

Mercury Study Report to Congress

Volume II:
An Inventory of Anthropogenic
Mercury Emissions in the
United States

SAB REVIEW DRAFT



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and
Office of Research and Development

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June 1996

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U.S. Environmental Protection Agency

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LIST OF SYMBOLS, UNITS AND ACRONYMS

AP-42	Compilation of Air Pollutant Emission Factors (U.S. EPA, 1988a)
APCD	Air Pollution Control Device
Btu	British Thermal Unit
CAA	Clean Air Act as Amended in 1990
CAS	Chemical Abstract Service
CFB	Circulized fluidized bed
CFR	Code of Federal Regulations
CIP	Carbon-in-pulp process
COC	Certification of Compliance
d	Day
dscf	Dry standard cubic foot
EEI	Edison Electric Institute
EMF	Emission modification factor
EPRI	Electric Power Research Institute
ESP	Electrostatic precipitator
FBC	Fluidized bed combustor
FF	Fabric filter
FGD	Flue gas desulfurization
FTP	Federal Test Procedure
g	Gram
GW	Gigawatt
HFET	Highway Fuel Economy Test
Hg	Mercury
HID	High Intensity Discharge
hr	Hour
ISGS	Illinois State Geological Survey
kg	Kilogram
kJ	Kilojoules
L	Liter
L&E	Locating and Estimating Document (U.S. EPA, 1993a)
lb	Pound
MB/REF	Mass burn/refractory wall
MB/RC	Mass burn/rotary waterwall
MB/WW	Mass burn/water wall
Mg	Megagram or metric ton (2200 pounds)
Mj	Megajoules
mm	Millimeter
MSW	Municipal solid waste
MW	Molecular weight
MWC	Municipal waste combustor
MWI	Medical Waste Incinerator
NEMA	National Electrical Manufacturers Association
Nm ³	Normal cubic meter
NSPS	New Source Performance Standard
NYCC	New York City Cycle
OPP	U.S. EPA Office of Pesticides Programs
OSHA	Occupational Safety and Health Administration

LIST OF SYMBOLS, UNITS AND ACRONYMS (continued)

OSW	U.S. EPA Office of Solid Waste
PM	Particulate matter
ppb	Parts per billion
ppm	Parts per million
ppmwt	Parts per million by weight
RCRA	Resource Conservation and Recovery Act
RDF	Refuse derived fuel
SDA	Spray dryer adsorber
SSI	Sewage sludge incinerator
TRI	Toxic Release Inventory
UDI	Utility Data Institute
μ mol	Micromole
USGS	United States Geological Service
VOC	Volatile Organic Compound
WDF	Waste derived fuel
yr	Year

EXECUTIVE SUMMARY

Section 112(n)(1)(B) of the Clean Air Act (CAA), as amended in 1990, requires the U.S. Environmental Protection Agency (U.S. EPA) to submit a study on atmospheric mercury emissions to Congress. The sources of emissions that must be studied include electric utility steam generating units, municipal waste combustion units and other sources, including area sources. Congress directed that the Mercury Study evaluate many aspects of mercury emissions, including the rate and mass of emissions, health and environmental effects, technologies to control such emissions and the costs of such controls.

In response to this mandate, U.S. EPA has prepared a seven-volume Mercury Study Report to Congress. This volume -- Volume II of the Report to Congress -- estimates emissions of mercury from anthropogenic sources and provides abbreviated process descriptions, control technique options, emission factors and activity levels for these sources. The information contained in this volume will be useful in identifying source categories that emit mercury, in selecting potential candidates for mercury emission reductions and in evaluating possible control technologies or materials substitution/elimination that could be used to achieve these reductions (as presented in Volume VII of this Report to Congress). The emissions data presented here also served as input data to U.S. EPA's local impact analyses and long-range transport model that assessed the dispersion of mercury emissions nationwide (as presented in Volume III of this Report to Congress).

Overview of Sources

In the CAA, Congress directed U.S. EPA to examine sources of mercury emissions, including electric utility steam generating units, municipal waste combustion units and other sources, including area sources. The U.S. EPA interpreted the phrase "... and other sources..." to mean that a comprehensive examination of mercury sources should be made and to the extent data were available, air emissions should be quantified. This report describes in some detail various source categories that emit mercury. In many cases, a particular source category is identified as having the potential to emit mercury, but data are not available to assign a quantitative estimate of emissions. The U.S. EPA's intent was to identify as many sources of mercury emissions to the air as possible and to quantify those emissions where possible.

The mercury emissions data that are available vary considerably in quantity and quality between different source types. Not surprisingly, the best available data are for source categories that U.S. EPA has examined in the past or is currently studying.

Sources of mercury emissions in the United States are ubiquitous. To characterize these emissions, the type of mercury emission is defined as either:

- *Natural mercury emissions* -- the mobilization or release of geologically bound mercury by natural processes, with mass transfer of mercury to the atmosphere;
- *Anthropogenic mercury emissions* -- the mobilization or release of geologically bound mercury by human activities, with mass transfer of mercury to the atmosphere; or
- *Re-emitted mercury* -- the mass transfer of mercury to the atmosphere by biologic and geologic processes drawing on a pool of mercury that was deposited to the earth's surface after initial mobilization by either anthropogenic or natural activities.

Anthropogenic mercury emissions can be further divided into area and point sources. Anthropogenic area sources of mercury emissions are sources that are typically small and numerous and usually cannot be readily located geographically. For the purpose of this report, mobile sources are included in the area source discussion. Point sources are those anthropogenic sources that are associated with a fixed geographic location. These point sources are further divided into combustion, manufacturing and miscellaneous source categories. Particular types of sources that fall into these various groups and that were examined in this study are outlined in Table ES-1.

A prerequisite for developing strategies for reducing mercury concentrations in surface waters, biota and ambient air is a comprehensive characterization of all sources of mercury releases to the environment. This would include a review not only of airborne emissions, but also direct discharges to surface water and soil as well as past commercial and waste disposal practices (e.g., historical applications of mercury-containing pesticides and fungicides that are presently banned) that have resulted in mercury contamination of different environmental media. Although the focus of this study is on air emissions in accordance with section 112(n) of the CAA, U.S. EPA recognizes that such past and current releases of mercury to other media can be important contributors to overall mercury loadings and exposures in some locations.

Moreover, a complete characterization of air emissions would include the identification of all significant mercury emission sources, both anthropogenic and natural, and would account for re-emitted mercury. The current state of knowledge about mercury emissions, however, does not allow for an accurate assessment of either natural or re-emitted mercury emissions. For example, approximately one-third of total current global mercury emissions are thought to cycle from the oceans to the atmosphere and back again to the oceans, but a major fraction of the emissions from oceans consists of recycled anthropogenic mercury. It is believed that much less than 50 percent of the oceanic emission is from mercury originally mobilized by natural sources. Similarly, an unknown but potentially large fraction of terrestrial and vegetative emissions consists of recycled mercury from previously deposited anthropogenic and natural emissions (Expert Panel, 1994).

Given the considerable uncertainties regarding the levels of natural and re-emitted mercury emissions, this report focuses only on the nature and magnitude of mercury emissions from anthropogenic sources. Further study is needed to determine the importance of natural and re-emitted mercury.

Approach for Estimating Anthropogenic Emissions

For most anthropogenic source categories, an emission factor-based approach was used to develop both facility-specific estimates for modeling purposes and nationwide emission estimates. This approach requires an emission factor, which is a ratio of the mass of mercury emitted to a measure of source activity.¹ It also requires an estimate of the annual nationwide source activity level. Examples of measures of source activity include total heat input for fossil fuel combustion and total raw material used or product generated for industrial processes. Emission factors are generated from emission test data, from engineering analyses based on mass balance techniques, or from transfer

¹ The emission factors used in developing this mercury emissions inventory are generally consistent with those presented in the U.S. EPA document entitled Locating and Estimating Air Emissions from Sources of Mercury and Mercury Compounds (EPA/454/R-93-023) September 1993. (Some of the nationwide emission estimates may vary slightly between the two documents because this report uses the most recently available data, whereas the emission factor document mentioned above is based on a baseline year of 1990.)

Table ES-1
Sources of Anthropogenic Mercury Emissions Examined in this Inventory

Area	Point		
	Combustion	Manufacturing	Miscellaneous
Electric lamp breakage	Utility boilers	Chlor-alkali production	Oil shale retorting
Paints use	Commercial/industrial boilers	Lime manufacturing	Mercury catalysts
Laboratory use	Residential boilers	Primary mercury production	Pigment production
Dental preparations	Municipal waste combustors	Mercury compounds production ^a	Explosives manufacturing ^a
Mobile sources ^a	Medical waste incinerators	Battery production	Geothermal power plants
Agricultural burning ^a	Sewage sludge incinerators	Electrical apparatus manufacturing	Turf products
Landfills ^a	Hazardous waste combustors	Carbon black production	
Sludge application ^a	Wood-fired boilers	Byproduct coke production ^a	
	Residential woodstoves ^a	Primary copper smelting	
	Crematories	Cement manufacturing	
		Primary lead smelting	
		Petroleum refining ^a	
		Instrument manufacturing	
		Secondary mercury production	
		Zinc mining ^a	
		Fluorescent lamp recycling	

^a Potential anthropogenic sources of mercury for which emissions were not estimated.

of information from comparable emission sources. Emission factors reflect the "typical control" achieved by the air pollution control measures applied across the population of sources within a source category.

The emission factor-based approach does not generate exact emission estimates. Uncertainties are introduced in the estimation of emission factors, control efficiencies and the activity level measures. Ideally, emission factors are based on a substantial quantity of data from sources that represent the source category population. For trace pollutants like mercury, however, emission factors are frequently based on limited data that may not have been collected from representative sources. Changes in processes or emission measurement techniques over time may also result in biased emission factors. Emission control estimates are also generally based on limited data; as such, these estimates are imprecise and may be biased. Further uncertainty in the emission estimates is added by the sources of information used on source activity levels, which vary in reliability. Table ES-2 presents anthropogenic source categories for which U.S. EPA had sufficient data to estimate national emissions.

Anthropogenic Emissions Summary

Table ES-3 summarizes the estimated national mercury emission rates by source category. While these emission estimates for anthropogenic sources have important limitations, they do provide insight into the relative magnitude of emissions from different groups of sources. Table ES-4 shows the distribution of estimated emissions among the four major classes of anthropogenic emission sources (area sources, combustion point sources, manufacturing point sources, and miscellaneous point sources). Tables ES-5 through ES-8 illustrate the distributions among individual source categories for these four classes, along with the date of underlying data, the degree of uncertainty, and the basis for the emission estimates. All of these emissions estimates should be regarded as best estimates given available data.

Of the estimated 220 Megagrams (Mg) (243 tons) of mercury emitted annually into the atmosphere by anthropogenic sources in the United States, approximately 85 percent is from combustion point sources, 13 percent is from manufacturing point sources, 1 percent is from miscellaneous sources and 1 percent is from area sources. Four specific source categories account for approximately 83 percent of the total anthropogenic emissions--medical waste incineration (27 percent), municipal waste combustion (23 percent), utility boilers (21 percent), and commercial/industrial boilers (12 percent). It should be noted that the U.S. EPA has finalized mercury emission limits for municipal waste combustors, and has proposed mercury emission limits for medical waste incinerators. These emission limits will reduce mercury emissions from these sources by 90 percent.

All four of the most significant sources represent high temperature waste combustion or fossil fuel processes. For each of these operations, the mercury is present as a trace contaminant in the fuel or feedstock. Because of its relatively low boiling point, mercury is volatilized during high temperature operations and discharged to the atmosphere with the exhaust gas.

For the long-range transport analysis, the emissions inventory was mapped for the continental U.S. The continental U.S. was divided into 40-km square grid cells and the magnitude of the mercury emissions were calculated for each cell. For the most part, the location (at least to the city level) of the mercury point sources described in this document were known.

Table ES-2
Anthropogenic Mercury Sources With Sufficient
Data to Estimate National Emissions

Area	Point		
	Combustion	Manufacturing	Miscellaneous
Electric lamp breakage	Utility boilers	Chlor-alkali production	Geothermal power plants
Laboratory use	Commercial/industrial boilers	Cement manufacturing	
Dental preparation	Residential boilers	Battery production	
	Municipal waste combustors	Electrical apparatus manufacturing	
	Medical waste incinerators	Instrument manufacturing	
	Sewage sludge incinerators	Secondary mercury production	
	Wood-fired boilers	Carbon black production	
	Hazardous waste combustors ^a	Primary lead smelting	
	Crematories	Primary copper smelting ^b	
		Lime manufacturing	
		Fluorescent lamp recycling	

^a Emissions in 1995 were estimated for hazardous waste incinerators and lightweight aggregate kilns; however, these 1995 estimates were not used in any of the modeling analyses.

^b Estimates were made for one source only. This source ceased operations in February 1995.

Table ES-3
Best Point Estimates of National Mercury Emission Rates by Category

Source of mercury ^a	1990-1993 Mg/yr ^{b,c}	1990-1993 tons/yr ^{b,c}	% of Total Inventory
Area sources	2.8	3.1	1.3
Lamp breakage	1.4	1.5	0.6
General lab use	0.7	0.8	0.3
Dental prep and use	0.7	0.8	0.3
Mobile sources	d	d	d
Paint use	e	e	e
Agricultural burning	d	d	d
Landfills	d	d	d
Point sources	217.3	239.4	98.7
Combustion sources	186.9	205.9	84.9
MWIs ^f	58.8	64.7	26.7
MWCs	50	55	22.7
Utility boilers	46.5	51.3	21.2
Coal	(46.3) ^g	(51)	(21.0)
Oil	(0.23)	(0.25)	(0.1)
Natural gas	(0.002)	(0.002)	(0.0)
Commercial/industrial boilers	26.3	29	12.0
Coal	(20.7)	(22.8)	(9.4)
Oil	(5.5)	(6.0)	(2.5)
Residential boilers	3.2	3.5	1.4
Coal	(0.5)	(0.6)	0.2
Oil	(2.7)	(3.0)	(1.2)
SSIs	1.7	1.8	0.7
Crematories	0.4	0.4	0.2
Wood-fired boilers ^h	0.3	0.3	0.1
Hazardous waste combustors ⁱ	d	d	d
Manufacturing sources	29.1	32	13.2
Primary lead	8.2	9.0	3.7
Secondary Hg production	6.7	7.4	3.1
Chlor-alkali	5.9	6.5	2.7
Portland cement	5.9	6.5	2.7
Primary copper ^j	0.6	0.7	0.3
Lime manufacturing	0.6	0.7	0.3
Electrical apparatus	0.42	0.46	0.2
Instruments	0.5	0.5	0.2
Carbon black	0.23	0.25	0.1
Fluorescent lamp recycling	0.005	0.006	0.002
Batteries	0.02	0.02	0.0
Primary Hg production	d	d	d
Mercury compounds	d	d	d
Byproduct coke	d	d	d
Refineries	d	d	d
Miscellaneous sources	1.3	1.4	0.6
Geothermal power	1.3	1.4	0.6
Turf products	e	e	e
Pigments, oil, etc.	e	e	e
TOTAL	220.1	242.5	100.0

^a MWC = Municipal waste combustor; MWI = medical waste incinerator; SSI = sewage sludge incinerator.

^b Numbers do not add exactly because of rounding.

^c Where available, emissions estimates for 1995 are discussed in the text. However, these 1995 estimates were not used in any of the modeling analyses.

^d Insufficient information to estimate 1990 emissions.

^e Mercury has been phased out of use.

^f In the course of an MWI rulemaking, with the receipt of new data, U.S. EPA expects to revise the mercury emission estimate for MWIs downward.

^g Parentheses denote subtotal within a larger point source category.

^h Includes boilers only; does not include residential wood combustion (wood stoves).

ⁱ In 1995 incinerators and lightweight aggregate kilns (not cement kilns) were estimated to emit 5.0 tons of mercury.

^j 1990 emissions are estimated for only one source, which ceased operations in February 1995. The nationwide estimate for 1995 is 0.08 tons.

**Table ES-4
Best Point Estimates of Mercury Emissions from Anthropogenic Sources: 1990-1993**

Source	Emissions		
	Mg/yr	Tons/yr	% of Total Inventory
Area	2.8	3.1	1.3
Combustion	186.9	205.9	84.9
Manufacturing	29.1	32.0	13.2
Miscellaneous	1.3	1.4	0.6
Total Inventory	220.1	242.5	100

Table ES-5
Best Point Estimates of Mercury Emissions from Anthropogenic Area Sources: 1990-1993

Source	Emissions			Date of Data ^a	Degree of Uncertainty ^b	Basis for Emissions Estimate
	Mg/yr	Tons/yr	% of total			
Electric lamp breakage	1.4	1.5	0.6	1989/1989	High	Industry estimate for this source category is 0.18 tons/year; this difference is explained in Section 3.2
Laboratory use	0.7	0.8	0.3	1973/1992	High	Engineering judgment
Dental preparations	0.7	0.8	0.3	1980/1992	High	Engineering judgment
Mobile sources	-	-	-	-	-	Insufficient information to estimate emissions
Paints use	-	-	-	-	-	Mercury phased out of paint use in 1991
Agricultural burning	-	-	-	-	-	Insufficient information to estimate national emissions; one study estimates 0.036 Mg/yr (0.04 tons/yr) from preharvest burning of sugarcane in Florida everglades area
Landfills	-	-	-	-	-	Insufficient information to estimate emissions
Total	2.8	3.1	1.3			

^a Date that data emission factor is based on/Date of activity factor to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

Table ES-6
Best Point Estimates of Mercury Emissions from Anthropogenic Combustion Point Sources: 1990-1993

Source	Emissions			Date of Data ^a	Degree of Uncertainty ^b	Basis for Emissions Estimate
	Mg/yr	Tons/yr	% of total			
Medical waste incinerators ^c	58.8	64.7	26.7	1983-92/1991	Medium	Test data
Municipal waste combustors	50	55	22.7	1986-92/1991	Medium	Test data; 1995 emissions estimate is 29 tons/year
Utility boilers - coal - oil - natural gas	46.5 (46.3) ^d (0.23) (0.002)	51.3 (51) (0.25) (0.002)	21.2	1994/1990	Medium	Test data; industry (Electric Power Research Institute) estimates are 44 tons/year for coal-fired utilities
Commercial/Industrial boilers - coal - oil	26.3 (20.7) (5.5)	29 (22.8) (6.0)	12.0	^e /1992	High	Mass balance; emissions may be overstated because emission factor assumes no control
Residential boilers - coal - oil	3.2 (0.5) (2.7)	3.5 (0.6) (3.0)	1.4	^e /1991	High	Mass balance; emissions may be overstated because emission factor assumes no control
Sewage sludge incinerators	1.7	1.8	0.7	1975-89/1990	High	Test data
Crematories	0.4	0.4	0.2	1992/1991	High	Engineering judgment
Wood-fired boilers ^f	0.3	0.3	0.1	1984-92/1980	Medium	Test data
Hazardous waste combustors	-	-	-	-	-	1995 emissions estimate is 5.0 tons/year for incinerators and light-weight aggregate kilns only
Total	186.9	205.2	84.9			

^a Date that data emission factor is based on/date of activity factor used to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

^c In the course of an MWI rulemaking, with the receipt of new data, U.S. EPA expects to revise the mercury emission estimate for MWIs downward.

^d Parentheses denote subtotal within a larger point source category.

^e Date of data used to develop emission factor was not determined.

^f Does not include residential emissions.

Table ES-7
Best Point Estimates of Mercury Emissions from Anthropogenic Manufacturing Sources: 1990-1993

Source	Emissions			Date of Data ^a	Degree of Uncertainty ^b	Basis for Emission Estimate
	Mg/yr	Tons/yr	% of total			
Primary lead smelting	8.2	9	3.7	1985/1990	High	Test data; 1995 emissions estimate is 1.5 tons/yr
Secondary mercury production	6.7	7.4	3.1	1973/1991	High	Test data
Chlor-alkali production	5.9	6.5	2.7	1991/1991	Medium	Section 114 industry survey responses
Cement manufacturing	5.9	6.5	2.7	1992/1990	Medium	Test reports; Industry estimates for this source category are 3.3 tons/yr; see Section 4.2.4
Primary copper smelting ^c	0.6	0.7	0.3	1992/1992	High	Test report; 1995 national emissions for currently operating smelters are 0.08 tons/yr
Lime manufacturing	0.6	0.7	0.3	1983/1992	High	Test data and mass balances
Electrical apparatus manufacturing	0.42	0.46	0.2	1973/1992	High	Engineering judgment
Instrument manufacturing	0.5	0.5	0.2	1973/1992	High	Survey questionnaire responses
Carbon black production	0.23	0.25	0.1	1980/1991	High	
Fluorescent lamp recycling	0.005	0.006	0.002	1993/1993	High	Test data and mass balances
Battery production	0.02	0.02	0.008	1986/1992	High	Engineering judgment
Primary mercury production	-	-	-	-	-	Insufficient data to estimate emissions
Mercury compounds production	-	-	-	-	-	Insufficient data to estimate emissions
Byproduct coke production	-	-	-	-	-	Insufficient data to estimate emissions
Petroleum refining	-	-	-	-	-	Insufficient data to estimate emissions
Total	29.1	32	13.2			

^a Date that data emission factor is based on/Date of activity factor used to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

^c Emissions are estimated for only 1 source, which ceased operation in February 1995; 1995 emissions reflect currently operating copper smelters.

Table ES-8
Best Point Estimates of Mercury Emissions from Miscellaneous Anthropogenic Emission Point Sources: 1990-1993

Source	Emissions			Date of Data ^a	Degree of Uncertainty ^b	Basis for Emissions Estimate
	Mg/yr	Tons/yr	% of Total			
Geothermal power plants	1.3	1.4	0.6	1977/1992	High	Test data
Turf products	-	-	-	-	-	No active registrations in the U.S. of mercury-containing turf products
Pigment production	-	-	-	-	-	No sources in U.S.
Oil shale retorting	-	-	-	-	-	No sources in U.S.
Mercury catalysts	-	-	-	-	-	Insufficient information to estimate emissions
Explosives manufacturing	-	-	-	-	-	No sources in U.S.
Total	1.3	1.4	0.6			

^a Date that data emission factor is based on/Date of activity factor used to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

Figure ES-1 illustrates the spatial distribution of mercury emissions across the U.S. based on this inventory. This distribution formed the basis of the long-range transport modeling and the resulting predictions of wet and dry deposition across the U.S.

Accuracy of the Inventory

The accuracy of the emission estimates is obviously a factor in assessing the inventory's usefulness for its intended purposes. Considering the admitted gaps in the inventory, the external peer review panel that reviewed this work concluded that the missing sources could contribute as much as 20 percent more mercury emissions to the U.S. total. For comparison, one reviewer submitted data on the amount of mercury emitted per person in some European countries (based on anthropogenic emissions only).

Based on the inventory presented in this document, the U.S. inventory represents 0.86 g mercury per person per year. Based on data submitted during the peer review process, 0.90 g mercury per person per year is emitted in the United Kingdom. In Germany (Western area), 0.75 g mercury per person per year is emitted. In Poland, 0.88 g mercury per person per year is estimated to be emitted. The European emission average is about 1.2 g mercury per person per year (Pacyna, 1995). The similarity between the U.S. inventory and other countries where coal is also the major source of energy lends credibility to the nationwide estimate presented in this report for the U.S.

Trends in Mercury Emissions

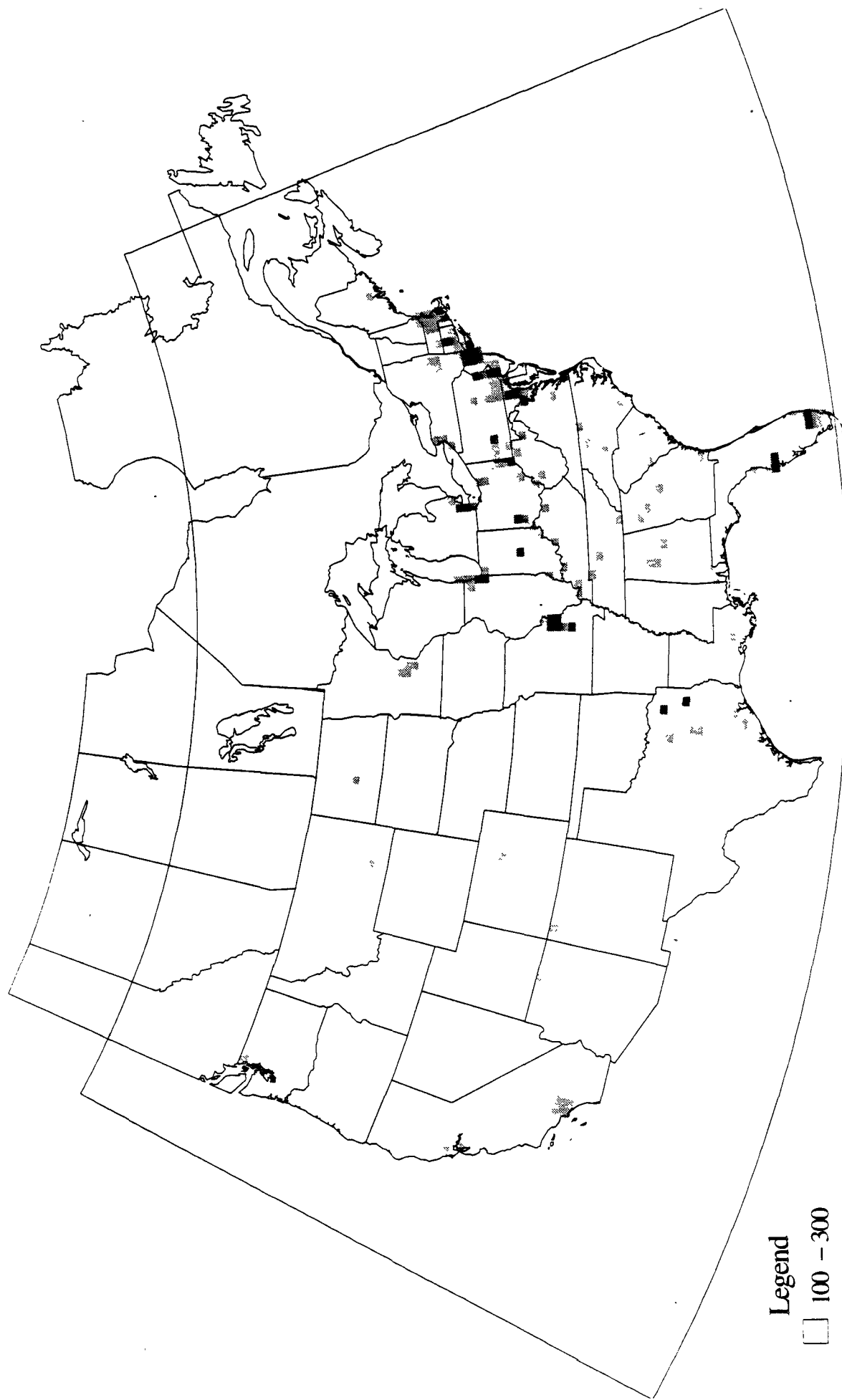
It is difficult to predict with certainty the temporal trends in mercury emissions for the U.S., although there appears to be a trend toward decreasing total mercury emissions from 1990 to 1995. This is particularly true for the combustion sources wherein mercury is a trace contaminant of the fuel. Also, as previously noted, there are a number of source categories where there is insufficient data to estimate current emissions let alone potential future emissions. Based on available information, however, a number of observations can be made regarding mercury emission trends from source categories where some information is available about past activities and projected future activities.

There has been a real success in the U.S. in the dramatic drop in mercury emissions from manufacturing over the past decade. Current emissions of mercury from manufacturing sources are generally low (with the exception of chlor-alkali plants using the mercury cell process). The emissions of mercury are more likely to occur when the product is broken or discarded. Therefore, in terms of emission trends, one would expect that if the future consumption of mercury remains consistent with the 1993 consumption rate, emissions from most manufacturing sources would remain about the same.

For industrial or manufacturing sources that use mercury in products or processes, the overall consumption of mercury is generally declining. Industrial consumption of mercury has declined by about two thirds between 1988 (1508 Mg) and 1993 (558 Mg). Much of this decline can be attributed to the elimination of mercury as a paint additive (20 percent) and the reduction of mercury in batteries (36 percent). Use of mercury by other source categories remained about the same between 1988 and 1993.

Secondary production of mercury (i.e., recovering mercury from waste products) has increased significantly over the past few years. Of the 558 Mg of mercury used in industrial processes in 1993, 63 percent was provided by secondary mercury producers. This is a two-fold increase since 1991. The number of secondary mercury producers is expected to increase as more facilities open to recover

Figure
Annual Mercury Emissions from Sources, All Species
Units: kg/y



mercury from fluorescent lamps and other mercury-containing products (e.g., thermostats). As a result there is potential for mercury emissions from this source category to increase.

The largest identifiable sources of mercury emissions currently are municipal waste combustors and medical waste incinerators. Emissions from these source categories are expected to decline significantly by the year 2000 due to regulatory action the U.S. EPA is taking under the statutory authority of section 129 of the CAA. As described in sections 4.1.1 and 4.1.2 of this document, the U.S. EPA has finalized rules for municipal waste combustors and proposed rules for medical waste incinerators that will reduce mercury emissions from both of these source categories by about 90 percent. In addition to this federal action, a number of states (including Minnesota, Florida and New Jersey) have implemented mandatory recycling programs to reduce mercury-containing waste, and some states have regulations that impose emission limits that are lower than the federal regulation. These factors will reduce national mercury emissions from these source categories even further.

After municipal solid waste and medical waste incinerators have been controlled, the largest remaining identified source of mercury emissions will be fossil fuel combustion by utility boilers, particularly coal combustion. Future trends in mercury emissions from this source category are largely dependent on both the nation's future energy needs and the fuel chosen to meet those needs. Another factor is the nature of actions the utility industry may take in the future to meet air quality requirements under the Clean Air Act.

Conclusions

The following conclusions are presented in approximate order of degree of certainty in the conclusion, based on the quality of the underlying database. The conclusions progress from those with greater certainty to those with lesser certainty.

- Numerous industrial and manufacturing processes emit mercury to the atmosphere. Mercury emissions from U.S. manufacturing sources, however, have dropped dramatically over the past decade.
- Prior to 1995, municipal waste combustors and medical waste incinerators were the largest identifiable source of mercury emissions to the atmosphere. Regulations finalized for municipal waste combustors and proposed medical waste incinerators will reduce emissions from these source categories by 90 percent.
- After emissions from municipal solid waste combustors and medical waste incinerators have been reduced, combustion of fossil fuels, particularly coal, will be the largest remaining source of mercury emissions to the atmosphere.
- Mercury is emitted, to a varying degree, from anthropogenic sources virtually everywhere in the United States.
- Natural sources of mercury and re-emission of previously deposited mercury are also sources of mercury to the atmosphere, although the magnitude of the contribution of these sources relative to the contribution of current anthropogenic sources is not well understood.
- Anthropogenic sources in the United States emit approximately 220 Mg (243 tons) of mercury annually into the atmosphere. This estimate is believed to be accurate to

within 30 percent. This estimate represents emissions calculated during the 1990-1993 timeframe. Emission estimates for 1995 are about 40 tons lower.

- In the United States, land areas east of the Rocky Mountains have the highest concentration of emissions from anthropogenic sources in the U.S.
- The land areas having the greatest concentration of mercury emissions from anthropogenic sources of total mercury (i.e., all chemical species) are the following: the urban corridor from Washington D.C. to Boston, the Tampa and Miami areas of Florida, the larger urban areas of the Midwest and Ohio Valley and two sites in northeastern Texas.
- The land areas having generally the lowest emissions are in the Great Basin region of the western United States and the High Plains region of the central United States. There are generally few large emission sources in the western third of the United States, with the exception of the San Francisco and Los Angeles areas and specific industrial operations.

There are many uncertainties in the emission estimates for individual source categories due to uncertainties inherent in an emission factor approach. The source of these uncertainties include the following:

- Variability in the estimates of source activity for each source category. Activity levels used in this Report were compiled over different time periods and by a variety of survey procedures.
- Emissions test data that are of poor quality or are based on very few analyses, which may not be representative of the full source population being studied.
- Changes in processes or emission measurement techniques over time (especially since about 1985). Earlier techniques may have measured too much mercury because of contamination problems.
- A lack of data for some source categories which either led to estimates based on engineering judgment or mass balance calculations. For a number of source categories there were insufficient data and, thus, no emissions estimates were made.
- Limited data on the effectiveness of air pollution control equipment to capture mercury emissions.

Understanding the public health and environmental impacts of current anthropogenic emissions is complicated by an incomplete understanding of the following factors:

- Global and transboundary deposition of mercury and the impact this has on deposition of mercury in the U.S.
- The magnitude and chemical nature of natural emissions.
- The magnitude and chemical nature of re-emitted mercury.

- The public health and environmental impacts of emissions from past uses of mercury (such as paint application) relative to current anthropogenic emissions.

To improve the emissions estimates, U.S. EPA would need the following:

- Source test data from a number of source categories that have been identified in this volume as having insufficient data to estimate emissions. Notable among these are mobile sources, landfills, agricultural burning, sludge application, coke ovens, petroleum refining, residential woodstoves, mercury compounds production and zinc mining.
- Improvements in the existing emissions information for a number of source categories including secondary mercury production (i.e., recycling), commercial and industrial boilers, electric lamp breakage, iron and steel manufacturing and primary lead smelting.
- Development and validation of a stack test protocol for speciated mercury emissions.
- More data on the efficacy of coal cleaning and the potential for slurries from the cleaning process to be a mercury emission source.
- More data are needed on the mercury content of various coals and petroleum and the trends in the mercury content of coal burned at utilities and petroleum refined in the U.S.
- Additional research to address the potential for methylmercury to be emitted (or formed) in the flue gas of combustion sources.
- Investigation of the importance (quantitatively) of re-emission of mercury from previously deposited anthropogenic emissions and mercury-bearing mining waste. This would include both terrestrial and water environments. Measuring the flux of mercury from various environments would allow a determination to be made of the relative importance of re-emitted mercury to the overall emissions of current anthropogenic sources.
- Determination of the mercury flux from natural sources to help determine the impact of U.S. anthropogenic sources on the global mercury cycle as well as the impact of all mercury emissions in the United States.
- More detailed emissions data to support the use of more sophisticated fate and transport models for mercury; in particular, more information is needed on the chemical species of mercury being emitted (including whether these species are particle-bound) and the temporal variability of the emissions.

Based on trends in mercury use and emissions, the U.S. EPA predicts the following:

- A significant (90 percent) decrease will occur in mercury emissions from municipal waste combustors and medical waste incinerators when the regulations put forth by U.S. EPA for these source categories are fully implemented.

- Manufacturing use of mercury will continue to decline with chlorine production from mercury cell chlor-alkali plants continuing to account for most of the mercury use in the manufacturing sector.
- Secondary production of mercury will continue to increase as more recycling facilities commence operation to recover mercury from discarded products and wastes.

1. INTRODUCTION

Section 112(n)(1)(B) of the Clean Air Act (CAA), as amended in 1990, requires the U.S. Environmental Protection Agency (U.S. EPA) to submit a study on atmospheric mercury emissions to Congress. The sources of emissions that must be studied include electric utility steam generating units, municipal waste combustion units and other sources, including area sources. Congress directed that the Mercury Study evaluate many aspects of mercury emissions, including the rate and mass of emissions, health and environmental effects, technologies to control such emissions, and the costs of such controls.

In response to this mandate, U.S. EPA has prepared a seven-volume Mercury Study Report to Congress. The seven volumes are as follows:

- I. Executive Summary
- II. An Inventory of Anthropogenic Mercury Emissions in the United States
- III. An Assessment of Exposure from Anthropogenic Mercury Emissions in the United States
- IV. Health Effects of Mercury and Mercury Compounds
- V. An Ecological Assessment of Anthropogenic Mercury Emissions in the United States
- VI. Characterization of Human Health and Wildlife Risks from Anthropogenic Mercury Emissions in the United States
- VII. An Evaluation of Mercury Control Technologies and Costs

This volume (Volume II) estimates mercury emissions from anthropogenic sources and provides abbreviated process descriptions, control technique options, emission factors, and activity levels for these sources. Also, if sufficient information is available, locations by city and state are given for point sources.

This report updates the U.S. EPA document entitled *National Emissions Inventory of Mercury and Mercury Compounds: Interim Final Report* (U.S. EPA, 1993d). The most significant changes to the inventory are the utility boiler and municipal waste combustor source categories. Other source categories (e.g., lamp breakage and chlor-alkali plants) were also updated. These changes are noted in the text.

1.1 Overview of Sources

In the CAA, Congress directed U.S. EPA to examine sources of mercury emissions, including electric utility steam generating units, municipal waste combustion units and other sources, including area sources. The U.S. EPA interpreted the phrase "... and other sources..." to mean that a comprehensive examination of mercury sources should be made and to the extent data were available, air emissions should be quantified. This report describes in some detail various source categories that emit mercury. In many cases, a particular source category is identified as having the potential to emit mercury, but data are not available to assign a quantitative estimate of emissions. The U.S. EPA's intent was to identify as many sources of mercury emissions to the air as possible and to quantify those emissions where possible.

The mercury emissions data that are available vary considerably in quantity and quality between different source types. Not surprisingly, the best available data are for source categories that U.S. EPA has examined in the past or is currently studying.

Sources of mercury emissions in the United States are ubiquitous. To characterize these emissions, the type of mercury emission is defined as either:

- *Natural mercury emissions* -- the mobilization or release of geologically bound mercury by natural processes, with mass transfer of mercury to the atmosphere;
- *Anthropogenic mercury emissions* -- the mobilization or release of geologically bound mercury by human activities, with mass transfer of mercury to the atmosphere; or
- *Re-emitted mercury* -- the mass transfer of mercury to the atmosphere by biologic and geologic processes drawing on a pool of mercury that was deposited to the earth's surface after initial mobilization by either anthropogenic or natural activities.

Anthropogenic mercury emissions can be further divided into area and point sources. Anthropogenic area sources of mercury emissions are sources that are typically small and numerous and usually cannot be readily located geographically. For the purpose of this report, mobile sources are included in the area source section. Point sources are those anthropogenic sources that are associated with a fixed geographic location. These point sources are further divided into combustion, manufacturing and miscellaneous source categories. Particular types of sources that fall into these various groups are outlined in Table 1-1.

A prerequisite for developing strategies for reducing mercury concentrations in surface waters, biota and ambient air is a comprehensive characterization of all sources of mercury releases to the environment. A complete characterization would include: (1) all sources of airborne emissions, including natural and anthropogenic emissions as well as re-emitted mercury; (2) direct discharges to surface water and soil; and (3) past commercial and waste disposal practices that have resulted in mercury contamination in different environmental media. The focus of this study, however, is only on air emissions in accordance with section 112(n) of the CAA. In addition, the current state of knowledge about airborne emissions does not allow for an accurate assessment of either natural mercury emissions or re-emitted mercury. The U.S. EPA recognizes that an assessment of the relative public health and environmental impact that can be attributed to current anthropogenic emissions is greatly complicated by releases to other media, natural mercury emissions, and previous emissions of mercury that have subsequently deposited. This report provides the basis for a nationwide mercury emission characterization from anthropogenic sources. For each source category, the processes yielding mercury emissions and the emission control measures are described. The procedures used to estimate nationwide mercury emissions from each category are also delineated.

1.2 Study Approach and Uncertainties

This report contains mercury emission factors available from the U.S. EPA document *Locating and Estimating Air Emissions from Sources of Mercury and Mercury Compounds* (L&E document, U.S. EPA, 1993a). Other information sources used include recently published reports, journal articles and information from trade associations. Mercury emission rates presented in this report are estimates only. To the degree that information is available, the sources of uncertainty in the emission estimates are discussed (at least qualitatively) as the estimates are discussed throughout the report.

For most source categories, an emission factor-based approach was used to calculate nationwide emission rate estimates. This approach requires an emission factor, which is a ratio of the mass of mercury emitted to a measure of source activity, as well as an estimate of the annual nationwide source activity level. Examples of measures of source activity include total heat input for

Table 1-1
Sources of Anthropogenic Mercury Emissions Examined In This Inventory

Area	Point		
	Combustion	Manufacturing	Miscellaneous
Electric lamp breakage	Utility boilers	Chlor-alkali production	Oil shale retorting
Paints use	Commercial/industrial boilers	Lime manufacturing	Mercury catalysts
Laboratory use	Residential boilers	Primary mercury production ^a	Pigment production
Dental preparations	Municipal waste combustors	Mercury compounds production	Explosives manufacturing
Mobile sources ^a	Medical waste incinerators	Battery production	Geothermal power plants
Agricultural burning ^a	Sewage sludge incinerators	Electrical apparatus manufacturing	Turf products
Landfills ^a	Hazardous waste combustors	Carbon black production	
Sludge application ^a	Wood-fired boilers	Byproduct coke production ^a	
	Residential woodstoves ^a	Primary copper smelting	
	Crematories	Cement manufacturing	
		Primary lead smelting	
		Petroleum refining ^a	
		Instrument manufacturing	
		Secondary mercury production	
		Zinc mining ^a	
		Fluorescent lamp recycling	

^a Potential anthropogenic sources of mercury for which emissions were not estimated.

fossil fuel combustion and total raw material used or product generated for industrial processes. Activity levels used in this report were compiled over different time periods and with a variety of survey procedures. Emission factors are generated from emission test data, engineering analyses based on mass balance techniques, or transfer of information from comparable sources. Generally, emission factors are based on a limited set of test data that may not be representative of the full source population being studied. Emission factors used to estimate nationwide emissions reflect "typical control" achieved by the air pollution control measures applied across the population of sources within a source category. The emission factors and control levels used to develop emission estimates contained in this report were generally taken from the L&E document (U.S. EPA, 1993a). Emission factors from the L&E document were not used for estimating emissions from utility boilers, chlor-alkali plants using the mercury cell process or fluorescent lamp breakage. Additional test data for utility boilers became available after the L&E document was published. More recent information was also available directly from chlor-alkali plant managers. A mass balance approach was used for lamp breakage.

The emission factor-based approach does not generate exact nationwide emission estimates. Uncertainties are introduced in the emission factors, the estimates of control efficiency and the nationwide activity level measures. Ideally, emission factors are based on a substantial quantity of data from sources that represent the source category population. For trace pollutants like mercury, however, emission factors are frequently based on limited data that may not have been collected from representative sources. Also, changes in processes or emission measurement techniques over time may result in biased emission factors. In particular, analytical methods for detecting mercury have changed, especially since about 1985. Emission control estimates are also generally based on limited data; as such these estimates are imprecise and may be biased. Control efficiencies based on data collected using older test methods may be biased because the older test methods tended to collect mercury vapor inefficiently. (Currently, U.S. EPA Method 301 from 40 CFR Part 63, Appendix A can be used to validate the equivalency of new methods.) Finally, activity levels used in this study were based on the most recent information that was readily available. The sources of data used vary in reliability, adding further uncertainty to the emission estimates.

Generally, quantitative estimates of the uncertainty in the emission factors, control efficiency estimates and activity level measures are not available; these uncertainties are discussed qualitatively. Potential biases in the final emission estimates are also discussed. Table 1-2 presents source categories for which U.S. EPA had sufficient data to estimate national emissions. Table 1-3 presents source categories for which information is insufficient to estimate national emissions.

1.3 Organization of the Rest of the Document

The remainder of this report consists of seven chapters and three appendices. Chapter 2 discusses trends in the environmental mercury burden and in the industrial consumption of mercury. Chapter 3 characterizes mercury emissions from area sources such as engines, light bulbs and dental preparations. It describes the emitting process and presents the basis for the emission estimates. Chapter 4 provides a summary of emission estimates from point sources, including combustion, manufacturing and miscellaneous sources. Chapter 5 summarizes mercury emission estimates from area and point sources; Chapter 6 presents overall conclusions; Chapter 7 identifies further research needs; and all of the references used are listed in Chapter 8. Appendix A contains information on activity levels, source locations and emissions from some source categories. Appendix B presents available data on the mercury removal efficiencies of particulate matter and acid gas controls for utilities. Finally, Appendix C presents emission factors used to estimate emissions from utility boilers.

Table 1-2
Anthropogenic Mercury Sources With Sufficient
Data to Estimate National Emissions

Area	Point		
	Combustion	Manufacturing	Miscellaneous
Electric lamp breakage	Utility boilers	Chlor-alkali production	Geothermal power plants
Laboratory use	Commercial/industrial boilers	Cement manufacturing	
Dental preparation	Residential boilers	Battery production	
	Municipal waste combustors	Electrical apparatus manufacturing	
	Medical waste incinerators	Instrument manufacturing	
	Sewage sludge incinerators	Secondary mercury production	
	Wood-fired boilers	Carbon black production	
	Hazardous waste combustors ^a	Primary lead smelting	
	Crematories	Primary copper smelting ^b	
		Lime manufacturing	
		Fluorescent lamp recycling	

^a Emissions in 1995 were estimated for hazardous waste incinerators and lightweight aggregate kilns; however, these 1995 estimates were not used in any of the modeling analyses.

^b Estimates were made for one source only. This source ceased operations in February 1995.

Table 1-3
Mercury Sources With Insufficient Information to Estimate National Emissions

Natural	Anthropogenic			
	Area	Point		
		Combustion	Manufacturing	Miscellaneous
Oceans and other natural waters	Mobile sources	Residential woodstoves	Primary mercury production ^a	Oil shale retorting ^a
	Paint use ^a			Mercury catalysts ^a
Vegetation	Agricultural burning		Mercury compounds production	Pigment production ^a
Volcanoes	Landfills		Byproduct coke production	Explosives manufacturing ^a
Rocks	Sludge application		Petroleum refining	Turf products ^a
Soils			Zinc mining	
Wildfires				

^a Mercury is no longer used in U.S. manufacture. However, this is not meant to imply that these previous activities are no longer having an impact on the environment due to mercury's persistence in the environment.

2. MERCURY TRENDS

Contemporary anthropogenic emissions of mercury are only one component of the global mercury cycle. Releases from human activities today are adding to the mercury reservoirs that already exist in land, water, and air, both naturally and as a result of previous human activities.

This chapter discusses the global mercury cycle and trends in mercury concentrations and fluxes observed in the environment. It also briefly reviews historical and recent patterns in mercury supply and demand in the U.S. This discussion provides general background and context for the more detailed discussion of anthropogenic sources in the chapters that follow.

2.1 Trends in the Atmospheric Mercury Burden

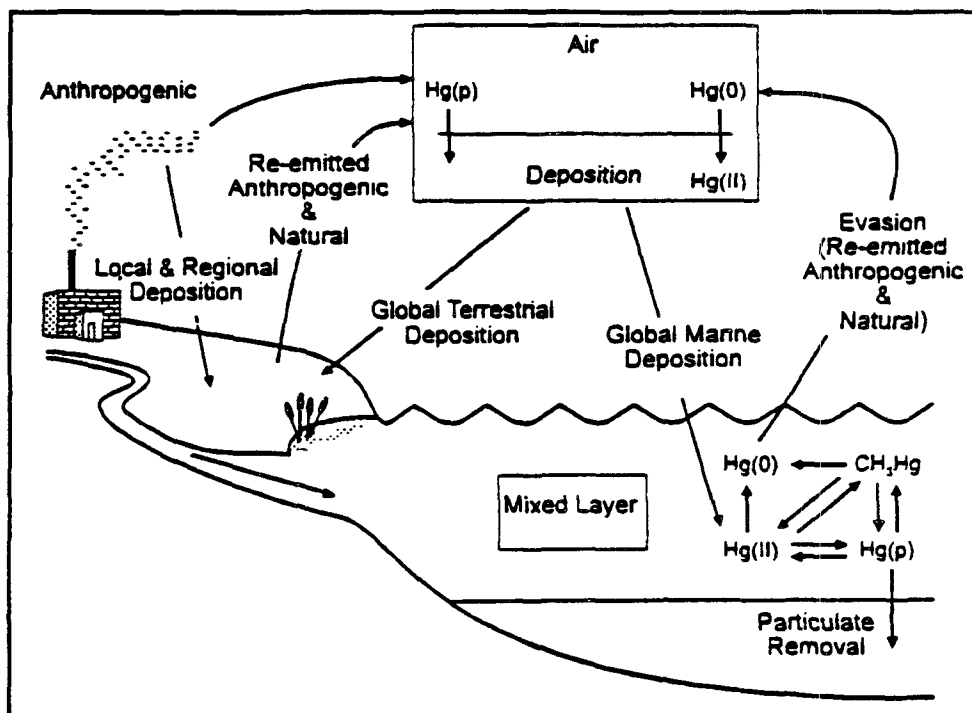
Given the present understanding of the global mercury cycle, the flux of mercury from the atmosphere to land or water at any one location is comprised of contributions from:

- The natural global cycle,
- The global cycle perturbed by human activities,
- Regional sources, and
- Local sources.

Understanding of the global mercury cycle (shown schematically in Figure 2-1) has improved significantly with continuing study of source emissions, mercury fluxes to the earth's surface, and the magnitude of mercury reservoirs that have accumulated in soils, watersheds and ocean waters. Although considerable uncertainty still exists, it has become increasingly evident that anthropogenic emissions of mercury to the air rival or exceed natural inputs. Recent estimates place the annual amounts of mercury released into the air by human activities at between 50 and 75 percent of the total yearly input to the atmosphere from all sources. Recycling of mercury at the earth's surface, especially from the oceans, extends the influence and active lifetime of anthropogenic mercury releases (Expert Panel, 1994).

A better understanding of the relative contribution of mercury from anthropogenic sources is limited by substantial remaining uncertainties regarding the level of natural emissions as well as the amount and original source of mercury that is re-emitted to the atmosphere from existing reservoirs. Recent estimates indicate that of the approximately 200,000 tons of mercury emitted to the atmosphere since 1890, about 95 percent resides in terrestrial soils, about 3 percent in the ocean surface waters, and 2 percent in the atmosphere (Expert Panel, 1994). More study is needed before it is possible to accurately differentiate natural fluxes from these reservoirs from re-emissions of mercury originally released from anthropogenic sources. For instance, approximately one-third of total current global mercury emissions are thought to cycle from the oceans to the atmosphere and back again to the oceans, but a major fraction of the emissions from oceans consists of recycled anthropogenic mercury. It is believed that much less than 50 percent of the oceanic emission is from mercury originally mobilized by natural sources. Similarly, a potentially large fraction of terrestrial and vegetative emissions consists of recycled mercury from previously deposited anthropogenic and natural emissions (Expert Panel, 1994).

Figure 2-1
The Global Mercury Cycle



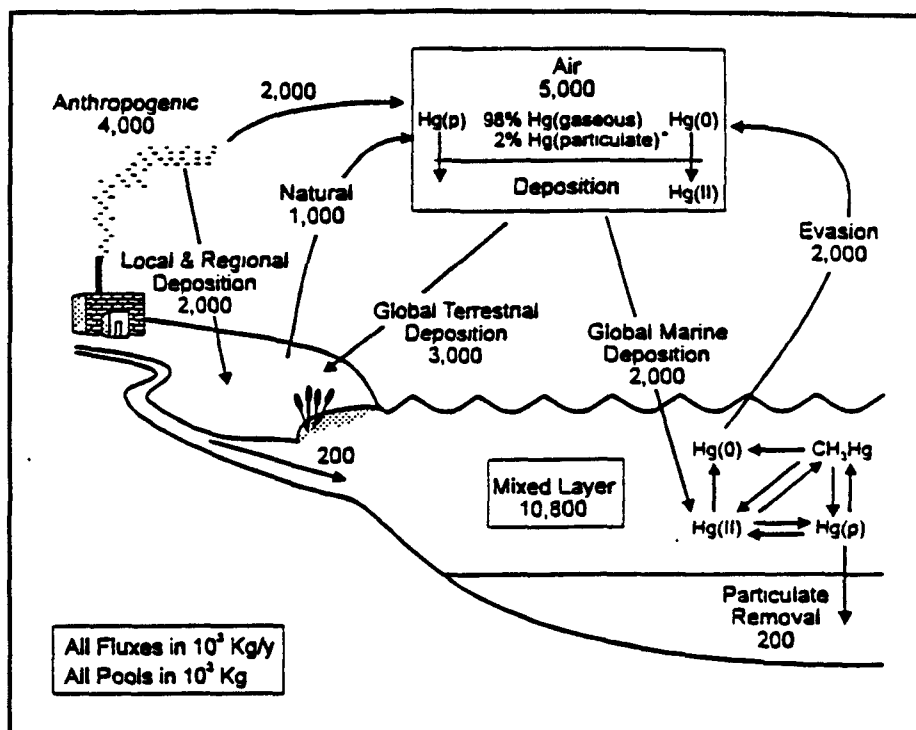
Source: Adapted from Mason et al., 1994.

Comparisons of contemporary (within the last 15-20 years) measurements and historical records indicate that the total global atmospheric mercury burden has increased since the beginning of the industrialized period by a factor of between two and five (see Figure 2-2). For example, analysis of sediments from Swedish lakes shows mercury concentrations in the upper layers that are two to five times higher than those associated with pre-industrialized times. In Minnesota and Wisconsin, an investigation of whole-lake mercury accumulation indicates that the annual deposition of atmospheric mercury has increased by a factor of three to four since pre-industrial times. Similar increases have been noted in other studies of lake and peat cores from this region, and results from remote lakes in southeast Alaska also show an increase, though somewhat lower than found in the upper midwest U.S. (Expert Panel, 1994).

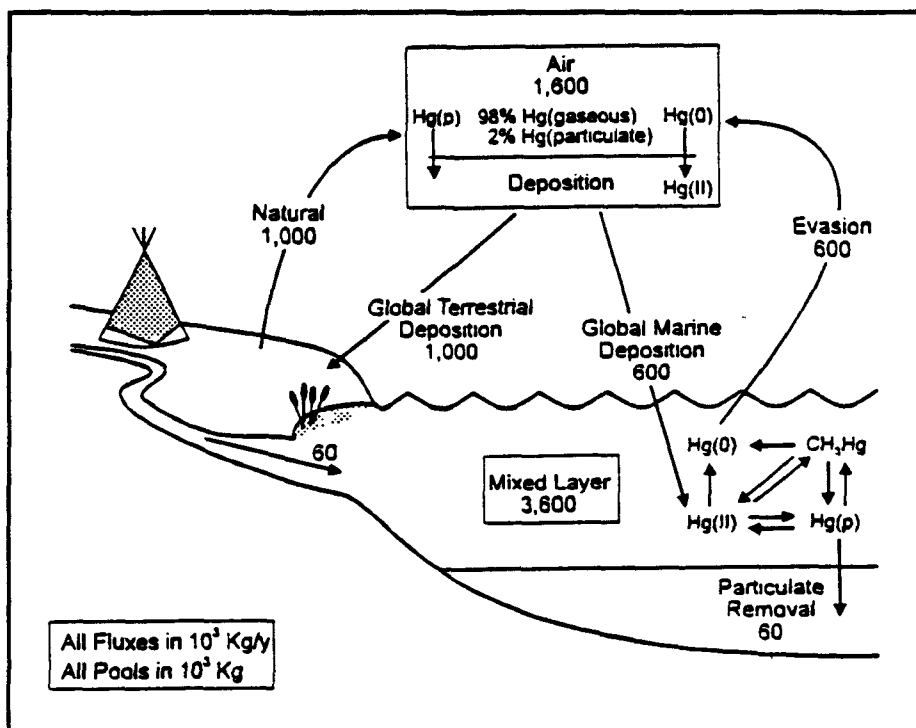
While the overall trend in the global mercury burden since pre-industrial times appears to be increasing, there is some evidence that mercury concentrations in the environment in certain locations have been stable or decreasing over the past few decades. For example, preliminary results for eastern red cedar growing near industrial sources (chlor-alkali, nuclear weapons production) show peak mercury concentrations in wood formed in the 1950s and 1960s, with stable or decreasing concentrations in the past decade (Expert Panel, 1994). Some results from peat cores and lake sediment cores also suggest that peak mercury deposition occurred prior to 1970 (Benoit et al., 1994; Swain et al., 1992; and Engstrom, 1994). Data collected over 25 years from many locations in the United Kingdom on liver mercury concentrations in two raptor species and a fish-eating grey heron indicate that peak concentrations occurred prior to 1970. The sharp decline in liver mercury

Figure 2-2
Comparison of Current and Pre-Industrial
Mercury Budgets and Fluxes

Current Mercury
 Budgets and
 Fluxes



Pre-Industrial
 Mercury Budgets
 and Fluxes



Source: Adapted from Mason et al., 1994.

concentrations in the early 1970s suggests that local sources, such as agricultural uses of fungicides, may have led to elevated mercury levels two to three decades ago (Newton et al., 1993). Similar trends have been noted for mercury levels in eggs of the common loon collected from New York and New Hampshire (McIntyre et al., 1993). This downward trend in mercury concentrations observed in the environment over the last few decades generally tracks with mercury use and consumption patterns over the same timeframe (discussed in the following section).

2.2 Trends in Mercury Consumption

The mercury available for use in the U.S. comes from five main sources: (1) primary production (mining); (2) by-product production (i.e., mercury by-product from gold mining operations); (3) secondary production (recovery) from industrial recycling operations; (4) sales from excess government stocks, including those held by the Department of Energy (DOE) and the Defense Logistics Agency (DLA) within the Department of Defense; and (5) imports. Table 2-1 illustrates the relative contributions of these sources to the U.S. mercury supply from 1988 through 1993. The table also shows the total industrial demand or consumption levels for that same period.

Figure 2-3 plots mercury supply and demand levels since 1955. Supplies associated with by-product production are not shown in this figure because data for this category are not available prior to 1990. Similarly, DLA sales are not presented in Figure 2-3 because data for such sales are not available prior to 1982.

These data show a general decline in domestic mercury use since demand peaked in 1964. Domestic demand fell by 74 percent between 1980 and 1993, and by more than 50 percent since 1988. The rate of decline, however, has slowed since 1990. Further evidence of the declining need for mercury in the U.S. is provided by the general decline in imports since 1988 and the fact that exports have exceeded imports since at least 1989. Approximately 78 percent of the net U.S. exports of mercury during the last five years has come from federal sales, with a steadily increasing portion of the federal sales coming from the National Defense Stockpile managed by DLA. Federal sales accounted for 97 percent of the U.S. demand in 1993 (Ross & Associates, 1994).

Most recently, there has been a sharp drop in Federal sales. In July 1994, DLA suspended future sales of mercury from the Department of Defense stockpile until the environmental implications of these sales are addressed. In addition, in past years, DLA sold mercury accumulated and held by the Department of Energy, which is also considered excess to government needs. DLA suspended these mercury sales in July 1993 for an indefinite period in order to concentrate on selling material from its own mercury stockpile (Ross & Associates, 1994).

In general, these data suggest that industrial manufacturers that use mercury are shifting away from mercury except for uses for which mercury is considered essential. This shift is believed to be largely the result of Federal bans on mercury additives in paint and pesticides; industry efforts to reduce mercury in batteries; increasing state regulation of mercury emissions sources and mercury in products; and state-mandated recycling programs. A number of Federal activities are also underway to investigate pollution prevention measures and control techniques for a number of sources categories (see Volume VII of this Report to Congress).

Table 2-1
U.S. Mercury: Supply, Demand, Imports, Exports
(Mg)

Category	1988	1989	1990	1991	1992	1993
Supply:						
Mine production ^a	379	414	448	0	0	0
By-product production ^b	W ^c	W	114	58	64	W
Industrial recovery	278	137	108	165	176	350
DLA sales	52	170	52	103	267	543
DOE sales	214	180	193	215	103	0
Subtotal: federal sales	266	350	245	318	370	543
Imports	329	131	15	56	92	40
Total supply	1,252	1,032	930	597	702	933
Federal sales as % of total supply	21.2%	33.9%	26.3%	53.3%	52.7%	58.1%
Demand:	1,503	1,212	720	554	621	558
Federal sales as % of domestic demand	17.6%	28.9%	34%	57.4%	59.6%	97.3%
Imports:	329	131	15	56	92	40
Exports:	N/A ^d	221	311	786	977	389
Exports minus imports:	N/A	90	296	730	885	349

Source: U.S. Bureau of Mines, 1994.

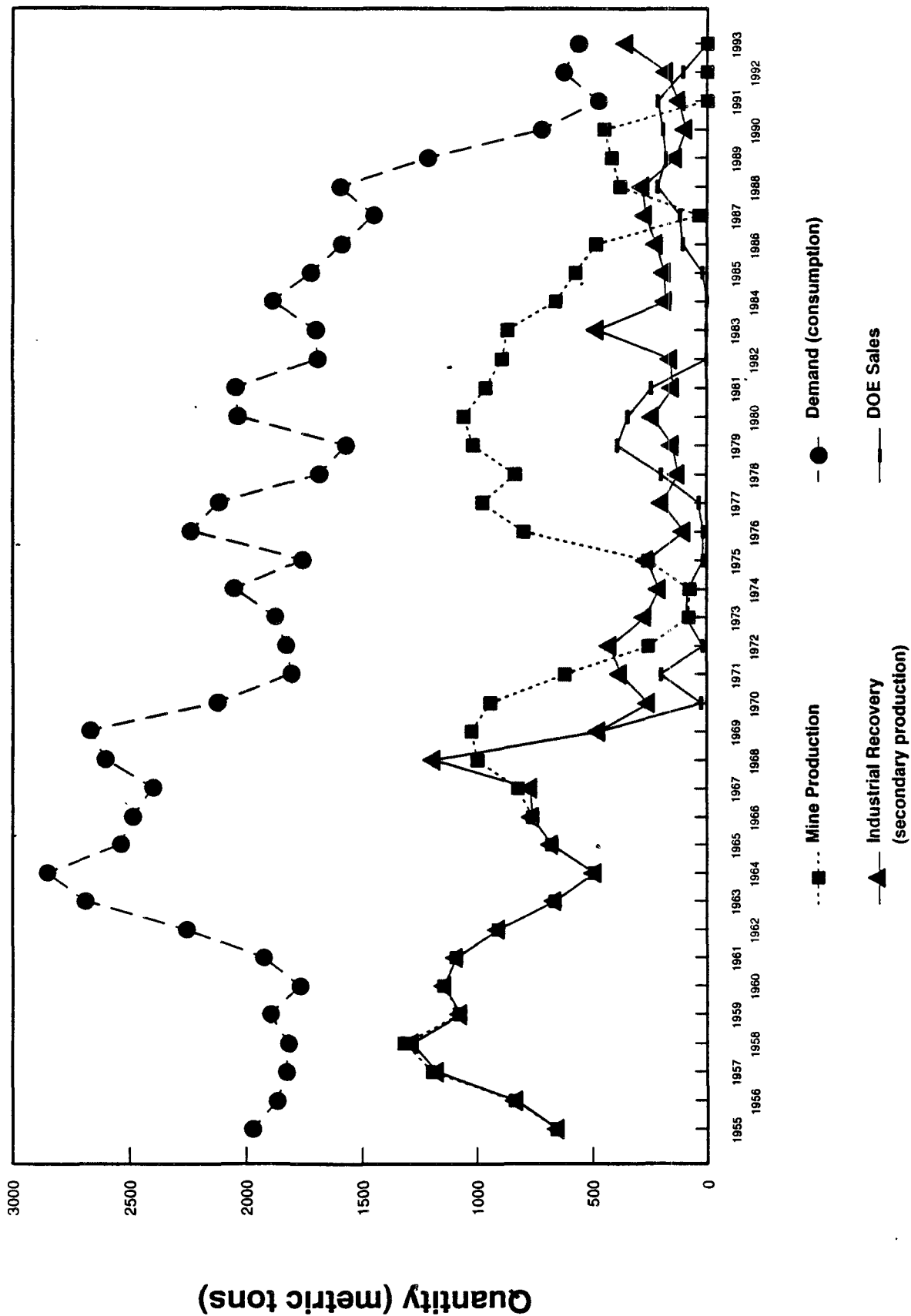
^a Mercury production from McDermitt mine; closed November 1990.

^b Mercury by-product from nine gold mining firms.

^c Withheld for proprietary reasons.

^d Not available.

Figure 2-3
U.S. Mercury: Supply, Demand, Secondary Production



3. ANTHROPOGENIC AREA SOURCES OF MERCURY EMISSIONS

Area sources account for approximately 1.3 percent of mercury emissions from anthropogenic sources. Table 3-1 summarizes the estimated annual quantities of mercury emitted from area sources. Each of these source categories is discussed in turn in the sections that follow.

3.1 Electric Lamp Breakage

Electric lamps containing mercury include fluorescent, mercury vapor, metal halide and high-pressure sodium lamps. These lamps are used for both indoor and outdoor applications including heat lamps, lights for high-ceiling rooms, film projection, photography, dental exams, photochemistry, water purification and street lighting. When these electric lamps are broken during use or disposal, a portion of the mercury contained in them is emitted to the atmosphere. The amount of mercury emitted to the atmosphere when mercury-containing lamps are disposed of will be a function of many factors. These include the chemical form of mercury in the lamp and the size of the particulate forms of mercury in the lamp powder. Approximately 643 Mg of mercury were discarded in U.S. municipal solid waste (MSW) in 1989. The amount of mercury entering the MSW system from the disposal of used mercury-containing lamps in 1989 is estimated to have been 24.3 Mg (26.8 tons), or 3.8 percent of the total mercury content of MSW (Truesdale et al., 1993).

Since the mid-1980s, electrical manufacturers have reduced the average amount of mercury in each fluorescent bulb from an average of 48.2 mg to an average of 22.8 mg of mercury. A certain amount of mercury is needed, however, in order to maintain desirable properties. The present practical limit needed for full-rated life performance of a 4-foot fluorescent lamp has been thought to be about 15 mg of mercury (National Electrical Manufacturers Association, 1995). However, one manufacturer recently announced that it will be manufacturing four-foot lamps with less than 10 mg of mercury by late 1995 (Walitsky, 1995). Table 3-2 presents the estimated mercury content of fluorescent bulbs, as provided in four different sources.

The average lifetime of a High Intensity Discharge (HID) lamp is between 10,000 and 24,000 hours. (Some small volume specialty products have lifetimes less than 10,000 hours or greater than 24,000 hours.) HID lamps last three to six years in typical applications. Low-pressure fluorescent lights typically have a rated lifetime of 20,000 hours (Truesdale et al., 1993).

Approximately 550 million lamps containing mercury are sold annually in the United States (National Electrical Manufacturers Association, 1992). Of these, 22 million are of the HID variety; the remaining 528 million are fluorescent bulbs. Table 3-3 contains production rates from 1970 through 1989 including exports and imports. Since 1970, there has been an increase in the production of HID lamps (U.S. EPA, 1992a). Table 3-4 presents the mercury content of HID lamps and their manufacturers.

Table 3-1
Best Point Estimates of Mercury Emissions from Anthropogenic Area Sources: 1990-1993

Source	Emissions			Date of Data ^a	Degree of Uncertainty ^b	Basis for Emissions Estimate
	Mg/yr	Tons/yr	% of total			
Electric lamp breakage	1.4	1.5	0.6	1989/1989	High	Engineering judgment and mass balance; Industry estimate for this source category is 0.18 tons/year; this difference is explained in section 3.2.
Laboratory use	0.7	0.8	0.3	1973/1992	High	Engineering judgment
Dental preparations	0.7	0.8	0.3	1980/1992	High	Engineering judgment
Mobile sources	-	-	-	-	-	Insufficient information to estimate emissions
Paints use	-	-	-	-	-	Mercury phased out of paint use in 1991
Agricultural burning	-	-	-	-	-	Insufficient information to estimate national emissions; one study estimates 0.036 Mg/yr (0.04 tons/yr) from preharvest burning of sugarcane in Florida everglades area
Landfills	-	-	-	-	-	Insufficient information to estimate emissions
Total	2.8	3.1	1.3			

^a Date that data emission factor is based on/Date of activity factor used to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

Table 3-2
Mercury Content of Fluorescent Bulbs^a

Year	Average Mercury Content (mg) per Bulb		
	NEMA	CWF	3M
1970-1984		75	
1985-1989	48.2	55	
1990	41.6		
1992		40	15-30
1995	22.8		

^a Cole et al., 1992; National Electrical Manufacturers Association, 1992; Tanner, 1992; National Electrical Manufacturers Association, 1995.

Table 3-3
Mercury (HID) Lamp Production - 1970 to 1989^a

Year	Quantity ^b (1000 bulbs)	Year	Quantity ^b (1000 bulbs)
1970	6,841	1982	20,891
1971	7,684	1983	22,146
1972	8,420	1984	25,636
1973	9,349	1985	25,529
1974	9,158	1986	22,206
1975	8,737	1987	28,143
1976	10,383	1988	24,479
1977	10,853	1989	28,090
1978	12,175		
1979	13,532		
1980	30,187		
1981	21,397		

^a Cole et al., 1992; U.S. EPA, 1992a.

^b Production rate = Domestic shipments - Exports + Imports.

Table 3-4
Mercury Content of HID Lamps^a

Manufacturer	Type	Mercury Content (mg)
Phillips	250 watt HID	45
	400 watt HID	60
	1000 watt HID	70
Sylvania	250 watt HID	46
	400 watt HID	75
	1000 watt HID	75

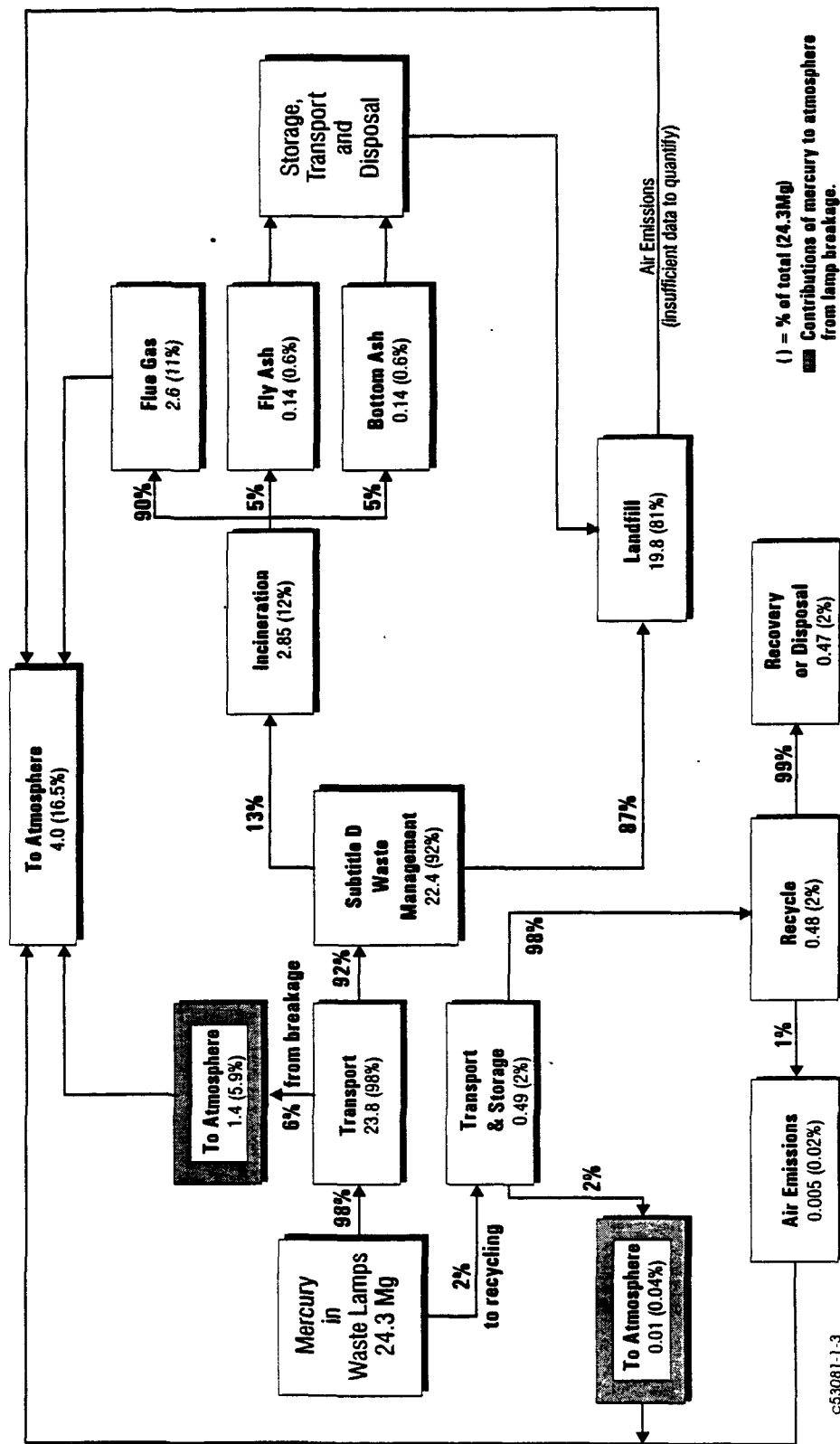
^a Cole et al., 1992; U.S. EPA, 1992a.

Mercury and metal halide lamps consist of an inner quartz arc tube enclosed in an outer envelope of heat resistant glass. The quartz arc tube contains a small amount of mercury ranging from 20 mg in a 75-watt lamp up to 250 mg in a 1000 watt lamp. According to the National Electrical Manufacturers Association, no other substance has been found to replace mercury. High-pressure sodium lamps consist of an inner, high-purity alumina ceramic tube enclosed in an outer envelope of heat-resistant glass. The ceramic tube contains a small amount of sodium/mercury amalgam, ranging from 8.3 mg of mercury in a 50-watt lamp up to 25 mg in a 1000-watt lamp (National Electrical Manufacturers Association, 1992).

The fate of used lamps is tied to the disposal of MSW. The three primary options for MSW disposal are landfilling, combustion and recycling. Landfilling accounts for 82 percent of MSW disposal, incineration accounts for 16 percent and recycling accounts for 2 percent. One study traced the path of used lamps in MSW to each of the primary disposal options. Figure 3-1 diagrams the flow of used mercury-containing lamps through the national MSW management system. Ninety-eight percent of used lamps are managed as MSW under Subtitle D (the solid, non-hazardous waste program) of the Resource Conservation and Recovery Act (RCRA), with the remaining 2 percent being recycled. Mercury emissions from lamp breakage occur during transportation and storage of lamps. A total of 1.4 Mg/yr (1.5 tons/yr) is estimated to be emitted during transport and storage (Truesdale et al., 1993). Additional mercury emissions from electric lamps are associated with MSW incineration, lamp recycling activities and landfills. Mercury emissions from MSW incineration are accounted for in Section 4.1. Lamp recycling activities are discussed in Section 4.2.7. Insufficient data exist to estimate mercury emissions from landfills.

Discarded lamps may be transported in two ways: in garbage trucks as household or commercial trash and in closed vans or trailers as part of a bulk relamping program. Of the 98 percent of mercury from lamps in the MSW stream, 80 percent is transported in garbage trucks along with other solid waste and 20 percent is transported in group re-lamping trucks holding lamps alone. Emissions from both transport mechanisms were estimated using the waste pile mass transfer model developed for the RCRA air emissions standards.

Figure 3-1
Overall Fate of Mercury from
Used Mercury-Containing Fluorescent Lamps



Reference: Adapted from Truesdale et al., 1993.

For transportation in a garbage truck, it was assumed that all lamps are broken in the truck and that the mercury present in the vapor is emitted to the atmosphere. The mercury concentration in the lamps was assumed to be 0.14 ppm. For relamping programs, the discarded lamps are packed in the corrugated containers from which the new lamps were taken and are then loaded into enclosed vans or trailers for removal. In this case, fewer lamps are broken; a 10 percent breakage was assumed (Truesdale et al., 1993).

The modeling exercise predicted that approximately 6 percent of the mercury being transported by garbage trucks and from group re-lamping is emitted to the atmosphere. This amounts to 1.4 Mg/year (1.5 tons/year).

Mercury emissions from transporting and storing lamps sent to recycling plants were also estimated using the waste pile emission model. Emissions were based on a 30-day storage time and an average of 5 percent breakage for the transport and storage steps. Emissions from storage facilities were estimated to comprise about 90 percent of the recycling transport and storage emissions, amounting to approximately 0.008 Mg. Total mercury emissions from transport and storage of waste lamps is estimated to be 0.01 Mg, or 0.04 percent of the mercury from lamps entering the MSW (Truesdale et al., 1993).

The total mercury emissions from mercury lamp breakage, including emissions from transport and storage, are estimated to be 1.4 Mg/yr (1.5 tons/yr). This estimate may have a high degree of error associated with it due to the number of assumptions made and the lack of test data. This estimate represents a change from U.S. EPA's earlier estimate of 8 Mg (8.8 tons/year) (U.S. EPA, 1993d). The current estimate is based on more plausible assumptions than the earlier analysis which assumed a large fraction of the mercury was emitted to the atmosphere upon lamp breakage. These earlier estimates were thought to be high considering that only a small percentage of mercury in the lamp is in the vapor state.

The industry estimate of mercury emissions from discarded fluorescent lamps is 0.16 Mg/year (0.18 tons/year) (National Electrical Manufacturers Association, 1995). The industry estimate assumes that most lamps are landfilled within a couple of days after their disposal and are covered with 0.5 to 1 foot of soil at that time. Simulating this landfilling practice and measuring the amount of mercury released led to an estimated mercury evaporation rate of 0.8 percent after 20 days when the lamps were covered by 0.5 feet of soil, and 0.2 percent after 20 days when the lamps were covered by 1 foot of soil (rather than the 6.6 percent estimated in Truesdale et al., 1993, which is the basis for U.S. EPA's estimate). The 0.8 percent evaporation rate was used to calculate the annual rate of 0.16 Mg/year (0.18 tons/year). The National Electrical Manufacturers Association study also measured the maximum mercury evaporation rate from a broken lamp to be 6.35 percent after 50 days. However, as explained above, the industry calculation of national emissions assumes that all discarded lamps are covered by soil within a couple of days of being discarded.

3.2 General Laboratory Use

Mercury is used in laboratories in instruments, as a reagent, and as a catalyst. In 1992, an estimated 0.7 Mg (0.8 ton) of mercury were emitted into the atmosphere from general laboratory use. An emission factor of 40 kg of mercury emitted for each megagram of mercury used in laboratories was estimated in a 1973 report (Anderson, 1973). Because this emission factor was based on engineering judgment and not on actual test data, and because it is dated, the reliability of this emission factor is questionable. As with most industries using mercury, there was a decline in mercury consumption in general laboratory use, with consumption dropping from 32 Mg (35 tons) in

1990 to 18 Mg (20 tons) in 1992 (Bureau of Mines, 1992). The annual emission estimate is the product of this consumption rate and the emission factor noted above. The limitations of that emission factor make the emission estimate uncertain.

3.3 Dental Preparation and Use

Mercury is used in the dental industry, primarily in amalgam fillings for teeth, although it may also be used in other dental equipment and supplies. In 1992, an estimated 0.7 Mg (0.8 ton) of mercury was emitted from dental preparation and use. This is an underestimate because it is derived using an emission factor that applies only to emissions of mercury from spills and scrap during dental preparation and use (2 percent of mercury used is emitted into the atmosphere) (Perwak, 1981). The total amount of mercury used in the dental industry is 37 Mg (41 tons) and includes mercury used in all dental equipment and supplies, not just the amount used in dental preparation and use (Bureau of Mines, 1992). Mercury emissions not accounted for in dental preparation and use are most likely accounted for in the emission estimates for municipal waste combustors and crematories. Mercury discharges from dental offices to publicly owned sewage treatment facilities are also known to occur but are not addressed in this report.

3.4 Mobile Sources

Mobile sources are defined in this report as diesel- and gasoline-powered, on-road, light-duty vehicles. Of these types, gasoline-powered vehicles make up the most significant mobile emission sources. A 1983 study indicated an estimated mercury emission factor of 1.3×10^{-3} milligram per kilometer (mg/km) (4.6×10^{-9} pound per mile [lb/mile]) traveled for tail-pipe emissions from motor vehicles (Pierson and Brachaczek, 1983). These data were for particulate mercury emissions derived from neutron activation analysis of particulate filters. The population of vehicles studied was 81.9 percent gasoline-powered passenger cars, 2.4 percent gasoline-powered trucks and 15.7 percent diesel trucks. The data are of questionable reliability for the current vehicle population because this emission factor is based on a 1977 ambient sampling study, which predated the broad use of catalytic converters and unleaded gasoline, widely mandated 'State-regulated inspection and maintenance programs and diesel-powered vehicle emission control requirements. It is unknown what effect these measures might have on mercury emissions.

A 1979 study characterized regulated and unregulated exhaust emissions from catalyst and non-catalyst equipped light-duty gasoline-powered automobiles operating under malfunction conditions (Urban and Garbe, 1979). An analysis for mercury was included in the study, but no mercury was detected in tail-pipe emissions. The analytical minimum detection limit was not stated. A 1989 study measured the exhaust emission rates of selected toxic substances for two late model gasoline-powered passenger cars (Warner-Selph and DeVita, 1989). The two vehicles were operated over the Federal Test Procedure (FTP), the Highway Fuel Economy Test (HFET) and the New York City Cycle (NYCC). Mercury was among the group of metals analyzed but was not present in detectable quantities. The analytical minimum detection limits for mercury in the three test procedures were the following: FTP 0.025 mg/km (8.9×10^{-8} lb/mile) HFET 0.019 mg/km (6.7×10^{-8} lb/mile) and NYCC 0.15 mg/km (53.2×10^{-8} lb/mile) (Warner-Selph and Lapp, 1993). These minimum detection limits are more than ten times higher than the estimated emission factor presented in the 1983 study.

Given the uncertainties associated with these data, tail-pipe mercury emissions from mobile sources were not calculated. The U.S. EPA also recognizes that various components of motor vehicles may contain mercury (e.g., certain truck and hood light switches, used motor oil, certain headlights and remote controls). Mercury emissions from the disposal or breakage of these components were not

estimated in this study. The potential for mercury emissions from other types of mobile sources, including ships, were not assessed in this study.

3.5 Paint Use

Four mercury compounds -- phenylmercuric acetate, 3-(chloromethoxy) propylmercuric acetate, di(phenylmercury) dodecenylsuccinate, and phenylmercuric oleate -- have been registered as biocides for interior and exterior paint (U.S. EPA, 1990). Mercury compounds are added to paints to preserve the paint in the can by controlling microbial growth. Prior to 1991, much larger amounts of mercury were added to preserve the paint film from mildew after paint was applied to a surface. During and after application of paint, these mercury compounds can be emitted into the atmosphere. As of May 1991, all registrations for mercury biocides used in paints were voluntarily canceled by the registrants, thus causing a drastic decrease in the use of mercury in paint (Agocs et al., 1990). In addition to the paint industry reformulating its paints to eliminate mercury, U.S. EPA banned the use of mercury in interior paint in 1990 and in exterior paint in 1991. The paint industry's demand for mercury in 1989 was 192 Mg (211 tons) but fell to 6 Mg (7 tons) in 1991, and had been completely eliminated in 1992 (Bureau of Mines, 1992).

Because Bureau of Mines data show no mercury usage in paint in 1992, emissions from this source were assumed to be zero. This presumes that all mercury emissions are generated from paint application the year the paint was produced. The U.S. EPA recognizes that current stocks of paint that are still being sold may include paint that contains mercury. Data were unavailable to estimate potential mercury emissions from this existing paint supply.

Prior to 1992, latex paints contributed significantly to atmospheric emissions. A 1975 study, performed when the demand for mercury in paint was high, estimated that 66 percent of the mercury used in paints was emitted into the atmosphere (Van Horn, 1975). Limited information suggests that emissions could occur for as long as seven years after initial application of paint to a surface, although the distribution of emissions over this time period is unknown (U.S. EPA, 1992a). Even so, this source category is a good example of past industrial uses and releases of mercury to the environment. Assuming the estimate is correct that 66 percent of the mercury in paint is emitted, as recently as 1989 as many as 140 tons of mercury were emitted from paint application alone in one year. Whether current levels of mercury in the environment are more likely the result of historical emissions like these or are attributable to current anthropogenic sources is still being debated.

3.6 Agricultural Burning

Mercury contamination of freshwater fish in the Florida Everglades has led to the investigation of possible mercury sources in south Florida. The preharvest burning of sugarcane has been proposed as a potential source of mercury to this area. One study estimated the atmospheric loading of mercury from burning sugarcane stalks and leaves and muck soils (Patrick, et al., 1994). An emission factor of 0.0002 kg mercury per hectare of burned crop was calculated. This resulted in 0.036 Mg (0.04 tons) of mercury emitted to the atmosphere from the preharvest burning of 174,00 acres of the Everglades Agricultural Area sugarcane crop.

Other types of agricultural burning may also contribute to mercury emissions, for example land-clearing activities. For this report, a national estimate of mercury emissions from sugarcane burning or other agricultural activities was not calculated because of the limited emissions data and a lack of data on the magnitude and frequency of these activities. The above study is presented to illustrate the potential magnitude of mercury from these activities in one area of the country.

3.7 Other Area Sources

As discussed throughout this volume, a variety of mercury-containing wastes are disposed in non-hazardous (*municipal and industrial*) and hazardous waste landfills. These landfills can serve as broad sources of airborne emissions of mercury as the disposed materials are broken or degraded, not only while the landfill is actively receiving and disposing of wastes but also after the landfilling stops and waste materials are covered with soil. Although mercury emissions have been estimated for fluorescent lamps discarded in landfills (see Section 3.1), insufficient information was available to estimate mercury emissions from landfills more generally.

Sludge application is another recognized area source of airborne emissions of mercury. This includes the agricultural and lawn application of municipal sewage sludge, which contains a number of nutrients beneficial to plants, as well as the land application of municipal and industrial sludges as a disposal method. Insufficient data were available to estimate national emissions of mercury from this activity.

4. ANTHROPOGENIC POINT SOURCES OF MERCURY EMISSIONS

A point source is a stationary location or fixed facility from which pollutants are discharged or emitted. Point sources account for approximately 99 percent of mercury emissions from anthropogenic sources. Table 4-1 presents the estimated aggregate mercury emissions from combustion, manufacturing and miscellaneous point sources. The sections that follow discuss the basis for the point source estimates for each source category within these three groups.

Table 4-1
Best Point Estimates of Annual Mercury Emissions from Combustion, Manufacturing and
Miscellaneous Point Sources: 1990-1993

Source	Emissions	
	Mg/yr	Tons/yr
Combustion	186.9	205.9
Manufacturing	29.1	32.0
Miscellaneous	1.3	1.4
Total Point Source Emissions	217.3	239.4

4.1 Combustion Sources

Combustion sources include medical waste incinerators, municipal waste combustors, fossil fuel-fired boilers, residential boilers, wood combustion, sewage sludge incinerators and crematories. Mercury emissions from these sources (excluding wood-fired residential heaters) account for an estimated 186.9 Mg/yr (205.9 tons/yr) or 85 percent of the mercury emissions generated annually in the United States. These types of combustion units are commonly found throughout the country and are not concentrated in any one geographic region. Information concerning emissions, fossil fuel consumption on a per-State basis and locations is presented in Appendix A.

Mercury exists naturally as a trace element in fossil fuels and can also be found in wastes. It is a highly volatile metal that vaporizes at the temperatures reached during the combustion zones of the processes discussed here. Consequently, mercury is emitted as a trace contaminant in the gas exhaust stream when waste materials containing mercury or fuels such as coal, oil, or wood are fired.

This section provides background information on each of the combustion sources and discusses the methodology used to estimate mercury and mercury compound emissions from the following:

(1) medical waste incinerators (MWIs), (2) municipal waste combustors (MWCs), (3) utility boilers, (4) commercial/industrial boilers, (5) residential boilers, (6) sewage sludge incinerators (SSIs), (7) crematories, (8) wood combustors, and (9) hazardous waste combustors. For each of these source types, processes and control measures currently in place are discussed, along with emission estimates

and the bases for those estimates. When a high degree of uncertainty within specific data is known, it is noted. Table 4-2 presents the estimated emissions from each source category.

4.1.1 Medical Waste Incinerators

Medical waste incinerators (MWIs) are small incineration units that charge from 0.9 Mg/d (1 ton/d) to 55 Mg/d (60 tons/d) of infectious and noninfectious wastes generated from facilities involved in medical or veterinary care or research activities. These facilities include hospitals, medical clinics, offices of doctors and dentists, veterinary clinics, nursing homes, medical laboratories, medical and veterinary schools and research units, and funeral homes. The Resource Conservation and Recovery Act (RCRA) (as amended November 1, 1988) defines medical waste as "...any solid waste which is generated in the diagnosis, treatment, or immunization of human beings or animals, in research pertaining thereto, or in the production or testing of biologicals" (U.S. EPA 1994a).

The estimated annual uncontrolled mercury emissions from MWIs are currently 58.8 Mg/yr (64.7 tons/yr). In the course of developing NSPS and emission guidelines for MWIs, with the receipt of new data, U.S. EPA expects to revise this estimate of current emissions downward. In addition, the recently proposed NSPS and emission guidelines for MWIs would decrease national mercury emissions from MWIs by almost 85 percent, to an estimated level of 3.7 Mg/yr (4.1 tons/year) after control (see the box below for more detail).

Several states including New York, California and Texas have adopted relatively stringent regulations in the past few years limiting emissions from MWIs. The implementation of these regulations has brought about very large reductions in MWI emissions of mercury in those states. It has also significantly reshaped how medical waste is managed in those states. Many facilities have responded to state regulations by switching to other medical waste treatment and disposal options to avoid the cost of add-on pollution control equipment. The two most commonly chosen alternatives have been off-site contract disposal in larger commercial incinerators and on-site treatment by other means (e.g., steam autoclaving).

Mercury emissions from MWIs occur when mercury, which exists as a contaminant in the medical waste, is combusted at high temperatures, vaporizes and exits the combustion gas exhaust stack. Known mercury sources in medical waste include batteries, fluorescent lamps, high-intensity discharge lamps, thermometers, paper and film coatings, plastic pigments, antiseptics, diuretics, skin preparations, pigments in red infectious waste bags and CAT scan paper. Much of the mercury in the medical waste stream is thought to be emitted as mercuric chloride, due to the large amount of chlorinated plastic products disposed.

U.S. EPA estimates that about 0.204×10^6 Mg/yr (0.268×10^6 tons/yr) of pathological waste and 1.431×10^6 Mg/yr (1.574×10^6 tons/yr) of general medical waste are processed annually in the United States (U.S. EPA, 1993a). Medical waste may consist of any of the following, in any combination: human and animal anatomical parts and/or tissue; sharps (syringes, needles, vials, etc.); fabrics (gauze, bandages, etc.); plastics (trash bags, IV bags, etc.); paper (disposable gowns, sheets, etc.); and waste chemicals.

As of 1985, there were a reported 6,872 known hospitals in the United States. Manufacturers estimate that at least 90 percent of these hospitals have an incinerator on-site, if only a small retort-type for burning pathological wastes (Epner, 1992). The number of larger, controlled-air MWIs is not

Table 4-2
Best Point Estimates of Mercury Emissions from Anthropogenic Combustion Point Sources: 1990-1993

Source	Emissions		Date of Data ^a	Degree of Uncertainty ^b	Basis for Emissions Estimate
	Mg/yr	Tons/yr % of total			
Medical waste incinerators ^c	58.8	64.7 26.7	1983-92/1991	Medium	Test data
Municipal waste combustors	50	55 22.7	1986-92/1991	Medium	Test data; 1995 emissions estimate is 29 tons/year
Utility boilers - coal - oil - natural gas	46.5 (46.3) ^d (0.23) (0.002)	51.3 (51) (0.25) (0.002)	1994/1990	Medium	Test data; industry estimates are 44 tons/year for coal-fired utilities
Commercial/Industrial boilers - coal - oil	26.3 (20.7) (5.5)	29 (22.8) (6.0)	*/1992	High	Mass balance; emissions may be overstated because emission factor assumes no control
Residential boilers - coal - oil	3.2 (0.5) (2.7)	3.5 (0.6) (3.0)	*/1991	High	Mass balance; emissions may be overstated because emission factor assumes no control
Sewage sludge incinerators	1.7	1.8	1975-89/1990	High	Test data
Crematories	0.4	0.4	1992/1991	High	Engineering judgment
Wood-fired boilers ^f	0.3	0.3	1984-92/1980	Medium	Test data
Hazardous waste combustors	-	-	-	-	1995 emissions estimate is 5.0 tons/year for incinerators and light-weight aggregate kilns only
Total	186.9	205.2 84.9			

^a Date that data emission factor is based on/Date of activity factor used to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

^c In the course of an MWI rulemaking, with the receipt of new data, U.S. EPA expects to revise the mercury emission estimate for MWIs downward.

^d Parentheses denote subtotal within a larger point source category.

^e Date of data used to develop emission factor was not determined.

^f Does not include residential emissions.

New Source Performance Standards and Emission Guidelines for MWIs

On February 27, 1995, EPA proposed NSPS for new MWIs and emission guidelines for existing MWIs (60 FR 10654). The standards would apply to all new and existing MWIs that commence construction or modification after February 27, 1995, although sources combusting only pathological wastes would be subject to only certain reporting and recordkeeping provisions. Overall, the NSPS and emission guidelines would implement sections 111 and 129 of the Clean Air Act Amendments of 1990, including the requirement for MWIs to control emissions of air pollutants to levels that reflect the maximum degree of emissions reduction achievable, taking into consideration costs, any non-air-quality health and environmental impacts, and energy requirements (a standard commonly referred to as "maximum achievable control technology" or MACT).

The proposed NSPS would reduce emissions from MWIs by establishing standards that limit emissions from new MWIs. For mercury, the standard would be 0.47 mg/dscm or an 85 percent reduction in mercury emissions (based on dry injection/fabric filter with an activated carbon injection system). The proposed NSPS also would require training and qualification of MWI operators, incorporate siting requirements, specify testing and monitoring requirements to demonstrate compliance with the emission limits, and establish reporting and recordkeeping requirements.

The proposed emission guidelines would reduce emissions from MWIs by requiring States to develop regulations that limit emissions from existing MWIs. The mercury emission limit proposed in the guidelines is the same as that proposed in the NSPS, i.e., 0.47 mg/dscm or an 85 percent reduction in mercury emissions (based on dry injection/fabric filter with an activated carbon injection system). Consistent with the NSPS, the emission guidelines also would require training and qualification of MWI operators, specify testing and monitoring requirements, and establish reporting and recordkeeping requirements. Existing MWIs would have to meet one of the following two compliance schedules: (1) full compliance with an EPA-approved State plan within one year after approval of the plan, or (2) full compliance with the State plan, provided the owner or operator submits information on measurable and enforceable incremental steps of progress that will be taken to comply with the State plan.

known. An estimated 1,200 systems have been installed at U.S. hospitals in the past 20 years (Epner, 1992). About 3,700 MWIs currently operate throughout the country; geographic distribution is relatively even (see Table A-11, Appendix A). Of these 3,700 units, about 3,000 are hospital incinerators, about 150 are commercial units, and the remaining units are distributed among nursing homes, laboratories and other miscellaneous facilities (U.S. EPA, 1994a).

There are an additional 1,305 incinerators burning only pathological waste which are not technically considered to be MWIs. These units are used for disposal of tissue only and are most commonly found at veterinary facilities or animal research facilities. The primary source of mercury in medical waste is mercury-containing products, not tissue. These small incinerators are estimated to contribute 0.12 Mg/year (0.13 tons/year) to the total MWI mercury estimate of 58.8 Mg/year (64.7 tons/year). The reader should note that the proposed NSPS and emission guidelines for MWIs do not apply to either incinerators for pathological waste only or crematories. In this document, crematories are discussed in Section 4.1.7.

The primary functions of MWIs are to render the waste biologically innocuous and to reduce the volume and mass of solids that must be landfilled by combusting the organic material contained within the waste. Currently, three major MWI types operate in the United States: continuous-duty, intermittent-duty and batch type. All three have two chambers that operate on a similar principle. Waste is fed to a primary chamber, where it is heated and volatilized. The volatiles and combustion gases are then sent to a secondary chamber, where combustion of the volatiles is completed by adding

air and heat. All mercury in the waste is assumed to be volatilized during the combustion process and emitted with the combustion stack gases.

A number of air pollution control systems are used to control PM and gas emissions from MWI combustion stacks. Most of these systems fall into the general classes of either wet or dry systems. Wet systems typically comprise a wet scrubber, designed for PM control (venturi scrubber or rotary atomizing scrubber), in series with a packed-bed scrubber for acid gas removal and a high-efficiency mist elimination system. Most dry systems use a fabric filter for PM removal, but ESPs have been used on some of the larger MWIs. These dry systems may use sorbent injection (e.g., lime) via either dry injection or spray dryers upstream of the PM control device for acid gas control. All of these systems have limited success in controlling mercury emissions. Recent U.S. EPA studies, however, indicate that sorbent injection/fabric filtration systems can achieve improved mercury control by adding activated carbon to the sorbent material (U.S. EPA, 1993a). (Activated carbon injection for mercury control at MWIs is discussed in Volume VII of this Report to Congress.)

The estimated mercury emission factors for MWIs were determined by analyzing test data from several MWI facilities tested by U.S. EPA. The uncontrolled emission data collected at eight facilities show wide variability, with mercury emission factors ranging from 0.043 to 317 g/Mg of waste charged (8.6×10^{-4} to 6.3×10^{-1} lb/ton) (U.S. EPA, 1993a). These data represent a variety of waste types (mixed medical waste, red bag [infectious] waste only and pathological waste) and incinerator types (continuous and intermittent units with varied operating practices). Because emissions are strongly related to waste characteristics, separate uncontrolled emission factors were developed for the different waste types. The average uncontrolled mercury emission factor for mixed medical waste was calculated to be 20 g/Mg of waste charged (4.0×10^{-2} lb/ton). The average uncontrolled mercury emission factor for red bag waste was calculated to be 16 g/Mg of waste charged (3.2×10^{-3} lb/ton). The average uncontrolled mercury emission factor for pathological waste was calculated to be 5×10^{-3} g/Mg waste charged (1×10^{-3} lb/ton) (U.S. EPA, 1993a).

Mercury emission estimates for the MWI source category were calculated using model units. Model units are representative facilities that are developed to represent an industry when information from each plant is unavailable. Seven model units were developed to represent the variety of MWIs in the United States (U.S. EPA, 1994b). The models differed by type of MWI operation (continuous, batch and intermittent), waste feed rate and operating hours, among other parameters. All seven model units represent uncontrolled MWIs. The aggregate uncontrolled mercury emissions from the MWI industry were estimated by multiplying the number of MWIs assigned to each model, the emission factors, feed rates and operating hours. Table A-12 in Appendix A presents the characteristics of each model MWI, the number of MWIs in the U.S. that were assigned to each model and the estimated mercury emissions from them. The total uncontrolled mercury emissions were estimated to be 58.8 Mg/yr (64.7 tons/yr). As mentioned above, with the receipt of more recent data, the U.S. EPA expects to revise this estimate downward.

4.1.2 Municipal Waste Combustors

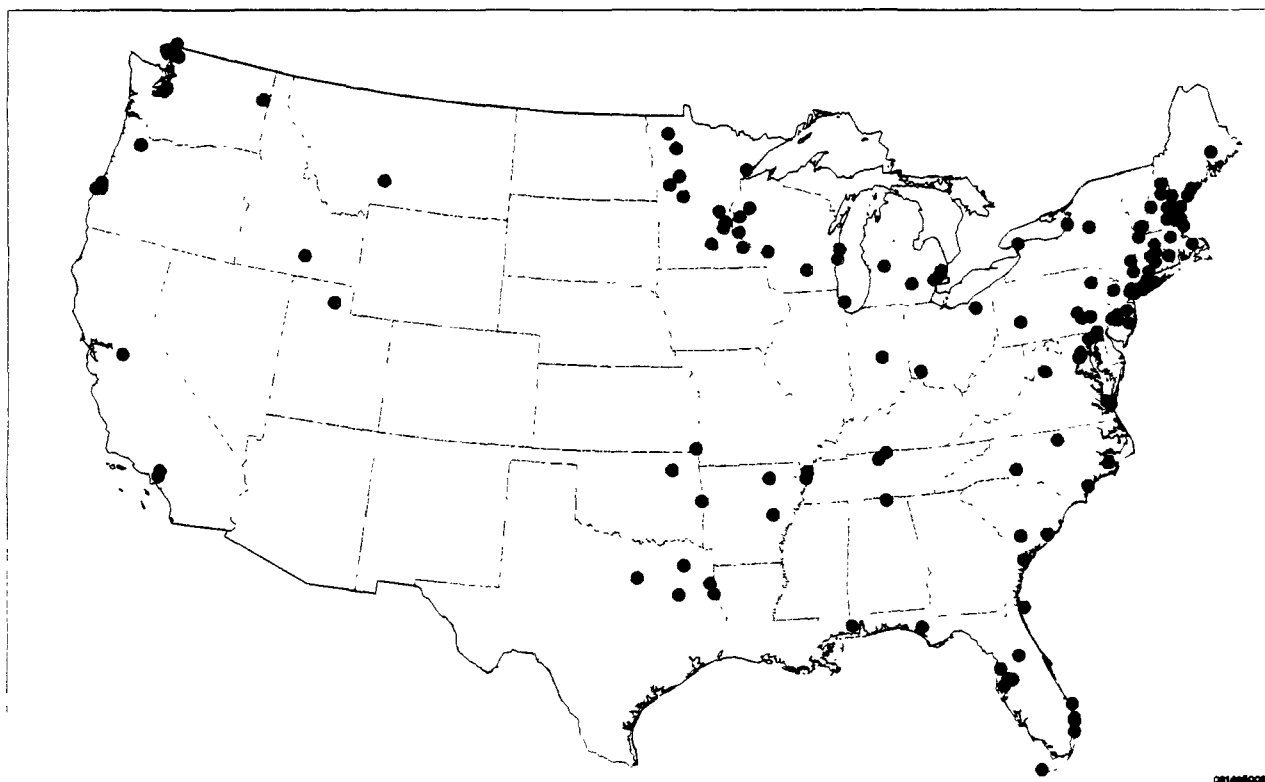
Municipal waste combustors (MWCs) fire municipal solid waste (MSW) to reduce the volume of the waste and produce energy. There are three main types of technologies used to combust MSW: mass burn combustors, modular combustors and refuse-derived fuel-fired (RDF) combustors. A fourth type, fluidized-bed combustors (FBCs), is less common and can be considered a subset of the RDF technology. Modular MWCs characterize the low end of the MWC size range, whereas the mass burn and RDF MWCs tend to be larger. Both the mass burn and modular MWCs fire waste that has undergone minimal pre-processing, other than the removal of bulky items. The RDF combustors fire

MSW that has been processed to varying degrees, from simple removal of bulky and noncombustible items, to extensive processing to produce a fuel suitable for co-firing in pulverized coal-fired boilers. Of the three main combustor types, mass burn combustors are the predominant technology used and are found in three kinds: mass burn/waterwall (MB/WW), mass burn/refractory wall (MB/REF) and mass burn/rotary waterwall (MB/RC). The MB/WW technology is the most common type, especially for newer MWCs. With the exception of the refractory wall combustors and some of the modular combustors, the majority of MWCs incorporate energy recovery (Fenn and Nebel, 1992).

At the beginning of 1995, there were over 130 MWC plants with aggregate capacities greater than 36 Mg/d (40 tons/d) of MSW operating in the United States. There have been a number of plant closures in this source category since 1991. The inventory described here represents 37 fewer facilities in this size range than reported by U.S. EPA in 1993 (U.S. EPA, 1993d). The number of combustion units per facility ranges from one to six, with the average being two. Total facility capacity ranges from 36 to 2,700 Mg/d (40 to 3,000 tons/d). These plants have a total capacity of approximately 90,000 Mg/d (99,000 tons/d). A geographic distribution of the MWCs and their capacities are presented in Table A-9, Appendix A (Fenn and Nebel, 1992). This distribution reflects MWC's that were operational in January 1995.

In addition to the MWCs discussed above, there are a number of smaller MWCs in the United States (with plant capacities of less than 36 Mg/d [40 tons/d]). This population of smaller MWCs comprises less than one percent of the nation's total MWC capacity (Fenn and Nebel, 1992). Since 1991, there have been 13 MWCs in this size range that have closed. Table A-9 in Appendix A, as well as the map shown in Figure 4-1, reflects the 1995 MWC population.

Figure 4-1
Municipal Waste Combustor Facilities



4.1.2.1 Mercury Emissions and Controls

Mercury emissions from MWCs occur when mercury in the MSW vaporizes during combustion and is exhausted through the combustor stack. There are numerous sources of mercury in MSW. These include electric switches and lighting components, paint residues and thermometers.

More than 85 percent of the MWC plants (99 percent of the MWC capacity) in the United States employ some kind of APCD (Fenn and Nebel, 1992). These controls range from the use of electrostatic precipitators (ESPs) alone to control PM, to the use of acid gas controls (e.g., dry lime injection, spray drying) in combination with an ESP or a fabric filter. New MWCs employ the latter combination of controls plus the application of activated carbon injection technology. Mercury control in APCDs without supplemental carbon injection technology is variable since mercury exists as a vapor at the typical APCD operating temperatures. Factors that enhance mercury control are low temperatures in the APCD system (less than 150 to 200°C [300 to 400°F]), the presence of an effective mercury sorbent and a method to collect the sorbent (Nebel and White, 1992). In general, carbon present in the fly ash enhances mercury sorption onto PM, which can then be captured in the PM control device. Most modern MWCs, excluding RDF combustors, have low levels of carbon in the fly ash and good carbon burnout, representative of efficient and complete combustion; thus, there is little carbon to adsorb the mercury. RDF combustors generally have higher PM loadings and higher carbon contents at the combustor exit because of the suspension firing of the RDF in the combustor. As a result, mercury levels for RDF MWCs with acid gas control alone (flue gas cooling) are lower than for other combustors (Nebel and White, 1991). With the additional application of carbon injection technology, non-RDF combustors achieve 85 to 95 percent mercury control with resulting emissions similar to RDF combustors. Since 1994, 15 MWC units have initiated commercial operation with carbon injection technology for mercury. The average performance level is 93 percent mercury control.

Add-on mercury control techniques include the injection of activated carbon or Na_2S into the flue gas prior to the PM control system. These technologies are now being used commercially on some MWCs in the U.S., and on MWCs in Europe, Canada and Japan where removal efficiencies have been reported to range from over 50 percent to 90 percent. Recent test programs using activated carbon and Na_2S injection conducted in the U.S. showed mercury removal efficiencies ranging from 50 percent to over 95 percent (U.S. EPA, 1993a). There are currently four MWCs in the U.S. that are being controlled with activated carbon injection in conjunction with PM control. Greater than 95 percent control of mercury emissions is being achieved. State regulations in Florida and New Jersey will require MWCs in these states to retrofit with activated carbon injection by the end of 1995.

Emission factors for mercury have been developed from test data gathered at several MWCs. The emission factors for various combinations of combustors and control devices are presented in Table A-10, Appendix A. Estimated mercury emissions were determined based on the tonnage of the waste being combusted (Table A-9 in Appendix A) and on these emission factors (U.S. EPA, 1992b; Waste Age, 1991). Multiplying the processing rates by the uncontrolled emissions and taking into account the different control efficiencies (all found in Table A-10, Appendix A) gives an estimated total baseline mercury emissions of 50 Mg/yr (55 tons/yr) in 1990. As described below, the 1995 emission estimate for MWCs is considerably lower.

Industry estimates of mercury emissions from this source category for 1990 are 40 Mg/year (44 tons/year) (Kiser, 1991). The difference between the assumptions that U.S. EPA used and the assumptions that industry used are described in Table A-9, Appendix A. The primary differences in

the assumptions are related to the average mercury emission rate and the extent of mercury reduction that can be attributed to an ESP.

Mercury emissions from MWCs have declined since 1990 and will continue to decline in the future for three important reasons. First, under section 129 of the CAA, U.S. EPA is required to develop emission limits for mercury (and a number of other pollutants) being emitted from MWCs. On October 31, 1995, the U.S. EPA Administrator signed New Source Performance Standards (NSPS) and emission guidelines for new and existing MWCs that have the capacity to burn more than 35 Mg MSW/day (39 tons/day) (see box below). The NSPS and emission guidelines, when fully implemented, are estimated to reduce mercury emissions by about 90 percent, from the 1990 baseline of 50 Mg/year (55 tons/year) to 4.0 Mg/year (4.4 tons/year).

New Source Performance Standards and Emission Guidelines for MWCs

On September 20, 1994, the U.S. EPA proposed New Source Performance Standards (NSPS) and Emission Guidelines (EG) applicable to MWC plants larger than 35 Mg/day (39 tons per day) capacity. The U.S. EPA finalized these regulations on October 31, 1995. The NSPS (Subpart Eb) applies to new MWC plants constructed after September 20, 1994 and the EG (Subpart Cb) applies to MWC plants constructed before September 20, 1994. For some of the pollutants regulated by the NSPS and EG, the NSPS is more stringent than the EG. For mercury, the same emission control requirements apply to new MWCs (NSPS) and existing MWCs (EG). The final mercury standard for new and existing MWCs is 0.08 mg/dscm or about 90 percent control.

Second, as described in the following sections, many of the mercury-containing components that comprise MSW have declined. These include household batteries where mercury use is being discontinued and paint residues and pigments where mercury additives have been phased out. Based on the status of all MWC facilities in 1995, the U.S. EPA estimates national mercury emissions from MWCs to be 26.3 Mg/yr (29 tons/yr). This estimate incorporates changes in MWC mercury emission levels resulting from (1) installation of APCDs on new and some existing MWCs that achieve moderate mercury control, (2) retirement of several existing MWCs, and (3) significant reductions in the mercury content of mercury-containing components of municipal waste, as described above. As a result, the inlet concentration of mercury in the MWC waste stream is estimated to be, on average, half of what the concentration was in 1990. As mentioned above, full implementation of the 1995 emissions guidelines (retrofit of carbon injection technology to existing MWCs) will result in national mercury emissions from MWCs being reduced to 4.4 tons per year.

Third, some States have enacted either MWC legislation requiring the use of activated carbon injection, recycling or bans on the sale of certain mercury-containing products. These efforts will decrease both the amount of mercury being emitted from MWCs and the amount of mercury in MSW in general. Florida, New Jersey and Minnesota have led State efforts in this area. Volume VII of this Mercury Report to Congress summarizes the legislative, regulatory and other programs of several states that influence mercury use and disposal.

4.1.2.2 MSW Components and Trends

MSW consists primarily of household garbage and other commercial, institutional and industrial solid wastes. The known sources of mercury in MSW are batteries (mercuric oxide), discarded electrical equipment and wiring, fluorescent bulbs, paint residues and plastics. In 1989, the

estimated mercury content of MSW was 664 Mg (709 tons), with concentrations ranging from 1 to 6 ppm by weight and a typical value being 4 ppm by weight (U.S. EPA, 1993a).

The U.S. EPA's Office of Solid Waste (OSW) estimates that 55 to 65 percent of MSW comes from residential sources, while 35 to 45 percent comes from commercial sources (U.S. EPA, 1992g). One recent study identified and reported a number of specific sources of mercury in MSW, as summarized in Tables 4-3 and 4-4. The data from Table 4-4 are shown graphically for the year 1989 in Figure 4-2. These figures show that in 1989 household batteries were the largest contributing source of mercury to MSW. Light bulbs, paint residues, thermometers, thermostats, and pigments contribute most of the remainder of mercury to MSW. However, as discussed in the subsections that follow, mercury in batteries and paint residues have decreased significantly in the 1990s.

In general, from an examination of Bureau of Mines data for mercury use, it can be inferred that the components of MSW that will be the main sources of mercury in the future will be in the electrical lighting and wiring devices and switches sectors, as well as fever thermometers.

Batteries

Major types of batteries include alkaline, mercuric oxide, silver oxide, and zinc air batteries. Another kind of battery, carbon zinc, is produced and discarded at a substantially lower rate.

In 1989, alkaline batteries accounted for about 419 tons or close to 60 percent of the mercury in the MSW stream (U.S. EPA, 1992a). Although the quantity of mercury in each alkaline battery is minimal, the large number sold and discarded has made these batteries the largest single source of mercury in MSW historically. The contribution from this source category, however, is declining dramatically due largely to industry initiatives to reduce and ultimately eliminate mercury from alkaline batteries.

Mercury has been used in alkaline manganese batteries as an additive to suppress formation of internal gases which would lead to leakage, possible explosions and/or short shelf life. In the U.S., alkaline batteries in the mid-1980's contained mercury in amounts from about 0.8 percent to about 1.2 percent of the battery weight. Between late 1989 and early 1991, all U.S. manufacturers converted production so that the mercury content, except in button and "coin" cells, did not exceed 0.025 percent mercury by weight (National Electrical Manufacturers Association, undated).

Mercuric oxide batteries include cylinder-shaped batteries (such as those used in hospital applications) and button-shaped batteries (such as those used in hearing aids, electronic watches, calculators, etc.). Larger mercuric oxide batteries are used in a variety of medical devices. The mercury content of mercuric oxide batteries is 30 to 40 percent of the weight of the battery and cannot be reduced without proportionately reducing the energy content of the battery. In 1989, these batteries contributed an estimated 196 tons (or about 28 percent) of mercury discards to MSW. Although mercuric oxide batteries are estimated to continue to be a large source of mercury in MSW on a percentage basis (Solid Waste Association of North America, 1993), the total tonnage of mercury discarded in such batteries is expected to decline in the future due to the increase in use of alkaline and zinc air batteries for these applications. Presently, mercuric oxide batteries are essentially the remaining source of mercury in MSW from batteries.

Table 4-3
Estimated Discards of Mercury in Products in Municipal Solid Waste (in tons)^a

Products	In Tons ^{b,c}						
	1970	1975	1980	1985	1989	1995	2000
Batteries							
Alkaline	4.1	38.4	158.2	352.3	419.4	41.6*	0.0
Mercuric oxide	301.9	287.8	266.8	235.2	196.6	131.5*	98.5*
Others	4.8	4.7	4.5	4.5	5.2	3.5*	0.0
Subtotal Batteries	310.8	330.9	429.5	592.0	621.2	176.6*	98.5*
Electric Lighting							
Fluorescent Lamps	18.9	21.5	23.2	27.9	26.0	14.7 ^d	11.6 ^d
High Intensity	0.2	0.3	1.1	0.7	0.8	1.0	1.2
Lamps	19.1	21.8	24.3	28.6	26.7	15.7	12.6
Subtotal Lighting							
Paint Residues	30.2	37.3	26.7	31.4	18.2	2.3	0.5
Fever Thermometers	12.2	23.2	25.7	32.5	16.3	16.9	16.8
Thermostats	5.3	6.8	7.0	9.5	11.2	8.1	10.3
Pigments	32.3	27.5	23.0	25.2	10.0	3.0	1.5
Dental Uses	9.3	9.7	7.1	6.2	4.0	2.9	2.3
Special Paper Coating	0.1	0.6	1.2	1.8	1.0	0.0	0.0
Mercury Light Switches	0.4	0.4	0.4	0.4	0.4	1.9	1.9
Film Pack Batteries	2.1	2.3	2.6	2.8	0.0	0.0	0.0
Total Discards	421.8	460.5	547.5	730.4	709.0	227.6	144.6

^a U.S. EPA, 1992a (except for fluorescent lamps estimates).

^b Discards before recovery.

^c One ton equals 2000 pounds.

^d The estimated contribution of mercury from fluorescent lamps disposal to MSW was calculated based on industry estimates of a 4 percent growth rate in sales in conjunction with a 53 percent decrease in mercury content between 1989 and 1995, and a further 34 percent decrease in mercury content by the year 2000 (to 15 mg of mercury per 4 foot fluorescent lamp) (National Electric Manufacturers Association, 1995).

* NOTE: The estimates for the years 1995 and 2000 do not reflect recent state, Federal or battery manufacturers' efforts to reduce the mercury content of batteries. Since the referenced report was released (U.S. EPA, 1992a), several states have restricted the mercury content of alkaline batteries and/or banned the sale of mercuric oxide batteries. Federal legislation to restrict mercury use in batteries is pending. The battery industry has eliminated mercury as an intentional additive in alkaline batteries, except in button cells.

Table 4-4
Estimated Discards of Mercury in Products in Municipal Solid Waste (in percent) ^a

Products	In Percent of Total Discards ^b						
	1970	1975	1980	1985	1989	1995	2000
Batteries^c							
Alkaline	1.0	8.3	28.9	48.2	59.2	17.0*	0.0
Mercuric oxide	71.6	62.5	48.7	32.2	27.7	53.6*	68.0*
Others	1.1	1.0	0.8	0.6	0.7	1.4*	0.0
Subtotal Batteries	73.7	71.9	78.4	81.1	87.6	72.0*	68.0*
Electric Lighting							
Fluorescent Lamps	4.5	4.7	4.2	3.8	3.7	13.3	8.0
High Intensity Lamps	0.0	0.1	0.2	0.1	0.1	0.4	0.8
Subtotal Lighting	4.5	4.7	4.4	3.9	3.8	13.7	8.7
Paint Residues	7.2	8.1	4.9	4.3	2.6	0.9	0.4
Fever Thermometers	2.9	5.0	4.7	4.4	2.3	6.9	11.6
Thermostats	1.3	1.5	1.3	1.3	1.6	3.3	7.1
Pigments	7.7	6.0	4.2	3.5	1.4	1.2	1.0
Dental Uses	2.2	2.1	1.3	0.8	0.6	1.2	1.6
Special Paper Coating	0.0	0.1	0.2	0.2	0.1	0.0	0.0
Mercury Light Switches	0.1	0.1	0.1	0.1	0.1	0.8	1.3
Film Pack Batteries	0.5	0.5	0.5	0.4	0.0	0.0	0.0
Total Discards	100.0	100.0	100.0	100.0	100.0	100.0	100.0

^a U.S. EPA, 1992a.

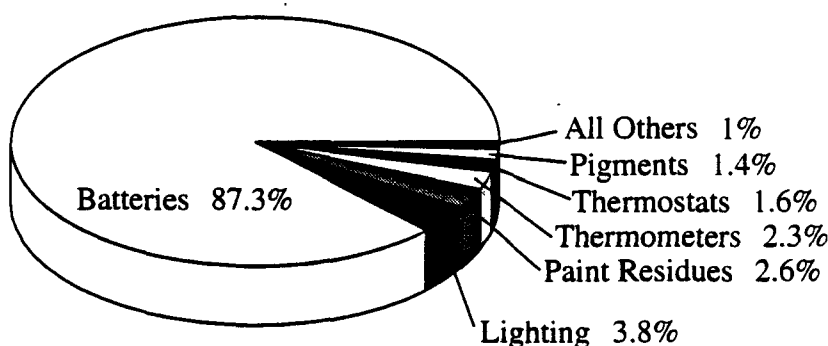
^b Discards before recovery.

^c Data were based on the assumption that batteries are discarded 2 years after purchase.

Details may not add to totals due to rounding.

* NOTE: The estimates for the years 1995 and 2000 do not reflect recent state, Federal or battery manufacturers' efforts to reduce the mercury content of batteries. Since the referenced report was released (U.S. EPA, 1992a), several states have restricted the mercury content of alkaline batteries and/or banned the sale of mercuric oxide batteries. Federal legislation to restrict mercury use in batteries is pending. The battery industry has eliminated mercury as an intentional additive in alkaline batteries, except in button cells.

Figure 4-2
Discards of Mercury in Municipal Solid Waste, 1989



Total mercury discards = 709 tons

Silver oxide, zinc air and carbon zinc batteries contributed an estimated 5 tons (or about 1 percent) of mercury discards in MSW in 1989. Because production of carbon zinc batteries is declining, and because these batteries have been converted to "no mercury added" designs, discards of mercury in carbon zinc batteries will decline. Production and discards of silver oxide and zinc air batteries are increasing, but mercury use has been discontinued in these types of batteries since 1992 (National Electric Manufacturers Association, undated).

Table 4-5 presents the estimated amount of mercury entering the MSW stream by year and battery type. However, it is important to note the estimates for the years 1995 and 2000 do not reflect recent state, Federal or battery manufacturers' efforts to reduce the mercury content of batteries.

Recent efforts to reduce the mercury content of batteries have included either passed or pending legislation in (at the time of this writing) at least 18 states. As just one example, Minnesota has passed a law requiring manufacturers to sell alkaline batteries containing no more than 0.025 percent mercury by weight and, by January 1, 1996, prohibiting manufacturers from selling alkaline batteries containing any mercury. Efforts at the Federal level include pending legislation that would eliminate the use of mercury in all types of batteries used in both industrial and household activities by January 1, 1997. The battery industry, as mentioned previously, also has eliminated mercury as an intentional additive in alkaline batteries, except in button cells. Several communities have established programs to recover household batteries in order to prevent the disposal of batteries in landfills or incinerators. At the time of this writing, three U.S. firms had the facilities necessary to recycle the mercury in batteries. U.S. EPA estimates that approximately 5 percent of the mercury in batteries will be recovered through recycling in 1995 and approximately 20 percent will be recovered by 2000.

Table 4-5
Estimated Discards of Mercury in Batteries^a

In Tons ^b				Year Discarded
Alkaline	Mercuric Oxide	Silver Oxide	Zinc Air	
4.1	301.9	0.1	0.0	1970
38.4	287.8	0.2	0.2	1975
158.2	266.8	0.3	0.3	1980
352.3	235.2	0.5	0.7	1985
443.6	182.5	1.1	2.4	1990
390.5	172.0	1.1	2.9	1991
41.6*	131.5*	0.7	2.0	1995
0.0	98.5*	0.0	0.0	2000

^a U.S. EPA, 1992a.

^b One ton equals 2000 pounds.

* NOTE: The estimates for the years 1995 and 2000 do not reflect recent state, Federal or battery manufacturers' efforts to reduce the mercury content of batteries. Since the referenced report was released (U.S. EPA, 1992a), several states have restricted the mercury content of alkaline batteries and/or banned the sale of mercuric oxide batteries, Federal legislation to restrict mercury use in batteries is pending. The battery industry has eliminated mercury as intentional additive in alkaline batteries, except in button cells.

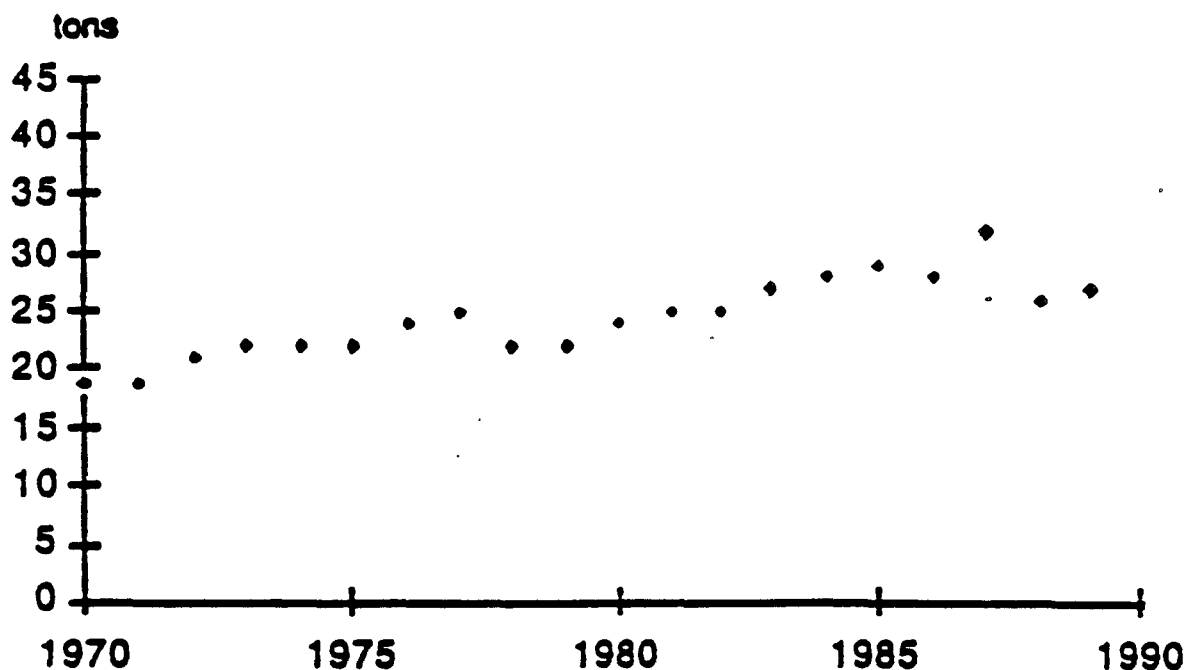
Chapter 4 of Volume VII of this Mercury Study Report to Congress provides additional discussion of various laws and programs for controlling battery production, use and disposal.

Electric Lighting

Fluorescent lamps (bulbs) and high intensity lamps (bulbs) used in lighting streets, parking lots, etc. were considered the second largest source of mercury in MSW in 1989 (U.S. EPA, 1992a). It is estimated that fluorescent lamps accounted for about 26 tons of mercury in MSW (or 3.7 percent of total discards) in 1989. All lighting sources were estimated to contribute about 27 tons of mercury in the same year. Figure 4-3 illustrates the estimated historical discards of electric lighting sources.

Future projections of mercury discards from electric lighting sources depend on the sales of lamps and their mercury content. Sales of fluorescent lamps increase between 3 and 5 percent a year. As described in section 3.2 of this Volume, the mercury content of fluorescent lamps has decreased by 53 percent between 1989 and 1995 to 22.8 mg of mercury per lamp. Assuming a 4 percent increase in sales and a 53 percent decrease in mercury, estimated discards of mercury would be 14.7 tons in 1995. Assuming a 4 percent increase in sales and an additional 34 percent decrease in mercury content between 1995 and 2000 (to 15 mg mercury per lamp) leads to an estimated 11.6 tons per year in discards in the year 2000.

Figure 4-3
Estimated Discards of Mercury in Electric Lighting in Municipal Solid Waste



Paint Residues

Mercury is no longer used in paint manufacture; however, paint cans with traces of mercury are still discarded. It was estimated that about 18 tons of mercury were discarded in paint residues in 1989. Mercury from paint residues is expected to decline significantly due to U.S. EPA's recent ban on mercury use in interior and exterior paints. Table 4-6 presents estimated mercury discards from paint residues from 1970 to 2000.

Fever Thermometers

An estimated 16.3 tons of mercury were discarded in thermometers in 1989. It is estimated that digital thermometers will gain an additional 1 to 2 percent of the market each year from 1990 through 2000, and the mercury content of mercury thermometers will remain constant (U.S. EPA, 1992a). Tables 4-3 and 4-4 illustrates the estimated discards of mercury from thermometers in MSW from 1970 to 2000.

Table 4-6
Estimated Discards of Mercury in Paint Residues^a

Year	Total Discards in Residues (In Tons)
1970	30.2
1975	37.3
1980	26.7
1985	31.4
1988	23.1
1990	17.5
1995	2.3
2000	0.5

^a U.S. EPA, 1992a.

Thermostats

Mercury thermostats are being replaced with digital thermostats. It is expected that thermostats, however, will still be a source of mercury in MSW through the year 2000 because of the long life of mercury thermostats. Mercury thermostats contributed an estimated 11 tons of mercury to the MSW stream in 1989 (U.S. EPA, 1992a). The estimated historical trends in mercury thermostat discards are presented in Table 4-7. Federal legislation finalized in 1995 will encourage the recycling of thermostats rather than their disposal. Recyclings efforts are discussed in section 4.2.7.1 of this Volume. As a result of recycling programs, mercury discards from thermostats are expected to decline.

Table 4-7
Estimated Discards of Mercury in Thermostats^a

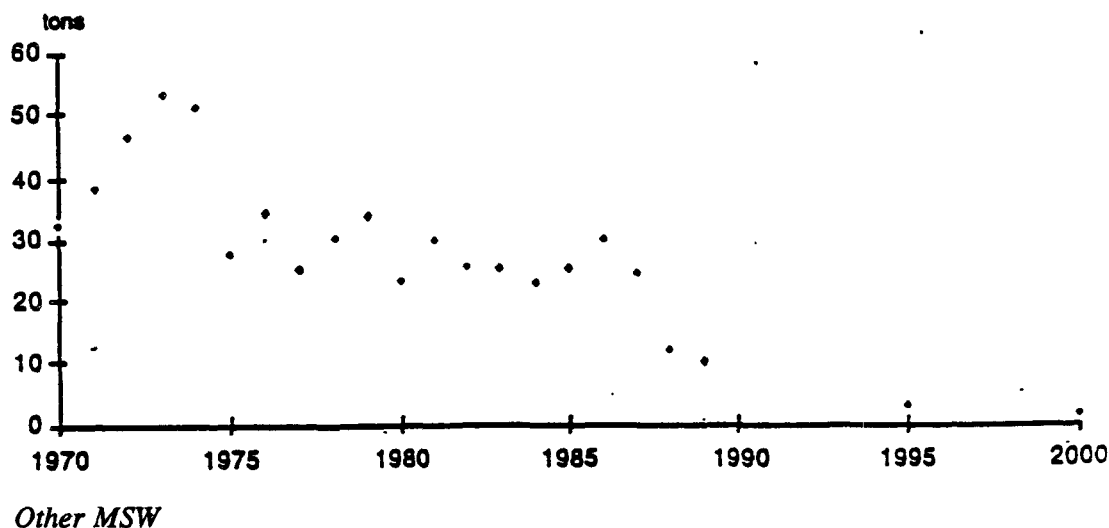
Year	Total Mercury (In Tons)
1970	5.3
1975	6.8
1980	7.0
1985	9.5
1988	10.7
1989	11.2
1995	8.1
2000	10.3

^a U.S. EPA, 1992a.

Pigments

Based on available data, one report estimated that 10 tons of mercury in pigments were discarded in 1989. This accounted for less than 2 percent of total mercury discards. Most of the mercury used in pigments is used in plastics, paints, rubber, printing inks, and textiles. As shown in Figure 4-4, estimated discards of mercury in MSW pigments have generally been trending downwards since 1970 (U.S. EPA, 1992a).

Figure 4-4
Estimated Discards of Mercury in Pigments in Municipal Solid Waste



Dental amalgams, a special paper coating used with cathode ray tubes, and mercury light switches contributed less than 1 percent of the mercury in MSW in 1989. Plans are underway to discontinue manufacture of the special paper by 1995. Mercury light switches are an increasing source of mercury in MSW. One study projects that 2 tons of mercury will be discarded to MSW from mercury light switches in the year 2000, which would account for about 1 percent of total discards in that year (U.S. EPA, 1992a).

Several additional sources of mercury have been found in MSW, but have not been quantified. For example, mercury was a component of batteries used in instant camera film packs, but these batteries were discontinued in 1988. Mirrors, glass, felt, outdoor textiles, and paper are other sources of mercury to MSW.

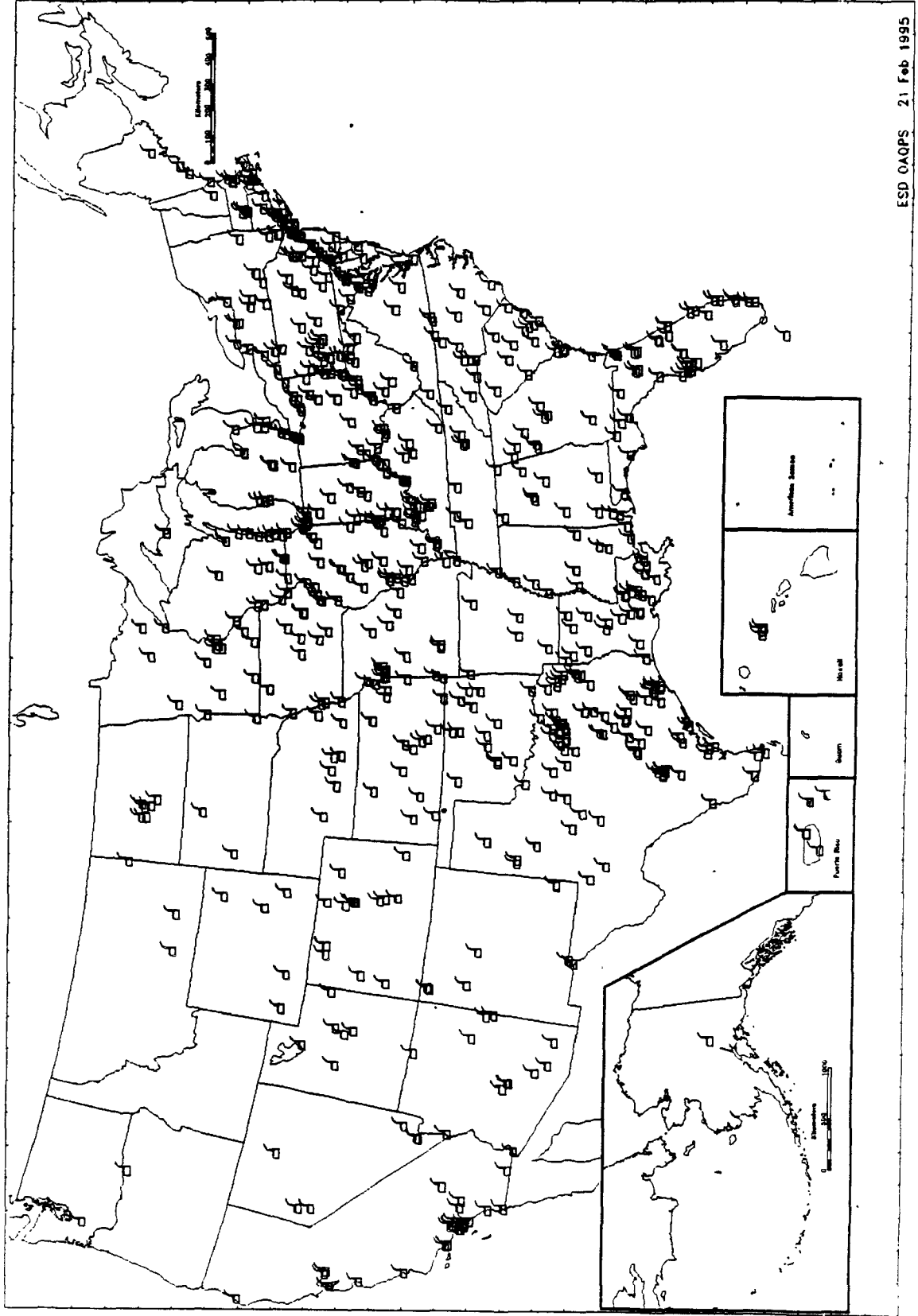
In the production of paper, mercury compounds were formerly used as slimicides to prevent the growth of green slime on the manufacturing equipment. Mercury compounds also were used to prevent the growth of mold and bacteria on pulp during storage, but this practice has been discontinued (U.S. EPA, 1992a).

4.1.3 Utility Boilers

Utility boilers are large boilers used by public and private utilities to generate electricity. Such boilers can be fired by coal, oil, natural gas, or some combination of these fuels (U.S. EPA, 1993a). Figure 4-5 shows the locations of operating utility boilers across the United States.

In 1990, utility boilers consumed fossil fuel at an annual level of 21×10^{12} megajoules (MJ) (20×10^{15} British thermal units [Btu]). About 80 percent of this total energy consumption resulted from coal combustion, 6 percent from oil and petroleum fuels and 14 percent from natural gas consumption (U.S. Department of Energy, 1992). In terms of coal usage, the majority of total nationwide coal combustion (about 84 percent) is in utility boilers. Almost all of the coal burned in the U.S. is bituminous and subbituminous (95 percent) while only 4 percent is lignite (Brooks, 1989). The combustion processes used for these different coals are comparable. The most common liquid

Figure 4-5
Utility Boiler Locations



fuel used by utility boilers is fuel oil derived from crude petroleum. Fuel oils are classified as either distillate or residual.

4.1.3.1 Description of the Different Utility Boiler Types

Because there is no evidence to show that mercury emissions are affected by boiler type, this section presents only a brief discussion of different boiler types and combustion techniques. More information on boiler types may be found in the Air Pollution Engineering Manual, *AP-42, Steam: Its Generation and Use*, and the L&E document (Buonicore and Davis, 1992; U.S. EPA, 1988a; Babcock and Wilcox, 1975; U.S. EPA, 1993a).

Although several options are available for each component of a utility operation, the overall process for coal-fired utility boilers is straightforward. Coal is received at the plant, typically by rail or barge, unloaded and transferred to storage piles or silos. From storage, the coal is subjected to mechanical sizing operations and then charged to the boiler. Coal-fired boilers are typically suspension-fired pulverized coal or cyclone systems. The other major process component is the ash-handling system for the bottom ash and the fly ash that is collected in the air pollution control system (U.S. EPA, 1988a).

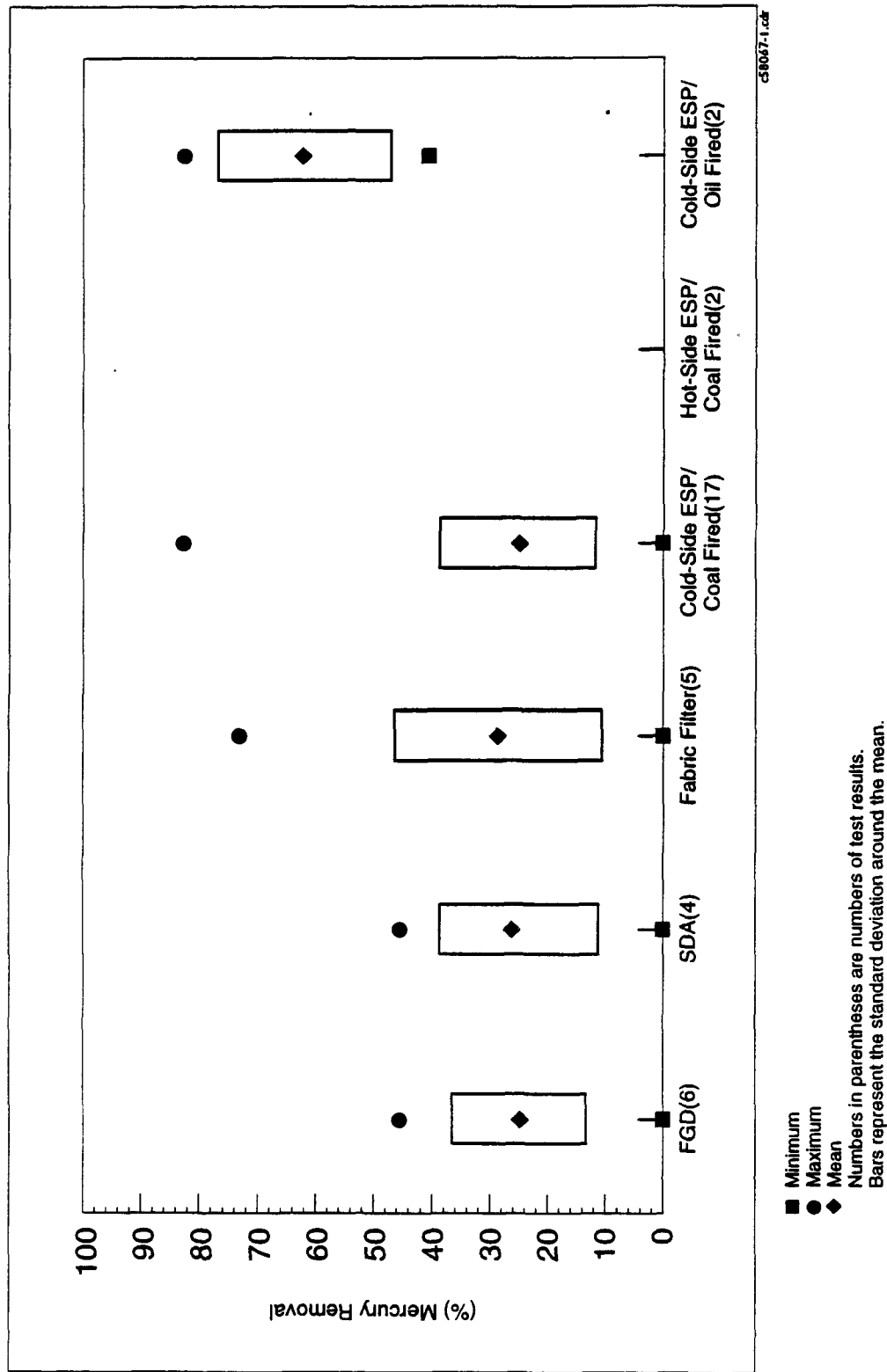
Oil-fired utility boilers are even simpler and have less variation in design than do the coal-fired systems. Oil is received by barge, rail, truck, or pipeline and transferred to storage tanks. From there the oil is fired to the boiler system. The main components of the system are the burner and the furnace. The primary difference in systems that fire distillate and residual oils is the presence of an oil preheater in residual systems (U.S. EPA, 1988a; Buonicore and Davis, 1992).

4.1.3.2 Effectiveness of Particulate Matter and Acid Gas Air Pollution Controls for Mercury

Although small quantities of mercury may be emitted as fugitive particulate matter (PM) from coal storage and handling, the primary source of mercury from both coal and combustion in utility boilers is the combustion stack. Because the combustion zone in boilers operates at temperatures above 1100°C (2000°F), mercury in the coal and oil is vaporized and exhausted as a gas. Some of the gas may cool and condense as it passes through the boiler and the air pollution control device (APCD). The primary types of control devices used for coal-fired utility boilers include electrostatic precipitators (ESPs); wet scrubbers; fabric filters or baghouses (FFs), which are typically used as a component of a dry flue gas desulfurization system (FGDs); and mechanical collectors. Mercury control efficiencies for each of the control devices are presented in Figure 4-6. The test data used to calculate the removal efficiencies described below are shown in more detail in Appendix B.

ESPs are the most widely used control device by the fossil fuel-fired electric utility industry. Because mercury in electric utility flue gas is predominantly in the vapor phase (Clarke and Sloss, 1992), with only about 5 to 15 percent in the fly ash (Noblett et al., 1993), ESPs are relatively ineffective at removing mercury compounds from flue gases. Cold-side ESPs, located after the air preheater have a median mercury removal efficiency of 16.2 percent for coal-fired units, with actual test data ranging from no control (zero percent removed) to 82.4 percent removal (Interpoll Laboratories, 1992a; Interpoll Laboratories, 1992b; Interpoll Laboratories, 1992c; Radian Corporation, 1993a; Interpoll Laboratories, 1992d; Interpoll Laboratories, 1992e; Radian Corp., 1992a; Radian Corp., 1993a; Radian Corp., 1993b; Radian Corp., 1993e; Radian Corp., 1994a; Battelle, 1993a; Battelle, 1993c; EPRI, 1993a; EPRI, 1993b; EERC, 1993; and Weston, 1993b). Cold-side ESPs were found to have a median mercury removal efficiency of about 62.4 percent in two tests of oil-fired units, with a range from 41.7 to 83 percent removal (Carnot, 1994b; Carnot, 1994c). Data from one

Figure 4-6
Comparison of Mercury Removal Efficiencies Without Activated Carbon Injection



Data and references used to produce this figure are presented in Appendix B.

emission test for a hot-side ESP, located before the air preheater, indicated no mercury control on a coal-fired unit (Southern Research Institute, 1993b).

Scrubbers or FGD units for coal-fired plants are generally used as devices for removal of acid gases (mainly SO₂ emissions). Most utility boilers have an ESP or a FF before the wet FGD units to collect the majority of PM. FGD units have a median mercury removal efficiency of about 17.3 percent, with a range from 0 percent to 59 percent removal (Interpoll Laboratories, 1991; Interpoll Laboratories, 1990a; Radian, 1993a; Radian, 1993b; EPRI, 1993a, Battelle, 1993a). One emission test across an ESP/wet-FGD (spray-tower absorber) system showed a mercury removal efficiency of 82 percent (Radian Corporation, 1993b).

A spray dryer adsorber (SDA) is a dry scrubbing system followed by a particulate control device. A lime/water slurry is sprayed into the flue gas stream and the resulting dried solids are collected by an ESP or a FF. Tests conducted on a SDA/FF system had a median mercury removal efficiency of 24 percent, with a range from 0 percent to 55 percent removal (Radian 1993c; Southern Research Institute, 1993a; Interpoll Laboratories, 1991; Interpoll Laboratories, 1990b).

Fabric filters are more effective than ESPs at collecting fine particles. This performance may be important in achieving better mercury removal. Also, the mercury may adsorb onto the fly ash cake that is collected on the fabric and allow more residence time for mercury removal. FFs have a median mercury removal efficiency of 8 percent, with a range from no control (zero percent removal) to 73 percent removal (Radian Corporation, 1993d; Carnot, 1994a; Interpoll, 1992d; Battelle, 1993b; Weston, 1993a).

Mechanical collectors typically have very low PM collection efficiencies, often lower than 20 percent for particles less than or equal to 1 µm in size. These devices are used as gross particulate removal devices before ESPs or as APCDs on oil-fired units. Venturi scrubbers can be effective for particulate control, but require high pressure drops (more than 50 or 60 in. of water) for small particles. Even with high pressure drops, ESPs and FFs are normally more effective for submicron particles. Mechanical collectors and venturi scrubbers are not expected to provide effective mercury removal, especially for those mercury compounds concentrated in the sub-micron PM fractions and in the vapor phase.

4.1.3.3 Estimated National Mercury Emissions from Utility Boilers

To estimate national mercury emissions from utility boilers, data were gathered on the type of fuel burned, the mercury content of each fuel and the amount of fuel consumed per year by each individual unit (boiler). Data on plant configurations, unit fuel usage and stack parameters (on a boiler-specific basis) were obtained from the Utility Data Institute (UDI)/Edison Electric Institute (EEI) Power Statistics database (1991 edition). The UDI/EEI database is compiled from Form EIA-767, which electric utilities submit on a yearly basis to the U.S Department of Energy's Energy Information Administration. Emissions were only calculated for operational or stand-by units. Previous estimates were based on the assumption that all the mercury present in the fuel would be emitted in the stack gas (U.S. EPA, 1993d). In addition, previous estimates did not attribute any mercury reductions to coal cleaning. As explained below, the estimates presented in this report do account for reductions in the mercury content of coal due to coal cleaning and considers any mercury reductions achieved by existing control devices.

Calculation of utility mercury emissions was a two-step process. First, the amount of mercury in the fuel was estimated as described below. The calculated mercury concentration in the fuel

multiplied by the fuel feed rate resulted in an estimate of the amount of mercury (in kg/year) entering each boiler. Next, based on test data, "emission modification factors" (EMFs) were developed that are specific to various boiler configurations and control devices. The EMFs basically represent the level of mercury control seen across various boiler configurations and control devices. (The control devices are those that are currently installed on boilers principally for nitrogen oxide, sulfur dioxide and PM control.) The EMFs developed from the tested units were applied to all other similar units in the U.S. to give mercury emission estimates on a per-unit basis.

Only coal, oil and natural gas were considered because these fuels account for nearly 100 percent of the fuels fired by utility boilers. The mercury content of these fuels varies greatly, with coal containing the most mercury and natural gas containing almost none.

Mercury Concentrations in Oil and Natural Gas. The mercury concentration in as-fired oil and natural gas was estimated from emissions test data for boilers burning these fuels. In the estimation of mercury emissions, all oil-fired units were assumed to burn residual oil because trace element data were available only for residual oil. An average density of 8.2 lb/gal was chosen to represent all residual oils. Trace element analysis of natural gas was performed for only two available emissions tests; these concentrations were averaged. The calculated mercury concentration in the oil and natural gas multiplied by the fuel feed rate resulted in an estimate of the amount of mercury (in kg/year) entering each oil- and natural gas-fired boiler.

Mercury Concentrations in Coal. Mercury concentrations were estimated for bituminous, subbituminous and lignite coals. The mercury concentration of anthracite coal was not calculated because only 6 (out of approximately 2000) utility boilers fire anthracite and account for only 0.4 percent of the coal burned annually. For the purposes of calculating mercury emissions, units burning anthracite were assumed to burn bituminous coal.

A database of trace element concentrations in coal, by state of coal origin, was compiled by the United States Geological Survey (USGS), which analyzed 3,331 core and channel samples of coal. These samples came from 50 coal beds having the highest coal production in the U.S. Industry reviewed these data and under a separate effort screened the data to remove about 600 entries representing coal seams that could not be mined economically (EPRI, 1994). The mercury concentration of the screened data set was virtually the same as the mercury concentration when the full USGS data set was used, so U.S. EPA chose to use the USGS data in its entirety. The mercury concentration of the samples ranged from 0.003 ppmwt to 3.8 ppmwt (Bragg, 1992).

The average mercury content of each of these beds was calculated. The location of each bed was then matched with a state. Using the UDI database and records of actual coal receipts, the state from which each utility purchased the majority of its coal was identified. With three exceptions, the mercury content of the coal fired by each utility was then assigned based on the average concentration of mercury calculated for each coal bed. Exceptions were made for Colorado bituminous, Illinois coal, and Wyoming coal where data were available from as-fired coal samples. These data were used directly to estimate emissions from utility boilers firing these coals. There were two sets of data for coal originating in Arizona and Washington. These two sets were averaged. Since no data were available for coal from Louisiana, data from Texas lignite coal were substituted for Louisiana lignite coal.

Mercury Reductions Due to Coal Cleaning. The USGS database contains concentrations of mercury in as-mined coal but does not include analyses of coal shipments (i.e., "as-fired" coal). The concentration of mercury in as-mined coal may be higher than the concentration in shipped coal because in the process of preparing a coal shipment, some of the mineral matter in coal - and the

associated mercury - may be removed by coal cleaning processes. Since approximately 77 percent of the eastern and midwestern bituminous coal shipments are cleaned in order to meet customer specifications for heating value (Akers et al., 1993), ash and sulfur content, analyses were done to estimate the average amount of mercury reduction that could be attributed to coal cleaning. As a result of these analyses, a 21 percent reduction in mercury concentration was attributed to coal cleaning for those boilers purchasing coal from states where coal washing is common practice. The highlight box below discusses how this mercury reduction value was determined. No coal cleaning reductions were applied to lignite or subbituminous coals, or bituminous coal when the state of coal origin was west of the Mississippi River.

For example, for a unit burning bituminous coal, the amount of mercury entering the boiler was estimated by multiplying the average mercury content of the coal (specific to state of coal origin) by 0.79 to account for a 21 percent reduction due to coal cleaning. This product was multiplied by the unit's annual fuel consumption rate to give the inlet mercury in kg/year.

Calculation of Mercury Emission Estimates. Emissions data were available from 51 emission tests conducted by U.S. EPA, the Electric Power Research Institute (EPRI), the Department of Energy (DOE), and individual utilities. Not all known boiler configurations or control devices could be tested. In order to estimate emissions from all units in the U.S., EMFs were developed for specific boiler configurations and control devices from the test data and applied to similar units.

The EMFs were calculated by dividing the amount of mercury exiting either the boiler or the control device by the amount of mercury entering the boiler. The average EMF for specific boiler configurations and control devices was calculated by taking the geometric mean of the EMFs for that type of configuration or control device. (The geometric mean was chosen rather than the arithmetic mean because the distribution of emission factors followed a lognormal distribution.) The EMFs for various boiler configurations and control devices are shown in Appendix C. To calculate the control efficiency, the EMF is subtracted from 1.

Boiler-specific emission estimates were then calculated by multiplying the calculated inlet mercury concentration by the appropriate EMF for each boiler configuration and control device.² Figures 4-7 and 4-8 illustrate how mercury emission estimates were calculated for coal-fired boilers and for oil- or natural gas-fired boilers. As displayed in Table 4-9, national estimates of mercury emissions from utility boilers are 51.3 tons per year, of which 51 tons are attributed to coal-fired units, 0.25 tons are attributed to oil-fired units, and 0.002 tons are attributed to natural gas-fired units.

4.1.4 Commercial/Industrial Boilers

Commercial/industrial boilers are large boilers found in businesses and industrial plants throughout the United States. These boilers may use coal, oil, or natural gas as fuels. As with utility boilers, mercury vaporizes during combustion and appears as a trace contaminant in the gas exhaust stream.

² Limestone is used in circulating fluidized bed (CFB) boilers to control sulfur dioxide emissions. The EPA recognizes that the limestone may contribute to trace metal emissions, including mercury. For the 19 CFB units in the U.S., the potential contribution of limestone to the unit's mercury emissions was included in the mercury emissions estimate for each boiler.

EFFECT OF COAL CLEANING ON MERCURY CONCENTRATIONS

U.S. EPA requested data on the concentrations of trace elements (including mercury) in coal from the National Coal Association, but limited data were available for two reasons. First, few shipments are analyzed for trace element concentrations, and second, many coal companies consider such information proprietary. EPA did receive data on the concentrations of trace elements in coal shipments from the ARCO Coal Company on 145 samples of Wyoming coal and on 30 samples of bituminous Colorado coal; the Illinois State Geological Survey (ISGS) on 34 samples of Illinois coal; and the Electric Power Research Institute (EPRI) on mercury concentrations in 100 various samples.

Since no other data were available on the concentration of mercury in actual coal shipments, arithmetic averages of the mercury concentrations provided by the ARCO Coal Company and the ISGS were considered as-fired samples. These values were used directly to estimate the amount of mercury in bituminous Colorado coal, subbituminous Wyoming, and bituminous Illinois coal shipments.

The mercury concentrations in the raw coal, the clean coal, and the percent reduction achieved by cleaning are shown in Table 4-8. As shown, some of the mercury reductions are negative. At first, this would seem to suggest that the mercury has been increased or enriched in the clean coal. Negative percentages occur when part of the coal is removed, but the mercury is not contained in the extracted portion. As a result, the same weight of mercury that was contained in the uncleaned coal is contained within a relatively smaller weight of the cleaned coal. Since the weight of the mercury was not changed, negative removal percentages were interpreted to mean that no mercury reduction occurred, or in other words, that the mercury reduction was zero percent.

As shown in Table 4-8, the mercury reductions ranged from -200 percent (effectively zero percent removal) to 64 percent. There is also variation in mercury reduction from cleaned coals originating from the same coal seam. For example, the mercury reduction ranged from -20 percent to 36 percent for Pittsburgh seam coals. The variation may be explained by several factors. The data may represent different cleaning techniques, and the effectiveness of the cleaning processes will depend on how much mercury was contained in the coal. Also, considerable variation may result from the mercury analytical technique.

Because of the variability of the data, typical mercury removal was estimated by taking the arithmetic average of the removal data listed in Table 4-8. Any negative value was taken as a zero, and the zero values were included in the average. The resulting 21 percent average reduction was used to estimate mercury emissions from utility boilers that burn bituminous coal from states east of the Mississippi River. Note that this reduction was assumed for all such boilers, even though data indicate that only 77 percent of the eastern and midwestern bituminous coal shipments are cleaned. As stated above, no coal cleaning reductions were applied to lignite or subbituminous coals, or bituminous coal when the state of coal origin was west of the Mississippi River.

As these data demonstrate, coal cleaning can result in mercury reductions that are higher or lower than the average 21 percent value applied in this analysis. It is expected that significantly higher mercury reductions can be achieved with the application of emerging coal preparation processes, such as selective agglomeration and advanced column floatation.

Figure 4-7
Mercury Emissions from Oil- and Natural-Gas Fired Boilers

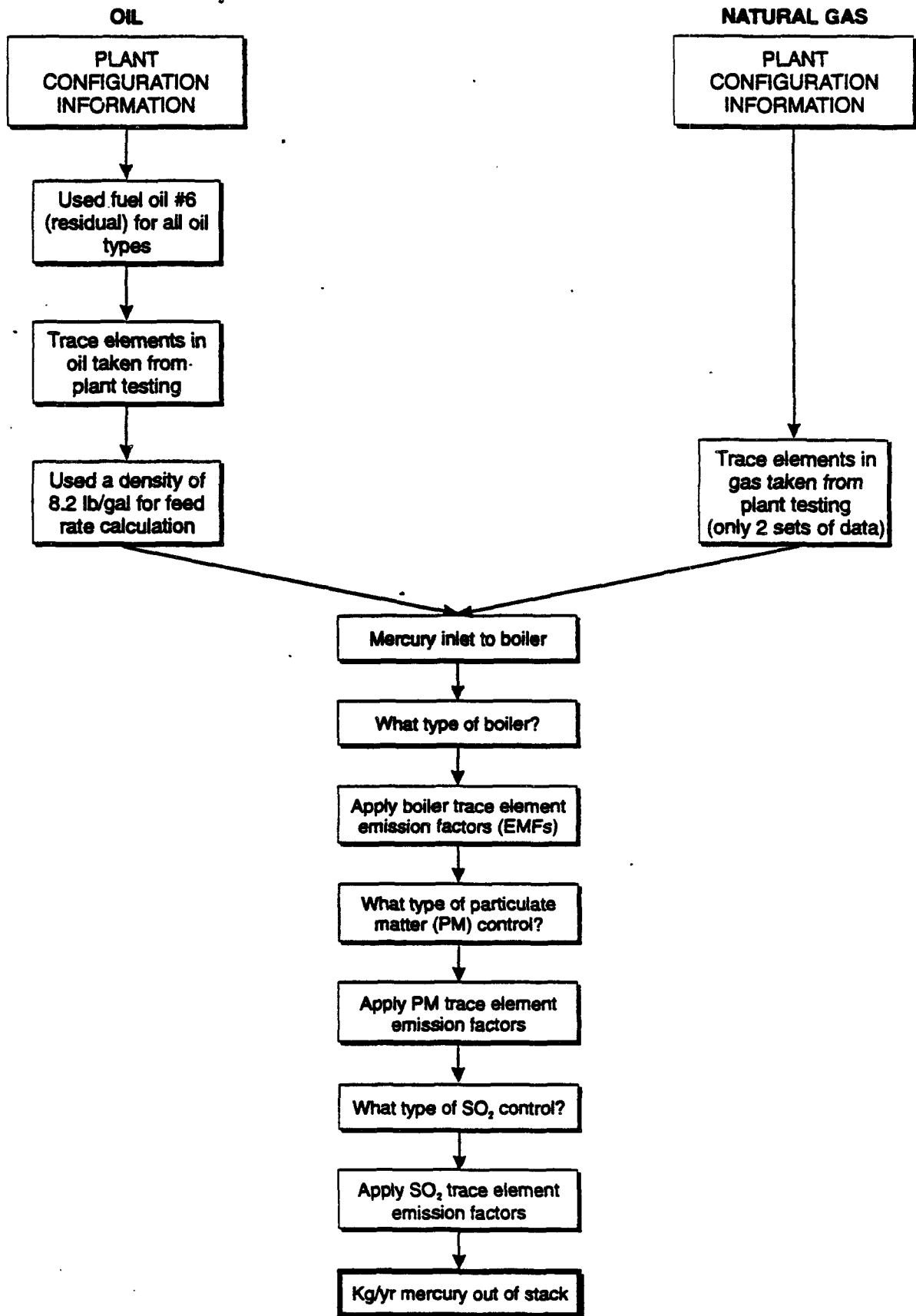


Figure 4-8
Mercury Emissions from Coal-Fired Boilers

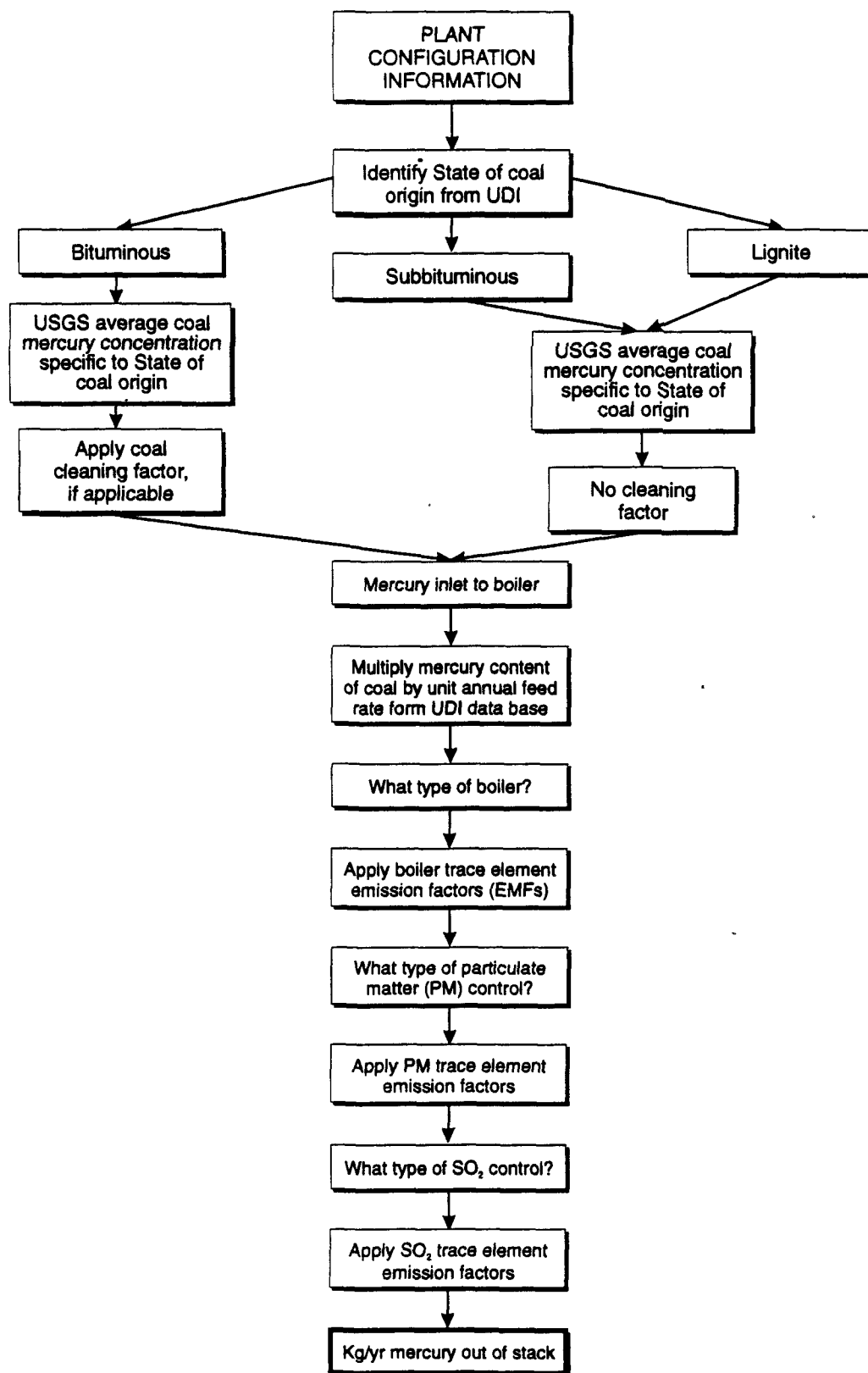


Table 4-8
Comparison of Mercury Concentrations in Raw and Cleaned Coal

Seam	State	Raw Coal Mercury (ppm)	Cleaned Coal Mercury (ppm)	Percent Removal
Central Appalachian Coal Sample A		0.09	0.1	-11.11
Central Appalachian Coal Sample B		0.12	0.11	8.33
Il #6	IL	0.14	0.08	42.86
Pittsburgh A	PA	0.15	0.11	26.67
Pittsburgh B	PA	0.14	0.09	35.71
Pittsburgh C	PA	0.14	0.13	7.14
Pittsburgh D	PA	0.1	0.12	-20.00
Pittsburgh E	PA	0.1	0.08	20.00
Pittsburgh	PA	0.1	0.08	20.00
Upper Freeport	PA	0.03	0.09	-200.00
Lower Kittanning	PA	0.44	0.34	22.73
Sewickley	PA	0.18	0.18	0.00
Pittsburgh	PA	0.13	0.11	15.38
Pittsburgh	PA	0.13	0.12	7.69
Il #6	IL	0.12	0.13	-8.33
KY #9 and 14	KY	0.16	0.14	12.50
Pratt/Utley	AL	0.28	0.22	21.43
Pratt	AL	0.29	0.28	3.45
Utley	AL	0.34	0.27	20.59
Pratt	AL	0.34	0.24	29.41
Upper Freeport	PA	0.7	0.25	64.29
Upper Freeport	PA	0.7	0.28	60.00
Il 2,3,5	IL	0.24	0.2	16.67
Il 2,3,5	IL	0.24	0.14	41.67
Ky #11	KY	0.15	0.12	20.00
ISGS	IL	0.2	0.09	55
Minimum				-200.00
Maximum				64.29
Average				21.21

Reference: Akers et al., 1993 for every seam but ISGS; Demir et al., 1993 for ISGS.

Table 4-9
Best Point Estimate of Mercury Emissions from Utility Boilers: 1990

Fuel Type	Emission Rate		Comments
	Mg/Yr	Tons/Yr	
Coal	46.3	51	The industry (Electric Power Research Institute) estimate for coal-fired units is 44 tons/year.
Oil	0.23	0.25	
Natural Gas	0.002	0.002	
Total	46.5	51.3	

Mercury emissions from commercial/industrial boilers, estimated at 26.3 Mg/yr (29 tons/yr), are directly related to the amount of fuel used in the combustion process (U.S. EPA, 1993a). Mercury emissions from natural gas combustion could not be estimated because a reliable emission factor does not exist (U.S. EPA, 1993a). Commercial/industrial boilers consume energy at an annual rate of 25×10^{12} MJ/yr (23×10^{15} Btu). About 12 percent of this energy consumption results from coal combustion, 39 percent from oil and petroleum fuel combustion, and 48 percent from natural gas combustion (U.S. Department of Energy, 1992). Estimates of coal and oil consumption from these boilers on a per-State basis are presented in Table A-2, Appendix A.

Because there is no evidence to show that mercury emissions are affected by boiler type, this section presents only a brief discussion of commercial/industrial boiler types and combustion techniques. More information on boiler types may be found in the Air Pollution Engineering Manual AP-42 and the L&E document (Buonicore and Davis, 1992; U.S. EPA, 1988a; U.S. EPA, 1993a).

As with utility boilers, the configuration of commercial/industrial boilers can vary, but the overall system is straightforward. Coal or oil is received and transferred to storage where it is held until it is transferred to the boiler. Because this source category encompasses a wide range of boiler sizes, the types of boilers used are more varied than those used in the utility sector. Larger coal-fired industrial boilers are suspension-fired systems like those used in the utility sector, while moderate and smaller units are grate-fired systems that include spreader stokers, overfeed traveling and vibrating grate stokers and underfeed stokers. Oil-fired furnaces, which may use either distillate or residual fuel oil, typically comprise a burner, a combustion air supply system, and a combustion chamber. All coal-fired facilities, and some oil-fired facilities, also have ash-handling systems.

Mercury emission factors for coal combustion in commercial/industrial boilers were developed using mass-balance calculations with the assumption that all mercury fired with the coal is emitted in the stack gas as a function of coal type (U.S. EPA, 1993a). The emission factors do not account for coal washing because the U.S. EPA believes that buyers for commercial/industrial boilers do not purchase washed coal; their source of coal is primarily the spot market. An estimated emission factor of $7.0 \text{ kg}/10^{15} \text{ J}$ ($16 \text{ lb}/10^{12} \text{ Btu}$) was used for bituminous coal combustion, and $7.6 \text{ kg}/10^{15} \text{ J}$ ($18 \text{ lb}/10^{12} \text{ Btu}$) was used for anthracite coal combustion. Estimates of mercury emissions on a per-

state basis from coal-fired commercial/industrial boilers are provided in Table A-3, Appendix A. These values were determined by using the referenced emission factors and the coal consumption estimates for the states presented in Table A-2, Appendix A. In estimating emissions, it was assumed that mercury emissions from commercial/industrial boilers were not controlled. The total estimated annual emissions for coal-fired boilers are 20.7 Mg/yr (22.8 tons/yr). Because mercury reductions from coal washing and any other reductions that may occur across existing control devices are not accounted for, the emissions may be overestimated.

Mercury emissions for oil combustion in commercial/industrial boilers were estimated on a per-state basis using an emission factor of 2.9 kg/10¹⁵ J (6.8 lb/10¹² Btu) for residual oil and 3.0 kg/10¹⁵ J (7.2 lb/10¹² Btu) for distillate oil and the oil consumption estimates for States given in Table A-2, Appendix A. These calculated emission values are presented in Table A-4, Appendix A. The total estimated annual emissions for oil-fired commercial/industrial boilers are 5 Mg/yr (6 tons/yr).

4.1.5 Residential Boilers

Residential boilers are relatively small boilers used in homes and apartments. These boilers may use coal, oil, or natural gas as fuels; however, mercury emissions from natural gas combustion are negligible. As with the other types of boilers, mercury vaporizes during combustion in the coal- and oil-fired residential boilers and the emissions appear as a trace contaminant in the exhaust gas.

The estimated annual mercury emissions from residential boilers, 3.2 Mg/yr (3.5 tons/yr), are related to the amount of fuel used in the combustion process. Estimates of coal and oil consumption from these boilers on a per-state basis are presented in Table A-4, Appendix A. Residential boilers consume energy at an annual rate of 6.2×10^{12} MJ/yr (5.8×10^{15} Btu/yr). About 1 percent of this energy consumption results from coal combustion, 22 percent from oil and petroleum fuel combustion and 77 percent from natural gas combustion (U.S. Department of Energy, 1992).

Because there is no evidence to link mercury emissions to boiler type, this section does not describe residential boiler types. Information on boiler types may be found in the Air Pollution Engineering Manual, AP-42 and the L&E document (Buonicore and Davis, 1992; U.S. EPA, 1988).

Estimated mercury emission factors for coal combustion in residential boilers are the same as those used for other coal combustion processes. These calculations include the assumption that all mercury fired with the coal is emitted as stack gas. An estimated emission factor of 7.0 kg/10¹⁵ J (16 lb/10¹² Btu) was used for bituminous coal combustion, and 7.6 kg/10¹⁵ J (18 lb/10¹² Btu) was used for anthracite coal combustion. Estimates of mercury emissions on a per-state basis from coal-fired residential boilers were determined by using these emission factors and the coal consumption estimates for the states as presented in Table A-5, Appendix A. These calculated emission values are presented in Table A-6, Appendix A. In estimating emissions, it was assumed that mercury emissions from residential boilers were not controlled. The total annual estimated emissions for coal-fired residential boilers is 0.5 Mg/yr (0.5 tons/yr).

The estimated mercury emissions for oil combustion were estimated by using an emission factor of 2.9 kg/10¹⁵ J (6.8 lb/10¹² Btu) for residual oil and 3.0 kg/10¹⁵ J (7.2 lb/10¹² Btu) for distillate oil and the oil consumption estimates for the states given in Table A-5, Appendix A. These estimated emissions values are presented in Table A-7, Appendix A. The total annual estimated emissions for oil-fired residential boilers is 2.7 Mg/yr (3.0 tons/yr).

4.1.6 Sewage Sludge Incinerators

Sewage sludge incinerators (SSIs) are operated primarily by U.S. cities and towns as a final stage of the municipal sewage treatment process. The mercury in sewage comes from households, commercial and industrial sources and industries discharging industrial wastewater into the sewer systems and flows to sewage treatment plants. After treatment at the sewage treatment plant, the sludge is usually landfilled or incinerated. Only a small percentage of U.S. cities use sewage sludge incinerators. The estimated annual mercury emissions from SSIs account for 1.65 Mg/yr (1.82 tons/yr). Mercury emissions occur when mercury in the sewage is combusted at high temperatures, vaporizes and exits through the gas exhaust stack. Landfilled sludge or sludge applied to farmland are also potential sources of mercury emissions. These sources are not addressed in this inventory.

A total of 206 SSIs currently operate in the United States. An estimated 1.5×10^6 Mg (1.65×10^6 tons) of sewage sludge on a dry basis are incinerated annually (U.S. EPA, 1993b). Table A-13, Appendix A shows the geographic distribution of sewage sludge incinerators throughout the country. Most facilities are located in the Eastern United States, but a substantial number also are located on the West Coast. New York has the largest number of SSI facilities with 32, followed by Pennsylvania and Michigan with 21 and 19, respectively.

Within the SSI category, three combustion techniques are used: multiple-hearth, fluidized-bed and electric infrared. Multiple-hearth units predominate; over 80 percent of the identified SSIs are multiple hearth. About 15 percent of the SSIs in operation are fluidized bed units, about 3 percent are electric infrared and the remainder co-fire sewage sludge with municipal waste (U.S. EPA, 1993b).

The sewage sludge incinerator process involves two primary steps: dewatering the sludge and incineration. The primary source of mercury emissions from SSIs is the combustion stack. Most SSIs are equipped with some type of wet scrubbing system for PM control. Because wet systems provide gas cooling, as well as PM removal, these systems can potentially provide some mercury control.

The U.S. EPA's Compilation of Air Pollutant Emission Factors (U.S. EPA, 1988a) (otherwise known as the AP-42) for SSIs lists five mercury emission factors for various types of SSIs and controls: 0.005 g/Mg (1.0×10^{-5} lb/ton) for multiple hearth combustors controlled with a combination of venturi and impingement scrubbers, 0.03 g/Mg (6.0×10^{-5} lb/ton) for fluidized bed combustors controlled with a combination of venturi and impingement scrubbers, 2.3 g/Mg (4.6×10^{-3} lb/ton) for multiple hearth combustors controlled with a cyclone scrubber, 1.6 g/Mg (3.2×10^{-3} lb/ton) for multiple hearth combustors controlled with a combination of cyclone and venturi scrubbers, and 0.97 g/Mg (1.94×10^{-3} lb/ton) for multiple hearth combustors controlled with an impingement scrubber (U.S. EPA, 1993b). Given that combustor and control types are not known for all SSIs currently operating in the United States, average emission factors were calculated: 0.0175 g/Mg (3.5×10^{-5} lb/ton) for SSIs controlled with a combination of venturi and impingement scrubbers and 1.623 g/Mg (3.25×10^{-3} lb/ton) for SSIs controlled by any other type or combination of types of scrubbers. Of the SSIs where data are available, 32.6 percent of SSIs are controlled by a combination of venturi and impingement scrubbers, and 67.4 percent are controlled by some other means. These percentages were assumed to apply to the total population of SSIs. Multiplying the total amount of sewage sludge incinerated annually, 1.5×10^6 Mg (1.65×10^6 tons), by the appropriate percentage and emission factor gives a mercury emission estimate of 0.009 Mg/yr (0.01 tons/yr) for SSIs controlled with a combination of venturi and impingement scrubbers and an estimate of 1.64 Mg/yr (1.81 tons/yr) for SSIs controlled by some other means. The overall mercury emissions estimate from SSIs is, thus, 1.65 Mg/yr (1.82 tons/yr).

4.1.7 Crematories

Volatilization of mercury from the mercury alloys contained in amalgam tooth fillings during cremation of human bodies is a potential source of mercury air emissions. In 1991, there were 400,465 cremations in the slightly more than 1,000 crematories located throughout the United States (Cremation Association of North America, 1992). Table A-8 in Appendix A lists the estimated number of crematories located in each State and the estimated number of cremations performed. Information was not available on the location of individual crematories (Vander Most and Veldt, 1992).

No data are available for the average quantity of mercury emitted for a cremation in the United States. Three estimated levels have been cited for European countries (Switzerland, Germany and the United Kingdom) with an estimated emission rate of 1 gram of mercury per cremation recommended as a typical value (Springer, 1993). This emission factor may not be applicable to cremations in the United States because dental care programs in the United States differ markedly from those in Europe. Consequently, the average number of mercury amalgam fillings per person may differ considerably, with Europeans believed to have more fillings per person than Americans. Because the average number of fillings per person and the average mercury content per filling have a direct impact on the estimated mercury emissions, this European emission factor may not provide an accurate estimate of mercury emissions from cremations in the United States. Multiplying the European emission factor of 1 gram of mercury per cremation by the total number of U.S. cremations in 1991 (400,465) gives a mercury emission estimate of 0.4 Mg/yr (0.4 tons/yr).

4.1.8 Wood Combustion

Wood and wood wastes are used as fuel in both the industrial and residential sectors. In the industrial sector, wood waste is fired in industrial boilers to provide process heat, while wood is used in fireplaces and wood stoves in the residential sectors. Studies have shown that wood and wood wastes may contain mercury. Insufficient data are available, however, to estimate the typical mercury content of wood and wood wastes.

Wood waste combustion in boilers is mostly confined to industries in which wood waste is available as a byproduct. These boilers, which are typically of spreader stoker or suspension-fired design, are used to generate energy and alleviate possible solid waste disposal problems. In boilers, wood waste is normally burned in the form of hogged wood, sawdust, shavings, chips, sanderdust, or wood trim. Heating values for this waste range from about 9,300 to 12,000 kJ/kg (4,000 Btu/lb to 5,000 Btu/lb) of fuel on a wet, as-fired basis. The moisture content is typically near 50 weight percent but may vary from 5 to 75 weight percent, depending on the waste type and storage operations. As of 1980, about 1,600 wood-fired boilers were operating in the United States, with a total capacity of approximately 30.5 gigawatts (GW) (1.04×10^{11} Btu/hr) (U.S. EPA, 1982). No specific data on the distribution of these boilers were identified but most are likely to be located where pulp and paper plants or logging operations are located (i.e., in the Southeast, the Pacific Northwest States, Wisconsin, Michigan, and Maine) (U.S. EPA, 1993a).

Wood-fired boilers use PM control equipment, which may provide some reduction in mercury emissions. The most common control devices used to reduce PM emissions from wood-fired boilers are mechanical collectors, wet scrubbers, ESPs, and fabric filters. Only the last three have the potential for mercury reduction. The most widely used wet scrubbers for wood-fired boilers are venturi scrubbers, although no data have been located on the performance of these systems relative to mercury emissions. No data are available on mercury emission reduction for fabric filters for wood

combustors, but results for other combustion sources suggest that efficiencies will be low, probably 50 percent or less (U.S. EPA, 1993a).

The data on mercury emissions from wood-fired boilers are limited. A recent AP-42 study provided a range and average typical emission factor for wood waste combustion in boilers based on the results of seven tests. The average emission factor of 0.34×10^{-5} kg/Mg (0.67×10^{-5} lb/ton) of wood burned is recommended as the best typical emission factor for wood waste combustion in boilers (U.S. EPA, 1992c). Dividing the total capacity of wood-fired boilers, 30.5 GW (1.04×10^{11} Btu/hr), by the average heating value of wood, 10,600 kJ/kg (4,560 Btu/lb), gives the total hourly rate: 10,367 Mg/hr (11,404 tons/hr) (U.S. EPA, 1982). Assuming that wood-fired boilers operate at capacity at 8,760 hr/yr and multiplying by the above emission factor gives a mercury emission estimate for wood-fired boilers of 0.3 Mg/yr (0.33 tons/yr). This estimate has a high degree of uncertainty given the limited data available.

Wood stoves, which are commonly used as residential space heaters, are of three different types: (1) the conventional wood stove, (2) the noncatalytic wood stove and (3) the catalytic wood stove. Fireplaces are used primarily for aesthetic effects and secondarily as a supplemental heating source in homes and other dwellings. Wood is most commonly used as fuel, but coal and densified wood "logs" also may be burned.

All of the systems described above operate at temperatures that are above the boiling point of mercury. Although some wood stoves use emission control measures to reduce volatile organic compound (VOC) and carbon monoxide (CO) emissions, these techniques are not expected to affect mercury emissions. Consequently, any mercury contained in the wood will be emitted with the combustion gases via the exhaust stack.

For residential wood combustion, only one emission factor, 1.3×10^{-2} kg/Mg (2.6×10^{-2} lb/ton) is available, which is based on a single test burning a single type of wood (pine) at a single location (DeAngelis et al., 1980). In 1987, the Department of Energy estimated that 22.5 million households burned approximately 42.6 million cords of wood (Phillips, 1993). Given that the densities of wood vary greatly depending on wood type and the moisture content of the wood, and because the above emission factor is from a single test, nationwide emissions of mercury for residential wood combustion were not estimated.

4.1.9 Hazardous Waste Combustors

For the purpose of this emissions inventory, hazardous waste combustors include hazardous waste incinerators and lightweight aggregate kilns. Although hazardous waste burning cement kilns are also typically classified as hazardous waste combustors, mercury emissions from cement manufacturing are considered separately (see Section 4.2.4).

Based on the U.S. EPA's 1995 emission estimates (U.S. EPA, 1995b), hazardous waste incinerators and lightweight aggregate kilns currently combine to emit a total of 4.5 Mg/year (5.0 tons/year) of mercury. Of this amount, hazardous waste incinerators are estimated to emit 4.3 Mg/year (4.7 tons/year), or approximately 95 percent of the total, while lightweight aggregate kilns are estimated to emit 0.25 Mg/year (0.27 tons/year), or about 5 percent of the total.

4.1.9.1 Hazardous Waste Incinerators

A hazardous waste incinerator is an enclosed, controlled flame combustion device that is used to treat primarily organic and/or aqueous waste, although some incinerators burn spent or unusable ammunition and/or chemical agents. These devices may be fixed (in situ) or mobile (such as those used for site remediation). Major incinerator designs include rotary kilns, liquid injection incinerators, fluidized bed incinerators and fixed hearth incinerators.

Currently, 162 permitted or interim status incinerator facilities, having 190 units, are in operation in the U.S. According to the U.S. EPA's *List of Hazardous Waste Incinerators* (November 1994), another 26 facilities are proposed (i.e., new facilities under construction or in the process of being permitted). Of the 162 facilities, 21 are commercial sites that burn about 700,000 tons of hazardous waste annually. The remaining 141 are onsite or captive facilities that burn about 800,000 tons of waste annually.

Hazardous waste incinerators are equipped with a wide variety of air pollution control devices. Typical devices include packed towers, spray dryers, or dry scrubbers for acid gas (e.g., HCl, Cl₂) control, as well as venturi scrubbers, wet or dry ESPs or fabric filters for particulate control. Most incinerators use wet systems to scrub acid emissions (three facilities use dry scrubbers). Activated carbon injection for controlling dioxin and mercury is being used at only one incinerator. New control technologies, such as catalytic oxidizers and dioxin/furan inhibitors, have recently emerged but have not been used on any full-scale incinerators in the U.S.

Major Designs for Hazardous Waste Incinerators

Rotary Kilns. Rotary kiln systems typically contain two incineration chambers: the rotary kiln and an afterburner. The shell of the kiln is supported by steel trundles that ride on rollers, allowing the kiln to rotate around its horizontal axis at a rate of one to two revolutions per minute. Wastes are fed directly at one end of the kiln and heated by primary fuels. Waste continues to heat and burn as it travels down the inclined kiln, which typically operates at 50-200 percent excess air and at temperatures of 1600-1800°F. Flue gas from the kiln is routed to an afterburner, operating at 100-200 percent excess air and 2000-2500°F, where unburnt components of the kiln flue gas are more completely combusted. Some rotary kiln incinerators, known as slagging kilns, operate at high enough temperatures that residual materials leave the kiln in molten slag form. The molten residue is then water-quenched. Ashing kilns operate at a lower temperature, with the ash leaving as a dry material.

Liquid Injection Incinerators. A liquid injection incineration system consists of an incineration chamber, waste burner and auxiliary fuel system. Liquid wastes are atomized as they are fed into the combustion chamber through waste burner nozzles.

Fluidized Bed Incinerators. A fluidized bed system is essentially a vertical cylinder containing a bed of granular material at the bottom. Combustion air is introduced at the bottom of the cylinder and flows up through the bed material, suspending the granular particles. Waste and auxiliary fuels are injected into the bed, where they mix with combustion air and burn at temperatures from 840-1500°F. Further reaction occurs in the volume above the bed at temperatures up to 1800°F.

Fixed Hearth Incinerators. These systems typically contain a primary and a secondary furnace chamber. The primary chamber operates in "starved air" mode and the temperatures are around 1000°F. The unburnt hydrocarbons reach the secondary chamber where 140-200 percent excess air is supplied and temperatures of 1400-2000°F are achieved for more complete combustion.

4.1.9.2 Lightweight Aggregate Kilns

The term lightweight aggregate refers to a wide variety of raw materials (such as clay, shale or slate) that after thermal processing can be combined with cement to form concrete products. Lightweight aggregate concrete is produced either for structural purposes or for thermal insulation purposes. A lightweight aggregate plant is typically composed of a quarry, a raw material preparation area, a kiln, a cooler and a product storage area. The material is taken from the quarry to the raw material preparation area and from there is fed into the rotary kiln.

Major Design and Operating Features of Lightweight Aggregate Kilns

Rotary kilns at lightweight aggregate plants typically consist of a long (30 to 60-meter) steel cylinder lined with refractory bricks. The cylinder is capable of rotating about its axis and is inclined at an angle of about 5 degrees.

Prepared raw material is fed into the kiln at the higher end, while firing takes place at the lower end. The dry raw material fed into the kiln is initially preheated by hot combustion gases. Once the material is preheated, it passes into a second furnace zone where it melts to a semiplastic state and begins to generate gases that serve as a bloating or expanding agent. In this zone, specific compounds begin to decompose and form gases (such as SO₂, CO₂, SO₃, and O₂) that eventually trigger the desired bloating action within the material. As temperatures reach their maximum (approximately 2100°F), the semiplastic raw material becomes viscous and entraps the expanding gases. This bloating action produces small, unconnected gas cells, which remain in the material after it cools and solidifies. The product exits the kiln and enters a section of the process where it is cooled with cold air and then conveyed to the discharge.

There are approximately 36 lightweight aggregate kiln locations in the U.S. Of these sites, there are currently seven facilities that burn hazardous waste in a total of 15 kilns.

Lightweight aggregate kilns use one or a combination of air pollution control devices, including fabric filters, venturi scrubbers, spray dryers, cyclones and wet scrubbers. All of the facilities utilize fabric filters as the main type of emissions control, although one facility uses a spray dryer, venturi scrubber and wet scrubber in addition to a fabric filter.

4.2 Manufacturing Sources

Manufacturing sources, including processes that use mercury directly and those that produce mercury as a byproduct, account for an estimated 29.1 Mg/yr (32 tons/yr) of mercury emissions generated in the United States. Emissions from these sources are presented in Table 4-10 and are discussed below.

4.2.1 Primary Lead Smelting

Primary lead smelters recover lead from a sulfide ore, which may contain mercury. The smelters emitted an estimated 8.2 Mg (9 tons) of mercury into the atmosphere in 1990. Table 4-11 lists the locations and 1990 production rates of the three primary lead smelters that are currently operating in the United States; the locations of these smelters are displayed in Figure 4-9.

Table 4-10
Best Point Estimate of Mercury Emissions from Anthropogenic Manufacturing Sources: 1990-1993

Source	Emissions			Date of Data ^a	Degree of Uncertainty ^b	Basis for Emission Estimate
	Mg/yr	Tons/yr	% of total			
Primary lead smelting	8.2	9	3.7	1985/1990	High	Test data; 1995 emissions estimate is 1.5 tons/yr
Secondary mercury production	6.7	7.4	3.1	1973/1991	High	Test data
Chlor-alkali production	5.9	6.5	2.7	1991/1991	Medium	Section 114 industry survey responses
Cement manufacturing	5.9	6.5	2.7	1992/1990	Medium	Test reports; industry estimates for this source category are 3.3 tons/yr; see Section 4.2.4
Primary copper smelting ^c	0.6	0.7	0.3	1992/1992	High	Test report; 1995 national emissions for currently operating smelters are 0.08 tons/yr
Lime manufacturing	0.6	0.7	0.3	1983/1992	High	Test data and mass balances
Electrical apparatus manufacturing	0.42	0.46	0.2	1973/1992	High	Engineering judgment
Instrument manufacturing	0.5	0.5	0.2	1973/1992	High	Survey questionnaire responses
Carbon black production	0.23	0.25	0.1	1980/1991	High	
Fluorescent lamp recycling	0.005	0.006	0.002	1993/1993	High	Test data and mass balances
Battery production	0.02	0.02	0.008	1986/1992	High	Engineering judgment
Primary mercury production	-	-	-	-	-	Insufficient data to estimate emissions
Mercury compounds production	-	-	-	-	-	Insufficient data to estimate emissions
Byproduct coke production	-	-	-	-	-	Insufficient data to estimate emissions
Petroleum refining	-	-	-	-	-	Insufficient data to estimate emissions
Total	29.1	32	13.2			

^a Date that data emission factor is based on/Date of activity factor used to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

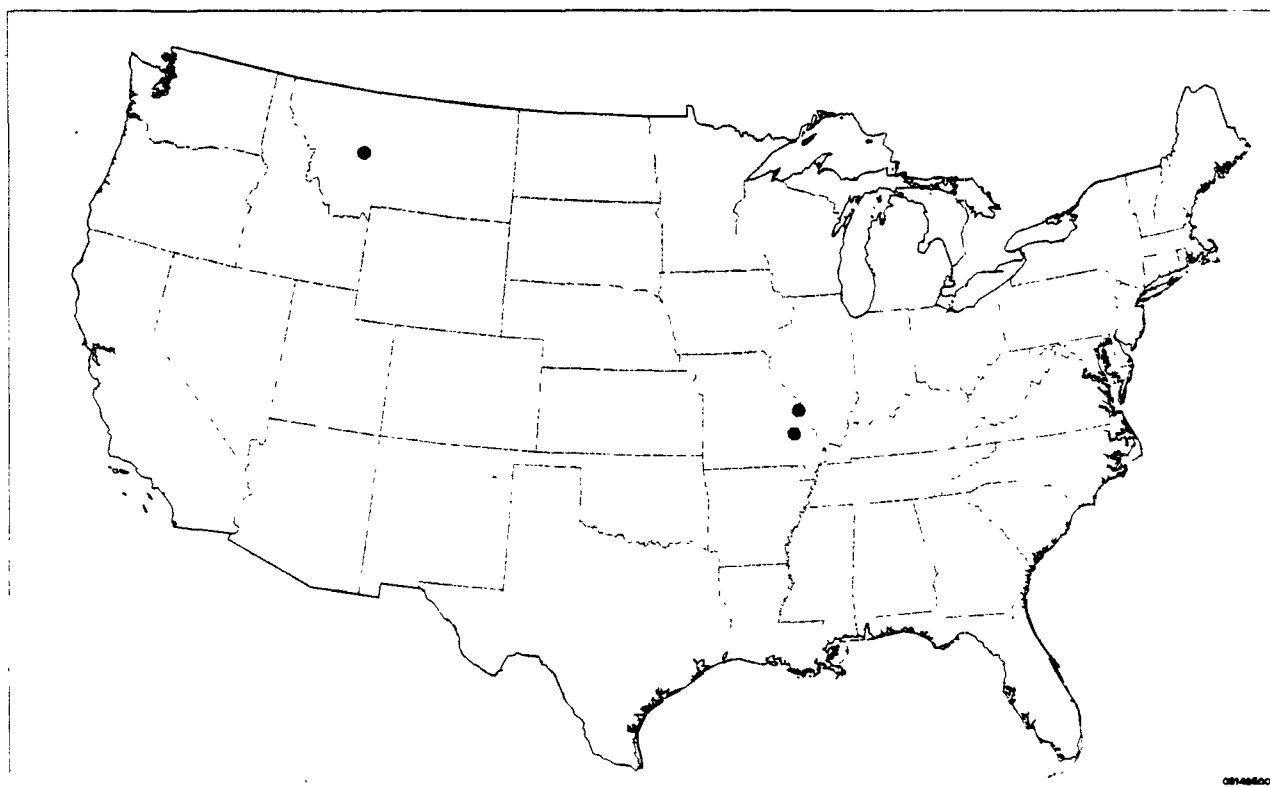
^c Emissions are estimated for only one source, which ceased operation in February 1995; 1995 emissions reflect currently operating copper smelters.

Table 4-11
1990 U.S. Primary Lead Smelters and Refineries^a

Smelter	Refinery	1990 Lead Production Tons (Megagrams)
ASARCO, East Helena, MT	ASARCO, Omaha, NE	65,800 (72,500)
ASARCO, Glover, MO	ASARCO, Glover	112,000 (123,200)
Doe Run (formerly St. Joe)	Doe Run, Herculaneum, MO	231,000 (254,100)

^a Woodbury, 1992.

Figure 4-9
Primary Lead Smelters



The Primary Lead Smelting Process

Recovery of lead from the lead ore in primary lead smelters consists of three main steps: sintering, reduction and refining. The sintering machine, which converts lead sulfide in the ore to lead and lead oxide, is a continuous steel pallet conveyor belt. Each pallet consists of perforated grates, beneath which are wind boxes connected to fans to provide a draft through the moving sinter charge. The sintering reactions on the grate take place at about 1000°C (1832°F). Because mercury and its compounds volatilize below this temperature, most of the mercury present in the ore is emitted as a vapor in the sintering machine exhaust gas as elemental mercury or as mercuric oxide.

Reduction of the sintered lead is carried out in a blast furnace at a temperature of 1600°C (2920°F). The furnace is charged with a mixture of sinter (80 to 90 percent of charge), metallurgical coke (8 to 14 percent of charge) and other materials, such as limestone, silica, litharge, and other slag-forming constituents. In the blast furnace, the lead sulfate and lead oxide in sinter is reduced to lead. The heat for the reaction is supplied by the combustion of coke. Impurities are removed from the furnace as slag, which is either processed at the smelter for its metal content, shipped to treatment facilities, or landfilled. The impurities include arsenic, antimony, copper, and metal sulfides and silicates. Lead bullion, which is the primary product, undergoes a preliminary treatment to remove impurities, such as copper, sulfur, arsenic, antimony, and nickel, before carrying out further refining. Any residual mercury left in the ore after sintering will be emitted during the reduction step (U.S. EPA, 1988).

The lead bullion is refined in cast iron kettles. Refined lead, which is 99.99 to 99.999 percent pure is cast into pigs for shipment (U.S. EPA, 1988). Mercury emissions from refining operations are expected to be negligible.

Primary lead smelters use high-efficiency emission control systems to reduce the levels of PM and SO₂ from the blast furnace and sintering machines. Centrifugal collectors (cyclones) are used in conjunction with baghouses or ESPs for PM control. Control of SO₂ emissions from sintering is achieved by absorption to form sulfuric acid in the sulfuric acid plants, which are commonly part of lead smelting plants. Because mercury is emitted from these as a vapor and these PM control systems often operate at temperatures at which mercury has a significant vapor pressure, these PM control devices are expected to have little effect on mercury emissions from the sintering machine and blast furnace. In contrast, sulfuric acid plants are expected to be relatively well controlled for mercury because of the low temperatures and high particulate removal efficiency of the APC device. No data are available, however, on performance of these systems with respect to mercury emissions (U.S. EPA, 1988).

Mercury, which may be present in the ore, may be emitted during the sintering and blast furnace steps and in the dressing area because these processes take place at high temperatures.

No recent mercury emission factors are available for the three currently operating primary lead smelters. Mercury emission factors were provided by industry for a custom smelter operated by ASARCO in El Paso, Texas which ceased operating in 1985 (Richardson, 1993). These emission factors were used in the preliminary analysis to evaluate the potential mercury emissions from primary lead smelting. Based on these emission factors and primary lead production estimates, EPA estimated total mercury emissions at 8.2 Mg (9 tons) for primary lead smelters. This estimate is reflected in this document as the 1990 estimate and was used as the basis for subsequent analyses in this Report (e.g., the long-range transport analysis). Because numerous assumptions were used to convert the emission factors to a lead production basis, this emission estimate was considered to have a high degree of uncertainty.

Due to the relatively high emission potential identified for primary lead smelters, EPA has recently reviewed the emission estimates. The review determined that the 1990 estimate overstated

potential emissions. As stated above, the 1990 estimate was based on data provided for the El Paso smelter. This smelter obtained lead ore from several sources from both within and outside of the United States. These ores had a variable mercury content depending upon the source of the ore. Two of the three current smelters are not custom smelters; they typically process concentrates from southeast Missouri. These concentrates have a very low mercury content. The ASARCO-East Helena plant, although a custom smelter, also processes low mercury concentrates. In addition, the earlier El Paso emissions data included emissions from processes that do not take place at the three operating primary lead smelters. Based on the findings of this review, the U.S. EPA's current estimate of national mercury emissions from primary lead smelters is about 1.3 Mg/yr (1.5 tons/yr).

4.2.2 Secondary Mercury Production

Secondary mercury production (mercury recycling) involves processing scrapped mercury-containing products, industrial waste and scrap, and scrap mercury from government stocks. Secondary mercury production is estimated to have accounted for approximately 6.7 Mg (7.4 tons) of mercury emissions in 1991. Major sources of recycled mercury include dental amalgams, scrap mercury from instrument and electrical manufacturers (lamps and switches), phosphor from discarded fluorescent lamps, wastes and sludges from research laboratories and electrolytic refining plants, and mercury batteries (U.S. EPA, 1993a). Table 4-12 lists the five major companies that were involved in secondary mercury production in 1989.

Table 4-12
1989 U.S. Mercury Recyclers^a

Adrow Chemical Company	Wanaque, NJ
Bethlehem Apparatus Company, Inc.	Hellertown, PA
D. F. Goldsmith Chemical and Metals Corp.	Evanston, IL
Mercury Refining Company, Inc.	Latham, NY
Wood Ridge Chemical Company	Newark, NJ

^a Bureau of Mines, 1991.

During secondary mercury production, emissions may potentially occur from the retort or furnace operations, the distillation process and the discharge to the atmosphere process (Reisdorf and D'Orlando, 1984; U.S. EPA, 1984). The major mercury emission sources are the condenser exhaust and fugitive vapor emissions that occur during unloading of the retort chamber. Mercury emissions can also occur in the filling area if the flask overflows and during the bottling process.

Information on specific emission control measures is limited and site-specific. If a scrubber is used, mercury vapor or droplets in the exhaust gas may be removed by condensation in the spray. No information was found for other control measures that are being used in secondary mercury production processes. Concentrations in the workroom air due to mercury vapor emissions from the hot retort may be reduced by the following methods: containment, local exhaust ventilation, dilution ventilation, isolation, and/or personal protective equipment. No information was provided to indicate that these methods are used in conjunction with any type of emission control device (Reisdorf and D'Orlando, 1984). Vapor emissions due to mercury transfer during the distillation or filling stages may be reduced by containment, ventilation (local exhaust or ventilation), or temperature control.

Secondary Mercury Production Processes

Secondary mercury production (recycling) can be accomplished by one of two general methods: chemical treatment or thermal treatment (U.S. EPA, 1993a). The most common method of recycling metallic mercury is through thermal treatment. Generally, the mercury-containing scrap is reduced in size and is heated in retorts or furnaces at about 538°C (1000°F) to vaporize the mercury. The mercury vapors are condensed by water-cooled condensers and collected under water (Reisdorf and D'Orlando, 1984; U.S. EPA, 1984).

Vapors from the condenser, which may contain PM, organic compounds and possibly other volatile materials from the scrap, are combined with vapors from the mercury collector line. This combined vapor stream is passed through an aqueous scrubber to remove PM and acid gases (e.g., hydrogen chloride [HCl], SO₂). From the aqueous scrubber, the vapor stream passes through a charcoal filter to remove organic components prior to discharging into the atmosphere (U.S. EPA, 1984).

The collected mercury is further purified by distillation and then transferred to the filling area. In the filling area, special filling devices are used to bottle small quantities, usually 0.464 kg (1 lb) or 2.3 kg (5 lb) of distilled mercury. With these filling devices, the mercury flows by gravity through tubing from a holding tank into the flask until the flask overflows into an overflow bottle. The desired amount of mercury is dispensed into the shipping bottle by opening a valve at the bottom of the flask. The shipping bottle is then immediately capped after the filling and sent to the storage area (Reisdorf and D'Orlando, 1984).

Chemical treatment can encompass several methods for aqueous mercury-containing waste streams. To precipitate metallic mercury, the waste stream can be treated with sodium borohydride or the stream can be passed through a zinc-dust bed. Mercuric sulfide can be precipitated from the waste streams by treatment with a water-soluble sulfide, such as sodium sulfide. Ion-exchange systems can be used to recover ionic mercury for reuse, while mercuric ions can be trapped by treatment with chemically modified cellulose (Cammarota, 1975).

Because the secondary mercury production process has not been recently tested, virtually no data are available for this process. In 1973, emission factors were estimated to be 20 kg of mercury emitted per megagram of mercury processed (40 lb/ton) due to uncontrolled emissions over the entire process (Anderson, 1973). These data have a high degree of uncertainty associated with them for the following reasons: (1) they are 20 years old, and processes may have changed substantially since they were generated; and (2) no information is available on their bases.

In 1991, 122 Mg (134 tons) of mercury were recycled from industrial scrap and 215 Mg (237 tons) from government stocks in the United States (Josinski, 1992). This total (337 Mg [371 tons]) does not include in-house mercury reclamation at industrial plants using mercury. Multiplying the total mercury recycled by the emission factor gives a mercury emission estimate of 6.7 Mg (7.4 tons) for 1991. Again, this estimate has a high degree of uncertainty because of the limited reliability of the emission factor.

Fluorescent lamp recycling

In order to reduce the net amount of mercury released to the environment, recycling of fluorescent lamps has become a more common practice. The recycling process begins with the crushing of the lamps to extract the white phosphor powder in them, which contains the bulk of mercury in lamps. Lamps can be crushed either by a mobile crushing unit at the point of collection, or by a centralized stationary crushing unit. Mercury emissions from crushing operations may be reduced using a vacuum collection system. In a vacuum collection system, air is passed through a cyclone to remove glass particles, followed by a filter to remove the phosphor powder, and a carbon adsorber to capture the mercury vapor, before being exhausted (Battye et al., 1994).

Mercury is recovered from crushed lamps by heating the crushed material to vaporize the mercury and then cooling the offgas stream to condense liquid elemental mercury (Battye et al., 1994). This can be accomplished in closed vessels called retorts or in open-hearth furnaces, ovens, or rotary kilns referred to as roasters. Retorting generally gives higher recovery rates than does roasting and is well suited to wastes containing volatile forms of mercury (Battye et al., 1994).

As presented previously in Figure 3-1, 2 percent of fluorescent lamps are estimated to be recycled each year. Air emission and mass balance information for fluorescent lamp recycling facilities was only available from one company. Based on this information, it was determined that only 1 percent of the mercury entering the recycling facility is emitted. This is equal to 0.005 Mg, or 0.02 percent of the mercury entering the MSW system (Truesdale et al., 1993). This number may be increasing due to the increasing number of recycling plants. Most lamp recycling facilities are located in California and Minnesota (Battye et al., 1994).

4.2.3 Chlor-alkali Production Using the Mercury Cell Process

Chlor-alkali production using the mercury cell process, which is the only chlor-alkali process using mercury, accounted for 14.7 percent of all U.S. chlorine production in 1993 (Dungan, 1994). Although most chlor-alkali plants use diaphragm cells, the mercury cell is still in use at some facilities. Each mercury cell may contain as much as 3 tons of mercury, and there are close to 100 cells at each mercury cell plant, making chlor-alkali plants a well-known source of mercury release. The chlor-alkali industry, however, is gradually moving away from mercury cell production and toward a membrane cell process because the membrane cell process does not use mercury and is more energy efficient than the mercury cell process (Rauh, 1991). Companies have been waiting until major capital investments are required for current installations before converting to processes that do not use mercury. When chlor-alkali plants replace mercury cells with alternative technologies, thousands of tons of mercury have to be disposed of as hazardous waste. There is currently no approved disposal method for mercury; only recovery/recycling of mercury is currently allowed under RCRA.

Table 4-13 lists U.S. mercury-cell chlor-alkali production facilities and their capacities. Figure 4-10 shows the location of these facilities across the U.S. The chlor-alkali industry is the largest user of mercury; however, the amount of chlorine produced using mercury cells has declined over the past 20 years (Cole et al., 1992). According to the Chlorine Institute, there are 14 chlor-alkali plants that currently use mercury cells compared to 25 facilities, 20 years ago (The Chlorine Institute, 1991). With the downward trend of chlor-alkali production, there are no plans for construction of new mercury-cell chlor-alkali facilities (Rauh, 1991).

The three primary sources of mercury air emissions are the (1) byproduct hydrogen stream, (2) end box ventilation air and (3) cell room ventilation air. The byproduct hydrogen stream from the decomposer is saturated with mercury vapor and may also contain fine droplets of liquid mercury. The quantity of mercury emitted in the end box ventilation air depends on the degree of mercury saturation and the volumetric flow rate of the air. The amount of mercury in the cell room ventilation air is variable and comes from many sources, including end box sampling, removal of mercury butter from end boxes, maintenance operations, mercury spills, equipment leaks, cell failure, and other unusual circumstances (U.S. EPA, 1984).

Mercury cell chlor-alkali facilities use pollution prevention methods to minimize mercury emissions to the environment. In the United States many facilities are installing thermal desorption or alternate technology to reduce mercury discharges to land (hazardous waste disposal sites). The amount of training provided to employees and the number of inspections have been increased to

Table 4-13
1991 U.S. Mercury-Cell Chlor-Alkali Production Facilities^a

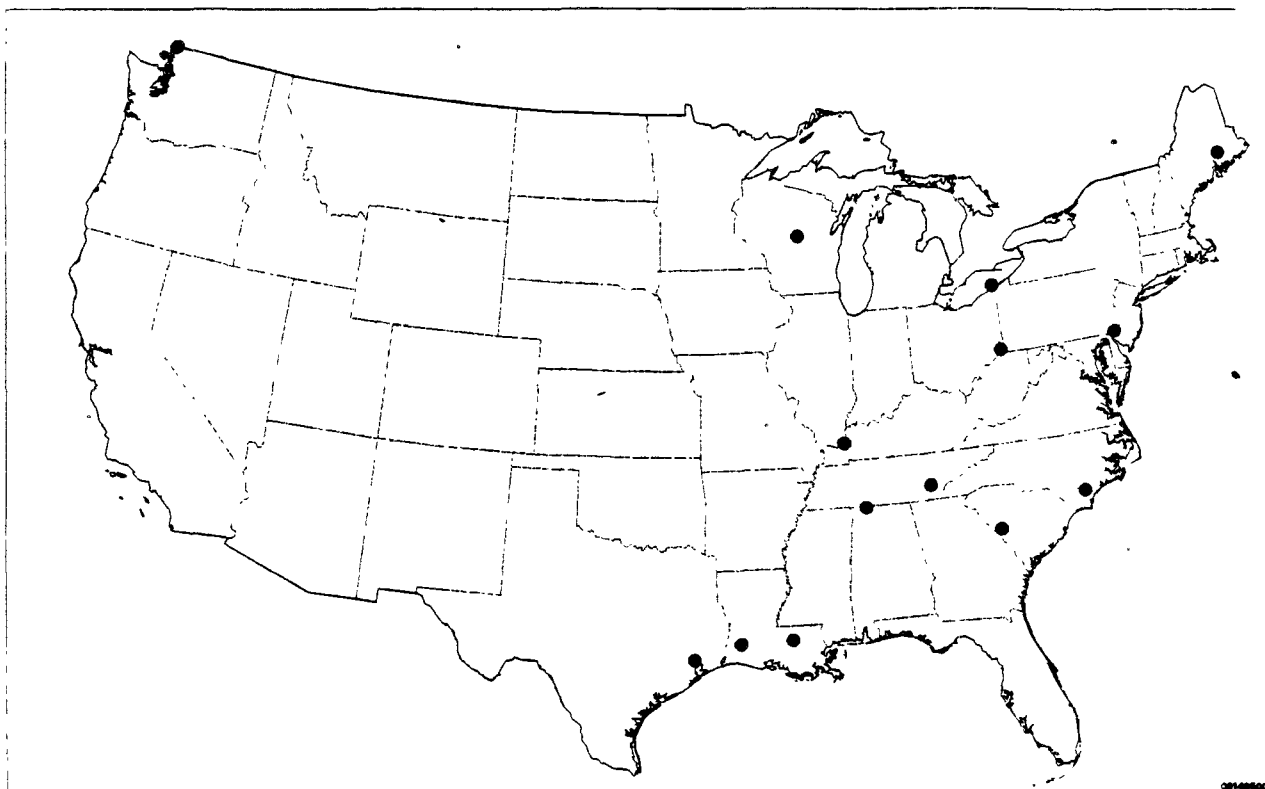
Facility	Location	Capacity, 10 ³ Mg/yr	Capacity, 10 ³ tons/yr	1991 emissions ^b lb/yr
Georgia-Pacific Corp., Chemical Division	Bellingham, WA	82	90	200
BF Goodrich, Chemical Group	Calvert City, KY	109	120	1,206
Hanlin Group, Inc., LCP Chemicals Division	Reigelwood, NC	48	53	528
	Orrington, ME	76	80	735
ASHTA Chemicals, Inc.	Ashtabula, OH	36	40	N/A
Occidental Petroleum Corporation, Electrochemicals Division	Deer Park, TX	347	383	1,290
	Delaware City, DE	126	139	532
	Muscle Shoals, AL	132	146	184
Olin Corporation, Olin Chemicals	Augusta, GA	102	112	1,540
	Charleston, TN	230	254	1,892
Pioneer Chlor Alkali Company, Inc.	St. Gabriel, LA	160	176	1,240
PPG Industries, Inc., Chemicals Group	Lake Charles, LA	233	256	1,440
	New Martinsville, WV	70	77	1,085
Vulcan Materials Company, Vulcan Chemicals Division	Port Edwards, WI	65	72	1,030
TOTAL		1,816	1,998	12,902 (5,858 kg/yr)

N/A = Not available from survey questionnaires. It is assumed that facilities not reporting mercury emissions produce no mercury emissions.

^a SRI International, 1991

^b BF Goodrich, 1992; Georgia-Pacific, 1993; LCP Chemicals, 1993a; LCP Chemicals, 1993b; Occidental, 1993a; Occidental, 1993b; Occidental, 1993c; Olin Chemicals, 1993a; Olin Chemicals, 1993b; Pioneer Chlor-Alkali, 1993; PPG Industries, 1993a; PPG Industries, 1993b; Vulcan Materials, 1993

Figure 4-10
Chlor-Alkali Production Facilities



The Mercury-Cell Chlor-Alkali Process

The mercury-cell chlor-alkali process consists of two electrochemical cells, the electrolyzer and the decomposer. A purified solution of saturated sodium or potassium brine flows from the main brine saturation section, through the inlet end box and into the electrolyzer. The brine flows between stationary activated titanium anodes suspended in the brine from above and a mercury cathode, which flows concurrently with the brine over a steel base (U.S. EPA, 1984).

Chlorine gas is formed at the electrolyzer anode and is collected for further treatment. The spent brine is recycled from the electrolyzer to the main brine saturation section through a dechlorination stage. Sodium is collected at the electrolyzer cathode, forming an amalgam containing from 0.25 to 0.5 percent sodium. The outlet end box receives the sodium amalgam from the electrolyzer, keeping it covered with an aqueous layer to reduce mercury emissions. The outlet end box also allows removal of thick mercury "butter" that is formed through the outlet end box into the second cell (the decomposer) (U.S. EPA, 1984).

The decomposer is a short-circuited electrical cell in an electrolytic sodium hydroxide solution. This cell has the sodium amalgam as the anode and graphite or metal as the cathode. Water added to the decomposer reacts with the sodium amalgam to produce elemental mercury, sodium hydroxide and hydrogen gas (a byproduct). The mercury, stripped of sodium, is recirculated to the cell through the inlet end box. The caustic soda solution typically leaves the decomposer at a concentration of 50 percent (by weight) and is filtered and further concentrated by evaporation. The byproduct hydrogen gas may be vented to the atmosphere, burned as a fuel, or used as a feed material for other processes (U.S. EPA, 1984).

reduce the possibilities of mercury releases. In addition, equipment has been upgraded to reduce the likelihood of mercury spills (The Chlorine Institute, 1991).

The control techniques that are typically used to reduce the level of mercury in the hydrogen streams and in the ventilation stream from the end boxes are these: (1) gas stream cooling, (2) mist eliminators, (3) scrubbers, and (4) adsorption on activated carbon or molecular sieves. Mercury emissions via the cell room air circulation are not subject to specific emission control measures. Concentrations are maintained, however, at acceptable worker exposure levels through good housekeeping practices and equipment maintenance procedures (U.S. EPA, 1984).

Gas stream cooling may be used as the primary mercury control technique or as a preliminary removal step to be followed by a more efficient control device. The hydrogen gas stream from the decomposer exits at 93 to 127°C (200 to 260°F) and passes into a primary cooler. In this indirect cooler, a shell-and-tube heat exchanger with ambient temperature water is used to cool the gas stream to 32 to 43°C (90 to 110°F). A knockout container following the cooler is used to collect the mercury. If additional mercury removal is desired, the gas stream may be passed through a more efficient cooler or another device. Direct or indirect coolers using chilled water or brine provide for more efficient mercury removal by decreasing the temperature of the gas stream to 3 to 13°C (37 to 55°F). Regardless of the gas stream treated, the water or brine from direct contact coolers requires *water treatment prior to reuse or discharge because of the dissolved mercury in the liquid* (U.S. EPA, 1984).

Mist eliminators (most commonly the filter pad type) can be used to remove mercury droplets, water droplets, or PM from the cooled gas streams. Particles trapped by the pad are removed by periodically spraying the pad and collecting and treating the spray solution (U.S. EPA, 1984).

Scrubbers are used to absorb the mercury chemically from both the hydrogen stream and the end box ventilation streams. The scrubbing solution is either depleted brine from the mercury cell or a sodium hypochlorite (NaOCl) solution. These solutions are used in either sieve plate scrubbing towers or packed-bed scrubbers. Mercury vapor and mist react with the sodium chloride or hypochlorite scrubbing solution to form water-soluble mercury complexes. If depleted brine is used, the brine solution is transferred from the scrubber to the mercury cell, where it is mixed with fresh brine, and the mercury is recovered by electrolysis in the cell (U.S. EPA, 1984).

Sulfur- and iodine-impregnated carbon adsorption systems are commonly used to reduce the mercury levels in the hydrogen gas stream if high removal efficiencies are desired. This method requires pretreatment of the gas stream by primary or secondary cooling followed by mist eliminators to remove about 90 percent of the mercury content of the gas stream. As the gas stream passes through the carbon adsorber, the mercury vapor is initially adsorbed by the carbon and then reacts with the sulfur or iodine to form the corresponding mercury sulfides or iodides. Several adsorber beds in series can be used to reduce the mercury levels to the very low parts per billion (ppb) range (U.S. EPA, 1984).

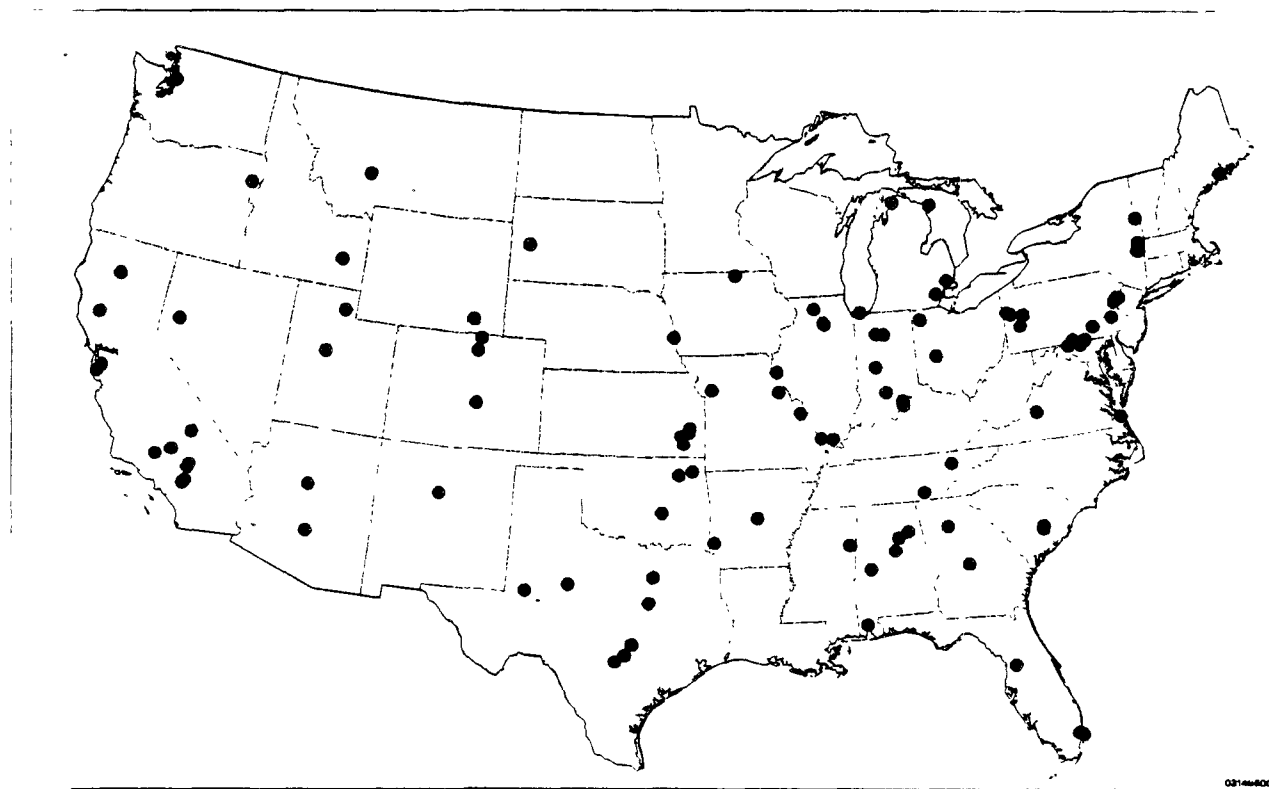
Mercury emissions data from chlor-alkali facilities were obtained from Clean Air Act section 114 survey questionnaires (BF Goodrich, 1992; Georgia-Pacific, 1993; LCP Chemicals, 1993a; LCP Chemicals, 1993b; Occidental, 1993; Olin Chemicals, 1993a; Olin Chemicals, 1993b; Pioneer Chlor Alkali, 1993; PPG Industries, 1993a; PPG Industries, 1993b; Vulcan Materials, 1993). The data reported are for 1991. The reported mercury emissions were 5.9 Mg (6.5 tons) and included 13 of the 14 mercury cell chlor-alkali production facilities listed in Table 4-11. Data are also available from the Toxic Release Inventory (TRI) (U.S. EPA, 1992e). Those facilities not reporting mercury emissions in

the section 114 questionnaire responses or in the 1991 TRI were assumed to produce no mercury emissions because it is a Federal law to report such emissions.

4.2.4 Cement Manufacturing

United States cement kiln capacity data for 1990 showed a total of 212 U.S. cement kilns with a combined total capacity of 73.5×10^6 Mg (81×10^6 tons) (U.S. EPA, 1993a). Of this total, 201 kilns were active and had a total clinker capacity of 71.8×10^6 Mg (79.1×10^6 tons) (U.S. EPA, 1993a). Because the majority (96 percent) of this cement was portland cement, portland cement production processes and emissions are the focus of this section (U.S. EPA, 1993a). Total mercury emissions from the portland cement process are estimated to be 5.9 Mg (6.5 tons). In 1990, 68 percent of portland cement was produced by the dry process and 32 percent by the wet process (Portland Cement Association, 1991). The locations of active cement manufacturing plants in the continental U.S. are shown in Figure 4-11.

Figure 4-11
Cement Manufacturing Plants



The primary sources of mercury emissions from portland cement manufacturing are expected to be from the kiln and preheating/precalcining steps. Small quantities of mercury may be emitted as a contaminant in the PM from process fugitive emission sources. Process fugitive emission sources include materials handling and transfer, raw milling and drying operations in dry process facilities and finish milling operations. Typically, PM emissions from these process fugitive sources are captured by a ventilation system controlled with a fabric filter. No data are available on the ability of these systems to capture mercury emissions from cement kilns.

The Portland Cement Manufacturing Process

The portland cement manufacturing process can be divided into four major steps: raw material acquisition and handling, kiln feed preparation, pyroprocessing, and finished cement grinding (U.S. EPA, 1993a).

The initial step in the production of portland cement manufacturing is acquiring raw materials, including limestone (calcium carbonate) and other minerals such as silica.

Raw material preparation, the second step in the process, includes a variety of blending and sizing operations designed to provide a feed with appropriate chemical and physical properties. Raw material processing differs somewhat for the "wet" and "dry" processes. At dry process facilities, the moisture content in the raw material, which can range between 2 and 35 percent, is reduced to less than 1 percent. Heat for drying is often provided by the exhaust gases from the pyroprocessor (i.e., kiln). At facilities where the wet process is used, water is added to the raw material during the grinding step, thereby producing a pumpable slurry containing approximately 65 percent solids.

Pyroprocessing (thermal treatment) of the raw material is carried out in a rotary kiln, which is the heart of the Portland cement manufacturing process. During pyroprocessing, the raw material is transformed into clinkers, which are gray, glass-hard, spherically shaped nodules that range from 0.32 to 5.1 cm (0.125 to 2.0 in.) in diameter.

The rotary kiln is a long, cylindrical, slightly inclined, refractory-lined furnace. The raw material mix is introduced in the kiln at the elevated end, and the combustion fuels are introduced into the kiln at the lower end, in a countercurrent manner. The rotary motion of the kiln transports the raw material from the elevated end to the lower end. Fuel such as coal or natural gas (or occasionally oil) is used to provide energy for calcination and sintering. Other fuels, such as shredded municipal garbage, chipped rubber, petroleum coke, and waste solvents are also being used more frequently. Mercury is present in coal and oil and may also be present in appreciable quantities in the waste-derived fuels mentioned above. Because mercury evaporates at approximately 350°C (660°F), most of the mercury present in the raw materials may be emitted during the pyroprocessing step. Combustion of fuel during the pyroprocessing step also contributes to mercury emissions. Pyroprocessing can be accomplished by one of four different processes: wet process, dry process, dry process with a preheater, and dry process with a preheater/precalciner. These processes accomplish the same physical and chemical steps described above.

The last step in the pyroprocessing is cooling the clinker. This process step recoups up to 30 percent of the heat input to the kiln system, locks in desirable product qualities by freezing mineralogy, and makes it possible to handle the cooled clinker with conventional conveying equipment. Finally, after the cement clinker is cooled, a sequence of blending and grinding operations is carried out to transform the clinker into finished portland cement.

In the pyroprocessing units, PM emissions are controlled by fabric filters and ESPs. Clinker cooler systems are controlled most frequently with pulse jet or pulse plenum fabric filters. No data are available on the ability of these control systems to capture mercury emissions from cement kilns.

Mercury present in the raw material and the fuel is likely to be emitted from all four cement processes summarized in the box on the next page. Cement kiln test reports were reviewed from a number of facilities performing Certification of Compliance (COC) tests which are required of all kilns burning waste-derived fuel (WDF). Emission tests from two other kilns were also reviewed in this analysis. In all, 15 test runs provided enough information to calculate an emission factor (some of

these were from the same kiln). This information included clinker production as well as mercury emission rates and process conditions.

Although the emission tests were COC tests, the data used to calculate the emission factor were taken from the "background" test runs when the usual fuel was being fired at normal operating conditions. If abnormal operating conditions were noted in the test report (e.g., hotter than normal clinkering), then the test results were not used. In some test reports, the mercury emissions were similar when either coal or WDF was being fired. If the emissions from the WDF burn was no higher than the emissions when coal alone was fired, the emissions data across test runs were averaged.

The calculated emission factors ranged from 0.0023 to 0.49 g/Mg of clinker (4.5×10^{-6} to 9.7×10^{-4} lb/ton). The average emission factor for the 15 test runs was 0.087 g/Mg of clinker (1.7×10^{-4} lb/ton)(U.S. EPA, 1993a).

Kilns firing WDF or coal only may have different mercury emissions than the average emission factor would indicate. As a result, the emissions estimated here for the portland cement industry as a whole (i.e., including kilns that fire fossil fuel or WDF) may underestimate or overestimate emissions as the mercury content of both WDF and coal may vary. Seven percent of the entire fuel consumption of the cement manufacturing industry in 1990 consisted of WDF. Coal was the largest fuel source, at over 71 percent of consumption, followed by coke at over 13 percent of consumption (Carrol, 1994).

The total production of portland cement in 1990 was 67.5×10^6 Mg (74.5×10^6 tons) (95.7 percent of the total cement production) (U.S. EPA, 1993a). Of the total production of portland cement, 96 percent was clinker, and the remaining 4 percent was other ingredients (U.S. EPA, 1993a). Multiplying the total amount of clinker produced in 1990 (total production of portland cement multiplied by 96 percent) by the above average emission factor gives an estimate of 5.9 Mg (6.5 tons) of mercury emissions from portland cement manufacturing for 1990. This emissions estimate is expected to be updated in 1995 during development of U.S. EPA's Combustion Strategy as well as for rulemaking for this source category under section 112 of the CAA.

Industry estimates for this category (based on a review of the same test data) are 2.9 Mg (3.3 tons) of mercury emitted in 1990. This estimate is calculated using an estimated average emission factor of 9.1×10^{-5} lb of mercury per ton of clinker (Portland Cement Association, 1995).

4.2.5 Primary Copper Smelting

Copper is recovered from a sulfide ore principally by pyrometallurgical smelting methods. The ore contains significant quantities of arsenic, cadmium, lead, antimony and mercury. Table 4-14 gives the locations and 1992 production capacities of primary copper smelters currently operating in the United States; these smelter locations are displayed in Figure 4-12.

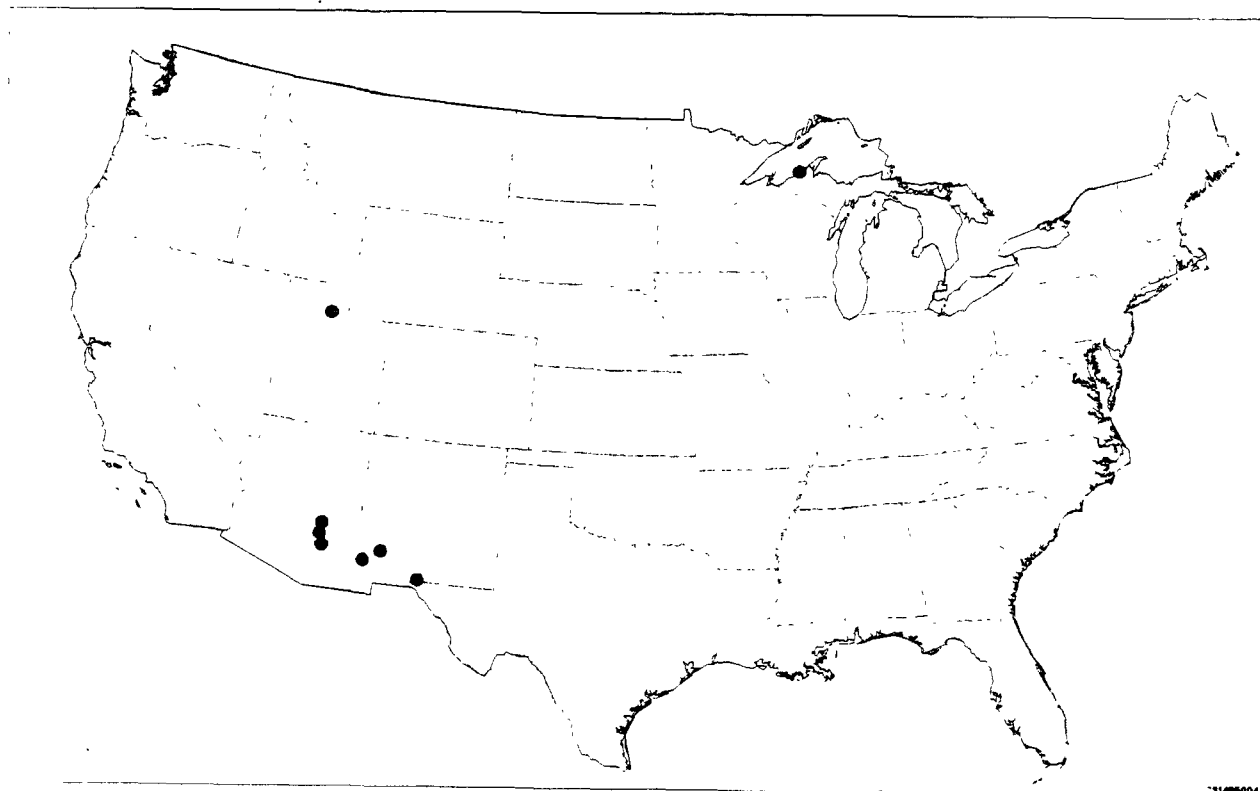
Copper smelters use high efficiency air pollution control options to control PM and SO₂ emissions from smelting furnaces and convertors. Electrostatic precipitators are the most common PM control device at copper smelters. Control of SO₂ emissions is achieved by absorption to sulfuric acid in the sulfuric acid plants, which are common to all copper smelters.

Table 4-14
1992 U. S. Primary Copper Smelters and Refineries

Smelter	Location	1992 Capacity, Mg (tons)
ASARCO Inc.	Hayden, AZ	190,900 (210,200)
Cyprus Miami Mining Co.	Globe, AZ	180,000 (198,000)
BHP Copper Co.	San Manuel, AZ	290,000 (319,000)
Copper Range Co. ^a	White Pine, MI	60,000 (66,000)
Phelps Dodge	Hidalgo, NM	190,000 (209,000)
Chino Mines Co.	Hurley, NM	170,000 (187,000)
ASARCO Inc.	El Paso, TX	104,000 (114,400)
Kennecott	Garfield, UT	210,000 (231,000)

^a Ceased operations in February 1995.

Figure 4-12
Primary Copper Smelters



The Primary Copper Smelting Process

The copper smelting process sequentially involves drying ore concentrates, smelting of ore concentrates to produce matte, converting matte to produce blister copper, and fire refining the blister copper in an anode furnace. After fire refining, the 99.5 percent pure copper is cast into "anodes" and sent to an electrolytic refinery for further impurity removal (Buonicore and Davis, 1992).

All of the currently operating copper smelters use either fluid bed or rotary kiln dryers to dry the concentrate. Temperatures in the dryer are not high enough to vaporize any mercury in the ore concentrate. Roasting of ores is no longer used because the off gases from the roasting process were too low in SO_2 to be processed in the sulfuric acid plant.

Smelting produces a copper matte by melting the hot ore concentrates with siliceous flux in a furnace. The mattes produced by domestic smelters range from 35 to 65 percent copper. Smelting furnace technologies operate at temperatures well above the boiling point of mercury with operating ranges as high as 2500°C (4530°F). Any mercury contained in the concentrate will likely be emitted during the flash smelter process step and directed to the sulfuric acid plant (Buonicore and Davis, 1992). The gas stream to the sulfuric acid plant passes through three to five control devices, such as dry ESPs, cyclones, scrubbing towers, cooling towers and acid mist ESPs. These control devices are required to remove metal impurities to prevent destruction of the catalyst in the acid plant. Any mercury volatilizing in the smelting furnace is removed in these multistage control systems and in the sulfuric acid plant. Limited data on sulfuric acid plant sludges show that the mercury is present in measurable concentrations. This mercury is recycled back to the flash converter and vaporized again into the control system. This appears to set up an internal recycling loop for the mercury, which is ultimately discarded with the solid waste.

The final step in the production of molten "blister" copper is converting. Converting eliminates remaining iron and sulfur impurities, leaving 98.5 to 99.5 percent pure copper. Converting involves molten matte, siliceous flux and scrap copper being charged in a rotating cylindrical shell, where air or oxygen rich air is blown through the molten matte. Blowing and slag skimming are repeated until relatively pure Cu_2S , called "white metal" accumulates in the bottom of the converter. A renewed air blast then oxidizes the copper sulfide to SO_2 , leaving blister copper. Blister copper is then removed and transferred to refining facilities. Further purification may involve fire refining and electrolytic refining (Buonicore and Davis, 1992).

Mercury emission data for primary copper smelting facilities are very limited. One emission test report for the Copper Range Company located in White Pine, Michigan containing metals analysis results was available (TRC Environmental, 1992). Before ceasing operations in February 1995, this facility operated a reverberatory furnace with an ESP to control PM. The exhaust stream from the converter (which is uncontrolled) was mixed with the exhaust from the ESP outlet and was routed through the main stack and discharged into the atmosphere. Testing for metals was performed at a location in the main stack downstream from the point where the two exhaust streams (from the ESP outlet and the converter) are combined. Mercury emissions were measured for three modes of converter operation: slag-blow, copper-blow and converter idle (no blow) cycles. The mercury level during the converter idle cycle was measured to be the highest, corresponding to a mercury emission rate of 0.0753 kg/hr (0.166 lb/hr). During the slag-flow and copper-flow periods, the emission rates were 0.0494 kg/hr (0.109 lb/hr) and 0.0635 kg/hr (0.140 lb/hr), respectively. Additionally, the plant capacity was reported to be approximately 38 Mg/hr (42 tons/hr) of feed, which consists of mill concentrate, limestone, iron pyrites and recycled material. The actual process rate during the test is not known, so an emission factor cannot be calculated from this test.

Because the feed mix varies from facility to facility and because the Copper Range Company is the only facility in the United States that operated a reverberatory furnace without any acid plants, the emission data from the Copper Range Company are not representative of current industry practice. A mercury emissions estimate of 0.6 Mg/yr (0.7 tons/yr) from this one facility was, therefore,

calculated using a weighted average of the emission rates for the three modes of converter operation (0.068 kg/hr [0.15 lb/hr]), and an operating schedule of 8,760 hr/yr. The Copper Range facility also is no longer active.

A more recent analysis of the seven copper smelters currently operating in the U.S. has been performed since the 1990 emissions estimate described above. Mercury emission rates from these seven smelters are presented in Table 4-15 along with the mercury concentration of ore. These data show that emissions range from less than 1 lb/year to 40 lbs/year. These emission rates are based on both stack testing and engineering judgment. As a result, the U.S. EPA estimates 1995 nationwide mercury emissions from primary copper smelters to be about 0.09 Mg/year (0.08 tons/year).

Table 4-15
Mercury Ore Concentrate and Emissions from
Primary Copper Smelters in the U.S.

Smelter	Mercury in Ore Concentrate lb/yr	Mercury Emissions lb/yr	Basis of Emission Values
ASARCO - El Paso	1,769	1.8	Emissions Test
ASARCO - Hayden	2,444	35	Emissions Test and Engineering Judgment
Copper Range	940	1,951	Emissions Test
Cyprus Miami	CBI ^a	34	Emissions Test
Kennecott	NA ^a	35	Emissions Test and Engineering Judgment
BHP Copper Co.	2,240	40	Emissions Test and Engineering Judgment
Phelps Dodge-Hidalgo	5,768	0.09	Engineering Judgment
Phelps Dodge-Chino	585	7.5	Engineering Judgment

^a CBI means Confidential Business Information that is unavailable to the public. NA means not available.

4.2.6 Lime Manufacturing

Lime is produced in various forms, with the bulk of production yielding either hydrated lime or quicklime. In 1992, producers sold 16.4×10^6 Mg (18×10^6 tons) of lime produced at 113 plants in 32 States and Puerto Rico. The 1992 production represented a 4 percent increase over 1991 production. The leading domestic uses for lime include steelmaking, flue gas desulfurization, pulp and paper manufacturing, water purification, and soil stabilization (Miller, 1993).

Table 4-16 identifies the top 10 lime-producing plants in the United States, in order of total output for 1991 (Hammond, 1993). Lime production is geographically concentrated as demonstrated by 1989 production data, when 63 percent of the U.S. total was produced in seven States (in order of decreasing production: Missouri, Ohio, Pennsylvania, Alabama, Kentucky, Texas and Illinois) (Bureau of Mines, 1991).

Table 4-16
Leading 1991 U.S. Lime Producing Plants^a

Plant	Company	Location
Ste. Genevieve	Mississippi Lime Company	Ste. Genevieve County, MO
Maysville Division	Dravo Lime Company	Mason County, KY
Black River Division	Dravo Lime Company	Pendleton County, KY
Montavello Plant	Allied Lime Company	Shelby County, AL
Woodville Plant	Martin Marietta Magnesia	Sandusky County, OH
Longview Division	Dravo Lime Company	Shelby County, AL
South Chicago Plant	Marblehead Lime Company	Cook County, IL
Nelson Plant	Chemstar, Inc.	Yavapai County, AZ
Clifton Plant	Chemical Lime, Inc.	Bosque County, TX
Annville Plant	Wimpey Minerals PA, Inc.	Lebanon, PA

^a Hammond, 1993.

Commercial production of the various forms of lime involves the following basic steps.

1. Quarrying raw limestone (or limestone and dolomite);
2. Stone processing or crushing and sizing in preparation for calcination;
3. Calcining the crushed stone in high temperature kilns (producing quicklime);
4. Hydrating the processed lime (to produce hydrated lime from quicklime); and
5. Miscellaneous transfer, storage and handling processes.

Emissions from quarrying or stone extraction are largely restricted to fugitive dust, as are the emissions from stone processing, crushing and sizing. Mercury emissions are expected to be negligible from these initial steps in lime production.

Calcining, which uses high-temperature kilns to convert carbonate to oxide (removing CO₂), is the lime production step from which most mercury emissions are expected. Rotary kilns are primarily used in the calcining step in the United States, accounting for 90 percent of domestic lime kilns. Other types of lime kilns include vertical kilns, rotary hearths and fluidized bed kilns. During calcination, kiln temperature may reach as high as 1820°C (3300°F) (U.S. EPA, 1993a). Because mercury has a boiling point of 350°C (660°F), most of the mercury that exists as impurities in the processed stone will likely emit as an air pollutant during calcination.

Fuels, including primarily coal, oil, petroleum coke, or natural gas, are used to provide the energy for calcination. Petroleum coke is usually used in combination with coal. Auxiliary fuels may include shredded municipal garbage, chipped rubber, or waste solvent. Mercury is expected to be present in the coal, oil, and possibly in appreciable quantities in any waste-derived fuels. Any mercury emitted from fuel combustion will occur during the calcination step and will be discharged as vapor kiln exhausts.

The quicklime that is produced by calcination can be hydrated with water to produce hydrated lime or slaked lime ($\text{Ca}(\text{OH})_2$). The hydration step may be immediately preceded by some crushing, pulverizing and separation of dolomitic quicklime to form high calcium and dolomitic quicklime. These processes and handling, storage and transfer are not likely sources for mercury emissions during lime production.

Air pollution control devices for lime kilns are primarily used to recover product or control fugitive dust and PM emissions. Calcination kiln exhaust is typically routed to a cyclone for product recovery and then routed through a fabric filter or ESPs to collect fine particulate emissions. Other emission controls found at lime kilns include wet scrubbers (typically venturi scrubbers). How well these various air pollution control devices perform relative to vapor phase mercury emissions in lime production is not well documented. The control efficiencies are expected to be similar to those observed in the production of portland cement, however, because of the similarities in the process and control devices.

Representative estimates of mercury emissions from lime manufacturing are not possible based on the available data from lime kilns around the country. An ongoing U.S. EPA study to update AP-42, Section 8.15, on lime manufacturing emission factors has reviewed and summarized test data for lime calcining at 93 kilns (U.S. EPA, 1992f). Pollutants identified and noted in a summary of the test data did not indicate any mercury emissions and gave little or no indication that emissions tests at lime kilns have sampled and analyzed for trace metals. Two very limited estimation efforts for mercury emissions are offered in the following discussion: one using 1983 mercury emission test data from five Wisconsin lime plants; and, the other from a draft report on 1983 mercury emission data from a pulp and paper lime kiln, which is referenced in SPECIATE.

An emission estimate based on mass balances generated from information for mercury content in limestone from the five operating lime kilns in Wisconsin in 1983 was 18 kg/yr (39 lb/yr) for all the kilns combined (Bureau of Air Management, 1986). In 1983, these five lime plants produced 0.29×10^6 Mg (0.32×10^6 tons) of lime (Miller, 1993b). Assuming uniform emissions for each ton of production suggests that 5.53×10^{-5} kg (1.22×10^{-4} lb) of mercury were emitted for each Mg (ton) of lime produced. These data do not account for any differences in fuel used to heat the kilns or any differences in raw materials used. Natural gas, which is believed to contain negligible amounts of mercury, is used to fire 33 percent of the lime kilns. Total estimated annual emissions would, therefore, be reduced by 33 percent to reflect the lack of mercury emissions from natural gas.

If the Wisconsin data are extrapolated to the lime production in the United States in 1992, an annual estimate of mercury emissions from lime kilns of 0.91 Mg/yr (1.00 ton/yr). Assuming that 33 percent of lime kilns use natural gas as their fuel source and produce no mercury emissions reduces this estimate to 0.6 Mg/yr (0.7 tons/yr). This estimate has a high level of uncertainty because material composition could vary significantly across the country, and the fuel type(s) used in Wisconsin may not be representative of those used nationally.

4.2.7 Electrical Apparatus Manufacturing

Mercury is one of the best electrical conductors among the metals and is used in five areas of electrical apparatus manufacturing: electric switches, thermal sensing elements, tungsten bar sintering, copper foil production, and fluorescent light production. Overall mercury emissions from electrical apparatus manufacturing were estimated to be 0.5 Mg (0.5 ton) in 1992. No information on locations of manufacturers of electrical apparatus that specifically contain mercury is available.

4.2.7.1 Electric Switches

The primary use of elemental mercury in electrical apparatus manufacturing is in the production of electric switches (electric wall switches and electric switches for thermostats). Wall switches consist of mercury, metal electrodes (contacts) and an insulator in button-shaped metal cans. Electric switches containing mercury have been manufactured since the 1960s with approximately one million produced annually.

Electric Switch Manufacturing Process

The wall switches are manufactured by first assembling a component consisting of a metal ring, a glass preform, a ceramic center, and a center contact. This subassembly is then transferred to a rotating multistation welding machine, located in an isolation room, where it is filled with approximately 3 g (0.11 oz.) of mercury. The filled subassembly is placed in the button-shaped can, evacuated, and welded shut. The assembled buttons then leave the isolation room and are cleaned, zinc-plated and assembled with other components to form the completed wall switches (Reisdorf and D'Orlando, 1984).

Thermostat switches are constructed using a short glass tube with wire contacts sealed in one end of the tube. First, metal electrodes (contacts) are inserted into small tubes. The tubes are then heated at one end, constricted and crimped closed around the electrodes (sealing the electrodes into the glass tube), and the apparatus is cleaned. The subassembly is then transferred to the isolation fill room where mercury is added. The open end of the mercury-filled tube is then heated, constricted and sealed. The filled tubes then leave the isolation room, and wire leads are attached to the electrode contacts, which completes the switch assembly (Reisdorf and D'Orlando, 1984).

During electric switch manufacture, mercury may be emitted during welding or filling operations, as a result of spills or breakage, during product testing, and as a result of product transfer. Often, emissions can be controlled by using effective gaskets and seals to contain mercury in the process streams. Also, good work practices, such as discarding rejected and broken switches under water and reducing the temperature in the fill room, can effectively suppress mercury vaporization. Furthermore, local exhaust ventilation, custom-designed to fit specific equipment, can reduce mercury vapor and mercury PM (Reisdorf and D'Orlando, 1984).

The amount of mercury used for the manufacture of switches and thermostats decreased 50 percent from 155 tons in 1989 to 77 tons in 1990. This decrease in mercury use for the manufacture of electric switches may be attributable to the shift to solid state devices and other alternatives. The recent decrease in the construction of houses may have also contributed to the decrease in mercury use for electric switch manufacture (Cole et al., 1992).

The amount of mercury disposed each year in electric switches compared to the amount of mercury in electric switches in use is small. One recent study estimated that 10 percent of switches

are discarded after 10 years, 40 percent after 30 years and the remaining 50 percent after 50 years (U.S. EPA, 1992a). Average unit life for mercury thermostats exceeds 20 years, with upgrading, remodeling or building demolition being the principal causes for removal from service (National Electrical Manufacturers Association, 1995). In addition, a few will be discarded due to leakage or some other failure.

Table 4-17 summarizes the discards of mercury in electric switches. In these estimates it was assumed that there is no recycling of mercury from discarded switches. In 1994, however, Honeywell, Inc., a major manufacturer of thermostats announced a pilot project in Minnesota to recycle mercury thermostats. Homeowners and contractors can send unneeded thermostats back to Honeywell so the mercury can be removed and recycled. In addition, in 1995, U.S. EPA announced a "Universal Waste Rule" (which includes thermostats) that effectively allows for the transportation of small quantities of mercury from specific products. This ruling should encourage recycling. Until programs such as these are fully implemented, it is unclear how much the mercury discards from this type of product will decline in MSW.

Table 4-17
Discards of Mercury in Electric Switches^a

Year	Electric Switch Production	Weight of Mercury in Switches (tons)	Weight of Mercury Discarded in MSW (tons)
1987	1,000,000	3.9	0.39
1988	1,000,000	3.9	0.39
1989	1,000,000	3.9	0.39
1995	1,000,000	3.9	1.93
2000	1,000,000	3.9	1.93

^a U.S. EPA, 1992a.

4.2.7.2 Thermal Sensing Instruments and Tungsten Bar Sintering

A thermal sensing instrument consists of a temperature-sensing bulb, a capillary tube, a mercury reservoir and a spring-loaded piston. The bulbs are made by cutting metal tubing to the correct size, welding a plug to one end of the tube and attaching a coupling piece to the other end. A capillary is cut to a specified length and welded to the coupling at the open end of the bulb. The other end of the capillary is welded to a "head" that houses the mechanical section of the sensor. The bulb and capillary assembly are filled with mercury by a multistation mercury filling machine that is housed in a ventilated enclosure. After filling, the sensor is transferred to a final assembly station, where a return spring and plunger are set into a temporary housing on the head of the sensor. In order to complete the temperature instrument, the sensor is then attached to a controller and/or indicating device (Reisdorf and D'Orlando, 1984).

Mercury is also used in tungsten bar sintering. Tungsten is used as a raw material in manufacturing incandescent lamp filaments. The manufacturing process starts with tungsten powder

pressed into long, thin bars of a specified weight. These bars are presintered and then sintered using a high-amperage electrical current. During the tungsten bar sintering process, mercury is used as a continuous electrical contact. The mercury contact is contained in pools (mercury cups) located inside the sintering unit.

After the sintering process is completed, the bars are cooled to ambient temperature to determine the density of the tungsten bar. Metallic mercury is normally used in these measurements because of its high specific gravity. In order to calculate the density of the tungsten bar, the tungsten bars are dipped into a pool of mercury and the weight of the displaced mercury is determined. When the bar is removed from the mercury pool, the mercury is brushed off into a tray of water that is placed in front of the pool (Reisdorf and D'Orlando, 1984).

No specific information on emission control measures for thermal sensing elements and tungsten bar sintering was found in the literature. It is assumed that mercury is emitted during the filling process for thermal sensing elements and during sintering and final density measurements for tungsten bar sintering (U.S. EPA, 1993a).

4.2.7.3 Copper Foil Production

High-purity copper foil, used as a laminate in printed circuit boards, is produced by an electrodeposition process using mercury as the electrical contacts. The initial step in the foil production process is the dissolution of scrap copper in sulfuric acid to form copper sulfate. The solution is then fed to the plating operation, where the copper ions are electrodeposited on rotating drums as copper metal. During the electrodeposition process, a current passes between a lead anode and a rotating drum cathode. As the drum rotates, the copper metal is electrodeposited on the drum surface in the form of a continuous thin foil sheet. The rotating drum requires using a rotating electrical contact between the electrical connection and the drum surface. Elemental mercury is used as the continuous contact between the rotating shaft of the drum and the electric connections. The liquid mercury is contained in a well located at one end of the rotating drum shaft (Reisdorf and D'Orlando, 1984).

During copper foil production, mercury can be emitted from the drum room and the treatment room of the copper plating process. Ventilated enclosures, with exhaust gases directed to mercury vapor filters, can be used to control mercury emissions, as can reducing the temperature of the mercury wells (Reisdorf and D'Orlando, 1984).

4.2.7.4 Fluorescent Lamps

All fluorescent lamps contain elemental mercury as mercury vapor inside the glass tube. Mercury has a unique combination of properties that make it the most efficient material for use in fluorescent lamps. Of the 550 million mercury-containing lamps sold in the United States annually, approximately 96 percent are fluorescent lamps. It is estimated that approximately the same amount of lamps are disposed of on an annual basis (National Electrical Manufacturers Association, 1992). In fluorescent lamp production, precut glass bulbs are washed, dried and coated with a liquid phosphor emulsion that deposits a film on the inside of the lamp bulb. Mount assemblies are fused to each end of the glass lamp bulb, which is then transferred to an exhaust machine. On the exhaust machine, the glass bulb is exhausted and a small amount (15 to 250 mg [3.3×10^{-5} to 5.5×10^{-4} lb]) of mercury is added. Some of the mercury combines with the emulsion on the interior of the bulb over its life. The glass bulb is filled with an inert gas and sealed. After the lamp bulbs are sealed, metal bases are attached to the ends and are cemented in place by heating.

The names and division headquarters of the fluorescent lamp manufacturers in the United States in 1995 are shown in Table 4-18 (U.S. EPA, 1993a).

Table 4-18
1992 U.S. Fluorescent Lamp Manufacturers' Headquarters^a

Company	Division headquarters
Duro-Test Corp. General Electric OSRAM Corp. ^b Philips Lighting Company	North Bergen, NJ Cleveland, OH Montgomery, NY Somerset, NJ

^a U.S. EPA, 1993a.

^b National Electrical Manufacturers Association, 1995.

During fluorescent lamp manufacturing, mercury can be emitted by transfer and parts repair during mercury handling; by the mercury injection operation; and from broken lamps, spills and waste material. Mercury air levels during lamp production steps are reduced by process modifications, containment, ventilated enclosures, local exhaust ventilation, and temperature control (Reisdorf and D'Orlando, 1984).

4.2.7.5 Emissions Summary for Electrical Apparatus Manufacturing

While mercury may be emitted from all of the aforementioned areas of electrical apparatus manufacturing, no specific data for mercury emissions from these areas were found in the literature and no emission test data were available to calculate mercury emissions from each area. One 1973 U.S. EPA report presents an emission factor of 4 kg of mercury emitted for each megagram of mercury used (8 lb/ton) in overall electrical apparatus manufacture (Anderson, 1973). This factor only pertains to emissions generated at the point of manufacture. This emission factor should be used with extreme caution, however, as it was based on engineering judgment and not on actual test data and because production and mercury control methods have probably changed considerably since 1973 to prevent waste and limit worker exposure. The emission factor may, therefore, substantially overestimate mercury emissions from this source.

In 1992, 104 Mg (116 tons) of mercury were used in all electrical apparatus production (36 Mg [40 tons] for electric lighting and 69 Mg [76 tons] for wiring devices and switches) (Bureau of Mines, 1992).³ Multiplying the emission factor above by the 1992 usage gives a mercury emission estimate of 0.42 Mg (0.46 ton) for electrical apparatus manufacture. Because of the lack of reliability of the emission factor, a high degree of uncertainty is associated with this emission estimate.

³ The Bureau of Mines previously reported mercury use for electric lighting to be 61 tons. This number is being revised to an estimated 40 tons (National Electric Manufacturers Association, 1995).

4.2.8 Instrument (Thermometers) Manufacturing

Mercury is used in many medical and industrial instruments for measurement and control functions. These instruments include thermometers, pressure-sensing devices and navigational devices. In 1992, an estimated 0.5 Mg (0.5 ton) of mercury was emitted from instrument manufacture; however, this estimate should be used with caution as discussed below.

It is beyond the scope of this study to discuss all instruments that use mercury in some measuring or controlling function. Although there is potential for mercury emissions from all instruments containing mercury, this section focuses only on the production of thermometers because they represent the most significant use, are usually disposed of in household waste (U.S. EPA, 1992a), and more information is available on thermometer manufacture than on the manufacture of other instruments.

There are generally two types of clinical thermometers: 95 percent are oral/rectal/baby thermometers, and 5 percent are basal (ambient air) temperature thermometers. An oral/rectal/baby thermometer contains approximately 0.61 grams of mercury and a basal thermometer contains approximately 2.25 grams (U.S. EPA, 1992a).

The Glass Thermometer Manufacturing Process

The production of glass thermometers begins by cutting glass tubes into required lengths and bore sizes. Next, either a glass or metal bulb, used to contain the mercury, is attached to the base of the tube. The tubes are filled with mercury in an isolated room. A typical mercury filling process is conducted inside a bell jar. Each batch of tubes is set with open ends down into a pan, and the pan set under the bell jar, which is lowered and sealed. The tubes are heated to approximately 200°C (390°F), and a vacuum is drawn inside the bell jar. Mercury is allowed to flow into the pan from either an enclosed mercury addition system or a manually filled reservoir. When the vacuum in the jar is released, the resultant air pressure forces the mercury into the bulbs and capillaries. After filling, the pan of tubes is manually removed from the bell jar. Excess mercury in the bottom of the pan is refiltered and used again in the process (Reisdorf and D'Orlando, 1984).

Excess mercury in the tube stems is forced out the open ends by heating the bulb ends of the tubes in a hot water or oil bath. The mercury column is shortened to a specific height by flame-heating the open ends (burning-off process). The tubes are cut to a finished length just above the mercury column, and the ends of the tubes are sealed. All of these operations are performed manually at various work stations. A temperature scale is etched onto the tube, completing the assembly (Reisdorf and D'Orlando, 1984).

During the production of thermometers, mercury emissions can be generated from mercury purification and transfer, the mercury filling process, the heating-out/burning-off steps, and accidents including spills of mercury and broken thermometers (U.S. EPA, 1993a). Within the industry, vapor emissions from mercury purification and transfer are typically controlled by containment procedures, local exhaust ventilation, temperature reduction to reduce the vapor pressure, dilution ventilation, or isolation of the operation from other work areas. The bore sizing step can be modified to reduce the use of mercury and be performed in an isolated room. Other measures that may be applied to this step are use of local exhaust ventilation, dilution ventilation and temperature control (U.S. EPA, 1993a).

Disposal of thermometers also may result in releases. There are currently no recycling efforts underway for mercury thermometers. The long life and small number of thermometers make a recycling effort impracticable. Mercury thermometers enter the waste stream by being discarded from residential and clinical settings. The thermometer is usually cracked or broken. In 1989, an estimated 16.3 tons of mercury were discarded in thermometers, or just over 2 percent of total discards of mercury (Kiser, 1991). No information was available on how much of that total was landfilled as opposed to incinerated or the emissions generated from each.

No specific data for mercury emissions from manufacturing thermometers or any other instrument containing mercury were found in the literature. One 1973 U.S. EPA report, however, presents an emission factor of 9 kg of mercury emitted for each megagram of mercury used (18 lb/ton) in overall instrument manufacture (Anderson, 1973). This emission factor should be used with caution, however, as it was based on survey responses gathered in the 1960s and not on actual test data. Instrument production and the mercury control methods used in instrument production have probably changed considerably since the time of the surveys.

In 1992, 52 Mg (57 tons) of mercury was used in all instrument production (Anderson, 1973). Multiplying the emission factor above by the 1992 usage gives a mercury emission estimate of 0.5 Mg (0.5 ton) for instrument manufacture. Again, a large degree of uncertainty is associated with this estimate because of the concerns about the reliability in the emission factor.

Trends in mercury emissions from thermometer use and production are relatively stable. Since 1984, digital thermometers have begun to replace clinical mercury thermometers in clinics, hospitals and doctors' offices. It is expected that this trend will continue. Mercury thermometers will continue to be used in residential settings because of infrequent use and the higher cost for digital thermometers. The decrease in mercury thermometer use attributable to the switch to digital thermometers in professional settings will likely be offset by an increase in mercury thermometers purchased due to increased population. The mercury content of thermometers will probably remain the same. Overall mercury entering the waste stream from thermometers will likely remain stable (U.S. EPA, 1992a).

4.2.9 Carbon Black Production

The majority of U.S. manufactured carbon black (over 98 percent) is produced using a highly aromatic petrochemical or carbochemical heavy oil feedstock containing mercury. In 1991, mercury emissions from carbon black production were estimated to be 0.23 Mg (0.25 ton). This estimate is expected to be an overestimate because it is based on production capacity and not on actual production. Table 4-19 lists the names, locations and annual capacities of U.S. producers of carbon black in 1991 (SRI International, 1992).

High-performance fabric filters are reported to be used to control PM emissions from main process streams during the manufacture of carbon black. The fabric filters can reduce PM emissions to levels as low as 6 milligrams per normal cubic meter (mg/Nm^3) (0.003 gr/dscf). Mercury emissions from the reactor are primarily in the vapor phase, and these emissions will proceed through the main process streams to the fabric filters as a vapor. If the mercury remains in the vapor phase, the mercury control efficiency of the fabric filters is expected to be low. If the product gas stream is cooled to below 170°C (325°F), the fabric filter may capture a significant fraction of the condensed mercury, thus providing some degree of emission control (Taylor, 1992).

The Carbon Black Production Process

Three primary raw materials used in the production of carbon black are preheated feedstock (either the petrochemical oil or carbochemical oil), which is preheated to a temperature between 150 and 250°C (300 and 480°F), preheated air and an auxiliary fuel such as natural gas. A turbulent, high-temperature zone is created in the reactor by combusting the auxiliary fuel, and the preheated oil feedstock is introduced in this zone as an atomized spray. In this zone of the reactor, most of the oxygen is used to burn the auxiliary fuel, resulting in insufficient oxygen to combust the oil feedstock. Thus, pyrolysis of the feedstock is achieved, and carbon black is produced. Most of the mercury present in the feedstock is emitted in the hot exhaust gas from the reactor (Taylor, 1992; Yen, 1975).

The product stream from the reactor is quenched with water, and any residual heat in the product stream is used to preheat the oil feedstock and combustion air before the carbon is recovered in a fabric filter. Carbon recovered in the fabric filter is in a fluffy form. The fluffy carbon black may be ground in a grinder, if desired. Depending on the end use, carbon black may be shipped in fluffy form or in the form of pellets. Pelletizing is done by a wet process in which carbon black is mixed with water along with a binder and fed into a pelletizer. The pellets are subsequently dried and bagged prior to shipping (Taylor, 1992; Yen, 1975).

Mercury, which is present in the oil feedstock, can be emitted during the pyrolysis step. No data are available, however, on the performance of the fabric filter control systems for mercury emissions. The only available data are for emissions from the oil-furnace process. These data show mercury emission to be 1.5×10^{-4} kg/Mg (3×10^{-4} lb/ton) from the main process vent (Serth and Hughes, 1980). The source of these data could not be obtained in order to validate the emission factors. Because the factors are not verified, they are considered to be of limited reliability.

In 1991, the total capacity for carbon black production was 1.55×10^6 Mg (1.7×10^6 tons) (SRI International, 1992). Multiplying the total capacity by the emission factor above gives a mercury emission estimate of 0.23 Mg (0.25 tons). This estimate may be greater than the actual emissions because it is based on production capacity and not on actual production. On the other hand, this estimate may understate the actual mercury emissions because the data are from the oil-furnace process only and not the main process streams.

4.2.10 Battery Production

Historically, mercury has been used in batteries for two purposes. The first use is as a component in the zinc-mercury amalgam used as the anode in mercury oxide (also known as mercury-zinc) and alkaline batteries and as a component in the cathode of mercury oxide batteries. The second use was to inhibit side reactions and corrosion of the battery casing material in carbon-zinc and alkaline batteries. Prior to the late 1980s, most primary batteries and some storage batteries contained mercury in the form of mercuric oxide (HgO), zinc amalgam (Zn-Hg), mercuric chloride (HgCl₂), or mercurous chloride (Hg₂Cl₂) (White and Jackson, 1993). As a result of technological improvements made by the battery industry, the use of mercury is being phased out of battery production. From 1989 to 1992, the use of mercury in battery production decreased 94 percent (Bureau of Mines, 1992). Because only one type of battery, mercuric oxide batteries, still used mercury to any measurable degree as of the end of 1992, it is the only battery discussed in this section. In 1992, an estimated 0.02 Mg (0.02 ton) of mercury was emitted from the production of batteries. Table 4-20 lists the

Table 4-19
1992 U.S. Carbon Black Production Facilities^a

Company	Location	Type of process ^b	Annual capacity ^c	
			10 ³ Mg	10 ³ tons
Cabot Corporation North American Rubber Black Division	Franklin, Louisiana	F	141	155
	Pampa, Texas	F	32	35
	Villa Platte, Louisiana	F	127	140
	Waverly, West Virginia	F	82	90
Chevron Corporation, Chevron Chemical Company, subsidiary, Olefins and Derivatives Division	Cedar Bayou, Texas	A	9	10
Degussa Corporation	Arkansas Pass, Texas	F	59	65
	Belpre, Ohio	F	64	70
	New Iberia, Louisiana	F	109	120
Ebonex Corporation	Melvindale, Michigan	C	4	4
General Carbon Company	Los Angeles, California	C	0.5	0.5
Hoover Color Corporation	Hiwassee, Virginia	C	0.5	0.5
J.M. Huber Corporation	Baytown, Texas	F	102	112
	Borger, Texas	F and T	79	87.5
	Orange, Texas	F	61	67.5
Phelps Dodge Corporation Colombian Chemical Company, subsidiary	El Dorado, Arkansas	F	50	55
	Moundsville, West Virginia	F	77	85
	North Bend, Louisiana	F	109	120
	Ulysses, Kansas	F	36	40
Sir Richardson Carbon & Gasoline Company	Addis, Louisiana	F	66	72.5
	Big Spring, Texas	F	52	57.5
	Borger, Texas	F	98	107
Witco Corporation Continental Carbon Company, subsidiary	Phoenix City, Alabama	F	27	30
	Ponca City, Oklahoma	F	114	125
	Sunray, Texas	F	45	50
	TOTAL		1,546	1,700

^a SRI International, 1992.

^b A = acetylene decomposition; F = furnace; C = combustion; T = thermal.

^c Capacities are variable and based on SRI estimates as of January 1, 1992.

Table 4-20
1992 U.S. Mercuric Oxide, Alkaline Manganese, or
Zinc-Carbon Button Cell Battery Manufacturers^a

Manufacturer	Production site	1990 Mercury TRI emissions kg (lb) ^b
Alexander Manufacturing Company (AMC, Inc.)	Mason City, IA	0 (0)
Duracell, USA	Cleveland, TN	NR ^c
	LaGrange, GA	NR
	Lancaster, SC	9 (20)
	Lexington, NC	3 (70)
Eagle-Picher Industries, Inc.	Colorado Springs, CO	NR
Eveready Battery Company, Inc.	Maryville, MO	14 (30)
	Red Oak, IA	NR
	Fremont, OH	NR
	Bennington, VT	1 (2)
	Asheboro, NC (2 plants)	2 (5)
Mutec ^d	Columbus, GA (Corporate offices)	NR
Rayovac Corp.	Madison, WI	0 (0)
	Fennimore, WI	5 (10)
	Portage, WI	NR

^a U.S. EPA, 1993a.

^b U.S. EPA, 1992e.

^c NR = Not reported, company did not report mercury emissions in 1990 TRI.

^d Mutec is a joint venture between Eastman Kodak and Panasonic.

manufacturers of mercuric oxide, alkaline manganese and zinc-carbon batteries and the associated emissions reported in the 1990 TRI (U.S. EPA, 1992e). The TRI does not distinguish the type of battery each facility produces.

Mercuric oxide batteries fall into two categories: button cells and larger sizes. Most mercuric oxide batteries sold for personal use are button cells. Button cells are small, circular, relatively flat batteries that are used in transistorized equipment, walkie-talkie's, hearing aids, electronic watches, and other items requiring small batteries. Mercuric oxide batteries are widely used for applications that require reliability and a constant rate of discharge, including medical and military applications. Larger mercuric oxide batteries, which often resemble 9-volt or fat AA batteries in size or shape, are produced for a variety of medical, industrial, military, and other non-household devices (Dierlich, 1994). The mercury content in mercuric oxide batteries is typically 33 percent to 50 percent mercury by weight and cannot be reduced without proportionally reducing the energy content of these batteries. Acceptable alternative batteries are available for almost all applications of household mercuric oxide batteries (Cole et al., 1992; Balfour, 1992).

Mercuric oxide-zinc cells use mercuric oxide (mixed with graphite and manganese dioxide) as the cathode and a zinc amalgam at the anode. In producing the cathodes, granulated mercuric oxide, manganese dioxide, and granulated graphite are manually metered through a hopper to the blending area (U.S. EPA, 1984). This mixture is then pelletized in a rotary press. The pellets are consolidated into plastic trays and are then sent to the production lines for cell assembly. For the production of the anodes, elemental mercury and zinc powder are blended along with electrolyte and a binder to produce an anode gel (Rauh, 1991). The completed anodes and cathodes are then sent to the cell manufacturing area. Separators, electrolytes and other components are assembled with the anode and cathode to produce the HgO-Zn cell. Assembly may be automatic or semiautomatic. The assembled cathode, anode, electrolyte, and cover are sealed with a crimper.

During the manufacture of mercuric oxide batteries, mercury may be emitted from grinding, mixing, sieving, pelletizing, and/or consolidating operations as PM and as vapor emissions. Baghouses are used to control PM emissions from the mixing/blending and processing steps in the production of cathodes. Mercury vapor emissions from the anode processing and cell manufacturing areas are generally discharged to the atmosphere uncontrolled. Ventilation air in the assembly room is recirculated through PM filters. One plant reported an average of 73 percent mercury vapor removal efficiency in the cell assembly room when an air handler system, consisting of a PM prefilter and a charcoal filter, was operated using 75 percent recirculating air and 25 percent fresh air (Reisdorf and D'Orlando, 1984).

The only reported emission factor for a mercuric oxide production facility was for one plant in Wisconsin (Bureau of Air Management, 1986). This facility used a combination of a baghouse and charcoal filter to treat the exhaust ventilation air. Annual use of mercury was 36.07 Mg (39.8 tons), and annual emissions were reported as 36.3 kg (80 lb) of mercury as HgO particles. The mercury emission factor for battery manufacture based on these data is 1.0 kg/Mg (2.0 lb/ton) of mercury used.

Several factors limit the reliability of this emission factor. First, the facility no longer produces mercuric oxide batteries. The processes and emission controls may be substantially different for existing mercuric oxide facilities, although no information on different process or controls was provided to U.S. EPA from one current manufacturer. Second, no information is presented on the bases of the emission factor, but the mercury emission quantity is presumed to be an engineering estimate by the manufacturer because no reference is made to any emissions testing performed at the facility. Finally, this factor is based on only one specific site, and that facility may not represent all mercuric oxide battery manufacturing facilities.

Emission source data from a study of an integrated mercury button cell plant are summarized in Table 4-21 (U.S. EPA, 1984). Major emission points were the pelletizing and consolidating operations (up to 42.46 g/d [0.094 lb/d]) and cell assembly (28.58 g/d [0.063 lb/d]). Emission controls were not in place for mercury vapor emissions from the main plant (U.S. EPA, 1984). This plant reported total mercury emissions of 3.2 kg (7 lb) in the 1990 TRI (U.S. EPA, 1992e).

In 1992, 16 Mg (18 tons) of mercury were used in the production of batteries in the United States (Bureau of Mines, 1992). Multiplying the mercury usage by the emission factor developed for the facility in Wisconsin gives a mercury emission estimate of 0.02 Mg (0.02 tons) for 1992. This estimate is highly uncertain, however, because of the concerns discussed above about the reliability of the emission factors (U.S. EPA, 1993a). Mercury emissions to the atmosphere when batteries are disposed are accounted for in the emission estimate for MWCs and MWIs, as discussed in Section 4.1 of this report.

Table 4-21
Emission Source Parameters for an Integrated
Mercury Button Cell Manufacturing Facility^a

Building/source description ^b	Emission rate ^c		Exit temp. (K); control device
	g/d	lb/d	
Main plant			
Control room			
1. Blending, slugging, compacting, granulating	6.12	0.0135	297; Baghouse
2. Slugging, granulating	1.22	0.0027	297; Baghouse
3. Pelleting, consolidating	1.63 ^d	0.0036 ^d	295; Baghouse
4. Pelleting, consolidating	42.46	0.0936	297; Baghouse
4a. Pelleting, consolidating	6.53	0.0144	297; Baghouse
5. Blending, compacting, granulating, pelleting, consolidating	1.36 ^d	0.003 ^b	297; Baghouse
Anode room			
6. Amalgam, dewatering	1.82 ^d	0.004 ^d	297; Uncontrolled
6a. Vacuum dryer	0.46 ^d	0.001 ^d	297; Uncontrolled
6b. Blending	0.91 ^d	0.002 ^d	297; Uncontrolled
7. Pelleting, zinc amalgam	4.08 ^d	0.009 ^d	295; Baghouse
Cell assembly area			
8. Assembling calls	28.58	0.0630	295; Baghouse for PM. Vapor by recirculating air through prefilters and charcoal filters

^a U.S. EPA, 1984.

^b Source names are those used by facility.

^c Emission rates were measured by facility except where noted.

^d Estimated emission rate by facility.

4.2.11 Primary Mercury Production

Mercury is currently only produced in the United States as a byproduct from the mining of gold ores and is no longer produced from mercury ore. The last U.S. mercury ore mine, the McDermitt Mine in McDermitt, Nevada, ceased operation in 1990, and all its equipment has since been dismantled, sold, landfilled, or scrapped (U.S. EPA, 1993a).

Since the closure of the McDermitt Mine, recovery of mercury as a byproduct from gold ores is the only remaining ore-based production process. In 1991, nine U.S. gold mines (seven in Nevada, one in California and one in Utah) produced metallic mercury as a byproduct. Mines that do produce mercury represent only a small percentage of all domestic gold mines. The names and locations of these mines are shown in Table 4-22. No information was available on the amount of mercury recovered at each facility, although the Bureau of Mines reported that 64 Mg (70 tons) of mercury was produced as a byproduct of gold ore mining in 1992 (Bureau of Mines, 1994). Data are insufficient at this time to estimate the quantity of mercury emissions generated as a byproduct of gold ore mining.

Table 4-22
1992 U.S. Byproduct Mercury-Producing Gold Mines^a

Mine	County/State	Operator
Alligator Ridge	White Pines, NV	USMX, Inc.
Getchell	Humboldt, NV	FirstMiss Gold
Carlin Mines Complex	Eureka, NV	Newmont Gold Co.
Hog Ranch	Washoe, NV	Western Hog Ranch Co.
Jerritt Canyon (Enfield Bell)	Elko, NV	Independence Mining Co., Inc.
McLaughlin ^c	Napa, CA	Homestake Mining Co.
Mercur	Tooele, UT	Barrick Resources (USA) Inc.
Paradise Peak	Nye, NV	Permanently closed in September 1993
Pinson and Kramer Hill	Humboldt, NV	Pinson Mining Co.

^a Bureau of Mines, 1994.

Potential sources of mercury emissions from gold processing facilities are at locations where furnaces, retorts, or other high-temperature sources are used in the process and where the mercury is removed from the launders. The treated gas discharged to the atmosphere is also a source of mercury emissions (U.S. EPA, 1993a).

No emission data have been published for facilities producing mercury as a byproduct of gold ore; therefore, no estimate of mercury emissions from gold ore mining can be made at this time.

Primary Mercury Production Processes

This description of production processes and emission controls used at gold mines does not necessarily reflect any specific gold mine but summarizes the types of processes and controls a gold mine could use to produce mercury and control mercury emissions. These processes vary from site to site.

The incoming gold ore is crushed using a series of jaw crushers, cone crushers and ball mills. If the incoming ore is an oxide-based ore, no pretreatment is required and the crushed ore is mixed with water and sent to the classifier. If the ore is a sulfide-based ore, it must be pretreated using either a fluid bed or multiple hearth pretreatment furnace (roaster) to convert metallic sulfides to metallic oxides. The exhaust gas from either of these units is sent through wet ESPs and, if necessary, through carbon condensers. The exhaust gas then passes through a lime sulfur dioxide (SO_2) scrubber prior to discharging to the atmosphere. If the treated sulfide ore is high in mercury content, the primary mercury recovery process occurs from the wet ESPs. If the concentration is low, no attempt is made to recover the mercury for sale. The pretreated ore is mixed with water and sent to the classifier, where the ore is separated (classified) according to size. Ore pieces too large to continue in the process are returned to the crusher operation (U.S. EPA, 1993a).

From the classifier, the slurry passes through a concentrator and then to a series of agitators containing the cyanide leach solution. From the agitators, the slurry is filtered, the filter cake sent to disposal and the filtrate containing the gold and mercury is transferred to the electrowinning process. If the carbon-in-pulp (CIP) process is used, the cyanide pulp in the agitators is treated with activated carbon to adsorb the gold and mercury. The carbon is filtered from the agitator tanks and treated with an alkaline cyanide-alcohol solution to desorb the metals. This liquid is then transferred to the electrowinning tanks. In the electrowinning process, the gold and mercury are electrodeposited onto a stainless steel wool cathode, which is sent to a retort to remove mercury and other volatile impurities. The stainless steel wool, containing the gold, is transferred from the retort to a separate smelting furnace, where the gold is volatilized and recovered as crude bullion (U.S. EPA, 1993a).

The exhaust gas from the retort, containing mercury, SO_2 , PM, water vapor, and other volatile components, passes through condenser tubes, where the mercury condenses as a liquid and is collected under water in the launders. From the launders, the mercury is purified and sent to storage. After passing through the condenser tubes, the exhaust gas goes through a venturi and impinger tower to remove PM and water droplets and then moves through the SO_2 scrubber prior to discharging to the atmosphere (U.S. EPA, 1993a).

When pretreatment roasting is required, the exhaust gases from the furnace pass through a cyclone to remove PM and then move through wet ESPs to remove arsenic, mercury and some of the SO_2 . If the mercury concentration in the gold ore is high, the ESPs will not remove all of the mercury, and an activated carbon adsorber bed may be required for additional mercury removal. The gas passes through a lime scrubber to remove SO_2 ; if the SO_2 concentration is low, a caustic scrubber may be used. From the scrubber, the gas is discharged through the stack to the atmosphere. Essentially, the same emission control measures are used for the exhaust gas from the retort. After the gas passes through the condenser tubes to remove the mercury, a venturi and a cyclone are used to remove PM and water droplets. These controls are followed by the lime scrubber to remove the SO_2 prior to discharging to the atmosphere.

Gold ores in open heaps and dumps can also be treated by cyanide leaching. In this process, the gold ore is placed on a leaching pad and sprayed with the cyanide solution. The solution migrates down through the ore to a collection system on the pad and then is sent to a pregnant solution pond. From this pond, the leachate liquors, containing gold and mercury, are transferred to the gold recovery area. In this area, the liquor is filtered and sent to the electrowinning process (U.S. EPA, 1993a).

According to an industry representative, all gold mines that produce mercury control their emissions because the objective is to recover as much mercury as possible (Barringer and Johnson, 1995).

No specific data on emission factors from potential sources of mercury emissions from mercury ore mining have been published since 1973 (U.S. EPA, 1993a). The 1973 report gives a total emission factor of 0.171 kg of mercury emitted for each megagram of mercury ore mined (0.342

lb/ton), which was based on stack tests conducted in the early 1970s (Anderson, 1973). However, this emission factor is for mercury emissions from mercury ore mining only and cannot be used for mercury emissions from gold ore mining. No mercury emissions from gold ore mining were, therefore, estimated for this report.

4.2.12 Mercury Compounds Production

The production of mercury compounds presents a potential source of mercury emissions into the atmosphere. Common mercury compounds include mercuric chloride, mercuric oxide, and phenylmercuric acetate (PMA). Table 4-23 presents a list of several producers of inorganic and organic mercury compounds.

Because numerous mercury compounds are produced in the United States, it is beyond the scope of this study to present process descriptions for each one. Process descriptions of the more common mercury compounds can be found in the mercury L&E document (U.S. EPA, 1993a).

During the production of mercury compounds, emissions of mercury vapor and particulate mercury compounds may occur at the following sources: reactors, driers, filters, grinders, and transfer operations. No information was found on specific emission control devices to remove or treat the mercury emissions, but the literature did contain information on methods designed to reduce the workplace concentrations without subsequent treatment (Reisdorf and D'Orlando, 1984). Typically, these procedures included some combination of enclosure or containment, process modifications, exhaust ventilation, dilution ventilation, and personal protective equipment (Reisdorf and D'Orlando, 1984). In some cases, ventilation systems are reported to be ducted to cyclone dust collectors to reduce dust emissions, but no information was located on mercury vapor controls (U.S. EPA, 1993a). No information was available from the literature on mercury emissions or emission factors from the production of mercury compounds; therefore, no mercury emission estimate could be developed. As shown in Table 4-23, the only company that reported significant emissions (227 kg [500 lb]) in the 1991 TRI was Mallinkrodt, Inc.

4.2.13 Byproduct Coke Production

Byproduct coke, also called metallurgical coke, is a primary feedstock for the integrated iron and steel industry. Because no information concerning mercury emissions from the production of byproduct coke could be found in the literature, no nationwide mercury emission estimates were generated. Table 4-24 lists U.S. byproduct coke oven facilities in 1991 (Huskanen, 1991) and Figure 4-13 shows the locations of these facilities.

Coke is currently produced in two types of coke oven batteries: the slot oven byproduct battery and the nonrecovery battery. The slot oven byproduct type is by far the most commonly used battery; over 99 percent of coke produced in 1990 was produced in this type of battery (Easterly et al.; U.S. EPA, 1988).

The byproduct coke oven battery consists of a series (ranging from 10 to 100) of narrow ovens, 0.4 to 0.6 m (1.3 to 2 ft) wide, and 12 to 18 m (40 to 60 ft) long. The height of the ovens may range between 3 and 6 m (10 and 20 ft). Depending on the dimensions, the production capacity may range between 6.8 and 35 Mg (7.5 and 39 tons) of coke per batch. A heating flue is located between each oven pair (Easterly et al.; U.S. EPA, 1988).

Table 4-23
1991 U.S. Mercury Compound Producers^a

Producer	Location	1991 TRI emissions, kg (lb) ^b	Compound(s)
Atochem North America, Inc., Chemical Specialties Division	Tulsa, OK	NR ^c	HgF ₂
Atomergic Chemetals Corp.	Farmingdale, NY	NR	Thimerosal (Merthiolate)
Cambrex Corp., CasChem, Inc., Subsidiary (formerly Cosan Chem. Corp.)	Carlstadt, NJ	18 (40)	Phenylmercuric acetate (PMA), Phenylmercuric oleate
W.A. Cleary Corp.	Somerset, NJ	NR	Phenylmercuric acetate (PMA)
Deepwater, Inc.	Carson, CA	NR	HgI ₂
GFS Chemicals, Inc.	Columbus, OH	NR	HgBr ₂ , HgI ₂ , Hg(NO ₃) ₂ , HgSO ₄
Hüls America, Inc.	Elizabeth, NJ	0 (0)	Phenylmercuric acetate (PMA)
Imsera Group, Inc., Mallinkrodt Inc., Subsidiary, Mallinkrodt Specialty Chem. Co.,	Erie, PA	227 (500)	HgCl ₂ on carbon support (catalyst for vinyl chloride manufacture)
Morton International, Inc., Specialty Chemicals Group, Advanced Materials, CVD Inc. Subsidiary	Woburn, MA	NR	Highly purified dimethylmercury, (CH ₃) ₂ Hg, for chemical vapor deposition (CVD) of thin films
Polychemical Laboratories, Inc.	Melville, NY	NR	Thimerosal (Merthiolate)
R.S.A Corporation	Ardsley, NY	NR	Hg(SCN) ₂
Troy Chemical Corp.	Newark, NJ	0 (0)	Phenylmercuric acetate (PMA)

^a SRI International, 1991.

^b U.S. EPA, 1992e.

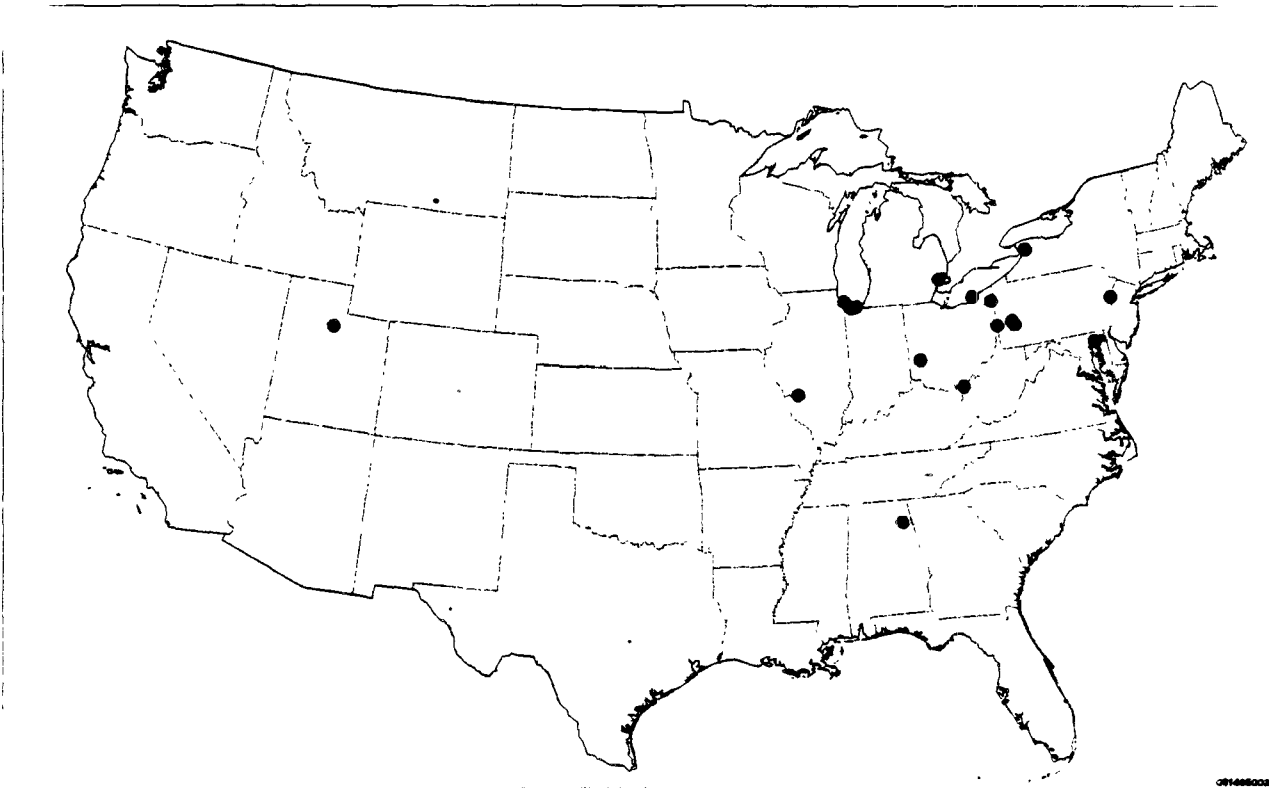
^c NR = Not reported; company did not appear in 1991 TRI.

Table 4-24
1991 U.S. Byproduct Coke Producers^a

Facility	No. of batteries	Total No. of ovens	Total capacity, Mg/d (ton/d)
Acme Steel, Chicago, IL	2	100	1,450 (1,600)
Armco, Inc., Ashland, KY	2	146	2,450 (2,700)
Armco, Inc., Middletown, OH	3	203	4,130 (4,540)
Bethlehem Steel, Bethlehem, PA	3	284	3,580 (3,940)
Bethlehem Steel, Burns Harbor, IN	2	164	3,980 (4,380)
Bethlehem Steel, Lackawanna, NY	2	152	1,700 (1,870)
Bethlehem Steel, Sparrows Point, MD	3	210	3,700 (4,070)
Geneva Steel, Orem, UT	1	208	2,050 (2,250)
Gulf States Steel, Gadsden, AL	2	130	2,550 (2,800)
Inland Steel, East Chicago, IN	6	446	5,250 (5,780)
LTV Steel, Pittsburgh, PA	5	315	4,910 (5,400)
LTV Steel, Chicago, IL	1	60	1,450 (1,600)
LTV Steel, Cleveland, OH	2	126	2,910 (3,200)
LTV Steel, Warren, OH	1	85	1,360 (1,500)
National Steel, Granite City, IL	2	90	1,380 (1,520)
National Steel, Ecorse, MI	1	78	840 (925)
USS, Div. of USX Corp., Clairton, PA	12	816	11,490 (12,640)
USS, Div. of USX Corp., Gary, IN	6	422	6,490 (7,140)
Wheeling-Pittsburgh Steel, East Steubenville, WV	4	224	3,450 (3,800)
Total	58	4,259	65,120 (71,660)

^a Huskanen, 1991.

Figure 4-13
Coke Oven Locations



Pulverized coal, which is the feedstock, is fed through ports located on the top of each oven by a car that travels on tracks along the top of each battery. The ports are sealed upon charging, and gaseous fuel is combusted in the flues located between the ovens to provide the energy for the pyrolysis. The coking process takes between 12 and 20 hours, at the end of which almost all the volatile matter from the coal is driven off, thus forming coke. The coke is then unloaded from the ovens through vertical doors on each end of the oven into a rail car, where it is cooled by being sprayed with several thousand gallons of water. The rail car then unloads the coke in a separate area, where the coke is allowed to cool further (Easterly et al.; U.S. EPA, 1988).

Mercury is present in coal in appreciable quantities. Consequently, the volatile gases that evolve from the coking operation are likely to contain mercury (Easterly et al.; U.S. EPA, 1988).

Emissions at byproduct coke plants are generated during coal preparation, oven charging operations and other operations. Emissions are also generated from door leaks and from the battery stack. The battery stack emissions are primarily a result of leakage from the oven into the flue. Mercury emissions can be generated in small quantities during coal preparation and handling as fugitive PM because mercury is present as a trace contaminant in coal. Mercury also may be volatilized and released during charging and pushing operations as well as from the battery stacks and door and topside leaks.

There are no mercury data for coke ovens in the U.S., so an estimate of U.S. mercury emissions from this source category is not included in this report. There are European emission factors available however, so a rough estimate can be calculated if only to give a sense of the potential

magnitude of this source category's emissions. Emission factors used in Germany for coke production range from 0.01 to 0.03g mercury per Mg of coke produced (Jockel and Hartje, 1991). One difference between European coke producers and U.S. coke producers is that U.S. coke producers use a very high quality cleaned coal while their European counterparts do not. If it is assumed that an emission factor of about 0.025 g mercury per Mg of coke produced is relevant for the U.S. (assuming a 20 percent reduction of mercury by the coal cleaning process), then potential mercury emissions for this source category would be 0.6 Mg/year (0.7 tons/year).

4.2.14 Petroleum Refining

Petroleum refining involves converting crude petroleum oil into refined products, including liquified petroleum gas, gasoline, kerosene, aviation fuel, diesel fuel, fuel oils, lubricating oils, and feedstocks for the petroleum industry. Mercury is reported to be present in petroleum crude, with its content ranging from 0.023 to 30 ppmwt (U.S. EPA, 1990).

As of January 1992, there were 32 oil companies in the United States with operable atmospheric crude oil distillation capacities in excess of 100,000 barrels per calendar day. These oil companies operated refineries at a total of 110 different locations. In addition, there are 72 companies with distillation capacities of less than 100,000 barrels per calendar day (National Petroleum Refiners Association, 1992).

The operations at refineries are classified into five general categories: separation processes, petroleum conversion processes, petroleum treating processes, feedstock and product handling, and auxiliary facilities. In the separation process, crude oil is separated into its constituents (including paraffinic, naphthionic and aromatic hydrocarbon compounds) by either atmospheric distillation, vacuum distillation, or gas processing (recovery of light ends). Conversion processes include cracking, coking and visbreaking, which breaks large molecules into smaller molecules; isomerization and reforming processes to rearrange the structures of molecules; and polymerization and alkylation to combine small molecules into larger ones (U.S. EPA, 1993a).

Petroleum treatment processes include hydrodesulfurization, hydrotreating, chemical sweetening, acid gas removal, and deasphalting. These treatment methods are used to stabilize and upgrade petroleum products. Feedstock and product handling includes storage, blending, loading, and unloading of petroleum crude and petroleum products. Auxiliary facilities include boilers, gas turbines, wastewater treatment facilities, hydrogen plants, cooling towers, and sulfur recovery units (U.S. EPA, 1993a).

Control of VOC emissions from distillation, catalytic cracking, coking, blowdown system, sweetening, and asphalt blowing is achieved by flares. In some cases, the VOC-laden gas stream is also used as fuel in process heaters. Cyclones in conjunction with ESPs are used to reduce emissions from catalytic cracking (U.S. EPA, 1993a). These control measures are expected to have little effect on mercury emissions.

The primary source of mercury emissions in petroleum refining is the separation process, although mercury emissions can also be expected in the petroleum conversion and petroleum treating processes (U.S. EPA, 1993a). Data were unavailable, however, to calculate an emission factor. As a result, no estimate of mercury emissions could be made for this source category. More analyses of oils and refinery emissions are needed to evaluate this source.

4.3 Miscellaneous Sources

Sources not readily classified as combustion or manufacturing sources of mercury or that once emitted mercury but currently do not are considered miscellaneous sources. These sources account for an estimated 1.3 Mg/yr (1.4 tons/yr) of mercury emissions generated in the United States. They include geothermal power plants, pigments, oil shale retorting, mercury catalysts and explosives. Table 4-25 presents mercury emissions from these miscellaneous sources.

4.3.1 Geothermal Power Plants

Geothermal power plants are either dry-steam or water-dominated and emitted an estimated 1.3 Mg (1.4 tons) of mercury in 1992. For dry-steam plants, steam is pumped from geothermal reservoirs to turbines at a temperature of about 180°C (360°F) and a pressure of 7.9 bars absolute (U.S. EPA, 1993a). For water-dominated plants, water exists in the producing strata at a temperature of approximately 270°C (520°F) and at a pressure slightly higher than hydrostatic (U.S. EPA, 1993a). As the water flows towards the surface, pressure decreases and steam is formed, which is used to operate the turbines. As of 1992, there were 18 geothermal power plants operating in the United States (Marshall, 1993). Table 4-26 lists the names, locations and capacities of these facilities.

No data on the mercury content of steam or water cycled through geothermal facilities are available. Likewise, no information exists on emission control systems for geothermal power plants (U.S. EPA, 1993a).

Mercury emissions at geothermal power plants are documented to result from two sources: off-gas ejectors and cooling towers. Table 4-27 contains the mercury emission factors for these two sources, which are based on measurements taken in 1977 (Robertson et al., 1977). No process data are given in the documentation containing the test results, and the primary draft source of these data could not be obtained in order to verify the validity of the emission factors (U.S. EPA, 1993a). If significant process modifications or changes in control strategies have been incorporated since 1977, the emission factors reported in Table 4-27 may no longer be valid.

Multiplying the emission factors in Table 4-27 by the total capacity shown in Table 4-26 (assuming that geothermal power plants operate 24 hr/d, 365 d/yr) gives a mercury emission estimate of 1.3 Mg (1.4 tons) for geothermal power plants in 1992. Because the emission factors used to generate this estimate have limited reliability, this emission estimate has a high degree of uncertainty.

4.3.2 Pigments, Oil Shale Retorting, Mercury Catalysts, Turf Products and Explosives

Pigments, oil shale retorting, mercury catalysts, turf products and explosives were once sources of mercury emissions but no longer. Domestic production of mercury-containing pigments ceased in 1988 (U.S. EPA, 1992a). There are currently no oil shale retorts in the United States (U.S. EPA, 1981). As of 1994, there are no active registrations of mercury-containing turf products in the United States. All registrations have been cancelled or are in the process of cancellation following voluntary cancellation by the registrants. No emissions of mercury from production mercury catalysts could be accounted for (U.S. EPA, 1993a). Commercial mercury use in explosives ceased prior to 1970 (U.S. EPA, 1992a).

Table 4-25
Best Point Estimates of Mercury Emissions from Miscellaneous Anthropogenic Emission Sources: 1990-1993

Source	Emissions			Date of Data ^a	Degree of Uncertainty ^b	Basis for Emissions Estimate
	Mg/yr	Tons/yr	% of Total			
Geothermal power plants	1.3	1.4	0.6	1977/1992	High	Test data
Turf products	-	-	-	-	-	No active registrations in the U.S. of mercury-containing turf products
Pigment production	-	-	-	-	-	No sources in U.S.
Oil shale retorting	-	-	-	-	-	No sources in U.S.
Mercury catalysts	-	-	-	-	-	Insufficient information to estimate emissions
Explosives manufacturing	-	-	-	-	-	No sources in U.S.
Total	1.3	1.4	0.6			

^a Date that data emission factor is based on/Date of activity factor used to estimate emissions.

^b A "medium" degree of uncertainty means the emission estimate is believed to be accurate within ± 25 percent. A "high" degree of uncertainty means the emission estimate is believed to be accurate within ± 50 percent.

Table 4-26
1992 U.S. Geothermal Power Plants^a

Facility	Type	Net Capacity (MW)
The Geysers, CA	Dry-steam	1,805.7
Salton Sea, CA	Water-dominated	218.3
Heber, CA	Water-dominated	47.0
East Mesa, CA	Water-dominated	106.0
Coso, CA	Water-dominated	247.5
Casa Diablo, CA	Water-dominated	34.0
Amedee, CA	Water-dominated	2.0
Wendel, CA	Water-dominated	0.7
Dixie Valley, NV	Water-dominated	57.0
Steamboat Hot Springs, NV	Water-dominated	19.3
Beowawe Hot Springs, NV	Water-dominated	16.7
Desert Peak, NV	Water-dominated	9.0
Wabuska Hot Springs, NV	Water-dominated	1.7
Soda Lake, NV	Water-dominated	15.7
Stillwater, NV	Water-dominated	12.5
Empire and San Emidio, NV	Water-dominated	3.2
Roosevelt Hot Springs, UT	Water-dominated	20.0
Cove Fort, UT	Water-dominated	12.1
Total		2,628.4

^a Marshal, 1993.

Table 4-27
Mercury Emission Factors for Geothermal Power Plants^a

Source	Emission factor range	Average emission factor	
	g/MWe/hr	g/MWe/hr	lb/MWe/hr
Off-gas ejectors	0.00075 - 0.02	0.00725	0.00002
Cooling tower exhaust	0.026 - 0.072	0.05	0.0001

^a Robertson et al., 1977.

5. EMISSIONS SUMMARY

Table 5-1 summarizes the estimated national mercury emission rates by source category. These emissions estimates should be regarded as best estimates given available data.

The emissions data presented in this document served three primary purposes. First, the inventory identifies source categories that emit a significant amount of mercury. This information will be useful for decision makers when selecting potential candidates for mercury emissions reductions and in evaluating possible control technologies or pollution prevention measures that could be used to achieve emission reductions. Second, the inventory was used to identify source types with the potential to have public health or environmental impacts when evaluated as singular point sources. The source types so identified were modeled in the local impact analysis to assess the potential public health and environmental impacts from a single source. Third, the emissions data summarized in this document served as input to U.S. EPA's long-range transport model which assessed the nationwide dispersion and deposition of mercury from all of the identified mercury sources in the U.S. The local impact analysis and long-range transport modeling are described in detail in Volume III of the Mercury Study Report to Congress -- An Assessment of Exposure From Anthropogenic Mercury Emissions in the United States.

Accuracy of the Inventory

The accuracy of the emission estimates is obviously a factor in assessing the inventory's usefulness for its intended purposes. Considering the admitted gaps in the inventory, the peer review panel that reviewed this work concluded that the missing sources could contribute as much as 20 percent more mercury emissions to the U.S. total. For comparison, one reviewer submitted data on the amount of mercury emitted per person in some European countries (based on anthropogenic emissions only).

Based on the inventory presented in this document, the U.S. inventory represents 0.86 g mercury per person per year. Based on data submitted during the peer review process, 0.90 g mercury per person per year is emitted in the United Kingdom. In Germany (Western area), 0.75 g mercury per person per year is emitted. In Poland, 0.88 g mercury per person per year is estimated to be emitted. The European emission average is about 1.2 g mercury per person per year (Pacyna, 1995). The similarity between the U.S. inventory and other countries where coal is also the major source of energy lends credibility to the nationwide estimate presented in this report for the U.S.

Use of the Inventory for the Local Impact and Control Technology Analyses

While the emission estimates have limitations, they do provide insight into the relative magnitude of emissions from different groups of sources. Table 5-2 shows the distribution of estimated emissions among the four major classes of sources of anthropogenic emissions (area sources, combustion point sources, manufacturing point sources, and miscellaneous point sources).

Of the estimated 220 Mg (243 tons) of annual mercury emission into the atmosphere by anthropogenic sources: combustion point sources currently account for 85 percent, manufacturing point sources contribute 13 percent, area sources contribute 1 percent, and miscellaneous point sources contribute 1 percent. Four specific source categories account for approximately 83 percent of the total anthropogenic emissions -- medical waste incineration (27 percent),

Table 5-1
Best Point Estimates of National Mercury Emission Rates by Category

Source of mercury ^a	1990-1993 Mg/yr ^{b,c}	1990-1993 tons/yr ^{b,c}	% of Total Inventory
Area sources	2.8	3.1	1.3
Lamp breakage	1.4	1.5	0.6
General lab use	0.7	0.8	0.3
Dental prep and use	0.7	0.8	0.3
Mobile sources	d	d	d
Paint use	e	e	e
Agricultural burning	d	d	d
Landfills	d	d	d
Point sources	217.3	239.4	98.7
Combustion sources	186.9	205.9	84.9
MWIs ^f	58.8	64.7	26.7
MWCs	50	55	22.7
Utility boilers	46.5	51.3	21.2
Coal	(46.3) ^g	(51)	(21.0)
Oil	(0.23)	(0.25)	(0.1)
Natural gas	(0.002)	(0.002)	(0.0)
Commercial/industrial boilers	26.3	29	12.0
Coal	(20.7)	(22.8)	(9.4)
Oil	(5.5)	(6.0)	(2.5)
Residential boilers	3.2	3.5	1.4
Coal	(0.5)	(0.6)	0.2
Oil	(2.7)	(3.0)	(1.2)
SSIs	1.7	1.8	0.7
Crematories	0.4	0.4	0.2
Wood-fired boilers ^h	0.3	0.3	0.1
Hazardous waste combustors ⁱ	d	d	d
Manufacturing sources	29.1	32	13.2
Primary lead	8.2	9.0	3.7
Secondary Hg production	6.7	7.4	3.1
Chlor-alkali	5.9	6.5	2.7
Portland cement	5.9	6.5	2.7
Primary copper ^j	0.6	0.7	0.3
Lime manufacturing	0.6	0.7	0.3
Electrical apparatus	0.42	0.46	0.2
Instruments	0.5	0.5	0.2
Carbon black	0.23	0.25	0.1
Fluorescent lamp recycling	0.005	0.006	0.002
Batteries	0.02	0.02	0.0
Primary Hg production	d	d	d
Mercury compounds	d	d	d
Byproduct coke	d	d	d
Refineries	d	d	d
Miscellaneous sources	1.3	1.4	0.6
Geothermal power	1.3	1.4	0.6
Turf products	e	e	e
Pigments, oil, etc.	e	e	e
TOTAL	220.1	242.5	100.0

^a MWC = Municipal waste combustor; MWI = medical waste incinerator; SSI = sewage sludge incinerator.

^b Numbers do not add exactly because of rounding.

^c Where available, emissions estimates for 1995 are discussed in the text. However, these 1995 estimates were not used in any of the modeling analyses.

^d Insufficient information to estimate 1990 emissions.

^e Mercury has been phased out of use.

^f In the course of an MWI rulemaking, with the receipt of new data, U.S. EPA expects to revise the mercury emission estimate for MWIs downward.

^g Parentheses denote subtotal within a larger point source category.

^h Includes boilers only; does not include residential wood combustion (wood stoves).

ⁱ In 1995 incinerators and lightweight aggregate kilns (not cement kilns) were estimated to emit 5.0 tons of mercury.

^j 1990 emissions are estimated for only one source, which ceased operations in February 1995. The nationwide estimate for 1995 is 0.08 tons.

Table 5-2
Best Point Estimates of Mercury Emissions from Anthropogenic Sources: 1990-1993

Source	Emissions		
	Mg/yr	Tons/yr	% of Total Inventory
Area	2.8	3.1	1.3
Combustion	186.9	205.9	84.9
Manufacturing	29.1	32.0	13.2
Miscellaneous	1.3	1.4	0.6
Total Inventory	220.1	242.5	100

municipal waste combustion (23 percent), utility boilers (21 percent), and commercial/industrial boilers (12 percent).

Based on this information, six source categories were selected for the local impact analyses in Volume III of this report and the control technology assessment described in Volume VII of this report. The source categories were selected based on the magnitude of their mercury emissions either in the aggregate as a source category or as single point sources. The source categories were coal- and oil-fired utility boilers, municipal waste combustors, medical waste incinerators, chlor-alkali plants, primary copper smelters and primary lead smelters. Model plants representing these categories were developed for both the local impact analyses and the control technology assessment. The model plants for the local impact analyses are described in detail in Appendix F to Volume III of this report and for the control technology assessment, in Appendix B of Volume VII of this report.

Use of the Inventory for the Long-range Transport Analysis

For the long-range transport analysis, the emissions inventory was mapped for the continental U.S. The continental U.S. was divided into 40-km square grid cells and the magnitude of the mercury emissions were calculated for each cell. For the most part, the location (at least to the city level) of the mercury point sources described in this document were known.

For area sources where the sources are small, diffuse and numerous, exact locations were not known. There were a number of source categories where this was the case. The emissions for these source categories were allocated or apportioned to each county in the U.S. based on a variety of information. The area sources and the method used to allocate their emissions are shown in Table 5-3.

Figure 5-1 illustrates the spatial distribution of mercury emissions across the U.S. based on this inventory. This distribution formed the basis of the long-range transport modeling and the resulting predictions of wet and dry deposition across the U.S.

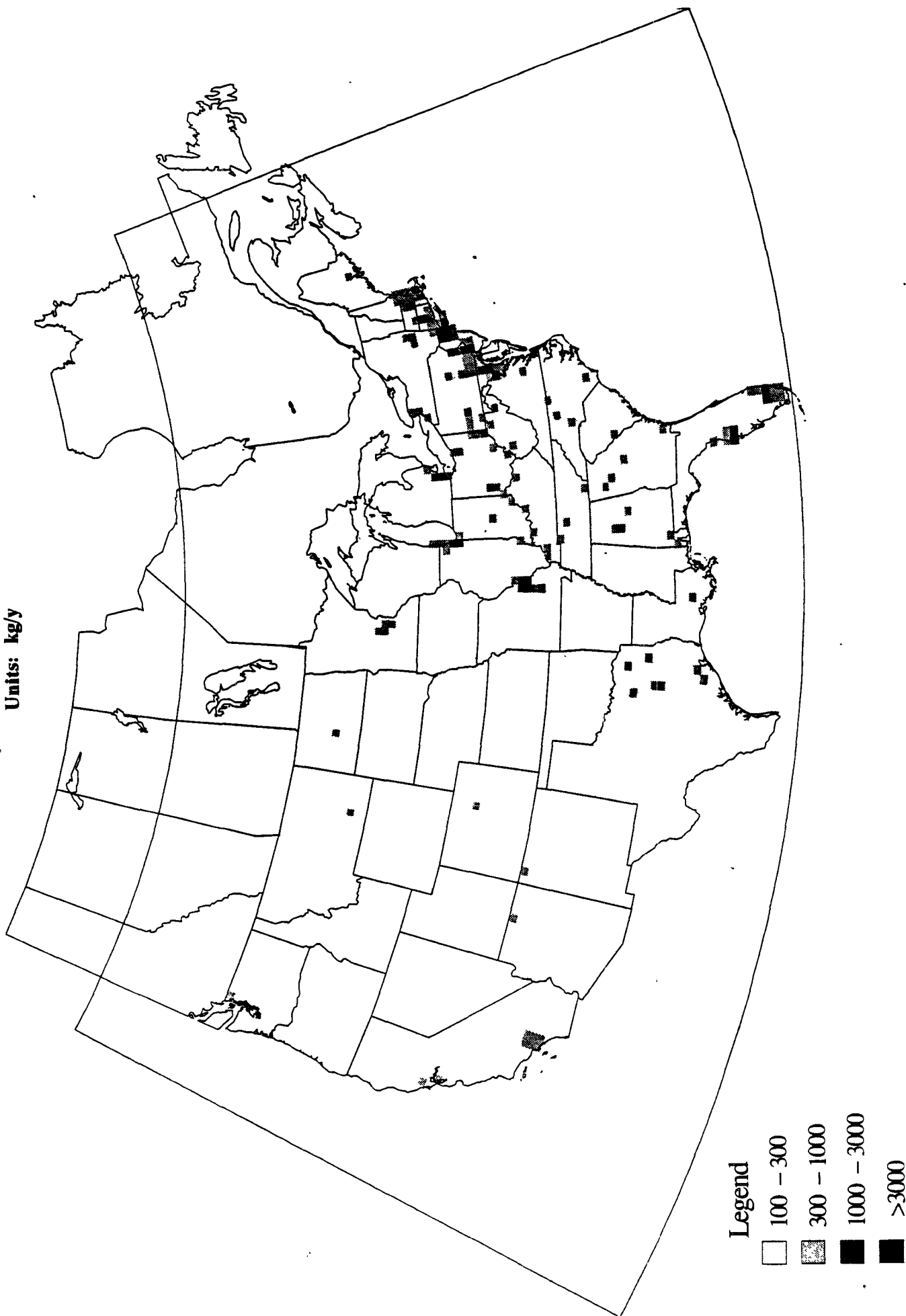
Throughout development of the inventory, an attempt was made to incorporate, to the extent possible, the most recent emissions data for each of the source categories. Not all of the changes could be incorporated into the subsequent modeling analyses. As a result, there are three discrepancies between the inventory described in this document and the emissions modeled in the long-range transport analysis. First, in this document, aggregate emissions for the chlor-alkali industry are estimated to be 5.9 Mg/year (6.5 tons/year) based on 1991 industry surveys. By comparison, the long-range transport analysis modeled 6.5 Mg/year (7.2 tons/year) based on earlier data submitted to the Toxic Release Inventory. The second discrepancy is for mercury emissions from paint application. The mercury inventory was updated to reflect that no mercury has been added to paint since 1991. The long-range transport analysis uses a previous estimate of 4 Mg/year (4.4 tons/year) to account for previously applied paint that may still be out-gassing mercury (U.S. EPA, 1993d). The third discrepancy is that the inventory was revised to reflect 0.23 Mg/year (0.25 tons/yr) of emissions from oil-fired utility boilers. This revision was based on a re-analysis of the detection limits for residual oil samples. The long-range transport analysis uses an earlier estimate of 3.25 tons/yr for this source category. None of these discrepancies is believed to be significant enough to affect the mercury deposition patterns predicted by the long-range transport model.

Table 5-3
Mercury Area Sources Allocation Methodology

Area Source Category	Emissions Mg/year (tons/yr)	Allocation Method
Latex Paint Application	4 (4.4)	Nationwide estimate allocated on a per capita basis (1980 Census).
Mercury Lamp Breakage	1.4 (1.5)	Nationwide estimate allocated to counties on a per capita basis (1980 Census).
General Laboratory Usage	0.7 (0.8)	Nationwide estimate allocated on a per capita basis (1980 Census data).
Dental Preparation	0.7 (0.8)	Nationwide estimate allocated by number of dental establishments. SIC 8020, 8072.
Residential Coal Combustion	0.5 (0.6)	Nationwide estimate allocated to counties that reported residential coal combustion in the 1980 Census. Apportionment to counties on a per capita basis.
Residential Oil Combustion	2.7 (3.0)	Nationwide estimate allocated by State based on fuel consumption (U.S. Department of Energy, 1992). Apportionment to counties within State on a per capita basis.
Industrial/Commercial Boilers Coal Oil	20.7 (22.8) 5.5 (6.0)	Nationwide estimates allocated by State based on fuel consumption (U.S. Department of Energy, 1992). Apportionment to counties within State on a per capita basis.
Crematories	0.4 (0.4)	Statewide emissions estimate allocated to counties on a per capita basis.
Medical Waste Incinerators	58.8 (64.7)	Nationwide estimate allocated by the number of hospital beds per county.

SIC = Standard Industrial Classification

Figure 5-1
Annual Mercury Emissions All Sources, All Species
Units: kg/y



In addition, it should be noted that the long-range transport analysis utilized the emission estimates for 1990 from this inventory. For some source categories, estimates for 1995 are given where new information became available prior to publication. However, it is the 1990 estimates that were modeled.

Trends in Mercury Emissions

It is difficult to predict with certainty the temporal trends in mercury emissions for the U.S., although there appears to be a trend toward decreasing total mercury emissions from 1990 to 1995. This is particularly true for the combustion sources where mercury is a trace contaminant of the fuel. Also, as previously noted, there are a number of source categories where there is insufficient data to estimate current emissions let alone potential future emissions. Based on available information however, a number of observations can be made regarding mercury emission trends from source categories where some information is available about past activities and projected future activities.

There has been a real success in the U.S. in the dramatic drop in mercury emissions from manufacturing over the past decade. Current emissions of mercury from manufacturing sources are generally low (with the exception of chlor-alkali plants using the mercury cell process). The emissions of mercury are more likely to occur when the product is broken or discarded. Therefore, in terms of emission trends, one would expect that if the future consumption of mercury remains consistent with the 1993 consumption rate, emissions from most manufacturing sources would remain about the same.

For industrial or manufacturing sources that use mercury in products or processes, the overall consumption of mercury is generally declining. Industrial consumption of mercury has declined by about two thirds between 1988 (1508 Mg) and 1993 (558 Mg). Much of this decline can be attributed to the elimination of mercury as a paint additive (20 percent), and the reduction of mercury in batteries (36 percent). Use of mercury by other source categories, as described in Chapter 2 of this document, remained about the same between 1988 and 1993.

Secondary production of mercury (i.e., recovering mercury from waste products) has increased significantly over the past few years. Of the 558 Mg of mercury used in industrial processes in 1993, 63 percent was provided by secondary mercury producers. This represents a two-fold increase since 1991. The number of secondary mercury producers is expected to increase as more facilities open to recover mercury from fluorescent lamps and other mercury containing products (e.g., thermometers). As a result there is potential for mercury emissions from this source category to increase.

The largest identifiable sources of mercury emissions currently are municipal waste combustors and medical waste incinerators. Emissions from these source categories are expected to decline significantly by the year 2000 due to regulatory action the U.S. EPA is taking under the statutory authority of section 129 of the CAA. As described in sections 4.1.1 and 4.1.2 of this document, the U.S. EPA has final rules for MWCs and proposed rules for MWIs that will reduce mercury emissions from both of these source categories by about 90 percent. In addition to this federal action, a number of states (including Minnesota, Florida and New Jersey) have implemented mandatory recycling programs to reduce the mercury-containing waste, and some states have regulations that impose emission limits that are lower than the federal regulation. These factors will reduce national mercury emissions from these source categories even further.

After municipal solid waste and medical waste incinerators have been controlled, the largest remaining identified source of mercury emissions will be fossil fuel combustion by utility boilers, particularly coal combustion. Future trends in mercury emissions from this source category are largely

dependent on both the nation's future energy needs and the fuel chosen to meet those needs. Another factor is what actions the utility industry may take in the future to meet air quality requirements under the Clean Air Act.

6. CONCLUSIONS

The following conclusions are presented in approximate order of degree of certainty in the conclusion, based on the quality of the underlying database. The conclusions progress from those with greater certainty to those with lesser certainty.

- Numerous industrial and manufacturing processes emit mercury to the atmosphere. Mercury emissions from U.S. manufacturing sources, however, have dropped dramatically over the past decade.
- Prior to 1995, municipal waste combustors and medical waste incinerators were the largest identifiable source of mercury emissions to the atmosphere. Regulations finalized for municipal waste combustors and proposed for medical waste incinerators will reduce emissions from these sources by 90 percent.
- After emissions from municipal solid waste combustors and medical waste incinerators have been reduced, combustion of fossil fuels, particularly coal, will be the largest remaining source of mercury emissions to the atmosphere.
- Mercury is emitted, to a varying degree, from anthropogenic sources virtually everywhere in the United States.
- Natural sources of mercury and re-emission of previously deposited mercury are also sources of mercury to the atmosphere, although the magnitude of the contribution of these sources relative to the contribution of current anthropogenic sources is not well understood.
- Anthropogenic sources in the United States emit approximately 220 Mg (243 tons) of mercury annually into the atmosphere. This estimate is believed to be accurate to within 30 percent. This estimate represents emissions calculated during the 1990-1993 timeframe. Emission estimates for 1995 are about 40 tons lower.
- In the United States, land areas east of the Rocky Mountains have the highest concentration of emissions from anthropogenic sources in the U.S.
- The land areas having the greatest concentration of mercury emissions from anthropogenic sources of total mercury (i.e., all chemical species) are the following: the urban corridor from Washington D.C. to Boston, the Tampa and Miami areas of Florida, the larger urban areas of the Midwest and Ohio Valley and two sites in northeastern Texas.
- The land areas having generally the lowest emissions are in the Great Basin region of the western United States and the High Plains region of the central United States. There are generally few large emission sources in the western third of the United States, with the exception of the San Francisco and Los Angeles areas and specific industrial operations.

There are many uncertainties in the emission estimates for individual source categories due to uncertainties inherent in an emission factor approach. The source of these uncertainties include the following:

- Variability in the estimates of source activity for each source category. Activity levels used in this Report were compiled over different time periods and by a variety of survey procedures.

- Emissions test data that are of poor quality or are based on very few analyses, which may not be representative of the full source population being studied.
- Changes in processes or emission measurement techniques over time (especially since about 1985). Earlier techniques may have measured too much mercury because of contamination problems.
- A lack of data for some source categories that either led to estimates based on engineering judgment or mass balance calculations. For a number of source categories there were insufficient data and, thus, no emissions estimates were made.
- Limited data on the effectiveness of air pollution control equipment to capture mercury emissions.

Understanding the public health and environmental impacts of current anthropogenic emissions is complicated by an incomplete understanding of the following factors:

- Global and transboundary deposition of mercury and the impact this has on deposition of mercury in the U.S.
- The magnitude and chemical nature of natural emissions.
- The magnitude and chemical nature of re-emitted mercury.
- The public health and environmental impacts of emissions from past uses of mercury (such as paint application) relative to current anthropogenic emissions.

To improve the emissions estimates, U.S. EPA would need the following:

- Emissions test data for source categories where there is currently insufficient data to estimate national emissions, including mobile sources, landfills, agricultural burning, sludge application, coke ovens, petroleum refining, residential woodstoves, mercury compounds production and zinc mining.
- Improvements in the existing emissions information for a number of source categories which could include these: secondary mercury production (i.e., recycling), commercial and industrial boilers, electric lamp breakage, iron and steel manufacturing and primary lead smelting.

Based on trends in mercury use and emissions, the U.S. EPA predicts the following:

- A significant (90 percent) decrease will occur in mercury emissions from municipal waste combustors and medical waste incinerators when the regulations put forth by U.S. EPA for these source categories are fully implemented.
- Manufacturing use of mercury will continue to decline with chlorine production from mercury cell chlor-alkali plants continuing to account for most of the mercury use in the manufacturing sector.
- Secondary production of mercury will continue to increase as more recycling facilities commence operation to recover mercury from discarded products and wastes.

7. RESEARCH NEEDS

Throughout this volume an effort has been made to characterize the uncertainties (at least qualitatively) in the emissions estimates for the various source categories described. As noted in Chapter 1, there are inherent uncertainties in estimating emissions using emission factors. To reduce these uncertainties, a number of research needs remain, including the following:

- Source test data are needed from a number of source categories that have been identified in this volume as having insufficient data to estimate emissions. These source categories are listed in Table 1-3. Notable among these are mobile sources, landfills, agricultural burning, sludge application, coke ovens, petroleum refining, residential woodstoves, mercury compounds production and zinc mining. A number of manufacturing sources were also identified as having highly uncertain emissions estimates. Notable among this category are secondary mercury production, commercial and industrial boilers, electric lamp breakage, primary metal smelting operations and iron and steel manufacturing. The possibility of using emissions data from other countries could be further investigated.
- Development and validation of a stack test protocol for speciated mercury emissions is needed.
- More data are needed on the efficacy of coal cleaning and the potential for slurries from the cleaning process to be a mercury emission source.
- More data are needed on the mercury content of various coals and petroleum and the trends in the mercury content of coal burned at utilities and petroleum refined in the U.S.
- Additional research is needed to address the potential for methylmercury to be emitted (or formed) in the flue gas of combustion sources.
- The importance (quantitatively) of re-emission of mercury from previously deposited anthropogenic emissions and mercury-bearing mining waste needs to be investigated. This would include both terrestrial and water environments. Measuring the flux of mercury from various environments would allow a determination to be made of the relative importance of re-emitted mercury to the overall emissions of current anthropogenic sources.
- Determination of the mercury flux from natural sources would help determine the impact of U.S. anthropogenic sources on the global mercury cycle as well as the impact of all mercury emissions in the United States.
- The use of more sophisticated fate and transport models for mercury will require more detailed emissions data, particularly more information on the chemical species of mercury being emitted (including whether these species are particle-bound) and the temporal variability of the emissions.

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APPENDIX A

**INFORMATION ON LOCATIONS OF AND EMISSIONS FROM
COMBUSTION SOURCES**



Table A-1
Mercury Emissions From Utility Boilers, By State and Fuel Type

State	Combined cycle ^a			Coal			Natural Gas			Oil			Total		
	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr
Alaska	0	0	0	1	0.005	0.006	0	0	0	0	0	0	1	0.005	0.006
Alabama	0	0	0	38	1.892	2.086	2	0.000	0.000	0	0	0	40	1.892	2.086
Arkansas	0	0	0	5	0.411	0.453	6	0.000	0.000	0	0	0	11	0.411	0.453
Arizona	0	0	0	14	0.706	0.778	14	0.000	0.000	0	0	0	28	0.706	0.778
California	4	0.001	0.001	0	0	0	96	0.000	0.000	38	0.008	0.009	138	0.009	0.010
Colorado	0	0	0	24	0.830	0.915	3	0.000	0.000	0	0	0	27	0.830	0.915
Connecticut	0	0	0	1	0.071	0.079	2	0.000	0.000	19	0.010	0.011	22	0.081	0.090
Delaware	0	0	0	6	0.137	0.151	2	0.000	0.000	2	0.001	0.001	10	0.138	0.152
Florida	2	0.001	0.001	29	1.258	1.387	51	0.000	0.000	49	0.062	0.069	131	1.321	1.457
Georgia	0	0	0	36	1.753	1.932	6	0.000	0.000	2	0.000	0.000	44	1.753	1.932
GU	0	0	0	0	0	0	0	0	0	4	0.002	0.003	4	0.002	0.003
Hawaii	0	0	0	0	0	0	0	0	0	14	0.017	0.010	14	0.017	0.010
Iowa	0	0	0	31	0.780	0.860	0	0	0	0	0	0	31	0.780	0.860
Illinois	0	0	0	57	1.265	1.395	5	0.000	0.000	10	0.002	0.002	72	1.267	1.397
Indiana	0	0	0	71	2.195	2.420	2	0.000	0.000	5	0.000	0.000	78	2.195	2.420
Kansas	0	0	0	19	0.466	0.514	17	0.000	0.000	0	0	0	36	0.466	0.514
Kentucky	0	0	0	55	1.766	1.946	0	0.000	0.000	1	0.000	0.000	56	1.766	1.946
Louisiana	2	0.000	0.000	6	0.737	0.812	46	0.000	0.000	0	0.000	0.000	55	0.737	0.812
Massachusetts	0	0	0	9	0.288	0.318	8	0.000	0.000	19	0.011	0.013	36	0.299	0.331
Maryland	0	0	0	14	0.937	1.033	4	0.000	0.000	13	0.008	0.009	31	0.945	1.042
Maine	0	0	0	0	0	0	0	0	0	8	0.003	0.003	8	0.003	0.003

Table A-1 (continued)
Mercury Emissions From Utility Boilers, By State and Fuel Type

State	Combined cycle ^a			Coal			Natural Gas			Oil			Total		
	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr
Michigan	0	0	0	52	1.726	1.902	1	0.000	0.000	6	0.002	0.002	59	1.728	1.904
Minnesota	0	0	0	26	0.672	0.741	1	0.000	0.000	0	0.000	0.000	27	0.672	0.741
Missouri	0	0	0	34	1.378	1.519	4	0.000	0.000	0	0	0	38	1.378	1.519
Mississippi	0	0	0	6	0.173	0.190	15	0.000	0.000	2	0.001	0.001	23	0.174	0.191
Montana	0	0	0	6	0.357	0.393	0	0.000	0.000	0	0	0	6	0.357	0.393
North Carolina	0	0	0	36	1.253	1.381	0	0	0	0	0	0	36	1.253	1.381
North Dakota	0	0	0	13	1.106	1.219	0	0	0	0	0	0	13	1.106	1.219
Nebraska	0	0	0	13	0.443	0.488	3	0.000	0.000	0	0	0	16	0.443	0.488
New Hampshire	0	0	0	4	0.130	0.144	0	0	0	3	0.002	0.002	7	0.132	0.146
New Jersey	0	0	0	8	0.173	0.190	13	0.000	0.000	15	0.005	0.005	36	0.178	0.195
New Mexico	0	0	0	10	0.396	0.437	9	0.000	0.000	0	0	0	19	0.396	0.437
Nevada	0	0	0	8	0.253	0.278	9	0.000	0.000	6	0	0	23	0.253	0.278
New York	0	0	0	32	1.208	1.332	32	0.000	0.000	41	0.057	0.063	105	1.265	1.395
Ohio	0	0	0	91	3.613	3.982	1	0.000	0.000	3	0.000	0.000	95	3.613	3.982
Oklahoma	4	0.001	0.001	10	0.533	0.587	26	0.000	0.000	0	0	0	40	0.534	0.588
Oregon	0	0	0	1	0.034	0.038	0	0	0	0	0	0	1	0.034	0.038
Pennsylvania	0	0	0	58	4.657	5.133	2	0	0	8	0.006	0.007	68	4.663	5.140
Puerto Rico	0	0	0	0	0	0	0	0	0	18	0.028	0.031	18	0.028	0.031
Rhode Island	0	0	0	0	0	0	4	0.000	0.000	1	0.000	0.000	5	0.000	0.000
South Carolina	0	0	0	24	0.547	0.603	2	0.000	0.000	4	0.000	0.000	30	0.547	0.603
South Dakota	0	0	0	2	0.154	0.170	1	0.000	0.000	0	0	0	3	0.154	0.170

Table A-1 (continued)
Mercury Emissions From Utility Boilers, By State and Fuel Type

State	Combined cycle ^a			Coal			Natural Gas			Oil			Total		
	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr	No.	Mg/yr	tons/yr
Tennessee	0	0	0	33	1.362	1.501	0	0	0	0	0	0	33	1.362	1.501
Texas	3	0.001	0.001	34	5.599	6.172	154	0.000	0.000	5	0.000	0.000	196	5.599	6.172
Utah	0	0	0	12	0.170	0.188	1	0.000	0.000	0	0	0	13	0.170	0.188
Virginia	0	0	0	24	0.564	0.621	1	0	0	4	0.002	0.003	29	0.564	0.621
VI	0	0	0	0	0	0	0	0	0	2	0.000	0.000	2	0.000	0.000
Washington	0	0	0	2	0.181	0.199	0	0	0	0	0.000	0.000	2	0.181	0.199
Washington D.C.	0	0	0	0	0	0	0	0	0	2	0.001	0.001	2	0.001	0.001
Wisconsin	0	0	0	40	1.063	1.172	2	0.000	0.000	0	0	0	42	1.063	1.172
West Virginia	0	0	0	33	1.911	2.107	0	0	0	0	0	0	33	1.911	2.107
Wyoming	0	0	0	15	0.851	0.938	0	0	0	0	0	0	15	0.851	0.938
Total	15	0.004	0.004	1043	45.999	50.710	545	0.001	0.002	305	0.230	0.250	1908	46.234	50.966

^a These units burn a combination of fuels.

Note: Totals shown here differ slightly from those shown elsewhere in this volume due to rounding. For each source category, a value of "0" means that the emissions estimate is zero whereas "0.000" means that the estimate is less than 0.001, the minimum value that could be shown in the space allotted.

Table A-2
Estimates of Coal, Natural Gas, and Oil Consumption
in the Commercial/Industrial Sector Per State
(Trillion Btu)

State	Coal			Natural gas	Petroleum
	Bituminous coal and lignite	Anthracite	Total		Distillate and residual
Alabama	144.6	0.4	145	185.0	45.6
Alaska	5.1	0.0	5.1	277.2	16.0
Arizona	13.3	0.0	13.3	48.3	21.1
Arkansas	5.8	0.0	5.8	153.6	23.4
California	65.1	0.0	65.1	900.6	138.2
Colorado	16.2	0.0	16.2	133.4	18.1
Connecticut	0.0	0.1	0.1	56.7	23
Delaware	5.8	0.0	5.8	21.4	4.5
Dist. of Col.	1.1	0.0	1.1	13.6	2.9
Florida	30.1	0.1	30.2	133.7	39.2
Georgia	56.5	^a	56.5	217.2	31.1
Hawaii	0.7	0.0	0.7	2.4	7.7
Idaho	9.6	0.0	9.6	32.8	17.9
Illinois	154.5	0.1	154.6	486.5	53.4
Indiana	350.6	0.1	350.7	300.8	32.7
Iowa	56.2	0.7	56.9	135.2	27.0
Kansas	3.8	0.0	3.8	213.8	24.4
Kentucky	89	0.5	89.5	107.8	34.2
Louisiana	16	0.0	16	1242.4	84.8
Maine	6.0	0.1	6.1	3.7	13.9
Maryland	58.1	^a	58.1	88.2	22.3
Massachusetts	2.3	0.4	2.7	98.1	49.0
Michigan	122.2	^a	122.2	468.3	29.9
Minnesota	25.9	0.0	25.9	167.0	33.0
Mississippi	6.3	0.0	6.3	129.8	36.4
Missouri	34.5	0.0	34.5	115.1	23.6
Montana	4.7	0.0	4.7	24.5	16.9

Table A-2 (continued)
Estimates of Coal, Natural Gas, and Oil Consumption
in the Commercial/Industrial Sector Per State
(Trillion Btu)

State	Coal			Natural gas	Petroleum
	Bituminous coal and lignite	Anthracite	Total		Distillate and residual
Nebraska	4.6	^a	4.6	61.3	25.5
Nevada	3.9	^a	3.9	23.2	21.0
New Hampshire	0.5	0.3	0.8	8.4	9.4
New Jersey	6.8	0.3	7.1	211.2	57.2
New Mexico	1.0	0.0	1	115.0	16.4
New York	84.0	2.1	86.1	305.7	95.6
North Carolina	77.1	0.0	77.1	121.2	28.3
North Dakota	87.5	0.0	87.5	22.3	16.0
Ohio	258.0	0.3	258.3	444.7	39.5
Oklahoma	12.7	0.0	12.7	350.7	21.1
Oregon	1.4	0.0	1.4	71.0	24.4
Pennsylvania	382.3	15.1	397.4	393.5	69.3
Rhode Island	0.0	0.1	0.1	8.3	5.3
South Carolina	57.9	0.2	58.1	105.1	14.9
South Dakota	3.9	0.0	3.9	14.7	13.1
Tennessee	100.0	0.3	100.3	158.7	20.7
Texas	61.7	^a	61.7	2373.3	169.9
Utah	52.7	0.0	52.7	77.8	10.9
Vermont	0.0	0.1	0.1	3.9	6
Virginia	121.4	0.3	121.7	121.1	31.6
Washington	6.1	0.0	6.1	120.7	38.2
West Virginia	127.0	0.1	127.1	84.6	18.2
Wisconsin	47.2	0.1	47.3	189.4	31.6
Wyoming	42.9	0.0	42.9	83.1	14.5
United States	2824.6	21.8	2842.5	11226.0	1668.8

^a Number less than 0.05

Source: U.S. Department of Energy. State Energy Data Report. Report No. DOE/EIA-0214(40). May 1992.

Table A-3
Estimates of Mercury Emissions From Coal-Fired
Commercial/Industrial Boilers on a Per-State Basis For 1991

State	Coal consumption, trillion Btu			Mercury emissions ^a	
	Bituminous coal and lignite	Anthracite	Total	Ton/Yr ^c	Mg/Yr
Alabama	144.6	0.4	145	1.2	1.1
Alaska	5.1	0.0	5.1	0.0	0.0
Arizona	13.3	0.0	13.3	0.1	0.1
Arkansas	5.8	0.0	5.8	0.0	0.0
California	65.1	0.0	65.1	0.5	0.5
Colorado	16.2	0.0	16.2	0.1	0.1
Connecticut	0.0	0.1	0.1	0.0	0.0
Delaware	5.8	0.0	5.8	0.0	0.0
Dist. of Col.	1.1	0.0	1.1	0.0	0.0
Florida	30.1	0.1	30.2	0.2	0.2
Georgia	56.5	b	56.5	0.5	0.4
Hawaii	0.7	0.0	0.7	0.0	0.0
Idaho	9.6	0.0	9.6	0.1	0.1
Illinois	154.5	0.1	154.6	1.2	1.1
Indiana	350.6	0.1	350.7	2.8	2.6
Iowa	56.2	0.7	56.9	0.5	0.4
Kansas	3.8	0.0	3.8	0.0	0.0
Kentucky	89	0.5	89.5	0.7	0.7
Louisiana	16	0.0	16	0.1	0.1
Maine	6.0	0.1	6.1	0.0	0.0
Maryland	58.1	b	58.1	0.5	0.4
Massachusetts	2.3	0.4	2.7	0.0	0.0
Michigan	122.2	b	122.2	1.0	0.9
Minnesota	25.9	0.0	25.9	0.2	0.2
Mississippi	6.3	0.0	6.3	0.1	0.0
Missouri	34.5	0.0	34.5	0.3	0.3
Montana	4.7	0.0	4.7	0.0	0.0
Nebraska	4.6	b	4.6	0.0	0.0
Nevada	3.9	b	3.9	0.0	0.0
New Hampshire	0.5	0.3	0.8	0.0	0.0
New Jersey	6.8	0.3	7.1	0.1	0.1
New Mexico	1.0	0.0	1	0.0	0.0
New York	84.0	2.1	86.1	0.7	0.6
North Carolina	77.1	0.0	77.1	0.6	0.6
North Dakota	87.5	0.0	87.5	0.7	0.6
Ohio	258.0	0.3	258.3	2.1	1.9
Oklahoma	12.7	0.0	12.7	0.1	0.1
Oregon	1.4	0.0	1.4	0.0	0.0
Pennsylvania	382.3	15.1	397.4	3.2	2.9
Rhode Island	0.0	0.1	0.1	0.0	0.0
South Carolina	57.9	0.2	58.1	0.5	0.4
South Dakota	3.9	0.0	3.9	0.0	0.0
Tennessee	100.0	0.3	100.3	0.8	0.7
Texas	61.7	b	61.7	0.5	0.4
Utah	52.7	0.0	52.7	0.4	0.4
Vermont	0.0	0.1	0.1	0.0	0.0
Virginia	121.4	0.3	121.7	1.0	0.9
Washington	6.1	0.0	6.1	0.0	0.0
West Virginia	127.0	0.1	127.1	1.0	0.9
Wisconsin	47.2	0.1	47.3	0.4	0.3
Wyoming	42.9	0.0	42.9	0.3	0.3
United States	2824.6	21.8	2842.5	22.8	20.7

^a Mercury emission factors of 16 lb Hg/trillion Btu and 18 lb Hg/trillion Btu were used for bituminous and anthracite coal, respectively. No control of emissions from commercial/industrial boilers was assumed.

^b Number less than 0.05.

^c Emissions less than 100 pounds/year for an entire State are reported as zero.

Table A-4
Estimates of Mercury Emissions From Oil-Fired
Commercial/Industrial Boilers On a Per-State Basis For 1991

State	Petroleum consumption, trillion Btu	Mercury emissions ^a	
	Distillate and residual	Ton/Yr	Mg/Yr
Alabama	45.6	0.16	0.15
Alaska	16.0	0.06	0.05
Arizona	21.1	0.08	0.07
Arkansas	23.4	0.08	0.08
California	138.2	0.50	0.45
Colorado	18.1	0.07	0.06
Connecticut	23	0.08	0.08
Delaware	4.5	0.02	0.01
Dist. of Col.	2.9	0.01	0.01
Florida	39.2	0.14	0.13
Georgia	31.1	0.11	0.10
Hawaii	7.7	0.03	0.03
Idaho	17.9	0.06	0.06
Illinois	53.4	0.19	0.17
Indiana	32.7	0.12	0.11
Iowa	27.0	0.10	0.09
Kansas	24.4	0.09	0.08
Kentucky	34.2	0.12	0.11
Louisiana	84.8	0.31	0.28
Maine	13.9	0.05	0.05
Maryland	22.3	0.08	0.07
Massachusetts	49.0	0.18	0.16
Michigan	29.9	0.11	0.10
Minnesota	33.0	0.12	0.11
Mississippi	36.4	0.13	0.12
Missouri	23.6	0.08	0.08
Montana	16.9	0.06	0.06
Nebraska	25.5	0.09	0.08
Nevada	21.0	0.08	0.07
New Hampshire	9.4	0.03	0.03
New Jersey	57.2	0.21	0.19
New Mexico	16.4	0.06	0.05
New York	95.6	0.34	0.31
North Carolina	28.3	0.10	0.09
North Dakota	16.0	0.06	0.05
Ohio	39.5	0.14	0.13
Oklahoma	21.1	0.08	0.07
Oregon	24.4	0.09	0.08
Pennsylvania	69.3	0.25	0.23
Rhode Island	5.3	0.02	0.02
South Carolina	14.9	0.05	0.05
South Dakota	13.1	0.05	0.04
Tennessee	20.7	0.07	0.07
Texas	169.9	0.61	0.56
Utah	10.9	0.04	0.04
Vermont	6	0.02	0.02
Virginia	31.6	0.11	0.10
Washington	38.2	0.14	0.13
West Virginia	18.2	0.07	0.06
Wisconsin	31.6	0.11	0.10
Wyoming	14.5	0.05	0.05
United States	1,668.8	6.01	5.46

^a Mercury emission factor for distillate oil is 7.2 lb Hg/trillion Btu. Calculation was performed assuming that all pollution control devices provide no mercury reduction.

^b Number less than 0.05.

Table A-5
Estimates of Coal, Natural Gas, and Oil Consumption
in the Residential Sector Per State (Trillion Btu)

State	Coal			Natural gas	Petroleum distillate and residual
	Bituminous coal and lignite	Anthracite	Total		
Alabama	0.9	0.0	0.9	46.7	0.1
Alaska	2.7	0.0	2.7	13.4	10.2
Arizona	a	0.0	0	29.3	0.1
Arkansas	a	0.0	0	39.5	a
California	0.2	0.0	0.2	530.8	1.3
Colorado	0.4	0.0	0.4	92.4	0.2
Connecticut	0.0	0.2	0.2	38.7	66.6
Delaware	0.2	0.0	0.2	7.4	5.6
Dist. of Col.	0.6	0.0	0.6	15.3	0.9
Florida	a	0.0	0	14.1	1.4
Georgia	0.2	0.0	0.2	92.7	1.5
Hawaii	0.0	0.0	0	0.6	a
Idaho	0.5	0.0	0.5	8.8	3.1
Illinois	2.1	a	2.1	451.9	7.0
Indiana	4.3	a	4.3	143.1	10.0
Iowa	2.0	a	2	71.9	4.6
Kansas	a	0.0	0	71.3	0.1
Kentucky	1.3	0.0	1.3	58.5	3.8
Louisiana	0	0.0	0	55.6	0.1
Maine	0.3	0.2	0.5	0.7	29.3
Maryland	0.4	0.1	0.5	68.2	25.0
Massachusetts	0.3	0.4	0.7	110.5	100.7
Michigan	2.3	0.0	2.3	342.2	24.3
Minnesota	1.1	0.0	1.1	107.4	18.8
Mississippi	0.0	0.0	0	25.9	a
Missouri	2.2	0.0	2.2	117.2	2.1
Montana	0.4	0.0	0.4	17.3	1.7
Nebraska	a	0.0	0	40.8	1.0
Nevada	a	a	a	17.7	1.4
New Hampshire	0.0	0.2	0.2	6.0	19.8
New Jersey	0.0	0.2	0.2	176.0	67.0
New Mexico	a	0.0	0	29.7	0.1
New York	1.2	2.0	3.2	347.8	154.5
North Carolina	1.4	0.0	1.4	36.1	20.7
North Dakota	0.7	0.0	0.7	9.5	4.9
Ohio	5.5	a	5.5	321.0	23.8
Oklahoma	0.0	0.0	0	66.9	a
Oregon	a	0.0	0	23.9	10.4
Pennsylvania	2.9	14.7	17.6	248.9	99.1
Rhode Island	0.0	0.1	0.1	18.2	14.9
South Carolina	0.1	0.0	0.1	18.9	5.9
South Dakota	a	0.0	0	10.4	4.7
Tennessee	1.8	0.1	1.9	48.0	1.4
Texas	0.1	0.0	0.1	220.8	a
Utah	2.2	0.0	2.2	47.3	0.8
Vermont	0.0	0.1	0.1	2.1	11.2
Virginia	2.1	a	2.1	53.6	29.8
Washington	0.5	0.0	0.5	41.6	17.5
West Virginia	1.6	0.0	1.6	34.9	3.3
Wisconsin	a	a	0	114.7	27.0
Wyoming	0.9	0.0	0.9	12.6	0.1
United States	43.4	18.3	61.7	4,518.8	837.8

^a Number less than 0.05.

Source: U.S. Department of Energy. State Energy Data Report. Report No. DOE/EIA-0214(40). May 1992.

Table A-6
Estimates of Mercury Emissions From
Coal-Fired Residential Boilers on a Per-State Basis For 1991

State	Coal consumption, trillion Btu			Mercury emissions ^a	
	Bituminous coal and lignite	Anthracite	Total	Ton/Yr	Mg/Yr
Alabama	0.9	0.0	0.9	0.007	0.007
Alaska	2.7	0.0	2.7	0.022	0.020
Arizona	b	0.0	0	0.000	0.000
Arkansas	b	0.0	0	0.000	0.000
California	0.2	0.0	0.2	0.002	0.001
Colorado	0.4	0.0	0.4	0.003	0.003
Connecticut	0.0	0.2	0.2	0.002	0.002
Delaware	0.2	0.0	0.2	0.002	0.001
Dist. of Col.	0.6	0.0	0.6	0.005	0.004
Florida	b	0.0	0	0.000	0.000
Georgia	0.2	0.0	0.2	0.002	0.001
Hawaii	0.0	0.0	0	0.000	0.000
Idaho	0.5	0.0	0.5	0.004	0.004
Illinois	2.1	b	2.1	0.017	0.015
Indiana	4.3	b	4.3	0.034	0.031
Iowa	2.0	b	2	0.016	0.015
Kansas	b	0.0	0	0.000	0.000
Kentucky	1.3	0.0	1.3	0.010	0.009
Louisiana	0	0.0	0	0.000	0.000
Maine	0.3	0.2	0.5	0.004	0.004
Maryland	0.4	0.1	0.5	0.004	0.004
Massachusetts	0.3	0.4	0.7	0.006	0.005
Michigan	2.3	0.0	2.3	0.018	0.017
Minnesota	1.1	0.0	1.1	0.009	0.008
Mississippi	0.0	0.0	0	0.000	0.000
Missouri	2.2	0.0	2.2	0.018	0.016
Montana	0.4	0.0	0.4	0.003	0.003
Nebraska	b	0.0	0	0.000	0.000
Nevada	b	b	b	0.000	0.000
New Hampshire	0.0	0.2	0.2	0.002	0.002
New Jersey	0.0	0.2	0.2	0.002	0.002
New Mexico	b	0.0	0	0.000	0.000
New York	1.2	2.0	3.2	0.028	0.025
North Carolina	1.4	0.0	1.4	0.011	0.010
North Dakota	0.7	0.0	0.7	0.006	0.005
Ohio	5.5	b	5.5	0.044	0.040
Oklahoma	0.0	0.0	0	0.000	0.000
Oregon	b	0.0	0	0.000	0.000
Pennsylvania	2.9	14.7	17.6	0.156	0.141
Rhode Island	0.0	0.1	0.1	0.001	0.001
South Carolina	0.1	0.0	0.1	0.001	0.001
South Dakota	b	0.0	0	0.000	0.000
Tennessee	1.8	0.1	1.9	0.015	0.014
Texas	0.1	0.0	0.1	0.001	0.001
Utah	2.2	0.0	2.2	0.018	0.016
Vermont	0.0	0.1	0.1	0.001	0.001
Virginia	2.1	b	2.1	0.017	0.015
Washington	0.5	0.0	0.5	0.004	0.004
West Virginia	1.6	0.0	1.6	0.013	0.012
Wisconsin	b	b	0	0.000	0.000
Wyoming	0.9	0.0	0.9	0.007	0.007
United States	43.4	18.3	61.7	0.512	0.465

^a Mercury emission factors of 16 lb Hg/trillion Btu and 18 lb Hg/trillion Btu were used for bituminous and anthracite coal, respectively. No control of emissions from residential boilers was assumed.

^b Number less than 0.05.

Table A-7
Estimates of Mercury Emissions From Oil-Fired
Residential Boilers on a Per-State Basis For 1991

State	Petroleum consumption, trillion Btu	Mercury emissions ^a	
	Distillate and residual	Ton/Yr	Mg/Yr
Alabama	0.1	0.0004	0.0003
Alaska	10.2	0.0367	0.0334
Arizona	0.1	0.0004	0.0003
Arkansas	*	0.0000	0.0000
California	1.3	0.0047	0.0043
Colorado	0.2	0.0007	0.0007
Connecticut	66.6	0.2398	0.2180
Delaware	5.6	0.0202	0.0183
Dist. of Col.	0.9	0.0032	0.0029
Florida	1.4	0.0050	0.0046
Georgia	1.5	0.0054	0.0049
Hawaii	*	0.0000	0.0000
Idaho	3.1	0.0112	0.0101
Illinois	7.0	0.0252	0.0229
Indiana	10.0	0.0360	0.0327
Iowa	4.6	0.0166	0.0151
Kansas	0.1	0.0004	0.0003
Kentucky	3.8	0.0137	0.0124
Louisiana	0.1	0.0004	0.0003
Maine	29.3	0.1055	0.0959
Maryland	25.0	0.0900	0.0818
Massachusetts	100.7	0.3625	0.3296
Michigan	24.3	0.0875	0.0795
Minnesota	18.8	0.0677	0.0615
Mississippi	*	0.0000	0.0000
Missouri	2.1	0.0076	0.0069
Montana	1.7	0.0061	0.0056
Nebraska	1.0	0.0036	0.0033
Nevada	1.4	0.0050	0.0046
New Hampshire	19.8	0.0713	0.0648
New Jersey	67.0	0.2412	0.2193
New Mexico	0.1	0.0004	0.0003
New York	154.5	0.5562	0.5056
North Carolina	20.7	0.0745	0.0677
North Dakota	4.9	0.0176	0.0160
Ohio	23.8	0.0857	0.0779
Oklahoma	*	0.0000	0.0000
Oregon	10.4	0.0374	0.0340
Pennsylvania	99.1	0.3568	0.3243
Rhode Island	14.9	0.0536	0.0488
South Carolina	5.9	0.0212	0.0193
South Dakota	4.7	0.0169	0.0154
Tennessee	1.4	0.0050	0.0046
Texas	*	0.0000	0.0000
Utah	0.8	0.0029	0.0026
Vermont	11.2	0.0403	0.0367
Virginia	29.8	0.1073	0.0975
Washington	17.5	0.0630	0.0573
West Virginia	3.3	0.0119	0.0108
Wisconsin	27.0	0.0972	0.0884
Wyoming	0.1	0.0004	0.0003
United States	837.8	3.02	2.74

^a Mercury emission factor for distillate oil is 7.2 lb Hg/trillion Btu. Calculations performed under the assumption that air pollution control devices provide no mercury reduction.

^b Number less than 0.05.

Table A-8
1991 U.S. Crematory Locations by State^{a,b}

State	No. of crematories	No. of cremations ^c	State	No. of crematories	No. of cremations
Alabama	6	1,138	Montana	12	2,502
Alaska	7	790	Nebraska	6	1,139
Arizona	26	10,189	Nevada	11	5,009
Connecticut	10	4,260	New York	40	23,946
Delaware	4	1,165	North Carolina	24	4,749
District of Columbia	1	d	North Dakota	1	d
Florida	95	46,775	Ohio	41	12,552
Georgia	14	2,684	Oklahoma	9	1,372
Hawaii	10	3,495	Oregon	34	9,020
Idaho	12	1,949	Pennsylvania	44	12,153
Illinois	44	12,083	Rhode Island	5	1,842
Indiana	21	3,636	South Carolina	10	1,764
Iowa	15	2,241	South Dakota	4	d
Kansas	10	1,559	Tennessee	8	1,712
Kentucky	5	1,192	Texas	36	9,340
Louisiana	6	1,853	Utah	5	769
Maine	4	2,656	Vermont	5	1,570
Maryland	17	5,587	Virginia	25	6,097
Massachusetts	13	8,104	Washington	46	15,673
Michigan	38	13,431	West Virginia	6	582
Minnesota	18	5,662	Wisconsin	29	5,541
Missouri	19	4,637			

^a Vander Most and Veldt, 1992.

^b Does not include the number of cremations in the District of Columbia, Arkansas, California, New Hampshire, New Mexico, Colorado, North Dakota, South Dakota, and Wyoming, or other U.S. territories.

^c 1990 data; 1991 data unavailable.

^d No information available.

Table A-9
Existing MWC Facilities (As of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Parsons (SOHIO) ^a	Endicott	AK	4,380	3,982
Juneau	Juneau	AK	25,550	23,227
Kyparuk (ARCO) ^a	Kyparuk	AK	4,380	3,982
Prudhoe Bay ^a	Prudhoe Bay	AK	36,500	33,182
Shemya (Air Force Base) ^a	Shemya	AK	7,300	6,636
Sitka (Sheldon Jackson College)	Sitka	AK	9,125	8,295
Huntsville	Huntsville	AL	251,850	228,955
Tuscaloosa ^a	Tuscaloosa	AL	109,500	99,545
Augusta ^a	Augusta	AR	7,300	6,636
Batesville	Batesville	AR	36,500	33,182
Blytheville	Blytheville	AR	25,550	23,227
Kensett ^a	Kensett	AR	5,475	4,977
North Little Rock ^a	North Little Rock	AR	36,500	33,182
Osceola	Osceola	AR	18,250	16,591
Stuttgart	Stuttgart	AR	21,900	19,909
Los Angeles County	Commerce	CA	138,700	126,091
Long Beach (SERRF)	Long Beach	CA	503,700	457,909
Stanislaus County	Modesto	CA	292,000	265,455
Bridgeport	Bridgeport	CT	821,250	746,591
Bristol	Bristol	CT	237,250	215,682
MID-Connecticut	Hartford	CT	730,000	663,636
New Cannan ^a	New Cannan	CT	45,625	41,477
Southeastern	Preston	CT	219,000	199,091
Stamford II ^a	Stamford	CT	131,400	119,455
Stamford I ^a	Stamford	CT	54,750	49,773
Wallingford	Wallingford	CT	153,300	139,364

^a No longer operational.

Table A-9 (continued)
Existing MWC Facilities (as of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Windham ^a	Windham	CT	39,420	35,836
Washington ^a	Washington	DC	365,000	331,818
Wilmington (Newcastle) ^a	Wilmington	DE	219,000	199,091
Hillsborough County	Brandon	FL	438,000	398,182
Fort Meade ^a	Fort Meade	FL	9,490	8,627
Broward County (South)	Ft. Lauderdale	FL	821,250	746,591
Pasco County	Hudson	FL	383,250	348,409
Monroe County	Key West	FL	54,750	49,773
Lakeland ^a	Lakeland	FL	109,500	99,545
Mayport NAS	Mayport	FL	17,520	15,927
Dade County	Miami	FL	1,095,000	995,455
Miami (Airport)	Miami	FL	21,900	19,909
Lake County	Okahumpka	FL	192,720	175,200
Bay County	Panama City	FL	186,150	169,227
Broward County (North)	Pompano Beach	FL	821,250	746,591
Pinellas County	St. Petersburg	FL	1,095,000	995,455
McKay Bay	Tampa	FL	365,000	331,818
Palm Beach County	West Palm Beach	FL	730,000	663,636
Savannah	Savannah	GA	182,500	165,909
Honolulu	Honolulu	HI	788,400	716,727
Honolulu (Waipahu) ^a	Honolulu	HI	219,000	199,091
Ames ^a	Ames	IA	73,000	66,364
Burley (Cassia County)	Burley	ID	18,250	16,591
Chicago NW	Chicago	IL	584,000	530,909
Indianapolis RRF	Indianapolis	IN	862,130	783,755
Louisville ^a	Louisville	KY	365,000	331,818
Springfield RRF	Agawam	MA	131,400	119,455

^a No longer operational.

Table A-9 (continued)
Existing MWC Facilities (as of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Fall River ^a	Fall River	MA	219,000	199,091
Framingham ^a	Framingham	MA	182,500	165,909
Haverhill	Haverhill	MA	602,250	547,500
Lawrence	Lawrence	MA	259,150	235,591
Millbury	Millbury	MA	547,500	497,727
North Andover RESCO	North Andover	MA	547,500	497,727
Pittsfield RRF	Pittsfield	MA	131,400	119,455
Rochester (SEMASS)	Rochester	MA	657,000	597,273
Saugus RESCO	Saugus	MA	547,500	497,727
Springfield ^a	Springfield	MA	131,400	119,455
Harford County	Aberdeen	MD	131,400	119,455
Baltimore (Pulaski)	Baltimore	MD	438,000	398,182
Baltimore (RESCO)	Baltimore	MD	821,250	746,591
Biddeford	Biddeford	ME	219,000	199,091
Aroostook County ^a	Frenchville	ME	18,250	16,591
Harpwell	Harpwell	ME	5,110	4,645
Penobscot (Orrington)	Orrington	ME	262,800	238,909
Portland	Portland	ME	182,500	165,909
Grosse Point Clinton	Clinton Township	MI	219,000	199,091
Greater Detroit Resource Recovery Authority	Detroit	MI	1,204,500	1,095,000
Fisher Guide Division ^a	Detroit	MI	36,500	33,182
Kent County District Waste to Energy Facility	Grand Rapids	MI	228,125	207,386
Jackson County	Jackson	MI	73,000	66,364
SE Oakland County ^a	Madison Heights	MI	219,000	199,091
Central Wayne County Sanitation Authority	Inkster	MI	36,500	16,590

^a No longer operational.

Table A-9 (continued)
Existing MWC Facilities (as of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Alexandria	Alexandria	MN	26,280	23,891
Duluth	Duluth	MN	146,000	132,727
Anoka County (Elk River)	Elk River	MN	547,500	497,727
Fergus Falls	Fergus Falls	MN	34,310	31,191
Polk County	Fosston	MN	29,200	26,545
Mankato	Mankato	MN	262,800	238,909
Hennepin County	Minneapolis	MN	438,000	398,182
Perham	Perham	MN	41,610	37,827
Red Wing	Red Wing	MN	262,800	238,909
Red Wing	Red Wing	MN	26,280	23,891
Olmstead County	Rochester	MN	73,000	66,364
Savage	Scott	MN	21,900	19,909
Thief River Falls (Hospital)	Thief River Falls	MN	1,825	1,659
Thief River Falls (Hospital)	Thief River Falls	MN	36,500	33,182
Ft. Leonard Wood ^a	Ft. Leonard Wood	MO	28,470	25,882
Pascagoula	Moss Point	MS	54,750	49,773
Livingston (Park County)	Livingston	MT	26,280	23,891
Mecklenburg Co.	Charlotte	NC	85,775	77,977
Cherry Point Marine St.	Cherry Point	NC	365	332
RTP/NIEHS	RTP	NC	14,600	13,273
New Hanover County	Wilmington	NC	73,000	66,364
New Hanover County	Wilmington	NC	91,250	82,955
Wrightsville Beach ^a	Wrightsville Beach	NC	18,250	16,591
Auburn	Auburn	NH	1,825	1,659
Candia	Candia	NH	5,475	4,977
Canterbury ^a	Canterbury	NH	3,650	3,318
Claremont	Claremont	NH	73,000	66,364

^a No longer operational.

Table A-9 (continued)
Existing MWC Facilities (as of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Concord	Concord	NH	182,500	165,909
Durham	Durham	NH	39,420	35,836
Lincoln	Lincoln	NH	8,760	7,964
Litchfield	Litchfield	NH	8,030	7,300
Nottingham	Nottingham	NH	2,920	2,655
Pelham	Pelham	NH	8,760	7,964
Pittsfield ^a	Pittsfield	NH	17,520	15,927
Plymouth	Plymouth	NH	5,840	5,309
Wilton	Wilton	NH	10,950	9,955
Wolfeboro	Wolfeboro	NH	5,840	5,309
Atlantic County Jail	Atlantic County Jail	NJ	5,110	4,645
Camden County	Camden	NJ	383,250	348,409
Fort Dix	Fort Dix	NJ	29,200	26,545
Essex County	Newark	NJ	831,105	755,550
Warren County	Oxford Township	NJ	146,000	132,727
Union County	Rahway	NJ	525,600	477,818
Gloucester County	West Deptford	NJ	209,875	190,795
Albany ^a	Albany	NY	219,000	199,091
Babylon	Babylon	NY	273,750	248,864
SW Brooklyn ^a	Brooklyn	NY	350,400	318,545
Cattaraugus ^a	Cuba	NY	40,880	37,164
Ellis Island	Ellis Island	NY	4,380	3,982
Fire Island	Fire Island	NY	4,380	3,982
Glen Cove ^a	Glen Cove	NY	91,250	82,955
Hempstead	Hempstead	NY	846,435	769,486
Huntington	Huntington	NY	273,750	248,862
Islip	Islip	NY	189,070	171,884

^a No longer operational.

Table A-9 (continued)
Existing MWC Facilities (as of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Liberty Island	Liberty Island	NY	4,380	3,982
Long Beach	Long Beach	NY	73,000	66,364
Betts Ave (NY City) ^a	New York	NY	365,000	331,818
Niagara Falls-Occidental	Niagara Falls	NY	730,000	663,636
Oswego County	Oswego County	NY	73,000	66,364
Westchester County	Peekskill	NY	821,250	746,591
Dutchess County	Poughkeepsie	NY	146,000	132,727
Oneida County	Rome	NY	73,000	66,364
Akron	Akron	OH	365,000	331,818
Columbus ^a	Columbus	OH	730,000	663,636
South Montgomery County	Dayton	OH	328,500	298,636
North Montgomery County	Dayton	OH	328,500	298,636
Euclid ^a	Euclid	OH	73,000	66,364
Miami	Miami	OK	38,325	34,841
Poteau	Poteau	OK	9,125	8,295
Tulsa	Tulsa	OK	273,750	248,864
Wilburton ^a	Wilburton	OK	6,570	5,973
Bendon	Benton	OR	9,490	8,627
Marion County	Brooks	OR	200,750	182,500
Coos Bay	Coos Bay	OR	36,500	33,182
Courthouse-Coquille	Coquille	OR	54,750	49,773
Courthouse-Coquille	Coquille	OR	4,745	4,314
Delaware County (Chester)	Chester	PA	981,120	891,927
Lancaster County	Conoy Township	PA	438,000	398,182
Westmoreland County	Greensburg	PA	18,250	16,591
Harrisburg	Harrisburg	PA	262,800	238,909
York County	Manchester Township	PA	490,560	445,964

^a No longer operational.

Table A-9 (continued)
Existing MWC Facilities (as of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Philadelphia EC ^a	Philadelphia	PA	273,750	248,864
Philadelphia NW ^a	Philadelphia	PA	273,750	248,864
Montgomery County	Plymouth Township	PA	438,000	398,182
Charleston County	Charleston	SC	219,000	199,091
Hampton	Hampton	SC	87,600	79,636
Davidson County	Davidson County	TN	67,525	61,386
Dyersburg ^a	Dyersburg	TN	36,500	33,182
Galletin	Galletin	TN	73,000	66,364
Lewisburg ^a	Lewisburg	TN	21,900	19,909
Nashville	Nashville	TN	408,800	371,636
Carthage City	Carthage	TX	14,600	13,273
Center	Center	TX	14,600	13,273
Cleburne	Cleburne	TX	41,610	37,827
Gatesville (Prison) ^a	Gatesville	TX	7,300	6,636
Walker County ^a	Huntsville	TX	9,125	8,295
Walker County (Prison) ^a	Huntsville	TX	9,125	8,295
Grimes County ^a	Navasota	TX	9,125	8,295
Anderson County	Palestine	TX	9,125	8,295
Quitman	Quitman	TX	7,300	6,636
Waxahachie ^a	Waxahachie	TX	18,250	16,591
Davis County	Layton	UT	146,000	132,727
Alexandria	Alexandria	VA	355,875	323,523
Arlington (Pentagon)	Arlington	VA	18,250	16,591
Galax ^a	Galax	VA	20,440	18,582
Hampton	Hampton	VA	73,000	66,364
Harrisonburg	Harrisonburg	VA	36,500	33,182
Fairfax County	Lorton	VA	1,095,000	995,455

^a No longer operational.

Table A-9 (continued)
Existing MWC Facilities (as of December, 1991)

Facility	City	State	Capacity	
			tons/yr	Mg/yr
Norfolk Navy Yard	Norfolk	VA	730,000	663,636
Norfolk Naval Station ^a	Norfolk	VA	131,400	119,455
Salem ^a	Salem	VA	36,500	33,182
Readsboro	Readsboro	VT	4,745	4,314
Rutland ^a	Rutland	VT	87,600	79,636
Stamford	Stamford	VT	3,650	3,318
Bellingham	Ferndale	WA	36,500	33,182
Fort Lewis	Fort Lewis	WA	43,800	39,818
Friday Harbor	Friday Harbor	WA	4,380	3,982
Skagit County	Mt. Vernon	WA	64,970	59,064
Spokane	Spokane	WA	292,000	265,455
Tacoma	Tacoma	WA	109,500	99,545
Barron Co.	Almena	WI	29,200	26,545
La Crosse County ^a	La Crosse	WI	146,000	132,727
Madison (Power Plant) ^a	Madison	WI	43,800	39,818
Muscoda ^a	Muscoda	WI	43,800	39,818
Waukesha ^a	Waukesha	WI	64,240	58,400
Total			41,729,720	37,936,109

Source: Memorandum from Fenn, D., and K. Nebel, Radian Corporation, to Stevenson, W., U. S. Environmental Protection Agency. March 9, 1992.

^a No longer operational.

Table A-10
Mercury Emissions From MWCs by Combustor Type For 1994

Combustor type	Control status	Process rate, ^a 10 ⁶ Mg/yr	Uncontrolled emission factor, g/Mg	Control efficiency, %	Annual Emissions	
					Mg/yr	Tons/yr
Mass burn	U	0.43	2.82	0	1.2	1.3
Mass burn	SD	10.8	2.82	50	15.2	16.7
Mass burn	DSI	0.85	2.82	50	1.2	1.3
Mass burn	ESP	6.3	2.82	0	17.8	19.6
Mass burn	CI	1.5	2.82	85	0.6	0.7
RDF	SD	8.1	2.77	50	11.2	12.3
Modular	ESP	1.0	2.82	0	2.8	3.1
Modular	SD	0.73	2.82	50	1.0	1.1
Unknown	U	0.03	2.82	0	0.07	0.08
Total					51.1	56.2

Key:

- SD = Spray dryer with either ESP or fabric filter
- ESP = Electrostatic precipitator
- DSI = Duct sorbent injection with either ESP or fabric filter
- U = Uncontrolled
- CI = Activated carbon injection

^a Annual totals assume that plants operate at 85% of design capacity.

Basis of Input Data for EPA's Emissions Calculations

- Under the assumption that ESP's provide essentially no control, the facility-average concentrations at 7 percent oxygen for uncontrolled and ESP-controlled mass burn (including modular) and RDF systems were averaged to obtain the following "typical" concentrations:
 Mass Burn - 696 µg/dscm
 RDF - 561 µg/dscm
- The F-factor for municipal waste combustors was assumed to be 0.257×10^{-6} dscm/J at 0 percent oxygen and the heating values were assumed to be 10,500 kJ/kg for MSW and 12,800 kJ/kg for RDF. The F-factor was converted from 0 percent oxygen to 7 percent oxygen (at which concentrations are based) using a factor of 1.5.
- Spray dryer or dry sorbent injection systems combined with fabric filters or ESP's and wet scrubber systems are assumed to achieve 50 percent removal. No other control measures achieve appreciable mercury control.

Calculations

Uncontrolled Emission Factors

- Mass burn/modular - $696 \mu\text{g/dscm} \times 0.257 \times 10^{-6} \text{ dscm/J} \times 10,500 \text{ kJ/kg} \times 1.5 = 2.82 \text{ g/Mg}$
- RDF - $561 \mu\text{g/dscm} \times 0.257 \times 10^{-6} \text{ dscm/J} \times 12,800 \text{ kJ/kg} \times 1.5 = 2.77 \text{ g/Mg}$

Controlled Emissions

Annual Emissions

$$= \text{Process rate} \times \text{emission factor} \times \frac{(100 - \text{efficiency})}{100}$$

Source: Locating and Estimating Air Emissions from Sources of Mercury and Mercury Compounds. U.S. Environmental Protection Agency. July 1993.

Industry Estimates for MWC Mercury Emissions

Industry estimates of mercury emissions for the MWC source category are 40 Mg/year (44 tons/year) (Kiser, 1991). The differences between EPA's emission estimates and the industry estimates stem from essentially three differences in the data assumed for the emissions calculations. The basis for the industry estimate is as follows.

Basis of Input Data for Industry's Emissions Calculations

1. The average level of mercury from MWCs was assumed to be 500 ug/dscm compared to 696 ug/dscm that EPA used for mass-burn combustors and 561 ug/dscm that EPA used for RDF combustors.
2. Industry estimates attribute a 10 percent reduction in mercury for MWCs equipped with an ESP. EPA estimates do not credit any mercury removal to MWCs equipped with ESPs.
3. Industry estimates attribute an 80 percent reduction in mercury for MWCs equipped with a spray dryer and firing RDF. EPA credits a 50 percent reduction for these facilities.

Table A-11
MWI Population By State

State	Commercial units		Onsite units		
	No.	Capacity range, lb/hr	No.	Capacity range, lb/hr	Facilities included
Alabama	2	Up to 60,000 lb/wk	250	ND	H,N
Alaska	1	ND	10 to 12	ND	H
Arizona	1	ND	97	ND	All
Arkansas	1	ND	150	ND	ND
California	10	ND	157	ND	H,N,O
Colorado	2	225	46	13-1,000	H
Connecticut	1 ^a	1,200	44 ^b	20-1,500	All
Delaware	0	NA	20	ND	H,V
Florida	12	300-3,750	273	ND	H,F,V,A,L,O
Georgia	1	ND	ND	ND	
Hawaii	0	NA	6 or 7	100-1,000	H
Idaho	ND	ND	20-25 permitted	ND	H
Illinois	2	ND	259	2-1,500	H,F,V,A,L,O
Indiana	4	ND	91	Most <7 tons/d	H
Iowa	0	NA	ND	≤125	H
Kansas	Some	ND	ND	ND	
Kentucky	1	1,500	ND	ND	
Louisiana	1	1,500	100-125	<500 to >1,000 (most <500)	H
Maine	1	150	22	20-1,000	H,N,L
Maryland	ND	ND	121	ND	All
Massachusetts	3	350-1,200	200	3-1,875	All
Michigan	1	720	160	ND	H,N
Minnesota	0	NA	145	50-1,250	H
Mississippi	0	NA	125-175	ND	All
Missouri	2	ND	100	Most <500	H
Montana	ND	ND	<50	ND	ND
Nebraska	0	NA	80	ND	H
Nevada	0	NA	17	40-360	H
New Hampshire	1	ND	27	Most 75-150	H
New Jersey	1	5,000	154	20-1,560	All
New Mexico	3	75-1,000	31	25-360	H,F,V,A,L
New York	2	1,950	599	3-3,000	All
North Carolina	7	150-3,250	29	60-2,100	H,L
North Dakota	0	NA	50	Most <500	H
Ohio	15	500-1,700	125	25-2,500	H,F,V,A,L,O
Oklahoma	1	ND	93	ND	H,N,L
Oregon	2	200-1,000	31	25-750	H
Pennsylvania	8	ND	186	ND	H
Rhode Island	1	975	11	50-1,500	H
South Carolina	3	6,250	70	<500 to 1,000	H

^a Closed as of 1994.

^b Twenty-two facilities as of 1994.

Table A-11 (continued)
MWI Population By State

State	Commercial units		Onsite units		
	No.	Capacity range, lb/hr	No.	Capacity range, lb/hr	Facilities included
South Dakota	0	NA	30	ND	H
Tennessee	≥1	ND	126	ND	H
Texas	2	ND	ND	ND	
Utah	0	NA	20	<500	H
Vermont	0	ND	9	ND	H
Virginia	1 or 2	ND	ND	ND	
Washington	6	1,600-7,500	137	40-825	All
West Virginia	ND	ND	50	ND	H
Wisconsin	4	ND	ND	ND	H
Wyoming	0	-NA	30	ND	

KEY:

General

ND = no data

NA = not applicable

Facility types

H = hospital/medical centers

F = funeral homes

V = veterinaries

A = animal shelters

L = laboratories, clinical and research

N = nursing homes

O = other/unidentified facilities

All = all MWI facilities (using current broad working definition)

Source: U.S. EPA, 1994. Medical Waste Incinerators--Background Information for Proposed Standards and Guidelines: Industry Profile Report for New and Existing Facilities. EPA-453/R-94-042a. Research Triangle Park, NC.

Table A-12
Mercury Emissions From Model Medical Waste Incinerators

Model Number	Type of Unit	Operating Hours/year	Number of Units	Waste feed capacity		Annual Emissions	
				Mg/yr	Tons/yr	Mg/yr	Tons/yr
1	Continuous	7,760	154	0.68	0.75	14.93	16.47
2	Continuous	3,564	182	0.45	0.5	5.40	5.96
3	Intermittent	4,212	171	0.68	0.75	9.00	9.93
4	Intermittent	4,212	742	0.27	0.3	15.62	17.23
5	Intermittent	3,588	2,097	0.09	0.1	12.54	13.83
6	Batch	3,520	335	0.23	0.25	1.05	1.16
7	Pathological	2,964	1,305	0.09	0.1	0.12	0.13
Total			4,986	2.5	2.75	58.7	64.71

Table A-13
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Anchorage	Anchorage	AK	401.5	365
Petersburg	Petersburg	AK	N/A	N/A
Wrangell	Wrangell	AK	73	66.4
Barstow	Barstow	CA	912.5	829.5
Lake Arrowhead	Lake Arrowhead	CA	1,752	1,592.7
Martinez	Martinez	CA	N/A	N/A
Palo Alto	Palo Alto	CA	2,190	1,990.9
Redwood City	Redwood City	CA	N/A	N/A
Sacramento	Sacramento	CA	2,628	2,389.1
San Mateo	San Mateo	CA	1,788.5	1,625.9
South Lake Tahoe	South Lake Tahoe	CA	292	265.5
Tahoe Truckee	Truckee	CA	N/A	N/A
Central Contra Costa	Walnut Creek	CA	16,060	14,600
Yosemite	Yosemite National Park	CA	1,241	1,128.2
Mattabassett	Cromwell	CT	4,562.5	4,147.7
Mattabassett	Cromwell	CT	N/A	N/A
Hartford WPCF	Hartford	CT	122,640	111,490.9
New Canaan ^a	New Canaan	CT	14,198.5	12,907.7
East Shore WPCF	New Haven	CT	10,658	9,689.1
New London WPCF	New London	CT	18,907	17,188.2
Norwalk ^a	Norwalk	CT	13,140	11,945.5
Stamford	Stamford	CT	34,565.5	31,423.2
Waterbury WPCF ^a	Waterbury	CT	2,372.5	2,156.8
West Haven	West Haven	CT	N/A	N/A
Willimantic WPCF	Willimantic	CT	N/A	N/A
Jacksonville	Jacksonville	FL	10,366	9,423.6
Pensacola WWTP	Pensacola	FL	N/A	N/A
R.M. Clayton WWTP	Atlanta	GA	N/A	N/A

^a Not operational as of 1994.

Table A-13 (continued)
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Atlanta (Utoy)	Atlanta	GA	2,956.5	2,687.7
Atlanta (Bolton Rd)	Atlanta	GA	47,304	43,003.6
Decatur	Decatur	GA	16,352	14,865.5
Gainesville	Gainesville	GA	2,007.5	1,825
Cobb County	Marietta	GA	7,227	6,570
Savannah	Savannah	GA	4,380	3,981.8
San Island WWTF	Honolulu	HI	9,453.5	8,594.1
Honouliuli WWTP	Honouliuli	HI	N/A	N/A
Oahu	Oahu	HI	N/A	N/A
Cedar Rapids WPCF	Cedar Rapids	IA	8,869.5	8,063.2
Davenport	Davenport	IA	12,994	11,812.7
Dubuque	Dubuque	IA	20,440	18,581.8
Decatur STP	Decatur	IL	N/A	N/A
Indianapolis-Belmont	Indianapolis	IN	132,458.5	120,416.8
Kansas City	Kansas City	KS	6,570	5,972.7
Kaw Point	Kansas City	KS	14,600	13,272.7
Mission Township STP	Mission	KS	N/A	N/A
Turkey Creek MSD #1	Shawnee Mission	KS	6,497	5,906.4
Cynthiana	Cynthiana	KY	N/A	N/A
Kenton County	Kenton	KY	N/A	N/A
New Orleans West Bank STP	Algiers	LA	N/A	N/A
Lake Charles	Lake Charles	LA	2,190	1,990.9
Lake Charles-Plant B	Lake Charles	LA	N/A	N/A
Lake Charles-Plant C	Lake Charles	LA	N/A	N/A
East Bank STP #2	Lake Charles	LA	14,965	13,604.5
Natchitoches	Natchitoches	LA	N/A	N/A
East Bank STP #1	New Orleans	LA	10,950	9,954.5
Attleboro Advanced WTF	Attleboro	MA	N/A	N/A
Chicopee	Chicopee	MA	2,628	2,389.1

Table A-13 (continued)
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Chicopee	Chicopee	MA	2,628	2,389.1
Fall River	Fall River	MA	N/A	N/A
Fitchburg East WWTP	Fitchburg	MA	14,198.5	12,907.7
Lynn	Lynn	MA	N/A	N/A
Upper Blackstone WWTP	Millbury	MA	12,811.5	11,646.8
New Bedford WWTP	New Bedford	MA	5,913	5,375.5
Greater Lawrence SD WWTP	North Andover	MA	33,142	30,129.1
Annapolis City SIP	Annapolis	MD	N/A	N/A
Patapsco	Baltimore	MD	35,916	32,650.9
Ocean City	Ocean City	MD	2,920	2,654.5
Cox Creek WWTP	Riviera Beach	MD	N/A	N/A
Ann Arbor	Ann Arbor	MI	19,710	17,918.2
Battle Creek	Battle Creek	MI	N/A	N/A
Bay City STP	Bay City	MI	1,168	1,061.8
Bay County STP	Bay County	MI	N/A	N/A
Detroit (1)	Detroit	MI	148,920	135,381.8
Detroit (2)	Detroit	MI	245,937	223,579.1
East Lansing	East Lansing	MI	11,826	10,750.9
Grand Rapids	Grand Rapids	MI	11,826	10,750.9
Kalamazoo WWTP	Kalamazoo	MI	17,520	15,927.3
Lansing WWTP	Lansing	MI	N/A	N/A
Niles WWTP	Niles	MI	N/A	N/A
Owosso WWTP	Owosso	MI	N/A	N/A
Pontiac STP	Pontiac	MI	23,652	21,501.8
Port Huron	Port Huron	MI	2,774	2,521.8
Trenton WWTP	Trenton	MI	N/A	N/A
Warren	Warren	MI	9,453.5	8,594.1
Wyandotte STP	Wyandotte	MI	88,768	80,698.2

Table A-13 (continued)
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Ypsilanti Community WWTP	Ypsilanti	MI	19,710	17,918.2
Duluth	Duluth	MN	12,410	11,281.8
Metropolitan TP	St. Paul	MN	283,824	258,021.8
Seneca TP	St. Paul	MN	7,081	6,437.3
Independence	Independence	MO	3,540.5	3,218.6
Kansas City	Kansas City	MO	16,571	15,064.5
Little Blue Valley	Little Blue	MO	N/A	N/A
St. Louis (Lenay STP)	St. Louis	MO	53,217	48,379.1
St. Louis (Bissel Point STP)	St. Louis	MO	118,260	107,509.1
Greensboro	Greensboro	NC	16,571	15,064.5
Manchester WWTP	Manchester	NC	N/A	N/A
Rocky Mount	Rocky Mount	NC	2,737.5	2,488.6
Shelby	Shelby	NC	5,913	5,375.5
Lebanon WWTP	Lebanon	NH	2,628	2,389.1
Merrimack WWTP	Merrimack	NH	N/A	N/A
Atlantic City	Atlantic City	NJ	9,453.5	8,594.1
Gloucester Township	Blackwood	NJ	3,504	3,185.5
Somerset Raritan Valley Authority	Bridgewater	NJ	5,110	4,645.5
West Side STP	Jersey City	NJ	5,037	4,579.1
Two Bridges	Lincoln Park	NJ	24,090	21,900
Parsippany	Parsippany	NJ	28,397	25,815.5
Rockaway Valley	Parsippany-Troy Hills	NJ	N/A	N/A
Stony Brook RSA STP #1	Princeton	NJ	14,417.5	13,106.8
Bayshore Regional Sewer Authority	Union Beach	NJ	10950	9,954.5
NW Bergen County Utilities	Waldwick	NJ	6,570	5,972.7
Wayne	Wayne	NJ	35,040	31,854.5

Table A-13 (continued)
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Mountain View Sewer Authority	Wayne Township	NJ	N/A	N/A
Round Hill	Round Hill Village	NV	3,285	2,986.4
Douglas County SID #1 WWTF	Zephyr Cove-Round Hill Village	NV	1,825	1,659.1
Albany (North)	Albany	NY	47,304	43,003.6
Albany (South)	Albany	NY	33,507	30,460.9
Amherst	Amherst	NY	N/A	N/A
Arlington	Arlington	NY	3,066	2,787.3
Auburn	Auburn	NY	14,782.5	13,438.6
Bath	Bath	NY	N/A	N/A
Beacon WPCP	Beacon	NY	3,540.5	3,218.6
Birds Island STP	Buffalo	NY	67,014	60,921.8
Southtowns Advanced WWTF	Buffalo	NY	15,768	14,334.5
Dunkirk STP	Dunkirk	NY	N/A	N/A
Numburg	Erie County	NY	105,120	95,563.6
Glen Cove	Glen Cove	NY	9,125	8,295.5
Glens Falls	Glens Falls	NY	N/A	N/A
NW Quadrant TP	Greece	NY	N/A	N/A
Little Falls	Little Falls	NY	1,423.5	1,294.1
Mamaroneck	Mamaroneck	NY	N/A	N/A
New Rochelle SD STP	New Rochelle	NY	N/A	N/A
Niagra County	Niagara County	NY	N/A	N/A
Utica	Oneida County	NY	21,900	19,909.1
Orangetown DPW	Orangetown	NY	6,132	5,574.5
East STP	Oswego	NY	N/A	N/A
West STP	Oswego	NY	N/A	N/A
Port Chester SDSTP	Port Chester	NY	N/A	N/A
Port Washington	Port Washington	NY	N/A	N/A
Gates Chile Ogden STP	Rochester	NY	13,140	11,945.5

Table A-13 (continued)
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Rochester (NW Quad)	Rochester	NY	17,520	15,927.3
Frank E. Van Lare WWTP	Rochester	NY	26,280	23,890.9
Saratoga	Saratoga	NY	N/A	N/A
Schenectady STP	Schenectady	NY	51,100	46,454.5
Two Mile Creek STP	Tonawanda	NY	N/A	N/A
Watertown	Watertown	NY	N/A	N/A
Watertown	Watertown	NY	7,665	6,968.2
Akron WWTP	Akron	OH	14,162	12,874.5
Canton WWTP	Canton	OH	18,250	16,590.9
Little Miami WWTP	Cincinnati	OH	N/A	N/A
Cincinnati (Millcreek)	Cincinnati	OH	61,466	55,878.2
Cleveland (Southerly WWTP)	Cleveland	OH	94,608	86,007.3
Cleveland (Westerly STP)	Cleveland	OH	70,956	64,505.5
Columbus (South)	Columbus	OH	16,571	15,064.5
Columbus (Jackson Pike WWTP)	Columbus	OH	14,198.5	12,907.7
Euclid WWTP	Euclid	OH	7,884	7,167.3
Warren County	Franklin	OH	N/A	N/A
Lorain	Lorain	OH	N/A	N/A
Willoughby-Eastlake WWTP	Willoughby	OH	7,665	6,968.2
Youngstown WWTP	Youngstown	OH	14,782.5	13,438.6
Tigard	Tigard	OR	5,475	4,977.3
Ambridge STP	Ambridge	PA	N/A	N/A
Kiski Valley WPCA	Appollo	PA	49,676.5	45,160.5
Bridgeport SIP	Bridgeport	PA	N/A	N/A
Delcora-Chester STP	Chester	PA	7,081	6,437.3
Hatfield Township STP	Colmer	PA	2,080.5	1,891.4

Table A-13 (continued)
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Duryea	Duryea	PA	9,453.5	8,594.1
Erie	Erie	PA	49,275	44,795.5
Hershey	Hershey	PA	14,782.5	13,438.6
City of Johnstown	Johnstown	PA	2,956.5	2,687.7
Cumberland City	Lemoyne	PA	N/A	N/A
E. Morristown Plymouth TP	Norristown	PA	N/A	N/A
Upper Gwynedd	North Wales	PA	401.5	365
Lower Lackawanna STP	Old Forge	PA	N/A	N/A
Alcosan WWTP	Pittsburgh	PA	N/A	N/A
Tyrone	Tyrone	PA	1,861.5	1,692.3
Trout Run WPCC	Upper Merion Township	PA	N/A	N/A
Hazeltown	West Hazeltown	PA	1,642.5	1,493.2
Wyoming Valley Sanitation Authority	Wilkes-Barre	PA	11,826	10,750.9
Upper Moreland-Hatboro TP	Willow Grove	PA	N/A	N/A
York	York	PA	14,162	12,874.5
Cranston	Cranston	RI	7,446	6,769.1
Harrisburg	Harrisburg	RI	49,421	44,928.2
Providence	Providence	RI	N/A	N/A
Charleston	Charleston	SC	11,826	10,750.9
Columbia	Columbia	SC	2,993	2,720.9
North Charleston	North Charleston	SC	N/A	N/A
Bristol	Bristol	TN	5,913	5,375.5
Maryville Regional STP	Maryville	TN	4,745	4,313.6
Central WWTP	Nashville	TN	33,580	30,527.3
Newport	Newport	TN	2,847	2,588.2
Alexandria STP	Alexandria	VA	N/A	N/A
Arlington COWPCP	Arlington	VA	12,702	11,547.3
Fairfax	Fairfax	VA	23,652	21,501.8

Table A-13 (continued)
U.S. Sewage Sludge Incinerators

Facility	Location	State	Capacity	
			Ton/yr	Mg/yr
Fairfax (Lower Potomac STP)	Fairfax	VA	33,142	30,129.1
Hopewell	Hopewell	VA	2,956.5	2,687.7
Boat Harbor	Newport News	VA	12,300.5	11,182.3
Lamberta Point WPCF	Norfolk	VA	21,133.5	19,212.3
Army Base WWTP (Hampton Rds.)	Norfolk	VA	9,307.5	8,461.4
Chesapeake-Elizabeth WPCF	Virginia Beach	VA	8,322	7,565.5
Williamsburg WPCF	Williamsburg	VA	20,330.5	18,482.3
Potomac River STP	Woodbridge	VA	N/A	N/A
Edmonds	Edmonds	WA	584	530.9
Lynnwood	Lynnwood	WA	255.5	232.3
Vancouver	Vancouver	WA	12,410	11,281.8
Brookfield STP	Brookfield	WI	1,423.5	1,294.1
Green Bay WWTP	Green Bay	WI	31,937.5	29,034.1
Milwaukee	Milwaukee	WI	2,591.5	2,355.9
Clarksburg	Clarksburg	WV	N/A	N/A
Huntington	Huntington	WV	N/A	N/A
Total			3,208,240.5	2,916,582.3

Source: Locating and Estimating Air Toxic Emissions from Sewage Sludge Incinerators. U.S. EPA. EPA-450/2-90-009. May 1990.

APPENDIX B

**MERCURY REMOVAL CAPABILITIES OF
PARTICULATE MATTER AND ACID GAS CONTROLS FOR UTILITIES**



APPENDIX B

MERCURY REMOVAL CAPABILITIES OF PARTICULATE MATTER AND ACID GAS CONTROLS FOR UTILITIES

Existing air pollution control devices (APCDs) on utilities typically control either particulate matter (PM) or sulfur dioxide (SO₂) emissions, or both. Nitrogen oxides may be controlled by an APCD, but are usually controlled by combustion modification. Generally, a wet scrubber is used to control SO₂ emissions only, while a dry scrubber can control SO₂ emissions and PM because it is usually built with a downstream PM collector. Devices that control PM only include fabric filters (FFs), electrostatic precipitators (ESPs), mechanical collectors (cyclones), and venturi scrubbers.

Mercury, however, is not well controlled by particulate matter APCDs because mercury is emitted as a mixture of solid and gaseous forms.

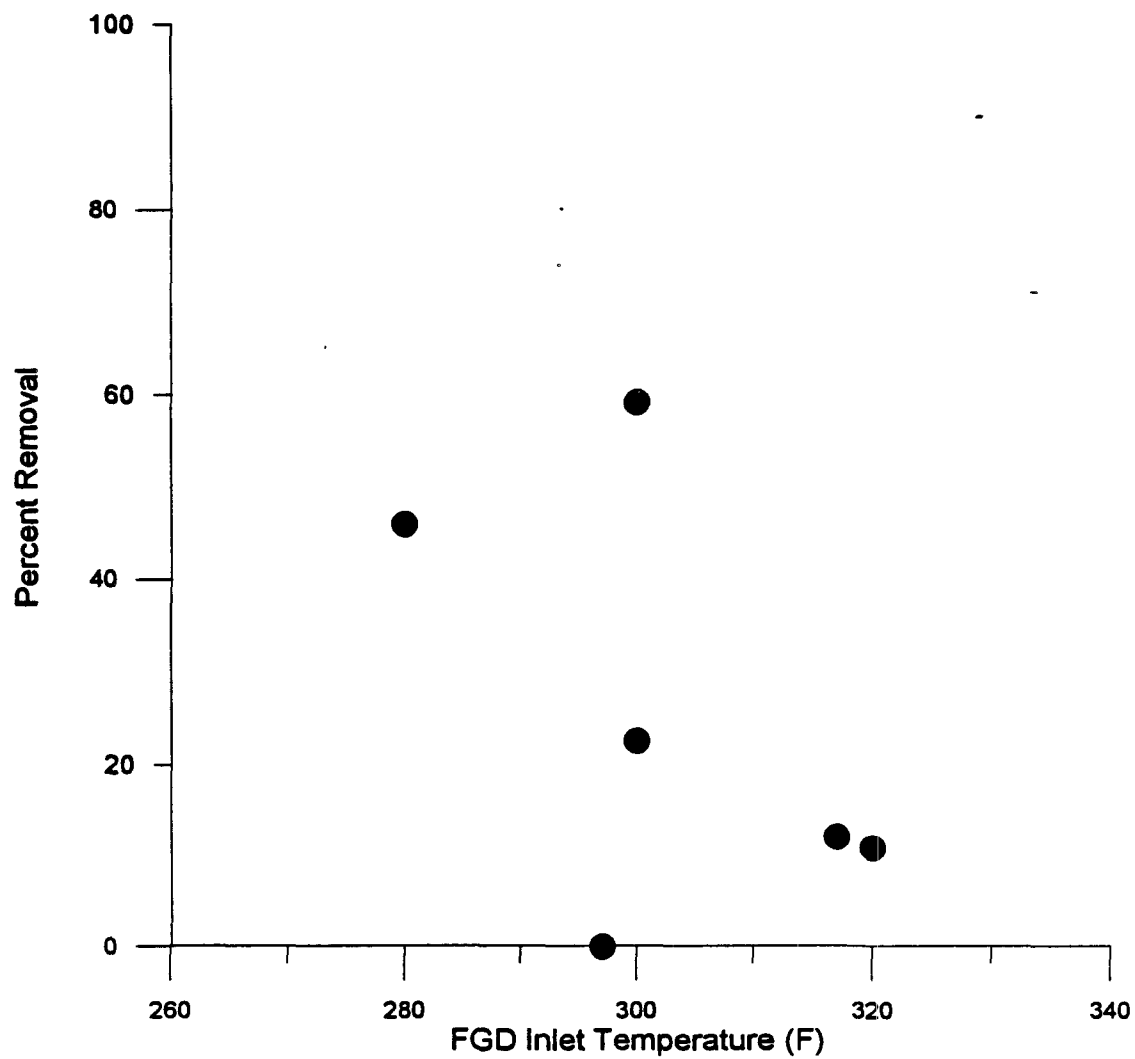
Mercury removal effectiveness is shown in this appendix as percent removal. Percent removal is equivalent to one minus the emission modification factor (EMF). For example, a 17.3 percent removal indicates an EMF of 0.827 or that 17.3 percent of the total mercury has been collected by that type of control device. Calculation of EMF's is described in Section 4.1.1.3. The EMF values are presented in Appendix C.

B.1 Scrubbers

Wet scrubbers or flue gas desulfurization (FGD) units for coal-fired plants are typically used to remove acid gases (mainly SO₂ emissions). Most utility boilers are equipped with an ESP or FF before the wet FGD units to collect PM.

Figure B-1 shows the relationship between mercury removal and the inlet temperature for wet FGD devices. Table B-1 summarizes available test data for FGD units. FGDs have a median mercury removal efficiency of about 17.3 percent, with a range from 0 percent to 59.3 percent removal. The correlation between FGD inlet temperature and mercury removal is difficult to determine. This difficulty is compounded by having only five data sites and two of the five test sites employ flue gas bypasses in their design. A bypass means that part of the flue gas is diverted around the FGD while the majority of the flue gas is treated.

Figure B-1
Removal of Mercury By An FGD (Coal)



**Table B-1
Test Data for FGD Units**

Unit	Control Device	Hg Removal %	Reference
EPRI Site 11	Wet limestone FGD (inlet Hg concentration of 9.9 µg/dscm)	10.87	Radian, 1993a
EPRI Site 12	Wet limestone FGD	0 ^a	Radian, 1993b
NSP Sherburne 1 & 2 Test A	Wet limestone FGD (inlet Hg concentration of 8.1 µg/dscm)	22.63	Interpoll, 1990a
NSP Sherburne 1 & 2 Test B	Wet limestone FGD (inlet Hg concentration of 11.6 µg/dscm)	59.3	Interpoll, 1991
DOE Yates	Wet limestone and jet bubbling reactor FGD (inlet Hg concentration of 6.0 µg/dscm)	45.91	EPRI, 1993a
DOE Coal Creek	Wet lime FGD (inlet Hg concentration of 10.0 µg/dscm)	12.05	Battelle, 1993a
	Median	17.34	
	Mean	25.13	
	Standard deviation	22.85	

^a This unit was re-tested for mercury as part of a ESP/FGD system. Since there was no way of determining which component (the ESP or the FGD) was responsible for any mercury removal, the ESP was given the full credit for removal, as shown in the site 12 ESP data in Table B-4.

B.2 SDA or Dry Scrubbing

A spray dryer adsorber (SDA) process is a dry scrubbing system followed by a particulate control device. A lime/water slurry is sprayed into the flue gas stream and the resulting dry solids are collected by an ESP or an FF.

Figure B-2 shows the relationship between mercury removal and the inlet temperature for the SDA/FF systems. Available SDA data are presented in Table B-2. SDA/FF systems have a median mercury removal efficiency of about 23.9 percent, with a range from 0 percent to 54.5 percent removal.

Figure B-2
**Removal of Mercury By A Spray Dryer Adsorber/
 Fabric Filter (Coal)**

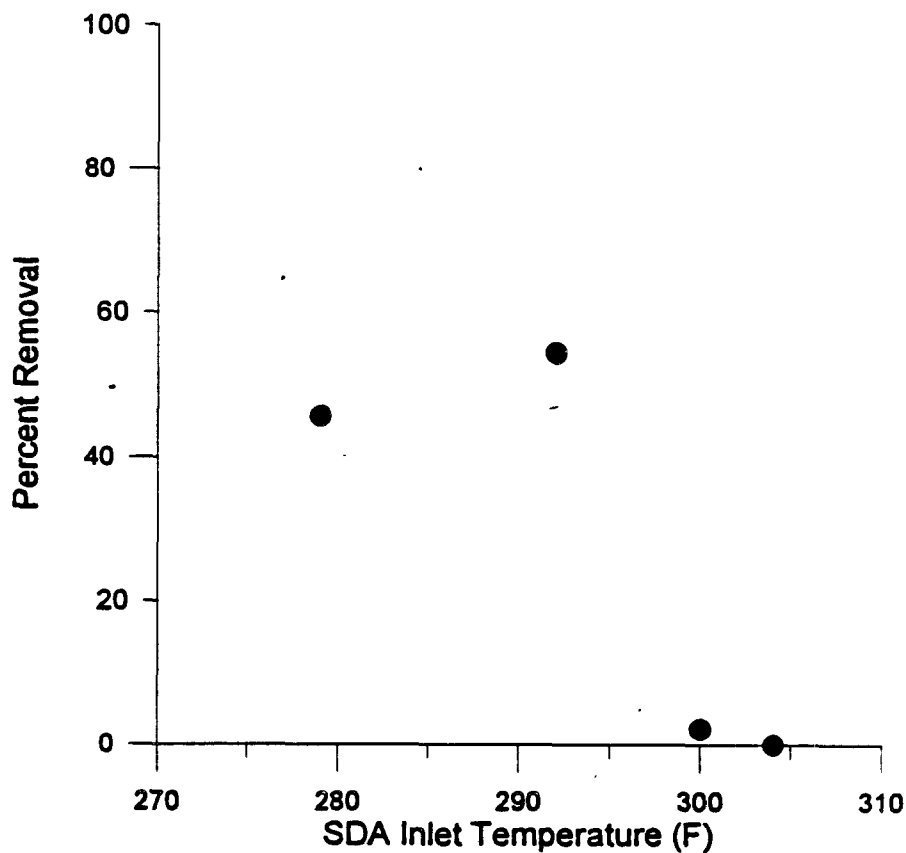


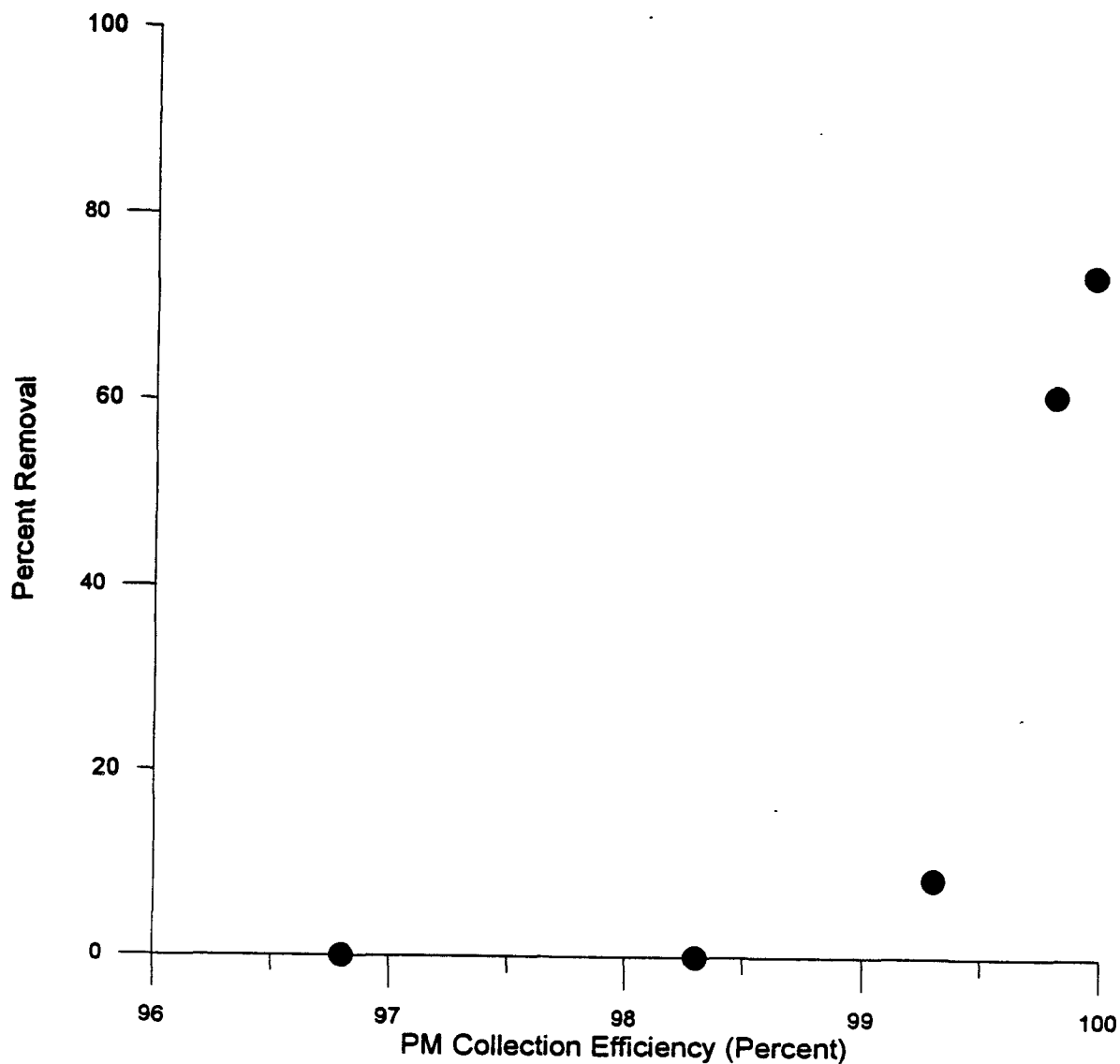
Table B-2
Spray Dryer Adsorption Data

Unit	Control Device	Hg Removal %	Reference
EPRI Site 14	SDA/FF (inlet Hg concentration of 1.0 µg/dscm)	0	Radian, 1993c
DOE Springerville	SDA/FF (inlet Hg concentration of 8.3 µg/dscm)	2.16	Southern Research Institute, 1993a
Sherburne 3 Test A	SDA/FF (inlet Hg concentration of 6.8 µg/dscm)	45.71	Interpoll, 1990b
Sherburne 3 Test B	SDA/FF (inlet Hg concentration of 13.4 µg/dscm)	54.5	Interpoll, 1991
	Median	23.94	
	Mean	25.59	
	Standard deviation	28.54	

B.3 Fabric Filters

Figure B-3 shows the relationship between mercury removal and the PM collection efficiency (percent) for FFs (controlling coal-fired units). Available FF data are presented in Table B-3. Fabric filters have a median mercury removal efficiency of about 8.39 percent, with a range from 0 percent to 73.36 percent removal.

Figure B-3
Removal of Mercury By A FF (Coal)



**Table B-3
Fabric Filter Data**

Unit	Control Device	Hg Removal %	Reference
EPRI Site 13	FF (inlet Hg concentration of 0.3 µg/dscm)	0	Radian, 1993d
EPRI Site 115	FF (inlet Hg concentration of 1.8 µg/dscm)	73.36	Carnot, 1994a
NSP Riverside 6 & 7	FF (inlet Hg concentration of 4.8 µg/dscm)	0	Interpoll, 1992a
DOE Niles #2 w/NO _x	FF (inlet Hg concentration of 25.8 µg/dscm)	8.39	Battelle, 1993b
DOE Boswell	FF (inlet Hg concentration of 6.4 µg/dscm)	60.59	Weston, 1993a
	Median	8.39	
	Mean	28.47	
	Standard deviation	35.61	

B.4 Electrostatic Precipitators

Electrostatic precipitators are the most widely used control device by the fossil fuel-fired electric utility industry. There are two design locations for ESPs, cold-side (CS) and hot-side (HS). Cold-side ESPs are located after the air preheater, thus it is subjected to a lower flue gas temperature than a hot-side ESP which is located before the air preheater.

Figure B-4 shows the relationship between mercury removal and the PM collection efficiency (percent) for cold-side ESPs (controlling coal-fired units). Table B-4 presents available test data for such ESPs. Cold-side ESPs have a median mercury removal efficiency of about 16.2 percent, with a range from 0 percent to 82.4 percent removal.

Figure B-5 shows the relationship between mercury removal and the PM collection efficiency (percent) for hot-side ESPs (controlling coal-fired units). Available test data for hot-side ESPs (controlling coal-fired units) are shown in Table B-5. There was no apparent control of mercury by a hot-side ESP. However, the data were collected from only one emission test where two separate sample runs were analyzed.

Figure B-6 shows the relationship between mercury removal and the PM collection efficiency (percent) for cold-side ESPs (controlling oil-fired units). Table B-6 presents available test data for such configurations. In these emission tests cold-side ESPs (controlling oil-fired units) had a median mercury removal efficiency of about 62.4 percent, with a range from 41.7 percent to 83 percent removal. It should be noted that data for mercury control by cold-side ESPs (controlling oil-fired units) were available from only two test sites.

Figure B-4
Removal of Mercury By Electrostatic Precipitators (Cold-Side, Coal)

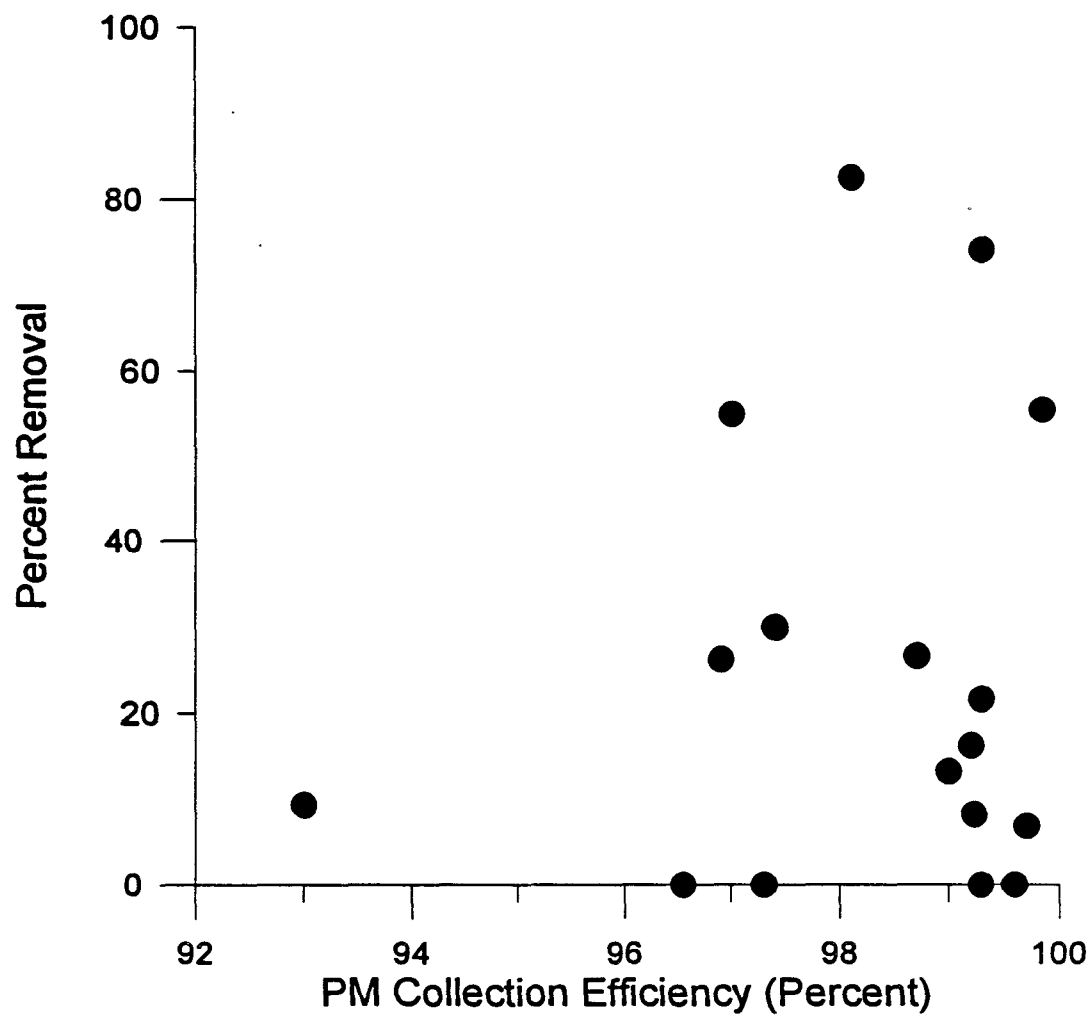


Table B-4
Test Data for Cold-Side Electrostatic Precipitators (Controlling Coal-Fired Units)

Unit	Control Device	Hg Removal %	Reference
EPRI Site 11	ESP, CS (inlet Hg concentration of 3.4 µg/dscm)	0	Radian, 1993a
EPRI Site 12	ESP, CS (inlet Hg concentration of 9.1 µg/dscm)	82.35	Radian, 1993b
EPRI Site 15	ESP, CS (inlet Hg concentration of 4.9 µg/dscm)	0	Radian, 1992a
EPRI Site 102	ESP, CS (inlet Hg concentration of 9.0 µg/dscm)	0	Radian, 1993e
NSP High Bridge 3,4,5,6	ESP, CS (inlet Hg concentration of 4.4 µg/dscm)	6.87	Interpoll, 1992b
NSP High Bridge 1,3,4	ESP, CS (inlet Hg concentration of 5.1 µg/dscm)	8.21	Interpoll, 1992c
NSP Black Dog #2	ESP, CS (inlet Hg concentration of 2.8 µg/dscm)	21.56	Interpoll, 1992d
NSP Riverside #8	ESP, CS (inlet Hg concentration of 2.9 µg/dscm)	0	Interpoll, 1992e
EPRI Site 114 / Test A	ESP, CS (inlet Hg concentration of 10.6 µg/dscm)	29.8	Radian, 1994a
EPRI Site 114 / Test B	ESP, CS (inlet Hg concentration of 10.6 µg/dscm)	16.16	Radian, 1994a
DOE Niles #2	ESP, CS (inlet Hg concentration of 24.7 µg/dscm)	26.55	Battelle, 1993c
DOE Yates	ESP, CS (inlet Hg concentration of 5.9 µg/dscm)	55.23	EPRI, 1993a
DOE Coal Creek	ESP, CS (inlet Hg concentration of 11.0 µg/dscm)	13.15	Battelle, 1993a
EPRI Site 16/OFA/LNO _x Burners	ESP, CS (inlet Hg concentration of 11.5 µg/dscm)	54.8	EPRI, 1993b
EPRI Site 16/OFA	ESP, CS (inlet Hg concentration of 7.6 µg/dscm)	9.38	EPRI, 1993b
DOE Cardinal	ESP, CS (inlet Hg concentration of 2.3 µg/dscm)	73.9	EERC, 1993
DOE Baldwin	ESP, CS (inlet Hg concentration of 7.0 µg/dscm)	26.13	Weston, 1993b
	Median	16.16	
	Mean	24.95	
	Standard deviation	26.33	

Figure B-5
Removal of Mercury By Electrostatic Precipitators (Hot-Side, Coal)

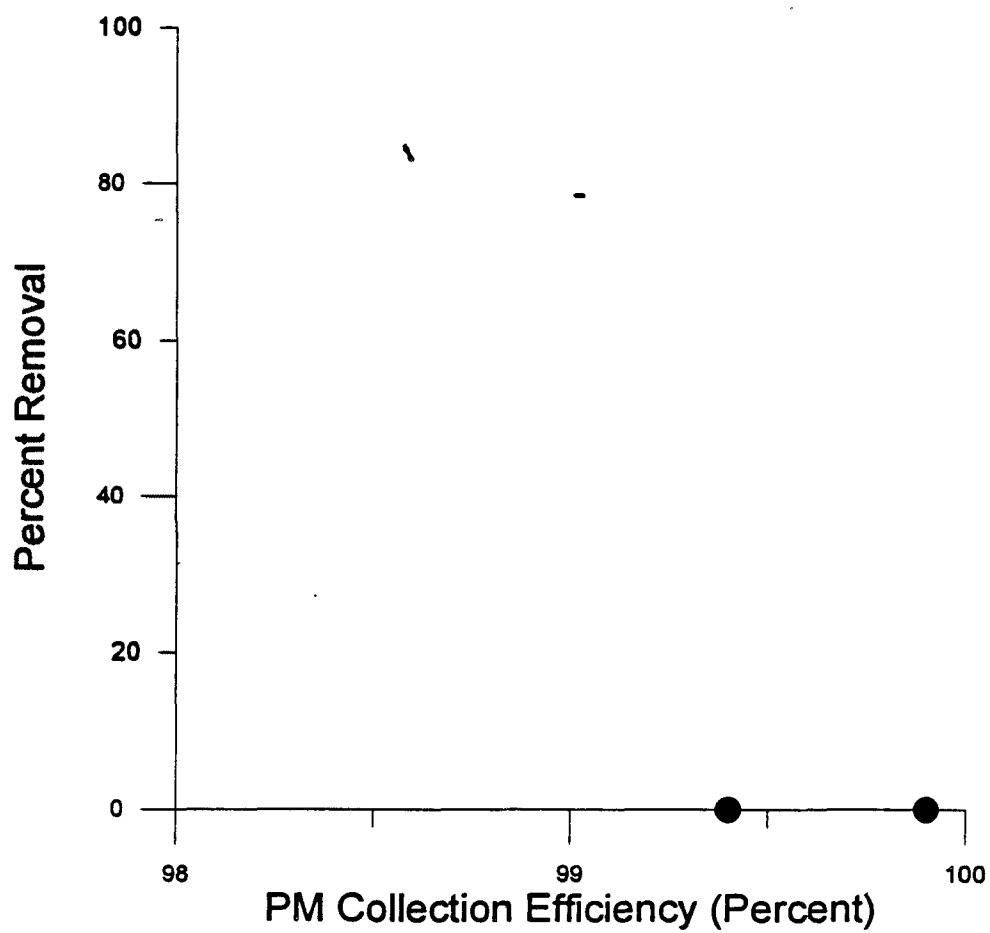


Table B-5
Test Data for Hot-Side Electrostatic Precipitators (Controlling Coal-Fired Units)

Unit	Control Device	Hg Removal %	Reference
EPRI Site 110	ESP, HS (inlet Hg concentration of 5.3 µg/dscm)	0	Southern Research Institute, 1993b
EPRI Site 110 with NO _x control	ESP, HS (inlet Hg concentration of 0.3 µg/dscm)	0	Southern Research Institute, 1993b
	Median	0 (see description in text)	

Figure B-6
Removal of Mercury By Electrostatic Precipitators (Oil)

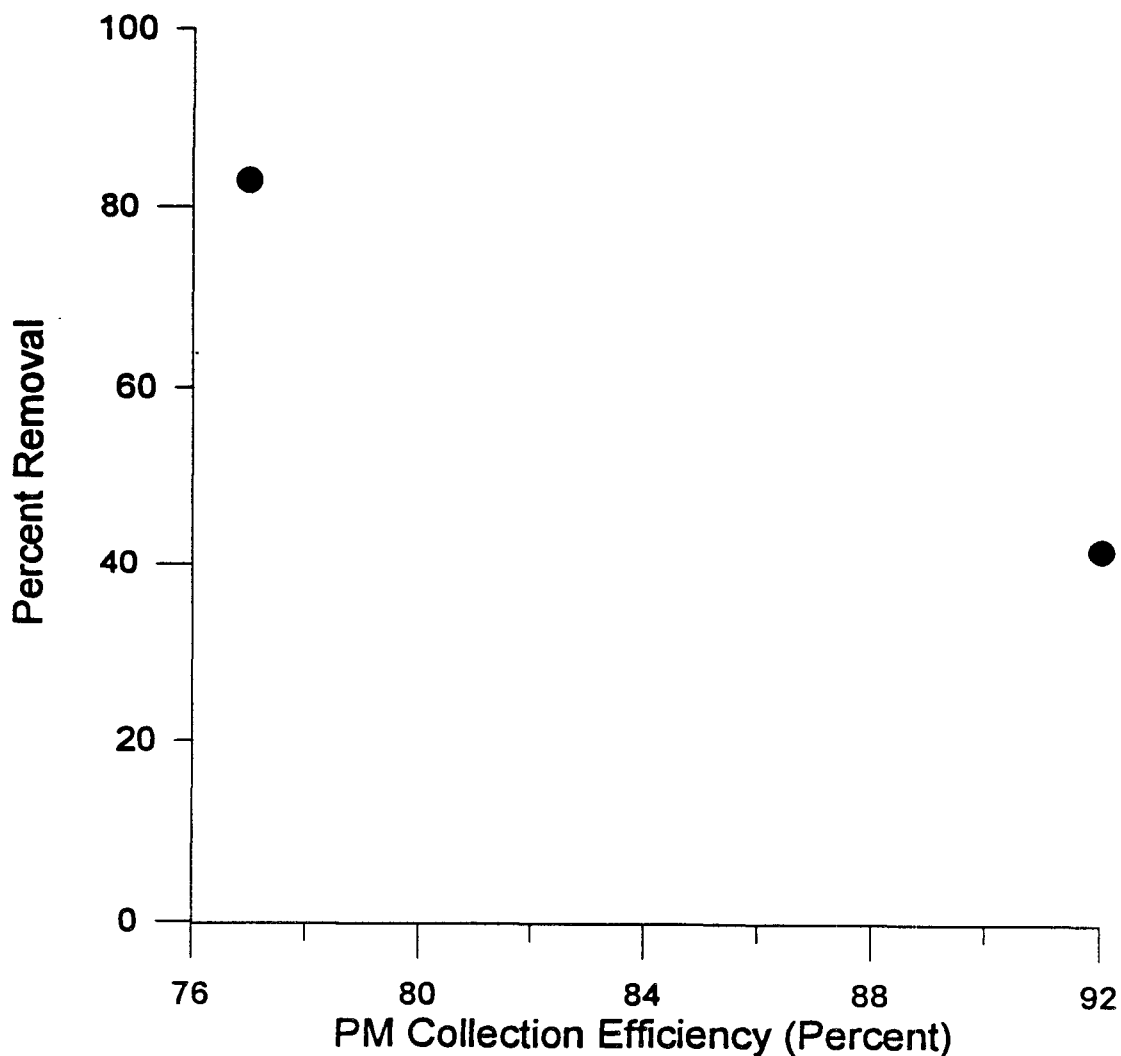


Table B-6
Test Data for Cold-Side Electrostatic Precipitators (Controlling Oil-Fired Units)

Unit	Control Device	Hg Removal %	Reference
EPRI Site 112 (oil-fired)	ESP, CS (inlet Hg concentration of 1.8 µg/dscm)	83	Carnot, 1994b
EPRI Site 118 (oil-fired)	ESP, CS (inlet Hg concentration of 1.4 µg/dscm)	41.7	Carnot, 1994c
	Median	62.35	
	Mean	62.35	
	Standard deviation	29.2	

B.5 Mechanical Collectors and Venturi Scrubbers

Mechanical collectors typically have very low collection efficiencies, often lower than 30 percent for particles in the 0 to 0.3 µm size range. These devices are used as gross particulate removal devices before ESPs or as APCDs on oil-fired units. Venturi scrubbers can be effective for particulate control but require high pressure drops (more than 50 or 60 in. of water) for small particles. Even with high pressure drops, ESPs and FFs are normally more effective for submicron particles. Mechanical collectors and venturi scrubbers are not expected to provide effective mercury removal, especially for those mercury compounds concentrated in the submicron PM fractions and in the vapor phase and, thus, are not discussed in this study.

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APPENDIX C

EMISSION MODIFICATION FACTORS FOR UTILITY BOILER EMISSION ESTIMATES

Table C-1
Emission Modification Factors for Utility Boiler Emission Estimates^a

Type of APCD or Boiler	EMF Factor
Fabric Filter	0.626
Spray Dryer Adsorber (includes a fabric filter)	0.701
Electrostatic precipitator (cold-side)	0.684
Electrostatic precipitator (hot-side)	1.000
Electrostatic precipitator (oil-fired unit)	0.315
Particulate matter scrubber	0.957
Fluidized gas desulfurization scrubber	0.715
Circulating fluidized bed combustor	1.000
Cyclone-fired boiler without NOx control (wet bottom, coal-fired)	0.856
Front-fired boiler without NOx control (dry bottom, coal-fired)	0.706
Front-fired boiler without NOx control (dry bottom, gas-fired)	1.000
Tangential-fired boiler without NOx control (before a hot-side ESP, coal-fired)	1.000
Tangential-fired boiler with NOx control (before a hot-side ESP, coal-fired)	0.748
Front-fired boiler without NOx control (dry bottom, oil-fired)	1.000
Front-fired boiler with NOx control (dry bottom, oil-fired)	1.000
Opposed-fired boiler without NOx control (dry bottom oil-fired)	0.040
Tangentially-fired boiler without NOx control (dry bottom, oil-fired)	1.000
Tangentially-fired boiler with NOx control (dry bottom, oil-fired)	1.000
Opposed-fired boiler with NOx control (dry bottom, coal-fired)	0.812
Front-fired boiler without NOx control (wet bottom, coal-fired)	0.918
Tangentially-fired boiler without NOx control (dry bottom, coal-fired)	1.000
Tangentially-fired boiler with NOx control (dry bottom, coal-fired)	0.625
Vertically-fired boiler with NOx control (dry bottom, coal-fired)	0.785

^a To calculate mercury control efficiency for a specific boiler/control device configuration, the EMF is subtracted from 1.

TECHNICAL REPORT DATA

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16. ABSTRACT <p>This volume of the draft Mercury Study Report to Congress describes mercury emissions from anthropogenic sources in the United States. The anthropogenic emissions summary estimates national mercury emissions rates by source category for area and point sources including combustion, manufacturing and miscellaneous point sources. Combustion point sources that dominate anthropogenic emissions are these: medical waste incineration, municipal waste combustion, utility boilers, and commercial/industrial boilers. National emission estimates are based on data from a 1990-1993 time-frame. Within the United States numerous industrial and manufacturing processes contribute mercury emissions to the atmosphere. These emissions occur nation-wide; however, the area east of the Mississippi River has the highest predicted concentrations of emissions from anthropogenic sources in the U.S. The land areas having the greatest concentrations of mercury emissions from anthropogenic sources of total mercury are the following: the urban corridor from Washington DC to Boston, the Tampa and Miami areas of Florida, the larger urban areas of the Midwest and Ohio Valley, and two sites in northern Texas. Sources of uncertainty and variability in these emissions categories are described in this volume. Data needed to improve these estimates are described in this volume.</p>					
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