumption rates for these two populations prevented IQ benefits modeling from considering inter-individual variability in fish consumption rates. While this is not problematic in the context of modeling overall (mean) benefits for the population as a whole, it does significantly limit the utility of considering per-capita IQ benefits in the context of distributional equity, since a key factor determining high-end benefits levels is not considered (i.e., high-end fish consumption rates which can result in disproportionately large IQ benefits, if matched to large mercury fish tissue concentration changes).

In order to provide additional insights into the distributional equity issue (and address to a certain extent this limitation in characterizing high-end fish consumption in modeling both the Hmong and Chippewa), EPA has included a Sensitivity Analysis specifically examining the issue of distributional equity for high fish consuming populations. As discussed below, this sensitivity analysis uses high-end (potentially bounding) fish consumption rates identified through NODA comments to support an analysis of distributional equity for Native American populations in the 37 state study area."

10.6.5 Sensitivity Analysis Examining the Economic Benefit Equity Issue in the Context of High Fish Consuming (subsistence) Populations Including Native Americans

There is the potential that, due to elevated fish consumption rates, individuals exhibiting subsistence-like fishing behavior could experience disproportionately higher benefits from this rule (i.e., the degree of per-capita IQ benefit for prenatally-exposed children of these fishers could be significantly higher than the degree of IQ benefit predicted for the general recreational angler population). Another way of viewing this issue is that EGU-attributable mercury impacts could disproportionately impact these high fish-consuming populations and consequently, this rule could have relatively larger benefits for these special populations by reducing their disproportionate health impacts. If this is the case (i.e., special high-consumption populations are found to have disproportionately larger benefits then the general recreational angler), then it be justified to support this rule on economic equity grounds, regardless of the net economic benefit-cost ratio.

Ideally, this distributional equity issue would be examined by systematically modeling population-level benefits for potentially high exposure populations including Native American populations exhibiting subsistence-like behavior. Specifically, using the population centroid model combined with (a) a probabilistic consideration of fish consumption rate variability in the study population, (b) demographic data on the distribution of these special populations across the study area and (c) modeled delta fish tissue concentrations for the CAMR rule, we would model the distribution of benefits (i.e., IQ loss reductions) across specific high consumption populations as a result of CAMR. In this context, modeling exposure using *fish consumption variability distributions* is critical since ultimately, we are interested in higher-end exposures for these populations which likely reflect a combination of high fish consumption rates paired with relatively higher mercury fish tissue concentrations changes (for the rule being modeled).

In developing the benefits model for this RIA, EPA attempted to conduct this type of focused benefits modeling for two key potentially high-consuming populations including the Hmong and Chippewa (the former located in Minnesota, Wisconsin and Michigan and the latter in Minnesota and Wisconsin - see Section 10.6.1). These two populations were chosen because they are known to have higher fish consumption rates and because mercury fish tissue concentrations in these areas are fairly large relative to other portions of the study area. However, several factors surfaced in modeling these two population which prevented a comprehensive analysis of these populations in the context of distributional equity including (a) it was not possible to establish fish consumption distributions for either populations since we could not identify a defensible upper-end percentile consumption rate for use in fitting a distribution and (b) ultimately, air quality modeling showed that the regions where these two populations are located are not generally associated with the highest EGU-related mercury deposition changes (these high deposition change areas are located primarily near the Ohio River valley). These two factors diminished the ability of the benefits analysis, as conducted for the Hmong and Chippewa to effectively address the potential for distributional equity concerns regarding these two populations.

However, given the potential importance that EPA places on distributional equity as a consideration in cost-benefit analysis, we have completed a sensitivity analysis designed to evaluate the potential for disproportionate health benefits for high fish consumption (subsistence) populations. The methodology used in this sensitivity analysis and the results of the analysis are presented in this section. Specifically, we have selected a high-end (near bounding) fish consumption rate (discussed below) for Native American populations and combined this value with upper-bound mercury fish tissue concentration changes (deltas) for Option 1 identified through several scenarios (described below) to estimate upper bound IQ benefits for these populations across the 37 state study area. This sensitivity analysis, while not allowing any enumeration of these populations, does allow us to determine whether there is the potential for disproportionate health impacts for Native American populations and other high fish consuming populations based on a combination of these conservative exposure assumptions.

A determination that disproportionate health benefits are experienced by a special population is ultimately based on consideration of two factors: (a) are the levels of health benefits (in this case IQ loss reductions for prenatally-exposed children) for the special population relatively larger than the general recreational angler population and (b) in absolute terms, are the magnitude of the health benefits experienced by the special population considered significant. This second point is important to emphasize. Even if a special population has relatively larger health benefits (e.g., an order of magnitude larger than the general recreational angler), if those health benefits are considered non-significant in absolute terms (e.g., less than 1 IQ point is saved based on conservative high-end exposure modeling), then it may be reasonable to conclude that there is little support for an distributional equity argument since the absolute level of health benefit involved is not considered significant.

<u>Fish Consumption Values Used in the Sensitivity Analysis</u>

Because this sensitivity analysis is considering high-end conditions for exposure in the context of this distributional equity analysis for Native American populations, the fish consumption rate selected needed to represent high-end (near bounding) behavior, if possible. The EPA recommends a 95^{th}% consumption rate of 170 g/day for Native American subsistence fishing populations in its Exposure Factors Handbook (EPA, 1997). This consumption rate would be a reasonable upper-bound for Native American populations and could arguably be used in this sensitivity analysis. However, in comments received for the NODA to this rule, fish consumption rates specifically for the Ojibwa in Minnesota, Wisconsin and Michigan were identified. These values covered consumption rates for the spring (189.6-393.8 grams/day) and fall (155.8 to 240.7 grams/day) spear fishing seasons. Because this sensitivity analysis is attempting to determine whether an equity issue exists (i.e., are there individuals with disproportionate health benefits), EPA decided that rather than using the EPA-recommended value of 170 g/day, it would develop an even more conservative (near bounding) value based on the NODA comment data. Specifically, EPA used the highest seasonal value provided by the commentor and apply that to the full year (i.e., assume that annual-averaged daily fish consumption is 393.8 grams/day). This value is very conservative and is not appropriate for other portions of this analysis since it is not possible to determine the percentile of the Ojibwe population that it represents (i.e., it could be a near max/bounding value, or it could be closer to a 95^{th}%). However, because this distributional equity analysis is intended to screen for potential disparities in health benefits, use of a very conservative bounding analysis is considered reasonable in this case.

<u>Selection of Fish Tissue Concentration Changes (deltas) Under Option 1 for the Sensitivity Analysis</u>

The sensitivity analysis was implemented using the change in fish tissue concentration (for selected locations) due to Option 1. This is the relevant metric to use in examining the distributional equity issue since we are interested in seeing whether special high consumption populations experience disproportionately higher health benefits <u>because of the CAMR rule</u>. However, this being said, there were several different ways in which specific fish tissue concentration values could be selected (e.g., use the Option 1 concentrations modeled in the RIA for the Chippewa, be more conservative and select the maximum fish tissue concentrations under Option 1 across the states where the Chippewa are located). Ultimately, EPA evaluated three different fish tissue concentration scenarios for this sensitivity analysis, each representing a different perspective on fish tissue concentration locations:

Scenario 1 - use fish tissue concentration values modeled in the RIA for the Chippewa: This scenario essentially represents applying the high-end fish consumption rate described above to the Chippewa population modeled above. With this scenario, we selected the maximum Option 1 fish tissue concentration modeled for the Chippewa in the RIA (i.e., the maximum ring-averaged fish tissue concentration estimated for the Chippewa using the Population Centroid Model.

Scenario 2 - select the maximum Option 1-attributable fish tissue concentration change predicted anywhere within Minnesota, Wisconsin and Michigan (i.e., identify a single maximum fish tissue concentration value for each state): This is a more conservative scenario than Scenario 1 since it assumes that Native Americans could be fishing anywhere in these three states. In addition, while Scenario 1 is based on averaging of standardized fish tissue concentrations across the 6 predicted (edible) fish species and across simulation years (which is reasonable for the economic analysis as conducted for the primary estimate), this scenario takes the more conservative approach of selecting the maximum fish tissue concentration change that will reflect a specific fish species in a specific year (i.e., Scenario 2 does not average across species or time and therefore will represent a higher-end concentration change value compared with Scenario 1).

Scenario 3 - identify the maximum Option 1 fish tissue concentration change for locations within the 37 state study area where Native Americans are located: This scenario expands the sensitivity analysis to include any HUCs co-located with tribal census tracts as defined by the US Census. Specifically, the maximum fish tissue concentration (not averaged across species or year) for each HUC was identified and used in the sensitivity analysis (Note: only the max value found across the entire study area is presented in the results table).

EPA believes that these three scenarios provide reasonable coverage for Native American Populations within the 37 state study area, although it is important to note that smaller populations of Native Americans may be located in other locations not covered by the sensitivity analysis. It is also important to note that, this sensitivity analysis (specifically Scenario 2) also provides coverage for the Hmong, since that scenario uses the maximum fish concentration change that is found in three of the states where Hmong are primarily located.

Results of the Sensitivity Analysis

The results of the sensitivity analysis examining the distributional equity issue are presented in Table 10-44. This table identifies both the mercury fish tissue concentration change (delta) values under Option 1 and the fish species (where applicable) and provides the IQ changes modeled for this analysis.[28] As described above, all results generated for this sensitivity analysis were based on a conservative (near bounding) consumption rate of 393.8 g/day which EPA identified through NODA comments. This value, while appropriate for a sensitivity analysis, was not used in other components of this analysis (e.g., primary benefits estimate) because it is not possible to clearly identify which percentile of the subsistence population this value represents.

[28] In generating these IQ changes, EPA used the same methodology applied in the general RIA for IQ change estimation with the exception that the consumption rate described above (393.8 g/day) was used (see Section 10.1.3.1 and 10.1.3.2 for additional details on IQ change calculation). Modeling of IQ change for the sensitivity analysis did include application of the cooking loss factor of 1.5 in predicting exposure levels.

Table 10-44. Results of the Sensitivity Analysis Examining Distributional Equity for Native American (subsistence) Populations

Sensitivity Analysis Scenario	Option 1-attributable mercury fish tissue concentration change (ppm)	Species modeled	Predicted IQ change for prenatally-exposed children
Scenario 1: maximum fish concentration change for Chippewa in the RIA			
Chippewa value	0.00083	average across 6 species	0.012
Scenario 2: maximum fish concentration change in MI, WI, and MN not limited to where Native Americans are located (and not averaged across species or year)			
Michigan	0.0212	Walleye	0.32
Wisconsin	0.0207	Walleye	0.31
Minnesota	0.0069	Walleye	0.10
Scenario 3: maximum fish concentration change in HUCs within the 37 state study area where Native Americans are located (based on Native American Census block designation)			
Max value (MS)	0.0406	Walleye	0.61

The results presented in Table 10-44 suggest that there may not be a strong argument for distributional equity for Native American (subsistence) populations within the 37 state study area (given the precision of the fisher exposure model used here - see discussion below). Even the most conservative scenario evaluated for the sensitivity analysis (involving the Option 3-attributable fish tissue concentration change of 0.0406 ppm paired with the high consumption rate used in this analysis), only produced a IQ change of 0.61 points. Although it is likely that high consuming Native American (subsistence) populations, as well as other high consuming populations do experience relatively higher benefits compared with the general recreational angler, because the absolute degree of health benefit (in terms of IQ points saved) is still relatively low (i.e., significantly less than 1), we conclude that a compelling argument can not be made on distributional equity grounds for this rule.[29]

The issue of distributional equity can also be raised in the context of high fish consuming (near subsistence) recreational anglers (i.e., is there a relatively smaller number of recreational anglers who consume at the high-end of the consumption distribution and experience health benefits significantly larger than the recreational freshwater angler of recreational anglers)? Because we conducted modeling for recreational anglers for the RIA with consideration for fish consumption variability by this population, it is possible to examine the tail of the *Option 1 - attributable IQ change distribution* to determine whether high-end consumers in this population might experience disproportionate health benefits. The maximum modeled IQ benefit for the recreational angler as a result of Option 1-related mercury fish tissue concentration reductions is

[29] The recreational angler population modeled for the RIA has a mean IQ change due to Option 1 attributable mercury fish tissue concentrations changes of 0.010, which is over an order of magnitude lower than the maximum IQ change predicted in the Sensitivity Analysis.

0.28 IQ points, which results in the same conclusion as drawn for the Native American populations (i.e., this degree of health benefit may be significantly higher than the general recreational angler population with its mean IQ change of 0.0010, but it is still considered not significant from an absolute perspective since it is significantly less than 1 IQ point).

Note, that the conclusions presented above are based on the fish consumption exposure model used in this RIA (with modifications as noted above) and consequently reflect the level of precision and accuracy inherent in this model. Because this economic benefits model was not developed to support site-specific analysis of individual waterbodies and the populations that fish at those waterbodies, there is the potential for this analysis to have overlooked individuals who may be subjected to higher absolute benefits because they fish at waterbodies where the Option 1 mercury fish concentration change is significantly larger than what is captured in the three Scenarios described above.

10.7 Discussion and Qualification of Results: Assumptions, Limitations, and Uncertainties

The previously described methods and results provide useful insights regarding:

1. The extent of potentially harmful mercury exposures due to consumption of noncommercial freshwater fish;

2. The size and distribution of resulting IQ losses among prenatally exposed children,;

3. The value of future earnings losses associated with these IQ decrements; and

4. The extent to which these losses could be avoided through emissions controls on coal-fired power plants.

However, these estimates are based on modeling approaches that require simplifying assumptions and are subject to limitations and uncertainties.

Uncertainty regarding the estimates developed in this section can arise from several sources. Some of the uncertainty can be attributed to model uncertainty. For example, to estimate exposures a number of different modeling approaches have been selected and combined, each of which simplifies potentially complex processes. The results therefore depend importantly on how these models are selected and specified.

One strategy used to evaluate the potential effects of model uncertainty has been to define alternative estimation approaches within the exposures assessment. For instance, two separate approaches—population centroid and angler destination—were developed to estimate the size of the exposed populations and their relation to mercury concentrations in fish. Also, to investigate exposures among high-risk groups three different approaches were summarized in Section 10.6. The ranges of exposure estimates reported for these different approaches help to demonstrate the sensitivity of results with respect to model selection.

Another important source of uncertainty can be characterized as input or parameter uncertainties. Each of the modeling components discussed in this report requires summary data and estimates of key model parameters. All of these inputs are measured with some degree of uncertainty and can affect, to differing degrees, the confidence range of our summary results. The discussion below identifies and highlights some of the key model parameters, characterizes the source and extent of uncertainties associated with them, and characterizes the potential effects of these uncertainties on the model results.

To organize this discussion, different components of the modeling framework are discussed separately. It first discusses issues related to estimating the mercury concentrations and then those related to estimating the exposed population. After that, it discusses issues related to matching these two components and then concludes by discussing the estimation of mercury ingestion through fish consumption.

10.7.1 Mercury Concentration Estimates

As described in Section 10.1.2, the core mercury concentration estimates for the analysis come from the fish tissue sample data in the NLFA. These estimates were then used to approximate mercury concentrations across the study area for a specific time period (2001) and for normalized conditions (size, species, and cut of fish). Some of the key assumptions, limitations, and uncertainties associated with these estimates are the following:

- The NLFA data themselves are subject to measurement and reporting error and variability. The NLFA is the largest and most detailed source of data on mercury in fish; however, the system was not centrally designed (e.g., by EPA) using a common set of sampling and analytical methods. Rather, states collected the data primarily to support the development of advisories, and the data are submitted voluntarily to EPA. Each state uses different methods and criteria for sampling and allocates different levels of resources to their monitoring programs. In addition, there are uncertainties regarding the precise locations (lat/long coordinates) of some of the samples. The heterogeneity and potential errors across state sampling programs can bias the results in any direction and contribute to uncertainty.

- The NLFA sampling data were assigned as either lake or river samples, based on the location coordinates for sampling sites in the NLFA and by mapping them with GIS to the nearest type of waterbody. This process also involves measurement error and may have resulted in misclassifications for some of the samples. These errors are not expected to bias results, but they contribute to uncertainty.

- The NDMMFT statistical model described in Section 10.1.1 was applied to the NLFA to estimate concentrations at specific locations and dates under normalized conditions (size, species, and cut of fish). These normalized estimates were then averaged across species and dates to create a single mercury concentration estimate for each of the roughly 5,200 sampling locations in the study area. This process of normalizing and averaging results involves both model uncertainty and estimation error. For example, if anglers systematically keep fish of different species or sizes than those included in the normalization and averaging process, then the modeling approach may lead to over- or underestimates of exposures.

- The normalized/averaged mercury concentration estimates (summarized in Table 10-2) were then spatially extrapolated, as discussed in Section 10.1.2. These spatial extrapolation processes are potentially significant sources of uncertainty and may also overstate actual mercury levels. In the population centroid approach, average concentrations within specified distance intervals from block group centroids were first extrapolated to all waterbodies within the interval, and then further averaged and extrapolated to other distance intervals as well. In the angler destination approaches, average concentrations in a HUC were used to characterize fish from all waterbodies in the HUC. All of these approaches assume that NLFA mercury samples are representative of "local" conditions in similar waterbodies. However, even though states use a variety of approaches to monitor and sample fish tissue contaminants, in some cases, the sampling sites are selected to target areas with high levels of angler activity and/or a high level of pollution potential. To the extent that sample selection procedures favor areas with relatively high mercury, the spatial extrapolation methods used in this report will tend to overstate exposures. These approaches also implicitly assume that mercury concentration estimates are strongly spatially correlated, such that closer sampling sites (i.e., from same HUC or distance interval) provide more information about mercury concentrations than more distant sites. To the extent that spatial correlation is weaker than assumed, this will increase the degree of uncertainty in the modeling results.

- To estimate mercury fish tissue concentrations under each of the emissions reductions scenarios, it was assumed that concentrations in any given area would decline in exactly the same proportion as modeled reductions in mercury deposition for the area. This assumption was based on Mercury Maps, which also assumes a linear, steady-state relationship between concentrations of methylmercury in fish and air deposition of mercury. However, Mercury Maps also recognizes that this condition may not be met in waterbodies that contain significant non-air sources of mercury. To assess the potential effect that nonair sources of mercury would have on the results reported in this section, EPA identified 56 HUCs in the 37-state study area with gold or mercury mines or chloralkali plants. EPA recalculated aggregate benefits in the study area for the 2001 Utility Emissions Zero-Out scenario assuming that concentrations in these 56 HUCs would not be reduced at all by the emissions controls. The resulting benefit estimates declined by less than 3 percent. These results suggest that overestimation of benefits due to exclusion of sources is relatively small. This analysis did not attempt to identify or otherwise account for naturally-occurring sources of mercury. To the extent that these are present and substantially affect concentrations in the water bodies of interest, the benefits will be overestimated.

10.7.2 Exposed Population Estimates

Section 10.1.2 describes two main approaches for estimating the annual number of children prenatally exposed to mercury because of their mothers' consumption of noncommercial freshwater fish. The population centroid approach addresses this objective by focusing on where these women of childbearing age are most likely to live in relation to mercury levels in freshwater fish. In contrast, the angler destination approach focuses on where angler households are most likely to go fishing, in relation to where mercury concentrations in freshwater fish are located.

Each approach has advantages and disadvantages. The main advantage of the population centroid approach is that it uses detailed Census data and allows us to characterize and subdivide populations with a high degree of confidence at a high level of spatial resolution (almost 165,000 block groups in the study area). A main disadvantage of this approach is that it requires strong assumptions for identifying the portion of these populations that live in freshwater angler households and for matching these populations to mercury levels in fish. The main advantage of the angler destination approach is that it starts by identifying angler populations and where they fish; however, it requires relatively strong assumptions for linking these populations to populations of women of childbearing age who consume their catch.

In certain respects, the two approaches rely on similar data sources and use them in similar ways; therefore, they are subject to similar uncertainties.

- Both approaches rely primarily on data from the NSFHWR to estimate state-level freshwater angler activity levels. The NSFHWR is based on a sample of over 10,000 freshwater anglers nationwide. The population centroid approach primarily uses the NSFHWR to estimate state-level freshwater fishing participation rates and lake-to-river day ratios. The angler destination approach uses data on the level of lake- and river-fishing days by state. Each of these data elements is measured with some error in the NSFHWR, but they are based on a relatively large sample. As discussed below, more uncertainty is generated in both approaches when these state-level estimates are applied or extrapolated to smaller spatial scales (block groups and HUCs).

- Both approaches also use state-level fertility rate data to approximate the rate of pregnancy among women of childbearing age in angler households for a smaller geographic area. The state-level fertility rates from the National Vital Statistics are estimated with relatively little error; however, applying these rates to specific block groups or HUCs (and specifically to women in angler households) does involve considerably more uncertainty.

In other respects, the two approaches use very different assumptions and are subject to different sources of uncertainty in measuring exposed populations. Moreover, as discussed below, some of these assumptions are likely to lead to underestimates of exposed populations in the population centroid approach and to overestimates in the other approach.

- The population centroid approach assumes that, in each block group, the percentage of women who live in freshwater angler *households* (i.e., households with at least one freshwater angler) is equal to the percentage of the state adult *population* that fishes. Applying the state-level participation rate to approximate the conditions at a block level creates uncertainty. More importantly, however, using individual-based fishing participation rates to approximate household rates is likely to underestimate the percentage of women living in freshwater angler households.[30] Unfortunately data on household participation levels in freshwater fishing are not readily available.

- In the angler destination approach, it is assumed that the percentage of self-caught fish from a HUC that goes to households with women of childbearing age is equal to the sum of (1) the state-level percentage of anglers who are women of childbearing age plus (2) the state-level percentage of anglers who are male, married, and in the same age range as women of childbearing age. In other words, it assumes that women who are exposed during pregnancy must either be anglers themselves or be married to an adult male angler below the age of 45. On the one hand, this approach may not capture women who receive noncommercial fish from other individuals (besides husbands younger than 45), in which case it would underestimate the size of the exposed population. On the other hand, and probably more importantly, this approach is likely to double count women of childbearing age who meet both criteria (anglers and married to angler men younger than 45). The extent of this double counting is not known, but it would lead to an overestimation of the exposed population.

- The angler destination approach also estimates exposures by estimating "angler-year equivalents" for each HUC. Angler-year equivalents can be thought of as groups of freshwater fishing days in a HUC, such that 16.42 fishing days in a year (U.S. annual average for freshwater anglers) are equivalent to one angler year. This approach assumes that the same expected *total* mercury ingestion by pregnant women will result from these fishing days, regardless of whether the days are all spent by one or by many anglers.[31]

10.7.3 Matching of Exposed Populations to Mercury Concentrations

Section 10.1.3 also describes how the two approaches were used to match the estimated exposed populations in each geographic area with corresponding mercury levels in freshwater fish. In the population centroid approach, this entails matching (1) different portions of the estimated number of pregnant women in angler households in each block group with (2) average mercury concentration estimates in different waterbody types and distance intervals. In the angler destination approach, this entails matching (1) different portions of freshwater fishing days in each state with (2) average mercury concentration estimates in each HUC in the state.

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[30] For example, hypothetically if one out of every three members in each household fished, the population rate would be 33 percent, but the household rate would be 100 percent.

[31] If the dose-response relationship for health effects with respect to mercury ingestion is linear (as is assumed), the aggregate health effects resulting from ingestion should only depend on the total mercury ingestion and not on how the ingestion is distributed among the exposed population.

In the population centroid approach, subpopulations are assigned to waterbody types based on state-level ratios of lake-to-river fishing days (from the NSFHWR). They are further assigned to distance intervals based on observed travel distance patterns in national fishing data (NSRE 1994). An important limitation of both methods is that neither one takes into account the physical characteristics of the area in which the population is located. In particular, the allocation of exposures to lakes or rivers at different distances from each block group does not take into account the presence or number of these waterbodies in each distance interval.

Using the NSRE 1994 data to assign subpopulations to distance intervals involves additional uncertainty because it is based on more aggregate data and on a much smaller sample of anglers than the NSFHWR. Additional uncertainty is also introduced by using *self-reported* travel distance estimates to calculate travel distance patterns[3232]. Moreover, the analysis does not control for the potential effect of trip frequency on trip distance and resulting exposures. For example, if anglers who fish more often are also more likely to fish closer to home, then mercury concentrations within the smaller distance intervals should be weighted more heavily in their average mercury exposure estimates. Because a constant average freshwater fish consumption rates is assumed in the main analysis (8 g/day), variations in fishing (and therefore fish consumption) intensity are not accounted for. If there is a negative correlation between trip frequency and average travel distance, and if more frequent fishers are located in areas closer to higher (lower) average mercury concentration, then the population centroid approach is likely to underestimate (overestimate) mercury ingestion levels for frequent fishers and for the exposed population as a whole.

The angler destination approach also combines state-level (NSFHWR) and national data (NSRE 1994) to match anglers with mercury concentrations. However, in contrast to the population centroid approach, it allocates state-level lake- and river-fishing days based on both physical and demographic data. Rather than allocating fishing days in proportion to just one of these characteristics (e.g., HUC area), data from the NSRE were used to estimate and apply a multivariate model described in Appendix E-2. Although based on a relatively small sample of fishing trip data and a somewhat crude estimate of angler activity in each HUC, the model results indicate that several factors, including population size and number of lake/river miles per HUC, have statistically significant effects in explaining variations in fishing levels across HUCs. The intended purpose of this model for predicting the allocation of fishing days across HUCs (rather than a simpler allocation rule) is to reduce uncertainty by including as much information as possible about the HUCs. However, it does not eliminate uncertainty in these estimates. The modeled coefficients inherently include statistical error, and other unmeasured factors are also likely to contribute to variations in fishing levels across HUCs.

One potentially important factor that is not included in either model for matching populations and mercury concentrations is the effect of fish consumption advisories on fishing behavior. Evidence summarized in Jakus, McGuinness, and Krupnick (2002) suggests that awareness of advisories by anglers is relatively low (less than 50 percent), and even those who

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[32] In addition to errors in respondents' own assessments of travel distance, their estimates may be a more accurate reflection of road distance than the straight line distance used in the exposure calculations. If so, the analysis most likely overestimates the percentage of trips to more distant waterbodies. The effect of this potential overestimation on the average and total mercury exposure estimates is not known.

are aware do not always alter their fishing behavior. Nonetheless, anglers are less likely to fish in areas with advisories. Unfortunately, we were not able to reliably quantify the reduction and redistribution of fishing trips in either model to account for fish advisories. By excluding these effects, the model estimates are likely to overstate mercury exposures.

10.7.4 Fish Consumption Estimates

Perhaps the most influential variable in both modeling approaches is the rate of noncommercial freshwater fish consumption. Based on recommendations in EPA's EFH, we have assumed 8 g/day for the general population in freshwater angler households. Unfortunately, data are not available to reliably vary this rate with respect to characteristics of the population across the entire study area. In Section 10.6, we applied alternative consumption rate assumptions for selected Asian and Native American populations because existing studies have estimated consumption rates specifically for these populations, but they represent a small portion of our study population.

Uncertainty regarding the true average fish consumption rate has a direct effect on uncertainty for the model results. Because a single consumption rate is applied uniformly across the entire exposed population and because it is a multiplicative factor in the model (see Eq. [10.1]), the two uncertainties are directly proportional to one another. As discussed in Section 10.1.3, the recommended 8 g/day rate is based on four studies with mean estimates ranging from 5 (37 percent less than 8) to 17 (113 percent more than 8) g/day. If it is assumed that this range of estimates represents the uncertainty in the mean freshwater fish consumption rate for the study population, then the resulting uncertainty range for the estimated mean mercury ingestion level will also be between –37 percent and +113 percent of the mean mercury ingestion level.

For a number of reasons, average consumption rates of freshwater fish may well differ from the recommended 8 g/day value. For example, the rate may be less for females than for male anglers and even less for females in angler households who do not fish themselves. To the extent that their consumption rates differ, the model estimates are likely to overstate mercury exposure.

Because of fish consumption advisories of various types, women of childbearing age and pregnant women in particular may well reduce their levels of consumption. Some evidence of these types of behavioral changes were found, for example, in a recent study by Oken et al. (2003). To the extent that these types of changes occur, they suggest that the current model results overstate mercury exposures in the population of interest.

A final potentially influential variable in both modeling approaches is the assumed conversion factor for mercury concentrations between uncooked and cooked fish. Studies have found that cooking fish tends to reduce the overall weight of fish by approximately one-third (Great Lakes Sport Fish Advisory Task Force, 1993) without affecting the overall amount of mercury. But these conversion rates depend on cooking practices and types of fish. Uncertainty regarding this conversion factor also has a proportionate effect on the modeling results.

10.7.5 Modelling and Valuation of IQ Related Effects

The models for estimating and valuing IQ effects based on (1) estimated mercury ingestion levels and (2) the size of the exposed population involve three main steps. The first step is translating maternal mercury ingestion rates to mercury levels in hair. The second step is translating differences in hair mercury concentrations during pregnancy to IQ changes in offspring. The third step is translating IQ losses into expected reductions in lifetime earnings. As discussed below, each of these steps also involves uncertainty.

- The conversion of mercury ingestion rate to mercury concentration in hair is based on uncertainty analysis of a toxicokinetic model for estimating reference dose (Swartout and Rice, 2000). The conversion factor was estimated by considering the variability and uncertainty in various inputs used in deriving the dose including body weight, hair-to-blood mercury ratio, half-life of MeHg in blood, and others. Therefore, there is uncertainty regarding the conversion factor between hair mercury concentration and mercury ingestion rate. Although, the median conversion factor (0.08 μg/kg-day/hair-ppm) is used, the ninety-percent confidence interval is from 0.037 to 0.16 μg/kg-day/hair-ppm. Any change in the conversion factor will proportionately affect the benefits results because of the linearity of the model.

- The dose-response model used to estimate neurological effects on children because of maternal mercury body burden is susceptible to various uncertainties, as discussed in Section 9.4. In particular there are three main concerns. First, there are other cognitive end-points that have stronger association with MeHg than IQ point losses. Therefore, using IQ points as a primary endpoint in the benefits assessment may underestimate the impacts. Second, blood-to-hair ratio for mercury is uncertain which can cause the results from analyses based on mercury concentration in blood to be uncertain. Third, uncertainty is associated with the design of the epidemiological studies used in deriving the dose-response models and the differences between these studies[33]. More discussion on the uncertainty in dose-response function is provided in Section 9.4.

- The valuation of IQ losses is based on a unit-value approach developed by EPA, which estimates that the average effect of a 1 point reduction in IQ is to reduce the present value of net future earnings by $8,807 (in 1999$). Three key assumptions of this unit-value approach are that (1) there is a linear relationship between IQ changes and net earnings losses and (2) the unit value applies to even very small changes in IQ, and (3) the unit value will remain constant (in real present value terms) for several years into the future. Each of these assumptions contributes to uncertainty in the results. The unit value estimate is itself subject to two main sources of uncertainty. The first source is directly related to uncertainties inherent in the statistical analysis by Salkever (1995), which provides estimates of average reductions in future earnings and years in school as a result of IQ changes. The average percent change estimates are subject to statistical error, modeling uncertainties, and variability across the population. The second main source of

[33] One uncertainty from the epidemiological studies is that the cognitive test were performed on young children who continued to be exposed to mercury after birth.

uncertainty are the estimates of average lifetime earnings and costs of schooling. Both of these estimates are derived from national statistics from the early 1990s, but they are also subject to statistical error, modeling uncertainties, and variability across the population.

10.7.6 Unquantified Benefits

In addition to the uncertainties discussed above associated with the benefit analysis of reducing exposures to methylmercury from recreational freshwater angling, there are several additional benefits that we are unable to quantify, which adds to the uncertainties in the final estimate of benefits of the CAMR.

Table 10-45 displays the health and ecosystem effects associated with methylmercury exposure that are discussed in Section 2 for which we are currently unable to quantify.

Table 10-45. Unquantified Health and Ecosystem Effects Associated with Exposure to Mercury

Category of Health or Ecosystem Effect	Potential Health or Ecosystem Outcomes
Neurologic Effects	Impaired cognitive development Problems with language Abnormal social development
Cardiovascular Effects*	Potential for fatal and non-fatal myocardial infarctions (heart attacks) - See Appendix B for a detailed discussion
Genotoxic Effects*	Associations with genetic effects
Immunotoxic Effects*	Possible autoimmunity effects in antibodies
Ecological Effects*	Neurological effects in wildlife (birds, fish, and mammals) that is similar to humans

* These are potential effects and are not quantified because the literature is either contradictory or incomplete.

For one of these health effects, cardiovascular disease, the Agency conducted a critical review of the available literature and determined that while some studies show that the effect may exist, it is premature to include analysis of cardiovascular effects in our benefit analysis. Studies investigating the relationship between methylmercury exposure and cardiovascular impacts have reached different conclusions. The findings to date and the plausible biologic mechanisms warrant additional research in this area. Appendix B provides an in-depth discussion of these epidemiology studies. If future scientific studies demonstrate this effect occurs, the benefits of reduced cardiovascular effects (from fatal and non-fatal heart attacks) if quantified could possibly be many times larger than those we are able to quantify in this section of the report due to the potential for mortality effects (monetized with the value of a statistical life which is much higher in value than IQ loss).

In addition to the health and ecosystem effects that we are not able to quantify, there are exposures to other segments of the U.S. population that we are currently unable to quantify. In Section 4 of this report, we discuss the other fish consumption pathways that lead to exposure to methylmercury, including: consumption of commercial seafood and freshwater fish (produced domestically as well as imported from foreign sources), and consumption of recreationally caught seafood from estuaries, coastal waters, and the deep ocean. These consumption pathways impact additional recreational anglers who are not modeled in our benefit analysis as well as the general U.S. population. Reductions in domestic fish-tissue concentrations can also impact the health of foreign consumers (consuming U.S. exports). Due to technical/theoretical limitations in the science, EPA is unable to quantify the benefits associated with several of these fish consumption pathways. For example, reductions in U.S. power plant emissions will result in a lowering of the global burden of elemental mercury, which will likely produce some degree of reduction in mercury concentrations for fish sourced from the open ocean and freshwater and estuarine waterbodies in foreign countries. In the case of mercury reductions for fish in the open ocean, complexities associated with modeling the linkage between changes in air deposition of mercury and reductions in biomagnification and bioaccumulation up the food chain (including open ocean dilution and the extensive migration patterns of certain high-consumption fish such as tuna) prevent the modeling of fish obtained from the open ocean. In the case of commercial fish obtained from foreign freshwater and estuarine waterbodies, while there are technical challenges associated with modeling long-range transport of elemental mercury and the subsequent impacts to fish in these distant locations, additional complexities such as accurately modeling patterns of harvesting and their linkages to commercial consumption in the U.S. prevent inclusion of foreign-sourced freshwater and estuarine fish in the primary benefits analysis.

Finally, with regard to commercially-produced freshwater fish sourced in the U.S. (i.e., fish from catfish, bass, and trout farms), we are unable to accurately quantify effects from this consumption pathway because many of the fish farms operating in the U.S. utilize feed that is not part of the aquatic foodweb of the waterbody containing the fish farm (e.g., use of agricultural-based supplemental feed). In addition, many of the farms involve artificial "constructed" waterbody environments that are a-typical of aquatic environments found in the regions where those farms are located, thereby limiting the applicability of Mercury Maps assumption in linking changes to mercury deposition to changes in mercury fish tissue concentrations (e.g., waterbodies may have restricted or absent watersheds and modified aquatic chemistry, which can effect methylation rates and impact time-scales for reaching steady-state mercury fish tissue concentrations following reductions in mercury deposition). Some research indicates that the recycling of water at fish farms can magnify the mercury concentration because the system does not remove mercury as it is recycled, while newly deposited mercury is added to the system. Thus, additional research on aquaculture farms is necessary before a benefit analysis can be conducted.

Exclusion of these commercial pathways means that this benefits analysis, while covering an important source of exposure to domestic mercury emissions (recreational freshwater anglers), excludes a large and potentially important group of individuals. As discussed in Section 4 of this report, recreational freshwater consumption accounts for approximately 10 - 17 percent of total U.S. fish consumption, and 90 percent is derived from commercial sources (domestic seafood, aquaculture, and imports). However, as is mentioned throughout this report, several of the other consumption pathways will not be affected by CAMR, or will be affected to at minimal levels.

Another area of unquantified benefits is associated with the study area selected for the benefit analysis conducted in this section, specifically the eastern 37 states of the continental U.S.. The focus on the eastern-half of the U.S. excludes evaluation of benefits from freshwater sources in the West as well as commercially produced fish in the Pacific (which produces 68% of the commercial fish supply in the U.S.). However, air quality modeling has shown that the largest change in deposition from U.S. power plants emissions of mercury will occur in the eastern-half of the U.S. so the unquantified benefits for this portion of the U.S. is expected to be quite small.

In conclusion, there are several unquantified benefits associated with this analysis that add to the overall uncertainty in the estimate of total benefits of CAMR. To the extent that CAMR will reduce mercury deposition from power plants over estuarine areas, coastal, open ocean waters, and in waterbodies in the western-half of the U.S., there would be a subsequent reduction in mercury fish tissue concentrations in these different waterbodies and an associated benefit from avoided decrements in IQ and other known health and ecosystem effects.

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Tables

SECTION 11

BENEFITS OF MERCURY REDUCTION CONSIDERING ESTABLISHED HEALTH-BASED BENCHMARKS AND OVERALL BENEFITS CONCLUSIONS

11.1 Introduction

Chapter 10 presents our estimates of the reductions in methylmercury exposure from recreationally-caught fish for our regulatory options. That chapter also presents our calculations of IQ loss for the case of a nonthreshold, linear dose-response curve (i.e., the analysis assumes a linear dose-response curve and no threshold).[1] Under this combination of assumptions, the risk assessor need not worry about background concentrations – all mercury exposure reductions have the same effect on IQ, regardless of the initial level of exposure from all sources of mercury. This set of assumptions is convenient but not consistent with EPA's mercury Reference Dose (RfD).[2] In fact, as discussed below, the effect of methylmercury on IQ is less likely below the benchmark dose (BMD) and increasingly more uncertain as exposures approach zero.

This chapter first presents some important summary material about EPA's mercury RfD and discusses the implications of the RfD for risk assessment and benefit analysis. Next, the exposure reduction and IQ loss estimates for the no-threshold case presented in Chapter 10 are scaled to quantify the IQ loss due to fetal mercury exposure under a variety of scenarios depicting our understanding of possible thresholds and background concentrations. The scenarios in this chapter represent the Agency's interpretation of benefits under the current RfD. The IRIS assessment (Methylmercury (MeHg), EPA's Integrated Risk Information System, CASRN 22967-92-6) USEPA, www.epa.gov/iris/subst/0073.htm, EPA 2001) reflects the Agency's formal position on the nature of risk of Mercury at low doses.

The chapter concludes with a presentation of EPA's final estimates of IQ-related benefits. To generate IQ-related benefits from reduced mercury exposure, we use three different models – two reflect health-based thresholds for mercury and the other assumes no threshold exist. The benefit estimates are arrayed in a heirarchy from most certain to less certain benefits.

11.2 The Mercury - IQ-loss Paradigm

In general, a risk threshold for any particular substance suggests that there is an exposure level that is without appreciable risk of adverse health effects. In the absence of data or compelling biological rationale indicating the contrary, EPA's default paradigm assumes such a

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[1] IQ was chosen as the endpoint for this analysis because the monetary implications associated with this endpoint are the most straightforward to model and provide a reasonable method for developing a concentration-response relationship. (EPA 2002)

[2] This linear, no threshold case serves a very useful purpose. In this chapter, we explain that as we empirically apply thresholds to the exposed population, we can scale the "no threshold" IQ loss by the change in exposed populations (weighted by change in exposure) to arrive at benefits for threshold scenarios

threshold for all noncancer health effects. This threshold is reflected in the derivation of the Reference Dose (RfD). "In general, 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 2001, p. 2) "This oral reference dose is based on the assumption that thresholds exist for certain toxic effects ..." [3] (EPA 2001, p. 2) The RfD is typically expressed in units of mg/kg bw/day.

The RfD paradigm is appropriate for methylmercury. We lack a biological rationale for how methylmercury causes neurobehavioral loss and other neurological effects[4]. Hence we cannot use our understanding of how methylmercury causes these effects to provide suggestive theories of the lack of a threshold. Further, for the most part, the data from the major studies examining developmental and neurological effects from mercury exposure involve exposure levels that are well above the exposure levels in the U.S. Hence, the underlying data do not provide an ability to empirically test whether a threshold does or does not exist at levels of exposure experienced by US citizens.

As described in Section 2.3 of this document, in updating the RfD, EPA considered a 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 bw/day using a one-compartment model. The BMDLs for these analyses (measured in

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[3] Note that it is conceptually possible to have a de minimus risk at the RfD, rather than an absolute threshold.

[4] There is limited evidence directly linking IQ and methylmercury exposure in the three large epidemiological studies that were evaluated by the NAS and EPA. Based on its evaluation of the three studies, EPA believes that children who are prenatally exposed to low concentrations of methylmercury 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. For this analysis, EPA is adopting IQ as a surrogate for the neurobehavioral endpoints that NAS and EPA relied upon for the RfD.

In the Faroes Island Study, a full scale IQ evaluation was not conducted. However, two core subtests were evaluated (similarities and block design) and one supplementary test was conducted (Digit Span). The similarities and block design tests are reported to be well correlated with the full WISC-R battery (0.885, see Bellinger (2005) paper), but how the Digit Span test relates is not reported. In the EPA analysis, we assume that it relates similarly. In the Faroes study, performance scores on the similarities and block design tests were not shown to be statistically related to cord blood or maternal mercury levels; the digit span test did show a statistical relationship with cord blood mercury.

Both the New Zealand and Seychelles study administered the WISC IQ test (WISC III in Seychelles, WISC R in New Zealand). A reanalysis of the New Zealand data found a positive association, but it was not statistically significant. No significant associations were seen in the Seychelles study. As displayed in Figure 5 of Ryan (2005), the confidence intervals for full scale IQ in both these studies include zero. However, Ryan conducted an integrative analysis, combining results from all three studies. When combined, the statistical power of the analysis increases. While the size of the dose-response relationship declined relative to past studies with a statistically significant finding, Ryan found a statistically significant relationship between IQ and mercury. The confidence interval did not include zero.

terms of mercury in cord blood) were all observed to be within a relatively close range, and the calculated RfDs converge at about 0.0001 mg/kg bw/day. These exposures were converted to dietary exposures of about 0.0005 mg/kg bw/day to 0.0019 mg/kg bw/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. EPA used the various BMDLs and then applied an uncertainty factor of 10 to account for interindividual toxicokinetic variability and pharmacodynamic variability and uncertainty. On this basis, EPA defined the updated RfD of 0.0001 mg/kg bw/day 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. The threshold is likely to be at or above the RfD. We discuss our assumptions of where the threshold falls for benefit analysis later in this section.

Its worth noting that other risk assessors and regulatory agencies from around the world use similar paradigms for benchmarking risks of noncancer health effects. Agencies use reference dose (RfD), acceptable daily intake (ADI), tolerable daily intake (TDI) and minimal risk level (MRL) as descriptors of their exposure benchmarks indicating acceptable exposure levels without appreciable risks. These concepts, in general, assume a threshold for effect, a level with no effect or acceptable effect.

How does EPA's RfD compare to other regulatory agencies benchmark for mercury risk? Health Canada established its Tolerable Daily Intake (TDI) level at twice the EPA's RfD. Their benchmark is .2 ug/kg bw/day. The Agency for Toxic Substances and Disease Registry (ATSDR) has set a Minimal Risk Level (MRI) of .3 ug/kg bw/day – three times EPA's RfD level. The World Health Organization's (WHO) benchmark is set at .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.

11.3 Quantifying IQ Benefits Associated with Mercury Emission Reductions

To assess the IQ benefits from reduced mercury exposure, we must have four pieces of information or data elements:

(1) The reduction in exposure from the regulatory option under investigation

(2) Assuming a linear dose-response curve, we need the slope of that curve.

(3) The level (if any) of the threshold where the dose-response curve intersects the x axis.

(4) If a threshold does exist, we also need to know the baseline exposure levels from all sources.[5]

[5] This baseline exposure level is necessary to know who is above the threshold. Exposure reductions for those individuals already below the threshold would not generate benefits. Hence, we need to know each fish consumer's methymercury exposure from all sources to ascertain whether they are above or below the threshold.

Exposure reductions [data element (1)] are derived in Chapter 10. The dose-response curve [data element (2)] is also discussed in a previous chapter. Our core analysis will use a primary does-response curve that implies an individual will avoid an IQ point decrement for every 1 ppm hair mercury reduction.

Data elements (3) and (4) require more discussion.

11.4 Data Element (3) – The Level of the Threshold

The preceding discussion of EPA's RfD does suggest we should model a threshold for calculating IQ benefits. However, as noted above, the RfD is a level of exposure without appreciable risks. It is plausible that the actual threshold could be higher than the RfD, it is unlikely to be lower. In the absence of data or a compelling biological rationale to suggest a threshold level, we will estimate IQ benefits using a variety of threshold assumptions. We will use some of the benchmark levels of exposure established by regulatory agencies (discussed above) as possible thresholds. Specifically, we will simulate benefits assuming: (1) a threshold equal to EPA's RfD and (2) a threshold in the neighborhood of the WHO and Health Canada benchmarks of .23 and .2 ug/kg bw/day respectively.

11.5 Date Element (4) – The Baseline Levels of Exposure for Consumers of Recreational-Caught Fish

To quantify the IQ benefits from the methyl mercury exposure reductions, we need to know the initial methylmercury exposure levels. If the mercury exposure of a woman eating recreationally-caught fish is below the proposed threshold, then further reductions would not yield IQ improvements. If the fish consumers were above the RfD, then IQ improvements would be expected from reductions in methylmercury exposure. Threshold models require information on the joint distribution of exposure from all sources and much more precise information about fish consumption patterns.

We also must account for the people currently above the RfD (due to all sources of methylmercury exposure) that would be taken below the RfD due to the rule. This would be very difficult to do because we would need the background exposure from all sources to determine who is currently exposed at a level above the RfD. For example, assume that there is an individual who is currently exposed at a level just over the RfD, but only a small part of this is due to the power plants. Looking at the change due to the rule would show a very small change in mercury exposure for this individual, but it could actually push him below the RfD threshold.

Unfortunately, EPA does not have the empirical information necessary to statistically link recreational fishing populations to baseline methylmercury levels. EPA's Notice of Data Availability (EPA 2004) discussed our interest in obtaining additional information on this issue and asked for any available information. We did not receive information that would help us estimate the joint probability distribution of recreational fish consumption and commercial fish consumption sufficient to quantify baseline mercury exposures. Although we have some information on the population at large, we do not have similar information for recreational anglers.

Hence, we are left without sufficient data to address this issue in an ideal manner. The technical approach for dealing with this data gap is addressed in this chapter below.

The remainder of this chapter derives and presents our estimates of IQ benefits.

11.6 Overview of Benefits Methodology

We make full use of the exposure estimates and IQ loss benefits calculated for the linear, no threshold case for each of our regulatory options presented in Chapter 10 to derive benefit estimates for our core threshold scenarios. The analytic steps we take are:

(1) Using the results from Chapter 10 for the Zero Out scenario, place each block group (described in Chapter 10) into categories or bins depending on the size of their exposure reduction (IQ reduction). By establishing bins or discrete categories, the computations, particularly for the uncertainty analysis are simplified greatly. The zero out scenario provides us with an estimate of the total mercury exposure due solely to electric utilities.

(2) Assign an average exposure reduction to and IQ losses for each bin (discrete category).

(3) Perform similar classifications and calculations for regulatory options.

(4) Assign each bin a "background" exposure based on blood mercury distribution from the 1999-2002 National Health and Nutrition Examination Survey (NHANES), assuming of course that the mercury from eating recreationally-caught food is a lower bound on total mercury exposure for any individual. This provides a distribution of total mercury exposure for the recreational fishing population.

(5) Impose thresholds (discussed above)

(6) Assess exposure reductions for each of the regulatory options for all bins above threshold.

(7) Scale benefits from the no-threshold case (presented in Chapter 10) to reflect only those exposure reductions that occur to people above the threshold.

11.7 Freshwater Fish Mercury Exposure

The model in Chapter 10 used 165,000 block groups to produce estimates of mercury exposure and the associated monetized benefits of avoided IQ decrements due to the rule. While, in principle, these block groups could be used to estimate the benefits for individuals above a threshold, a simpler approach can be used relying on a small number discrete interval categories or "bins" of data. In other words, we use the number of individuals experiencing an IQ decrement within a small range, as was illustrated in [Figure 11-11 from the draft Chapter 11 on 2-17-05]. The advantage to using a small number of bins is that an uncertainty analysis can be conducted to produce a range of estimates which would be much more difficult to conduct using the full sample.

The 2020 baseline results from the Population Centroid Model described in Chapter 10 provides an estimate of the average mercury concentration in maternal hair (ppm) from freshwater fish consumption for 481,987 individuals in the baseline. For purposes of producing bins, mercury concentrations are obtained using the mid point of IQ decrements in 0.1 intervals and the inverse of equation [EQ 11-18 from the draft Chapter 11 on 2-17-05]. That is

$$CHgH_j = dIQ_j / 0.131 \qquad \text{(11 Eq. 1)}$$

where

CHgH = average mercury concentration in maternal hair (ppm) for bin j;
dIQ_j = mid point of the IQ decrement interval for bin j.

These data are describe in Table 11-1.

Table 11-1. Freshwater Fish IQ Loss and Hair Mercury from the 2020 Baseline

Number of Individuals	IQ Loss Range	IQ Loss Mid Point	Hair Mercury (ppm)
416,844.83	0 - 0.1	0.05	0.385
45,666.60	0.1 - 0.2	0.15	1.154
11,157.90	0.2 - 0.3	0.25	1.923
4,095.02	0.3 - 0.4	0.35	2.692
1,870.15	0.4 - 0.5	0.45	3.462
899.31	0.5 - 0.6	0.55	4.231
432.62	0.6 - 0.7	0.65	5.000
339.36	0.7 - 0.8	0.75	5.769
206.28	0.8 - 0.9	0.85	6.538
176.41	0.9 - 1	0.95	7.308
74.69	1 - 1.1	1.05	8.077
68.73	1.1 - 1.2	1.15	8.846
15.76	1.2 - 1.3	1.25	9.615
23.49	1.3 - 1.4	1.35	10.385
31.21	1.4 - 1.5	1.45	11.154
22.41	1.5 - 1.6	1.55	11.923
9.77	1.6 - 1.7	1.65	12.692
11.20	1.7 - 1.8	1.75	13.462
17.96	1.8 - 1.9	1.85	14.231
1.56	1.9 - 2	1.95	15.000
0.92	2 - 2.1	2.05	15.769
3.84	2.1 - 2.2	2.15	16.538
5.07	2.2 - 2.3	2.25	17.308
1.35	2.3 - 2.4	2.35	18.077
2.17	2.5 - 2.6	2.55	19.615
1.39	3 - 3.1	3.05	23.462
1.00	4.9 - 5	4.95	38.077
3.97	5.1 - 5.2	5.15	39.615
2.44	5.7 - 5.8	5.75	44.231

The 2020 baseline results can be compared to three scenarios

1. A hypothetical “zero out” scenario, in which mercury emissions from coal-fired utilities are completely eliminated in 2020. This provide an estimate of the full range of potential IQ-related benefits from eliminating EGU-attributable mercury contamination of freshwater fish.
2. Option 1, in which mercury emissions from coal-fired utilities are capped at 38 tons per year in 2010 and capped at 15 tons per year in 2018.
3. Option 2, in which mercury emissions from coal-fired utilities are capped at 38 tons per year in 2010 and capped at 15 tons per year in 2015.

Due to the fact that the analysis is being done using data in bins, these three scenarios are reported as change in mercury levels. These changes are generated by comparing the IQ decrements from the baseline and the particular scenario under consideration. The difference in the IQ decrements are then place in a small number discrete interval categories, or bins, with the number of individuals in each change category. These changes can then be applied to the number of individuals in each exposure category to obtain the estimated monetized benefit of avoided IQ decrements. These data are reported in Table 11-2, 11-3, and 11-4 for the three scenarios.

Table 11-2. Change in IQ Loss and Hair Mercury from the 2020 Zero Out (No Threshold) Compared to the Baseline, and the Relative Probability of each Change Category

Number of Individuals	IQ Loss Range	Hair Mercury Range	Probability
477,526	0 - 0.025	0 - 0.192	99.0744%
3,640	0.025 - 0.05	0.192 - 0.385	0.7552%
537	0.05 - 0.075	0.385 - 0.577	0.1115%
186	0.075 - 0.1	0.577 - 0.769	0.0386%
53	0.1 - 0.125	0.769 - 0.962	0.0111%
16	0.125 - 0.15	0.962 - 1.154	0.0033%
10	0.15 - 0.175	1.154 - 1.346	0.0021%
9	0.175 - 0.2	1.346 - 1.538	0.0019%
2	0.2 - 0.225	1.538 - 1.731	0.0004%
1	0.225 - 0.25	1.731 - 1.923	0.0002%
4	0.4 - 0.425	3.077 - 3.269	0.0008%
2	0.475 - 0.5	3.654 - 3.846	0.0005%

Table 11-3. Change in IQ Loss and Hair Mercury from the CAMR Option 1 (No Threshold) Compared to the Baseline, and the Relative Probability of each Change Category

Number of Individuals	IQ Loss Range	Hair Mercury Range	Probability
481,555	0 - 0.025	0 - 0.192	99.910%
361	0.025 - 0.05	0.192 - 0.385	0.075%
48	0.05 - 0.075	0.385 - 0.577	0.010%
12	0.075 - 0.1	0.577 - 0.769	0.002%
4	0.1 - 0.125	0.769 - 0.962	0.001%
1	0.125 - 0.15	0.962 - 1.154	0.000%
2	0.15 - 0.175	1.154 - 1.346	0.000%
4	0.275 - 0.3	2.115 - 2.308	0.001%

Table 11-4. Change in IQ Loss and Hair Mercury from the CAMR Option 2 (No Threshold) Compared to the Baseline, and the Relative Probability of each Change Category

Number of Individuals	IQ Loss Range	Hair Mercury Range	Probability
481,324	0 - 0.025	0 - 0.192	99.862%
558	0.025 - 0.05	0.192 - 0.385	0.116%
72	0.05 - 0.075	0.385 - 0.577	0.015%
22	0.075 - 0.1	0.577 - 0.769	0.004%
2	0.1 - 0.125	0.769 - 0.962	0.000%
4	0.125 - 0.15	0.962 - 1.154	0.001%
2	0.15 - 0.175	1.154 - 1.346	0.000%
4	0.3 - 0.325	2.308 - 2.5	0.001%

11.8 Deriving Baseline Mercury Exposures from All Sources of Mercury

For this analysis, we assumed that the NHANES data provided coverage for recreational freshwater anglers (e.g., sampling of individuals from the general US population for inclusion within NHANES, include some fraction of recreational freshwater anglers). However, we are unable to identify the specific location of recreational anglers within the full NHANES distribution for purposes of establishing total mercury exposure for those modeled individuals. Therefore, we had to make certain assumptions in "relating" the recreational anglers modeled for the RIA to the NHANES distribution for purposes of predicting total mercury exposure for this population. Specifically, we assumed that the least exposed recreational angler would at least have total mercury exposure equal to the mercury exposure level they receive from the self-caught freshwater pathway (i.e., assuming they have zero background or commercial fish exposure). This is a reasonable lower-bound estimate of total mercury exposure for the recreational anglers modeled for the RIA and therefore represents a reasonable basis for establishing the lower bound of total recreational angler exposure within the NHANES distribution. The next step is to determine how recreational anglers are distributed across percentiles of the NHANES distribution above that lower bound percentile

The NHANES data used was provided in percentiles, with an average blood mercury concentration for each 1 percent of the exposed population. The blood mercury level (in ppb) was converted to a hair mercury level (in ppm) using the hair:blood ratio of 200, as described in Chapter 9. This implies that hair mercury levels (in ppm) are one-fifth of blood mercury levels (in ppb). The lowest hair mercury level in Table 11-1 is 0.385 ppm. This represents the mercury exposure from freshwater fish alone. Since the total exposure must be above this level, these individuals must be associated with a NHANES percentile with a mercury concentration higher than 0.385 ppm. The lowest NHANES percentile above this level is the 77th percentile, with a value of 0.4 ppm. Therefore, the 481,987 individuals in the 2020 baseline are assumed to be evenly distributed over the 77th through the 100th percentile (24 bins) in the NHANES data.[6] The exact number of individuals in table 11-1 is 481,987.41. Divided by 24 implies 20,082.81 in each NHANES percentile.[7]

The mercury exposure from freshwater fish and total mercury exposure is estimated by combining the data from Table 11-1 and the NHANES data. These are listed in Table 11-5. The sum of individual for any freshwater fish exposure level must equal the number of individuals in Table 11-1 for that exposure. The sum of individuals for any total exposure level must equal the number of individuals in the NHANES percentile. In the upper tail of the freshwater fish consumption, the freshwater fish exposure exceeds the top hair mercury level from the NHANES data. For these individuals, the freshwater fish mercury exposure is also taken to be their total exposure value.

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[6] These results suggest that the lowest modeled recreational fisher has total mercury exposure matching the 77th% US resident. In reality, if the threshold analysis had been conducted using the fully disaggregation set of 481,987 modeled individuals described in Chapter 10, the least exposed recreational angler would have been matched to a much lower NHANES percentile (probably less than the median). Note, however, that the mean for the recreational anglers would likely be larger than the NHANES mean and probably close to the 77th%, which is plausible given that recreational anglers consume self-caught fish in addition to their commercial fish exposure and therefore may reasonably be expected to have total fish consumption above the general population. The use of aggregated (clustered) data in the threshold analysis, while making the analysis tractable, did remove much of the inter-individual variability in modeled exposure from self-caught fish, which adds additional uncertainty into the threshold analysis both in terms of (a) predicting the number of individuals exceeding the threshold of concern due to total exposure and (b) estimating the fraction of total benefits exceeding those same thresholds. It is also important to note that the assumption that recreational anglers are evenly distributed across the 77th-100th percentiles of the NHANES distribution is also subject to uncertainty. Compelling arguments can be made that this assumption is either over- and under-conservative. Ultimately, without identifying data specifically defining the relationship between self-caught fish and commercial fish consumption for the recreational angler population, any assumptions regarding this relationship is subject to considerable uncertainty.

[7] We note that the conversion from blood mercury levels (in ppb) to hair mercury levels (in ppm) using a 5:1 ratio might produce higher values for hair mercury in the upper tail of the distribution than is found in the actual NHANES data on hair mercury levels. Since the NHANES data use for this analysis included only blood mercury levels, however, this type of conversion was necessary and is a common technique. If we were to try to account for this potential over-estimate, we might assume that individuals are at some lower percentile (say, the 50th) of the NHANES distribution and then impose the shape of the distribution beyond this point for total exposure. This would likely force more individuals into the lower exposure range and reduce the subsequent scaling factor. This type of sensitivity analysis, however, was not done.

Table 11-5. Joint Distribution of Mercury Exposure from Freshwater Fish and Total Mercury Exposure

Number of Individuals	Freshwater Fish Exposure	Total Exposure
20082.81	0.385	0.400
20082.81	0.385	0.420
20082.81	0.385	0.420
20082.81	0.385	0.440
20082.81	0.385	0.460
20082.81	0.385	0.480
20082.81	0.385	0.520
20082.81	0.385	0.540
20082.81	0.385	0.560
20082.81	0.385	0.600
20082.81	0.385	0.640
20082.81	0.385	0.680
20082.81	0.385	0.740
20082.81	0.385	0.780
20082.81	0.385	0.860
20082.81	0.385	0.920
20082.81	0.385	0.980
20082.81	0.385	1.100
20082.81	0.385	1.220
20082.81	0.385	1.360
15188.66	0.385	1.540
4894.15	1.154	1.540
20082.81	1.154	1.980
20082.81	1.154	2.460
606.83	1.154	7.780
11157.90	1.923	7.780
4095.02	2.692	7.780
1870.15	3.462	7.780
899.31	4.231	7.780
432.62	5.000	7.780
339.36	5.769	7.780
206.28	6.538	7.780
176.41	7.308	7.780
74.69	8.077	8.077
68.73	8.846	8.846
15.76	9.615	9.615
23.49	10.385	10.385
31.21	11.154	11.154
22.41	11.923	11.923
9.77	12.692	12.692
11.20	13.462	13.462
17.96	14.231	14.231
1.56	15.000	15.000
0.92	15.769	15.769
3.84	16.538	16.538
5.07	17.308	17.308

Number of Individuals	Freshwater Fish Exposure	Total Exposure
1.35	18.077	18.077
2.17	19.615	19.615
1.39	23.462	23.462
1.00	38.077	38.077
3.97	39.615	39.615
2.44	44.231	44.231

Thresholds

The introductory sections of this chapter present background on our RfD, other agencies' risk benchmarks and the nature of thresholds in risk assessment. As discussed in these sections, we use the two benchmark levels of exposure as possible thresholds.

a. 0.1 µg/kg bw/day - A threshold equal to EPA's RfD
b. 0.2 µg/kg-day - A threshold in the neighborhood of the WHO and Health Canada benchmarks of .23 and .2 µg/kg bw/day respectively.

These were converted to ppm of hair mercury based on Eq. 2 (11 - Eq. 11.17 from the draft Chapter 11 on 2-17-05] in Chapter 10. Specifically,

$$CHgH = (0.08)^{-1} * HgIkg \qquad \text{(11 Eq. 2)}$$

where
CHgH = average mercury concentration in maternal hair (ppm)
HgIkg = average daily mercury ingestion rate (mg/kg bw/day);

This conversion rate between average daily ingestion rate and maternal hair concentration is based on the one compartment model used by Swartout and Rice (2000) as described in Chapter 10. This implies the two thresholds of 1.25 ppm and 2.5 ppm.

As discussed above, there are other potential thresholds that could be considered. For example, from the Agency for Toxic Substances and Disease. These are not considered here because of the nature of the discrete nature of the NHANES data in percentile form. The 99th percentile of hair mercury in the NHANES data is 2.46 ppm. The 100th percentile data is 7.78 ppm. Because of this analysis is being conducted using data bins, the choice of any threshold between 2.46 and 7.78 ppm would produce the same results. Therefore, consideration of the benchmark values from these other agencies would produce the same results as our second threshold above.

11.9 Deriving Scaling Factors

Using the data from Table 11-5; the changes in Tables 11-2, 11-3, and 11-4 and the thresholds described above, it is possible to evaluate the effect of our two thresholds. Because of a number of uncertainties, including the level of total exposure, this evaluation is best conducted using a Monte Carlo simulation.

In a Monte Carlo simulation a model is run many times using different values for uncertain input parameters. Prior to the simulation, each uncertain parameter is assigned a probability distribution. For each iteration of the simulation, a random input value is chosen for each uncertain parameter, using the assigned distribution and a realization of the output value is calculated. The output value is recorded and another iteration with a different random choice of input values is conducted. After a very large number of iterations, a probability distribution can be developed for the output value.

The analysis in Chapter 10 calculates the change in mercury exposure due to this rule. What is unknown is the total exposure level of the individuals experiencing this change. To address this uncertainty, we conduct a Monte Carlo simulation. In this case, we treat the change in exposure as the uncertain input variable. We assume that the distribution in the change in exposure is equal to the change in exposure from the Population Centroid Model results from Chapter 10. For example, for the results from CAMR, a separate change in exposure is randomly chosen for each data bin using the probabilities in 11-3. This change is then applied to data bin listed in Table 11-5. For the scenarios in which a threshold is applied, change in exposure is assumed to not occur for any data bin in which the total exposure is below the threshold. The resulting distribution of the decision variable can then be used to determine the scaling factors for each scenario.

This approach assumes zero correlation between the total mercury exposure and the change in mercury exposure as a consequence of the regulatory option. However, the individual level results from Chapter 10 suggest a high degree of correlation between the mercury exposure self-caught fish and the changes in mercury exposure. This means that the zero correlation assumption above could understate the scaling factor because the individuals experiencing the highest change are likely above the thresholds. To account for this potential underestimate, a second scaling factor is developed assuming perfect correlation between total mercury exposure and the changes in mercury exposure. Since the changes listed in Tables 11-2, 11-3, and 11-4 are given in ranges, a uniform distribution was assumed, so that each data bin in Table 11-5 is associated with a progressively larger change in mercury exposure from the regulatory option.

The scaling factors are listed in Table 11-6. The No Threshold scenario is the reference value, and has a scaling value of 100%. The lower bound is based on the mean value from the Monte Carlo simulation. The upper bound is based on the perfect correlation scenario. Due to the fact that over 99% of the changes for all three scenarios occur in the lowest change category (a change of 0.0 - 0.192 ppm in hair mercury), there is no difference in the scaling factors between regulatory scenarios. These values can be used to scale the detailed benefits estimates from Chapter 10.

Table 11-6. Scaling Factors

	Zero Out	Option 1	Option 2
EPA RfD (Threshold = 0.1 µg/kg bw/day)	21 - 34%	21 - 34%	21 - 34%

WHO / Health Canada (Threshold = 0.2 - 0.23 μg/kg bw/day)	4 - 8%	4 - 8%	4 - 8%
No Threshold	100%	100%	100%

11.10 Monetization and Scaling of IQ Benefits

The IQ decrements associated with a change in mercury exposure are derived from the Dose-Response curve described in Chapter 9. The slope of that dose response curve is 0.131, so each 1 ppm in hair mercury is assumed to cause a decrement of 0.131 of an IQ point. The monetary value of losses resulting from IQ decrements are assessed in terms of foregone future earnings for the affected individuals and described in Chapter 10. These monetary value of these losses were estimated to be $8,807 (in 1999$) per IQ point lost. Therefore, the monetized benefit from a decrease in hair mercury is calculated as follows:

$$VN_i = N_i * VIQ * (0.131 * dCHgH_i) \qquad (11 - Eq.-3)$$

where

VN = Value of avoiding IQ loss for N_i individuals in bin i;
N_i = Number of individuals in bin i;
VIQ = Value per change in IQ point decrement (= $8,807 in 1999$);
dCHgH = Change in mercury concentration in maternal hair (ppm) for bin i.

The dose-response slope coefficient of 0.131 was modeled assuming a no threshold case, in which the dose-response line passes through the origin, and a zero IQ response only occurs with a zero dose. If there were a threshold, then ideally, we would reestimate the dose-response curve with the threshold assumption incorporated into the estimation procedure. If this was done, the new linear dose response curve would have a steeper slope than the one estimated under a no threshold case. This would affect the monetized benefits value from equation (11 Eq.-3). Unfortunately, we do not have the original data to reestimate the dose-response with a threshold assumption. However, since the coefficient enters this equation linearly, the benefits would simply be increased by the same proportion as the increase in the dose-response coefficient.

The benefits from Chapter 10 are used as inputs to our model to estimate benefits under our threshold cases. The monetary benefits and reduction in IQ point decrements can be obtained for the two thresholds cases by multiplying the scaling factor from Table 11-6 times these benefits. Tables 11-7 and 11-8 present our results for the two threshold models and for the no threshold case. The benefits are presented in order of increasing uncertainty. We are most certain of the benefits at high doses, given that the original data from the developmental studies fall mostly at high doses. Accordingly, the benefits for the highest threshold are most certain, followed by the lower threshold equal to EPA's RfD. The benefits estimated at exposure levels below the RfD are the most uncertain. The derivation of the unit value for each IQ point is discussed in Chapter 10. The monetized benefit of CAMR Option 1 over the baseline range from $.07 million to $2 million a year. For Option 2, the benefits range from $.1 million to $4.6 million per year, depending on the dose-response model and the choice of discount rates. The tables also present “discounted” IQ points (IQ points gained discounted to present values at 3 and 7 percent.

Table 11-7. IQ Benefits for CAMR Option 1 under Established Health-Based Benchmarks

Uncertainty Regarding Threshold	Benchmark Source	Level of Threshold	Discount Rate	Scaling	Benefits (millions 1999$)	Discounted IQ Points
More Certain	WHO / Health Canada	0.2 - 0.23 μg/kg bw/day	3%	4%	$0.07 - $0.12	8 - 14
				8%	$0.14 - $0.24	15 - 27
			7%	4%	$0.03 - $0.08	4 - 9
				8%	$0.06 - $0.16	7 - 18
	EPA RfD	0.1 μg/kg bw/day	3%	21%	$0.36 - $0.63	41 - 72
				34%	$0.58 - $1.0	66 - 116
			7%	21%	$0.17 - $0.42	19 - 48
				34%	$0.27 - $0.68	31 - 77
Less Certain	No Threshold	N/A	3%	100%	$1.7 - $3.0	193 - 341
			7%	100%	$0.8 - $2.0	91 - 277

Table 11-8. IQ Benefits for CAMR Option 2 under Established Health-Based Benchmarks

Uncertainty Regarding Threshold	Benchmark Source	Level of Threshold	Discount Rate	Scaling	Benefits (millions 1999$)	Discounted IQ Points
More Certain	WHO / Health Canada	0.2 - 0.23 μg/kg bw/day	3%	4%	$0.1 - $0.18	11 - 21
				8%	$0.2 - $0.37	23 - 42
			7%	4%	$0.05 - $0.12	5 - 14
				8%	$0.1 - $0.25	11 - 28
	EPA RfD	0.1 μg/kg bw/day	3%	21%	$0.53 - $0.97	60 - 110
				34%	$0.85 - $1.56	97 - 178
			7%	21%	$0.25 - $0.65	29 - 74
				34%	$0.41 - $1.05	46 - 120
Less Certain	No Threshold	N/A	3%	100%	$2.5 - $4.6	284 - 522
			7%	100%	$1.2 - $3.1	136 - 352

11.11 Uncertainties

This benchmark analysis is subject to uncertainty reflecting a number of factors. The first is the choice of the threshold value. We simply do not have enough data to test whether a threshold does or does not exist. The use of the RfD represents a best estimate of the population threshold. Another area of uncertainty is that the lack of data characterizing the correlation between self-caught fish consumption and commercial fish consumption in determining total fish consumption for recreational freshwater anglers, prevents a truly representative analysis of total mercury exposure for this population. The use of aggregated (clustered) exposure estimates for recreational freshwater anglers rather than the full set of block group-level results generated for the RIA introduces uncertainty into (a) the estimate of the number of individuals exceeding a given benchmark due to total mercury exposure and (b) the fraction of total Option 1 and Option 2 benefits exceeding that threshold. Finally, failure to consider an upward adjustment to the IQ loss function under assumptions of various thresholds considered in this analysis resulted in an under-prediction of benefits above the threshold.

11.12 Conclusions

This chapter presents modeled estimates of IQ benefits for three scenarios. (1) A threshold equal to EPA's Rfd of 0.1 ug/kg bw/day; (2) a threshold at a level near the Health Canada and World Health Organization's benchmark levels of 0.2 ug/kg bw/day and 0.23 ug/kg bw/day respectively; and (3) a no threshold benchmark. The chapter presents IQ benefit estimates for the CAMR Option 1 and CAMR Option 2 that are incremental to the expected reduction in mercury emissions under CAIR.

We are more confident of the likelihood of effects at higher exposures (i.e., above the WHO/ Health Canada value) because these exposure levels are closer to the observed level of exposures in the three underlying studies. However, if the threshold were set above the RfD level, at a level equivalent to the WHO level, we would be ignoring the potential health effects that might be gained from exposures above the EPA RfD. On the other hand, we are less certain of the benefits below the RfD in the no threshold analysis since the RfD identifies a level below which there is no appreciable risk of deleterious effects. Based on the NAS review of the science and their endorsement of the RfD value as being scientifically justifiable for the protection of public health, EPA has adopted the RfD as an appropriate construct for expressing the level of daily exposure which is likely to be without an appreciable risk of deleterious effects during a lifetime.

Under the EPA's RfD threshold scenario, we found that under CAMR Option 1, total IQ points gained were between 19 and 116 IQ points for children of recreationally-caught fish consumers. These IQ points were valued at a total benefit of $0.25 - $1.56 million.

We also simulated a threshold at a level near the Health Canada and World Health Organization benchmark levels of .2 ug/kg bw/day and .23 ug/kg bw/day respectively. The table indicates that benefits are indeed sensitive to the modeled thresholds. Going from the EPA threshold to the higher threshold suggested by the WHO and Health Canada reduced benefits by 19 to 24 percent.

Finally, not surprisingly, the no threshold case – the more uncertain of the estimates – has the largest benefits – roughly four times the benefits as the EPA RfD threshold case.

11.13 References

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Tables

SECTION 12

CO-BENEFITS RESULTING FROM REDUCTIONS IN EMISSIONS OF PM2.5

12.1 Introduction

Emissions control strategies adopted by power plants to meet cap-and-trade regulations implemented under CAMR are likely to result in co-benefits including reductions in direct emissions of PM2.5. These PM emissions reductions could result in decreased population-level exposure to PM2.5, which, in turn will produce reductions in adverse health effects including both morbidity and premature mortality for the U.S. population. Because of limitations associated with the Integrated Planning Model (IPM), EPA was not able to conduct a comprehensive assessment of health benefits associated with reductions in directly-emitted PM2.5 from coal-fired power plants (the IPM model, as currently configured, cannot project changes in directly emitted PM2.5 for the technology configurations relevant to this regulation). Instead, EPA conducted a illustrative analysis focused on direct PM2.5 and on the key health endpoint; premature adult mortality. Despite the use of a simplified modeling approach, the illustrative methodology used in evaluating the adult mortality endpoint for this rule is still considered robust enough to provide perspective on the likely magnitude of PM-related co-benefits of CAMR if Activated Carbon Injection with the addition of a polishing baghouse (TOXECON^{tm}) is used. In conducting the illustrative analysis, EPA has used standard methods, assumptions, and monetary values that are used in other EPA regulatory analyses and have undergone significant peer review. See the CAIR for a detailed discussion of the derivation of assumptions used in PM benefit analyses conducted by EPA (EPA, 2005).

It is important to note that this analysis does not consider a number of health endpoints and welfare effects that would add to the overall co-benefits if modeled, including: (a) PM-related morbidity (e.g., chronic bronchitis, hospital admissions for respiratory and cardiovascular events, non-fatal myocardial infarctions), (b) infant mortality, and (c) welfare effects including visibility improvements. In other analyses of PM2.5 benefits, the mortality endpoint has typically accounted for 85-95% of the total quantified benefits.

In addition, this analysis does not consider potential co-benefits resulting from reductions inprecursors to PM2.5, including SO2. In the case of SO2, given the SO2 caps under CAIR, we do not expect SO2 reductions beyond the CAIR levels under CAMR. There is also the potential that compliance with mercury regulations will simply shift the time-course of SO2 emissions changes, but not result in absolute changes in the magnitude of those emissions. Consequently, co-benefits resulting from reductions in the secondary formation of PM2.5 were not modeled.

<u>Options Evaluated for Co-Benefits</u>

As is discussed in Section 7 of this report, two scenarios were considered in the emissions modeling and thus modeled for the PM2.5 co-benefits analysis including (a) Option 1 (38 ton cap in 2010 and a 15 ton cap in 2018), and (b) Option 2 (38 ton cap in 2010 and a 15 ton cap in 2015). We also examined the effects of the possibility that *advanced sorbents* (i.e., advanced carbon injection (ACI)) technology are available earlier than anticipated (assumed available in 2013) with resulting reductions in PM2.5 co-benefits (see Section 7.5 for details on projected development/application of ACI technology and EPA, 2005). Without advanced sorbents, Option 2 yields the greatest PM2.5 co-benefits (5671 tons in 2020) due to less opportunity for banking of mercury allowances by power plants which reduces the opportunity to delay installation of mercury control equipment. Option 1 produces lower PM2.5 co-benefits (1920 tons in 2020) due to greater potential for banking which increases the potential for plants to delay installation of control equipment. With advanced sorbents, there are noticeably less PM2.5 emissions reductions (67 tons in 2020) due to the assumption that advanced sorbent technology will not require installation of additional bag houses (the source of the majority of the PM2.5 co-benefits predicted for these scenarios).[1]

Overview of Methodology

The PM co-benefits illustrative analysis was conducting using the following step-wise procedure:

The Integrated Planning Model (IPM) was used to estimate the fraction of coal-fired power plants expected to install bag houses (and the time course for those installations) for each of the regulatory scenarios under consideration. This information was then combined with (a) information on the relevant characteristics of key coal types used by these facilities (i.e., heating values and ash content) and (b) emissions factors for bag houses and electrostatic percipitators installed at power plants, to generate estimates of direct PM2.5 emissions reductions for each scenario.

The Source-Receptor Matrix (SR-MATRIX) model (E. H. Pechan and Associates, 1997), was used to predict population-level changes in PM2.5 exposure given emissions reduction estimates for direct PM2.5 from coal-fired power plants. The SR-MATRIX model uses a simplified methodology in conducting both air quality modeling and exposure modeling. In a full-scale benefits analysis, these two steps would have been conducted using more rigorous and sophisticated models such as CMAQ (for air quality modeling) and BenMAP (for exposure modeling).

Changes in population-level exposure to PM2.5 were then combined with baseline incidence data for adult mortality and used together with the concentration-response function for mortality derived from Pope et al., 2002, to estimate reductions in premature adult mortality. The Pope et al., 2002 -based concentration response function has been used in other benefits analysis including CAIR.

Adult mortality incidence reductions were then monetized using the value of statistical life (VSL) metric of 5.5 million dollars per statistical life saved that is used in other EPA regulatory analyses. We also incorporate other elements of the benefit methodology from CAIR (EPA, 2005), including consideration for: (a) a mortality reduction lag of 20 years (i.e., the

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[1] Because of time and resource considerations, and because Option 1 was the preferred option for CAMR, we only ran the advanced sorbents sensitivity analysis for Option 1, but we believe the results for Option 2 would be similar to what is reported here for Option 1 (e.g., approximately 67 tons in 2020).

decrease in mortality incidence is distributed over 20 years after the emissions change), (b) income elasticity associated with the valuation function (i.e., degree to which willingness-to-pay (WTP) for a reduction in mortality changes as income grows), (c) real growth in per-capita income and (d) growth in the adult population cohort experiencing the mortality reduction.

Organization of this Section

The remainder of this section is organized as follows. Section 12.2 provides an overview of emissions modeling conducted using IPM and other data sources in order to generate estimates of emissions reductions in direct PM2.5 from power plants. Section 12.3 provides a brief overview of the SR-MATRIX model and its role in air quality modeling and predicting population-level changes in exposure for the study population. Section 12.4 describes the use of the Pope et al., 2002 study in modeling long-term exposure-based mortality reductions linked to reduced PM2.5 exposure. This section also describes the lag period used in modeling mortality reductions. Section 12.5 describes the valuation of benefits including the VSL metric used, the income elasticity and related factors. Section 12.6 presents the results of the illustrative analysis of PM-related co-benefits. Finally, Section 12.7 describes some of the uncertainties related to this analysis.

12.2 Emissions Modeling

Potentially, several different technologies could be selected by coal-fired power plants to reduce mercury emissions including: (a) optimized existing emission controls (no direct PM2.5 co-benefit), (b) injection of halogenated or standard (non-halogenated) powdered activated carbon (PAC) in front of an electrostatic precipitator (no direct PM co-benefit)[2], (c) injection of standard PAC in front of a retrofit polishing baghouse - TOXECON™ application (direct PM2.5 co-benefit), and (d) other technology (may or may not have an impact on direct PM2.5 emissions). In option "c" described above, a polishing baghouse is retrofitted in after an existing electrostatic precipitator. Since the baghouse is added as an additional PM control and is generally a more efficient collector of PM2.5 than an ESP, a direct PM2.5 co-benefit is likely in this option. Therefore, the first step in estimating possible co-benefits associated with reductions in emissions of direct PM2.5, was to estimate the power plant capacity likely to select option "c", which involves the addition of polishing baghouses with associated reductions in direct PM2.5 emissions. Because different coal types (e.g., bituminous, subbituminous) are used by power plants and these coal have different characteristics (e.g., *ash content*) that can impact direct PM2.5 emissions, it is important to consider the distribution of these coal types across the subset of power plants projected to install baghouses to control mercury. The IPM model, which is a linear programming model used to predict the behavior of the U.S. electric utility sector in response to air-pollution-control-related regulatory scenarios, was used to determine the coal-specific capacities of coal-fired power plants likely to install baghouses in response to the two regulatory scenarios under consideration

[2] Concerns have been raised (EPA, 2005) that this option (injection of PAC in front of an electrostatic precipitator (ESP) could actually result in dis-benefits by producing increased arcing in the ESP which can degrade ESP PM capture performance. However, at this point it is unclear whether the rates of PAC injection likely to be utilized under this scenario would produced sustained increases in arcing such that PM removal is significantly reduced. Research is currently on-going looking at this issue in the context of longer-term facility performance (EPA, 2005).

Once the capacities of power plants projected to install baghouses under the two regulatory scenarios were known, the next step was to estimate reductions in direct PM2.5 emissions associated with these capacities. Calculation of reductions in direct PM2.5 emissions involves combining data on (a) the utilization of the four coal types at power plants projected to install baghouses (*annual heat input values* for each coal type obtained from IPM), (b) the relevant attributes of those coal types, including *heating values* and *ash content* (from the Information Collection Request (ICR) database), and (c) the PM2.5 emissions *factors* for electrostatic precipitators and baghouses (from AP-42)[3]. For each regulatory scenario, the *annual heat input* and the *average heating values* for a specific coal type were combined to determine the amount of that coal type used by power plants of interest. This usage value was then combined with the ash content value and emissions factors to generate the emissions reduction estimate for that coal type. Such estimates are obtained for each coal type. These estimates were then summed across the coal types to generate a single direct PM2.5 emissions reduction value for coal-fired power plants under the specific regulatory scenario.

EPA used the currently available information on mercury controls to develop the estimates of PM2.5 co-benefits described above (Note: EPA projects that, under Option 1, 13 units will have installed ACI by 2020 - see Section 7.5 for projections regarding installation of ACI devices). It is recognized, however, that mercury control technologies are under vigorous development and, therefore, control approaches other than those described above may be implemented in the future. Such actions may result in different PM2.5 co-benefits compared to estimates provided here.

12.3 Air Quality Modeling and Population-Level Exposure Estimation

The SR-MATRIX model (E. H. Pechan and Associates, 1997), uses county-to-county transfer factors to predict changes in PM2.5 air concentrations resulting from reductions in emissions of directly-emitted PM2.5 from (user-specified) source categories (Levi et al., 2003). These transfer factors are generated using an adjusted version of the Climatological Regional Dispersion Model (CRDM), which uses a sector-averaged dispersion model combined with summaries of 1990 meteorological data to produce these transfer factors. SR-MATRIX can model secondary formation of PM2.5 including nitrates and sulfates, however this functionality was not used in this illustrative analysis. All sources in the SR-MATRIX model are divided into four categories including three categories based on effective stack height and a forth category for all area sources. The SR-MATRIX model has been shown to generate reasonable predictions of population-level changes in exposure resulting from changes in directly emitted PM and

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[3] The PM2.5 emissions factors for electrostatic precipitators and baghouses are taken from Table 1.1.6 in AP-42, Fifth Edition, Volume 1 Chapter 1: External Combustion Sources, available at http://www.epa.gov/ttn/chief/ap42/ch01/final/c01s01.pdf. As is well known, emission factors are simply averages of all available data of acceptable quality, and are generally assumed to be representative of long-term averages for all facilities in the source category (i. e., a population average). In the absence of availability of specific data on direct PM2.5 co-benefit associated with TOXECON applications, EPA chose to use PM2.5 emission factors. It is noted however, that baghouse emission factor in AP-42 does not take into account any mercury-specific design changes that may take place. For example, if the baghouse to capture mercury-impregnated sorbent, with particles larger than PM2.5, was designed to use a fabric with a coarser weave (to keep pressure drop low), capture of PM2.5 in such a baghouse may be lower than that indicated in the emission factor. This, in turn, will result in reduced direct PM2.5 co-benefit.

consequently, this model was applied in modeling these direct PM-related co-benefits (Levy et al., 2003).

As applied in this analysis, the SR-MATRIX model was used for converting the estimates of reductions in direct PM2.5 emissions into changes in ambient PM2.5 concentrations. In modeling population-level exposure reductions, the SR-MATRIX generates results in the form of population-weighted PM2.5-related exposure reductions. The model provides estimates by region throughout the U.S. As with the benefit analysis of mercury reductions presented in Section 10, the eastern-half of the U.S. is analyzed, which is identified by the Midwest/Northeast (ME) and Southeast (SE) regions in the SR Matrix. We then averaged the results in these regions to derive total PM benefits. A more comprehensive analysis of power-plant related emissions reductions (and associated exposure reductions) conducted using BenMAP, would have used a more spatially-refined geographic grid that tracked both ambient PM2.5 reductions and associated exposure reductions with greater precision and specificity. Note, however, that it is not known whether the simplified modeling approach used here results in a net over- or under-prediction of population-level exposure since sources of uncertainty associated with this illustrative approach can result in both over- and under-predictions of ambient PM2.5 levels and population-level exposures. However, the more generalized approach used here is considered reasonable for a illustrative analysis of potential benefits. The SR-MATRIX model can be used for health effects incidence estimation and valuation, but the version used in this analysis did not have the latest mortality functions (Pope et al., 2002) or the latest valuation functions and consequently, both mortality incidence changes and valuation of those incidence reductions were estimated outside of the model (as discussed in detail below).

12.4 Modeling Changes in Health Endpoint (Mortality) Incidence

Exposure to PM2.5 has been linked to a variety of morbidity endpoints (e.g., respiratory and cardiovascular hospital admissions, chronic bronchitis events, asthma exacerbations) as well as mortality (both child and adult) (NRC, 2002). However, the majority of monetized benefits linked to health endpoints are associated with chronic (long-term exposure-related) mortality in adults. Consequently, this illustrative analysis focuses on this endpoint exclusively.

Both long- and short-term exposure to PM2.5 has been associated with mortality in adults (NRC, 2002). However, long-term exposure cohort studies, which are better able to capture the full public health impact of PM2.5 exposure over time and likely cover some of the shorter-term exposure mortality signal together with the longer-term exposure signal, have typically found higher levels of mortality risk than shorter-duration studies (Kunzli et al., 2001, NRC, 2002). Consequently, this illustrative analysis evaluated chronic exposure-related mortality in adults.

Long-term mortality studies examine the relationship between community-level exposure to PM2.5 over multiple years and annual mortality rates, also recorded at the community-level (NRC, 2002). More recently-conducted prospective cohort studies allow for better control for key confounders associated with mortality including individual-level information on key risk-related factors such as diet, occupational exposure and smoking. The EPA's Science Advisory Board recommends the use of long-term prospective cohort studies in estimating PM-related mortality (EPA-SAB-Council-ADV-99-005, 1999), a finding that was confirmed by the recent National Research Council Report (NRC, 2002).

The prospective cohort study selected for this illustrative analysis (Pope et al, 2002), has the broadest geographical coverage of the available prospective cohort studies examining PM-related mortality and uses the American Cancer Society's (ACS) dataset which has been subjected to extensive reexamination and reanalysis, and which has served to strengthen the confidence associated with its findings. The latest reanalysis of the ACS study data conducted by Pope (2002) provides additional refinements including (a) an extended follow-up period that triples the size of the mortality data set, (b) significant increase in the exposure dataset, including consideration for cohort exposure following implementation of the PM2.5 standard, (c) greater control for possible confounders (diet and workplace exposure) and (d) use of advanced statistical techniques to address concerns over spatial autocorrelation of survival times in communities located near each other. Both the NRC and SAB-HES recommended the use the Pope et al., 2002 study as the basis for modeling adult mortality linked to PM2.5 exposure as part of economic benefits analysis. The "all-cause" mortality category was selected as the endpoint modeled using this study.

Addressing Possible Mortality Reduction Lag

It is expected that reduction in ambient PM2.5 levels will produce a decrease in mortality that does not occur immediately, but is distributed over some number of years (i.e., lagged over time following the PM reduction) (NRC, 2002). However, the exact nature of the lag associated with PM-related mortality reductions is not currently known. Consideration of mortality-related lag periods is important since delays in reductions in mortality rates following PM reductions will result in discounting (lowering) of overall monetary benefits associated with those mortality reductions. Given limited information available for establishing a lag structure for PM-related mortality reductions, the SAB-HES has recommended the following provisional 20-year lag structure: 30% in the first year, 50% distributed over years 2-5 and 20% distributed over years 6-20. This structure reflects the following perspective towards PM-related mortality reduction: short-term exposure related reductions are assumed to occur in the first year, cardiovascular-related mortality reductions are assumed to occur in years 2-5 and longer-term respiratory as well as lung cancer mortality reductions are assumed to occur in years 5-20.

Consideration of Population Growth

The regulatory analysis (simulation) year for this illustrative analysis is 2020 for both Option 1 and Option 2 and for the Sensitivity Analysis scenario. Demographic change (growth in the adult-age population) needs to be considered in estimating mortality reductions for this future simulation year. Typically, in a comprehensive benefits analysis conducted using BenMAP, demographic growth would be projected separately for each geographic unit of analysis (e.g., county, or US Census block) for which mortality reductions are being projected. However, this illustrative analysis, does not include this level of spatial resolution and consequently, a simple demographic scaling ratio was being used to adjust mortality estimates to reflect potential growth in the adult-age cohort over the years leading up to the simulation year. This demographic scaling factor was simply the ratio of US adult population (>29yrs) predicted for the year 2020 divided by the adult population in 1999 (the year modeled in SR-MATRIX). Note, that because both Options being considered are modeled for the same simulation year, the same demographic scaling ratio was used in modeling each regulatory option.

Addressing Potential Uncertainty in the Mortality Function

The EPA is currently investigating uncertainty associated with the concentration-response function for chronic-duration PM2.5 mortality. A number of methods are being considered, including the use of expert elicitation to develop quantitative assessments of overall uncertainty associated with PM2.5-related mortality estimates. Because this analysis was conducted as an illustrative analysis, consideration of uncertainty in the mortality estimate was not quantitatively modeled. Section 12.6 provides a summary of the potential magnitude of uncertainty surrounding the mortality estimate based on benefit analysis of CAIR. The reader is referred to the CAIR RIA (EPA, 2005) for an in-depth discussion of uncertainty associated with PM2.5-related mortality estimates and ongoing efforts to characterize that uncertainty.

12.5 Valuation of Benefits

Valuation of benefits involved monetizing of the reduction in adult premature mortality associated with PM2.5 using a value of statistical life (VSL) of 5.5$ million. This value is based on several published meta-analyses examining value of statistical life (VSL) studies from the wage-risk literature (Mrozek and Taylor, 2002 and Viscusi and Aldy, 2003). See the Final Non-Road Diesel Rule (EPA, 2004) for a detailed discussion of the derivation of this value.

Monetized values for avoided mortality are subject to several additional adjustments reflecting factors relevant to health effects valuation including (a) discounting over the lag period established for the mortality effect, (b) consideration of real-world growth in per-capita income over the period leading up to the regulatory analysis year (i.e., 2020) and (c) income elasticity for the mortality endpoint (i.e., the degree to which WTP to reduce mortality will match real-world income growth). Following EPA and OMB guidelines for preparing economic analyses (EPA 2000b, OMB, 2003), discounting over the mortality lag period employed both 3% and 7% discount rates. Modeling of real-world growth in per-capita income is based on a projection of GDP growth (obtained from Kleckner and Neuman, 1999 and Standard and Poor's, 2000) as well as population growth for the adult cohort. With regard to income elasticity (for the mortality endpoint), research has shown that elasticity is related to the severity of the health endpoint. This has lead to the development of elasticities for different categories of health effect including minor, severe/chronic and mortality (Kleckner and Neuman, 1999). For this illustrative analysis, an income elasticity value of 1.2008 (in 2020) for mortality has been selected and applied in adjusting monetized estimates.

12.6 Presentation of Results

This illustrative analysis generated mortality incidence reduction estimates for the adult cohort for each regulatory option and provided two monetized benefits estimates for each regulatory option (a 3% discount-based estimate and a 7% discount-based estimate). The discount values are applied in valuing mortality results that are distributed across the lag period with the higher discount rate of 7% resulting in a lower overall monetized value for aggregated mortality incidence reductions compared with the 3% rate. Overall results for both options are presented in Table 12-1. As presented in Table 12-1, benefits for Option 1 range from $1.5 million to $44 million depending on the availability of advanced sorbents technology. Potential benefits for Option 2 range from $1.5 million to $130 million, again depending on the status of advanced sorbent technology.

Table 12-1. PM2.5 Co-Benefits Associated with CAMR Regulatory Options 1 and 2 in 2020

Regulatory Option		Annual Mortality Incidence Reduction (adults)	Benefits in 2020 (Millions of 1999 dollars)	
			3% discount rate	7% discount rate
Option 1				
	advanced sorbents <u>not available</u>	7	44	40
	advanced sorbents <u>available</u>	<1	1.5	1.4
Option 2				
	advanced sorbents <u>not available</u>	21	130	117
	advanced sorbents <u>available</u>	<1	1.5	1.4

12.7 Discussion of Uncertainties

Characterization of health-related benefits associated with PM reductions is a complex process which is subject to a variety of potential sources of uncertainty. Key assumptions underlying the estimate of avoided premature mortality include the following:

- Inhalation of fine particles is causally associated with premature death at concentrations near those experienced by most Americans on a daily basis. Although biological mechanisms for this effect have not yet been established, the weight of the available epidemiological and experimental evidence supports an assumption of causality.

- All fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because PM produced via transported precursors emitted from EGUs may differ significantly from direct PM released from automotive engines and other industrial sources. However, no clear scientific grounds exist for supporting differential effects estimates by particle type.

- The C-R function for fine particles is approximately linear within the range of ambient concentrations under consideration. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of PM including both regions that are in attainment with the fine particle standards and those that do not meet the standard.

- The forecasts for future emissions and associated air quality modeling are valid. Although recognizing the difficulties, assumptions, and inherent uncertainties in the overall enterprise, these analyses are based on peer-reviewed scientific literature and up-to-date assessment tools, and we believe the results are highly useful in assessing this rule.

Overall uncertainty in mortality-related benefits can be increased when a simplified methodology such as the illustrative analysis described here is employed due to potential limitations in capturing important relationships and inter-dependencies between key factors (e.g., use of coarse geographic scale which may miss important spatial gradients in emissions, air quality impacts and the location and density of exposed populations). Key sources of potential uncertainty associated with this illustrative analysis are briefly described below (Note: this discussion begins with a list of categories of potential benefits not considered quantitatively in this illustrative analysis):

- *Predicting power plant (sector) behavior in relation to mercury controls*: As shown in table 12-1, the benefits are highly dependent on the assumptions about what mercury-specific technologies will be chosen. For example, in the control option chosen in the final rule (option 1), the benefits range from $1.5 million (assume ACI works without an additional baghouse) to $42 million (assume ACI requires a baghouse). Note that costs and benefits may co-vary; if H-PAC is used then costs and benefits are lower and if ACI requires a baghouse then costs and benefits are higher. PM2.5 cobenefits presented in this section are ultimately dependent on decisions made (collectively) by industry in controlling mercury emissions under different cap-and-trade scenarios.

- *Unquantified benefits*: the illustrative approach used here, in focusing on the mortality endpoint, is likely to capture the majority of health-related benefits and the majority of total benefits (i.e., health plus welfare). However, important categories of potential PM-related benefits have been excluded to simplify modeling including: morbidity (e.g., hospital admissions for respiratory and cardiovascular endpoints, chronic bronchitis episodes, asthma exacerbations, non-fatal myocardial infarctions) and visibility benefits. Had these additional benefits been evaluated, overall benefits would be higher.

- *SR-MATRIX model*: In modeling population-level exposure reductions, the SR-MATRIX results in the form of population-weighted PM2.5-related exposure reductions generated for the Midwest/Northeast (ME) and Southeast (SE) regions were selected and averaged together, since these regions are where power plant-based emissions reductions are primarily expected to occur. A more comprehensive analysis of power-plant related emissions reductions (and associated exposure reductions) conducted using BenMAP, would have used a more spatially-refined geographic grid that tracked both ambient PM2.5 reductions and associated exposure reductions with greater precision and specificity. However, the more generalized approach used here is considered reasonable for a illustrative analysis of potential benefits. While the SR-MATRIX model has been shown to produce reasonably accurate predictions of direct PM emissions-related changes in ambient PM2.5 concentrations, the decision to average together population-weighted exposure changes (for ambient PM2.5) generated at the highly-aggregated ME and SE levels to produce a single value for use in this analysis does introduce considerable uncertainty. It is not known whether this simplifying approach biases the results in a more or less conservative direction.

- *Long-term cohort-based mortality function:* Use of the Pope et al., 2002-derived mortality function to support this analysis is associated with uncertainty resulting from:

(a) potential of the study to incompletely capture short-term exposure-related mortality effects, (b) potential mis-match between study and analysis populations which introduces various forms of bias into the results, and (c) failure to identify all key confounders and effects modifiers, which could result in incorrect effects estimates relating morality to PM2.5 exposure. EPA is researching methods to characterize all elements of uncertainty in the dose-response function for mortality. As is discussed in detail in the CAIR RIA (EPA, 2005), EPA has used two methods to quantify uncertainties in the mortality function, including: the statistical uncertainty derived from the standard errors reported in the Pope et al., 2002 study, and the use of results of a pilot expert elicitation conducted in 2004 to investigate other uncertainties in the mortality estimate. Because this analysis is a illustrative analysis, we do not quantify uncertainty with these two methods in this report. In the CAIR benefit analysis, the statistical uncertainty from the standard error of the Pope et al, 2002 study was twice the mean benefit estimate at the 95th percentile and one-fourth of the mean at the 5th percentile, while the expert elicitation provided mean estimates that ranged in value from less than one-third of the mean estimate from the Pope et al, 2002 study-based estimate to two and one-half times the Pope et al., 2002-based estimate. The confidence intervals from the pilot elicitation applied to the CAIR benefit analysis ranged in value from zero at the 5th percentile to a value at the 95th percentile that is seven times higher than the Pope et al., 2002-based estimate. These results are highly dependent on the air quality scenarios applied to the concentration-response functions of the Pope et all, 2002 study and the pilot expert elicitation. Thus, the characterization of uncertainty discussed in the CAIR RIA could differ greatly from what would be observed for CAMR due to differences in population-weighted changes in concentrations of PM2.5 (i.e., the location of populations exposure relative to the changes in air quality). EPA is continuing its research of methods to characterize uncertainty in total benefits estimates, and is conducting a full-scale expert elicitation. The full-scale expert elicitation is scheduled to be completed by the end of 2005.

- *Lag period for mortality reductions*: Failure to accurately capture the true lag period in mortality following reductions in ambient PM2.5 concentrations. Although the distributed lag approach described in Section 12.4 is reasonable given our current understanding of disease endpoint behavior including lung cancer, cardiovascular disease and respiratory disease (all of which are associated with PM2.5 exposure), the actual lag period for specific mortality causes linked to PM2.5 exposure could differ from that used in this analysis. In the CAIR RIA, we conducted sensitivity analyses to investigate how different lag structures could influence the total benefits, and concluded that substitution of the most plausible alternative lag structures had little overall impact on the estimate of total benefits (reductions are on the order of 5 to 15 percent).

- *WTP-based valuation for mortality*: The WTP-based value used in this analysis to monetize mortality incidence reductions is enhanced by being based on a variety of wage-risk studies evaluated using meta-analysis techniques. However, this WTP-based value may misrepresent the actual societal value for reductions in PM-related mortality if societal perception of mortality risk related to PM exposure differs significantly from that associated with job-related hazards.

12.8 References

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Tables

Figures

APPENDIX A-1

MERCURY LOAD REDUCTION ANALYSIS AND RESPONSE FOR EAGLE BUTTE (SOUTH DAKOTA)

A1.1 Introduction

This appendix contains technical details of input parameters, model calibration and scenario projection in response to atmospheric mercury loading reduction in Eagle Butte, South Dakota.

Eagle Butte is located in north/central portion of South Dakota on the Cheyenne River Sioux Tribal (CRST) Lands. The USGS Hydrologic Unit Code (HUC) for this watershed is: 01010302 (Upper Lake Oahe). The CRST inhabits the Cheyenne River Sioux Reservation, which is located within the former Dakota Territory, in an area that is known today as central South Dakota. The Reservation is over 2.8 million acres, comparable in size and shape to the State of Connecticut. The Reservation encompasses Dewey and Ziebach counties and is comprised of a portion of the aboriginal territory of the Great Sioux Nation and the United States. The boundaries of Cheyenne River Reservation are set forth in Section 5 of the Act of March 2, 1889, 25 Stat. 888 (1889). The Cheyenne River (Wakpa Waste) and the Missouri River (Mni Sose') form the southern and eastern Reservation boundaries, respectively. Bordering the Cheyenne River Sioux Reservation on the north is the Standing Rock Sioux Reservation. The world's largest earthen dam was constructed by the U.S. Army Corps of Engineers (USACE) on the Missouri River downstream of the Cheyenne River Sioux Reservation in 1958 and formed Lake Oahe.

The climate of the Cheyenne River Basin is characterized as semi-arid continental, with large variations in precipitation and temperature. The high plains of eastern Wyoming are relatively dry. During the period from 1931 to 1990, the mean annual temperature was about 46 degrees Fahrenheit (^{o}F). Orographic effects induced by the Black Hills are responsible for dramatic spatial changes in climate within western South Dakota. As elevation increases, precipitation generally increases and temperature generally decreases. Precipitation also varies spatially within the Black Hills, from a mean annual value of less than 12 inches in the southwest, to a maximum of more than 29 inches in the north.

The Cheyenne and Belle Fourche Rivers both originate in east-central Wyoming in areas of Eocene and Paleocene-age sedimentary rock exposures. The Paleocene-age rocks host thick, continuous coal seams. The two rivers around the domal Black Hills uplift where mostly metasedimentary rocks of precambrain age, carbonate rocks of Paleozoic age, and mixed sandstones and shales of early Mesozoic age are exposed. The two rivers stay within the outcrop belts of Late Cretaceous-age Pierre Shale, which is a marine shale that contains high concentrations of iron, manganese, and limestone concretions. The Pierre shale also is characterized by an abundance of low permeability bentonite clay. As a result, exposures of this unit are prone to high runoff during periods of intense or extended rainfall

The site modeled (Lee Dam), is a shallow, well-mixed system and has a water surface area of 0.2 km^2 and a watershed:lake area ratio of 22.6. There are power plants in the vicinity of this site, although the prevailing meteorology likely transports most emissions east of the site. Currently, consumption advisories are in place on reservation lands due to high levels of mercury in piscivorous fish. Atmospheric deposition data, total and methylmercury concentrations in sediments, water and biota have all been collected as part of this research. To mode this system, we used the empirical data collected at this site to parameterize both the SERAFM and WASP models.

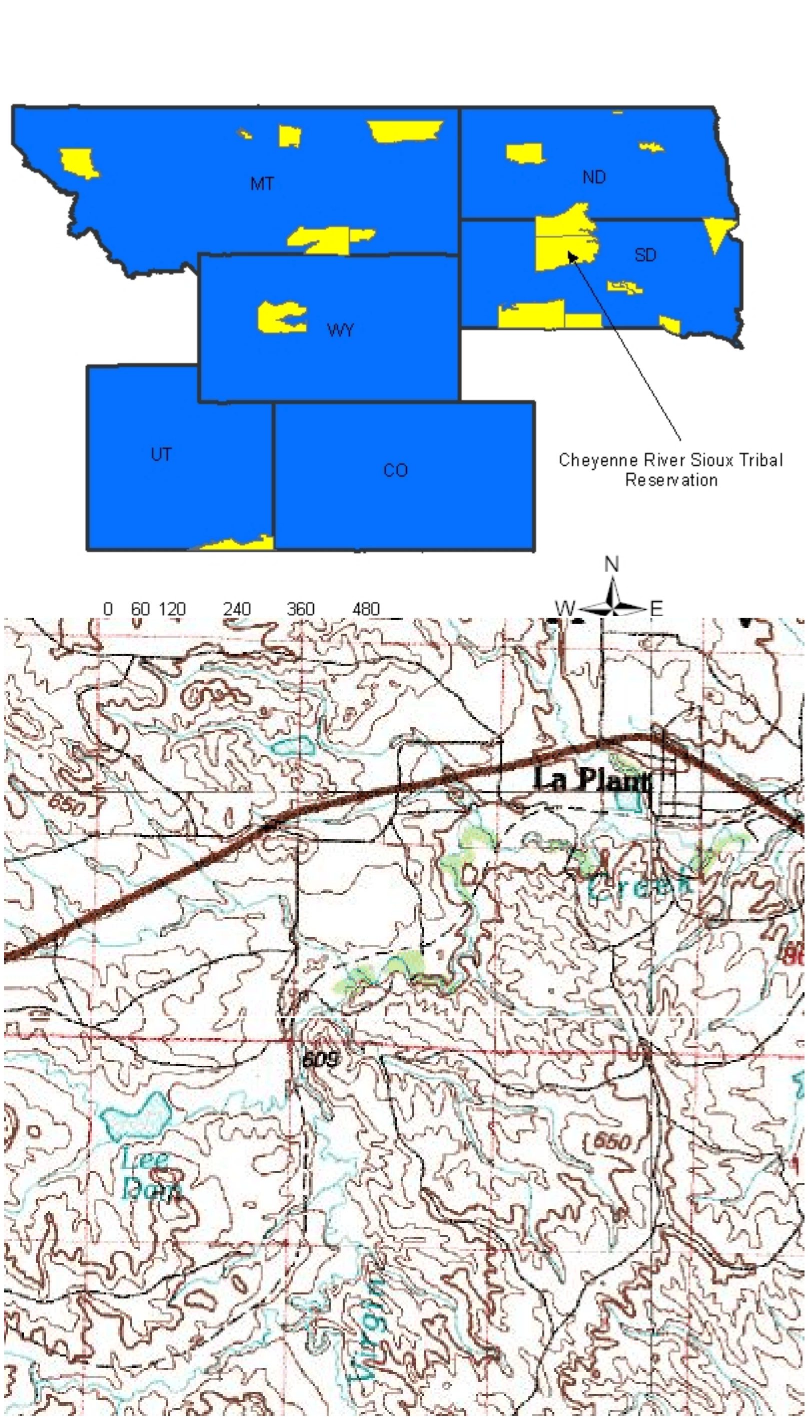

Figure A1-1. Location of Lee Dam (lower left quadrant) on La Plant SW quadrangle

Top map shows tribal lands in EPA Region 8 (highlighted in yellow)

A1.2 Site Characteristics and Model Parameterization

Mercury dynamics at this site are currently being studied as part of an EPA Regional Office RARE Grant awarded in 2003 (http://www.epa.gov/osp/regions/RARE_Region8.pdf). Data used to parameterize ecosystem and food web models were obtained from this study (Unpublished Data, John Johnston, EPA/ORD ERD-Athens).

Table A1-1 presents Northern Pike sample data collected at the Lee Dam site that were used to characterize empirically calibrated BAFs for this site. Data were normalized to correct for length/age variability, long transformed and summarized in Table A1-2.

Table A1-1. Observed Mercury Concentrations in Northern Pike from Lee Dam (DMA-80 results)

Sample ID	Weight	Height	[TOTHg] ppb	Mass Fish g	Length Fish mm
fpsd150401	0.1986	0.1829	683.96	1900	680
fpsd150402	0.1943	0.2779	1094.07	1980	854
fpsd150403a	0.1520	0.1695	824.79	2460	730
fpsd150403b	0.1363	0.1350	725.34	2460	730
fpsd150404	0.2250	0.2356	790.33	1272	590
fpsd150405	0.1683	0.1427	622.06	1758	669
fpsd150406	0.1885	0.2024	801.88	1184	580
fpsd150407	0.1524	0.1947	951.94	1858	662
fpsd150408	0.2157	0.2175	756.57	708	500
fpsd150409	0.2378	0.1690	525.43	550	468
fpsd150412	0.2898	0.0803	199.89	n/a	n/a
fpsd150410	0.2066	0.0843	294.42	n/a	n/a
fpsd150411	0.2616	0.0749	206.19	n/a	n/a

BAFs of both benthic macroinvertebrates and zooplankton were calculated directly from field data and used in the model. The expression used for mercury exposures under atmospheric loading reduction (dissolved organic mean concentration from field data used was 0.5 ng L^{-1}) is as follows:

$$C_{water}\ (\text{ng L}^{-1}) = 0.5\ \exp(-0.0011 * t(\text{day}))$$

This is consistent with the predictions of both SERAFM and WASP, such that after two years epilimnion concentrations were half of the starting concentration and after five years concentration reached a 90% reduction.

Annual water temperature was simulated by the following function:

$$T = 15 + 10*\sin[(0.0172*T(\text{days})-0.280)],\ (T_0 = \text{April } 1)$$

This is intended to capture the annual variability in temperature (which affects metabolic demand), matching minimum and maximum temperatures collected in the field data, as well as the lowest temperatures reached in January.

Table A1-2. Statistical summary of northern pike length normalized BAF (4 years) for Eagle Butte (used in SERAFM)

Northern Pike	Value
BAF – MeHg water (kg L^{-1})	8.9×10^5
Stdev BAF (kg L^{-1})	2.65×10^5
N samples	42
N of age 4 pike for length correction	10
Normalized HgT (μg g^{-1})	0.89
Stdev normalized Hg (μg g^{-1})	0.27

A1.3 SERAFM Simulations of Eagle Butte, Lee Dam

The following tables and parameter values summarize the application of the SERAFM model to simulate mercury dynamics in Eagle Butte, Lee Dam. The model was first run to steady state after being calibrated to the observed data and then used for scenario projections of 1) a 50% decline in atmospheric deposition and 2) removal of coal fired utilities from overall projected deposition using the CMAQ and REMSAD models.

Table A1-3. SERAFM Parameter Values for Eagle Butte

	Measured		Predicted
Parameter	Range	Mean	
Water Column MeHg Unfiltered (ng L^{-1})	0.4 – 2.9	1.0	0.82
Water Column HgT Unfiltered (ng L^{-1})	0.5 - 100	6.9	10.2
Sediment MeHg (ng g^{-1} dry)	0. 062 – 1.74	0.40	0.29
Sediment HgT (ng g^{-1} dry)	28.1 – 95	44.1	63.9
Age 4 Northern Pike Tissue Hg (μg/g)	0.5 – 2	0.89	0.97
Observed BAF: FishHg/MeHg Water		8.9×10^5	

Table A1-4. Calibrated SERAFM Rate Constants for Eagle Butte

Eagle Butte: SERAFM Calibrated Rate Constants			
Process	**Media**	**Value**	**Units**
Methylation*	Epilimnion	0.04	per day
	Sediment	0.001	per day
Demethylation	Epilimnion	0.1	per day
	Sediment	0.40	per day
Biotic Reduction	Water Column	0.03	per day
Photo-Degradation	Water Column	0.002	Per day per E/m^2-day
Photo-Reduction (Vis)	Water Column	0.003	Per day per E/m^2-day
Photo-Reduction (UV-B)	Water Column	2.825	Per day per E/m^2-day
Photo-Oxidation (UV-B)	Water Column	5.885	Per day per E/m^2-day
Dark Oxidation	Water Column	1.44	per day

*Note: Methylation rate constant in Lee Dam is increased to account for the reservoir effect of repeated flooding and drying, and the increased zone of redox potential where methylation is found to be increased.

Table A1-5. SERAFM 50% Load Reduction Scenario for Eagle Butte

Lee Dam, Eagle Butte 50% Loading Scenario			
	Slow	Medium	Fast
Water	2	2	4
Sediment	3	4	6
Fish	2	3	4

Note: Fast = 1 cm active sediment layer, D (macro-dispersion coefficient) = 10^{-4} cm^2 s^{-1}; Medium = 2 cm active sediment layer, D = 10^{-4} cm^2 s^{-1}; Slow –3 cm sediment, D = $5x10^{-5}$ cm^2 s^{-1}.

Table A1-6. SERAFM Zero-Out Scenario for Eagle Butte (Removal of Deposition attributed to coal-fired utilities) in the CMAQ and REMSAD Models

	Eagle Butte – Lee Dam		
	Slow	Med	Fast
Epilimnion	2	3	4
Hypolimnion	--	--	--
Sediment	3	4	6
Fish	2	3	4

A1.4 WASP7 Simulations of Eagle Butte, Lee Dam

Simulations were set up with the basic parameters from the SERAFM model of Lee Dam. This water body receives watershed loadings of solids and mercury, but has no significant outflow. Solids loadings from the watershed are balanced by in-lake mineralization and burial. Mercury loadings from direct deposition and from the watershed are subject to volatilization and burial losses.

WASP7 simulations were run for a total of 200 years. The first 100 years represent the buildup of mercury to steady-state levels using present loadings. Mercury loadings were then cut 50%, and the attenuation period was tracked for 100 years.

The solids balance is represented in the following figures (Figure 2-Figure 3). The water column equilibrated at a silt concentration of just over 2 mg L^{-1}. The organic matter (OM) represents biotic solids (including phytoplankton, periphyton, and macrophytes) and detritus.

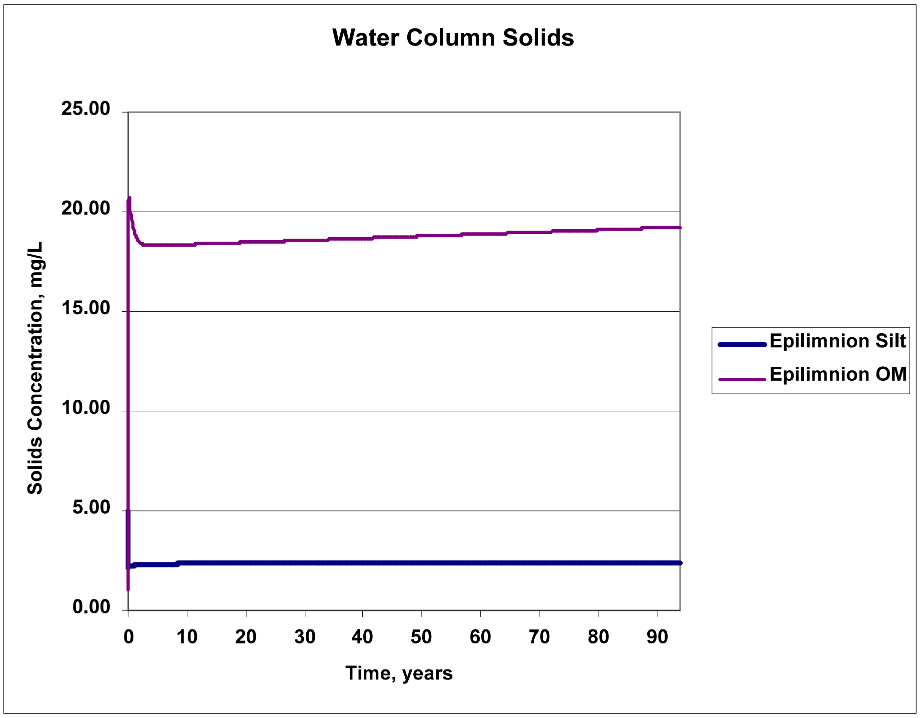

Figure A1-2. WASP Water Column Solids Calibration.

Sediment solids are balanced by the composition of the erosion load, the in-lake biotic production, and the mineralization of OM. The sediment composition is primarily abiotic silt (70%), with a sand fraction just under 20% and an organic fraction just over 10%.

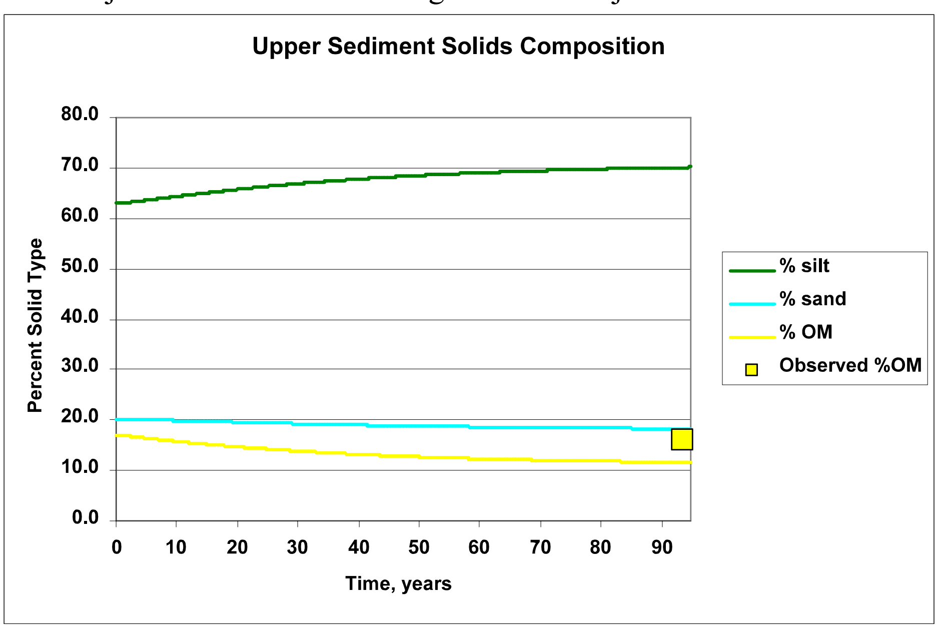

Figure A1-3. WASP Upper Sediment Solids Calibration

The burial rate is calculated internally from the solids balance. Given the specified loads, production, and mineralization rates, burial stabilized at about 0.24 cm/yr.

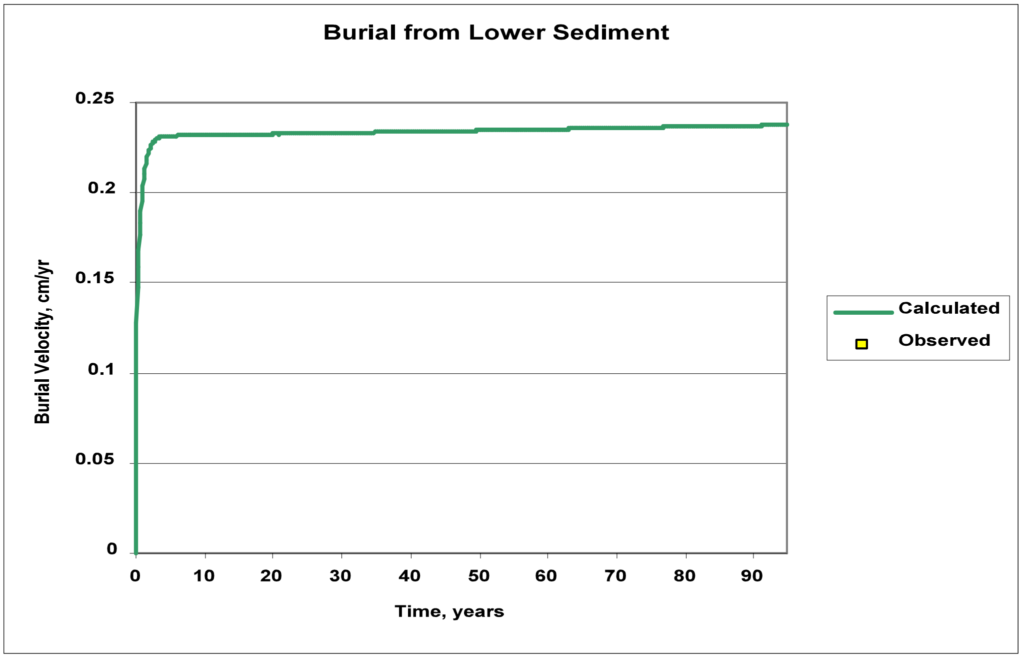

Figure A1-4. WASP Burial Rate Calibration

Total mercury built up over the first 100 years to 9.5 ng L^{-1} in the water, and 180 ng g^{-1} in the sediment. The sediment levels are higher than observations, indicating an additional loss mechanism (e.g., faster burial or uptake by macrophytes). Methyl mercury levels built up to reasonable concentrations in the water and the sediment.

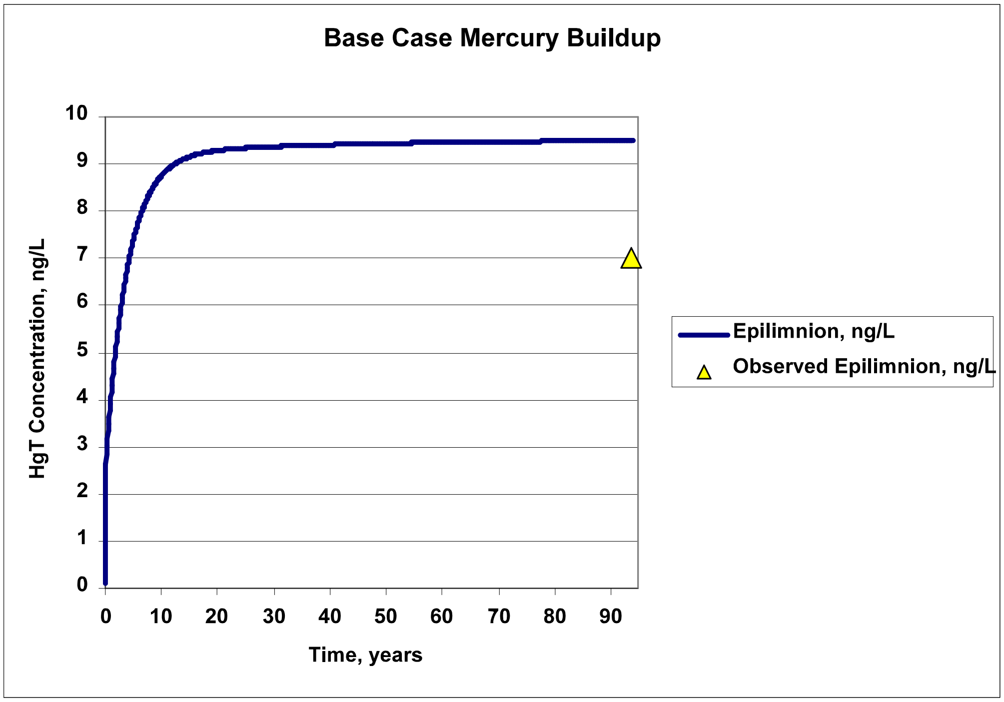

Figure A1-5. WASP Total Mercury Buildup in Water

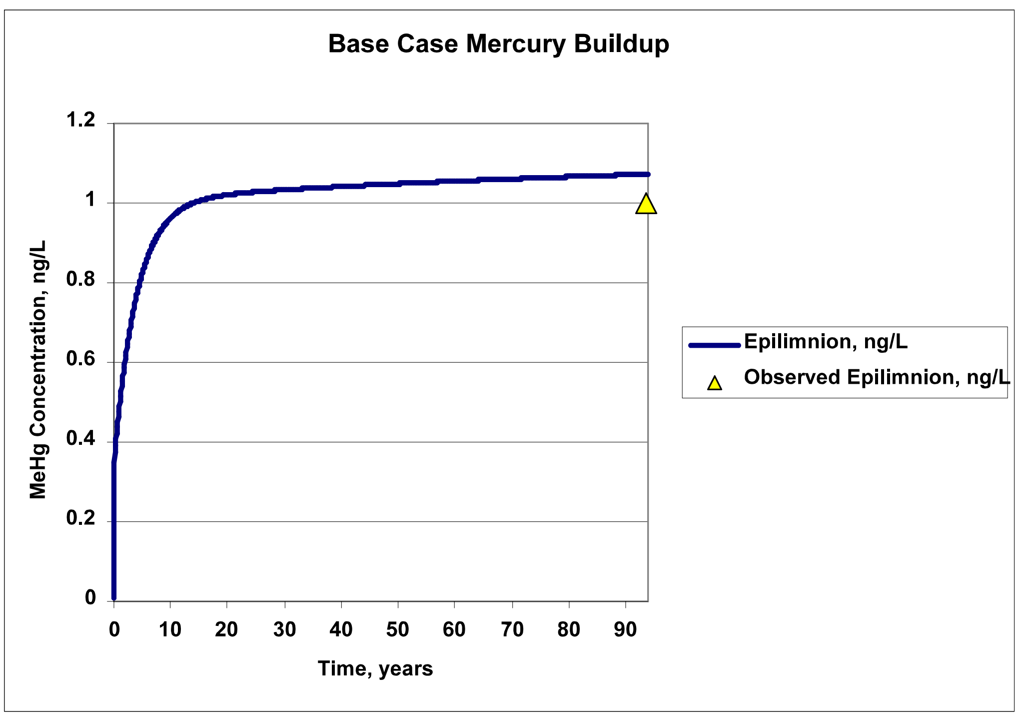

Figure A1-6. WASP Methyl Mercury Buildup in Water

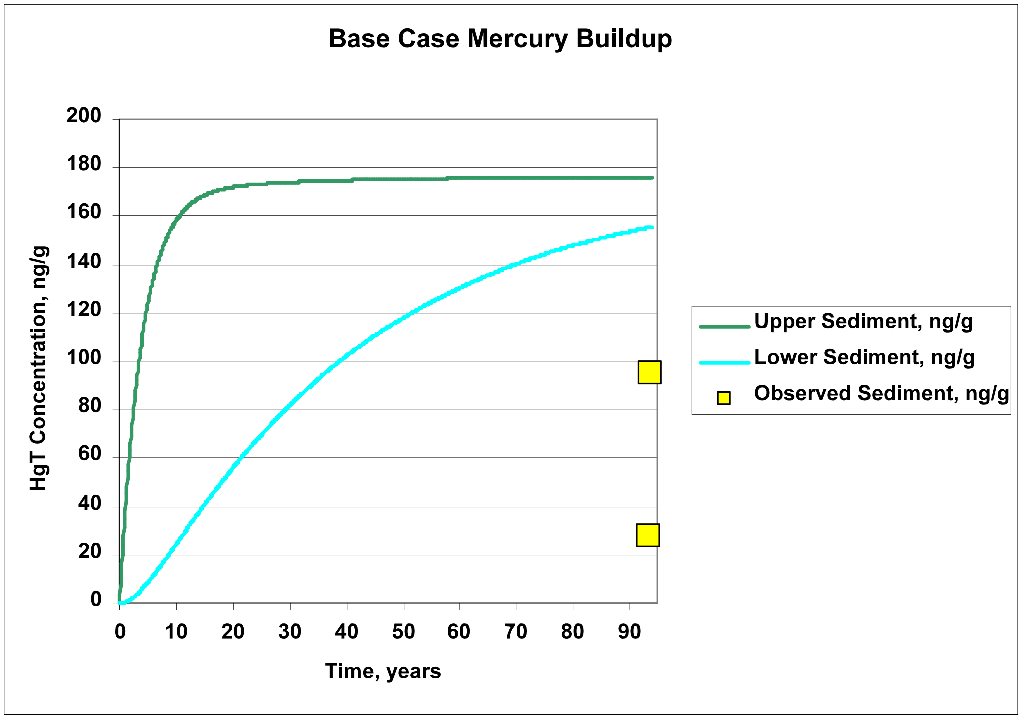

Figure A1-7. WASP Total Mercury Buildup in Sediment

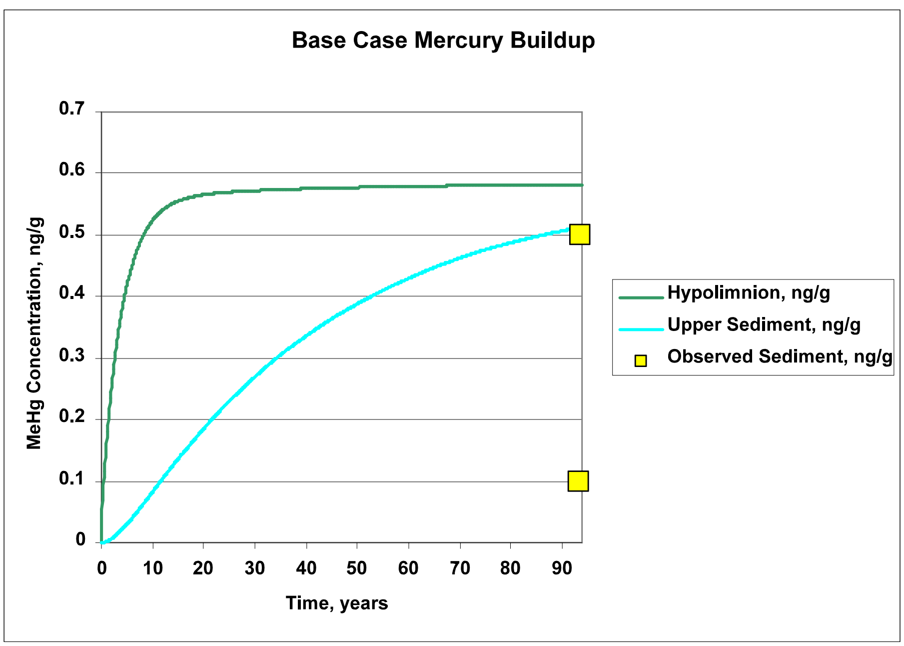

Figure A1-8. WASP Methyl Mercury Buildup in Sediment

After external mercury loads were reduced 50%, the mercury levels in the water column and surface sediment declined rapidly, on the order of years. Mercury levels in the lower sediment layer declined slowly over the following decades due to burial.

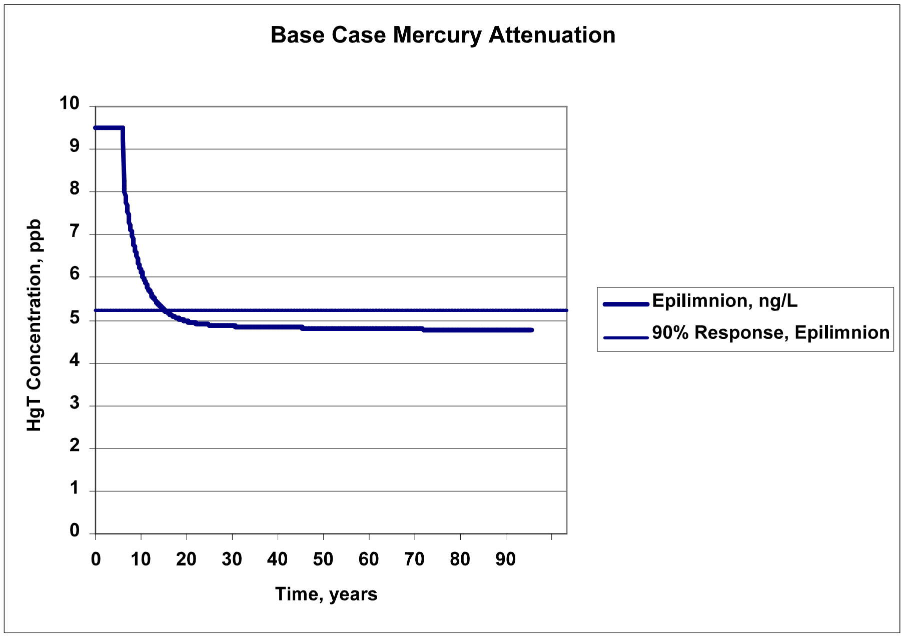

Figure A1-9. WASP Total Mercury Attenuation in Water

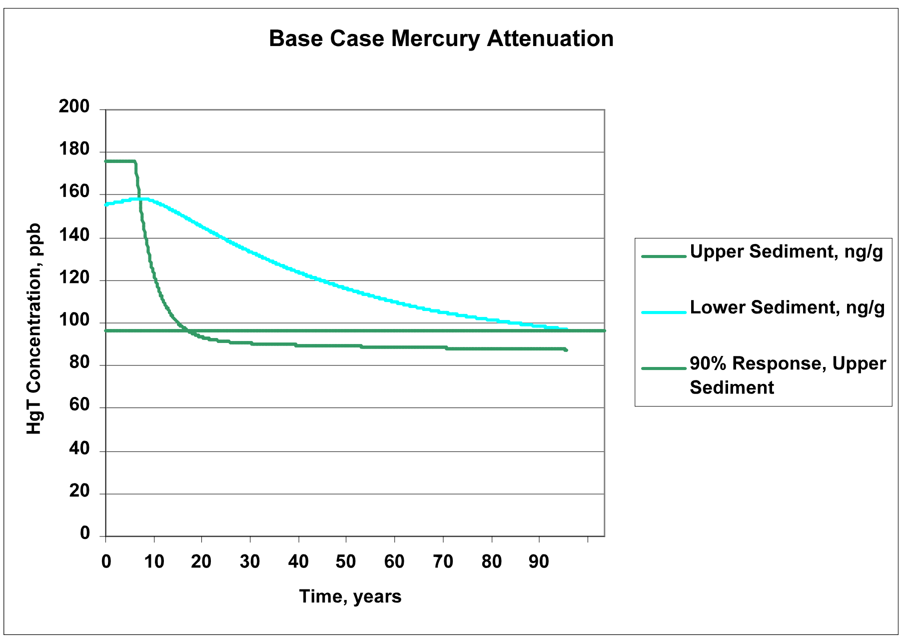

Figure A1-10. WASP Total Mercury Attenuation in Surface Sediment

Three scenarios were simulated representing fast, medium, and slow estimates of recovery. The upper sediment layer thickness was varied from 1 cm, 2 cm, and 3 cm. For the slow scenario, the sediment-water dispersion coefficient was reduced by half from 10^{-4} cm^2 sec^{-1}. Response times are summarized in the following table, and presented graphically for water and sediment.

Table A1-7. WASP Forecasted Mercury Concentrations in Eagle Butte Sediments in Response to 50% Loading Reduction Scenario

Mercury Response Times for Lee Dam, in years			
Compartment	Fast	Medium	Slow
Epilimnion	5	9	14
Surface Sediment	6	11	16

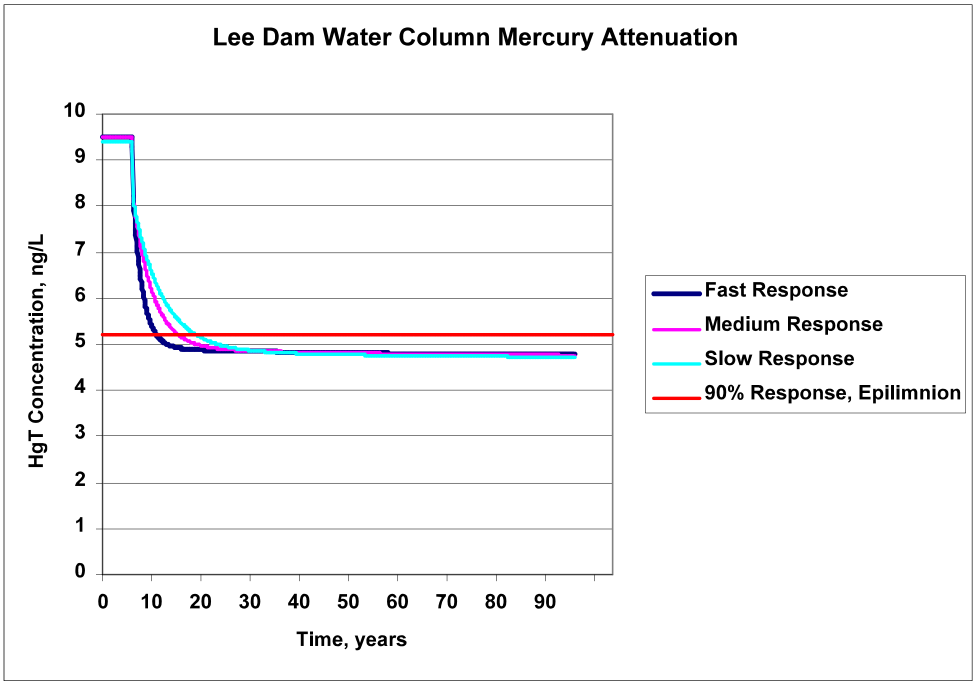

Figure A1-11. WASP Attenuation Sensitivity in Water

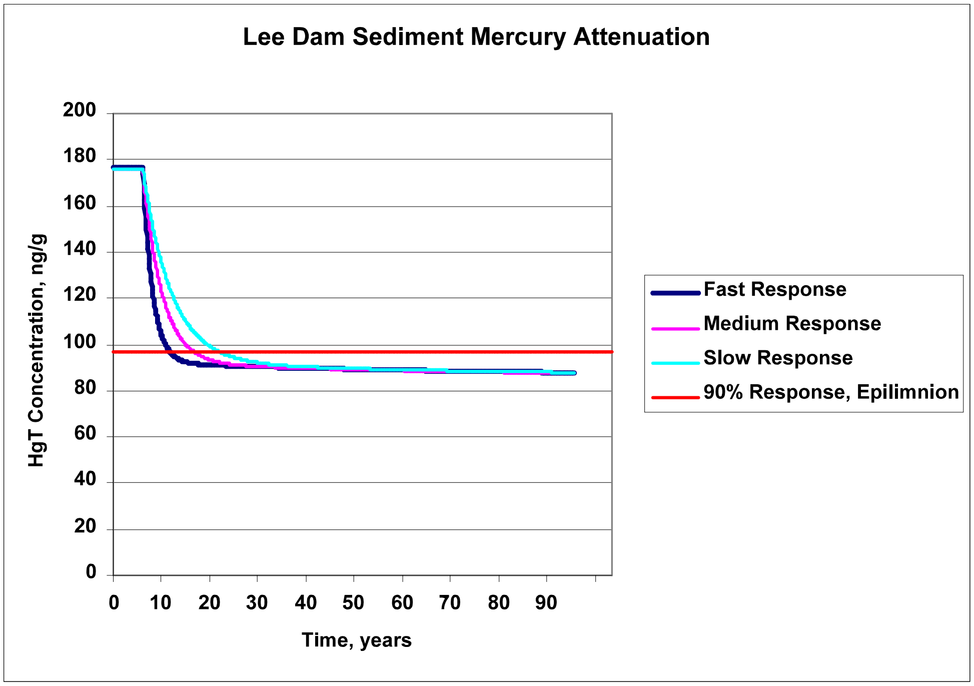

Figure A1-12. WASP Attenuation Sensitivity in Water

A1.5 BASS Model Simulations of Methylmercury and Fish Dynamics in Lee Dam, Eagle Butte, SD—Response to Changes in Mercury Loading

The following figures illustrate the response time to both loadings and load reductions of mercury from atmospheric sources. BASS is run in FGETS mode, i.e., without population dynamics, to simulate whole-body organic mercury chemical residues in weight and length classes of community members. BASS is calibrated by adjusting species growth rates, dietary composition and concentrations and bioaccumulation factors of benthic macroinvertebrates and zooplankton. Field data for length, weight and ages of species are corroborated by the BASS model. Predicted model concentrations for northern pike, yellow perch, black bass and black crappie agree with observed field concentrations for these species. Exposure to mercury occurs through food and gill uptake only. The macroinvertebrate BAF used is $1.14^{x}10^{5}$, and the zooplankton BAF used is $1.67^{x}10^{5}$, both of which are calculated from field data at Lee Dam.

Lee Dam is predominantly a northern pike/yellow perch community, but also includes subdominant black crappie, black bass and spottail shiner, all of which have been sampled for residue analysis over a two year period using a combination of gill nets and seine nets. An abundance of submerged aquatic vegetation provides suitable cover for these species. Northern pike is the top predator (SERAFM Level 4) in this lake system, feeding primarily on yellow perch.

Dissolved methylmercury concentrations are set at 0.5ng L^{-1} for the base case, and equilibrium is reached in 1500 days for length class 3 pike and 3500 days for length class 4 pike.

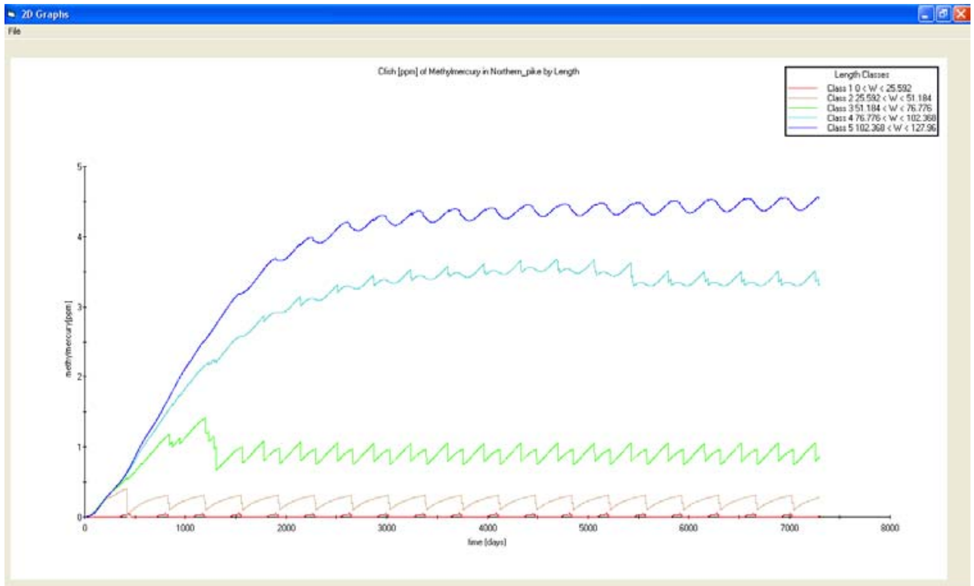

Figure A1-13. Base Case Response of Northern Pike to Methylmercury Exposure

Dynamics of the community co-dominants to reductions in atmospheric loading were simulated by decreasing epilimnion dissolved methylmercury concentrations from 0.5ng L^{-1} at time zero to approximately 0.25ng L^{-1} at year 2 and approximately 0.05ng L^{-1} at year 5 to be consistent with predicted water column dynamics from SERAFM and WASP for this lake system (reaching a 90% reduction on the order of 5 years).

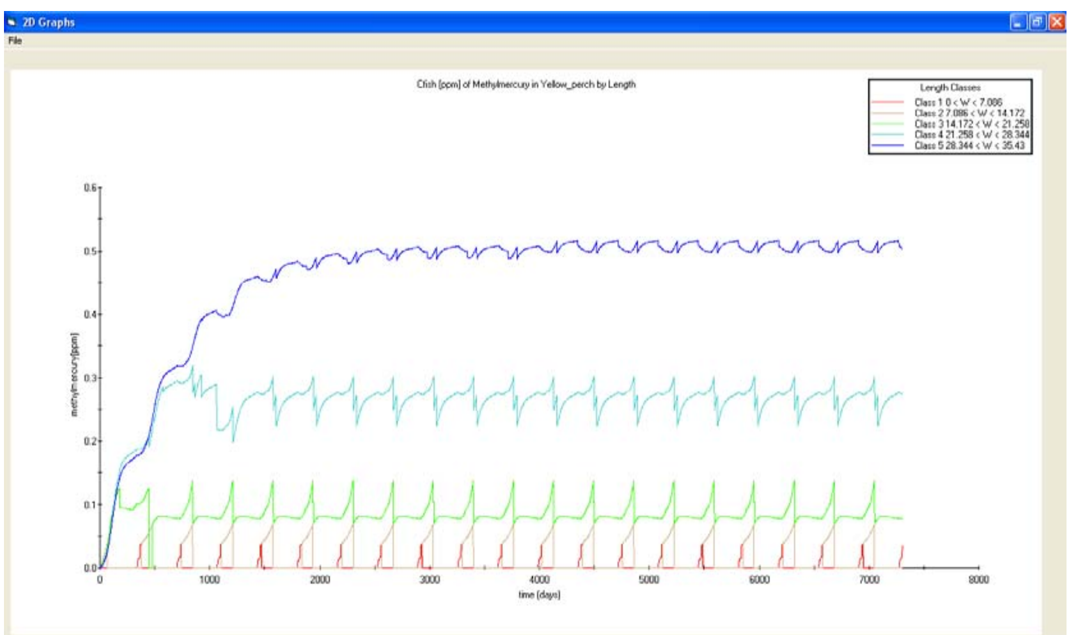

Figure A1-14. Base case response of yellow perch to methylmercury exposure (0.5ng/L)

The response time of the concentration of mercury in fish is roughly twice the duration of the simulated water concentration response. The largest northern pike length classes reach the action limit concentration of 0.3 ppm in approximately 5 - 10 years. The largest yellow perch reach the 0.3 ppm level in a span of 5 to 8 years. Note that a roughly equivalent time lag is predicted for the biotic response in this dynamic and relatively rapidly responding system (see SERAFM and WASP forecasts).

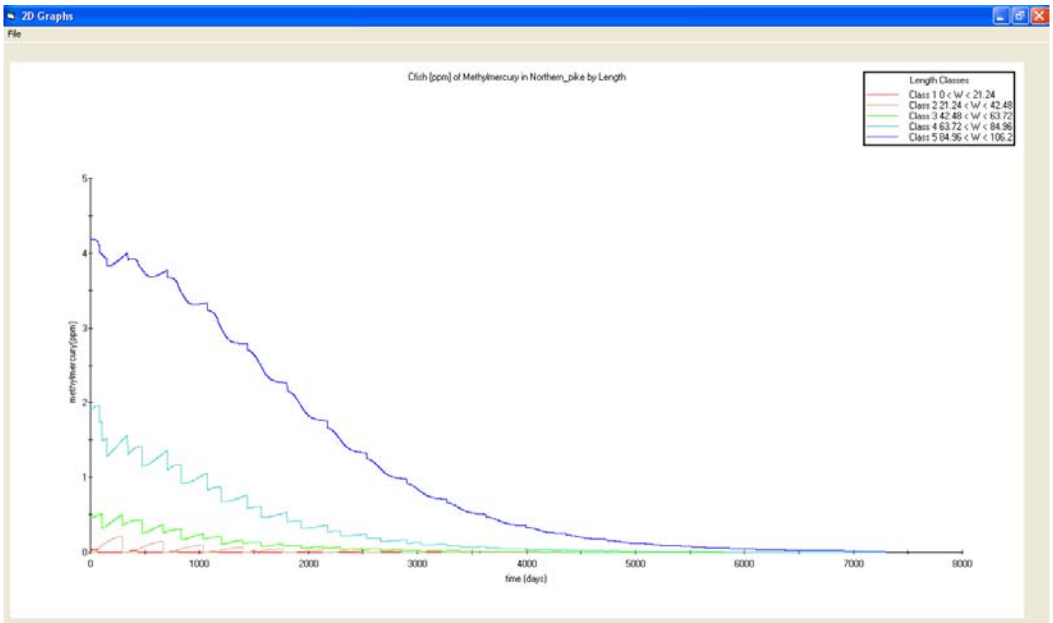

Figure A1-15. Attenuation of Methylmercury in Northern Pike after Load Reduction

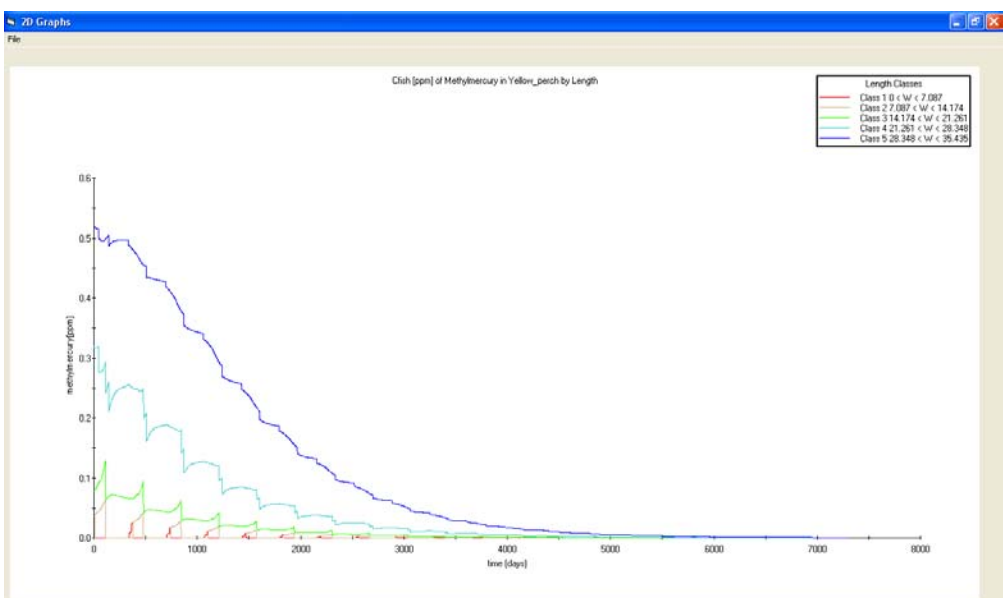

Figure A1-16. Attenuation of Methylmercury in Yellow Perch after Load Reduction

APPENDIX A-2

MERCURY LOAD REDUCTION ANALYSIS AND RESPONSE FOR PAWTUCKAWAY LAKE (NEW HAMPSHIRE)

A2.1 Introduction

Pawtuckaway Lake is a medium sized, seepage lake in Nottiham, New Hamsphire. This lake a water surface area of 3.6 km^2 and a watershed:lake area ratio of 13.7. Lakes in this region are characteristic of undisturbed lakes within the Northeastern Highlands Ecoregion (Omernik, 1987; US EPA, 2000). This lake was part of a recent study of mercury dynamics across a number of Vermont and New Hampshire Lakes funded by EPA's Office of Research and Development under the Regional Environmental Monitoring and Assessment Program (Kamman et al., 2004). Site specific data used to parameterize the mercury cycling models developed for Pawtuckaway Lake were from Kamman and Engstrom (Kamman and Engstrom, 2002) and Kamman (2004) or http://www.vtwaterquality.org/lakes/docs/lp_remap-datareport.pdf. An excerpt from Kamman and Engstrom (2002) describing the region is given below:

"Most lakes in this area occupy undisturbed forested catchments, which are a mix of deciduous or coniferous vegetation overlying soils ranging from stony to silty loams. Bedrock geology is largely schistose or granitic, and most watersheds are poorly buffered. Some shales and slates are in evidence near High Pond in Vermont, and the buffering capacity of this watershed is enhanced accordingly. These watersheds have experienced varying degrees of deforestation during settlement, but have regrown to forest in the past 75–150 years."

Fish data were normalized for age/length variability. Details of data transformations can be found in Kamman (2004). SERAFM was calibrated to forecast fish tissue concentrations in 4 year old yellow perch from this system. These data are summarized in Table 1.

Table A2-1. Summary of Yellow Perch Mercury Data from Pawtuckaway Lake

Pawtuckaway Lake	Value
Mean normalized Yellow Perch Hg (ug/g)	0.21
Range Hg Yellow Perch (ug/g)	0.12-0.40
Mean MeHg Water Concentration Epilimnion (ng/L)	0.19
Range MeHg Epilimnion (ng/L)	0.14-0.24
HgT Water Concentration (ng/L)	
Mean	1.65
Log Mean	0.18
Range HgT water (ng/L)	0.7-3.8

A2.2 SERAFM Application

Table A2-2. Pawtuckaway Lake Parameter Values

Parameter	Lake Pawtuckaway
Watershed Area	50,007,406 m^2
Percent Impervious	1%
Percent Forest	88%
Percent Riparian	10%
Percent Upland	1%
Lake Area	3,6412,200 m^2
Catchment/Lake Ratio	13.73
Epilimnion Depth	2 m
Hypolimnion Depth	3 m
Hypolimnion Anoxia	Yes
Hydraulic Residence Time	0.45 yrs
Inflow/Outflow	4.05×10^7 m^3/yr
Water pH	6.45
Epilimnion DOC	5.46 mg/L
Hypolimnion DOC	5.55 mg/L
Trophic Status	Dystrophic
Annual Precipitation	102 cm/yr
HgII Conc. in Precip	10 ng/L
Wet Deposition (HgII)	10.2 ug/m2/yr
Dry Deposition (HgII)	10.2 ug/m2/yr
Wet Deposition (HgII)	0.153 ug/m2/yr
Dry Deposition (HgII)	0.153 ug/m2/yr

Table A2-3. A Comparison of Measured and Baseline Steady State Values for Pawtuckaway Lake

	Measured		Predicted
Parameter	Range	Mean	
EPI MeHg Unfiltered	0.14 – 0.24 ng/L	0.19 ng/L	0.35 ng/L
EPI HgT Unfiltered	0.71 – 3.8 ng/L	2.26 ng/L	3.57 ng/L
HYP MeHg Unfiltered	2.38 – 3.44 ng/L	2.91 ng/L	0.38 ng/L
HYP HgT Unfiltered	6.94 – 34.54 ng/L	20.74 ng/L	5.63 ng/L
Sediment MeHg		7 ng/g	6 ng/g
Sediment HgT		290 ng/g	237 ng/g
Perch Tissue Hg	0.12 – 0.4 ug/g	0.21 ug/g	0.23 ug/g (0.21-0.26)
BAF: FishHg/MeHg	1.11×10^6		

Table A2-4. Lake Pawtuckaway SERAFM Calibrated Rate Constants

Process	Media	Value	Units
Methylation	Epilimnion	0	per day
	Hypolimnion	0.01	per day
	Sediment	0.01	per day
Demethylation	Epilimnion	0.0001	per day
	Hypolimnion	0.001	per day
	Sediment	0.4	per day
Biotic Reduction	Water Column	0.03	per day
Photo-Degradation	Water Column	0.002	per day per E/m^2-day
Photo-Reduction (Vis)	Water Column	0.003	per day per E/m^2-day
Photo-Reduction (UV-B)	Water Column	2.825	per day per E/m^2-day
Photo-Oxidation (UV-B)	Water Column	5.885	per day per E/m^2-day
Dark Oxidation	Water Column	1.44	per day

Table A2-5. Time to Reach 90% Steady State After 50% Reduction in Atmospheric Deposition

Lake Pawtuckaway	Fast	Medium	Slow
Epiliminion	59	115	179
Hypolimnion	79	154	>180
Sediments	80	125	>180
Fish	34	56	64

Fast = 1 cm active sediment layer, D (macro-dispersion coefficient) = 10^{-4} cm^2/s
Medium = 2 cm active sediment layer, D = 10^{-4} cm^2/s
Slow – Slow Responding Sediment: 3 cm sediment, D = $5x10^{-5}$ cm^2/s

Table A2-6. SERAFM Model Forecasts with Zero-Out Scenario for Coal-Fired Power Plants (Medium Response Time Scenario)

Lake Pawtuckaway	Time
Epilimnion HgT (ng/L)	95
Epilimnion MeHg (ng/L)	--
Hypolimnion (ng/L)	122
Sediment MeHg (ng/g)	125
Fish (ng/g)	47

A2.3 WASP Model Calibration

WASP simulations of Lake Pawtuckaway were set up with the basic parameters from the SERAFM model. In addition to direct deposition of mercury, this water body receives watershed loadings of solids and mercury. Solids loadings from the watershed and internal production of organic matter are balanced by outflow, in-lake mineralization, and burial. Mercury loadings from direct deposition and from the watershed are subject to outflow, volatilization and burial losses. WASP simulations were run for a total of 200 years. The first 100 years represent the buildup of mercury to steady-state levels using present loadings. Mercury loadings were then cut 50%, and the attenuation period was tracked for 100 years.

WASP calibration was conducted for a base case scenario with an active sediment depth of 2 cm and a sediment macro-dispersion coefficient (E) = 10^{-4} cm^2/sec. This calibration employed the same deposition and resuspension velocities for silts and organic matter in the SERAFM model. Solids were calibrated to match the observed porosity of 0.93, OM fraction of 35%, and burial rate of 0.05 cm/yr

The solids balance is represented in Figure 1 and Figure 4. The epilimnion solids are mostly organic, including runoff and plankton. Hypolimnion organic matter includes living periphyton and macrophytes, as well as detritus. Sediment solids are balanced by the composition of the erosion load, the in-lake biotic production, and the mineralization of OM. The sediment composition is a balance between silt and organic matter (35% each), and sand (30%). The burial rate is calculated internally from the solids balance. Given the specified loads, production and mineralization rates, burial stabilized at 0.04 cm/yr, close to observations.

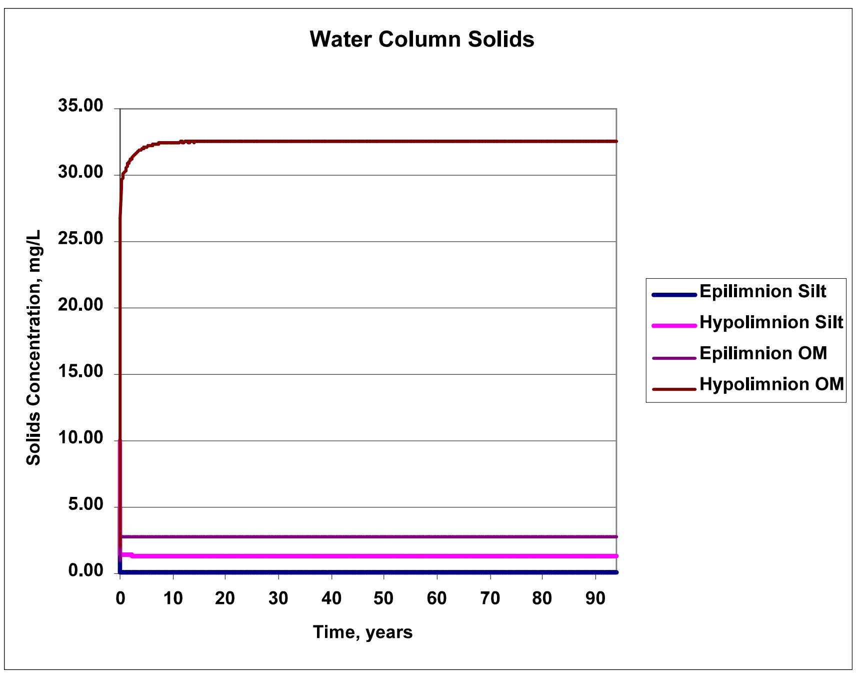

Figure A2-1. WASP Water Column Solids Calculation

Figure A2-2. WASP Solids Simulation for Surface Sediment

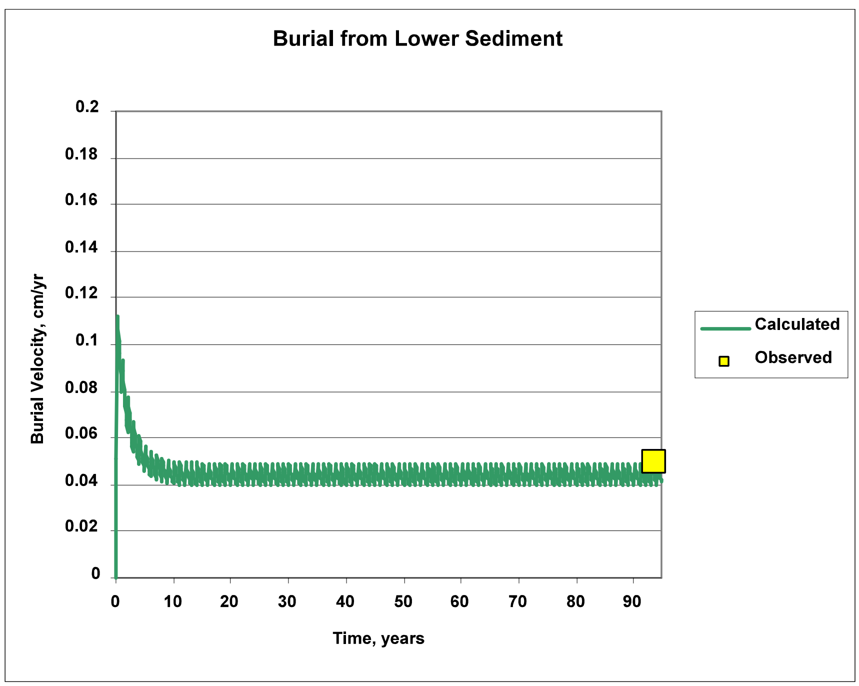

Figure A2-3. WASP Simulation of Burial Velocity

WASP was calibrated to better match total mercury levels by adjusting the fractions of sand and silt in the watershed erosion load resulting in a silt/sand/OM erosion composition of 50/45/5 percent. The hypolimnion methylation and demethylation rate constants were adjusted to try to obtain reasonable a MeHg concentration using reasonable coefficients. The final hypolimnion rate constants were the same as in the active sediment layer, reasoning that the hypolimnion of this lake has a lot of plant substrate and organic matter.

Total mercury built up over the first 100 years to 8 ng/L in the epilimnion, 15 ng/L in the hypolimion, and 300 ng/g in the upper sediment. Methyl mercury levels built up to 0.5 – 1 ng/L in the water, and 6 ng/g in the sediment. Although HgT is overpredicted in the epilimnion and MeHg is underpredicted in the hypolimnion, sediment levels match the data very well.

Note that while the model calculates high OM in the hypolimnion, much of it conceptually is macrophyte biomass. The epilimnion HgT concentration is high, while the hypolimnion HgT is reasonably good. The calculated sediment HgT for the upper 2 cm and the lower 10 cm bracket the observed value.

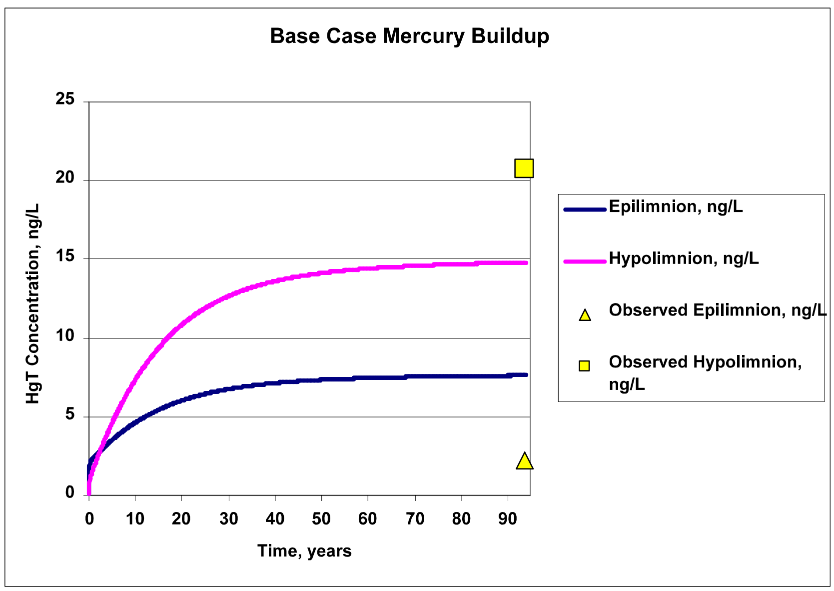

Figure A2-4. WASP Total Mercury Buildup in Water

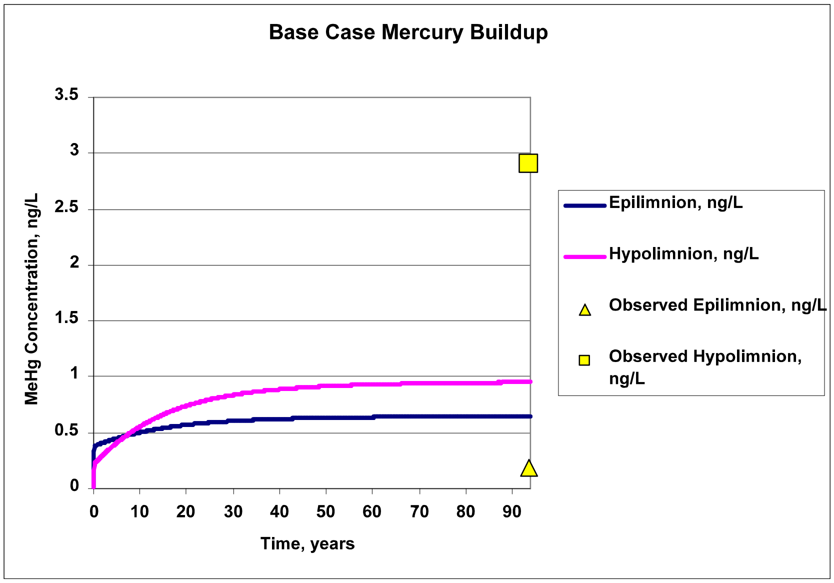

Figure A2-5. WASP Methyl Mercury Buildup in Water

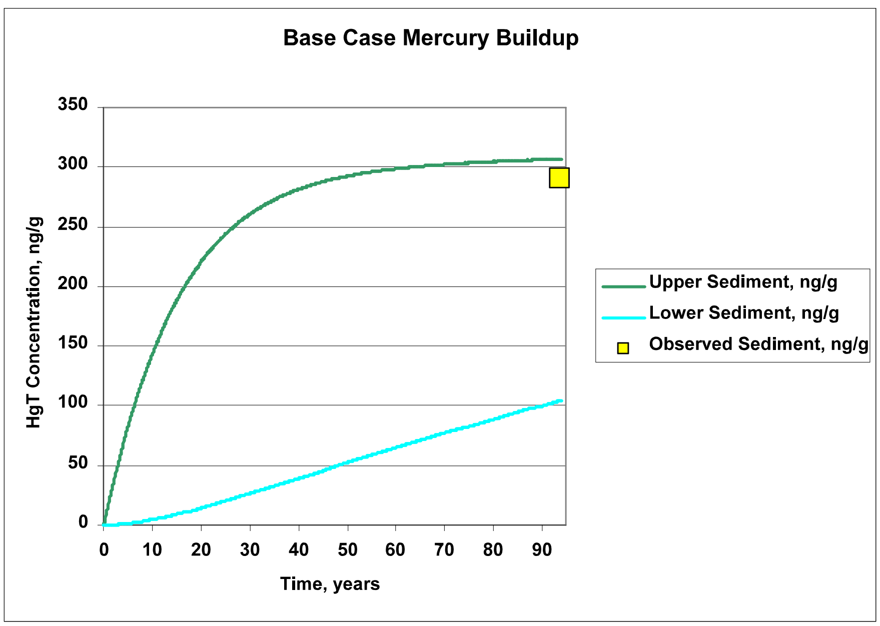

Figure A2-6. WASP Total Mercury Buildup in Sediment

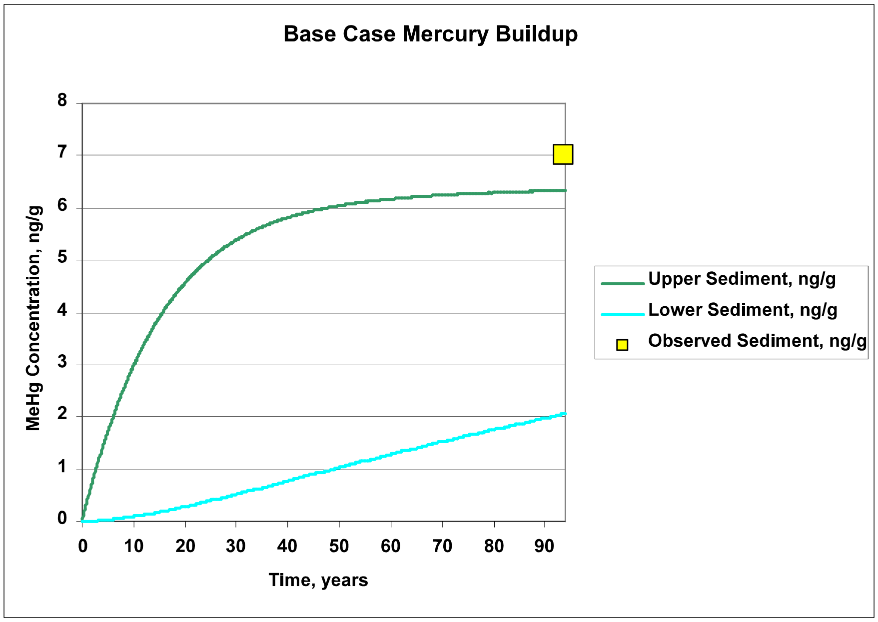

Figure A2-7. WASP Methyl Mercury Buildup in Sediment

After external mercury loads were reduced 50%, the mercury levels in the water column and surface sediment declined at moderate rates. Mercury attenuation seems to be controlled by the slow burial and by resuspension of silt and organic matter. Three scenarios were simulated representing fast, medium, and slow estimates of recovery. The upper sediment layer thickness was varied from 1 cm, 2 cm, and 3 cm. For the slow scenario, the sediment-water dispersion coefficient was reduced by half from 10^{-4} cm^2/sec. Response times are summarized in Table 7, and presented graphically for water and sediment in Figures A2-8 and A2-10.

Table A2-7. Mercury Response Times for Lake Pawtuckaway, in years

Compartment	Fast	Medium	Slow
Epilimnion	20	36	55
Hypolimnion	23	43	66
Surface Sediment	24	44	69

WASP is 2-3 times faster in recovery than SERAFM, probably due to the treatment of sediment solids balance. In particular, WASP resuspends silt and OM, which have high Kd's rather than bulk sediment (influenced by sand), which has a lower Kd.

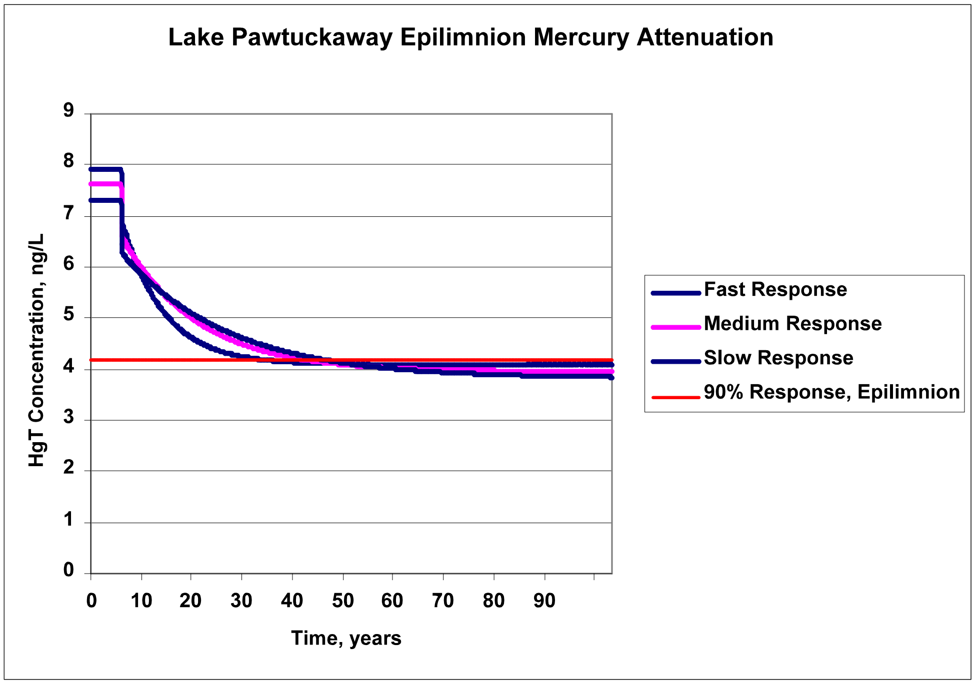

Figure A2-8. WASP Total Mercury Attenuation in Epilimnion

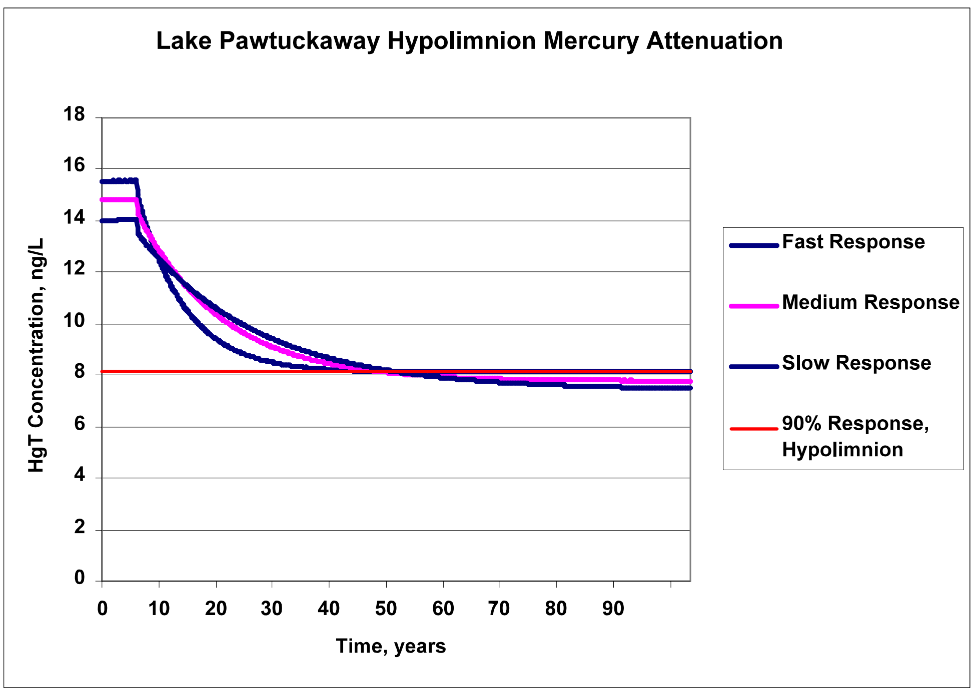

Figure A2-9. WASP Total Mercury Attenuation in Hypolimnion

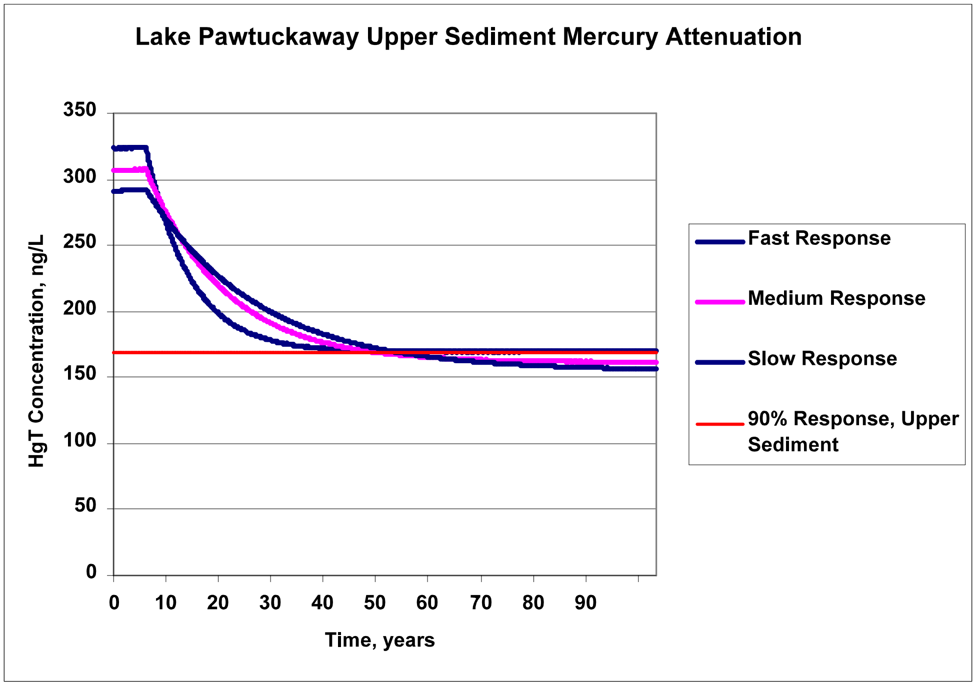

Figure A2-10. WASP Total Mercury Attenuation in Surface Sediment

A2.4 References

Kamman, N., Driscoll, C.T., Estabrook, B., Evers, D.C. and Miller, E.K., 2004. Biogeochemistry of Mercury in Vermont and New Hampshire Lakes An Assessment of Mercury in Water, Sediment and Biota of Vermont and New Hampshire Lakes Comprehensive Final Project Report May, 2004, Project Funding Provided by United States Environmental Protection Agency Office of Research and Development under the Regional Environmental Monitoring and Assessment Program.

Kamman, N.C. and Engstrom, D.R., 2002. Historical and present fluxes of mercury to Vermont and New Hampshire lakes inferred from Pb-210 dated sediment cores. Atmospheric Environment, 36: 1599-1609.

Omernik, J.M., 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). Annals of the Association ofAmerican Geographers, 77: 119-125.

US EPA, 2000. Level III ecoregions ofthe continental United States (revision of Omernik, 1987). US Environmental Protection Agency National Health and Environmental Effects Research Laboratory, Western Ecology Division, Corvallis, OR.

APPENDIX A-3

MERCURY LOAD REDUCTION ANALYSIS AND RESPONSE FOR LAKE WACCAMAW (NORTH CAROLINA)

A3.1 Introduction

Lake Waccamaw is a Large Bay lake in southeastern North Carolina. Lake Waccamaw has a water surface area of almost 35 km^2 and a catchment to lake area ratio of slightly more than six. In 1992, a survey of fish mercury concentrations North Carolina's Department of Environment, Health and Natural Resources in this region revealed that fish mercury concentrations exceeded 1 ppm in over 60% of the samples.

Waccamaw is a popular destination for recreational fishing and a fish consumption advisory is currently in place. The area surrounding Lake Waccamaw is typical of the region: flat terrain with ubiquitous wetlands and waterways. Very little commercial or industrial activity takes place in the area immediately surrounding the park, population density is relatively low and roadways are lightly traveled.

The nearest town is Whiteville, NC, located approximately 15 kilometers to the west-northwest of Lake Waccamaw. A variety of mercury sources are located in this region including at least two coal-fired electric utility boilers, a large municipal waste incinerator, several large coal or oil-fired industrial boilers, and a pulp and paper mill. By far the largest source of historic mercury emissions was the HoltraChem mercury cell chlor-alkali operation located in Riegelwood, NC, approximately 25 kilometers east-northeast of Lake Waccamaw.

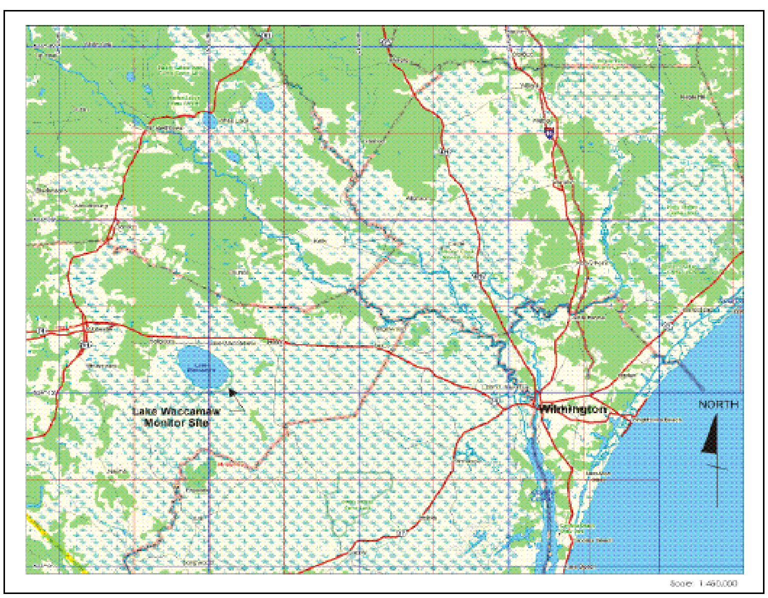

Figure A3-1. Southeastern North Carolina and Lake Waccamaw

A3.2 Empirical Data from Lake Waccamaw

Data in Tables 1 and 2 were provided by Debra A. Owen, Environmental Biologist, North Carolina Department of the Environment and Natural Resources, Department of Water Quality and Riggs et al., (Riggs et al., 2000). Fish data were normalized to two year old large mouth bass assuming a length of 10 inches (http://animaldiversity.ummz.umich.edu/site/accounts/information/Micropterus_salmoides.html)

Table A3-1. Observational Data from Lake Waccamaw

Quarter	Date	Hg, ng/L	MeHg, ng/L	Diss Hg, ng/L	Diss MeHg, ng/L	DOC, mg/L	Sulfate, mg/L
1-fall	11/20/02	4.380	0.317			31.90	13.40
2-winter	01/22/03	2.740	0.150			21.10	32.60
3-spring	05/13/03	18.400	4.040			49.80	1.37
4-summer	07/24/03	10.100	1.480	7.350	1.680	36.30	1.41
1-fall	11/20/02	5.990	0.568			28.20	9.90
2-winter	01/22/03	2.940	0.152			19.50	28.80
3-spring	05/13/03	18.100	4.990			50.40	1.99
4-summer	07/24/03	9.32	1.42	3.78	1.8	29.60	1.73
1-fall	11/20/02	1.060	0.132			8.63	5.40
2-winter	01/22/03	3.890	0.230			9.96	8.40
3-spring	05/13/03	2.680	0.233			14.00	4.60
4-summer	07/24/03	1.990	0.132	1.210	0.120	12.00	4.11

Table A3-2. Raw Fish Tissue Data Collected from Lake Waccamaw

DATE SAMPLED	LENGTH (cm)	WEIGHT (g)	Hg (µg g^{-1})	Common name	Scientific Name	Trophic Status
10/7/2003	27.6	286	0.22	BROWN BULLHEAD	ICTALURUS NEBULOSUS	Omnivore
			0.22	**BROWN BULLHEAD Average**		
10/7/2003	28.3	285	0.52	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	28.3	310	0.52	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	29.6	333	0.68	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	29.8	349	0.5	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	30.4	388	0.59	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	31.9	550	0.75	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	32.2	421	0.73	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	32.3	399	0.88	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	36.5	619	0.95	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	39.5	1055	0.87	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	27	295.3	0.77	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	29.5	428	0.84	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	40.5	1029	1	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	40.5	1229	1.1	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	43.5	1419	0.82	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
10/7/2003	52	2685	2	LARGEMOUTH BASS	MICROPTERUS SALMOIDES	Piscivore
	34.4875	737.1438	0.845	**LARGEMOUTH BASS Average**		
10/7/2003	23.2	272.5	0.11	REDEAR SUNFISH	LEPOMIS MICROLOPHUS	Insectivore
10/7/2003	25.5	385	0.2	REDEAR SUNFISH	LEPOMIS MICROLOPHUS	Insectivore
10/7/2003	28.5	421	0.48	REDEAR SUNFISH	LEPOMIS MICROLOPHUS	Insectivore
10/7/2003	24.3	376	0.31	REDEAR SUNFISH	LEPOMIS MICROLOPHUS	Insectivore
10/7/2003	25.4	453	0.29	REDEAR SUNFISH	LEPOMIS MICROLOPHUS	Insectivore
			0.278	**REDEAR SUNFISH Average**		

DATE SAMPLED	LENGTH (cm)	WEIGHT (g)	Hg ($\mu g\ g^{-1}$)	Common name	Scientific Name	Trophic Status
10/7/2003	43.5	1189	0.12	SPOTTED SUCKER	MINYTREMA MELANOPS	Insectivore
10/7/2003	38.7	752	0.18	SPOTTED SUCKER	MINYTREMA MELANOPS	Insectivore
10/7/2003	41	925	0.41	SPOTTED SUCKER	MINYTREMA MELANOPS	Insectivore
10/7/2003	43.5	1072	0.4	SPOTTED SUCKER	MINYTREMA MELANOPS	Insectivore
10/7/2003	44	1339	0.24	SPOTTED SUCKER	MINYTREMA MELANOPS	Insectivore
10/7/2003	45.2	1228	0.4	SPOTTED SUCKER	MINYTREMA MELANOPS	Insectivore
			0.291667	**SPOTTED SUCKER Average**		
10/7/2003	24	165.3	0.35	YELLOW PERCH	PERCA FLAVESCENS	Piscivore
10/7/2003	18.5	87.5	0.81	YELLOW PERCH	PERCA FLAVESCENS	Piscivore
			0.58	**YELLOW PERCH Average**		

Atmospheric Deposition Data

The Mercury Deposition Network Monitoring (MDN) Site NC08 corresponds to Lake Waccamaw. We averaged cumulative wet deposition between 1998 and 2000 to get wet deposition to Lake Waccamaw (North Carolina Division of Air Quality, 2002) and assumed that dry deposition is approximately half of total deposition.

Table A3-3. Annual Wet Deposition of Mercury at Waccamaw 1998-2000

Year	Cumulative Wet Deposition ($ug/m^2/yr$)	Precipitation (cm/yr)	Volume weighted concentration (Hg ng/L)
1998	15.8	126.6	11.6
1999	14.8	185.4	7.8
2000	12.6	133.8	9.4

A3.3 SERAFM Application: Lake Waccamaw

Table A3-4. Model Parameter Values

Parameter	Lake Waccamaw
Watershed Area	216,789,552 m^2
Percent Impervious	1%
Percent Forest	72%
Percent Riparian	0%
Percent Upland	27%
Lake Area	34,706,448 m^2
Catchment/Lake Ratio	6.25
Epilimnion Depth	2.3 m
Hydraulic Residence Time	0.66 yrs
Inflow/Outflow	1.20×10^8 m^3/yr
Parameter	Lake Waccamaw
Water pH	4.3
Epilimnion DOC	25.95 mg/L
Hypolimnion DOC	n/a
Trophic Status	Mesotrophic
Annual Precipitation	120.4 cm/yr
HgII Conc. in Precip	12.0 ng/L
Wet Deposition (HgII)	14.4 ug/m2/yr
Dry Deposition (HgII)	14.4 ug/m2/yr
Wet Deposition (HgII)	0.22 ug/m2/yr
Dry Deposition (HgII)	0.22 ug/m2/yr

Table A3-5. Measured and Baseline Steady State Values for Lake Waccamaw

	Measured		Predicted
Parameter	Range	Mean	
Water Column MeHg Unfiltered	0.132 – 4.99 ng/L	0.483 ng/L	0.17 ng/L
Water Column HgT Unfiltered	1.06 – 18.4 ng/L	4.79 ng/L	1.95 ng/L
Sediment MeHg	0. 033 – 0.2 ng/g	0.13 ng/g	0.2 ng/g
Sediment HgT	1.66 – 36.4 ng/g	22.8 ng/g	1.5 ng/g
Largemouth Bass Tissue Hg	0.5 – 2 ug/g	2 yr old lgmouth bass: 0.60 ug/g	0.21 ug/g
Observed BAF: FishHg/MeHg	1.24×10^6		

Table A3-6. SERAFM Calibrated Rate Constants for Lake Waccamaw

Process	Media	Value	Units
Methylation	Epilimnion	0	per day
	Sediment	0.07	per day
Demethylation	Epilimnion	0.0001	per day
	Sediment	0.40	per day
Biotic Reduction	Water Column	0.03	per day
Photo-Degradation	Water Column	0.002	per day per E/m^2-day
Photo-Reduction (Vis)	Water Column	0.03	per day per E/m^2-day
Photo-Reduction (UV-B)	Water Column	2.825	per day per E/m^2-day
Photo-Oxidation (UV-B)	Water Column	5.885	per day per E/m^2-day
Dark Oxidation	Water Column	1.44	per day

Table A3-7. SERAFM 50% Load Reduction Scenario for Lake Waccamaw

Compartment	Lake Waccamaw		
	Fast	Medium	Slow
Epilimnion	1	2	1
Sediment	3	6	12
Fish	1	1	2

Note: Fast = 1 cm active sediment layer, D (macro-dispersion coefficient) = 10^{-4} cm^2 s^{-1}; Medium = 2 cm active sediment layer, D = 10^{-4} cm^2 s^{-1}; Slow –3 cm sediment, D = 5x10^{-5} cm^2 s^{-}

Table A3-8. SERAFM Zero-Out Scenario for Lake Waccamaw (Removal of Deposition Attributed to Coal-fired Utilities) in the CMAQ and REMSAD Models

Compartment	Response Time (yrs)
Epilimnion HgT (ng/L)	2.8
Epilimnion MeHg (ng/L)	--
Sediment (ng/g)	5.8
Fish (ug/g)	3.8

A3.4 Lake Waccamaw WASP Model Calibration

WASP simulations of Lake Waccamaw were set up with the basic parameters from the SERAFM model. In addition to direct deposition of mercury, this water body receives watershed loadings of solids and mercury. Solids loadings from the watershed and internal production of organic matter are balanced by outflow, in-lake mineralization, and burial. Mercury loadings from direct deposition and from the watershed are subject to outflow, volatilization and burial losses. WASP simulations were run for a total of 200 years. The first 100 years represent the buildup of mercury to steady-state levels using present loadings. Mercury loadings were then cut 50%, and the attenuation period was tracked for 100 years.

WASP calibration was conducted for a base case scenario with an active sediment depth of 2 cm and a sediment macro-dispersion coefficient (E) = 10^{-4} cm^2/sec. This calibration employed the same deposition and resuspension velocities for silts and organic matter in the

SERAFM model. Solids were calibrated to match the observed porosity of 0.93, OM fraction of 35%, and burial rate of 0.05 cm/yr

The solids balance is represented in Figure A3-2- Figure A3-4. The epilimnion solids are mostly organic, including plankton, periphyton and macrophytes, as well as detritus. Sediment solids are balanced by the composition of the erosion load, the in-lake biotic production, and the mineralization of OM. The predicted sediment composition is primarily sand (90%), with a residual of silt and organic matter (10% and 5%). The burial rate is calculated internally from the solids balance. Given the specified loads, production and mineralization rates, burial rates averaged about 0.04 cm/yr.

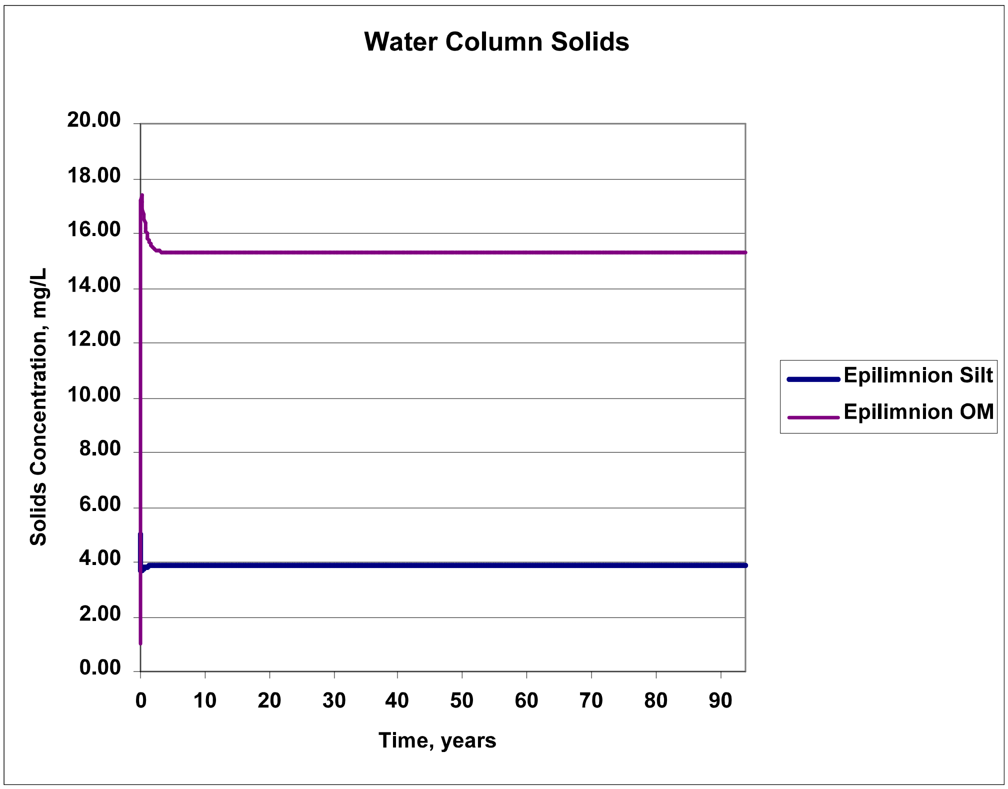

Figure A3-2. WASP Water Column Solids Calculation

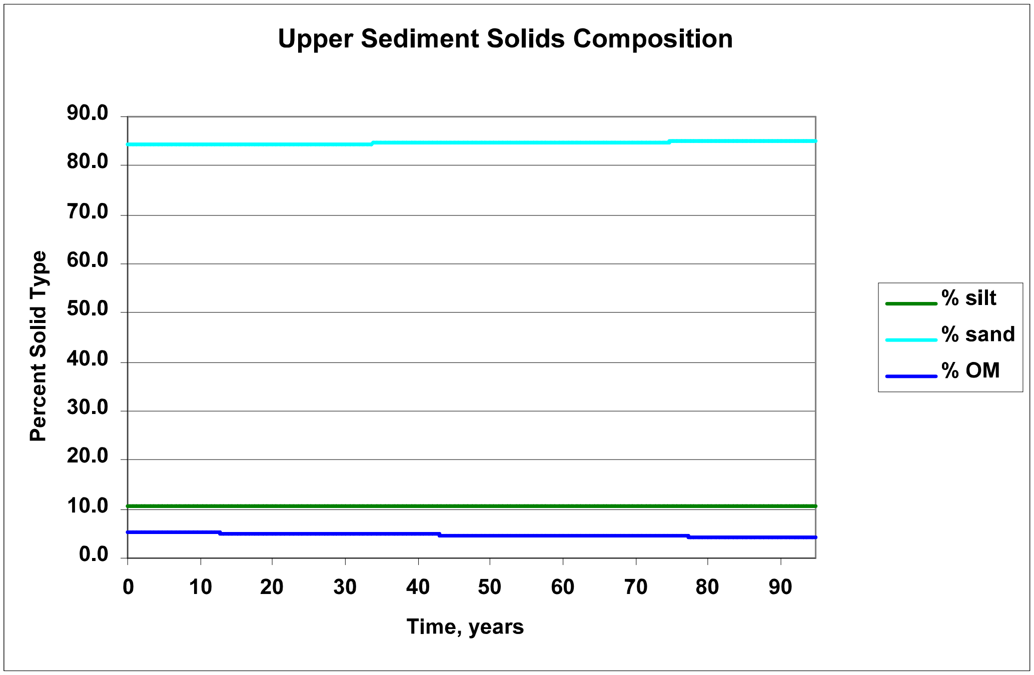

Figure A3-3. WASP Solids Simulation for Surface Sediment

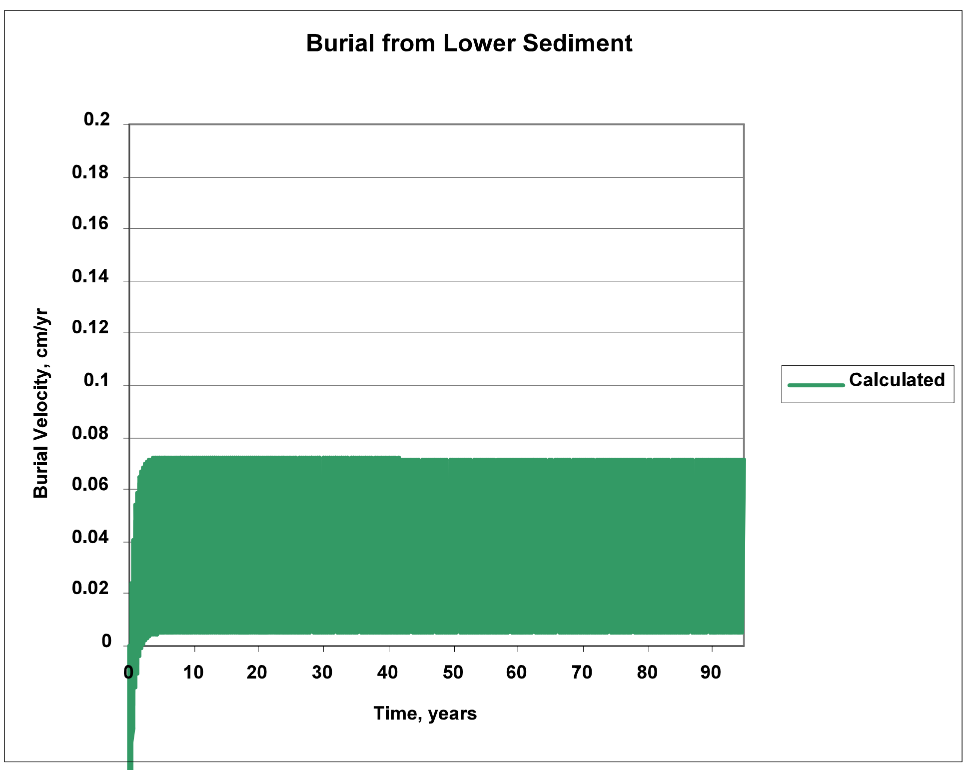

Figure A3-4. WASP Simulation of Burial Velocity

The WASP MeHg photo-degradation rate constant was adjusted to try to obtain more reasonable MeHg concentrations. Total mercury built up over the first 100 years to 8 ng/L in the epilimnion, 15 ng/L in the hypolimion, and 300 ng/g in the upper sediment. Methyl mercury levels built up to 0.5 – 1 ng/L in the water, and 6 ng/g in the sediment. Although HgT is overpredicted in the epilimnion and MeHg is underpredicted in the hypolimnion, sediment levels match the data very well.

Note that while the model calculates high OM in the hypolimnion, much of it conceptually is macrophyte biomass. The epilimnion HgT concentration is high, while the hypolimnion HgT is reasonably good. The calculated sediment HgT for the upper 2 cm and the lower 10 cm bracket the observed value.

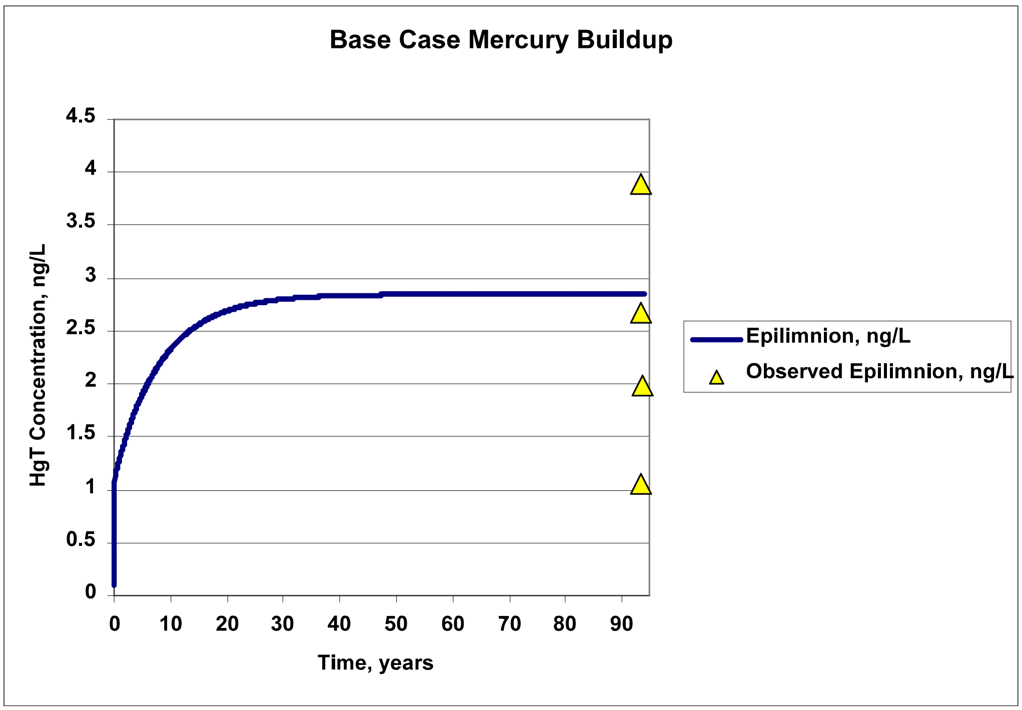

Figure A3-5. WASP Total Mercury Buildup in Water

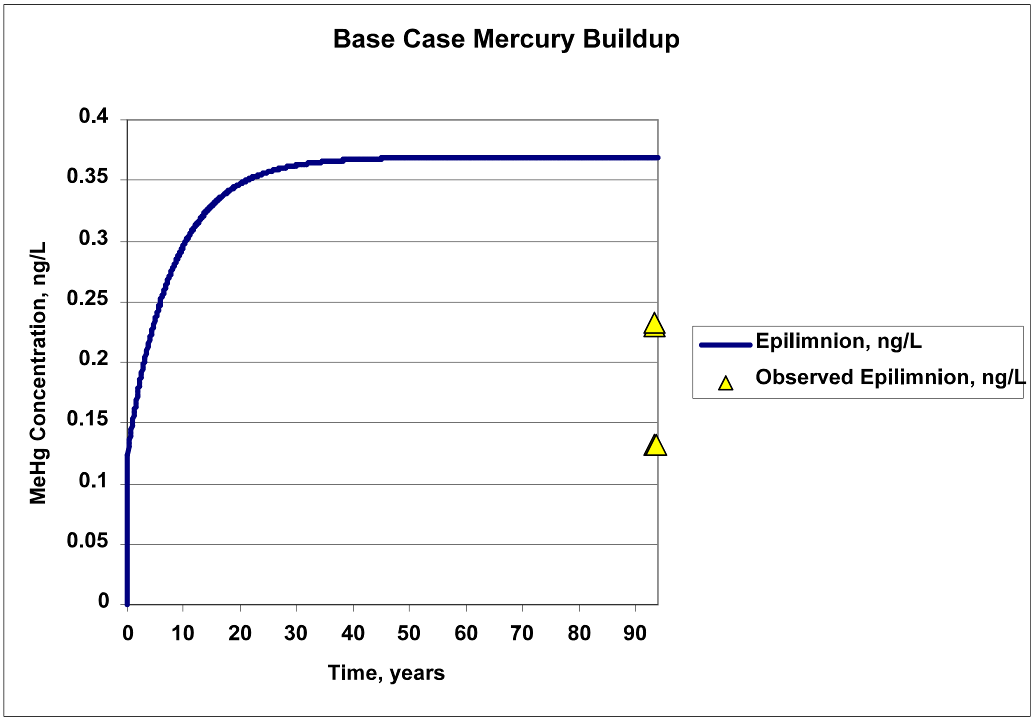

Figure A3-6. WASP Methyl Mercury Buildup in Water

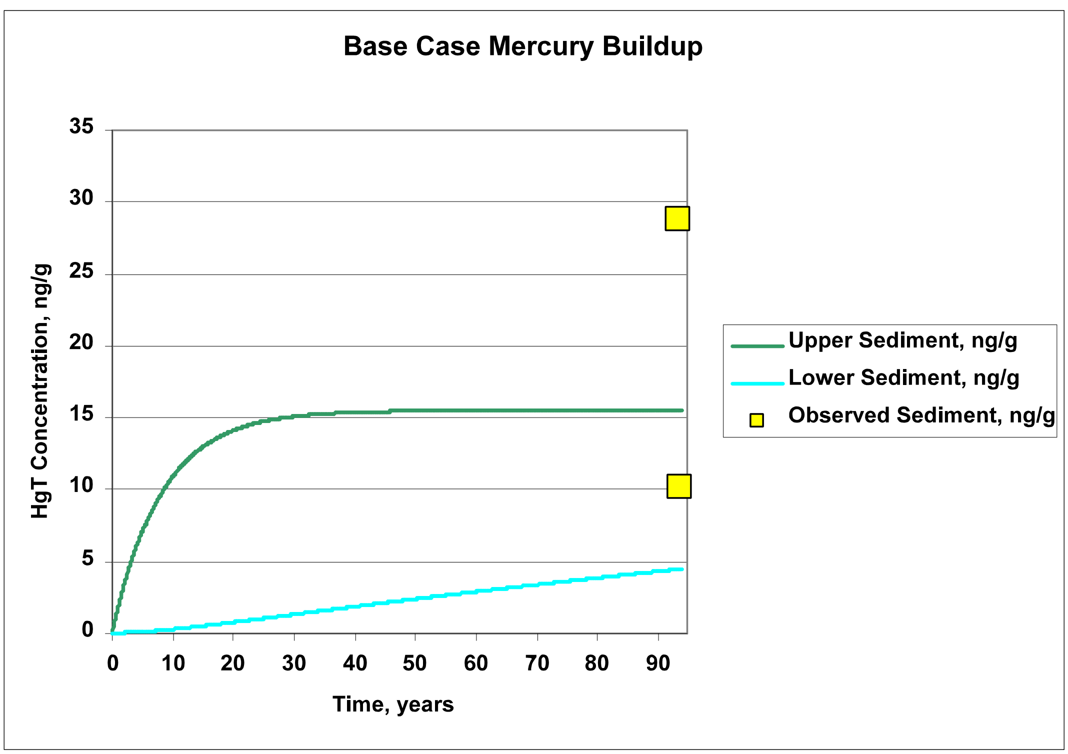

Figure A3-7. WASP Total Mercury Buildup in Sediment

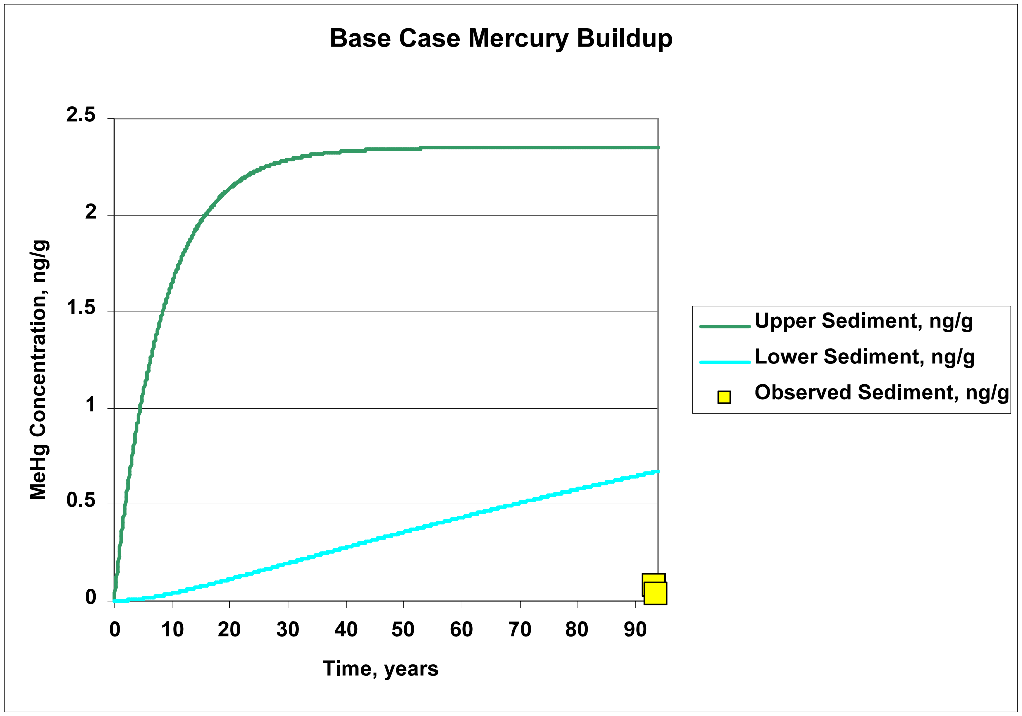

Figure A3-8. WASP Methyl Mercury Buildup in Sediment

After external mercury loads were reduced 50%, the mercury levels in the water column and surface sediment declined at relatively rapid rates. Mercury attenuation seems to be controlled by resuspension of silt and organic matter, followed by volatilization and export. Three scenarios were simulated representing fast, medium, and slow estimates of recovery. The upper sediment layer thickness was varied from 1 cm, 2 cm, and 3 cm. For the slow scenario, the sediment-water dispersion coefficient was reduced by half from 10^{-4} cm^2/sec. Response times are summarized in Table A3-9, and presented graphically for water and sediment in Figure A3-9 - Figure A3-10.

Table A3-9. WASP Response Time Estimates for Lake Waccamaw

Mercury Response Times for Waccamaw (years)			
Compartment	Fast	Medium	Slow
Epilimnion	8	11	11
Surface Sediment	10	15	17

SERAFM is 2-3 times faster in recovery than WASP, probably due to the treatment of sediment solids balance. In particular, WASP resuspends silt and OM, which have high Kd's rather than bulk sediment (influenced by sand), which has a lower Kd.

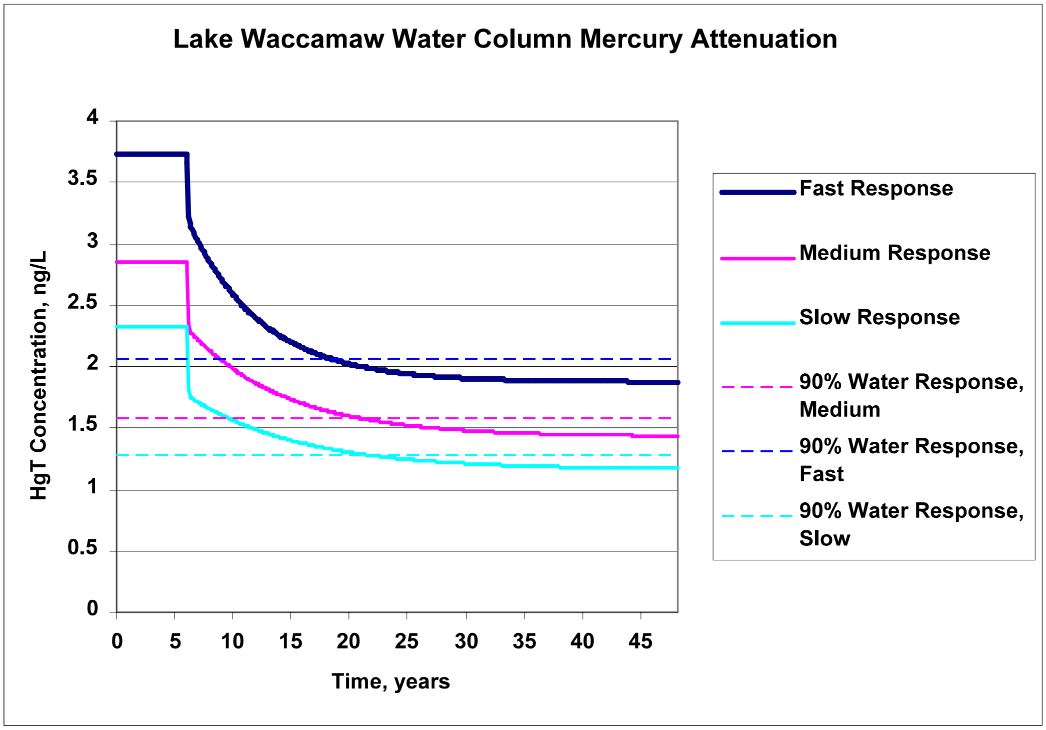

Figure A3-9. WASP Total Mercury Attenuation in Epilimnion

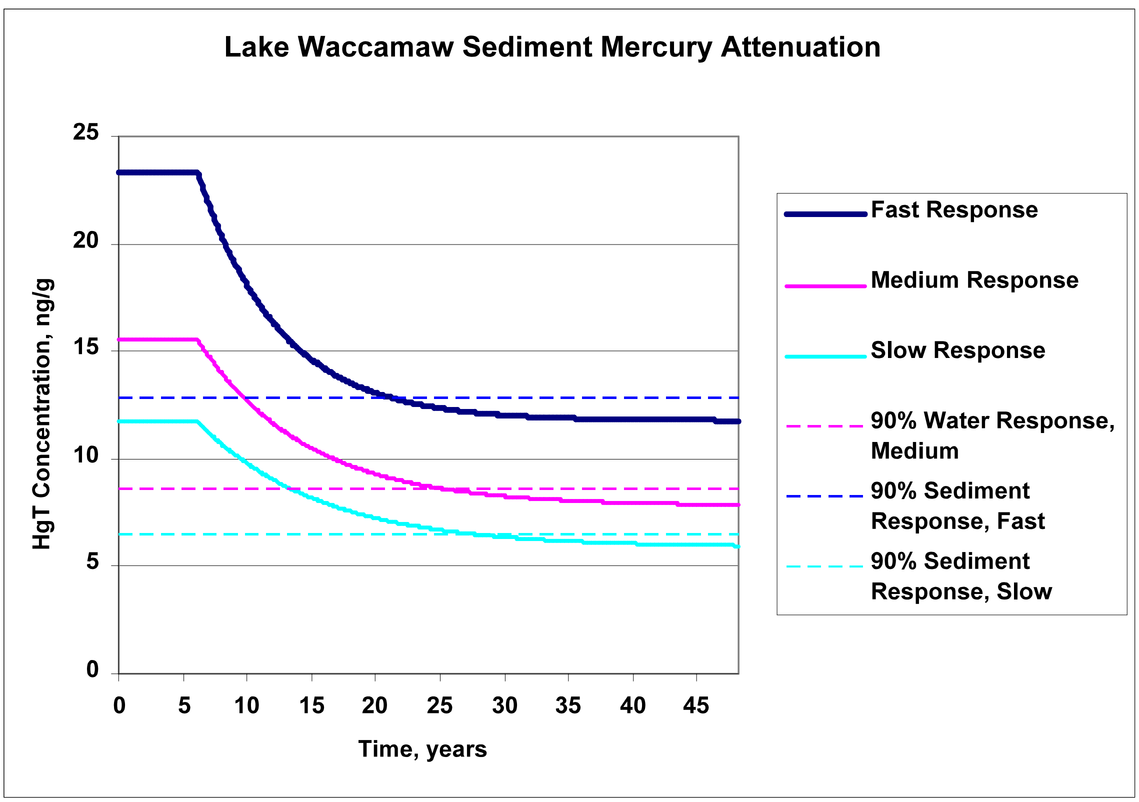

Figure A3-10. WASP Total Mercury Attenuation in Surface Sediment

A3.5 References

North Carolina Division of Air Quality, 2002. Waccamaw Atmospheric Mercury Study. Final Report to the United States Environmental Protection Agency Persistent Bioaccumulative and Toxic Chemical Program. GRANT AGREEMENT # X98493600-1

3/19/2002. http://daq.state.nc.us/toxics/studies/waccamaw/PBT_FINAL.pdf.

Riggs, S.R., Ames, D.V., Brant, D.R. and Sager, E.D., 2000. The Waccamaw Drainage System: Geology and Dynamics of a Coastal Wetland, Southeastern North Carolina, East Carolina University. Submitted to North Carolina Department of Environment and Natural Resources, Dvision of Water Resources. September 2000.

APPENDIX A-4

MERCURY LOAD REDUCTION ANALYSIS AND RESPONSE FOR THE BRIER CREEK WATERSHED (LOCATED IN THE SAVANNAH RIVER BASIN, GEORGIA)

A4.1 Background

The Brier Creek watershed is located in central/eastern portion of Georgia. The USGS Hydrologic Unit Code (HUC) for this watershed is: 03060108 (Brier). The Brier Creek watershed is presented in Figure A4-1.

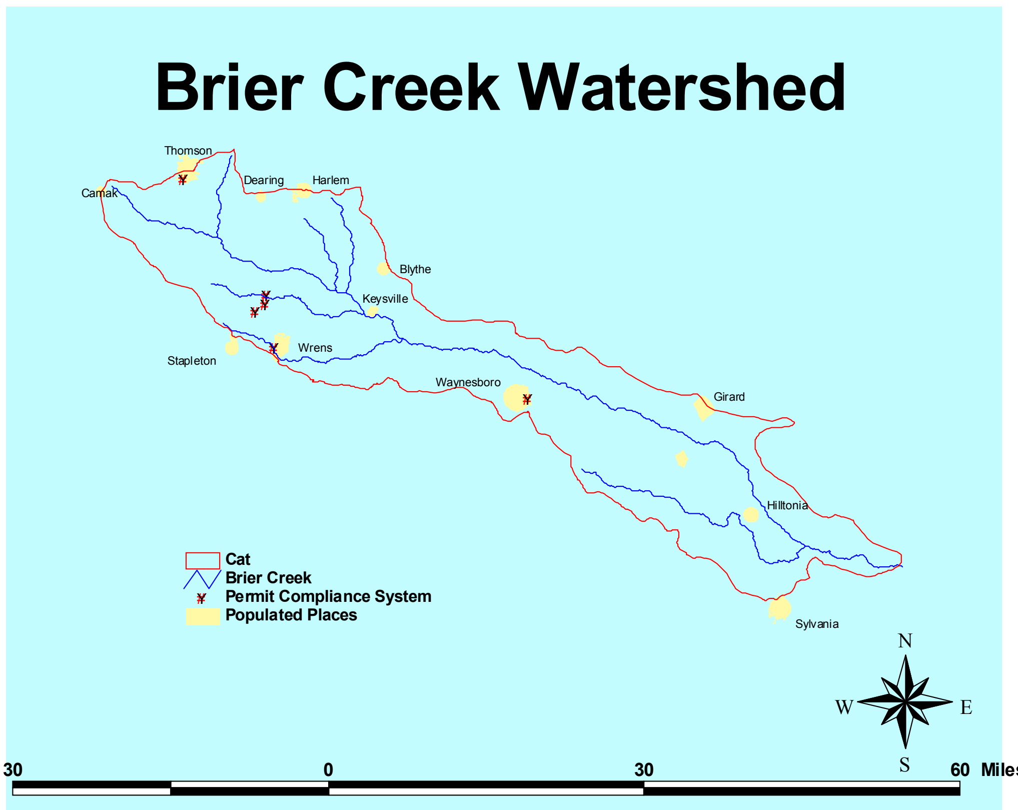

Figure A4-1. Brier Creek Watershed

The Brier Creek watershed has been divided into 11 subwatersheds for this analysis (Figure A4-3), representing all of the major tributaries to Brier Creek. A total mercury load will be determined for each of these subwatersheds to determine the impact of atmospheric deposition on the Brier Creek.

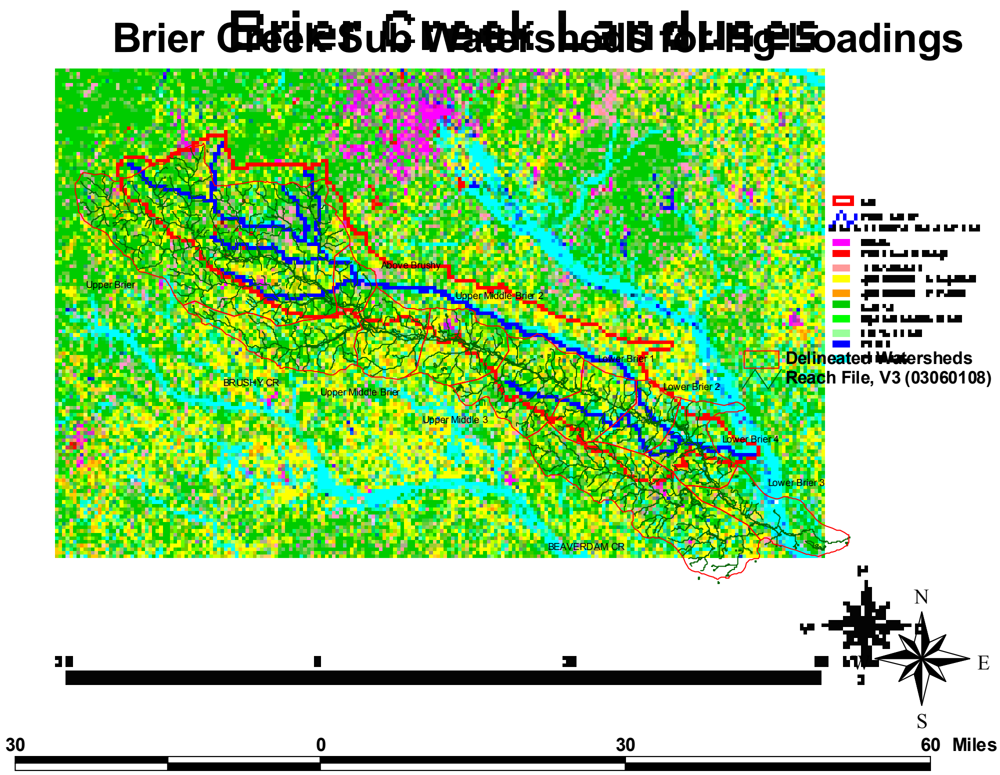

Figure A4-2. Brier Creek Subwatersheds for Hg Loadings

The watershed contains several different types of landuses. The landuses for the Brier Creek watershed are given in Figure A4-3. Different landuses collect and distribute mercury at different rates as a function of runoff and erosion.

Figure A4-3. Brier Creek Watershed Landuses

This analysis covers all waterbodies in the Brier Creek watershed. Because the spatial distribution of mercury contamination is not completely known in the streams and creeks throughout the watershed, and fish move throughout the watershed, this analysis is developed to protect all streams and creeks in the entire watershed from unacceptable accumulations of mercury in fish tissue. As discussed in previous sections of this document, the State of Georgia has issued a Fish Consumption Guideline for a segment of the Brier Creek watershed. This guideline was issued due to elevated levels of mercury found in fish flesh collected in the watershed.

A4.2 Mercury Deposition Network

The objective of the Mercury Deposition Network (MDN) is to develop a national database of weekly concentrations of total mercury in precipitation and the seasonal and annual flux of total mercury in wet deposition. The data will be used to develop information on spatial and seasonal trends in mercury deposited to surface waters, forested watersheds, and other sensitive receptors. Locations of the MDN sampling stations are shown on Figure A4-4. The sampling stations with the average mercury concentration in precipitation and mercury deposition are presented in Table A4-1.

Table A4-1. Average Mercury Deposition Hg Concentrations and Depositions Rates

Station Number	Station Name	Hg Conc in Precipitation (ng/L)	Hg Dry Deposition (ug/m^2/yr)
GA09	Okefenokee National Wildlife Refuge	14.47	13.25
GA22	Jefferson Street	15.7	13.15
GA40	Yorkville	15.37	10.49
SC03	Savannah River	13.63	12.07
SC19	Congaree Swamp	14.46	11.35
	Average	14.73	12.06

Using the MDN data, the average annual wet deposition rate was determined to be 14.7 ug/sq. meter and the dry deposition rate was determined to be 12.1 ug/sq. meter/year.

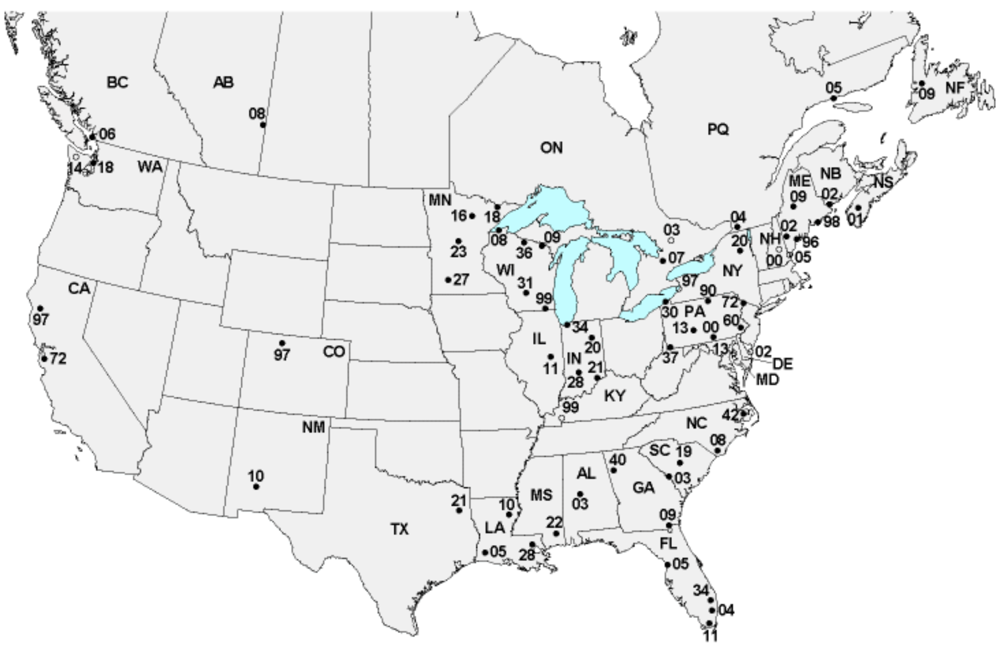

Figure A4-4. Mercury Deposition Network Sampling Locations

A4.3 Watershed Hydrologic and Sediment Loading Model

An analysis of watershed loading could be conducted at various levels of complexity, ranging from a simplistic gross estimate to a dynamic model that captures the detailed runoff from the watershed to the receiving waterbody. Because of the limited amount of data available for the Brier Creek watershed to calibrate a detailed dynamic watershed runoff model, a more simplistic approach is taken to determine the mercury contributions to the Brier Creek from the surrounding watershed and atmospheric components. Therefore, a scoping-level analysis of the watershed mercury load, based on an annual mass balance of water and sediment loading from the watershed is used for the analysis development.

Watershed-scale loading of water and sediment was simulated using the Watershed Characterization System (WCS). The complexity of this loading function model falls between that of a detailed simulation model, which attempts a mechanistic, time-dependent representation of pollutant load generation and transport, and simple export coefficient models, which do not represent temporal variability. The WCS provides a mechanistic, simplified simulation of precipitation-driven runoff and sediment delivery yet is intended to be applicable without calibration. Solids load, and runoff, can then be used to estimate pollutant delivery to the receiving waterbody from the watershed. This estimate is based on pollutant concentrations in wet and dry deposition and processed by soils in the watershed and ultimately delivered to the receiving waterbody by runoff, erosion and direct deposition.

A4.4 Water Quality Fate and Transport Model

WASP (Ambrose, et al., 1993) was chosen to simulate mercury fate in Brier Creek. WASP is a general dynamic mass balance framework for modeling contaminant fate and transport in surface waters. Based on the flexible compartment modeling approach, WASP can be applied in one, two, or three dimensions with advective and dispersive transport between discrete physical compartments, or segments. A body of water is represented in WASP as a series of discrete computational elements or segments. Environmental properties and chemical concentrations are modeled as spatially constant within segments. Each variable is advected and dispersed among water segments, and exchanged with surficial benthic segments by diffusive mixing. Sorbed or particulate fractions may settle through water column segments and deposit to or erode from surficial benthic segments. Within the bed, dissolved variables may migrate downward or upward through percolation and porewater diffusion. Sorbed variables may migrate downward or upward through net sedimentation or erosion.

Two WASP models are provided with WASP. The toxics WASP model combines a kinetic structure adapted from EXAMS2 with the WASP transport structure and simple sediment balance algorithms to predict dissolved and sorbed chemical concentrations in the bed and overlying waters. WASP simulates the transport and transformation of one to three chemicals and one to three types of particulate material. The three chemicals may be independent, such as isomers of PCB, or they may be linked with reaction yields, such as a parent compound-daughter product sequence. Each chemical exists as a neutral compound and up to four ionic species. The neutral and ionic species can exist in five phases: dissolved, sorbed to dissolved organic carbon (DOC), and sorbed to each of the up to three types of solids. Local equilibrium is assumed so that the distribution of the chemical between each of the species and phases is defined by distribution or partition coefficients. The model, then, is composed of up to six systems, three chemical and three solids, for which the general WASP mass balance equation is solved.

The WASP model was parameterized to simulate the fate and transport of mercury for the development of this analysis. Site specific and literature values were used to predict water column concentrations as a function of flow.

A4.5 Model Results

A4.5.1 Water Quality Model

The WASP toxic chemical program was set up to simulate mercury in the mainstem of the Brier Creek. The mainstem of the river was divided into 8 reaches. Each reach was further divided into 2 vertical compartments representing surface water and surficial sediment. The 2 cm deep surficial sediment layer actively exchanges silt and clay-sized solids as well as chemicals within the water column. In addition, this layer is the site for active microbial transformation reactions. Sediment-water column diffusion coefficients were set at 10^{-5} cm^2/sec.

Two solids classes were simulated: sand and silt. Sand makes up most of the benthic sediment compartments, which have a dry bulk density of 0.5 g/ml. Given a particle density of 2.7 g/ml, the sediment porosity is about 0.8 and the bulk density is 1.3 g/ml. Silt is found both suspended in the water column and in the sediment. These simulations assumed that 10 mg/L of silt enters the mainstem from the subwatersheds, settling out at an assumed velocity of 0.3 m/day. Silt in the surficial sediment compartments is assumed to resuspend at a velocity of 0.006 m/day, giving a concentration of about 0.005 g/ml, or about 1% of the surficial sediment. The exchanging silt carries sorbed mercury between the water column and surficial sediment.

Mercury was simulated as 3 components – elemental mercury, Hg^0; inorganic divalent mercury, Hg(II); and monomethylmercury, MeHg. Hg(II) and MeHg partition to solids and dissolved organic carbon (DOC). These are represented as equilibrium reactions governed by specified partition coefficients. The three mercury components are also subject to several transformation reactions, including oxidation of Hg^0 in the water column, reduction and methylation of Hg(II) in the water column and sediment layer, and demethylation of MeHg in the water column and sediment layer. These are represented as first-order reactions governed by specified rate constants. Reduction and demethylation are driven by sunlight, and the specified surface rate constants are averaged through the water column assuming a constant light extinction coefficient (here, 0.5 m^{-1}). In addition to these transformations, Hg^0 is subject to volatile loss from the water column. This reaction is governed by a transfer rate calculated from velocity and depth, and by Henry's Law constant, which was set to 7.1×10^{-3} L-atm/mole-K. Under average flow conditions, velocity ranges from 0.2 to 0.3 m/sec, while depth ranges from 0.37 to 0.69 m. The specified and calculated reaction coefficients used here are summarized in Table A4-2.

Table A4-2. Specified and Calculated Reaction Rates and Coefficients

Component	Reaction	Compartment	Coefficient Value
Hg^0	Volatilization	Water	0.3 - 3.0 day^{-1} (calc)
	Oxidation	Water	0.001 day^{-1}
Hg(II)	Reduction	Water surface Water column	0.10 day^{-1} 0.03-0.05 (calc)
	Methylation	Water	0.001 day^{-1}
	Methylation	Sediment	0.0005 day^{-1}
	Partitioning to silt	Water, Sediment	1×10^5 L/kg
	Partitioning to sand	Water, Sediment	1×10^3 L/kg
	Partitioning to DOC	Water, Sediment	1×10^5 L/kg

MeHg	Demethylation to Hg(II)	Sediment	0.005 day^{-1}
	Demethylation to Hg^0	Water surface Water column	0.05 day^{-1} 0.015 – 0.025
	Partitioning to silt	Water, Sediment	1×10^5 L/kg
	Partitioning to sand	Water, Sediment	1×10^2 L/kg
	Partitioning to DOC	Water, Sediment	2×10^5 L/kg

The Brier Creek simulation was conducted using annual average flow and load. The average flow simulation was run for 30 years, so that steady-state conditions are achieved in the water and surficial sediment. The flows, depths, length, widths, and volumes used for annual average conditions are summarized in Table A4-3.

Table A4-3. Flows, Depths, Length and Volumes used in WASP Model

Segment	Length (m)	Width (m)	Depth (m)	Flow (cfs)	Volume (cubic meters)
Upper Brier/Bushy	9494.0	64.0	1.2	360.0	729139.2
Upper Middle	13804.7	71.2	1.3	439.7	1295173
Upper Middle 2	13804.7	78.3	1.4	519.3	1553562
Upper Middle 3	13804.7	85.5	1.6	599.0	1835365
Lower 1	13804.7	92.7	1.7	678.7	2140582
Lower 2	13804.7	99.8	1.8	758.3	2469214
Lower 4	13804.7	107.0	1.9	838.0	2821260
Beaverdam	18180.0	107.0	1.9	838.0	3715447

The Watershed Characterization System calculates mercury loadings to each reach. These values are specified as constant Hg(II) and MeHg loadings for each surface water compartment. Loadings for average flow conditions reflect both wet and dry deposition throughout the watershed, followed by runoff and erosion to the tributary stream network.

Table A4-4 compares the measured sediments characteristics in Brier Creek with the predicted concentrations and conditions from WASP.

Table A4-4. Measured vs. Predicted for Sediment Components

River Station	Wasp Segment	Measured TSS, mg/L	Calculated TSS, mg/L	Measured VolS fraction	Calculated OM fraction*
BC01 water	1	16	15		
BC02 water	8	4	8		
BC01 sediment	9			0.09	0.07
BC02 sediment	16			0.02	0.04

Table A4-5 provides the predicted water column concentrations under annual average load and flow for the Brier Creek. The highest predicted water column concentration is used in the analysis calculation to determine the maximum annual average load that could occur and still achieve the target.

Table A4-5. Predicted and Observed Mercury Concentrations under Annual Average Load and Flow

Component	WASP Reach							
	1	2	3	4	5	6	7	8
HgT, ng/L	7.75	7.67	7.51	7.49	7.38	7.30	7.21	6.92
MeHg, ng/L	0.74	0.84	0.91	1.03	1.08	1.12	1.17	1.20
HgT, ng/g	35.5	32.9	31.3	28.3	27.0	24.0	23.0	20.0
MeHg, ng/g	3.4	3.3	3.2	3.1	3.0	2.7	2.7	2.4

A4.6 Brier Creek Watershed Results

Table A4-6 provides measured soil mercury concentrations for both the Brier Creek watershed and the larger Savannah River watershed which virtually surrounds Brier Creek.

Table A4-6. Soil Mercury Data in Local Region

Basin	Station	%VS	%Moisture	THg, ug/kg	MeHg ug/kg	% MeHg
Brier Creek	BC01	27.0	42	130.0	0.740	0.57
	BC02	11.0	20	75.0	0.110	0.15
Ogeechee	OG1	4.3	8.2	26.0	0.028	0.11
	OG2	1.8	3.6	13.0	0.035	0.27
	OG3	11.0	14	30.0	0.019	0.06
	OG4	15.0	47	47.0	0.940	2.00
Canoochee	CAN01	6.2	25	32.0	1.800	5.63
	CAN02	14.0	23	71.0	0.010	0.01
Savannah	Below Horse Cr.	20.0	32.7	133.0	0.054	0.04
	Below Horse Cr.	16.0	32.4	41.2	0.065	0.16
	Clarks Hill		29.8	67.2	2.050	3.05
	Below Clarks Hill		24.3	78.6	0.042	0.05
	Below Clarks Hill		24.3	80.8		0.00
	Below Butler Creek	10.0	20.9	33.1	0.031	0.09
	Below Upper Three Runs Creek	3.9	17.5	22.7	0.052	0.23
	Below Lower Three Runs Creek	2.8	17.1	56.8	0.003	0.00
	Below Brier Creek	5.2	9.2	43.6	0.257	0.59
	Clyo, USGS Gage	5.2	21.9	71.8	0.949	1.32
	Below Ebenezer Creek	6.9	5.5	33.9	0.011	0.03
	Butler Creek	7.1	3	43.8	0.063	0.14
	Horse Cr.		17.4	43.6	0.009	0.02
	Upper Three Runs Creek	7.8	17.5	56.4	0.013	0.02
	Lower Three Runs Creek	16.0	33.9	137.7	0.543	0.39
	Brier Creek	4.4	16	26.3	0.319	1.21
	Ebenezer Creek	3.9	8.3	28.1	0.109	0.39
Mean			20.6	56.9	0.344	0.66
Standard Deviation			11.4	34.6	0.569	1.27

A4.6.1 Brier Creek Soil Mercury Calibration

The WCS Mercury model was applied and run for 100 years to equilibrate the watershed soils with atmospheric conditions. Figure A4-5 illustrates the buildup of mercury in the soils over the 100 year period.

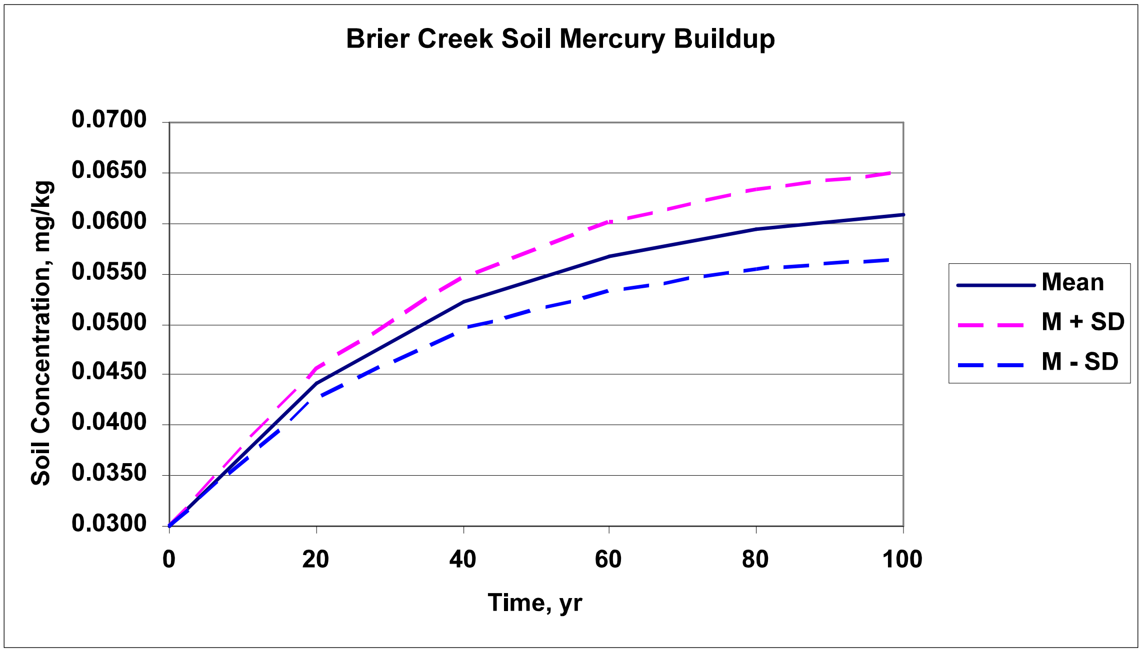

Figure A4-5. Brier Creek Soil Mercury Buildup

- Soil Partition Coefficient = 5000 L/kg
- Soil Reduction Rate Constant = 0.0005 day^{-1}
- Calibrated mean soil mercury concentration of 61 ng/g is close to the area mean of 57 ng/g. Modeled variability in Hg (standard deviation of 4 ng/g) is significantly less than the measured variability in the area (SD of 35 ng/g). The simulations used constant atmospheric wet and dry deposition fluxes, which suppressed the spatial variability.

A4.6.2 Mercury Loading Fluxes

Figure A4-6 illustrates the mercury loadings from the delineated subwatersheds as predicted by the WCS over the 100 year equilibration period.

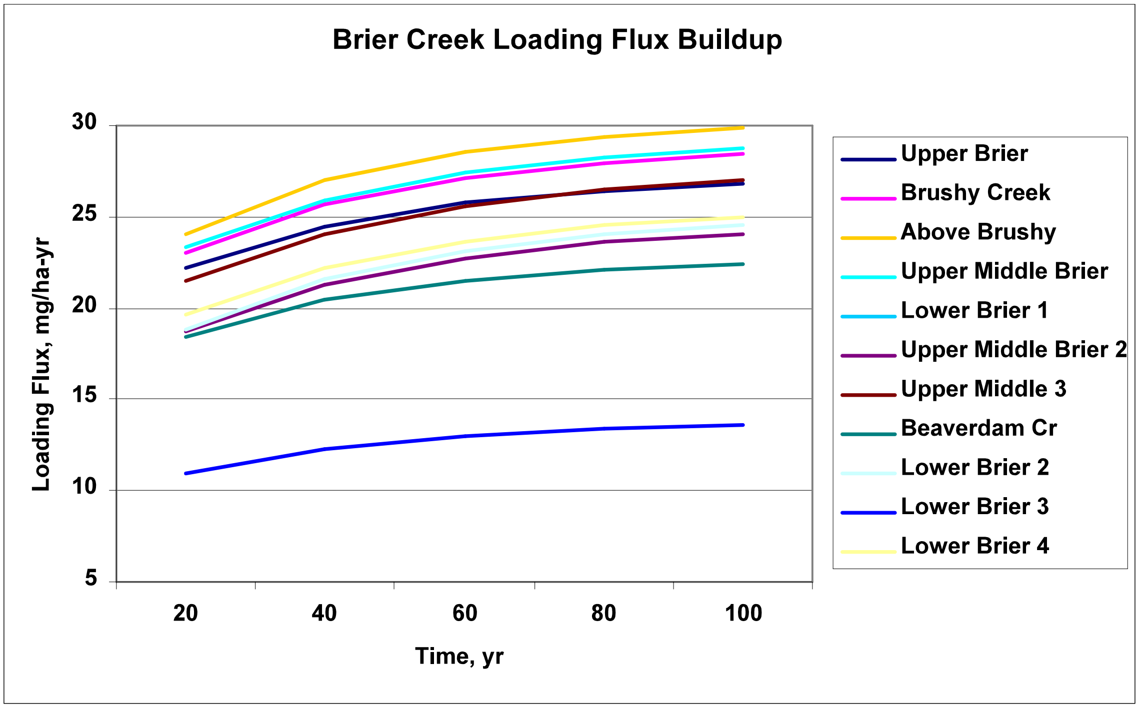

Figure A4-6. Brier Creek Loading Flux Buildup

There is significant variability in loading fluxes from the 11 subwatersheds in the Brier Creek watershed. Loads at year 100 were used in the water body calibration.

A4.6.3 Future Projections

For these projections, the atmospheric loads were cut in half, and the watershed response was followed for 100 years in the Upper Brier Creek subwatershed (Figure A4-7).

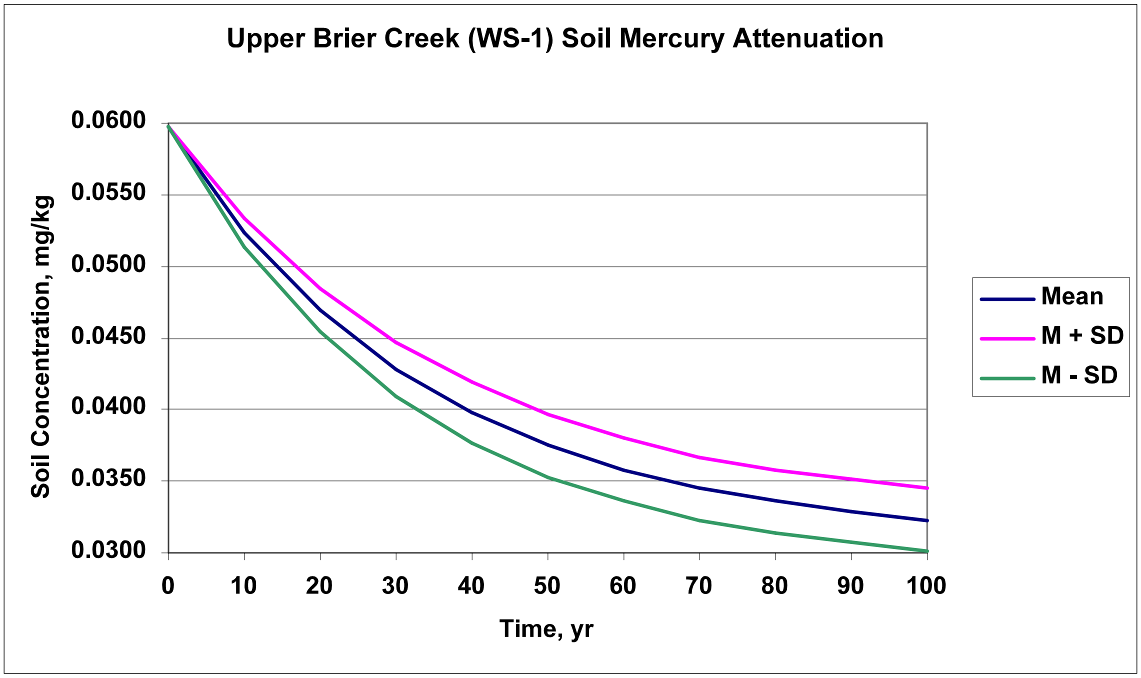

Figure A4-7. Upper Brier Creek Soil Mercury Attenuation

The half life of the soil attenuation response is about 25 years.

The projected loading flux for Upper Brier Creek was projected over 100 years. The loading fluxes for the other 10 subwatersheds were calculated over 50 years (Figure A4-8).

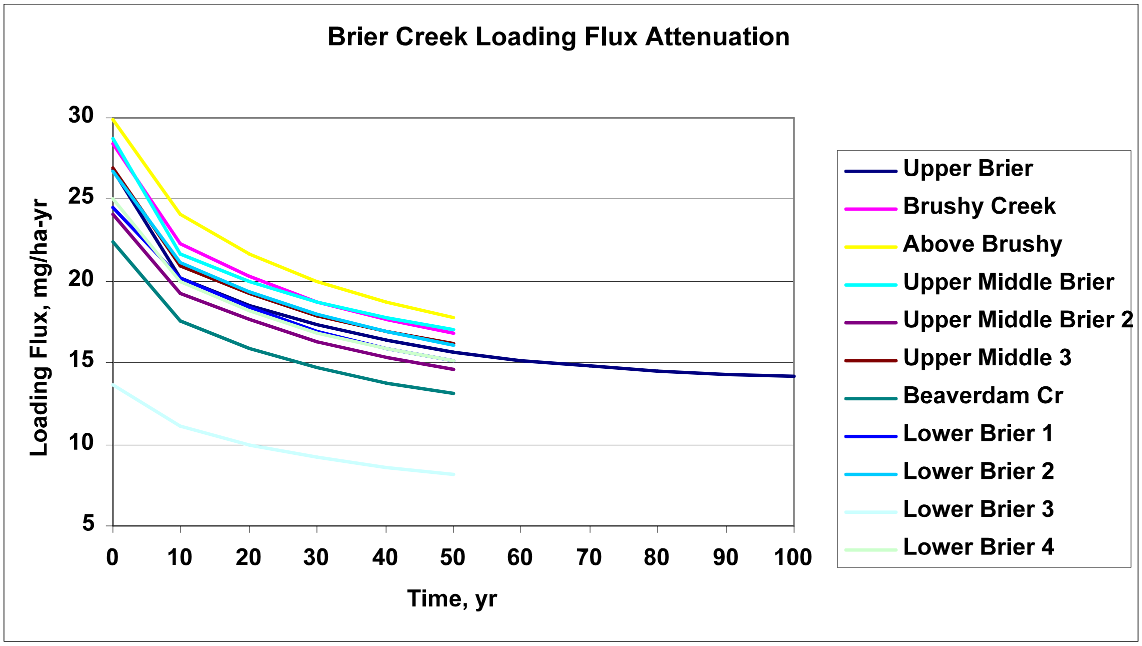

Figure A4-8. Brier Creek Loading Flux Attenuation

A4.6.4 Sensitivity of Temporal Response

The depth of soil incorporation significantly influences soil response time. The default of 1 cm was varied plus and minus 50% to get a range of response times. This parameter should not affect concentrations or loadings at steady-state. Figure A4-7 illustrates the response time of the upper soil layer to decreases in mercury loadings from the atmosphere.

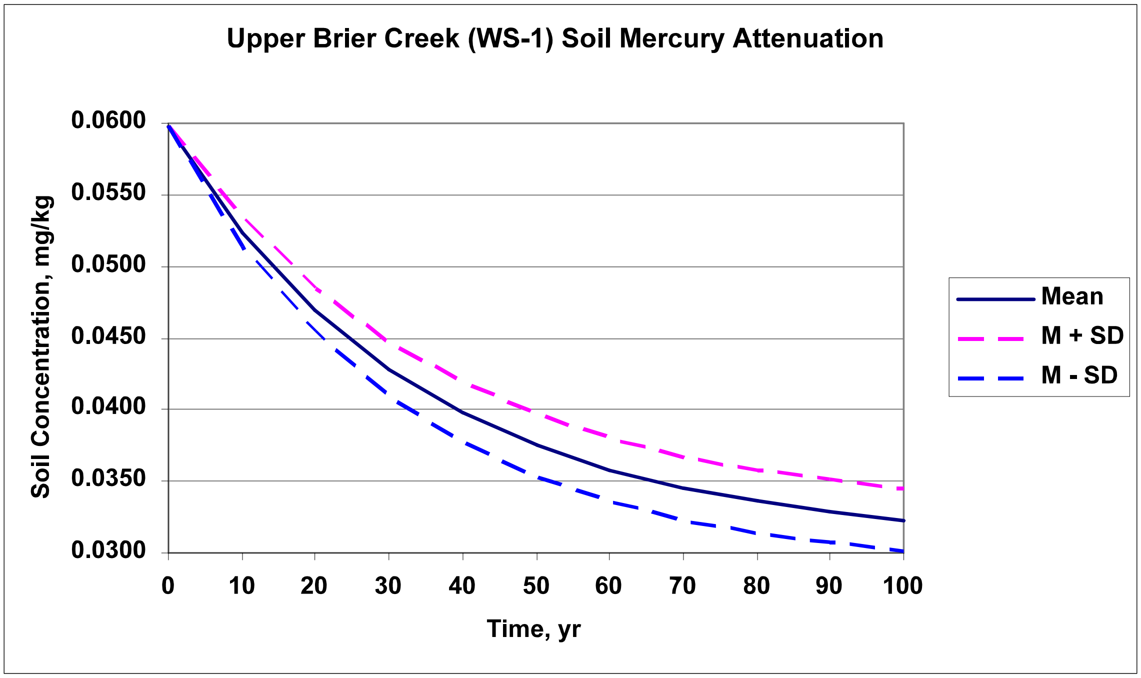

Figure A4-9. Upper Brier Creek Soil Mercury Attenuation

The half life of the soil mercury attenuation response for the base simulation in Upper Brier Creek was 25 years, and the 90% response was 90 years. Varying the incorporation depth 50% caused the half life to vary between 12 and 38 years. The 90% response time varied even more, between 45 and about 110 years.

The loading responses for Upper Brier Creek are given in Figure A4-10. An initial rapid drop-off in loading (due to instantaneous drop in deposition to water surfaces and impervious runoff) is followed by a slower drop-off in runoff and erosion fluxes, controlled by soil mercury concentrations. The 50% loading response varied between 8, 10, and 15 years for incorporation depths of 0.5, 1.0, and 1.5 cm. The 90% loading response times were much longer, varying between 35, 70, and 100 years, respectively.

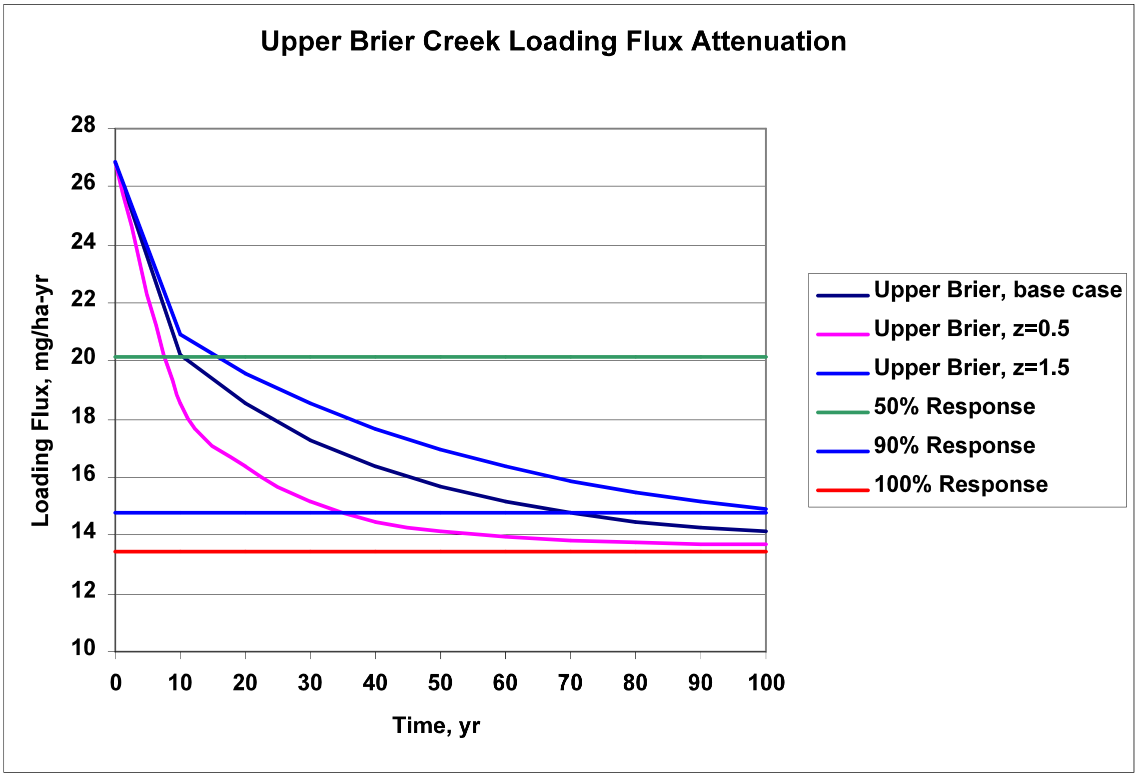

Figure A4-10. Upper Brier Creek Loading Flux Attenuation

Land use changes can also significantly affect future loading response from a watershed. In particular, changing pervious land use areas to impervious areas will directly affect the delivery of atmospheric deposition fluxes to the water body. In this model, impervious areas were assumed to deliver 100% of the deposition load, while pervious areas delivered a much smaller fraction through runoff and erosion. Although the fraction of the Brier Creek watershed covered by impervious surfaces is small (about 3% of the upper watershed), even modest growth over many years could increase the total watershed delivery of deposited mercury, working against the overall reductions in atmospheric emissions. The loading response of Upper Brier Creek to 3 land use scenarios is shown in the next figure. All scenarios assume an immediate 50% cut in atmospheric deposition. The base case assumes present land use patterns. The other two scenarios assume modest impervious surface growth rates of 0.5 % per year and 1 % per year. For the 0.5% scenario, total watershed loads reach a minimum in 80 years, with watershed loadings stalling at 40% of present levels. For the 1% scenario, total watershed loads reach a reduction level of 33% in 50 years, and then increase significantly. After 100 years, the 50% cut in atmospheric deposition would translate into a 25% drop in ambient watershed loading (Figure A4-11).

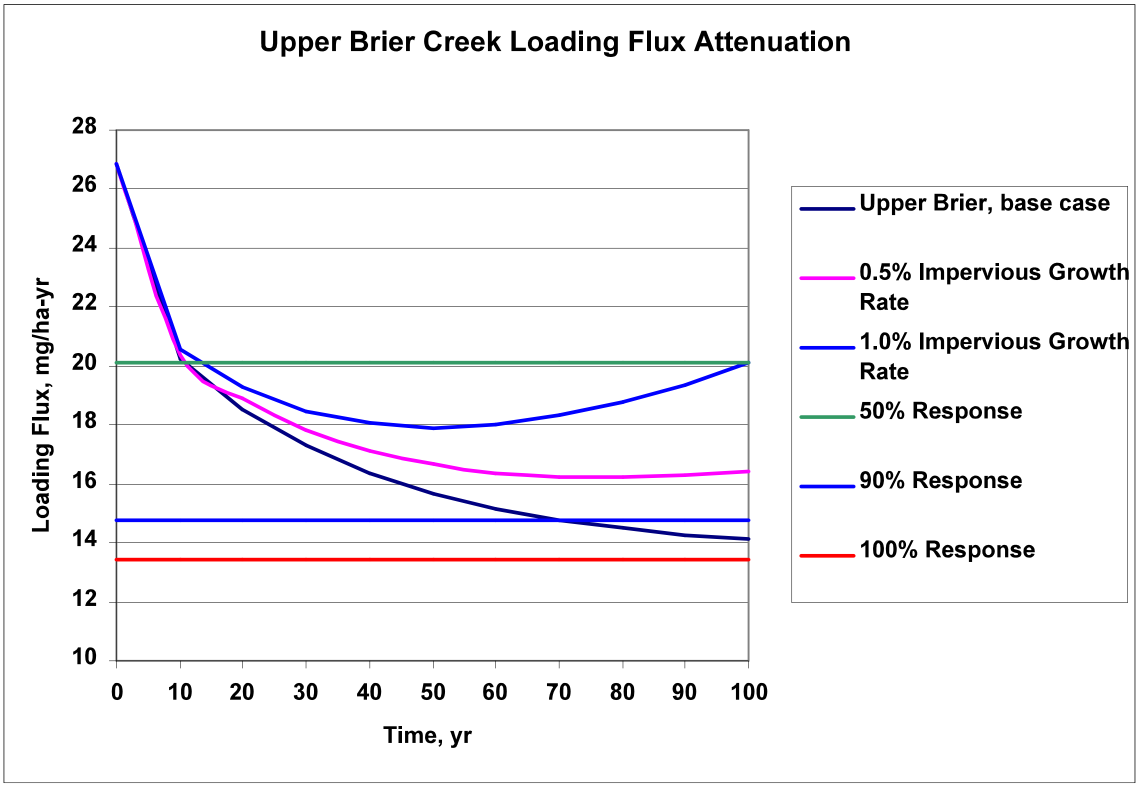

Figure A4-11. Watershed Loading Flux Attenuation considering Landuse Change

A4.7 Brier Creek Water Body Results

A4.7.1 Phase 1: Long Term Buildup

Long-term predicted watershed loadings were applied to the Brier Creek water body network to simulate the long-term buildup of mercury in the water and sediment. Average flows were used for this entire simulation. The watershed loadings from year 30 to 100 in the watershed simulation were used in the first 70 years of this water body simulation. For the last 20 years of the water body simulation, watershed loadings were held constant (Figure A4-12).

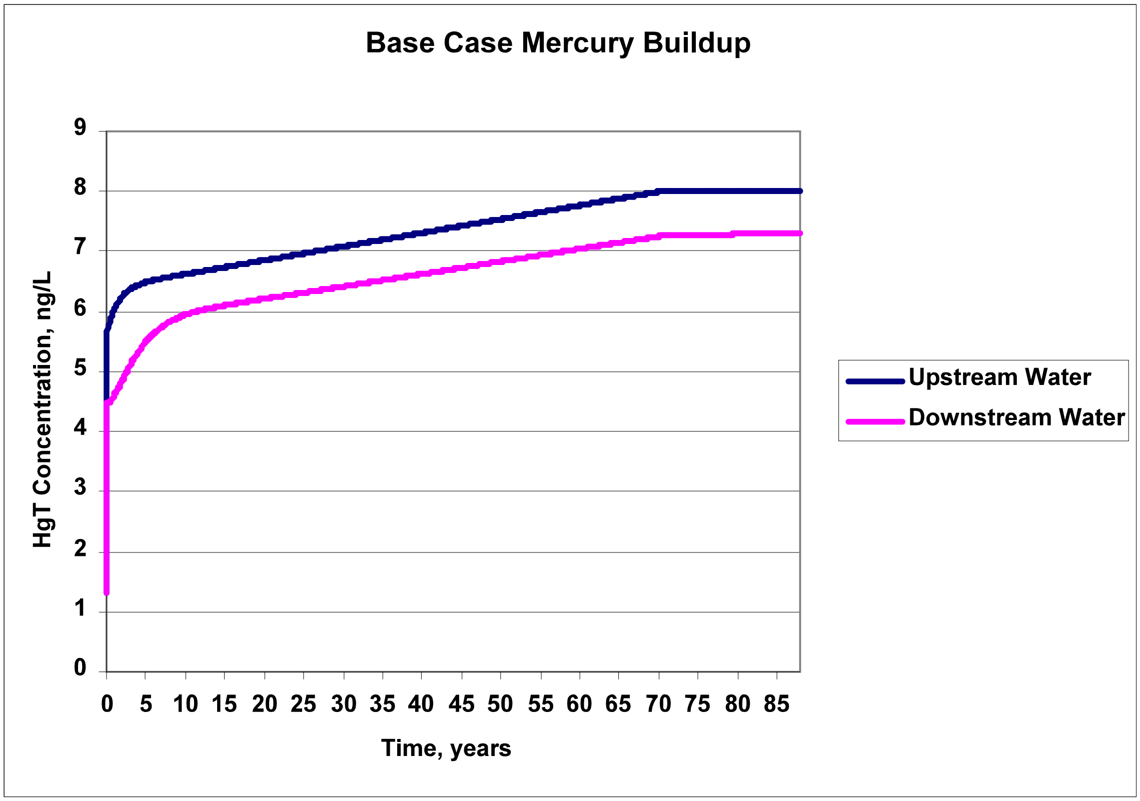

Figure A4-12 Base Case Water Column Mercury Concentration for Brier Creek

Initial low water column concentrations jumped quickly from a background of 1 ng/L to near 5 to 6 ng/L in response to the loadings. Following this initial response to the industrial era loadings, the water column mercury slowly increased in proportion to the slowly increasing watershed loadings.

Mercury concentrations in the upper sediment (2 cm layer) followed the same pattern, increasing from an initial 5 ng/g to a steady 20 – 35 ng/g. In response to slow internal mixing, mercury concentrations in the lower sediment (2 - 12 cm layer) slowly increased throughout the simulation from a background of 5 ng/g to the range of 10-20 ng/g. Lower sediment concentrations were still increasing at the end of this 85 year simulation.

The model dynamics follow the expected pattern of a water column and upper sediment responding relatively quickly to external loads (a few years), and a lower sediment layer responding slowly (decades).

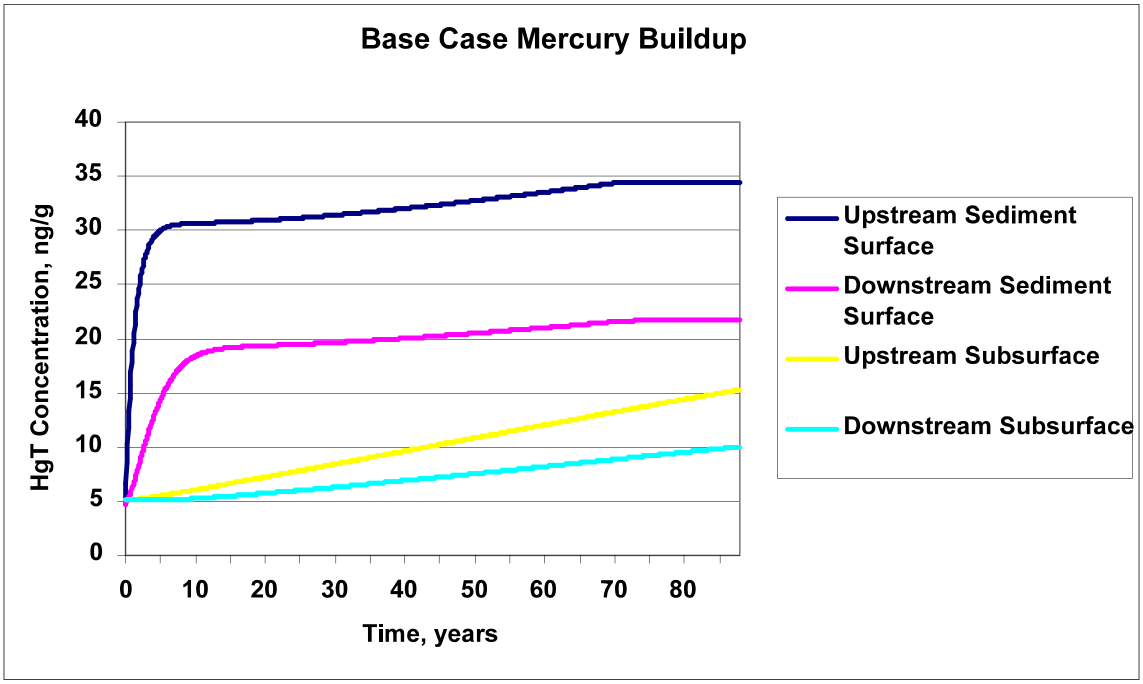

Figure A4-13. Base Case Sediment Mercury Concentration for Brier Creek

A4.7.2 Phase 2: Response to 2002 Flows

Mercury concentrations in Brier Creek were measured in a June, 2002 survey. Table A4-7 illustrates the model predictions for water column and sediment concentrations versus model predictions. Figure A4-14 - Figure A4-15 provide a graph of model predictions for both total mercury and methyl mercury over time. A more dynamic simulation was conducted using daily stream flow data for December 2001 – July 2002. Mercury loadings were assumed to be proportional to incremental inflows from the subwatersheds. In response to changing flows and depths, but constant inflow concentrations, simulated water column mercury levels fluctuated mildly. Sediment levels (not shown) changed very slowly, further buffering the water column from major fluctuations. The predicted mercury levels compared favorably with the observations.

Table A4-7. June 2003 Survey vs WASP Predictions for Mercury

River Station	Wasp Segment	Measured HgT, ng/L	Calculated HgT, ng/*	Measured MeHg, ng/*	Calculated MeHg, ng/*	Measured MeHg fraction	Calculated MeHg fraction
BC01 water	1	8.3	7.8	0.73	0.80	0.09	0.10
BC02 water	8	6.0	6.3	1.40	0.86	0.23	0.14
BC01 sediment	9+17	37.0	18.6	5.70	1.7	0.15	0.09
BC02 sediment	16+24	6.4	12.0	0.04	1.3	0.01	0.10
				* L or g			

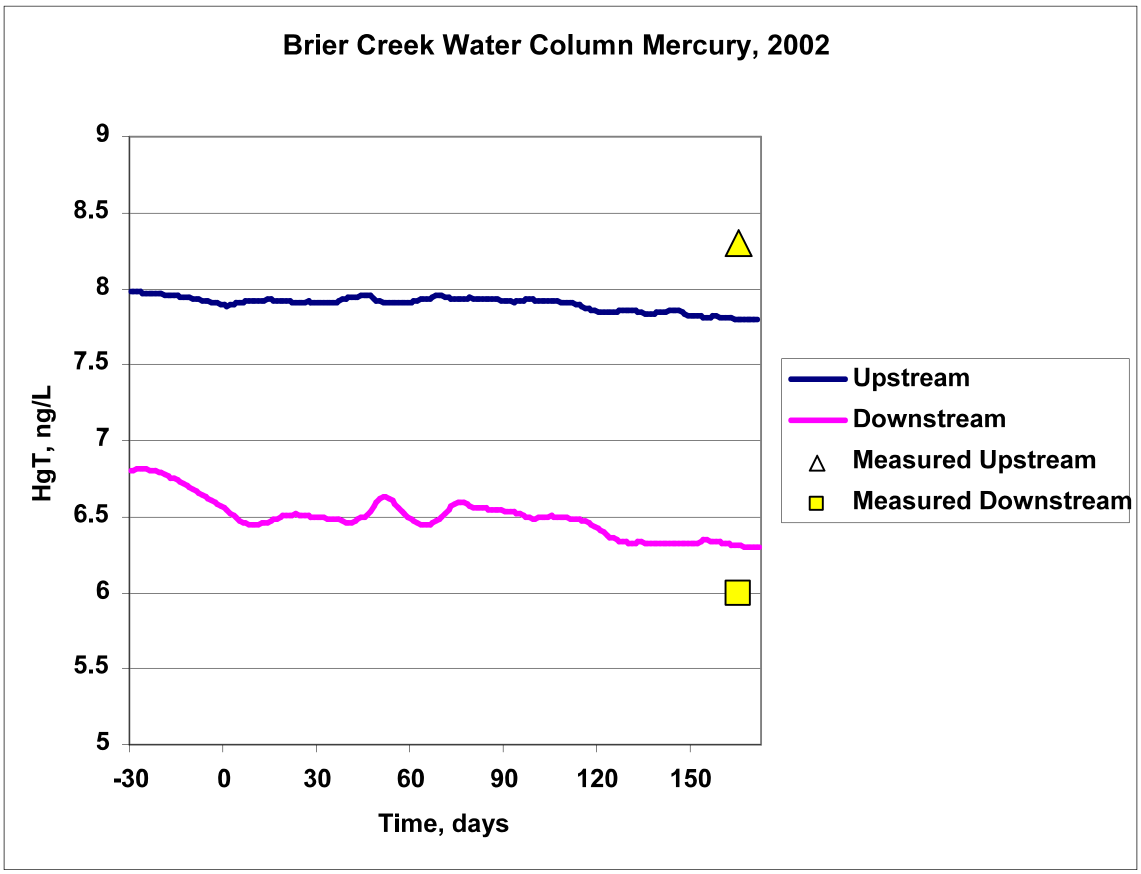

Figure A4-14. Brier Creek Total Mercury Water Column Concentration

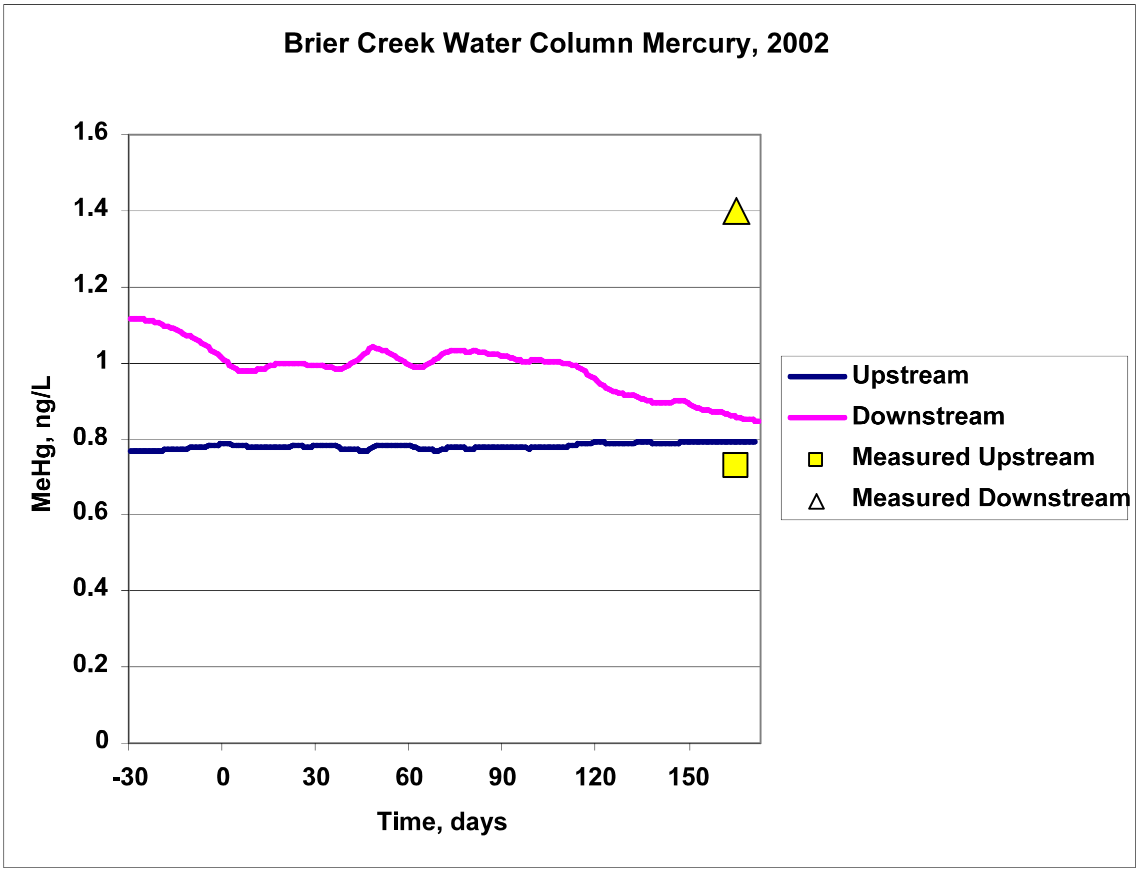

Figure A4-15. Brier Creek Methyl Mercury Water Column Concentration

A4.7.3 Phase 3: Future Attenuation

A 55 year simulation was conducted to explore mercury response to declining watershed loads due to an immediate decline in atmospheric deposition of 50% (year 5 of the water body simulation). Figure 16 illustrates the change in mercury concentration as a function of load reduction in the water column. Figure A4-17 illustrates the change in mercury concentration as a function of load reduction in the sediments.

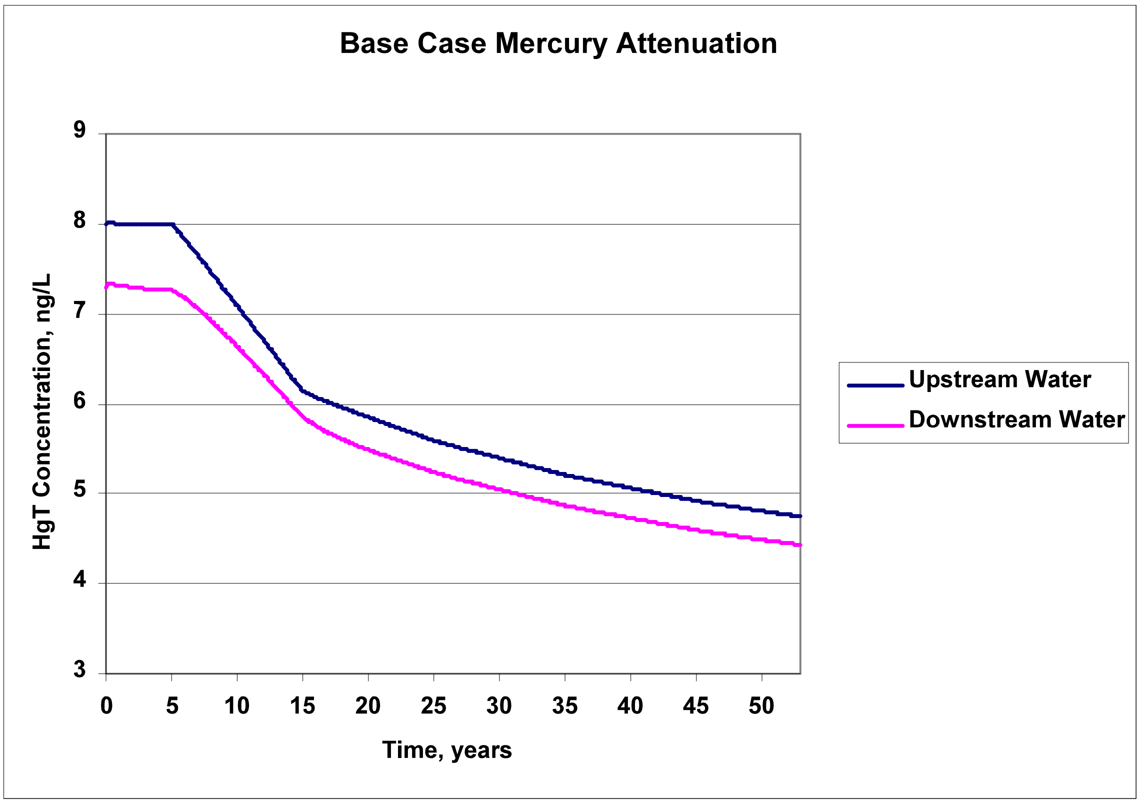

Figure A4-16. Mercury Attenuation over Time in Water Column

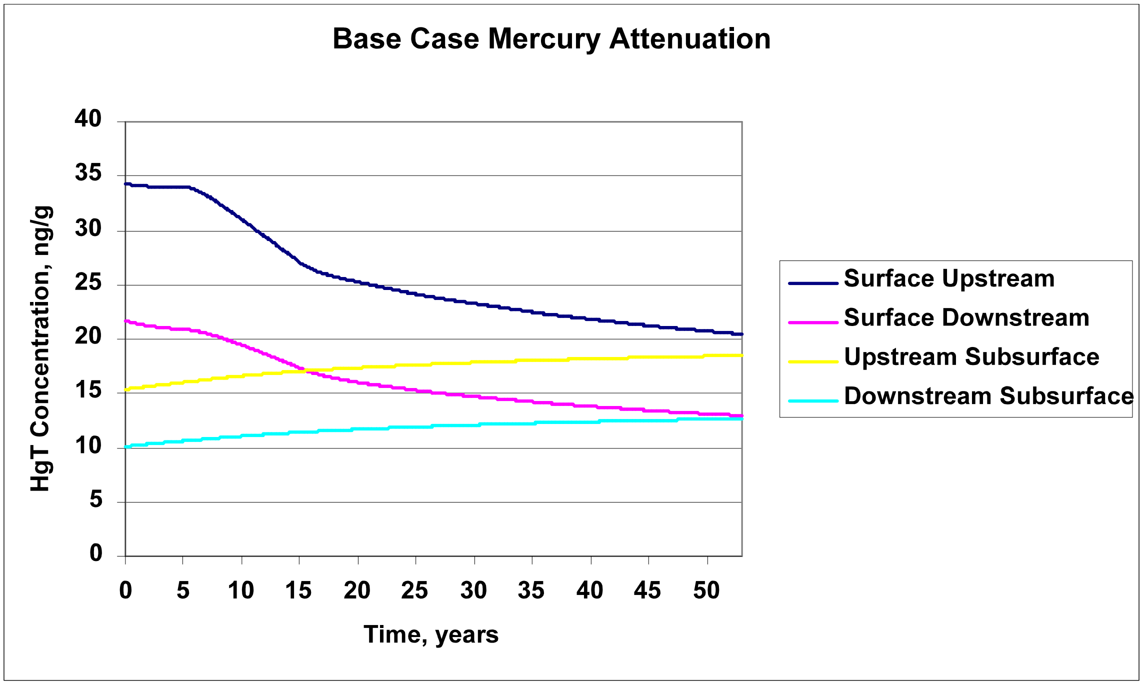

Figure A4-17. Mercury Attenuation over Time in Sediments

Water body and upper sediment concentrations were predicted to decline more rapidly in the first 10 years, and then more slowly for the next 4 decades. The initial decline is due to the rapid decline in direct loadings to the water surface and from impervious areas. The later slow decline is due to the slowly dropping soil concentrations and the internal recycling of sediment mercury. Ultimately, the predicted water body concentrations should decline proportionally to the declining loadings. The first half of this response takes about a decade, while 90% of the response should take on the order of 50 years. The remainder of the water body response should be even slower, as the lower sediment concentrations become the controlling factor.

Uncertainties in this response include watershed loadings and internal water body conditions, including infrequent occurrences of very high scouring flows, and sediment mixing characteristics.

A4.7.4 Sensitivity of Time Response

The depth of soil incorporation in the watershed significantly influences soil response time and loadings to the stream (see previous section), and thus concentrations in the stream. The default of 1 cm was varied plus and minus 50% to get a range of response times. Within the water body, the sediment incorporation depth should have a similar influence. The Brier Creek model was set up with a 2 cm active sediment layer and a 10 cm lower sediment layer. Transport between these layers was controlled by a bulk dispersion parameter, which mixes pore water, solids, and mercury. The calibrated base value of 10^{-8} cm^2/sec was varied between 10^{-6} and 10^{-9} cm^2/sec to get a range of response times. These parameters should not affect concentrations or loadings at steady-state.

These sensitivity runs demonstrate that attenuation of mercury in the Brier Creek water will be controlled more by watershed loadings than by internal mixing processes. The next figure shows the response of upper Brier Creek to the three reduction scenarios. The 50% watershed loading responses of 8, 10, and 15 years for incorporation depths of 0.5, 1.0, and 1.5 cm produced 50% stream concentration responses of 9, 12, and 18 years. The 90% loading response times were much longer. The fast, medium, and slow stream response scenarios reached the 90% mark in 40, 74, and 113 years, respectively.

The mercury concentration response in lower Brier Creek follows a more complicated time trend, influenced not only by the watershed loadings and internal mixing, but also by the upper Brier Creek dynamics. The 50% response times are 18, 15, and 22 years, for the fast, base, and slow scenarios. The downstream base and slow scenarios are 3 to 4 years longer than the upstream response. The downstream fast response scenario, however, takes 9 years longer than the upstream reach, as the enhanced mixing results in significant short term internal sediment fluxes. The 90% response times for the fast, medium, and slow response scenarios were 58, 89, and 132 years, which is 9 - 18 years longer than the upstream response.

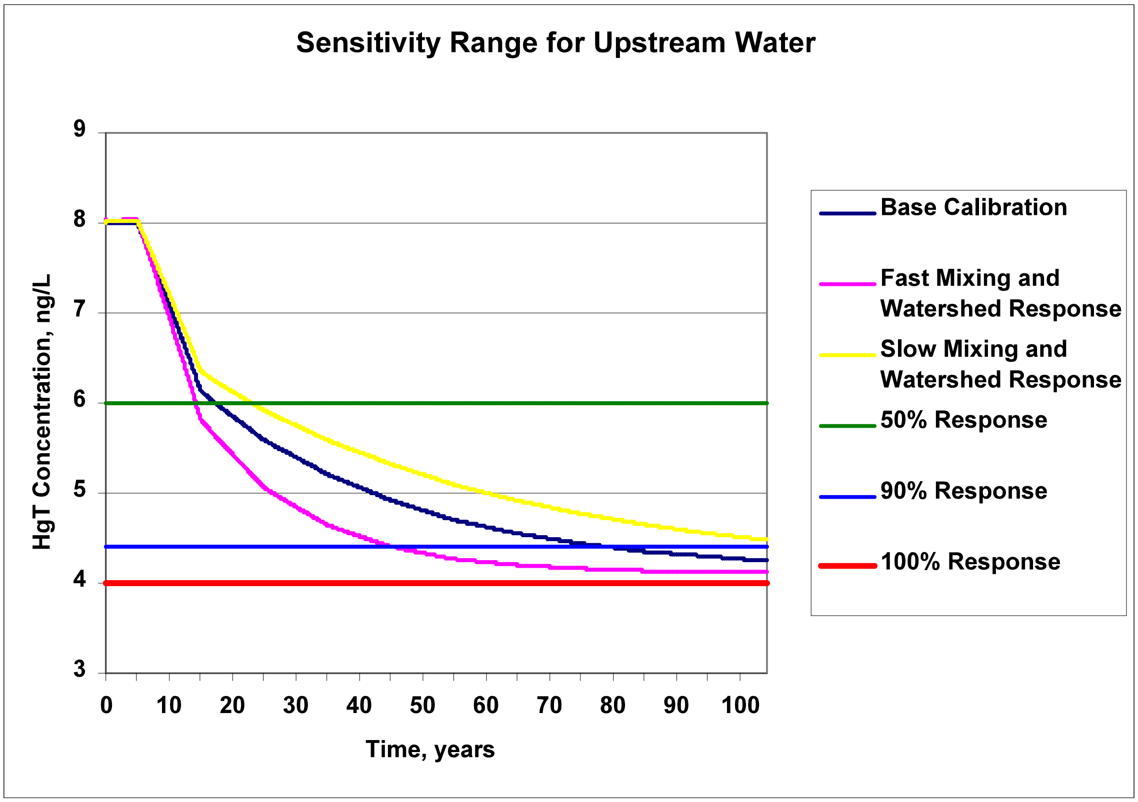

Figure A4-18. Sensitivity Range for Upstream Waters

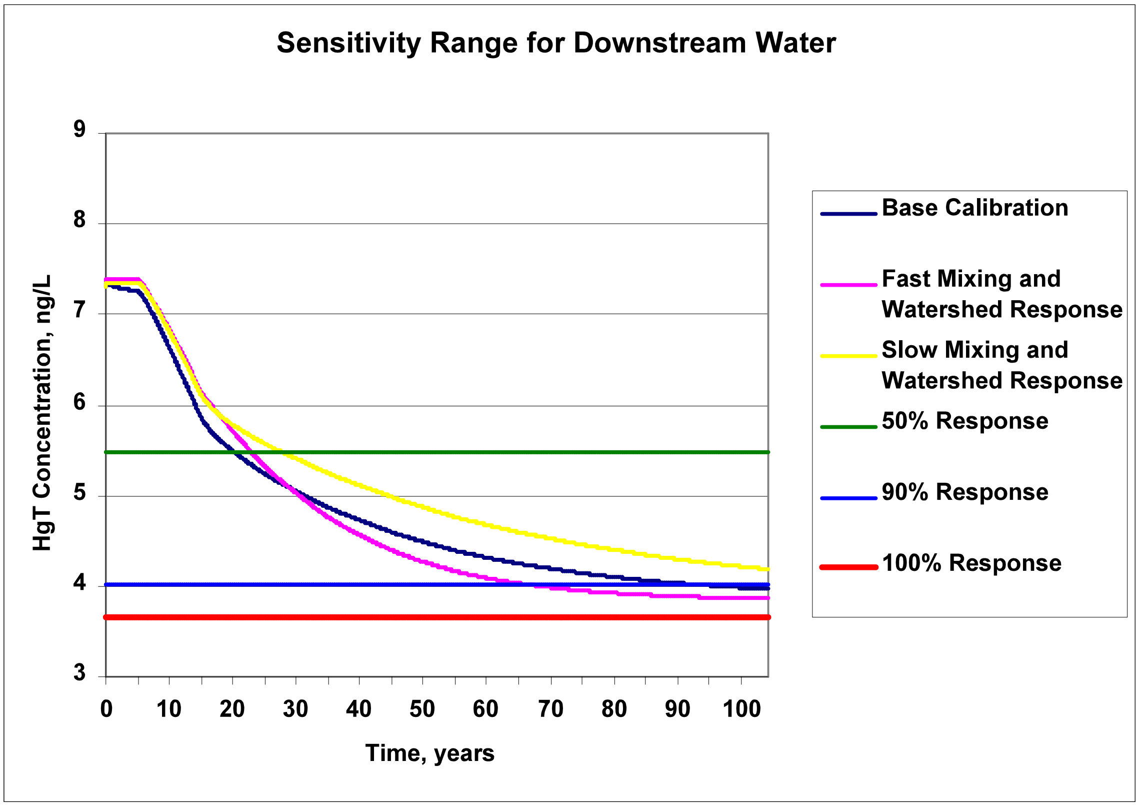

Figure A4-19. Sensitivity Range for Downstream Waters

APPENDIX A-5

MERCURY LOAD REDUCTION ANALYSIS AND RESPONSE FOR LAKE BARCO (FLORIDA)

A5.1 Introduction

Lake Barco is a small, seepage lake in northeast Florida with has a water surface area of 0.12 km^2 and a negligible catchment area. Lake Barco is located ca. 35 km east of Gainesville, Florida on the Ordway Preserve that is operated by the University of Florida. The Ordway is protected from direct human impacts, although some recreational fishing does take place. Hydrology and geochemistry of Lake Barco have been well characterized by past studies (Pollman, Lee et al. 1991; EPRI 2003). There are several nearby mercury sources. For example, Gainesville Regional Utilities (GRU) operates a medium-size coal-fired power plant approximately 40 km NW of Lake Barco. Emissions of Hg from the Deerhaven Unit No. 2 facility averaged ca. 30 kg/yr between 1998 and 2002 (range 13 to 47 kg/yr).

A5.2 Empirical Data from Lake Barco

Site specific data used to model Lake Barco were obtained from an EPRI report (EPRI 2003), and unpublished data collected by Tetra Tech, Inc. (Schofield 1998) and associates (Pers. Comm., C. Pollman, Tetra Tech, Inc., 2005).

A5.3 SERAFM Application: Lake Barco

Table A5-1. Lake Barco Parameter Values

Parameter	Lake Barco
Watershed Area	0
Percent Impervious	n/a
Percent Forest	n/a
Percent Riparian	n/a
Percent Upland	n/a
Lake Area	118,000 m^2
Catchment/Lake Ratio	0
Epilimnion Depth	3.7 m
Hypolimnion Depth	n/a
Hypolimnion Anoxia	n/a
Hydraulic Residence Time	n/a
Inflow/Outflow	0
Water pH	4.5
Epilimnion DOC	0.85 mg/L
Trophic Status	Oligotrophic
Annual Precipitation	134.8 cm/yr
HgII Conc. in Precip	11.5 ng/L
Wet Deposition (HgII)	15.5 ug/m2/yr
Dry Deposition (HgII)	15.5 ug/m2/yr
Wet Deposition (HgII)	0.23 ug/m2/yr
Dry Deposition (HgII)	0.23 ug/m2/yr

Table A5-2. Measured and Baseline Steady State Values for Lake Barco

Parameter	Measured	Predicted
Water Column MeHg Unfiltered	0.018 ng/L	0.040 ng/L
Water Column HgII Unfiltered	1.03 ng/L	0.96 ng/L
Sediment MeHg	1.9 – 6.9 ng/g [dry]	6.2 ng/g [dry]
Sediment HgII	152 - 186 ng/g [dry]	177.45 ng/g [dry]
Largemouth Bass Tissue Hg	2 yr old lgmouth bass: 0.55 ± 0.25 ug/g [wet]	1.2 ug/g (0.7-1.7)
Observed BAF: FishHg/MeHg	3.06×10^7	

Table A5-3. Lake Barco SERAFM Calibrated Rate Constants

Lake Barco Calibrated Rate Constants			
Process	**Media**	**Value**	**Units**
Methylation	Epilimnion	0	Per day
	Sediment	0.014	Per day
Demethylation	Epilimnion	0.0001	Per day
	Sediment	0.40	per day
Biotic Reduction	Water Column	0.03	per day
Photo-Degradation	Water Column	0.002	per day per E/m^2-day
Photo-Reduction (Vis)	Water Column	0.0003	per day per E/m^2-day
Photo-Reduction (UV-B)	Water Column	0.02825	per day per E/m^2-day
Photo-Oxidation (UV-B)	Water Column	0.05885	per day per E/m^2-day
Dark Oxidation	Water Column	1.44	per day

Table A5-4. Time to Reach 90% Steady State After 50% Reduction in Atmospheric Deposition

	Lake Barco		
	Fast	Med	Slow
Epilimnion	13	27	41
Hypolimnion	--	--	--
Sediment	14	28	45
Fish	14	28	43

Fast = 1 cm active sediment layer, D (macro-dispersion coefficient) = 10^{-4} cm^2/s
Medium = 2 cm active sediment layer, D = 10^{-4} cm^2/s
Slow=3 cm sediment, D = 5×10^{-5} cm^2/s

Table A5-5. SERAFM Model Forecasts with Zero-Out Scenario for Coal-Fired Power Plants
(Medium Response Time Scenario)

Lake Barco	Time
Epilimnion	27
Epilimnion MeHg	--
Hypolimnion	--
Sediment	28
Fish	28

A5.4 References

EPRI (2003). Factors Affecting Predicted Responses of Fish Mercury Concentrations to Changes in Mercury Loading. Palo Alto, CA, Electric Power Research Institute Report 1005521.

Pollman, C., T. Lee, et al. (1991). "Preliminary analysis of the hydrologic and geochemical controls on acid-neutralizing capcity in two acidic seepage lakes in Florida." Water Resources Research 27(9): 2321-2355.

Schofield, C. (1998). Mercury Bioaccumulation in Lake Barco Centrachids. Final Report. Gainesville, FL, Tetra Tech, Inc.

APPENDIX B

QUALITATIVE ECOLOGICAL REVIEW OF MERCURY LITERATURE

B.1 Introduction

Section 3 describes the Agency's approach for identifying ecological benefits that may result from reductions in mercury emissions. A comprehensive quantitative analysis is not possible at this time given the current state of the science. However, research on the ecological effects of mercury exposures provides some qualitative support to the notion that reductions in mercury emissions could contribute to improvements in overall ecosystem health. This Appendix expands on the analysis presented in Sections 2, 3 and Appendix A by reviewing some of the recent scientific findings, focusing on studies not included in and/or published since the release of the *Mercury Study Report to Congress: Volume VI* in December 1997. This review is not a quantitative ecological toxicity assessment and the information presented here does not identify a "safe" ecological exposure level. The bulk of this research, based on both laboratory and field studies, suggests that because mercury is persistent in the environment and biomagnifies up the food chain when methylated, a wide variety of species and ecosystems may be harmed by excessive levels of mercury in the environment. At the outset, however, it must be noted that these studies do not distinguish between mercury from emissions of U.S. coal-fired utilities, other current mercury emissions, and legacy mercury.

Numerous studies have generated field data on the levels of mercury in environmental media as well as a variety of wild species. Comparison of contemporary measurements of atmospheric mercury and historical measures from lake sediment indicate that the amount of global atmospheric mercury has risen by a factor of 2-5 since the start of the industrial period (Boening, 2000). However, both the sediment and glacier core record have shown declining trends in mercury deposition since the early 1990s (Schuster et al., 2002). The body of work examining the effects of these exposures is still incomplete. A large portion of the research conducted to date has been carried out in the laboratory setting rather than in the wild; conclusions about effects on natural populations and overall ecosystem health are difficult to make at this time.

Extensive laboratory-based studies of mercury toxicity using captive-bred animals (mice, rats, monkeys, etc.) have been conducted. From this research, much has been learned about the mechanisms of mercury toxicity; unfortunately, many of these captive-bred test species may be genetically different from their wild relatives since they are bred to ensure consistency in study results. Also, exposures of animals in laboratory-based studies are not always comparable to average exposures in wild. Often times, a laboratory-based study will identify a potential adverse effect in a specific species, but the ability to determine if this effect is present in the wild is limited by the understanding of mercury fate, transport and exposure dynamics in any given ecosystem. As a result, laboratory-based experiments using captive-bred animals lacking genetic variation and environmentally relevant exposures may not accurately reflect the effects of mercury toxicity in the natural environment. Further study of ecosystem-specific exposures will be necessary to determine the nature and magnitude of the risks posed to wild species by mercury.

The review that follows seeks to summarize some of the adverse effects seen in organisms exposed to mercury in both laboratory and wild settings. The following review should help to illustrate that all living organisms and the ecosystems they co-inhabit are influenced to some degree by mercury pollution. At this time, the magnitude of the power plant contribution to ecological exposures cannot be quantified nationally, and the corresponding risk for adverse effect is difficult to determine. While the benefit of further reducing mercury emissions cannot be quantified for ecosystems at this time, we have described this benefit qualitatively for context.

B.2 Potential Exposure Media

B.2.1 Mercury in Air

Atmospheric deposition of elemental mercury (Hg^0) in the vapor phase is the primary pathway of global deposition (Boening, 2000). Measurable concentrations of elemental mercury vapor may be detected in ambient air, especially near sources of mercury emission (U.S.EPA, 1997). The rate and mechanism by which mercury is deposited to the earth's surface depends on the chemical form present in the air (U.S.EPA, 1997). Atmospheric mercury also exists in the divalent (Hg^{2+}) state but is often associated with particulate matter when in this form. Mercury associated with particulate matter typically settles out of the atmosphere at a faster rate than Hg^0 in the vapor phase. Wet deposition of mercury with precipitation is also possible as Hg^{2+} is soluble in water.

B.2.2 Mercury in Water

Mercury can enter surface water as elemental mercury, divalent mercury, or methylmercury (MeHg or CH_3HgCl). Once in aquatic systems, mercury can exist in dissolved forms or in complexes sorbed to particles and can undergo a variety of physical or chemical transformations (such as oxidization, reduction, methylation, and demethylation) depending on its form and conditions. Dissolved Hg^0 can volatilize from the water back into the atmosphere, and particulate mercury forms can become buried in the sediment bed. The percent of total mercury in surface waters that exists as methylmercury varies, but it has been estimated to average about eight percent, with some estimates as high as 20 percent (U.S.EPA, 1997).

Within a surface water body, contaminated sediments can act as an important mercury reservoir for decades, with mercury recycling from the sediment back into the aquatic ecosystem. Biological processes, such as the methylation of Hg^{2+} by sulfate-reducing bacteria and accumulation of mercury in benthic invertebrates, mediate the availability of mercury to other aquatic animals in the food chain (U.S.EPA, 1997).

B.2.3 Mercury in Soil

Mercury compounds enter the soil through atmospheric deposition and form stable complexes with soil particles (U.S.EPA, 1997). Both Hg^{2+} and Hg^0 are present in soil; however, these mercury species may be transformed into methylmercury and back by bacteria or other living organisms (Jereb et al., 2003). Both organic and inorganic forms of mercury undergo environmental transformation (Boening, 2000). Aquatic organisms are known to be capable of transforming mercury species (Gilmour and Henry, 1991). Less is known about mercury transformation in soil ecosystems. A recent *in vivo* laboratory study of the terrestrial isopod crustacean, *Porcellio scaber,* found that both methylation and demethylation of mercury compounds are possible in this organism (Jereb et al., 2003).[1] Thus, many of the mechanisms affecting biomagnification in the aquatic ecosystem are also possible in terrestrial soil ecosystems.

B.3 Bioaccumulation of Mercury

Mercury is one of the few metals that has been demonstrated to biomagnify in food chains (U.S.EPA, 1997), resulting in increasing tissue concentrations of mercury in organisms at successively higher trophic levels. Three terms are commonly used to describe the mechanism by which a contaminant accumulates in living tissues – bioconcentration, biomagnification, and bioaccumulation. The term “bioconcentration” refers to the accumulation of a chemical that occurs as a result of direct contact of an organism with its surrounding medium (e.g., uptake by a fish from water through the gills and epithelial tissue or uptake by earthworms from soil pore water through the skin) and does not include the ingestion of contaminated food. The term “biomagnification” refers to the increase in chemical concentration in organisms at successively higher trophic levels as a result of the ingestion of contaminated organisms at lower trophic levels. The term “bioaccumulation” refers to the net uptake of a contaminant from all possible pathways and includes the accumulation that may occur by direct exposure to contaminated media as well as uptake from food. Mercury, in its various forms, can bioconcentrate, bioaccumulate, and biomagnify. Biomagnification of mercury is apparent in many aquatic ecosystems, particularly in aquatic systems with food chains that depend on benthic organisms which live in the sediments, the primary site of methylation of inorganic mercury. Mercury also magnifies in terrestrial food chains, but to a lesser extent than in aquatic ones owing to lower levels of mercury methylation, less accumulation of mercury in organisms at the base of the food chain, and shorter food chains.

All forms of mercury can bioaccumulate to some degree; however, methylmercury generally accumulates to a greater extent than other forms because of its ability to biomagnify. Methylmercury is absorbed into tissues quickly, where it becomes sequestered. Inorganic mercury can also be absorbed, but usually at a slower rate and with lower efficiency than methylmercury. Elimination of methylmercury from fish is so slow that long-term reductions of mercury concentrations in fish are often due mainly to growth of the fish (“growth dilution”), whereas other mercury compounds are eliminated relatively quickly. Therefore, methylmercury (and thus total mercury) concentrations tend to increase in aquatic organisms as the trophic level in aquatic food webs increases. In addition, the proportion of total mercury that exists as methylmercury generally increases with trophic level (U.S.EPA, 1997).

B.4 Exposure and Toxic Effects in Wildlife

[1] *In vivo* studies involve experiments on biochemical reactions inside living organisms or cells. *In vitro* studies investigate reactions outside of cells, and therefore may be less representative of responses by organisms in the natural environment than *in vivo* studies.

A large portion of mercury research conducted to date has focused on the effects of mercury exposure on aquatic ecosystems. In general, organic mercury species are more toxic to aquatic organisms than inorganic mercury, and toxicity increases with temperature and decreases with water hardness (Boening, 2000).

B.4.1 Aquatic Plant Species

Accumulation of mercury in the primary producers at the base of the aquatic food web can have substantial impacts on the amount of mercury available to aquatic animals, not only via direct ingestion of the plants, but also from ingestion of detritus derived from the decomposition of plant material (Boening, 2000). Effects of mercury on aquatic plants include senescence, growth inhibition, decreased chlorophyll content and dry weight, decreased protein and RNA content, inhibited catalase and protease activities, inhibited and abnormal mitotic activity, increased free amino acid production, discoloration of floating leaves, and leaf and root necrosis (U.S.EPA, 1997; Boening, 2000). The level of mercury that results in toxic effects varies greatly among aquatic plant groups and species (U.S.EPA, 1997). Inorganic mercury concentrations in water of approximately 1,000 µg/L can adversely affect aquatic plants. In general, organic mercury compounds are more toxic to aquatic plants than inorganic forms (Boening, 2000).

Studies of inorganic mercury uptake by aquatic plants indicate that bioconcentration (as measured by the ratio of mercury concentration in tissue to mercury concentration in water) decreases with increasing concentrations of mercury in the water column. In laboratory experiments with concentrations of mercuric chloride ($HgCl_2$) ranging from 50 to 20,000 µg/L, maximum bioconcentration in water cabbage (*Pistia stratiotes*) occurred at water concentrations of 6,000 µg/L or less (Boening, 2000). Although mercury concentrations in the plants did increase with increasing mercury concentrations in the water column, only 20 percent accumulated at the highest concentration. Other studies of vascular aquatic plants have shown that mercury uptake may occur in roots rather than stems or shoots, with two to four times the accumulation of mercury in the roots over the shoots (Boening, 2000).

B.4.2 Aquatic Invertebrate Species

In general, aquatic invertebrate species vary widely in their susceptibility to mercury toxicity, although most are more sensitive during the larval stage than during other life stages (Boening, 2000). The concentration and speciation of mercury, developmental stage of the organism, and the temperature, salinity, and hardness of the water all influence the toxicity to aquatic invertebrates (Boening, 2000). Acute toxicity values (i.e., the lethal concentration for 50 percent of the population or LC_{50}) of inorganic mercury compounds identified in recent literature for freshwater invertebrates range from 2.2 µg Hg^{2+}/L for a cladoceran (*Daphnia pulex*) to 2,000 µg Hg^{2+}/L for the larval forms of three insects (U.S.EPA, 1997), with organic mercury compounds from 10 to 100 times more toxic than inorganic mercury (Boening, 2000). In marine invertebrates, the gastrula stage was found to be the most sensitive period of development in embryo toxicity tests (Bellas et al., 2001).

Bellas et al. (2001) examined the effects of mercuric chloride on sperm viability, fertilization, embryogenesis, and larval attachment of the ascidian (*Ciona intestinalis*), a minute sedentary marine invertebrate, at concentrations ranging from 8 to 256 µg /L $HgCl_2$. Larval attachment and embryogenesis were the most sensitive endpoints, showing effects at concentrations of 16 and 64 μg/L, respectively. No significant differences across concentrations were observed in fertilization, and the authors noted that the effect of trace metals on sperm viability is controversial. However, embryonic development was substantially affected and decreased larval attachment was observed in the newly hatched larvae.

The toxicity of different forms of mercury to aquatic invertebrates depends in part on the bioavailability of those forms to the animals. Mercury speciation and concentration affect bioavailability; the organic content of the water and/or sediment also plays a role given the tendency of inorganic and organic mercury compounds to form complexes with dissolved organic matter and to sorb to organic particles. Divalent mercury and methylmercury (CH_3HgCl) are both particle reactive and have a strong affinity for organic matter (Bellas et al., 2001). Because particulate organic matter is a food source for many species of benthic invertebrates, it can serve as a source of mercury intake by benthic organisms. On the other hand, organic matter tends to bind with mercury compounds, making them less bioavailable. In experiments conducted with *Leptocheirus plumulosus*, Lawrence and Mason (2000) found that mercury accumulation in the estuarine amphipod was reduced in sediments enriched in organic matter. They also found that methylmercury was more readily available for uptake. Sjoblom et al. (2000) showed that in freshwater, most dissolved inorganic mercury is bound to dissolved organic matter. Dissolved humic substances in freshwater exert a strongly negative influence on the bioavailability of both inorganic mercury and methylmercury.

In addition, the beneficial role of consuming increased amounts of the algae (*Chlorella vulgaris*) was cited in one study. Ramirez-Perez et al. (2004) tested the age-specific responses to mercuric chloride of a rotifer (Brachionus calyciflorus), an organism found between phytoplankton and fish larvae in aquatic food chains, using two algal densities. With increasing mercury concentrations (ranging up to 5 μg/L $HgCl_2$), the researchers observed an increasingly negative effect on survivorship, reproduction, and lifespan of the rotifer, although less of an impact was seen when the food level (i.e., algal density) was higher.

Recent studies in aquatic invertebrates have demonstrated the ability of mercury to suppress the immune system. In marine bivalves, the internal immune system is based on the ability of circulating cells known as hemocytes to fend off foreign bacteria and viruses using phagocytic and microbicidal mechanisms (Cooper and Knowler, 1992; Fournier et al., 2001). An *in vivo* study demonstrated that clams (*Mya arenaria*) accumulate methylmercury to a much greater extent than $HgCl_2$ and that methylmercury leads to greater immune suppression (Fournier et al., 2001). Mercury exposure concentrations ranged from 10^{-9} to 10^{-5} M. No detrimental effects were observed at concentrations below 10^{-6} M, but at higher concentrations, a clear relationship between increasing mercury accumulation and decreasing phagocytic activity of hemocytes was evident from 7 to 28 days after exposure. In an *in vitro* analysis of a small aquatic worm (*Tubifex tubifex*), phagocytic responses were tested with methylmercury or mercuric chloride at concentrations from 10^{-9} to 10^{-4} M (Sauvé et al., 2002). Although the authors noted that short-term *in vitro* assays are often not as realistic as *in vivo* tests, the results are similar to Fournier et al. (2001) in that immunotoxicity, expressed as the concentration that resulted in a 50 percent reduction in phagocytic activity relative to controls, was observed at mercury levels around 10^{-6} M.

B.4.3 Fish and Amphibian Species

Effects of methylmercury on fish include reduced reproduction, impaired growth and development, behavioral abnormalities, altered blood chemistry, impaired osmoregulation and immunity, reduced feeding rates and predatory success, effects on oxygen exchange, and death (U.S.EPA, 1997). Bacterial methylation of inorganic mercury can occur either in sediment or in bacteria associated with the fish gills or gut. Greater than 90% of the mercury content in freshwater fish is in the methylmercury form (U.S.EPA, 1997). Symptoms of acute mercury poisoning in fish include increased secretion of mucous, flaring of gill opercula, increased respiration rate, loss of equilibrium, and sluggishness. When fish have been exposed to concentrations of mercury considered to be sublethal (i.e., below 30 µg/L), abnormalities ranging from physiological to reproductive to biochemical have been reported (Boening, 2000). Signs of chronic poisoning include emaciation, brain lesions, cataracts, inability to capture food, abnormal motor coordination, and various erratic behaviors (e.g., altered feeding behavior) (U.S.EPA, 1997).

Accumulation of mercury in fish depends on several factors. Although the highest mercury concentrations in fish generally occur in the blood, spleen, kidney, and liver, and may exceed those in muscle by 2 to 10 times, most of the mercury contained in a fish at any given time is associated with muscle tissues due to their larger mass relative to that of other tissues (U.S.EPA, 1997). Mercury concentrations in fish tissues tend to increase in both marine and freshwater fish with increasing age and size. For example, Redmayne et al. (2000) found that methylmercury concentrations in long-finned eels (*Anguilla dieffenbachia*) in New Zealand increase with both eel age and length. Also, mercury accumulation in fish is higher in waters with higher levels of dissolved organic carbon, which may assist mercury in entering the food chain. And at lower pH levels in water, methylmercury is a higher fraction of the total mercury in fish tissues than in waters at higher pH levels (Boening, 2000).

The toxicity of mercury to fish varies, depending on the fish's characteristics (e.g., species, life stage, age, and size), environmental factors (e.g., temperature, salinity, dissolved oxygen content, hardness, and the presence of other chemicals), and the form of mercury available. As with aquatic invertebrates, organomercury compounds, such as methylmercury, generally are much more acutely toxic than inorganic mercury to fish (U.S.EPA, 1997).

Recent studies provide examples of the toxic effects of mercury in fish at both laboratory and environmentally-relevant concentrations. For example, Houck and Cech (2004) exposed juvenile blackfish (*Orthodon microlepidotus*) to varying levels of methylmercury (0.21, 0.52 , 22.2, and 55.5 µg/g) (from 0.21 to 5.5 µg/g) over a 70-day period and found that the two highest dose groups exhibited decreased growth rates. In walleye (*Stizostedion vitreum*), Latif et al. (2001) assessed methylmercury effects on embryonic and larval stages. They found that increases in methylmercury concentrations at environmentally relevant concentrations in the water (from 0.0001, .002, 0.003, and 0.008 µg MeHg/L) were generally associated with linear declines in the hatching success of eggs; however, no statistical analysis was conducted. Furthermore, hatching success was not correlated with methylmercury concentrations in the eggs. In the embryonic stage, a decrease in heart rate was observed with higher waterborne methylmercury concentrations, but larval growth was not subsequently affected. In addition, information on the toxicity of mercury to fish at environmentally-relevant levels is found in the

British Columbia Ministry of Environment, Lands and Parks (2001) ambient water quality criteria and guideline for mercury.

Recent experiments have demonstrated important sublethal effects of mercury toxicity that might affect fish populations in the field. For example, Fjeld et al. (1998) found that although morphological disturbances were observed in only the group exposed to the highest level of methylmercury (i.e., 20 µg/L), there was long-term permanent impairment of feeding behavior in all exposed grayling (*Thymallus thymallus*) embryos dosed during the first 10 days of development to methylmercury at concentrations ranging from 0.8 to 20 µg/L. Berntssen et al. (2003) demonstrated the relative susceptibility of the Atlantic salmon (*Salmo salar*) brain to methylmercury tissue concentrations above a threshold of 10,000 µg/kg, as compared to kidney and liver tissues. There was no evidence of reduced growth in the salmon, but substantial reductions in neural enzyme activity and alterations in feeding behavior were observed. LeBlond and Hontela (1999) examined the effects of mercuric chloride and methylmercury on cellular synthesis of the hormone cortisol using an *in vitro* assay with rainbow trout (*Oncorhynchus mykiss*). Although the specific cellular site of action was not determined for the two mercury species, both disrupted production of cortisol.

Samson et al. (2001) demonstrated effects ranging from delayed mortality syndrome to physical (e.g., faint heartbeats) and behavioral abnormalities in embryonic zebrafish (*Danio rerio*) exposed to methylmercury levels of 5, 10, and 15 µg/L for several different periods of time. Prey capture ability was impaired in larvae exposed continuously to 10 µg/L, even after 4 days in clean water. Although morphological defects were not always evident, the authors conclude that functional impairment is a more subtle response to developmental toxicants than mortality or the production of morphological defects, which may not be sensitive enough endpoints for determining safe levels of a toxicant in the environment. Sweet and Zelikoff (2001) cite several *in vivo* and *in vitro* studies, with effect-inducing doses of 30 to 350 µg/L and 0.3 to 10 µM, respectively, that have demonstrated immunotoxic effects in fish due to mercury exposure ranging from depressed blood cell production and enzyme activity to enhanced cell death.

It is generally thought that toxic effects are unlikely to occur in fish in the environment, where the concentrations of mercury in surface waters are much lower than those in many of the laboratory experiments conducted. For example, Friedmann et al. (1996) examined 14 northern pike (*Esox luclus*) from Lake Champlain for reproductive status and mercury concentrations in tissues. This species tends to accumulate more mercury than many others because northern pike are top-level predators. Despite the fact that the average muscle concentration was 325 µg Hg/kg wet-weight, higher than the national average of 100 µg Hg/kg, no correlation was observed between mercury content, gonadosomatic index, and gonadal sex steroid levels. However, evidence on more subtle effects, including those discussed in the previous paragraph, shows that effects on behavior, reproduction, and development can occur at relatively low mercury concentrations in water (U.S.EPA, 1997). Many of the earlier toxicity studies focused on endpoints related to growth, reproduction, and mortality. More recent studies are demonstrating more subtle effects, including effects on neural and immunological endpoints.

Amphibians are similar to fish in sensitivity to mercury. Acute toxicity (i.e., LC_{50}) values for a variety of embryo-larval stage amphibians exposed to inorganic mercury compounds range from 1.3 to 107.5 µg Hg/L, which are similar to values found for fish (Boening, 2000). The toxicity of mercuric chloride to tadpoles ranges from approximately 50 to 760 µg Hg/L for the frog (*Rana hexadactyla*), clawed toad (*Xenopus laevis*), and toad (*Bufo melanstictus*) (Boening, 2000).

B.4.4 Terrestrial Plant Species

The effect of mercury pollution on terrestrial plants has not been studied as extensively as aquatic plants. Terrestrial plants typically acquire mercury through contaminated soil. The accumulation of mercury in terrestrial plants increases with increasing soil mercury concentration (Boening, 2000). Methylmercury is more toxic to terrestrial plants than Hg^{2+} (U.S. EPA, 1997). Wild plant communities located in areas with soil contaminated with a sufficient amount of mercury atmospheric deposition may be at risk for exhibiting mercury-induced chronic effects. It is probable that subtle disturbances to a community occur at lower concentrations (chronic exposures) of mercury than those suggested in the literature (based largely on acute exposures) (Boening, 2000). Mosses have shown the ability to acquire mercury directly through atmospheric deposition (Boening, 2000).

Terrestrial plants accumulate mercury primarily in their root structures (Greger et al., 2005). In a recent laboratory study, the soil of six plant species was dosed with a solution containing 200 µg/L $HgCl_2$. The plants included white clover (*Trifolium repens*), spring wheat (*Triticum aestivum*), sugar beet (*Beta vulgaris*), oil-seed rape (*Brassica napus*), willow (*Salix viminalis*), and garden pea (*Pisum sativum*). All of the examined plant species were able to take up mercury in the root. However, transport of mercury from the root to the shoot was low (0.17 – 2.5%). Mercury that reached the leaves was sequestered and not released into the ambient air through transpiration. These findings indicate that the studied plant species may serve as a reservoir of mercury; however, they are not capable of remobilizing terrestrial soil mercury deposits into the atmosphere. Such studies may eventually lead to development of plants to assist phytoremediation of mercury contaminated sites.

B.4.5 Terrestrial Invertebrate Species

The available information regarding the toxicity of mercury on terrestrial invertebrate species is limited, and the topic needs further study. Terrestrial invertebrates contribute to the diet of numerous species including birds and mammals. Some terrestrial species are known to transform Hg^{2+} to methylmercury and back (Jereb et al., 2003). A better understanding of the toxicity of mercury on terrestrial invertebrates would help to characterize whether, and if so, how the health of lower trophic level species affects the overall health of the ecosystem.

The phagocytic immune response of terrestrial invertebrate worms was assessed following *in vitro* exposure to either mercuric chloride or methylmercury (Sauvé et al., 2002). The authors dosed three earthworm species (*Lumbricus terrestris*, *Eisenia fetida*, *Aporrectodea turgida*) with methylmercury or mercuric chloride at concentrations from 10^{-9} to 10^{-4} M. Both $HgCl_2$ and CH_3HgCl inhibited the phagocytic immune response to challenge with carboxylate microsphere beads. The concentration necessary to reduce the phagocytic response by 20% varied across species from approximately 10^{-8} to 10^{-6} M for methylmercury and from 10^{-6} to 10^{-7} M for $HgCl_2$. Inhibition of the phagocytic response could potentially lead to increased infection by opportunistic pathogens such as viruses and/or bacteria. Species-specific considerations are clearly a factor when determining mercury immunotoxicity in invertebrate worm species.

Sauvé and Fournier (2005) continued the examination of the phagocytic immune response to methylmercury exposure in the earthworm (*Eisenia andrei*) and found age-specific differences. Four age groups were studied in both *in vitro* and *in vivo* assays. The youngest hatchling group exhibited lower *in vitro* phagocytic activity than that of adults. Neither adults, nor hatchlings showed a significant decrease in phagocytosis at methylmercury concentrations up to 10^{-7} M in invertebrate worms. Thus, hatchlings do not show a higher sensitivity to mercury exposure, but they have less capacity to respond to immune challenges requiring a phagocytic response.

A laboratory-based *in vivo* immune assay was employed in which *Eisenia andrei* worms were exposed to varying methylmercury concentrations for 5 days prior to evaluation of phagocytic response (Sauvé and Fournier, 2005). Adult worms exposed to filter paper dosed with concentrations of methylmercury (1 – 2 $\mu g/cm^2$ MeHg) showed an immune response ranging up to 300% greater than the control. This could potentially mean that adult worms exposed to sublethal methylmercury may become more effective in dealing with immune challenges by opportunistic pathogens; however, the long term effects of sublethal exposures to methylmercury were not examined in this study of invertebrate worms. The same effect was not observed in hatchling worms, and possible effects of elevated immune function over an extended period of time are not known.

B.4.6 Avian Species

In exposed birds, the liver and kidney are typically the sites of highest mercury levels (Boening, 2000). Sublethal effects of mercury on birds include neurobehavioral effects, reduced food consumption, liver and kidney damage, spinal cord damage, reduced cardiovascular function, impaired immunity, reduced muscular coordination, impaired growth and development, altered blood chemistry, and reproductive effects (U.S.EPA, 1997). Bird species occupying top predator roles in aquatic ecosystems frequently exhibit elevated levels of mercury. For example, mercury has been detected in adult spectacled (*Somateria fischeri*) eiders in northern Alaska and in German Northern Goshawks (*Accipiter gentiles*) (Kenntner et al., 2003; Wilson et al., 2004). In non-pisciviorous bird species inhabiting strictly terrestrial ecosystems, the mercury body-burden are typically lower (Boening, 2000). Nevertheless, terrestrial seed-eating bird species may be exposed through the application of methylmercury containing fungicides during agricultural practice (Boening, 2000).

The common loon (*Gavia immer*) has been used as a potential indicator of methylmercury contamination in lake ecosystems across the northern United States and Canada (Meyer et al., 1998; Evers et al., 2003). In a long term field study of the adverse effects of mercury on the loon, eggs were collected across the northern United States between 1995 and 2001 (Evers et al., 2003). Eggs collected in the same geographic territory showed adult female blood mercury concentrations that were highly correlated with blood mercury concentrations in eggs. In the New England region, egg volume significantly decreased with increasing mercury concentration; however, the authors did not find a significant relationship between egg mercury concentration and reproductive success. The average mercury concentration found in infertile eggs (0.78 ± 0.5 μg/g, n = 201) was not significantly different from the average mercury concentration (0.74 ± 0.54 μg/g, n = 205) of the fertile eggs. A previous study found chick production to be lower at lakes where chick blood mercury concentrations were elevated; however, decreased chick production was not associated with adult mercury exposures in the study (Meyer et al., 1998).

A dose-response laboratory study of common loons was conducted to investigate the adverse effects of mercury exposure on chick development (Kenow et al., 2003). Eggs were collected in a four county region of northern Wisconsin during the summers of 1999 and 2000 and dosed from hatch to age 105 days with varying levels of methylmercury contaminated rainbow trout. While the methylmercury administered was found to contain 20% ethyl mercury, no effect on food consumption, growth in body mass or body length was measured in any of the exposure groups following dosing. Further, no signs of neurotoxic effects, such as behavioral abnormalities or loss of muscle coordination, were observed. The authors speculated that rapid excretion of methylmercury during feather growth offered some protection against adverse effects during the first 105 days of loon development.

A field study conducted in Hong Kong, China examined the breeding success of two Ardeid bird species exposed to metals (Connell et al., 2002). Ardeidae are fish-eating birds that include species such as herons, egrets and bitterns (De Luca-Abbott et al., 2001). The feathers of the Little Egret (*Egretta garzetta*) and the Black-crowned Night Heron (*Nycticorax nycticorax*) were analyzed for concentrations of copper, iron, manganese, zinc, lead, cadmium, chromium, and mercury. An examination of the possible adverse effects of metals exposure included a probabilistic assessment of breeding success. The authors concluded that mercury (0.5 – 7.1 μg/g dry wt feathers) increased the likelihood of adverse effects on the breeding success of the Little Egret at one of the six sites monitored. At this site, a maximum Risk Quotient (RQ) of 1.37 was calculated based on a 3.0 μg/g dry wt feathers No Observed Adverse Effect Level (NOAEL) and an average measured egret feather mercury concentration of 4.1 μg/g. The 3.0 μg/g NOAEL was derived from the available scientific literature (Burger and Gochfeld, 1997; Connell et al., 2002). The same analysis did not find any evidence of mercury effects on the breeding success of the Black-crowned Night Heron.

Burger and Gochfeld (1997) attempted to relate adverse effects of mercury exposures observed in the laboratory to field biomonitoring observations in birds. The authors identify laboratory studies that indicate exposures as low as 1.5 ppm in eggs and/or 5 to 40 ppm in feathers are associated with adverse effects such as impaired reproduction (Burger and Gochfeld, 1997). Egg mercury concentrations as low as 0.5 – 6.0 ppm wet weight are capable of causing decreased egg weight, embryo malformations, lowered hatchability, decreased chick growth and lowered chick survival (Burger and Gochfeld, 1997).

An examination of the total mercury (ppm, dry weight) in bird eggs from the New York City region show levels of mercury which exceed the 0.5 ppm adverse effect level identified by the authors for a number of species; however, not every studied location found the same result (Burger and Gochfeld, 1997). At risk species included: Snowy Egret (*Egretta thula*), Black Skimmer (*Rynchops niger*), Common Tern (*Sterna hirundo*), Foraster's Tern (*Sterna forsteri*), Roseate Tern (*Sterna dougallii*), and Herring Gull (*Larus argentatus*). The feathers of these same species were also analyzed for total mercury and were found to be within the range identified in the scientific literature as being associated with reduced hatchability of eggs, behavioral abnormalities of adults, and infertility (Burger and Gochfeld, 1997).

Work by Wayland et al. (2002, 2003) examined the health of the northern common eider (*Somateria millisima borealis*). The common eider is a sea duck that inhabits coastal areas in Canada, Greenland as well as other Arctic territories. In a 1999 field analysis, measured liver mercury concentrations ranging from 1.3 to 6.5 µg/g dry weight were negatively correlated to abdominal fat mass, spleen mass and body mass in males at the time of capture (Wayland et al., 2002). Further study of this population in 2000 revealed that mercury concentrations were also negatively correlated with heart mass and body mass at the time of dissection (Wayland et al., 2003). Immune endpoints were also examined and no statistically significant relationship was found between mercury concentration and swelling response to an injection of the antigen, phytohemagglutinin-P.

A laboratory-based egg study in eighty pairs of mallard ducks (*Anas platyrhynchos*) exposed to 0, 5, 10 or 20 µg/g methylmercury found neurological effects in exposed offspring (Heinz and Hoffman, 2003). Females laid 15 eggs while unexposed and another 15 during the exposure period. Following the exposure period, another 30 eggs were laid and examined for mercury content. Even-numbered eggs were incubated and allowed to hatch while odd-numbered eggs were saved for mercury analysis. Mercury in the even-numbered eggs was estimated by averaging mercury content in the neighboring odd-numbered eggs. Neurological signs of mercury poisoning included loss of coordination and staggered gait. These altered behaviors were observed in ducklings hatching from eggs containing 2.3 µg/g estimated mercury on a wet-weight basis. Developmental deformities were also observed in eggs containing as little as 1 µg/g estimated mercury. The authors did not conduct a statistical analysis of the data; however, they conclude that methylmercury concentrations in excess of 2 µg/g on a wet-weight basis will harm the neurological development of sensitive mallard embryos.

B.4.7 Mammalian Species

The effects of mercury on mammalian wildlife are similar to those found in humans, with the primary target being the central nervous system. Most mammalian studies have been conducted in laboratories. Relatively few studies of mammalian populations in the wild have been published (Boening, 2000). Mammals drawing all or a portion of their dietary intake from aquatic ecosystems will likely have greater exposure than those that do not. Thus, aquatic bioaccumulation is a factor in determining mammalian exposures. Some bioaccumulation may also occur in the terrestrial setting. Deer mice (*Peromyscus maniculatus*) sampled in Isle Royale National Park in the state of Michigan have liver mercury concentrations that may pose an exposure risk to higher trophic level predators such as the red fox (*Vulpes vulpes*) (Vucetich et al., 2001).

Extensive laboratory studies of monkeys (*Macaca fascicularis*) have been conducted in an attempt to elucidate the toxicity of mercury in humans. Sensory system impairment was observed in a cohort of monkeys dosed in utero with methylmercury through age four (Rice and Gilbert, 1990; Rice and Gilbert, 1995; Rice, 1998; Rice and Hayward, 1999). Exposure symptoms included impaired hearing, visual function, and ability to detect vibration. Doses of 10 or 25 µg/kg/day resulted in evidence of delayed neurotoxicity as well as impairment of auditory function (Rice, 1998). These same mechanisms of toxicity described in the laboratory setting may also be present in the wild. However, the diet of the *M. fascicularis* monkey in the wild has not been extensively studied. The wild diet of *M. fascicularis* is speculated to consist primarily of fruit but they are believed to be opportunistic omnivores willing to consume terrestrial invertebrates and bird eggs (Kemp and Burnett, 2003). Regardless, estimates of mercury exposure in the wild are difficult to make, but mercury exposure is of less concern for primarily herbivorous mammals.

Concern has recently grown that mercury exposure may be a contributing factor to the decline in the endangered Florida Panther population (*Puma concolor coryl*) (Barron et al., 2004). Barron et al. (2004) performed a probabilistic risk assessment of retrospective and current mercury exposure using a dietary model that incorporated the variability and uncertainty in ingestion rate, diet, body weight, and mercury exposure of panthers. Under the worst-case modeling conditions, the current risk of panthers developing clinical symptoms that may lead to death was 4.6%. Thus, there was a 4.6% chance that any given panther would receive a lethal mercury dose under the worst case exposure scenario. The authors concluded that past mercury exposures likely did adversely affect panthers in the Florida Everglades, but current estimated risks are significantly lower than past risks because of an estimated 70-90% decline in mercury exposure over the past decade (Barron et al., 2004).

Mink (*Mustela vison*) are an example of a species acquiring a portion of their diet foraging in aquatic ecosystems. Fish compose about 25% of the mink diet (Ferreras and Macdonald, 1999; Yamaguchi et al., 2003). Yamaguchi et al. (2003) calculated Risk Quotients for mink at four locations along the Thames River after sampling for mercury contamination in perch (*Perca fluviatilis*), roach (*Rutilus rutilus*), dace (*Leuciscus leuciscus*), eel (*Anguilla anguilla*), and pike (*Esox lucius*). Not all species were available at each sampling site. An RQ greater than one indicates that the concentration of mercury in fish is likely greater than the No Observable Adverse Effects Concentration (NOAEC) threshold (Giesy et al., 1994; Henry et al., 1998; Yamaguchi et al., 2003) and thus poses a risk to the exposed species at the time of assessment. The calculated RQ values for each species consumed by the mink at each of the four locations ranged from less than one to near 8. The authors speculate that the RQ of mercury for mink may be closer to 1 since the fish sampled in the study were close to the upper limits of the prey size typically selected by mink.

B.5 Ecosystems Potentially Affected

Ecosystems that could be affected by mercury exposure include those that already have high mercury levels, particularly those with top carnivore populations with high mercury loads; ecosystems with long aquatic food chains and piscivorous wildlife ; and ecosystems with soils or sediments low in organic content and high in minerals – promoting more soluble and bioavailable forms of mercury. Mercury levels in all of these ecosystems are likely declining as a result of recent regulations, but the quantitative effect on the ecosystems is unclear because the decline is slow, depending on sediment burial as a primary mechanism.

More pristine aquatic ecosystems tend to have longer food chains than eutrophic systems. Assuming equal inputs of mercury into the ecosystem from atmospheric deposition, biomagnification through multiple trophic levels may thus result in exposures to top carnivores in pristine systems that are comparable to or higher than corresponding exposures in eutrophic systems. In mixing zones (e.g., river entering lake, estuaries), the higher levels of suspended sediments with sorbed mercury compounds correlate with higher rates of bioaccumulation of mercury in zooplankton, and presumably the rest of the food chain.

B.6 Conclusions

A quantitative analysis of the ecological benefits of reduced mercury emissions is not possible at this time given the current state of the science. Recent research on the ecological effects of mercury exposures summarized in this appendix does provide qualitative support to the notion that reductions in mercury emissions from various sources could lead to improvements in overall ecosystem health. The bulk of this research, based on both laboratory and field studies, suggests that because mercury is persistent in the environment and biomagnifies up the food chain when methylated, a wide variety of species and ecosystems may be harmed by excessive levels of mercury in the environment.

To some degree, mercury contamination is present in virtually all environmental media, but aquatic systems appear to experience the greatest exposures due to higher rates of biomagnification possible in those systems. Elimination of methylmercury from fish is so slow that long-term reductions of mercury concentrations in fish are often due to growth of the fish ("growth dilution"), whereas other mercury compounds are eliminated relatively quickly. Piscivorous avian and mammalian wildlife are exposed to mercury mainly through the consumption of contaminated fish and, as a result, bioaccumulate mercury to levels greater than those in prey items (U.S.EPA, 1997).

Numerous studies have generated field data on the levels of mercury in a variety of wild species. The body of work examining the effects of these exposures, particularly in real world settings, is growing, but our understanding of the consequences is still incomplete. Much of the research conducted to date has been carried out in laboratory settings rather than in the wild; so EPA believes reliable conclusions about overall ecosystem health cannot be made at this time. Nevertheless, numerous adverse effects have been identified at environmentally relevant doses as well as at doses slightly above environmental concentrations. Although the magnitude of the power plant contribution to ecological exposures cannot be quantified so the corresponding risk for adverse effect cannot be determined, reducing the presence of mercury in the environment should reduce the potential for adverse ecological impacts.

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APPENDIX C

CARDIOVASCULAR EFFECTS AND METHYLMERCURY

C.1 Introduction

Some recent epidemiological studies suggest that methylmercury may be a risk factor for myocardial events, such as acute myocardial infarction (AMI), coronary heart disease (CHD), cardiovascular disease (CVD) or other adverse cardiovascular effects such as carotid atherosclerosis, increased blood pressure, or decreased heart rate variability. Other recent studies did not observe a relationship between methylmercury levels and myocardial events.[1]

This appendix presents a qualitative discussion of recent studies that have looked at the relationship between potential for cardiovascular impacts from chronic low-dose methylmercury exposures. The results of several key peer-reviewed studies are summarized and study uncertainties and other relevant information are also discussed. This section also includes a discussion of the beneficial effects of fish consumption.

The potential for adverse cardiovascular effects due to consumption of fish containing methylmercury is of particular interest given the evidence for the *protective* cardiovascular effect believed to occur from an increased dietary fish intake. Strong evidence indicates that consumption of fish, particularly fatty fish, has a cardio-protective effect (Wang et al. 2004; 2005 Dietary Guidelines Advisory Committee 2004; NRC 2000). The presence of omega-3 (*n*-3) fatty acids in fish oils is hypothesized to drive the preventive effect on cardiovascular disease (Calder 2003). Several mechanisms of action are recognized, including the stabilization of the atherosclerotic plaque (which, when ruptured, may cause a heart attack) by reducing the infiltration of inflammatory and immune cells (lymphocytes and macrophages) into the plaque. However, those studies relevant to the general population show cardiovascular health benefits were for fish in the diet, not for isolated omega-3 fatty acids, such as in fish oil supplements, suggesting the potential for synergistic benefits. Thus, consumption of fish containing methylmercury is not necessarily detrimental even though some evidence suggests that the cardiovascular system may be a target system for methylmercury exposure. The cardio-protective effect of fish consumption has not been observed in all studies (Curb and Reed 1985; Morris et al. 1992; Folsom and Demissie 2004).

C.2 Acute Myocardial Infarctions and Major Cardiovascular Effects

Salonen et al. (1995), Virtanen et al. (2005), and Guallar et al. (2002) have reported an association between increased risk of AMI (or other major cardiovascular impacts) and exposures to methylmercury via fish consumption or mercury levels in the body. The findings of these studies are summarized in this section.

[1] As described in detail by NRC (2000) and elsewhere, consumption fish containing methylmercury is the primary route of exposure to methylmercury.

C.2.1 The Kuopio Ischemic Heart Disease Risk Factor Study (KIHD) Cohort

Salonen et al. (1995) investigated the association between methylmercury and AMI in a study of a subset of men from the KIHD study group (n = 1,833 Finnish males age 42 to 60 years). Rissanen et al. (2000) in a follow up to this study extended observation of the same cohort and Virtanen (2005) also utilized this cohort. In theory, the findings could be specific only for men in Eastern Finland, who traditionally have a high intake of meat, fish and saturated animal fat and a low intake of selenium and vitamin C and, most likely, other vegetable-derived antioxidants. However these studies provide new information about potential mechanisms in how methylmercury interacts with the cardiovascular system in humans, even though the consequences of methylmercury intake for the cardiovascular system may vary among populations due to different confounding factors (Salonen et al. 1995).

In the Salonen et al. (1995) study, all subjects' mercury levels were evaluated by hair analysis, and subjects were grouped into three exposure groups according to hair mercury levels. Urinalysis was also conducted for a subset of men who either had an AMI during the follow-up or acted as the controls to that subset. Fish consumption was measured through an interview-checked four-day food recording. Daily fish intake ranged from 0 to 619.2 g/day, and hair concentrations ranged from 0 to 15.67 ppm (μg/g). Stern (2005) observes that the estimated mean dietary methylmercury intake was 7.6 ug/day, which is only somewhat larger than the intake corresponding to the EPA RfD for a 70-kg (average) man of 7.0 ug/day (the corresponding mean hair mercury concentration was 1.92 μg/g (ppm) (Stern 2005). For the subset of men who had an AMI, two control subjects were matched to each patient according to age, municipality of residence, and date of baseline examination. Occurrence of AMI and deaths due to CHD and CVD were recorded over the course of a seven-year period. In the analysis of the results, Salonen et al. found that "the hair mercury (r = .27) and the urinary mercury (r = .47) correlated with the estimated fish intake." Men with the highest hair mercury content (≥ 2.0 μg/g and ranging up to 15.67 μg/g - Stern (2005) estimates that ≥ 2.0 μg/g is likely equivalent to the 90th percentile in U.S. men) had twice the risk of AMI when compared to men in the two lowest exposure groups when adjusting for age, examination year, ischemic exercise, electrocardiogram (ECG), and maximal oxygen uptake (relative risk [RR] 2.0; 95% confidence interval [CI] 1.2-3.1). The risk of AMI decreased slightly for men within the highest hair mercury category (≥ 2.0 μg/g) after adjusting for all confounders and risk factors[2] but was still statistically significant (RR = 1.7; 95% CI 1.03-2.8). The relative risk was similar for coronary deaths but not statistically significant due to the smaller number of events. However, men in this group also had elevated risks of cardiovascular death (RR = 2.9; 95% CI 1.2-6.6) and death by all causes (RR 2.3; 95% CI 1.4-3.6) after adjusting for all confounders and risk factors. Urinary mercury levels were also significantly (and independently) associated with risk of AMI after adjusting for the strongest risk factors (for each μg mercury excreted daily, the risk of AMI increased by 36%; 95% CI 1% to 82%). Based on these results, the study authors conclude that, "although consumption of fish may be healthy in general, some fish may contain agents that are

[2] Risk factors included age, examination year, ischemic exercise ECG, maximal oxygen uptake, family history of CHD, cigarette-years, mean systolic blood pressure, diabetes, socioeconomic status, place of residence (urban vs. rural), dietary iron intake, and serum apolipoprotein B, HDL_2 cholesterol, and ferritin concentrations.

not healthy for the human cardiovascular system." Further, the authors suggest that mercury is a risk factor for coronary and fatal CVD.

Two relevant follow-up studies have been conducted on the men in the KIHD population. Rissanen et al. (2000) examined the interaction of mercury and the serum *n*-3 end-product fatty acids docosahexaenoic acid (DHA) and docosapentaenoic acid (DPA). For this analysis, the men in the KIHD cohort study were divided into quintiles based on their level of serum fatty acids. Risk of acute coronary events was examined within the study cohort as a function of *n*-3 fatty acid levels. After adjusting for other risk factors,[3] men in the highest fifth (quintile) of *n*-3 fatty acid level exhibited a 44 percent reduced risk of acute coronary events (95% CI 11% to 65%) when compared to the lowest quintile. When the data were stratified by hair mercury concentration into those with hair mercury content above and below 2 μg/g, individuals with lower hair mercury who were also in the upper quintile of *n*-3 fatty acid level had a 67 percent reduced risk of acute coronary events (95% CI 19% to 87%) compared with men with higher hair mercury who were also in the upper quintile of *n*-3 fatty acid level. In each quintile, subjects with the higher hair mercury concentrations had a higher risk of acute coronary events, suggesting that the cardio-protective effects of the serum fatty acids was attenuated by the mercury. Based on these results, Rissanen et al. suggest that their data "provide for the concept that fish-oil derived fatty acids reduce the risk of acute coronary events. However, a high mercury concentration in fish could attenuate this protective effect."

In a more recent follow-up study of the KIHD cohort, the association between mercury and the risk of acute coronary events and mortality from CVD, CHD, and all causes was re-evaluated in a group of 1,887 men (Virtanen et al. 2005). The study also examined whether mercury could interfere with the beneficial effects of fish oils. The interaction between mercury and the serum *n*-3 end-product fatty acids DHA, DPA, and eicosapentaenoic acid was investigated. Deaths by CVD, CHD, and all-causes were recorded, and fish consumption was measured through an interview-checked 4-day food recording. Fish intake in men for those in the highest third (tertile) of hair mercury content was more than double that of the lowest tertile (65 vs. 30 g/day, respectively). High mercury content in hair was also most strongly associated with fish intake and serum DHA plus DPA concentrations. Men in the highest tertile of hair mercury content (>2.03 μg/g) had an increased risk for frank cardiovascular effects, including AMI (RR 1.60; 95% CI 1.24-2.06), CVD (RR 1.68; 95% CI 1.15-2.44), CHD (RR 1.56; 95% CI 0.99 to 2.46), and any death (RR 1.38; 95% CI 1.15-1.66), compared with men in the combined lower two thirds.[4] For each microgram of mercury in hair, the risk of acute coronary events increased, on average, by 11 percent (95% CI 6% to 17%). Based on these results, Virtanen et al. concluded that "high content of mercury in hair may be a risk factor for acute coronary events

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[3] Risk factors included age, examination years, body mass index, maximal oxygen uptake, hair mercury content, serum ferritin, serum LDL cholesterol, systolic blood pressure, serum insulin, ADP-induced platelet aggregation, socioeconomic status, ischemic findings in exercise test, smoking, place of residence, and dietary energy intake.

[4] Estimated increased risks were adjusted for age, examination year, high-density lipoprotein (HDL) and low-density lipoprotein (LDL) cholesterol, body mass index (BMI), family history of ischemic heart disease, systolic blood pressure, maximal oxygen uptake, urinary excretion of nicotine metabolites, serum selenium, alcohol intake, serum DHA + DPA as a proportion of all fatty acids in serum, and intake of saturated fatty acids, fiber, and vitamin C and E.

and CVD, CHD, and all-cause mortality in middle-aged eastern Finnish men." Furthermore, the authors concluded that "mercury may also attenuate the protective effects of fish on cardiovascular health."

C.2.2 The European Multicenter Case Control Study on Antioxidants, Myocardial Infarction and Cancer of the Breast (EURAMIC) Cohort

In a case-control study using subjects from the EURAMIC study population (n = 1,400 men from eight European countries and Israel), Guallar et al. (2002) investigated the relationship between the risk of a first myocardial infarction in men and mercury levels measured in toenail clippings[5] and DHA levels in adipose tissue. The investigators reported that mercury levels in patients who had suffered AMI were 15 percent higher (after adjusting for DHA level and coronary risk factors) than levels in controls. Additionally, men in the highest quintile of mercury exposures exhibited a risk-adjusted[6] 2.16-fold increased risk (odds ratio [OR]) of myocardial infarction (95% CI 1.09-4.29) when compared to the lowest quintile. DHA level also was inversely associated with the risk of myocardial infarction after adjusting for mercury level (OR 0.59; 95% CI 0.30-1.19). Consequently, toenail mercury level was directly associated with the risk of myocardial infarction and adipose tissue DHA level was inversely associated with the risk. One study location, which had higher mercury than the others, appeared to be influential in the analysis. Guallar et al. concluded that "high mercury content may diminish the cardio-protective effect of fish intake."

C.2.3 Mechanisms for Cardiovascular Impacts

Currently, there is a general lack of mechanistic evidence for the role of methylmercury in heart disease (Stern 2005). However, Salonen et al. (1995), Virtanen (2005), and Guallar et al. (2002) summarize several mechanistic bases by which mercury may increase the risk of adverse cardiovascular impacts. The increased risk may be related to a reduction in the body's antioxidative capacity and the promotion of free radical stress and lipid peroxidation. A reduction in antioxidative capacity may be due to the high affinity of mercury for sulfhydryl groups (thereby inactivating antioxidative thiolic compounds) and mercury's tendency to bind to selenium and form an insoluble complex (selenium is believed to be a factor in catalyzing the formation of free-radical scavengers). Mercury is a transitional metal and therefore can promote the formation of free radicals via Fenton-type reactions. Additionally, Virtanen et al. (2005) note that mercury inactivates paraoxonase, an extracellular enzyme that may help prevent AMI. Mercury may also may promote ADP-induced platelet aggregation and blood coagulation, inhibit endothelial-cell formation and migration, and affect apoptosis (i.e., programmed cell death) and inflammatory responses.

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[5] It should be noted that although measuring mercury exposure through toenail clippings appears to quantitatively reflect dietary intake, this method has not been well characterized in comparison to hair or blood mercury exposure. Consequently, it is not possible to distinguish elemental mercury exposure from that of methyl mercury (Stern 2005). Additionally, results from this study cannot be compared to those that measure mercury through hair or blood.

[6] Adjusted for age, DHA, BMI, waist:hip ratio, smoking status, alcohol intake, high-density lipoprotein cholesterol, diabetes, history of hypertension, parental myocardial infarction, α-tocopherol level, ß-carotene level, toenail selenium level, and toenail weight.

C.2.4 Other Studies Evaluating CVD and Mercury Levels

In contrast with the aforementioned studies that may demonstrate a correlation between methylmercury and AMI, CHD, and CVD, Yoshizawa et al. (2002) conducted a study specifically addressing the relationship between total mercury exposure and the risk of coronary heart disease, and reported no significant association. This study utilized the Health Professionals Follow-up Study as its study population and examined a subset of 470 patients, including men who had fatal coronary disease, nonfatal myocardial infarction, coronary-artery bypass surgery, or percutaneous transluminal coronary angioplasty, as well as controls. Mercury levels were measured via toenail clippings. After adjusting for age, smoking and other risk factors, toenail mercury was not associated with the risk of coronary heart disease (CHD). Adjustment for intake of *n*-3 fatty acids from fish did not appreciably change these results. When dentists were excluded from the analysis, an association of toenail mercury with CHD was observed by comparing the highest and lowest quintiles of mercury exposure (RR 1.27; 95% CI 0.62-2.59; *P* for trend = 0.43) (Yoshizawa et al. 2002). This relationship, however, was not statistically significant (possibly due to the 53 percent reduction in total number of cases, to 220), suggesting that further study with a larger sample may be warranted.

In a recent review of the cardiovascular health effects of methylmercury, Stern (2005) noted that interpretation of the Yoshizawa results may depend on whether it is hypothesized that total mercury exposure or specifically methylmercury exposure is responsible for the cardiovascular effects of mercury. If only methylmercury is responsible for cardiovascular effects then the elevated exposure to elemental mercury of dentists would tend to confound the underlying association of methylmercury and cardiovascular effects. Stern (2005) also notes that the strengthening of the association when an adjustment is made for n-3 fatty acids is consistent with the observations in other studies that overall risk occurs as a balance between the protective effect of n-3 fatty acids from fish and the adverse effects from methylmercury.

In a study that focused on inorganic mercury exposure from dental amalgams, Ahlqwist et al. (1999) examined associations between serum mercury and myocardial infarctions (among other health outcomes), among 1,462 Swedish women (ages 38 to 60 at study initiation) for 25 years. Mercury concentration in serum was measured in all subjects at the beginning of the study and, 13 years later, for a subsample of 142 women from one age group (all born in 1922). No statistically significant association between serum mercury and MI was found. However, Stern (2005) notes that it is difficult to assess the significance of these findings for methylmercury exposures that occur through fish consumption as serum mercury disproportionally reflects inorganic mercury exposure.

In a case-control study of both men and women with a first-time MI from a longitudinal cohort in northern Sweden (n=78), Hallgren et al. (2001) analyzed erythrocyte mercury concentration in order to study a possible association between increased fish consumption and reduced coronary heart disease (concentration of mercury in erythrocytes is often used as an index of fish consumption). The researchers summed the *n-3* fatty acids, EPA and DHA, to express the percentage of total plasma polyunsaturated fatty acids (P-PUFA). Mean levels of mercury in erythrocytes were reported as 4.44 ng/g in cases and 5.42 ng/g in controls. Stern (2005) notes that these concentrations translate to about 2.8 and 3.4 ng/L, respectively, for mercury concentrations in whole blood. Hallgren et al. (2001) concluded that erythrocyte mercury concentration alone was not significantly associated with the occurrence of MI but found that higher levels of P-PUFA's and erythocyte mercury (which was used as a marker of fish intake) are significantly associated with lower risk of myocardial infarction. Various combinations of high and low P-PUFA and high and low mercury concentration were assessed, and none of the combinations had a statistically significant odds ratio (OR) greater than 1.0, including the group of subjects with high mercury and low P-PUFA that might be expected to have a higher risk of heart disease. The high mercury-low P-PUFA group did have an OR greater than 1.0, but this group consisted of only four individuals and the result was not considered statistically significant.

In two studies of death certificates in the region near Minamata City, Japan, Tamashiro et al. (1984 and 1986) investigated the significance of heart disease as a primary or secondary cause of death. Tamashiro et al. (1984) studied the causes of death among those with official diagnoses of Minamata disease in a region including Minamata City in two groups: those who died from 1954 to 1969 (n=44), and those who died from 1970 to 1980 (n=334). In the first group, Minamata disease and diseases of the central nervous system were the underlying causes of death, but nonischemic heart disease accounted for 50 percent of the secondary causes of death (multiple causes of death could be reported). Analysis of the second group constituted a case-control study by matching individuals with a control in the same city or town by age, sex, and year of death. In no disease were the ORs observed to be significantly high or low. However, when non-Minamata diseases and Minamata diseases were mentioned on the same death certificate, non-ischemic heart disease (for males and females combined) was the only statistically significant cause of death for this second group.

Tamashiro et al. (1986) also investigated the causes of death from death certificates in the Minamata City area, but focused on fishermen and their families residing in a small coastal area of the city. Certificates from 1970 to 1981 were examined and standard mortality rates (SMRs) were calculated using age-specific death rates for Minamata City as a standard. The results showed that SMRs for all categories of heart disease and hypertensive disease were not significantly elevated in the study population. Stern (2005) note several uncertainties associated with these results. He notes that deaths from methylmercury-related cardiovascular death could have possibly peaked prior to 1970, even though incidence of "Minamata disease" was observed to peak after this date in the study area (thus the selection of the study time period). The exposure to methylmercury may have varied throughout the study area, and death rates in Minamata City itself were used as the denominator in calculating the SMRs. Therefore, although overall deaths were more prevalent in this study area, lower exposures to methylmercury that resulted in higher rates of heart disease could have occurred. Furthermore, there were no data reported on the type of fish consumed and the relative consumption of *n-3* fatty acids which may provide protection against the cardiovascular effects of methylmercury. Stern (2005) also notes in his review that differences in the results from the two studies by Tamashiro et al. are probably related to study design, and the weaknesses of the Tamishiro 1986 study likely explain the differences in the studies with respect to the identification of heart diseases as potentially associated with highly elevated methylmercury exposures resulting in Minamata disease. Stern also notes that for ischemic heart disease only males appear to be at risk.

C.3 Other Cardiovascular Effects

Another possible adverse cardiovascular effect related to methylmercury exposure is accelerated progression of carotid atherosclerosis (i.e., the progressive narrowing and hardening of the arteries over time). This effect was observed in men from the KIHD study population, in which high hair mercury content was the second strongest predictor for the four-year increase in the mean intima-media thickness (IMT), a marker of early artherosclerosis (Salonen et al. 2000). Predictors for IMT ranked as follows: systolic blood pressure > hair mercury content> antidyslipidemic medication> dietary iron intake > cigarette pack years > age. Additionally, there was an 8 µm incremental increase in the four-year IMT for each µg/g increase in hair mercury content.

The case for mercury-induced blood pressure effects is not as strong, but some evidence suggests a possible link between methylmercury exposure and hypertension and heart rate variability. Oka et al. (2003) reviewed electrocardiogram data, along with blood pressure and pulse pressure measurement, in Minamata patients who had been exposed to methylmercury *in utero* and were institutionalized for "fetal Minamata disease" as adults. Subjects had a significantly elevated resting heart rate and a slightly (but not significantly) decreased variability in heart rate when compared to controls. Blood pressure did not differ between the two groups, but pulse pressure was significantly decreased among the subjects. Considering the limited data available, it is difficult to draw any conclusions at this time regarding heart rate variability and methylmercury exposure; more research is necessary to fully explore this area.

Cardiovascular effects have also been reported for children. A decrease in heart rate variability, an important measure of the ability of the cardiovascular system to withstand stress, was reported for children in the Faroe Islands (917 seven-year-old children) exposed *in utero* to methylmercury through maternal consumption of fish and marine mammals (additional post-natal exposures, again though consumption of fish and marine mammals may have also occurred). At seven years, children who had been exposed before birth to higher levels of cord blood mercury (up to 10 µg/L) had increased blood pressure as well as a decrease in heart rate variability (Sorensen et al. 1999). Blood pressure was based upon a single measure for each child. At fourteen years, the effect on blood pressure in this cohort was no longer observed but heart rate variability remained low (Grandjean et al. 2004). It is unclear, however, whether this effect will persist beyond age 14 and what the significance of this finding is for adverse health outcomes later in life.

A recent study by Vupputuri et al. (2005) showed no statistically significant association between total blood mercury and blood pressure in a sample of 1,240 women aged 16 to 49 years from the National Health and Nutrition Examination Survey 1999-2000. Additionally, no association was observed between total blood mercury and blood pressure in fish consumers when data were stratified by dietary fish intake (subjects were separated into those who consume fish and those who consume no fish).[7] More research is required to reduce the uncertainty

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[7] A weak association was observed, however, between total blood mercury and blood pressure among subjects who did *not* consume any fish (i.e., systolic blood pressure increased by 1.83 mm mercury per 1.3 µg/L increase in total blood mercury; 95% CI 0.36-3.30; $P = 0.018$; adjusted for age, race, income, body mass index, pregnancy status, and dietary sodium, potassium, and total calories). Vupputuri et al. (2205) suggest that the oils in fish may counteract

associated with the possibility of other cardiovascular effects, including blood pressure, from exposure to methylmercury in both adults and children. A potential confounding effect in several of the dietary studies could be based on method of preparation of fish (Mozaffarian et al. 2004).

C.4 Cardiovascular Health Benefits of Fish Consumption

Current federal dietary recommendations about fish consumption are found in the 2005 Dietary Guidelines for Americans (DHHS and USDA 2005). The Guidelines were based on an expert scientific report from the 2005 Dietary Guidelines Advisory Committee (2005 Dietary Guidelines Advisory Committee 2004). The Advisory Committee report in turn references two major sources: the National Academy of Sciences Institute of Medicine (IOM) report on Dietary References Intakes for Energy, Carbohydrate, Fiber, Fat, Fatty Acids, Cholesterol, Protein, and Amino Acids (IOM 2002) and the DHHS Agency for Healthcare Research and Quality (AHRQ) evidence-based AHRQ Report Effects of Omega-3 Fatty Acids on Cardiovascular Disease (Wang et al. 2004). The main conclusions of the IOM and AHRQ reports are summarized below. The Advisory Committee also identified the main conclusions of six other national and international expert groups regarding recommendations for fish consumption.

Available evidence suggests that the cardiovascular health benefits of fish consumption are related to the long chain omega-3 polyunsaturated fatty acids in fish (LC omega-3 PUFA, also called LC omega-3 PUFA), which are unique nutrients in fish. The specific omega-3 fatty acids are docosahexaenoic acid (DHA) and eicosahexaenoic acid (EPA). In the general population (rather than in heart patients), most of the studies were observational (epidemiologic) studies of fish consumption and cardiovascular disease and not studies of omega-3 fatty acid supplements (for example, see Wang et al. 2004 and FDA 2004). Therefore, although evidence indicates that the cardiovascular health benefits of fish consumption are related to the omega-3 fatty acids in fish oil, DHA and EPA, the actual studies, relevant to the general population, showing cardiovascular health benefits were for fish in the diet, not for fish oil supplements. According to the Dietary Guidelines, the current federal dietary recommendations are that most fats should come from sources of polyunsaturated and monounsaturated fatty acids, such as fish, nuts and vegetable oils. Based on the scientific review of the Advisory Committee, the Dietary Guidelines note that limited evidence suggests an association between consumption of fatty acids in fish and reduced risks of mortality from cardiovascular disease for the general population. Other sources of EPA and DHA may provide similar benefit; however, more research is needed. The Dietary Guidelines further state that "evidence suggests that consuming approximately two servings of fish per week (approximately 8 ounces total) may reduce the risk of mortality from coronary heart disease and that consuming EPA and DHA may reduce the risk of mortality from cardiovascular disease in persons who have already experienced a cardiac event." The Dietary Guidelines explain that the Food and Drug Administration and EPA are advising women of childbearing age who may become pregnant, pregnant women, nursing mothers, and young children to avoid some types of fish and shellfish and to eat fish and shellfish that are lower in mercury, and refer readers to the FDA telephone hotline and web site.

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the potentially harmful effects of mercury on blood pressure.

The scientific review by the Dietary Guidelines Advisory Committee on fish consumption used as its starting point the comprehensive Institute of Medicine report on macronutrients (IOM 2002). The IOM report found that "a growing body of literature suggests that diets high in EPA and DHA may afford some degree of protection against CHD." The Advisory Committee review also relied upon an evidence-based review by AHRQ (Wang et al. 2004). The AHRQ report evaluated 22 prospective cohort studies that were conducted in the US and other developed countries. AHRQ noted that most of the cohorts had several thousand subjects; the range was 272 to 223,170 subjects, with most subjects at least age 40. AHRQ indicated that, despite some limitations, if viewed together, these studies provide evidence that is highly applicable to the U.S. population. Overall, AHRQ found that the evidence from the primary and secondary prevention studies supports the hypothesis that the consumption of omega-3 fatty acids, fish, and fish oil reduces all-cause mortality and various cardiovascular disease (CVD) outcomes. These outcomes include sudden death and cardiac death (coronary or myocardial infarct (MI) death). Thus, the AHRQ report concluded that the consumption of omega-3 fatty acids from fish or from supplements of fish oil reduces all-cause mortality and various CVD outcomes. Additionally, the Advisory Committee based its conclusions on its own analysis of epidemiologic studies of the cardioprotective effects of fish consumption among healthy populations, such as those by Dolecek (1992), Siscovick et al. (1995), Hu et al. (2002), and Mozaffarian et al. (2003) (cited in Dietary Guidelines Advisory Committee 2004).

FDA in 2004 (FDA 2004) announced the availability of a qualified health claim for reduced risk of coronary heart disease (CHD) on conventional foods that contain EPA and DHA omega-3 fatty acids. The FDA explained that, typically, EPA and DHA omega-3 fatty acids are contained in oily fish, such as salmon, lake trout, tuna and herring. FDA also stated that these fatty acids are not essential to the diet; however, scientific evidence indicates that these fatty acids may be beneficial in reducing CHD. An example of wording of the qualified health claim as it would appear on a food label would be, "Supportive but not conclusive research shows that consumption of EPA and DHA omega-3 fatty acids may reduce the risk of coronary heart disease. One serving of [name of food] provides [x] grams of EPA and DHA omega-3 fatty acids. [See nutrition information for total fat, saturated fat and cholesterol content.]"

C.5 Conclusions

In summary:

- Studies investigating the relationship between methylmercury and cardiovascular impacts have reached different conclusions. The findings to date and the plausible biological mechanisms warrant additional research in this arena (Stern 2005; Chan and Egeland 2004).

- Some recent epidemiological studies of men suggest that methylmercury is associated with a higher risk of acute myocardial infarction, coronary heart disease and cardiovascular disease in some populations. Other recent studies have not observed this association.

- Some studies have suggested that methylmercury attenuates the beneficial effects of fish consumption. A further possible explanation is that the observed effect may be present in certain populations but is not generalizable to other populations.

- There is a significant and well-recognized body of literature documenting the cardioprotective benefits of fish consumption.

- As the science on the impact of methylmercury on the risk of cardiovascular events remains uncertain, and the weight of the evidence, in fact, supports a positive association between fish consumption and potential cardiovascular benefits, the impacts of methylmercury from fish consumption are only discussed qualitatively.

C.6 References

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Tables

Figures

APPENDIX D

NORMALIZATION OF MERCURY IN FISH TISSUE SAMPLES

This section of the appendix provides a detailed description of the procedures used to normalize the fish tissue data discussed in Section 5 using the National Descriptive Model of Mercury in Fish (NDMMF). We also provide a general examination of model performance. It should be noted that the examination of model performance is based on a normalization of the NLFA data for the years 1990 to 2002. The validation demonstrates that the NDMMF performs well with this set of data. Additional data recently became available for the year 2003 for the NLFA and for the first two years of the NLFTS after the validation was conducted. Therefore, EPA updated the normalization with the new data for application in the final benefit analysis presented in Section 10.

D.1 Methods

D.1.1 National Descriptive Model of Mercury in Fish (NDMMF)

The United States Geological Survey developed a procedure called the National Descriptive Model of Mercury and Fish Tissue (NDMMF) (Wente 2004). The NDMMF model provides a translation factor to convert a mercury concentration taken from one species/size/sample method to an estimated concentration for any other user pre-defined species/size/sample method. The model provides species/size/sample estimates for each sampling site and year. New concentrations can only be estimated by the NDMMF if at least one sample has already been taken at a particular location. In other words, predictions can only be made for locations and dates where at least one sample has already been recorded.

The NDMMF is a statistical model related to covariance. The model is designed to allow the prediction of different species, cuts, and lengths of fish for sampling events, even when those species/lengths/cuts of fish were not sampled during those sampling events.

The NDMMF models mercury concentration as a power function of fish length, i.e., $y = ax^b$, where y = mercury concentration, x = length, and a,b are parameters. Potentially, that could be done for each species at each site, but the data are far too sparse for that, and, further, he wanted to devise some way to predict mercury concentrations for a species that was not even collected at some given site. So, the assumptions are made that: (i) a universal, national value of "b" for each tissue type from each fish species, and (ii) the value of "a" at a given site at a given time is the same for all species. Thus a prediction of mercury concentration for a given tissue from a fish of an arbitrary species of length x could then be predicted by "looking up" the right value of "b" for that species and tissue and the value of "a" for that site and time. The following is the SAS code used to implement the NDMMF.

```
data Hg.nlfwaStepTwo (keep = rec dl hg leng spc event upper lower llength);
set Hg.nlfwaStepOne;
rec = rec;
dl = dl;
hg = hg;
leng = leng;
spc = spc;
event = event;
upper = log(hg*1000+1);
if DL = 1 then lower = .; else lower = upper;
if Leng > 0 then llength = log(leng+1); else delete;
run;

proc lifereg noprint data=Hg.nlfwaStepTwo outest=Hg.b6;
      Class SPC Event;
      Model (lower, upper) = SPC*Llength Event/ d=normal noint;
      output out=Hg.Mod6Pred p=pred6;
run;
quit;
```

Code filename and variable Definitions:

Hg.nlfwaStepOne is the data filename

rec	=	unique ID for each input record;
dl	=	if dl = 1 then the sampled MeHg was below detection limits;
hg	=	Sampled MeHg;
leng	=	length of the sampled fish;
spc	=	unique code for every unique species/sample method combination starting with 1 and ending at n where n = number of unique combinations; and
event	=	unique code for every unique location/date combination starting with 1 and ending at n where n = number of unique combinations.

D.2 General Examination of Model Performance

D.2.1 NDMMF Estimated Values

The 2002 NLFA was cleaned and subset according to the methods described in chapter 5 and used to calibrate the NDMMF. The output slopes and intercepts were then used to estimate every observation that was an input to the model calibration. Thus, for this section, an input of a whole 3 in. chub sampled from waterbody A on January 1, 1992 is re-predicted using the NDMMF as a whole 3 in. chub sampled from waterbody A on January 1, 1992.

The input observed data characteristics and estimated NDMMF data characteristics are very similar. The mean fish tissue concentration of the observed input data set was a 0.39, and the mean of the NDMMF estimated fish tissue concentration was 0.36. The range changed from 0 to 9 ppm, to an NDMMF estimated 0 to 5 ppm. Hg fish tissue concentration. The standard deviation of the observed data was 0.4. The standard deviation of the estimated data was 0.36. Figure D-1 graphically depicts the characteristics of the observed and NDMMF estimated data through the use of box and whisker plots. These results include censored values (where input concentrations were below detection limits).

D.2.2 Accuracy of NDMMF Estimated Values

The NDMMF was used to estimate every observation that was entered into the model. The estimated concentrations are then compared with the observed data inputs to obtain residuals (estimated – observed = residual). Table D-1 details the statistical distribution of these residuals and compares them to the observed data. Figure D-2 graphically displays the distribution of the observed, NDMMF estimated, and residual values in box and whisker plots. The red x indicates the mean of the distributions. Boxes are drawn around the 75th and 25th percentiles. The ends of the whiskers are drawn at the minimum and maximum values. Figures D-3 and D-4 are scatter plots of observed, predicted, and residual values.

Table D-1. Statistical Distribution of Residuals

	NLFA Observed Data	NDMMF Estimated	Mean of Residual (Error)
Mean	0.39	0.36	–0.02
Min	0	0	–4.80
Max	8.9	5.3	4.8
Std. Dev.	0.42	0.36	0.2

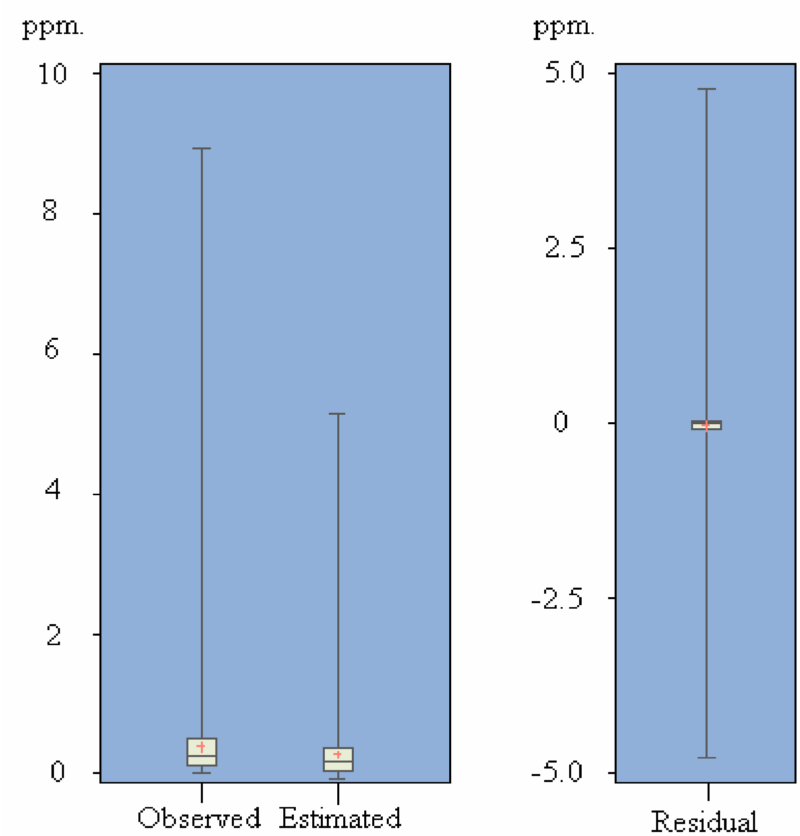

Figure D-1. Box and Whisker Plots of the NLFWA Observed, NDMMF Estimated, and Residuals Measurements in ppm

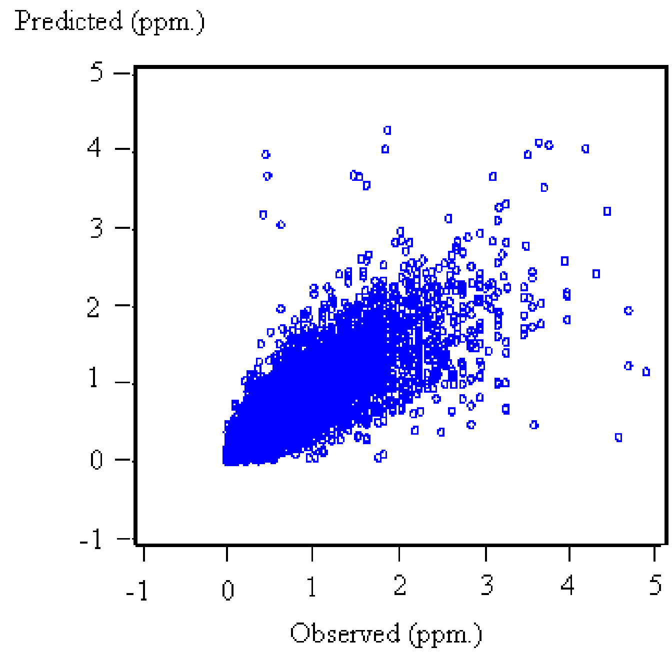

Figure D-2. Scatterplot of Predicted vs. Observed Measurements

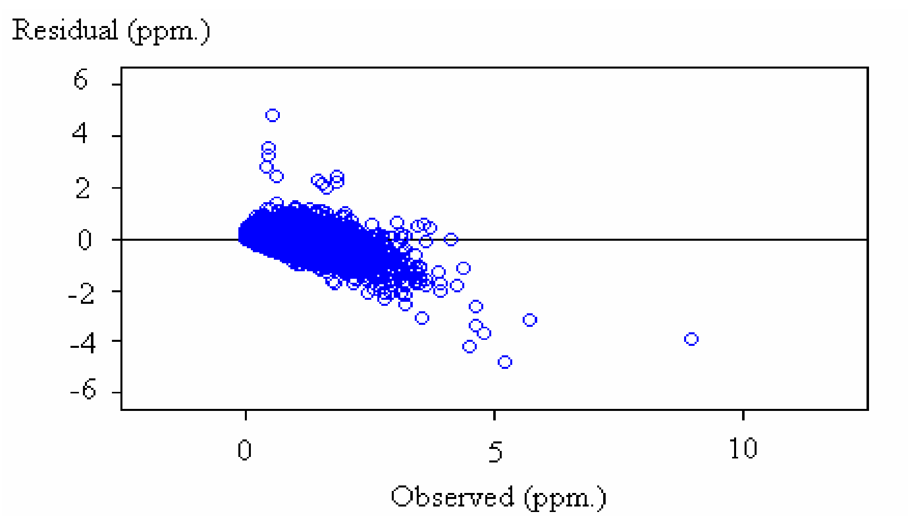

Figure D-3. Scatterplot of Residual vs. Observed Measurements

The box and whisker plots indicate that the distribution of the predicted data is a little narrower than the observed data. In both cases there are extreme high-end outliers. The distribution of the residuals from the NDMMF is centered around zero. The whiskers do indicate that there are some outliers within the residual data.

The average residual value of -0.02 is relatively small and we believe this indicates a fairly even mix between over and under predictions. The residual standard deviation of 0.2 indicates that most of the predictions are within 0.2 ppm. of the observed value. 0.2 ppm. is 51 percent of the mean fish tissue concentration. Figure D-4 indicates a systematic underestimation of higher values.

D.2.3 Spatial Examination of Model Performance

D.2.3.1 Accuracy Differences Between Lake and River Environments

The samples found within the NLFA, and used as inputs into the NDMMF, were taken from over 4,000 different locations across the U.S. These sample locations range in ecological character from Florida lakes to Wisconsin rivers.

The type of waterbody sampled was not recorded in the NLFA. To estimate the type of waterbody samples originated from, the distance to the nearest lake and nearest flowing water source was determined within a Geographic Information System (GIS). The shortest distance source waterbody type was assigned to each geocoded sample point. Approximately 1/3 of the observations in the NLFA were assigned a "lake" source type, and 2/3 were assigned "river" source type.

To examine if there is a difference in the performance or accuracy of estimates between observations originating from these two different types of sources, the observed concentrations, estimated concentrations, and residuals are reported by source type in Table D-2 below.

Table D-2. Differences between the Performance of Lake and River Samples Used as Inputs into the NDMMF

		Observed	NDMMF Estimated	Residual
Lake				
	Mean	0.36	0.33	–0.02
	Min	0	0	–3.64
	Max	5.7	2.8	1.16
	Std. Dev.	0.36	0.3	0.18
River				
	Mean	0.4	0.38	–0.02
	Min	0	0	–4.76
	Max	8.9	5.3	4.79
	Std. Dev.	0.45	0.39	0.22

An examination of the residuals shows that the NDMMF performs very similarly for lake and river source type predictions of Hg fish tissue concentrations. Any variability in model performance is not attributable to differences between rivers and lakes.

D.2.4 Predictive Examination of Model Performance (Withheld Data Set)

The examination of residuals where the prediction fish type is equal to the observed sample fish type is useful for evaluative purposes, but does not exactly parallel the planned future application of the NDMMF for data preparation for benefits analysis. To calculate benefits from Hg fish tissue samples, where the method of exposure is consumption, we must have either observations, or predictions, of Hg fish tissue concentrations found in consumable fish. For this reason, the NDMMF will be used to predict concentrations for fish types (species, length, sample method) that may not be equal to the observed sample fish type.

For example, if a 5 in. sunfish were sampled using a whole fish sample method, the NDMMF may be used to predict what the Hg fish tissue concentration would have been had the sample been taken from the fillet of an 11 in. bass. In this case, and in the application of the NDMMF for benefits analysis, the prediction fish type is not the same as the observed sample fish type.

To examine the NDMMF's performance within the context of its application for benefits analysis, where the observed and estimated fish types are likely to be different, a subset of the data (approximately 10 percent) was withheld to use as a validation dataset. The NDMMF was then implemented utilizing the remaining observations as a training dataset. The output slopes and parameter estimates were then used to predict the Hg fish tissue concentrations of the withheld data set. Figure D-4 shows the spatial distribution of withheld data observations.[1]

[1]Lack of observations throughout Pennsylvania, Kentucky, West Virginia, Tennessee, Ohio, Virginia, Kansas, and Missouri reflect to a lack of input observations in those states into the NDMMF. Filters (described in detail in chapter 5) for the most part removed these observations because no fish length or weight were recorded with the Hg concentration.

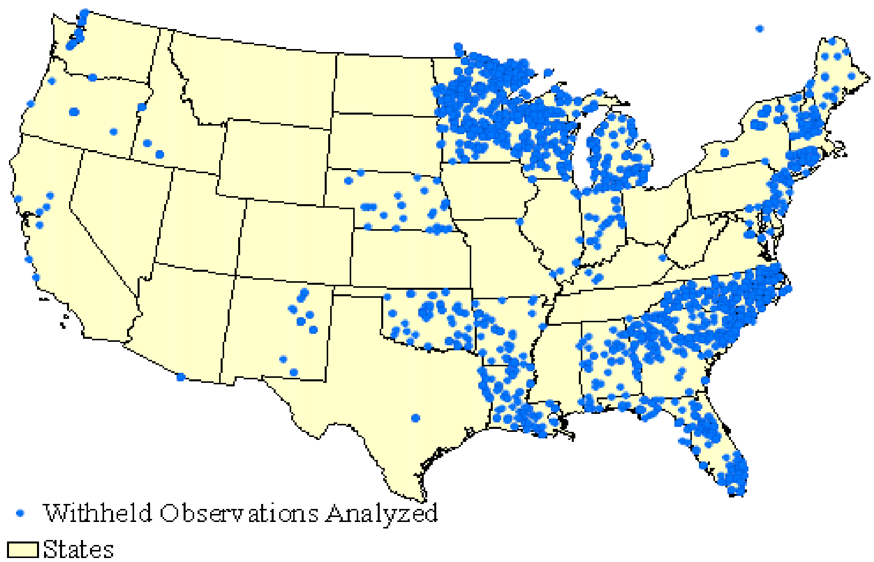

Figure D-4. Locations of Withheld Observations

The mean fish tissue concentration of the observed withheld data was 0.38 ppm. The mean predicted fish tissue concentration for the same withheld observations was 0.35 ppm. The range changed from 0 to 3.9 to 0 to 4.25. The standard deviation of the fish tissue concentrations of the withheld observed data concentrations was 0.41. The predicted concentration standard deviation was a 0.37. Table D-3 details the statistical distribution of the observed and predicted data, and the residuals. Figure D-5 graphically depicts the characteristics of the withheld observed and predicted concentrations and the residuals. Figures D-6 and D-7 are scatterplots of predicted and residual measurements vs. observed measurements for the withheld data.

Table D-3. Statistical Distribution of Residuals from Withheld Data Set

	NLFA Observed Data	NDMMF Estimated	Residual
Mean	0.39	0.35	–0.03
Min	0	0	–2.14
Max	3.9	4.25	3.05
Std. Dev.	0.41	0.37	0.24

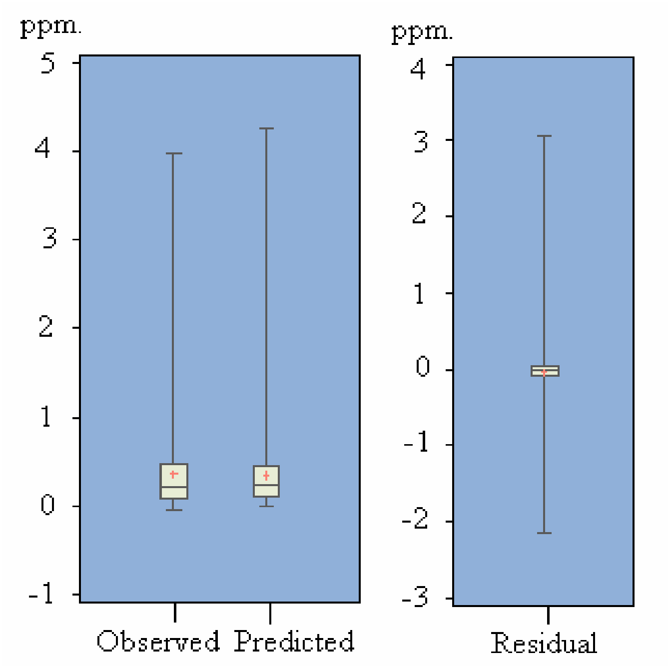

Figure D-5. Box and Whisker Plots of Observed, Predicted, and Residual (Error) Distributions for the Withheld Data Set

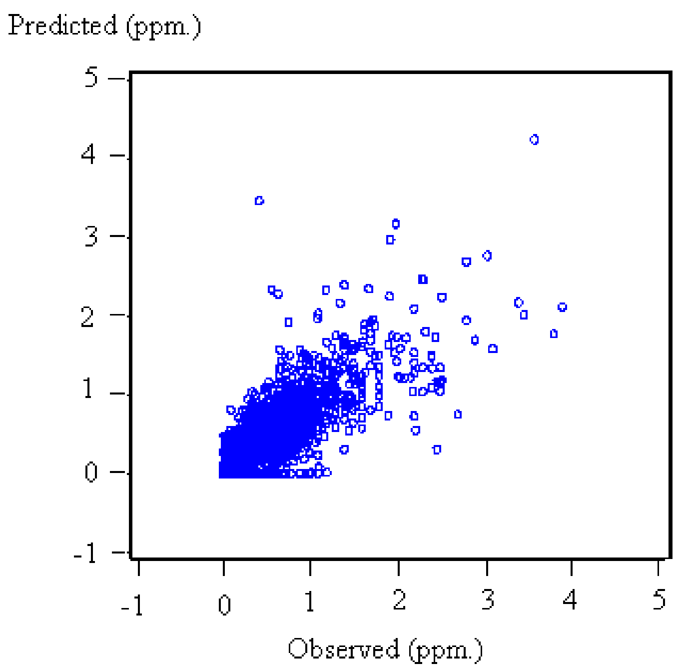

Figure D-6. Scatterplot of Predicted vs. Observed Measurements

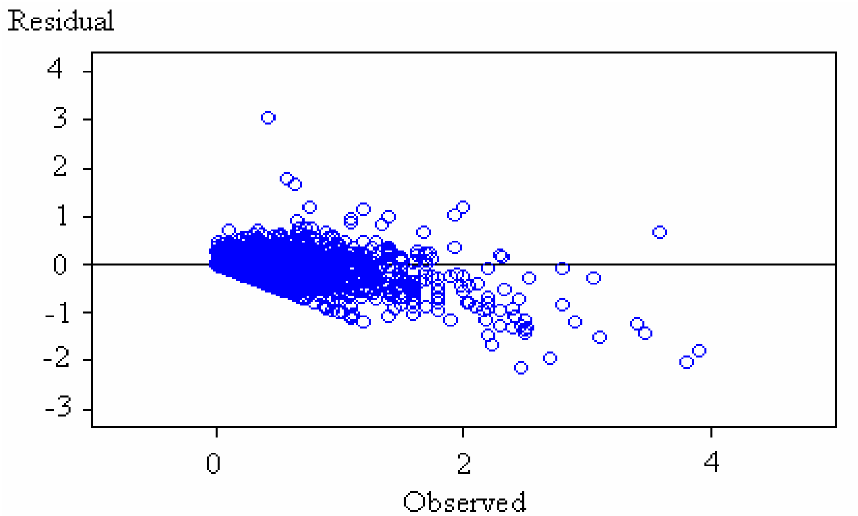

Figure D-7. Scatterplot of Residual vs. Observed Measurements

An analysis of the residuals from the withheld predicted data set shows that the average residual is –0.03 ppm. fish tissue concentration. EPA believes the closeness of the average residual to the mean, the visual analysis of the box and whisker plot, and the scatterplot of residual vs. observed measures, indicate that the model is neither systematically under or over-predicting the majority of the data. There is a slight under-prediction of higher values; however EPA believes it is small enough, and the observations are few enough, to be acceptable for benefits analysis. The range of individual observation prediction error is from –2.14 (an under-prediction) to 3.05 (an over-prediction). A residual standard deviation of 0.24 (62 percent of mean observed concentration) indicates that a large portion of the observations were predicted to within 0.24 ppm. fish tissue concentration.

It is evident that there are many issues surrounding the NLFA and sample selection, and how that influences the outputs of the NDMMF. Even given these less than ideal input data circumstances, EPA believes the NDMMF adds significantly to the value of a benefits analysis by generating a reasonably accurate estimate of Hg fish tissue concentrations from consumable fish too small for consumption or of a species not typically targeted by anglers. This enables the more complete use of the NLFA data set and allows EPA to generate benefit estimates for a larger portion of the population.

D.3 References

Wente, S.P. 2004. A Statistical Model and National Data Set for Partitioning Fish-Tissue Mercury Concentration Variation between Spatiotemporal and Sample Characteristic Effects: U.S. Geological Survey Scientific Investigations Report 2004-5199

Tables

Figures

APPENDIX E-1

ANALYSIS OF TRIP TRAVEL DISTANCE FOR FRESHWATER ANGLERS

As described in Section 10.1.2, using the population centroid approach to estimate exposures to mercury in freshwater fish requires information about how far individuals typically travel for freshwater fishing. This appendix describes the data and methods used to analyze travel distance patterns by freshwater anglers, and it reports the results that were used in applying the population centroid approach.

E1.1 Data

To conduct an analysis of trip travel distance for freshwater anglers, we used data from the NSRE 1994. As described previously, this 16,000-person survey elicited information on water-based recreation activities—specifically boating, fishing, swimming, and wildlife viewing—during the previous year. Respondents were asked about *their most recent trip* taken in each of the four categories. Of particular interest to this analysis is data concerning fishing trip characteristics for all respondents who fished in freshwater bodies during the previous year. Of the 3,220 respondents who had reported fishing, 2,482 visited either a lake, pond, river, or stream on their most recent trip.

The fishing module elicited location information about most recent fishing trip taken during the preceding 12 months. This trip was recorded as either a single- or multiday trip to a specific water body ("site") identified by the respondent. Subsequently, a series of questions were asked to gather location data on the specific site visited, including the site name, the state in which the site was located, and the name of the city or town nearest the site. To identify potential determinants of travel distance for a freshwater fishing trip, we analyzed the 2,384 available responses to the following survey question: "What was the one way travel distance, in miles from your home, to your destination on *site*?" Table E1-1 presents summary statistics for travel distance, which are reported separately for single-day, multiday, and aggregated trips. As would be expected, median travel distance varied according to trip type, from 20 miles for a single-day trip to almost 140 miles for a multiday trip. Across both trip types, the average travel distance was slightly under 100 miles.

Table E1-1. Reported Trip Travel Distance for Freshwater Anglers (miles)

	N	Min[a]	P5	P25	P50	Mean	P75	P95	Max
All trip types	2384[b]	0	2	10	20	91.9	45	125	3000
Single-day trips only	1791	0	2	10	20	41	45	125	1100
Multiday trips only	586	3	18	70	138	248.2	300	850	3000

[a] Seven respondents reported traveling 0 miles for their most recent trip; all were described as single-day trips.
[b] Seven respondents did not report whether trip was single-day or multiday.
Note: Ninety-eight respondents who visited freshwater bodies on their most recent fishing trip did not report the travel distance.

E1.2 Analysis of Travel Distance Data

The influence of multiple demographic characteristics on travel distance was tested using multivariate regression analysis. Table E1-2 reports descriptive statistics for the anglers included in this analysis. As indicated by the table, over 90 percent of the sample is white; males comprise a higher percentage of the sample (62 percent) than females. More than half the sample had completed at least some college and three-fourths of the sample reported being employed. The survey asked respondents to classify their place of residence as either rural, suburban, or urban. Approximately 40 percent described their area as rural, 37 percent as suburban, and 23 percent as urban. Respondents were assigned to a U.S. Census geographic region by matching their zip code to a corresponding state. The states were then aggregated to the appropriate Census region (http://www.census.gov/geo/www/us_regdiv.pdf). The majority of respondents resided in the South and Midwest, followed by the West and Northeast.

Table E1-3 presents additional characteristics on the demographic distribution of the sample. The average age of respondents was 38 years, while household size averaged approximately three members, with fewer than one person under the age of six. Respondents' average weekly leisure time was 28 hours. However, this varied significantly across the sample, from zero to 168 hours. In the survey, family income is reported as a categorical variable, with respondents selecting the income range that reflected family income in the previous year. The midpoint of this range was taken to produce a continuous income variable. Subsequently, this value was converted to (2000$) using the consumer price index. Median (mean) income was estimated to be $57,325 ($66,496) annually.

Table E1-2. Demographic Characteristics of Freshwater Anglers[a]

	N	Frequency
Gender	2267	62% Male
Race	2250	91% White
		4% Black
		2% Hispanic
		2% Other
Education	2262	11% Less than high school degree
		34% High school degree/equivalent
		55% Some college or more
Work status	2263	75% Employed
Geography	2237	23% Urban
		37% Suburban
		41% Rural
Region	2205	13% Northeast
		33% South
		31% Midwest
		23% West

[a] In total, 2,384 respondents reported information on trip travel distance to a freshwater destination.
Note: Values may not add to 100 percent due to rounding.

Table E1-3. Demographic Characteristics of Freshwater Anglers

	N	Mean	SD	Min	Max
Age	2245	38.4	14.5	16	92
Household size	2255	3.1	1.5	1	10
Persons $\leq$6 yrs	2270	0.3	0.7	1	5
Persons $\geq$16 yrs	2254	2.2	0.9	0	7
Weekly leisure time (hrs)	2025	27.7	23.9	0	168
Family income (2000$)	1851	66496	57324	8938	208547

Multivariate regression analysis was used to identify determinants of travel distance to freshwater fishing sites. The dependent variable in this analysis was the miles traveled to the most recent freshwater fishing site. The explanatory variables included several demographic and geographic characteristics of the respondents.

Separate regressions were conducted for the full sample (1), single-day trips only (2), and multiday trips only (3). The results are reported in Table E1-4. Family income was estimated to have a positive and highly significant effect in all three models. Dummy variables for urban and suburban location were also found to have positive and highly significant effects in all models. These results suggest that wealthier anglers and those living in or near metropolitan areas tend to travel further to fishing sites, relative to less-wealthy anglers and those living in rural areas. In models (1) and (2) dummy variables for the Midwest and West regions also had positive and highly significant effects on trip travel distance, relative to the South region. The Northeast region did not have a statistically significant effect on distance traveled. Education was estimated to be positively and significantly related to distance traveled in the first and second models. (Note that the respondent's level of education, recorded in the survey as a categorical variable, was recoded as a continuous variable for the regression analysis.) Neither age, race, nor gender had significant effects (at a 5 percent level) on travel distance in any of the models.

E1.3 Summary Results Applied in the Population Centroid Approach

Given the high significance of geographic area and family income across the regressions, nonparametric results (frequency distributions) were generated for four mutually exclusive subgroups of respondents and five travel distance categories. The result are reported in Table E1-5. Respondents were categorized into the four following groups:

- G1: family income ≥\$50,000 (in 2000 dollars) and urban or suburban resident
    - (N = 452 for single-day trips)
    - (N = 649 for single- and multiday trips)
- G2: family income ≤\$50,000 and urban or suburban resident
    - (N = 329 for single-day trips)
    - (N = 417 for single- and multiday trips)
- G3: family income ≥\$50,000 and rural resident
    - (N = 295 for single-day trips
    - (N = 376 for single- and multiday trips)
- G4: family income ≤\$50,000 and rural resident
    - (N = 309 for single-day trips)
    - (N = 386 for single- and multiday trips)

Table E1-4. OLS Regression Results for Determinants of Reported Trip Travel Distance (miles)

	(1) Full Sample (both single- and multiday trips)		(2) Single-Day Trips Only		(3) Multiday Trips Only	
Variable Description	**Coefficient**	**t-stat**	**Coefficient**	**t-stat**	**Coefficient**	**t-stat**
CONSTANT	0.6966	1.54	1.7954	3.89**	2.2493	3.26**
AGE	0.0044	1.83*	0.0011	0.44	0.001	0.28
GENDER	0.0572	0.83	0.0173	0.25	0.1446	1.39
EDUC	0.1729	2.48**	0.1552	2.21**	0.128	1.22
MINORITY	–0.0437	–0.36	0.0228	0.19	–0.1391	–0.76
FAMILY INCOME (log)	0.187	4.41**	0.0827	1.92*	0.1759	2.78**
URBAN	0.3491	3.95**	0.2799	3.12**	0.2121	1.62*
SUBURBAN	0.3422	4.48**	0.193	2.50**	0.4298	3.67**
NEAST	–0.0387	–0.36	–0.2549	–2.42**	0.1525	0.89
MIDWEST	0.3856	4.65**	0.1	1.21	0.4923	3.63**
WEST	0.6103	6.73**	0.3374	3.59**	0.3239	2.32**
	R^2 = 0.077		R^2 = 0.041		R^2 = 0.112	
	N = 1,798		N = 1,360		N = 434	

** = significant at 5 percent level.
* = significant at 10 percent level.

Table E1-5. Travel Distance Frequencies by Demographic Group (Percentage in each Distance Category)

Travel Distance (mi)	(G1) High-Income and Urban/Suburban Resident	(G2) Low-Income and Urban/Suburban Resident	(G3) High-Income and Rural Resident	(G4) Low-Income and Rural Resident
Single-day trips only (N = 1,385)				
N	(N = 452)	(N = 329)	(N = 295)	(N = 309)
Distance ≤10 mi	23%	32%	31%	34%
>10 mi to 20 mi	18%	23%	22%	24%
>20 mi to 50 mi	31%	20%	28%	26%
>50 mi to 100 mi	17%	19%	14%	11%
Distance >100 mi	11%	6%	5%	5%
Full sample (both single- and multiday trips) (N = 1,828)				
N	(N = 649)	(N = 417)	(N = 376)	(N = 386)
Distance ≤10 mi	16%	26%	24%	29%
>10 mi to 20 mi	13%	18%	18%	21%
>20 mi to 50 mi	24%	18%	25%	25%
>50 mi to 100 mi	19%	19%	16%	14%
Distance >100 mi	27%	18%	17%	11%

These categories were selected because they match categories that can be easily identified in Census data and because they split the sample into roughly similar group sizes. Travel distance was categorized into ranges reported in the first column of Table E1-5. The results are consistent with those generated from the regression analysis. Among respondents on single-day trips, the number that traveled longer distances (greater than 100 miles) increased from the low-income rural cohort (5 percent) to the higher-income urban/suburban cohort (11 percent). The same pattern holds for those taking either a single- or multiday trip. The number traveling longer distances more than doubled, from 11 percent among low-income rural respondents to 27 percent among high-income urban/suburban respondents. These results indicate higher-income urban/suburban anglers travel greater distances to freshwater destinations than lower-income urban/suburban anglers and rural anglers.

As described in Section 10.1.2, the trip frequency estimates reported in Table E1-5 for the full sample were used in the population centroid approach to weight exposures to mercury in fish according to distance from the Census Block Group centroid, income levels in the Block Group, and whether the Block Group is predominantly rural or urban/suburban.

APPENDIX E-2

METHODOLOGY FOR ESTIMATING FRESHWATER FISHING DAYS BY WATERSHED

This appendix describes how data from the NSRE and NSFHWR were used in our analysis to develop HUC-level approximations of fishing activity. These methods were developed specifically to support the angler destination approach described in Section 10.1.2 for estimating the size of exposed populations to mercury in freshwater fish.

As reported in Table 10-4, the NSFHWR provides estimates of the total number of lake- and river-fishing days *by state* (in 2001 for both resident and nonresident anglers). These state-level estimates are informative for estimating exposures through fish consumption, but they are limited by a low degree of spatial resolution. Particularly within larger states, there is likely to be considerable geographic variation in angler activity. Accounting for this variation should improve estimates of mercury exposures by anglers. Unfortunately, data comparable to the NSFHWR are not available for smaller or more homogeneous geographic units.

This appendix describes an empirically based method for distributing state-level estimates of fishing activity from the NSFHWR to the eight-digit HUCs within their boundaries. This distribution method takes into account the varying attributes across watersheds, which make them more or less likely to attract anglers. Using this approach we estimated lake- and river-fishing days for the 1,362 HUCs located in our 37-state study area. These HUCs range in size from 0.02 to 7,939 square miles, with an average (median) of 1,353 (1,186) square miles.

E-2.1 Data

To supplement the NSFHWR, the primary source of data for this analysis is the NSRE 1994. Anglers responding to NSRE 1994 provided detailed information regarding their last fishing trip, including where the fishing site was located and the number of times they visited the site over the previous year. The fishing site locations have been geocoded to identify the HUC corresponding to each trip. There are 2,111 identified HUCs in the continental United States, which are grouped into 18 regional "hydrologic units." Figure E2-1 shows the boundaries for hydrologic units in the United States.

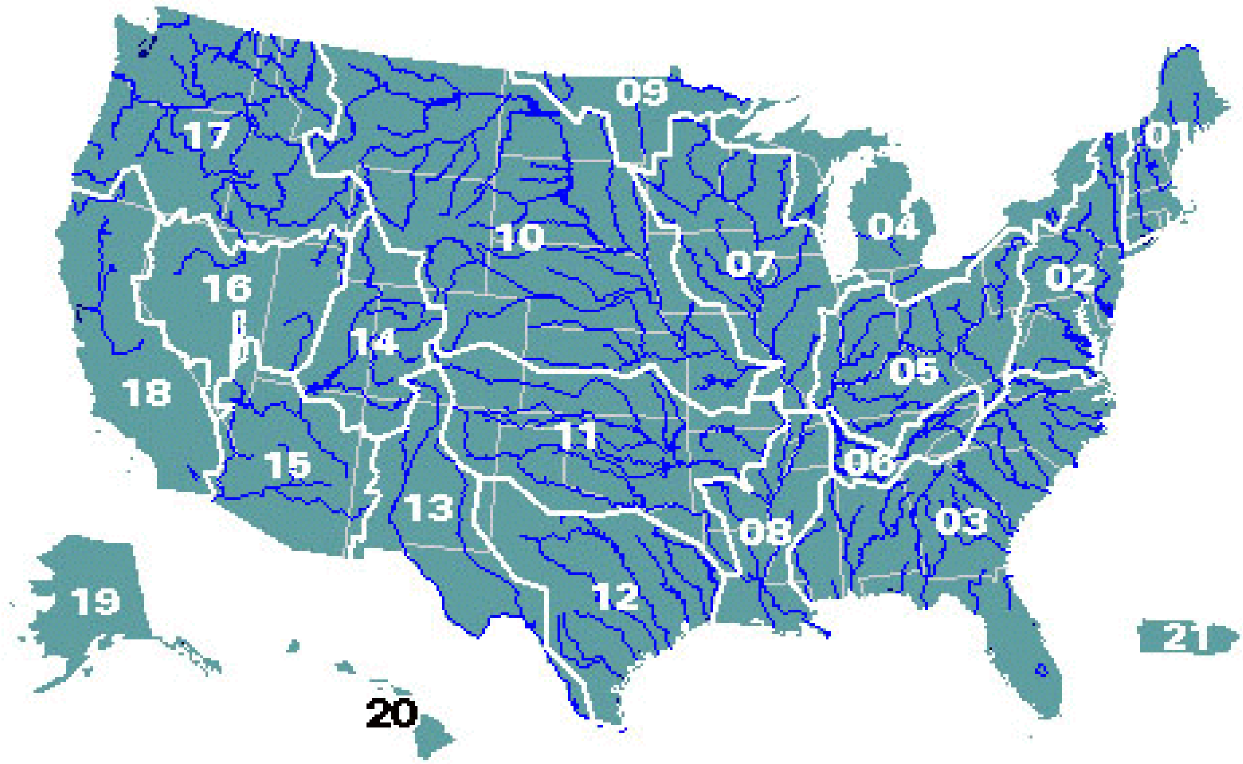

Figure E2-1. U.S. Hydrologic Regions

Source: U.S. Geological Survey. Hydrologic Unit Maps. Last update February 17, 1999, accessed March 31, 1999. <http://water.usgs.gov/public/GIS/huc.html>.

In addition to detailed information about last fishing trip destinations, NSRE 1994 contains information on the type of waterbody visited on the last trip. Waterbodies are broken down into four categories: lakes, rivers and streams, wetlands, and coastal areas. Of the 3,247 respondents who had fished in the continental United States in the previous year, 1,812 indicated that their last trip was to a lake, 694 to a river or stream, 5 to a wetland, and 598 to a coastal area. (Type of waterbody visited was not available for 141 responses.) Specific reach numbers, which include the HUC identifier, were able to be identified for 1,758 of the 2,506 noncoastal (i.e., freshwater) destinations. For example, none of the 123 trips to hydroregion 17 (Pacific Northwest) were able to be assigned a reach index, because the reach indexing system does not include this region.

After selecting only the freshwater fishing destinations from the NSRE, we grouped them by HUC. Using the freshwater last-trip destination information, we then constructed four "level-of-use" indicators. For each HUC, the first and second indicators (RESP_COUNT_LAKE and RESP_COUNT_RIV) are equal to the number of *respondents* in the survey who reported the HUC as their last-trip destination for lake and river fishing, respectively. The third and fourth indicators (TRIP_COUNT_LAKE and TRIP_COUNT_RIV) are equal to the total number of *trips* taken by respondents in the previous year to their last visited site, when this site was in the HUC. In all four cases, the observed trip frequency from the NSRE sample is taken as an indicator of aggregate visitation to each HUC.

Table E2-1 reports frequency distributions for these four indicators across 1,892 HUCs (excluding hydroregion 17). In all four cases, the number of HUCs in each level-of-use category gradually declines as the level-of-use indicator increases, which is broadly consistent with a Poisson/negative binomial distribution for this indicator.

To examine whether these level-of-use indicators are systematically related to other HUC-level characteristic, we use GIS along with geographic, Census, and EPA Reach File 3 data to create several HUC-level variables. Summary statistics and descriptions of these variables are provided in Table E2-2. These variables include continuous measures of the size of each HUC, the number of reach miles in each HUC (lake and river), and populations within 25 and 50 miles of the HUC centroid. All of these measures are expected to have a positive effect on level of use. We also included categorical variables to indicate whether a HUC is located along the coast, whether it is located along one of the Great Lakes, and which hydroregion it falls in.

To analyze the relationship between the HUC level-of-use indicators and these HUC-level characteristics, we used negative binomial regression analysis. The negative binomial is a variant of the Poisson count model. Using a simple Poisson regression model, we would assume that the probability of observing a given count (nonnegative integer value t_h) at HUC h for one of the lake or river level-of-use indicators (T_{lh} or T_{rh}) can be expressed as

$$\Pr(T_{ih} = t_h) = \frac{\exp(-\exp(\beta_i X_h))(\exp(\beta_i X_h)^{t_h}}{t_h!}$$

for i= r, l (E2.1)

Table E2-1. Frequency Distributions for HUC Level-of Use Indicators

Lake Trip Counts		River Trip Counts	
RESP_COUNT_LAKE	**Number of HUCs**	**RESP_COUNT_RIV**	**Number of HUCs**
0	1310	0	1554
1	317	1	234
2	112	2	61
3	64	3	32
4	39	4	6
5	20	5	4
6	12	6	0
7	4	7	1
8	4	8	0
9	2	9	0
10+	8	10+	0
Total	1892	Total	1892
TRIP_COUNT_LAKE	**Number of HUCs**	**TRIP_COUNT_RIV**	**Number of HUCs**
0	1312	0	1555
1	93	1	52
2	66	2	47
3	57	3	33
4	37	4	26
5	43	5	21
6	34	6	27
7	17	7	10
8	20	8	7
9	20	9	2
10	15	10	11
11–20	80	11–20	48
21–50	76	21–50	33
51–100	15	51–100	13
100+	7	100+	7
Total	1892	Total	1892

Table E2-2. Variable Definitions and Descriptive Statistics

Variables	Definitions	Mean	Standard Deviation
LAKEMILES	Number of lake shore miles (RF3)	90.87	153.17
RIVERMILES	Number of river/stream miles (RF3)	499.82	542.92
POP0_25	Population within 25 miles of HUC centroid (2000 Census)	197350	593910
POP25_50	Population between 25 and 50 miles of HUC centroid (2000 Census)	566266	1121439
HUCAREA	Surface area of the HUC (thousand square miles)	1.429	0.892
COASTHUC	=1 if HUC is located adjacent to the coast	0.1317	0.3382
GLAKEHUC	=1 if HUC is located adjacent to a Great Lake	0.0375	0.1899
H1-H18	Hydrologic region dummy variables	—	—

where X_h represents a vector of HUC-level characteristics and β_i represents a corresponding vector of coefficients. The Poisson specification assumes that the mean and variance of the distribution are the same. In contrast, the negative binomial allows the variance to differ and therefore includes an additional parameter (α), which is referred to as the overdispersion parameter. The coefficient vectors are estimated using maximum likelihood methods and are represented as $\left(\hat{\beta}_1, \hat{\alpha}_1\right)$ and $\left(\hat{\beta}_r, \hat{\alpha}_{\mathbf{r}}\right)$.

The regression results are reported in Table E2-3 for lake trips and Table E2-4 for river trips.[1] For lake use (both respondent- and trip-level indicators), as expected, the number of lake miles, size of proximate populations, geographic size, and Great Lakes proximity all have a positive and statistically significant (at a 10 percent level) effect.[2] Proximity to the coast has a negative and significant effect. For both river-use indicators, the number of river miles, size of population within 25 miles, and geographic size had positive and significant effects. For the

[1]The variable listed in the tables but not included in the regression results were found to be jointly not significantly different from zero (at a 5 percent level) in previous regressions.

[2]The lake mile and population variables were included in logarithmic form because their distributions are heavily skewed to the right.

Table E2-3. Estimated Determinants of HUC Level-of-Use Indicators for Lake Trips: Negative Binomial Regressions

	Dependent Variable:			
	RESP_COUNT_LAKE		TRIP_COUNT_LAKE	
Explanatory Variables	Coefficient	Standard Error	Coefficient	Standard Error
Log(LAKEMILES)	0.6339**	0.0426	0.7506**	0.0688
Log(POP0_25)	0.1517**	0.0429	0.1730*	0.0735
Log(POP25_50)	0.2317**	0.0535	0.3290**	0.0839
HUCAREA	0.0979*	0.0528	0.2584*	0.094
COASTHUC	–0.6371**	0.1291	–1.2226**	0.2531
GLAKEHUC	1.5940**	0.1626	2.3050**	0.4281
H1	—	—	1.1766*	0.5855
H2	—	—	1.2429*	0.5495
H3	-0.5832**	0.1699	0.9615*	0.5495
H4	—	—	0.9022	0.5717
H5	—	—	1.3917**	0.5349
H6	—	—	0.9678*	0.5556
H7	—	—	0.8245*	0.4998
H8	2.0139**	0.2983	4.3948**	0.6842
H10	—	—	1.1988*	0.5169
H11	—	—	1.2947*	0.5098
H12	—	—	0.8991	0.5606
H13	—	—	1.6459*	0.5556
H14	0.8110**	0.2377	1.1379*	0.4991
H15	1.0561**	0.308	2.0487*	0.6292
H16	—	—	1.0613	0.736
H18	1.0318**	0.205	2.8459**	0.5889
CONSTANT	–8.1065**	0.4529	–9.8803	1.0189
alpha	0.7761	0.1039	1.7149	0.064
N	1884		1884	
Pseudo R2	0.1735		0.0737	

** Significant at 1 percent level.
* Significant at 10 percent level.

Table E2-4. Estimated Determinants of HUC Level-of-Use Indicators for River Trips: Negative Binomial Regressions

	Dependent Variable:			
	RESP_COUNT_RIV		TRIP_COUNT_RIV	
Explanatory Variables	Coefficient	Standard Error	Coefficient	Standard Error
Log(RIVERMILES)	0.6265**	0.1123	0.5258**	0.1483
Log(POP0_25)	0.1371*	0.6511	0.4723**	0.0749
Log(POP25_50)	0.1519*	0.0802	—	—
HUCAREA	0.2977**	0.073	0.4323*	0.146
COASTHUC	0.2354*	0.1426	—	—
GLAKEHUC	0.6069*	0.2665	—	—
H1	—	—	—	—
H2	0.6038**	0.1841	0.7030*	0.3049
H3	—	—	0.5763*	0.33
H4	—	—	—	—
H5	—	—	0.6736*	0.3231
H6	0.6076*	0.2854	1.0691*	0.4612
H7	—	—	—	—
H8	3.8641**	0.7789	3.3431**	1.0372
H10	—	—	—	—
H11	—	—	—	—
H12	—	—	–1.1334**	0.3553
H13	0.8902*	0.4855	1.2390*	0.762
H14	0.6823*	0.3015	1.3690*	0.6687
H15	—	—	—	—
H16	—	—	1.7103*	0.9127
H18	0.4614	0.3222	—	—
CONSTANT	–9.4317**	0.8285	–8.9700**	1.0066
alpha	0.8416	0.1947	2.5765	0.0762
N	1884		1884	
Pseudo R2	0.1335		0.05	

** Significant at 1 percent level.
* Significant at 10 percent level.

respondent-level indicator, the size of the population beyond 25 miles, as well as the Great Lake and coastal dummy variables, were found to also have a positive and significant effect. In all cases, the overdispersion parameter (α) was found to be significantly different from zero, which indicates that the negative binomial model is preferable to a simple Poisson.

Using the results of the regressions reported in Tables E2-3 and E2-4, level-of-use indicators were then *predicted* for each HUC (h) and waterbody type, based on differences in HUC-level characteristics. For example, based on the regression results, the models predict relatively higher angler activity in HUCs that have more river and lake miles, larger populations within 25 and 50 miles, and larger surface area. For lakes (l) and rivers (r) respectively, we refer to these predicted values as $\hat{T}_{lh}, \hat{T}_{rh}$. From Eq. (E2.1) it follows that

$$\hat{T}_{ih} = E(T_{ih} | X_h) = \exp(\hat{\beta}_i X_h) \qquad \text{for i= r, l.} \tag{E2.2}$$

Table E2-5 provides a summary of the predicted level-of-use values.

Table E2-5. Predicted Level-of-Use Indicators for HUCs in Study Area: Negative Binomial Regression Model Predictions

Predicted Indicator	min	p1	p50	mean	p99	max
RESP_COUNT_LAKE $\hat{T}_l$	0.001	0.01	0.2788	0.6192	4.08	28.3667
TRIP_COUNT_LAKE $\hat{T}_l$	0	0	1.2264	4.3204	38.1314	578.688
RESP_COUNT_RIV $\hat{T}_r$	0	0	0.1507	0.241	1.5347	4.4308
TRIP_COUNT_RIV $\hat{T}_r$	0	0.01	0.8551	2.2631	17.8171	85.3717

The predicted-use indicators were then used as measures of the *relative* level of angler activities across HUCs within each state (s). Specifically, they were used to approximate the percentage of state-level lake- and river-fishing days that occur in each HUC in the state as follows:

$$p_{lhs} = \frac{\hat{T}_{lh}}{\sum_{hs} \hat{T}_{lh}} \tag{E2.3}$$

$$p_{rhs} = \frac{\hat{T}_{rh}}{\sum_{hs} \hat{T}_{rh}} . \tag{E2.4}$$

The denominators in each of these two expressions is the sum of predicted-use indicators across HUCs in the state.[3]

For the analysis summarized in Section 11, the predicted level-of-use indicators from the trip-based models (TRIP_COUNT_LAKE and TRIP_COUNT_RIV) were applied in Eq. (E2.3) and (E2.4) to estimate HUC-level indicators. The two models produce similar results regarding the spatial distribution of angler activity, but the trip-based model was selected because it includes more information about the frequency of specific trip destinations. In the analysis, these predicted values were truncated at the 99th percentile of their distribution (see Table E2-5) to prevent a small number of outliers from skewing the results.

To estimate *absolute* levels of angler activity in each HUC, these estimated percentages of fishing activity were combined with total state-level fishing estimates from the NSFHWR. In effect, the state-level estimates of lake- and river-fishing days by residents and nonresidents (D_{ls} and D_{rs}; see **Table** 10-4) were distributed to HUCs based on the estimated percentages. The number of lake- and river-fishing days in each HUC (in 2001) were estimated as:

$$A_{lh} = p_{lhs} * D_{ls} \tag{E2.5}$$

$$A_{rh} = p_{rhs} * D_{rs} \tag{E2.6}$$

For example, HUCs with a predicted level-of-use indicator of 3 were assumed to have three times as many angler days as HUCs with a predicted indicator of 1. The results of these calculations are summarized in Section 10.1.2.

[3]For HUCs located in more than one state, the predicted-use indicators were distributed across states in proportion to the percentage of HUC surface area located in each state.

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