

Mercury Study Report to Congress

Volume VII:
An Evaluation of Mercury
Control Technologies and Costs

SAB REVIEW DRAFT



Office of Air Quality Planning & Standards
and
Office of Research and Development

MERCURY STUDY REPORT TO CONGRESS

VOLUME VII:

**AN EVALUATION OF MERCURY CONTROL
TECHNOLOGIES AND COSTS**

SAB REVIEW DRAFT

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LIST OF SYMBOLS, UNITS AND ACRONYMS

AC	Activated carbon
APCD	Air pollution control device
ASME	American Society of Mechanical Engineers
CAA	Clean Air Act as Amended in 1990
CaS	Calcium sulfide
cf	Cubic feet
CFB	Circulating fluidized bed
cm	Cubic meter
CRF	Capital recovery factor
dscf	Dry standard cubic feet
dscm	Dry standard cubic meter
ESP	Electrostatic precipitator
DSI	Dry sorbent injection
EPRI	Electric Power Research Institute
FFDCA	Federal Food, Drug, Cosmetic Act
FFs	Fabric filters
FGD	Flue gas desulfurization
FIFRA	Federal Insecticide, Fungicide, Rodenticide Act
FWS	U.S. Fish and Wildlife Service
GACT	Generally available control technology
GLFCATF	Great Lakes Fish Consumption Advisory Task Force
GLNPO	Great Lakes National Program Office
g	Gram
gr	Grains
HAPs	Hazardous air pollutants
HCl	Hydrochloric acid
Hg	Mercury
HgCl	Mercuric chloride
HgI	Mercuric iodide
HgO	Mercuric oxide
HgS	Mercuric sulfide
HgSe	Mercuric selenite
HMTA	Hazardous Materials Transportation Act
HVAC	Heating, ventilating and air conditioning
IDLH	Immediately dangerous to life and health
INGAA	Interstate Natural Gas Association Of America
kg	Kilogram
kW	Kilowatt
MACT	Maximum achievable control technology
MB	Mass burn
MCL	Maximum contaminant level
Mg	Megagram
MSW	Municipal solid waste
MW	Megawatt
MWCs	Municipal waste combustors
MWIs	Medical waste incinerators

LIST OF SYMBOLS, UNITS AND ACRONYMS (continued)

NaCl	Sodium chloride
NaOH	Sodium hydroxide
ng	Nanogram
NIOSH	National Institute for Occupational Safety and Health
Nm ³	Normal cubic meter
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NSP	Northern States Power
NSPS	New source performance standard
OAQPS	Office of Air Quality Planning and Standards (U.S. EPA)
OECD	Organization for Economic Co-operation and Development
O&M	Operation and maintenance
OSHA	Occupational Safety and Health Administration
PCBs	Polychlorinated biphenyls
PELs	Permissible exposure limits
PM	Particulate matter
ppm	parts per million
ppmv	parts per million by volume
RQ	Reportable quantity
SARA	Superfund Amendments and Reauthorization Act
scf	Standard cubic feet
scm	Standard cubic meter
SD	Spray dryer
SDAs	Spray dryer absorbers
TCC	Total capital cost
TCLP	Toxicity characteristic leaching procedure
TMT	Trimercapto-s-triazine
tpd	Tons per day
TRI	Toxic Release Inventory
µg	Microgram
UNDEERC	University of North Dakota Energy and Environmental Research Center
WS	Wet scrubber
WW	Waterwall

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 report -- Volume VII of the Mercury Study Report to Congress -- provides information on mercury control technologies, associated costs and regulatory issues. It describes and analyzes additional technologies that could bring about reductions of mercury emissions, and existing state and federal programs that control the use and release of mercury. This report also describes management alternatives and U.S. EPA's statutory authority to control mercury emissions under the CAA.

Control Technologies and Associated Costs and Impacts

This report focuses on mercury control technologies, costs, and financial impact estimates for six industries: municipal waste combustors (MWCs), medical waste incinerators (MWIs), utility boilers, chlor-alkali plants, primary copper smelters and primary lead smelters. These source categories were chosen for control technology and cost analyses on the basis of either their source category emissions in the aggregate or their potential to be significant point sources of emissions. Consideration was also given to whether a particular source category was a feasible candidate for application of control technology (e.g., fluorescent lamp breakage would not be considered an appropriate mercury emission source category for a technology-based standard under section 112 of the Clean Air Act). Although this narrowed the analyses to a certain group of source categories, it was believed that this approach would give an overall sense of what could be required in terms of technologies and costs for the selected source categories. •

Control technology performance and cost information was obtained from the literature and pollution control technology vendors to develop cost effectiveness values for the various mercury controls applied to model plants for each industry. The estimated cost effectiveness values represent generalized costs and are not intended to be site-specific. Table ES-1 summarizes the results of the cost analysis. For each of the six emission sources, the applicable mercury controls and the estimated level of control and cost effectiveness are presented in units of dollars per pound of mercury removed and other measures (e.g., dollars per pound of medical waste incinerated for MWIs and dollars per pound of copper produced for primary copper smelters).

The financial impact of mercury controls was determined for each of the model plants representing the six industries examined. Affordability was based on financial ratios that were determined on the basis of whether an industry could pass the cost of mercury control on to consumers. For industries that can potentially pass control costs onto consumers (MWCs, MWIs and utility boilers), affordability was determined based on the ratio of total annual control costs to revenues. This ratio defines the potential increase in service price arising from the mercury control costs. For two of the industries that have little control over the price they are able to receive for their products and thus cannot pass mercury control costs on to consumers (chlor-alkali plants and primary

Table ES-1
Cost Effectiveness of Control Technologies

Source	Mercury Control Technique	Cost Effectiveness		Cost Comments
		\$/lb Hg Removed	Other Measures	
MWCs	Material separation (batteries)	1,450	\$0.37/ton MSW	Costs are very community specific; Results shown are based on one community's program
	Production substitution (e.g., batteries, fluorescent lights)	--	--	The potential for product substitutions requires that the specific circumstances of each situation be examined; general cost estimates are not possible
	Activated carbon injection	211-870	\$0.7-3.5/ton MSW	Costs assume an 85% reduction; range of costs cover the two model plants
	Carbon filter beds	513-1,083	\$5.44-9.39/ton MSW	Range of costs cover the two model plants
	Polishing wet scrubber	1,600-3,320	\$5.3-13.5/ton MSW	Costs assume an 85 percent reduction; range of costs cover the two model plants
MWIs	Material separation (batteries)	--	--	Costs vary on a site-specific basis; no costs were available; cost effectiveness for a hospital program would be assumed to be better than for a community program
	Activated carbon injection	228-955	\$10.3-43/ton medical waste	Costs assume an 85 percent mercury reduction; acid gas controls are assumed to be in place; range of costs cover the two model plants
	Polishing wet scrubber	310	\$12/ton medical waste	Costs developed for the large model plant only (see box on page 3-2 and 3-3); acid gas controls are assumed to be in place

Table ES-1
Cost Effectiveness of Control Technologies
(continued)

Source	Mercury Control Technique	Cost Effectiveness		Cost Comments
		\$/lb Hg Removed	Other Measures	
Utility Boilers	Carbon filter beds	33,000-38,000	2.7-3.1 mills/kWh	Range of costs covers two model plants (Models 1d and 2 in the box on page ES-2)
	Activated carbon injection (coal fired) 100% of Hg as HgCl ₂	4,970-27,700	0.41-2.26	Range of costs covers model plants burning coal (Models 1a, 3a, 1b, 3b and 1c)
	80% of Hg as HgCl ₂	6,190-34,600	0.41-2.26	
	50% of Hg as HgCl ₂	9,840-55,400	0.40-2.25	
	30% of Hg as HgCl ₂	16,300-92,000	0.40-2.25	
Chlor-Alkali Plants Using Mercury Cell Process	Process modification	4,590	\$39.6/ton chlorine produced	Cost effectiveness calculated using capital and electrical costs only
	Depleted brine scrubbing	1,040	\$6.7/ton chlorine produced	Cost data were scaled from 1972 dollars
	Treated activated carbon adsorption	769	\$5.1/ton chlorine produced	Cost data were scaled from 1972 dollars
Primary Copper Smelters	Selenium filters	497	\$11.3/ton copper produced	Cost data and information on mercury emissions at copper smelters are very limited
Primary Lead Smelters				
Sintering Stream	Selenium filters	541	\$2.7/ton lead produced	Cost data and information on mercury emissions at lead smelters are very limited
Furnace Stream	Selenium filters	520	\$5.8/ton lead produced	Cost data and information on mercury emissions at lead smelters are very limited

copper smelters), financial impact was determined based on both the ratio of total annual control cost to profits and the ratio of annual capital costs to total annual expenditures. These ratios define the financial impact of installing and operating the mercury controls. Information on profits and expenditures was not available for the primary lead smelter industry. Consequently, financial impact was estimated as the price increase in lead necessary to recover the control costs based on the ratio of total annual control costs to revenues.

The estimated financial impact of mercury controls for each industry examined is in Tables ES-2 and ES-3. These tables also present estimated annual revenues, profits and expenditures that were used to determine affordability.

For utility boilers, potential cost increases were calculated for a total of seven control technology-model plant combinations. The results of the financial analysis for utility boilers are summarized in Table ES-4.

Existing Federal and State Control Programs

Several federal agencies have authority and responsibility for controlling mercury uses, releases and exposures. For example, U.S. EPA has addressed for many years and continues to address the risks posed by mercury through regulations designed to limit releases to air, water and land. These regulations have been promulgated under the authority of the Safe Drinking Water Act, the Clean Water Act, the Clean Air Act, the Resource Conservation and Recovery Act and the Comprehensive Environmental Response, Compensation, and Liability Act. Other federal agencies that have mercury-related standards include the Food and Drug Administration (which regulates mercury in cosmetics, food and dental products), the Occupational Safety and Health Administration (which regulates mercury exposures in the workplace) and the Department of Transportation (which limits the potential for mercury releases during transportation). In general, existing federal standards can be categorized as environmental media standards, environmental source controls, or product controls.

Mercury control regulations are increasing rapidly at the state level. Many states are developing new regulations that will control the release of mercury from different environmental sources. Minnesota, for example, is currently drafting management standards for facilities that recycle mercury-containing waste and has proposed new combustion rules. States also have developed new monitoring and reporting requirements on mercury release from air and water point sources. In addition to health-based concerns, states are focusing on waste disposal problems associated with mercury-containing products. Many states have regulations that ban or limit the amount of mercury in products, establish recycling requirements and impose disposal restrictions on products containing mercury. For example, certain types of batteries containing mercury are banned in a number of states and 12 states have enacted laws that limit the amount of mercury in alkaline batteries to 0.025 percent by weight.

Management Alternatives

Effective control of mercury emissions may require a mix of strategies including pollution prevention, materials separation and conventional regulatory strategies to control mercury emissions at the stack. Pollution prevention would be suitable for those processes or industries where a mercury substitute is demonstrated and available (e.g., mercury cell chlor-alkali plants). Material separation is an appropriate approach for processes where mercury-containing products are disposed of by incineration, or where mercury can be reduced in the fuel prior to the fuel being combusted (e.g.,

Table ES-2
Estimated Annual Revenues and Potential Cost Increases^a for MWCs, MWIs and Lead Smelters

Parameter	Municipal Waste Combustors		Medical Waste Incinerators		Primary Lead Smelters
Total Annual Revenues (millions of dollars)	Small ^b :	\$3.4	Hospital:	\$32.5	\$66.8
	Large ^c :	\$38	Commercial:	\$2.0	
Potential Cost Increases					
Activated Carbon Injection	Small ^b :	6.9%	Hospital:	0.02%	N/A
	Large ^c :	1.3%	Commercial:	2.0%	
Carbon Filter Beds	Small ^b :	83%		-- ^e	N/A
	Large ^c :	35%		-- ^e	
Selenium Filter		N/A ^d		N/A	1.3%

^a Potential cost increase = total annual operating cost divided by total annual revenue. Represents the potential cost increase in service or product to cover the cost of controls.

^b Capacity = 180 Mg/day.

^c Capacity = 2,045 Mg/day.

^d N/A = control technology is not applicable.

^e The costs of the carbon filter beds were not estimated for this model plant.

Table ES-3
Estimated Annual Profits, Expenditures,
Revenues and Financial Impacts^a

Parameter	Chlor-Alkali Plants	Primary Copper Smelters
Total Annual Profits (millions of dollars)	\$12.9	\$22.0
Total Annual Expenditures (millions of dollars)	\$26.7	\$204
Financial Impact		
Membrane Cell Process	12% of Expenditures	N/A ^b
Depleted Brine	5.1% of Profits	N/A
Scrubbing	0.7% of Expenditures	
Treated Activated Carbon Adsorption	3.9% of Profits 0.5% of Expenditures	N/A
Selenium Filters	N/A	4.4% of Profits 0.2% of Expenditures

^a Financial impact = total annual control costs divided by profits and annual capital costs divided by total expenditures.

^b N/A = control technology is not applicable.

Note: The percentage of annual profits represents the amount of profit that would be needed to absorb the control costs. The percentage of annual expenditures provides a measure of the industry's ability to acquire the capital needed for the controls while still remaining competitive.

Table ES-4
Model Utility Boilers: Estimated Annual Revenues and Potential Cost Increases^a

Model ^b	Size (MW)	Fuel	Pre-Existing Controls	Estimated Revenues (\$ Million)	Mercury Control	Annual Control Costs (\$ Million)	Potential Cost Increase %
1a	975	Low-sulfur coal	ESP	287	AC injection ^c	10.1	3.5
1b	975	Low-sulfur coal	ESP	287	Spray cooler, AC injection, FF	7.9	2.8
1c	975	Low-sulfur coal	ESP	287	Spray cooler, AC injection	2.4	0.8
1d	975	Low-sulfur coal	ESP	287	Carbon filter bed	59.9	21
2	975	High-sulfur coal	ESP/FGD	287	Carbon filter bed	60.7	21
3a	100	Low-sulfur coal	ESP	29	AC injection	0.64	2.2
3b	100	Low-sulfur coal	ESP	29	Spray cooler, AC injection, FF	1.3	4.4

a Potential cost increase = total annual operating costs divided by total annual revenues, which represents the potential increase in electricity costs to cover the costs of control.

b See model plant description in box on pages 3-2 and 3-3 as well as Appendix B.

c AC = activated carbon.

medical waste incineration). The third approach, conventional regulatory strategies, may be applicable when mercury is emitted to the environment as a result of trace contamination in fossil fuel or other essential feedstock in an industrial process (e.g., smelting). Other non-traditional approaches such as emissions trading or application of a use tax, or other market-based approaches may also prove feasible for mercury control. In addition, emissions control is only one possible means for risk control; reduced human exposure, for example through the use of fish advisories, is another alternative that would need to be explored when selecting among strategies for reducing risks to human health (though not to ecosystems).

The analyses of control technologies and costs presented in this Report are not intended to replace a thorough regulatory analysis, as would be performed for a rulemaking. The information presented is intended to present the range of available options and provide a relative sense of the extent of mercury reductions achievable and the general magnitude of the cost of such reductions.

The three major types of control techniques reviewed are:

- Pollution prevention measures, including product substitution and process modification;
- Materials separation; and
- Flue gas treatment technologies.

Pollution Prevention Measures

One possible means of achieving reductions in mercury emissions is through the use of pollution prevention or source reduction. Such approaches to achieving reductions involve changes in processes or inputs to reduce or eliminate emissions of mercury from a particular product or process. They could include, for example, the replacement of mercury with an appropriate substitute or the use of low-mercury content inputs.

In considering opportunities for pollution prevention or source reduction it is important to consider both the potential reductions achievable and the costs of these options. Any consideration of the potential reductions, should examine whether (and the extent to which) emission reductions from the particular sources in question will yield reductions in risk to public health and the environment. It is also essential to understand the costs associated with implementing a pollution prevention measure, including any changes in the quality of the end product.

Materials Separation

Removing mercury-containing products such as batteries, fluorescent lights and thermostats from the waste stream can reduce the mercury input to waste combustors without lowering the energy content of the waste stream. The mercury removal efficiency would vary, however, depending on the extent of the separation. Many materials in wastes contain mercury. Materials that comprise a large portion of the waste stream, such as paper, plastic, dirt and grit and yard waste, contain very low concentrations of mercury. Therefore, obtaining appreciable mercury reduction from separation of these types of materials would require separating a large fraction of the total waste stream. Separating these materials would counter the intended purpose of the combustion process, which is to disinfect and reduce the volume of waste materials.

Other materials contain higher concentrations of mercury, but make up only a very small portion (less than 1 percent) of the waste stream. These materials include mercuric oxide batteries, fluorescent lights, thermostats and other electrical items. Separation of such materials can reduce mercury input to a combustor without removing any of the energy content of the waste stream. To evaluate a materials separation program, the feasibility and costs of separating a particular material should be compared with the mercury emission reduction achieved. Furthermore, the current and future mercury reduction achieved by separating a certain material should be considered since the mercury contribution of some materials such as household batteries has already declined considerably.

Coal cleaning is another option for removing mercury from the fuel prior to combustion. In some states, certain kinds of coal are commonly cleaned to increase its quality and heating value. Approximately 77 percent of the eastern and midwestern bituminous coal shipments are cleaned in order to meet customer specifications for heating value, ash content and sulfur content.

There are many types of cleaning processes, all based on the principle that coal is lighter than the pyritic sulfur, rock, clay, or other ash-producing impurities that are mixed or embedded in it. Mechanical devices using pulsating water or air currents can physically stratify and remove impurities. Centrifugal force is sometimes combined with water and air currents to aid in further separation of coal from impurities. Another method is dense media washing, which uses heavy liquid solutions usually consisting of magnetite (finely ground particles of iron oxide) to separate coal from impurities. Smaller sized coal is sometimes cleaned using froth flotation. This technique differs from the others because it focuses less on gravity and more on chemical separation.

Some of the mercury contained in coal may be removed by coal cleaning processes. Volume II of this report (An Inventory of Anthropogenic Mercury Emissions in the United States) presents available data on the mercury concentrations in raw coal, cleaned coal and the percent reduction achieved by cleaning. These data, which cover a number of different coal seams in four states (Illinois, Pennsylvania, Kentucky and Alabama), indicate that mercury reductions range from 0 to 64 percent, with an overall average reduction of 21 percent. This variation may be explained by several factors, including different cleaning techniques, different mercury concentrations in the raw coal and different mercury analytical techniques. It is expected that significantly higher mercury reductions can be achieved with the application of emerging coal preparation processes. These include selective agglomeration and advanced column flotation. Bench-scale testing is also being carried out to investigate the use of naturally-occurring microbes to reduce the mercury and other trace elements from coal.

Any reduction in mercury content achieved by coal cleaning results in a direct decrease in mercury emissions from boilers firing cleaned coals. The mercury removed by cleaning processes is transferred to coal-cleaning wastes, which are commonly in the form of slurries. No data are available to assess the emissions of mercury from coal-cleaning slurries.

Flue Gas Treatment Technologies

With the exception of mercury, most metals have sufficiently low vapor pressures at typical air pollution control device operating temperatures that condensation onto particulate matter is possible. Mercury, on the other hand, has a high vapor pressure at typical control device operating temperatures, and collection by particulate matter control devices is highly variable. Factors that enhance mercury control are low temperature in the control device system (less than 150 °Celsius [°C] [300 to 400 °Fahrenheit (°F)]), the presence of an effective mercury sorbent and a method to collect the sorbent. In general, high levels of carbon in the fly ash enhance mercury sorption onto particulate matter which

is subsequently removed by the particulate matter control device. Additionally, the presence of hydrogen chloride (HCl) in the flue gas stream can result in the formation of mercuric chloride (HgCl_2), which is readily adsorbed onto carbon-containing particulate matter. Conversely, sulfur dioxide (SO_2) in flue gas can act as a reducing agent to convert oxidized mercury to elemental mercury, which is more difficult to collect.

Add-on controls to reduce mercury emissions are described in detail in this volume, including information on commercial status, performance, applicability to the specified mercury emission sources, and secondary impacts and benefits. The controls described are:

- Carbon filter beds;
- Wet scrubbing;
- Depleted brine scrubbing;
- Treated activated carbon adsorption;
- Selenium filters; and
- Activated carbon injection.

The most important conclusions from the assessment of flue gas treatment technologies include:

- Conversion of mercury cell chlor-alkali plants to a mercury-free process is technically feasible and has been previously demonstrated.
- Control technologies designed for control of pollutants other than mercury (e.g., acid gases and particulate matter) vary in their mercury-removal capability, but in general achieve reductions no greater than 50 percent.
- Selenium filters are a demonstrated technology in Sweden for control of mercury emissions from lead smelters. Carbon filter beds have been used successfully in Germany for mercury control on utility boilers and MWC's. These technologies have not been demonstrated in the U.S. for any of these source types.
- Injection of activated carbon into the flue gas of MWC's and MWI's can achieve mercury reductions of at least 85 percent. The addition of activated carbon to the flue gas of these source types would not have a significant impact on the amount of particulate matter requiring disposal.
- No full-scale demonstrations of mercury controls have been conducted in the U.S. for utility boilers. Based on limited pilot-scale testing, activated carbon injection provides variable control of mercury for utility boilers (e.g., the same technology might capture 20 percent of the mercury at one plant and 80 percent at another). The most important factors affecting mercury control on utility boilers include the flue gas volume, flue gas temperature and chloride content, the mercury concentration and chemical form of mercury being emitted.
- The chemical species of mercury emitted from utility boilers vary significantly from one plant to another. Removal effectiveness depends on the species of mercury present. To date, no single control technology has been identified that removes all forms of mercury.

- The addition of activated carbon to utility flue gas for mercury control would significantly increase the amount of particulate matter requiring disposal.

Cost of Controls

The overall approach for assessing the cost of flue gas treatment technologies was to select a subset of source categories on the basis of either their source category emissions in the aggregate or their potential to be significant point sources of emissions. Consideration was also given to whether a particular source category was a feasible candidate for application of a control technology-based standard under section 112 of the CAA. This narrowed the analyses to six source categories: municipal waste combustors (MWC), medical waste incinerators (MWI), chlor-alkali plants, utility boilers, and primary lead and copper smelters.

In addition to determining the cost effectiveness of applying mercury control technology, a financial analysis was performed to evaluate the affordability of mercury control (in terms of potential price increases or impacts on financial impact) for the selected source categories.

Table ES-5 presents the six source categories for which a control technology and cost analysis was performed. The table presents the number of facilities in each category and the percent contribution of each to the national inventory. Potential national mercury reductions, potential national control costs and cost-effectiveness estimates are also presented. These estimates are based on the assumption that all plants within a source category will achieve the same reductions and incur the same costs as the model plants used in the analysis. Because this assumption would not be applicable in all circumstances, the estimates of potential reductions and costs should be used only for relative comparisons among the source categories to give an initial indication as to where mercury controls could provide the most emission reduction for the least cost.

The cost of mercury control incurred by any specific facility may be underestimated by the cost analysis presented in this Report because of variability inherent in the assumptions that were made in the analyses. These include the efficiency of the various control techniques for reducing mercury, the amount of mercury in the flue gas stream and other site-specific factors such as down-time and labor costs. In addition, costs for monitoring and record keeping were not included in the cost analyses. On the other hand, the costs represent retrofit application of controls. Installation of controls at new facilities can be significantly less expensive than retrofitting an existing facility.

The estimates of cost for mercury reduction also do not illustrate two important considerations. One is that all of the cost of control is attributed to mercury removal. Many of these controls achieve reductions of other pollutants as well. The benefits of these additional reductions should also be considered. Second, the technologies available for mercury control represent relatively new applications of these technologies. Thus, it is possible that new or emerging control technologies will improve the cost-effectiveness.

Clean Air Act Authorities

Mercury is a priority pollutant across numerous U.S. EPA programs including air, water, hazardous waste and pollution prevention. The statutory authorities under the CAA that could be used to control mercury emission sources are briefly summarized below.

Table ES-5
Potential Mercury Emission Reductions and Costs for Selected Source Categories

Mercury Source Category	Number of Facilities	% of U.S. Mercury Emission Inventory	Mercury Control Techniques	Potential National Reductions ^b	Potential National Annual Costs ^c	Cost-Effectiveness (\$/lb of mercury removed) ^d
Municipal waste combustors	149	23	Material separation Product substitution <u>Activated carbon injection</u> Carbon filter beds Polishing wet scrubber	50 tons (90% reduction)	\$56 million	\$211-870
Medical waste incinerators	~3,700	27	Material separation <u>Activated carbon injection</u> Polishing wet scrubber	60 tons (90% reduction)	\$24 million	\$228-955
Coal-fired utility boilers	426 (1,043 boilers)	21	Fuel switching Advanced coal cleaning Carbon filter beds <u>Activated carbon injection</u>	24-44 tons (50-90% reduction) ^e	\$2.9 billion	\$5,240-28,000
Chlor-alkali plants using the mercury cell process	14	2.7	<u>Process modification</u> Depleted brine scrubbing Treated activated carbon adsorption	6.5 tons (100% reduction)	\$70 million	\$4,590
Primary copper smelters	8	0.3 ^a	<u>Selenium filters</u>	>0.7 tons (90% reduction)	\$7.7 million	\$497
Primary lead smelters	3	3.7	<u>Selenium filters</u>	8 tons (90% reduction)	\$0.8 million	\$1,061
Total	~4,900	78			~\$3 billion	

NOTE: The underlined mercury control techniques are the techniques on which potential national reductions and potential national annual costs are based.

^a Reflects one smelter only; a national estimate would be higher.

^b Estimated reductions assuming every facility could achieve the reduction listed.

^c Potential national costs are estimates only and assume all facilities would incur the same costs as the model plants used in the analysis.

^d Where cost-effectiveness values are presented as a range, the values reflect the range across facilities of different sizes.

^e The range in potential national reductions reflects the variable efficiency of activated carbon injection to control mercury emissions from coal-fired utility boilers. Activated carbon injection has not been demonstrated for a full-scale utility boiler application. Control costs are based on the installation of spray cooler, fabric filter and carbon injection systems

Section 112(a) Lesser Quantity Emission Rates

The U.S. EPA Administrator has the discretion to redefine major sources by setting an emissions cutoff lower than the 10 tons per year emission rate level for a single pollutant or 25 tons per year emission rate for a mixture of pollutants. This is referred to as a lesser quantity emission rate (LQER). The CAA states that LQERs are pollutant-specific and should be based on public health or environmental effects.

The major implications of setting an LQER are that all the requirements for a major source, including setting maximum achievable control technology (MACT) standards, mandatory residual risk analyses, calculation of the MACT floor, modification provisions and Title V permitting requirements become applicable to what was previously defined as an area source category.

Section 112(c)(6) List of Specific Pollutants

Section 112(c)(6) requires that by 1995, sources accounting for not less than 90 percent of the aggregate emissions of each of seven specific pollutants must be listed on the source category list, and be subject to standards under 112(d)(2) or (4) no later than 2005. The pollutants are: alkylated lead compounds; polycyclic organic matter; hexachlorobenzene; mercury; polychlorinated biphenyls; 2,3,7,8-tetrachlorodibenzo-p-dioxin; and 2,3,7,8-tetrachlorodibenzofuran. This provision makes a specific reference to utility boilers. It reads: "This paragraph shall not be construed to require the Administrator to promulgate standards for such pollutants emitted by electric steam generating units."

Section 112(d) Emission Standards

Section 112(d) requires that emission standards be established for each source category listed on the source category list. The emission standards are applicable to both new and existing sources and are based on the application of MACT. MACT is defined differently for new and existing sources as explained by 112(d)(2) and (3). Under 112(d)(4), if the pollutant is a threshold pollutant (i.e., noncarcinogen), the emission standard can be based on a health threshold with an ample margin of safety. A health threshold is a level where the risk of an adverse effect from exposure to the pollutant is negligible. Section 112(d)(5) allows the Administrator the discretion to apply generally available control technology (GACT) to area sources rather than MACT (or any other technologies that may be required of the source category on account of residual risk analyses under 112(f)).

Section 112(f) Residual Risk Program

Section 112(f) required the U.S. EPA to report to Congress on the methods that will be used to calculate the risk remaining after the promulgation of MACT emission standards under Section 112(d). This report should address the public health significance of the risk and the actual health effects experienced by persons living in the vicinity of emitting sources, and make recommendations on legislation regarding the risk. This report is due to Congress on November 15, 1996. If Congress does not accept any of the recommendations provided for reducing the residual risk, the Administrator has the authority to promulgate any additional standards required in order to protect public health with an ample margin of safety. The report is currently under development.

Section 112(k) Urban Area Source Program

By 1995, a national strategy to control emissions of hazardous air pollutants (HAPs) from area sources in urban areas must be transmitted to Congress. The strategy must identify not less than 30

HAPs which present the greatest threat to public health in the largest number of urban areas. Source categories accounting for at least 90 percent of the aggregate emissions of each HAP must be listed on the source category list and be subject to 112(d) standards. The strategy, when implemented, is to achieve a 75 percent reduction in cancer incidence attributable to these sources.

The urban area source program is a section 112 authority that does not require that a finding of adverse effects be made for the area source category (as required for listing by 112(c)(3)). Mercury is a likely candidate for the urban area source program.

112(m) Atmospheric Deposition to Great Lakes and Coastal Waters (Great Waters)

The Great Waters study is an ongoing study with periodic reports to Congress required. This program must identify and assess the extent of atmospheric deposition of HAPs to the Great Waters, the environmental and public health effects attributable to atmospheric deposition and the contributing sources. The first report was submitted in May 1994 and is to be submitted biennially hereafter. Mercury was identified as a priority pollutant under the Great Waters program. The Administrator must determine if other provisions under Section 112 will adequately control these sources. If not, by 1995, further emission standards to control these sources must be promulgated.

The recommendations of the first Great Water Report to Congress were (1) the U.S. EPA should strive to reduce emissions of the identified pollutants of concern through implementation of the CAA; (2) a comprehensive approach should be taken both within the U.S. EPA and between the U.S. EPA and other federal agencies to reduce and preferably prevent pollution in the air, water, and soil; and (3) the U.S. EPA should continue to support research for emissions inventories, risk assessment and regulatory benefits assessment.

112(n)(1)(A) Study of Hazardous Air Pollutants for Electric Utility Steam Generating Units

The Utility Study is required to address the hazards to public health that are reasonably anticipated to occur as a result of emissions by electric utility steam generating units of ... [hazardous air pollutants] ... after imposition of the requirements of the Act. The list of 189 HAPs is presented in section 112(b) of the CAA. In the study, the U.S. EPA must develop and describe alternative control strategies for HAPs that may require regulation under section 112, and, if appropriate and necessary, the U.S. EPA is to proceed with rulemaking to control HAP emissions from utility boilers. Mercury is one of the pollutants of concern for utilities.

Section 129 Solid Waste Combustion

Under this section, the Administrator must establish emission guidelines and standards for solid waste incineration units, including municipal waste combustors, medical waste incinerators and commercial and industrial waste incinerators. The performance standards must specify numerical emission limits for mercury as well as a number of other pollutants. The U.S. EPA has already issued final rules for municipal waste combustors (59 FR 48198) and proposed rules for medical waste incinerators (60 FR 10654). Emission limits for hazardous waste combustors will be forthcoming under the Agency's Combustion Strategy.

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.

- Conversion of mercury cell chlor-alkali plants to a mercury-free process is technically feasible and has been previously demonstrated.
- Energy conservation and switching to low-mercury fuels would reduce the amount of mercury being emitted by utility boilers.
- Injection of activated carbon into the flue gas of MWC's and MWI's can achieve mercury reductions of at least 85 percent. The addition of activated carbon to the flue gas of these source types would not have a significant impact on the amount of particulate matter requiring disposal.
- Numerous opportunities exist for replacing mercury in various products with other materials, such as solid state electronics for mercury switches, digital thermometers for mercury thermometers and zinc-air batteries for mercury batteries.
- Removing mercury-containing products such as batteries, fluorescent lights and thermostats from the waste stream can reduce the mercury input to waste combustors without lowering the energy content of the waste stream. The mercury removal efficiency would vary, however, depending on the extent of the separation.
- Selenium filters are a demonstrated technology in Sweden for control of mercury emissions from lead smelters. Carbon filter beds have been used successfully in Germany for mercury control on utility boilers and MWC's. These technologies have not been demonstrated in the U.S.
- Activated carbon injection provides variable control of mercury for utility boilers, based on limited pilot-scale testing. The most important factors affecting mercury control on utility boilers include the flue gas volume, flue gas temperature and chloride content, the mercury concentration and chemical form of mercury being emitted. The addition of activated carbon to utility flue gas for mercury control would significantly increase the amount of particulate matter requiring disposal.
- The available data on coal cleaning indicate that mercury reductions ranged from zero to 64 percent. The average reduction was 21 percent. This variation may be due to several factors including different cleaning methods, different mercury concentrations in the raw coal and different mercury analytical techniques. There are no data available to assess the potential for mercury emissions from coal-cleaning slurries.
- Control technologies designed for control of pollutants other than mercury (e.g., acid gases and particulate matter) vary in their mercury-removal capability, but in general achieve reductions no greater than 50 percent.

There are many uncertainties associated with the cost analysis for individual source categories due to assumptions inherent in a model plant approach. The impact of these uncertainties on the analyses include the following:

- The cost of mercury control incurred by any specific facility may be underestimated by the cost analysis presented in this Report because of variability inherent in the assumptions that were made in the analyses. These include the efficiency of the various control techniques for reducing mercury, the amount of mercury in the flue gas stream and other site-specific factors such as down-time and labor costs. In addition, costs for monitoring and record keeping were not included in the cost analyses.
- The financial impacts calculated in this Report may not represent the financial status of all facilities within a source category. The affordability of mercury control will depend on a given facility's profits and expenditures.
- In the control technology analysis, the cost of control was attributed to mercury reduction only. Because many of the controls also reduce emissions of other pollutants, apportioning the costs across all pollutants reduced would better characterize the cost of mercury control.

To improve the control technology, cost and impacts assessment, the U.S. EPA would need the following:

- Data from full-scale testing of activated carbon injection at a coal-fired utility boiler.
- Additional data are needed on the efficiency of activated carbon injection, and various impregnated carbons, in reducing the different chemical species of mercury present in flue gas.
- Additional information on the efficiency and cost of other technologies for mercury control that are currently in the research stage. These include impregnated activated carbon, sodium sulfide injection and activated carbon fluidized bed.
- More data are needed on both the ability of conventional or advanced coal cleaning techniques to remove mercury from raw coal and advanced coal cleaning techniques such as selective agglomeration and advanced column floatation. The potential for mercury emissions from coal-cleaning slurries needs to be characterized.
- Additional analyses are required on the feasibility, cost effectiveness of other mercury emission prevention measures such as emissions trading, emissions averaging, energy conservation, renewable energy, and fuel switching.

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 for 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 report, Volume VII, includes a description and analysis of additional technologies that could bring about greater controls of mercury emissions, and existing state and federal programs that control the use and release of mercury. This report also describes management alternatives and the U.S. EPA's statutory authority under the CAA to control mercury emissions.

1.1 Risk Management Principles

Risk management is the process by which the U.S. EPA arrives at a regulatory decision. The analyses of human health effects, ecological effects and the exposure assessment presented in previous volumes of this Report were combined in the risk characterization (Volume VI). The risk characterization is designed to provide decision makers with the essential risk information they need for making risk management decisions. Risk managers must take into account not only the findings of the risk assessment, but also consider non-scientific factors such as statutory authorities, technological limitations and feasibility and economic and social consequences before reaching a regulatory decision.

This volume of the Mercury Study Report to Congress focuses on the risk management issues that decision makers need to weigh against the findings of the risk assessment and risk characterization. The first goal of this volume is to describe the analyses that were performed to characterize the non-scientific risk management factors listed above, and to present the results of these analyses. The second goal is to illustrate where there are regulatory gaps with respect to sources of airborne mercury by summarizing regulations already in place and describing other non-regulatory programs the U.S. EPA has underway to address emissions of mercury. Third, this report summarizes the regulatory options that are both available and feasible under the statutory authority of the CAA. In particular, sections 112 and 129 of the CAA contain provisions most pertinent to the control of mercury emissions.

The overall approach for the analyses was as follows. First, a subset of source categories was chosen for control technology and cost analyses on the basis of either their source category emissions in the aggregate or their potential to be significant point sources of emissions. Consideration was also given to whether a particular source category was a feasible candidate for application of control technology (e.g., fluorescent lamp breakage would not be considered an appropriate mercury emission source category for a technology-based standard under section 112 of the CAA). Although this narrowed the analyses to a certain group of source categories, it was believed that this approach would give the risk manager an overall sense of what could be required in terms of technologies and costs for the source categories potentially posing the greatest impact.

As mandated by the CAA, an analysis of control technologies and the costs of such technologies was undertaken. The analyses were primarily focused on "end-of-pipe" technologies and some pollution prevention options for six specific source categories. It is recognized that for a number of source categories, "end-of-pipe" technologies may not be the only remedy. For example, for the combustion source categories, a fuel switch could also be an effective way to achieve emission reductions. Removal of mercury-containing materials from the waste stream is another possible control option. Although these types of remedies are described qualitatively in the report, it was believed that it was beyond the scope of the analysis to quantitatively assess every combination of potential controls. In addition, it should be noted that the analyses presented here should not be considered to substitute for a regulatory analysis. Any source category that becomes subject to regulatory action should be thoroughly evaluated for a range of control options and not limited to those presented here.

In addition to determining the cost effectiveness of applying mercury control technology, a financial analysis was performed to evaluate the affordability of mercury control (in terms of potential price increases or impacts on financial impact) for the selected source categories.

Finally, existing state and Federal regulatory programs for mercury are described to help identify regulatory gaps, particularly for air emissions of mercury. Management alternatives and statutory authorities under the CAA are also summarized.

1.2 Report Organization

The remainder of this report is organized into the following five chapters:

- Chapter 2 describes opportunities for further mercury controls over and above the status quo, including pollution prevention measures (i.e., product substitution, process modification, and materials separation), coal cleaning, other management measures for utilities (e.g., emissions trading, conservation, renewable energy options and fuel switching) and end-of-pipe emission control technologies.
- Chapter 3 addresses the costs and financial impacts associated with further mercury controls.
- Chapter 4 summarizes mercury control strategies, including Federal and state authorities and activities that impact mercury emissions.
- Chapter 5 summarizes conclusions.
- Chapter 6 identifies research needs.

All references cited in the volume are listed after Chapter 6. The report also contains two appendices: Appendix A presents pilot test data on the mercury removal efficiency of activated carbon injection for utility boilers, and Appendix B presents model plant cost analyses for installing and operating applicable mercury control techniques at selected source types.

2. MERCURY CONTROLS

This chapter provides information on mercury controls that provide opportunities for significant further reductions of mercury emissions. Unlike the technologies described in Volume II of this Mercury Study, which are generally used to control other pollutants and reduce mercury emissions incidentally, the controls described in this chapter were assessed specifically for their potential to reduce mercury emissions.

The discussion focuses on mercury emission controls for six source categories: municipal waste combustors (MWCs), medical waste incinerators (MWIs), utility boilers, chlor-alkali plants, primary copper smelters and primary lead smelters. As discussed in Chapter 1, these sources were selected for analysis on the basis of their source category emissions in the aggregate, or their potential to be significant point sources of emissions. Three major types of control techniques are described:

- Pollution prevention measures, including product substitution and process modification;
- Management measures, including materials separation; and
- Flue gas treatment technologies.

Table 2-1 summarizes information on the control techniques applicable to the different source categories studied, including the level of mercury control and the present frequency of use. Each of the techniques is described in more detail in the sections that follow.

2.1 Pollution Prevention and Other Management Measures

Discussed below are four measures for reducing mercury emissions: substitution of non-mercury-containing products to reduce the amount of mercury entering waste streams ultimately managed in MWCs and MWIs; the conversion of mercury cell chlor-alkali plants to the membrane cell process, thereby eliminating all mercury emissions from that source category; separation of mercury-containing materials from the waste streams of MWCs and MWIs; and coal cleaning, which may remove some of the mercury contained in coal in addition to achieving other benefits (reducing ash and/or sulfur content and increasing heating value of coal). This section also provides a general discussion of various other management measures that could be used to reduce mercury emissions from utilities, including emissions trading and averaging programs, conservation measures, renewable energy options and fuel switching.

2.1.1 Product Substitution

One option for reducing the mercury content of municipal and medical waste streams is the replacement of items containing significant mercury levels. Although this substitution process is potentially applicable to a wide range of components, batteries have received the greatest attention because of their significant contribution to total mercury content in municipal and medical wastes.

The potential for product substitutions requires that the specific circumstances of each situation be examined. As a result, general estimates of potential mercury reductions and costs associated with product substitution are not possible at this point.

Table 2-1
Summary of Mercury Control Techniques for Selected Source Types

Mercury Control Technique	Applicable Source Type	Estimated Mercury Removal Efficiency	Cross-Media Impacts?^a	Other Pollutants Controlled	Comments
Product substitution (e.g., batteries, fluorescent lights)	MWCs, MWIs	Variable, depending on the extent of substitution	Yes	Could include other components of mercury-containing batteries, fluorescent lights and other products	<ul style="list-style-type: none"> • Product substitution has reduced the use of mercury in household batteries • Use of mercury-containing fluorescent lights has increased because of their energy efficiency, but lower mercury content is being achieved • The impact of product substitution to other areas depends on specific circumstances, including technical and economic feasibility
Process modification	Mercury cell chlor-alkali plants	100%	Yes	None directly	<ul style="list-style-type: none"> • In 1994, about one-half of the chlor-alkali plants used mercury-free processes • Because the membrane cell process has lower electricity demands than the mercury cell process, plant conversion results in an energy savings • Additional savings presumably also result by avoiding costs of recycling or disposing of mercuric wastes
Materials separation	MWCs and MWIs	Variable, depending on the extent of separation	Yes	Could include other components of mercury-containing wastes burned in MWCs or MWIs	<ul style="list-style-type: none"> • Separation of low-volume materials containing high mercury concentrations (e.g., batteries, fluorescent lights, thermostats and other electrical items) can reduce mercury input to a combustor without removing energy content of the waste stream • Household battery separation has been implemented by several communities; program efficiency ranges from 3 to 25 percent • Pilot studies conducted at hospitals have been successful

Table 2-1 (continued)
Summary of Mercury Control Techniques for Selected Source Types

Mercury Control Technique	Applicable Source Type	Estimated Mercury Removal Efficiency	Cross-Media Impacts? ^a	Other Pollutants Controlled	Comments
Carbon filter beds	MWCs, utility boilers, industrial boilers	99%	Yes	Residual organic compounds, other heavy metals, SO ₂ , acid gases	<ul style="list-style-type: none"> • Currently applied to five full-scale power plants in Germany, and planned to be installed on five hazardous waste incinerators in Europe • Technically feasible to other sources, such as MWIs or smelters, but has not been applied • Potential negative effects associated with the disposal of spent carbon and the potential for fires in the bed
Wet scrubbing (single-staged polishing scrubbers)	MWCs, MWIs, boilers	Can be >90% for water-soluble species; limited for elemental mercury	Yes	Acid gases, metals, particulate matter, dioxins, furans	<ul style="list-style-type: none"> • Applied to one MWI in the U.S. • Have not been applied to MWCs or boilers in the U.S., although they have been used at MWCs in Europe • Requires treatment of wastewater prior to disposal • May form more toxic, lesser-chlorinated dioxin and furan congeners
Depleted brine scrubbing	Chlor-alkali plants	98%	Yes	None	<ul style="list-style-type: none"> • Very little information is available on this technique
Treated activated carbon adsorption	Chlor-alkali plants	90%	Yes	Residual organic compounds, other heavy metals, SO ₂ , acid gases	<ul style="list-style-type: none"> • Very little information is available on this technique • In 1984, carbon bed systems were in use at 8 of the 20 chlor-alkali plants in operation in the U.S. at that time

Table 2-1 (continued)
Summary of Mercury Control Techniques for Selected Source Types

Mercury Control Technique	Applicable Source Type	Estimated Mercury Removal Efficiency	Cross-Media Impacts? ^a	Other Pollutants Controlled	Comments
Selenium filters	Primary copper smelters, primary lead smelters, and (more limited) MWCs, crematories, power plants	90%	Yes	Particulate matter, acid gases	<ul style="list-style-type: none"> Factors that influence performance include inlet mercury concentrations, flue gas temperature and flue gas dust content Four known applications at smelters as well as a MWC and a crematory in Sweden; known installation at a German power plant; potentially applicable to MWIs Spent filter containing selenium and mercury must be landfilled after use More information needed on the possibility of selenium being emitted from the filter itself
Activated carbon injection	MWCs, MWIs, utility boilers	50-90+%	Yes	Chlorinated dioxins and furans, potentially other semi-volatile organics	<ul style="list-style-type: none"> Activated carbon injection efficiencies reported for utility boilers are based on pilot-scale data and as such have a high degree of uncertainty Factors that influence performance include flue gas temperature, amount of activated carbon injected, type of particulate matter collector, concentration and species of mercury in flue gas and type of carbon used Addition of carbon could have significant impact on amount of particulate matter requiring disposal from utility boilers, but not from MWCs or MWIs

^a For the purpose of this table, cross-media impacts refer to the potential to transfer and release mercury to media other than air, such as soil, ground water, and surface water. For example, carbon filter beds and wet scrubbers remove mercury from air emissions but result in the generation and disposal of mercury-containing solid and liquid wastes, respectively. In the case of product substitution, cross-media impacts refer to the potential to decrease airborne emissions of mercury at one site but increase such emissions elsewhere.

2.1.1.1 Batteries

The battery industry has undertaken two initiatives to reduce the amount of mercury used in battery manufacturing. The first initiative is directed toward reducing and eventually eliminating mercury in household batteries. Historically, mercury has been used (1) to inhibit side reactions and corrosion of the battery casing material used with carbon-zinc and alkaline batteries, and (2) as a component in the zinc amalgam that forms the electrical anode of alkaline batteries. Through recent production improvements, use of mercury to inhibit casing material reactions and corrosion has been eliminated, and the level of mercury in the zinc amalgam has been reduced to trace levels (White and Jackson, 1992).

The battery industry's second initiative has been to develop alternatives to mercuric oxide batteries where small size and constant current supply are critical. These batteries use mercury as the cathode material. Unlike alkaline batteries, mercuric oxide batteries cannot maintain their performance at reduced mercury levels. The battery industry has developed alternative battery designs, such as zinc-air and silver-oxide batteries, that are capable of delivering comparable levels of performance at reasonable cost. As a result, the use of mercuric oxide batteries has declined, lowering the amount of mercury put into the municipal and medical waste streams (White and Jackson, 1992).

The applicability of product substitution to other areas will vary based on technical and economic feasibility -- both on the part of product manufacturer and user. For example, zinc-air batteries, although having similar current delivery performance to mercuric oxide batteries, have a shorter operating life. Because the number of batteries required on an annual basis would increase if consumers use zinc-air batteries, the annualized cost of zinc-air batteries would be higher than for mercury-zinc batteries.

Industry initiatives, combined with recent Federal and state control programs, have brought about a dramatic reduction in the mercury content of batteries (see Volume II of this Report for a more detailed discussion of battery trends). For example, alkaline batteries in the U.S. in the mid-1980s 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, 1995). Such efforts have resulted in a corresponding decline in the amount of mercury entering the municipal solid waste stream. One U.S. EPA study (U.S. EPA, 1992a) estimates that the discard of mercury in batteries has decreased from approximately 306 tons in 1970 to 176 tons in 1995, a 43 percent decrease. The actual discards in 1995 are expected to be even lower than this estimate, which does not reflect recent state, Federal, or battery manufacturer's efforts to limit the mercury content of batteries, discussed in Chapter 4 of this volume.

2.1.1.2 Fluorescent Lights and Other Products

Similarly, the use of mercury-containing fluorescent and other high-efficiency lighting systems is increasing because of the energy efficiency of these systems. Reducing or eliminating the use of mercury in these applications could result in continued use of less efficient technology thereby requiring increased electricity generation and associated air emissions.

As discussed in Volume II of this Report, 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 in

fluorescent lamps in 1995 would be 14.7 tons. This would represent a 44 percent decrease from the estimated 26 tons of mercury discarded along with fluorescent lamps in 1989.

Alternatives currently available to mercury amalgams in dental preparations include gold, ceramic, porcelain, polymers, composites and glass ionomers. The cold silver technique and gallium are among the most promising currently in the development phase. While alternatives to mercury amalgam have been developed, these alternatives have very limited use for a variety of reasons. Some of these variables are the location of the defect in the tooth, the extensiveness of the defect, the location of the afflicted tooth in the mouth, the amount of stress placed on the filling, and the probability for contact with moisture during placement of the filling material. Amalgam use is favored over composite resins by differences in strength durability, ease-of-placement, and the lower cost between mercury amalgam and alternatives. Amalgams resist dissolution and wear better, require a less precise technique during placement, and are lower in cost.

Mercury use by the dental profession decreases each year. Reduction of mercury used by the dental profession has resulted from an increased emphasis on prevention of dental decay. Fewer fillings are removed now than in previous years, and this trend is likely to continue. It has been suggested that with the technological progress being made in amalgam substitutes, these will become competitive and likely displace traditional amalgam within the next decade or two. However, since the general populace already has a great number of dental amalgam restorations in their teeth, the decreasing amount of amalgam discharge during replacement into the waste stream will remain a challenge for some time.

Finally, product substitutions are also expected to decrease the amount of mercury contained in such items as thermometers, thermostats, and solid state electronic equipment. For example, mercury thermometers and thermostats are being replaced with digital devices.

2.1.2 Process Modification for Mercury Cell Chlor-Alkali Plants

In mercury cell chlor-alkali plants, mercury is used as a flowing cathode in electrolytic cells. The mercury electrolytic cell consists of an electrolyzer and a decomposer. In the electrolyzer section, a brine solution (usually sodium chloride [NaCl]) flows concurrently with the mercury cathode. A high current density is applied between the mercury cathode and the carbon or metal anodes. Chlorine gas forms at the anode and an alkali amalgam forms at the mercury cathode. The amalgam is separated from the brine in a discharge end-box and then enters the decomposer section, where water is added. In the decomposer, the amalgam becomes the anode to a short-circuited graphite cathode resulting in formation of hydrogen gas and alkali metal hydroxide, and conversion of the amalgam back to mercury. The mercury is then recycled to the inlet end-box, where it reenters the electrolyzer. The major emissions of mercury from this process occur with the hydrogen gas, the end-box ventilation system and the electrolytic cell room ventilation air (U.S. EPA, 1973).

Mercury emissions from chlor-alkali operations can be eliminated by converting to the membrane cell process. Because the membrane cell process is more energy efficient (Fauh, 1991), the use of mercury cells is declining.

In the membrane cell, a synthetic cation exchange membrane separates the electrolytic reaction products. Chlorine gas is generated at the anode on one side of the membrane, and caustic soda and hydrogen gas are produced at the cathode on the other side. The membrane allows passage of only sodium ions from the anode to the cathode compartment. This produces caustic that is purer and more concentrated than that from other nonmercury technologies, such as the diaphragm cell. The solution

produced by membrane cells can be as much as 25 to 30 percent caustic by weight. This solution is then evaporated to obtain a 50 percent product (U.S. EPA, 1984).

When a mercury cell process is converted to a membrane cell process, certain parts of the process remain the same. Figure 2-1 presents a block diagram of the new process areas required when the mercury cells are replaced with membrane cells (Horvath, 1986). Because mercury levels exceeding 10 parts per million (ppm) in the brine system can greatly affect the membrane performance (O'Brien, 1983), a mercury removal system is required. The mercury removal process is needed until mercury is sufficiently purged from the brine (typically 1 or 2 years). The filters used for mercury removal can later be used for secondary brine treatment (Horvath, 1986). More detail on the conversion process is contained in the reference by O'Brien (1983).

The membrane cell process has lower electricity costs than the mercury cell process. As a result, there are electricity savings associated with plant conversion. As described in detail in Chapter 3 of this Volume, the estimated annual capital cost of converting the model plant to a membrane process from the mercury cell process (after deducting electricity savings) is about \$3.3 million, or about 12 percent of total annual expenditures. Additional savings would presumably also result from the avoidance of costs of recycling or disposing of mercuric wastes, although these management costs are unknown.

2.1.3 Materials Separation

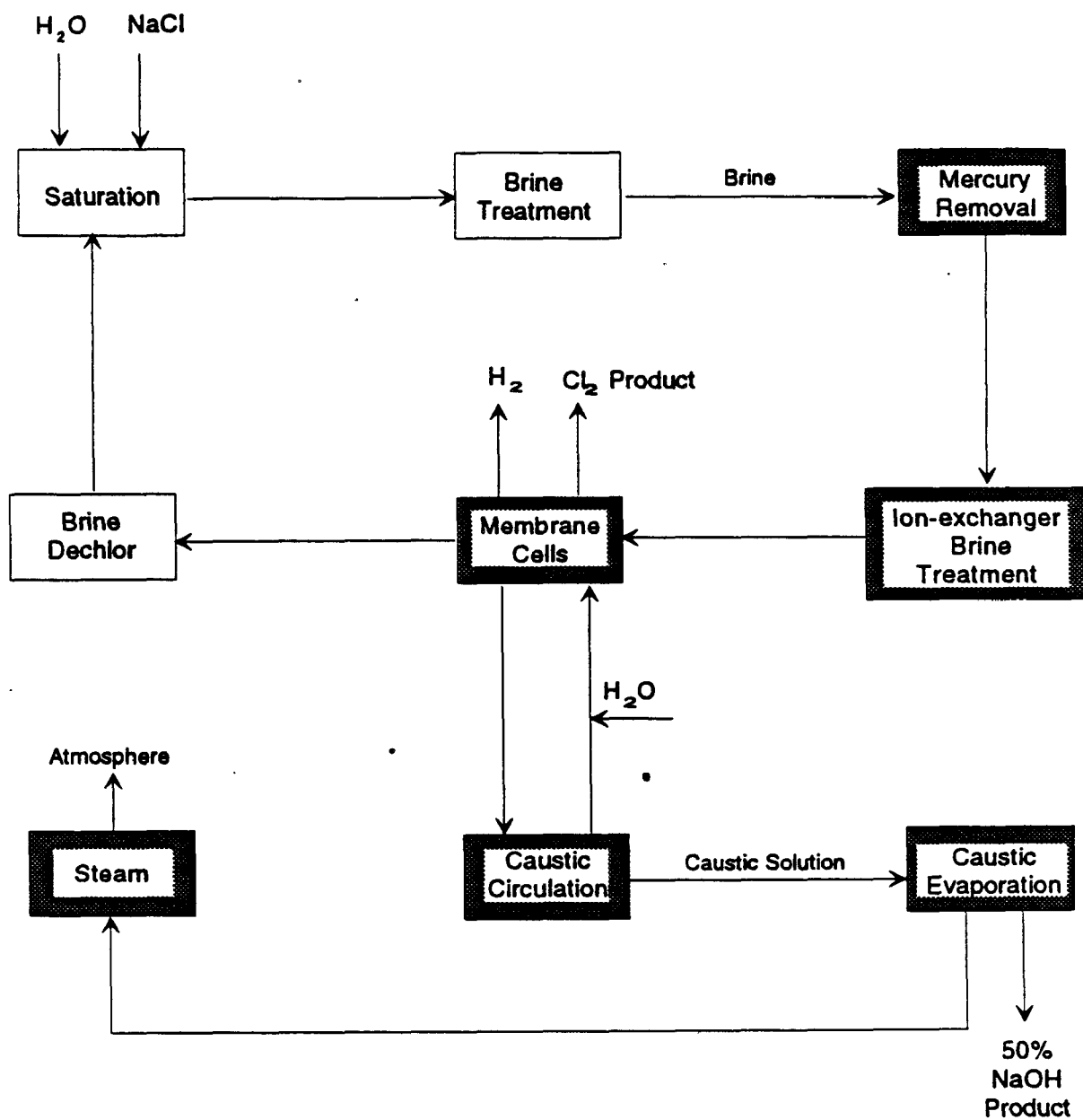
Unlike other metals, mercury has a high vapor pressure at the temperatures typically present in MWC and MWI air pollution control devices (APCD's). As a result, mercury does not condense as readily onto particulate matter (PM) and capture by the PM control device is highly variable. Separation of mercury-containing materials from the waste stream before combustion can reduce the amount of mercury in the exhaust gases from MWCs and MWIs (upstream of the APCD).

Many materials in wastes burned in MWCs and MWIs contain mercury. Materials that compose a large portion of the waste stream, such as paper, plastic, dirt and grit and yard waste (MWCs only), contain very low concentrations of mercury (U.S. EPA et al., 1992). Therefore, obtaining appreciable mercury reduction from separation of these types of materials would require separating a large fraction of the total waste stream. Separating these materials would counter the intended purpose of the combustion process, which is to disinfect and reduce the volume of waste materials.

Other materials contain higher concentrations of mercury, but make up only a very small portion (less than 1 percent) of the waste stream. These materials include mercuric oxide batteries, fluorescent lights, thermostats and other electrical items. Separation of such materials can reduce mercury input to a combustor without removing any of the energy content of the waste stream.

To evaluate a materials separation program, the feasibility and costs of separating a particular material should be compared with the mercury emission reduction achieved. Furthermore, the current and future mercury reduction achieved by separating a certain material should be considered since the mercury contribution of some materials such as household batteries has already declined considerably because of the substitution of mercury-free batteries. The pros and cons of separating several different materials are presented below.

Figure 2-1
Mercury Cell Replacement with Membrane Cells for
Chlor-Alkali Production Facilities



Existing Mercury Cell Process
 New Membrane Cell Process

2.1.3.1 Household Batteries

Batteries are usually the focus of materials separation efforts aimed at reducing mercury emissions. Household batteries, including alkaline and mercuric oxide batteries, are the largest source of mercury in MSW (U.S. EPA, 1992a). Both, the type and composition of batteries used in households affect the amount of mercury in the waste stream.

In 1989, household batteries accounted for an estimated 88 percent of all mercury from manufactured items in MSW and are expected to account for less than 68 percent by 2000 (U.S. EPA, 1992a). The expected mass of mercury from batteries discarded in MSW is expected to decline from 565 to 90 Mg (620 to 99 tons) by 2000 (U.S. EPA, 1992a; Shepherd, 1993). This decline is due largely to the reduction in mercury used in alkaline batteries (the type used, for example, in flashlights, toys and radios). In 1989, alkaline batteries contributed about 60 percent of the mercury from batteries in MSW. Because the use of mercury in alkaline batteries is rapidly declining, however, it is projected that alkaline batteries will contribute less than 1 percent to the total mercury from batteries by 2000 (U.S. EPA, 1992a; Shepherd, 1993).

By 2000, 99 percent of the battery-related mercury in MSW is expected to come from mercuric oxide button cells (the type used, for example, in hearing aids, watches and cameras) (U.S. EPA, 1992a). Therefore, any program to remove household batteries from MSW should target mercuric oxide button cells.

Several communities in the United States have implemented household battery separation programs in an effort to reduce mercury in the waste stream destined for MWCs. In Hennepin County, Minnesota, for example, small cardboard drop boxes are located at jewelry stores, camera shops and other establishments that sell button cells (Michaud, 1993). Customers can deposit used batteries in the box, free of charge. The batteries in the box are then collected by the county and sent to a vocational center for hand sorting to separate the batteries by type (mercury-zinc, silver-oxide, zinc-air and lithium). The mercury-zinc and silver-oxide batteries are sent to a company in New York to recover the metals in the batteries. A fee must be paid to the recycling company for acceptance of the mercury-zinc button cells, and a credit is provided for the silver button cells (Michaud, 1993).

Hennepin County also operates a program to separate other types of household batteries such as alkaline and carbon-zinc batteries. Collection barrels are located in shopping malls, drug stores, high schools and other locations. The county collects these batteries and ships them to a hazardous waste landfill, since there are no facilities in the United States for recycling them. Most other programs for separating button cells or larger household batteries in the United States are very similar to the Hennepin County program.

The proportion of batteries recovered through community materials separation programs in the United States (versus the proportion discarded in the MSW stream) is difficult to determine. Most estimates of the battery recovery efficiency of household battery separation programs, however, range from 3 to 25 percent (U.S. EPA, 1991b).

2.1.3.2 Medical Batteries

The concentration of mercury-zinc batteries in hospital waste may be even greater than in MSW, because industrial-type mercury-zinc instrument batteries are used in hospital diagnostic equipment (Shepherd, 1993; Cooper, 1993) and there is potential for a high proportion of elderly persons at hospitals to use hearing aids. Mercury-zinc instrument batteries contain over 20 times more

mercury per battery than button cells (14 grams (g) of mercury per cell versus 0.63 g of mercury per cell, respectively) (Shepherd, 1993). Industrial mercury-zinc batteries have been poorly characterized in most nationwide inventories of batteries in the waste stream. Therefore, they may be a more important source of mercury from MWIs than has been reported. Thus, the primary targets for separation at hospitals should be mercury-zinc button cells and instrument batteries.

Several successful programs have been implemented at hospitals to separate mercury-zinc button cells and instrument batteries from the waste stream. Initial results from these programs indicate that they are highly effective in removing mercury from the waste stream at hospitals (Shepherd, 1993). Results of emissions tests from the MWI at the Mayo Clinic in Rochester, Minnesota, showed a significant decline in mercury emissions after a battery separation program was established (White and Jackson, 1992). Similar types of separation programs designed to remove specific items containing high concentrations of mercury may also be feasible for some industrial, commercial, or military facilities.

2.1.3.3 Fluorescent Lights

Fluorescent lights accounted for about 3.8 percent of the mercury in MSW streams in 1989. It is projected that by 2000, fluorescent lights will account for about 8 percent of the mercury in MSW (U.S. EPA, 1992a). Currently, there are very few locations in the United States where the mercury from such lights can be recovered (in 1989, there were five mercury recycling facilities in the U.S.); however, the recycling capacity probably encompasses a majority of the lamps used in the U.S. In addition, recycling capacity is increasing as state regulations requiring recycling take effect. A program to separate such lights would be beneficial in reducing mercury emissions only if there are facilities available to accept and process the discarded lights in an environmentally sound manner. Because fluorescent lights are widely used in homes, schools, commercial facilities and other types of buildings, a program to separate, store and safely transport them to a processing facility would require a far-reaching program of education, training and cooperation throughout a community.

It also should be recognized that discarded fluorescent lights can break during handling, resulting in the direct release of mercury into the environment. Therefore, while separating fluorescent lights may reduce mercury inputs to waste combustors, consideration must also be given to the potential for mercury emissions from handling, storing, recycling and disposing of fluorescent lights by other means.

2.1.4 Coal Cleaning

In some states, certain kinds of coal are commonly cleaned to increase its quality and heating value. Approximately 77 percent of the eastern and midwestern bituminous coal shipments are cleaned in order to meet customer specifications for heating value, ash content and sulfur content (Akers et al., 1993).

There are many types of cleaning processes, all based on the principle that coal is lighter than the pyritic sulfur, rock, clay, or other ash-producing impurities that are mixed or embedded in it. Mechanical devices using pulsating water or air currents can physically stratify and remove impurities. Centrifugal force is sometimes combined with water and air currents to aid in further separation of coal from impurities. Another method is dense media washing, which uses heavy liquid solutions usually consisting of magnetite (finely ground particles of iron oxide) to separate coal from impurities. Smaller sized coal is sometimes cleaned using froth flotation. This technique differs from the others because it focuses less on gravity and more on chemical separation.

Some of the mercury contained in coal may be removed by coal cleaning processes. Volume II of this Mercury Study Report to Congress (*An Inventory of Anthropogenic Mercury Emissions in the United States*) presents available data on the mercury concentrations in raw coal, cleaned coal and the percent reduction achieved by cleaning. These data, which cover a number of different coal seams in four states (Illinois, Pennsylvania, Kentucky and Alabama), indicate that mercury reductions range from 0 to 64 percent, with an overall average reduction of 21 percent. This variation may be explained by several factors, including different cleaning techniques, different mercury concentrations in the raw coal and different mercury analytical techniques.

It is expected that significantly higher Hg reductions can be achieved with the application of emerging coal preparation processes. Proof-of-concept scale testing of selective agglomeration and advanced column flotation have indicated Hg removals as high as 80%. Bench-scale testing is also being carried out to investigate the use of naturally-occurring microbes to reduce the Hg (and other trace elements) from coal.

Any reduction in mercury content achieved by coal cleaning results in a direct decrease in mercury emissions from utility boilers firing cleaned coals. The mercury removed by cleaning processes is transferred to coal-cleaning wastes, which are commonly in the form of slurries. No data are available to assess the emissions of mercury from coal-cleaning slurries.

2.1.5 Other Management Measures

In addition to the pollution prevention measures and emission control technologies discussed above, there are a variety of flexible approaches for reducing the emissions of hazardous air pollutants. For example, under Title IV of the Clean Air Act Amendments of 1990, U.S. EPA established a new approach as part of its Acid Rain Program for limiting SO₂ and NO_x emissions from the electric-generating industry. This approach, built on the concept of allocating "emissions control allowances," represents a significant departure from the more traditional "command and control" approach to regulation. Overall, the emissions allowance approach will help improve efficiency and reduce compliance costs, while leaving intact the environmental benefits intended by regulation. Although this and other flexible programs administered by U.S. EPA have not been rigorously analyzed to determine their applicability and potential cost-effectiveness for mercury controls, such approaches are briefly described below.

2.1.5.1 Emissions Allowance Trading

Under emissions allowance trading, individual units that release SO₂ or NO_x from a utility are not assigned rigid emissions limits. Instead, each unit is allocated transferable emissions "allowances," each of which permits the holder to emit specified amounts of a pollutant. If the unit exceeds its allowances, it can still comply with the program by obtaining additional allowances from units whose emissions are smaller than their allowance allocations. Units with sources that are relatively inexpensive to control can reduce emissions more than would be required under uniform standards, to create a surplus of allowances. These surplus allowances can then be traded to other sources that are more costly to control, allowing these latter sources to reduce emissions less than would be otherwise required, while still achieving the same level of aggregate emission reductions.

A potential market for emissions allowances is created by this transferability, in which allowances may be bought, sold, auctioned and banked. The flexibility of this system will lower the costs of reducing emissions considerably, since the emissions reduction at the units with the lowest

costs of control will be able to substitute for the more costly emissions reduction by other units that would otherwise have been required.

A market for these allowances creates an incentive to develop more effective emissions control techniques. These incentives, in the long run, are intended to result in technological advances that make even tighter emissions standards feasible.

2.1.5.2 Emissions Averaging

In the emissions averaging program, sources for which emissions control is expensive need not meet a specific target, as long as other sources in the same "averaging group" reduce their emissions below their targets to compensate. When the necessary emissions reductions are made at other sources where emissions can be reduced at the lowest cost, the pool will meet the target "on average." If reducing emissions at one source is assumed to be equivalent to reducing emissions at any other source in the same averaging pool, an averaging program would likely result in a pure economic gain. Like emissions allowance trading, emissions averaging also could result in lower emission reductions. Further, emissions averaging can, in many instances, reduce the total cost of reaching a given emission limit.

2.1.5.3 The Opt-In Program

Though the Acid Rain Program is mandated only for utility sources, section 410 of Title IV provides opportunities for other SO₂ emitting sources to "opt in" to the program. The opt-in program is a voluntary economic incentive provision, developed to further help lower the cost of complying with acid rain limitations. Sources are given economic incentives to opt in if they can reduce emissions at costs lower than those for affected utility units. These reductions generate acid rain allowances that can then be transferred to utilities to meet mandatory reduction requirements in the utility sector. In this way, the program reduces the overall cost of the Acid Rain Program.

2.1.5.4 Energy Conservation and Renewable Energy

Title IV also encourages energy conservation measures and use of renewable energy as a long-term strategy for reducing air pollution and other adverse effects of energy production and use. Energy conservation is a cost-effective measure that increases the efficiency of the use of electricity provided by an electric utility to its customers. Renewable energy is defined as energy that is derived from biomass, solar, geothermal or wind.

As an incentive, qualified electric utilities receive allowances for SO₂ emissions avoided through implementing energy conservation measures or by the use of renewable energy. For example, for each ton of SO₂ emissions avoided by an electric utility through the use of qualified energy conservation measures or qualified renewable energy, the utility will be allocated a single allowance. These allowances can be banked, traded, or used for other sources that require expensive emissions control measures. U.S. EPA will allocate up to 300,000 allowances for all utilities from the Conservation and Renewable Energy Reserve.

2.1.5.5 Fuel Switching

Fuel switching refers to switching from one fuel to another (e.g., high-sulfur coal to low-sulfur coal, or coal to natural gas) for better environmental and economic benefits. For example, coal-

fired utilities could lower their mercury emissions by switching to a lower-mercury fuel (e.g., natural gas).

Installing pollution control equipment may not be cost-effective for sources that are not affected by Title IV regulations, which are generally smaller than affected utilities. Given the economic benefits of the opt-in program, fuel switching can be more cost-effective for such smaller sources.

2.2 Flue Gas Treatment Technologies

With the exception of mercury, most metals have sufficiently low vapor pressures at typical APCD operating temperatures that condensation onto PM is possible. Mercury, on the other hand, has a high vapor pressure at typical APCD operating temperatures, and collection by PM control devices is highly variable. Factors that enhance mercury control are low temperature in the APCD system (less than 150 °Celsius [°C] [300 to 400 °Fahrenheit (°F)]), the presence of an effective mercury sorbent and a method to collect the sorbent (Nebel and White, 1991). In general, high levels of carbon in the fly ash enhance mercury sorption onto PM. The ash is then removed by the PM control device. Additionally, the presence of hydrogen chloride (HCl) in the flue gas stream can result in the formation of mercuric chloride (HgCl₂), which is readily adsorbed onto carbon-containing PM. Conversely, sulfur dioxide (SO₂) in flue gas can act as a reducing agent to convert oxidized mercury to elemental mercury, which is more difficult to collect (Schager, 1990).

Add-on controls to reduce mercury emissions are discussed in the sections below. These include:

- Carbon filter beds;
- Wet scrubbing;
- Depleted brine scrubbing;
- Treated activated carbon adsorption;
- Selenium filters; and
- Activated carbon injection.

Each of these control technologies is described, including information on commercial status, performance, applicability to the specified mercury emission sources, and secondary impacts and benefits.

2.2.1 Carbon Filter Beds

2.2.1.1 Process Description

Carbon filter beds have been developed in Europe for use as a final cleaning stage to remove heavy metals (e.g., mercury), organic pollutants (e.g., dioxins and furans) and acid gases (e.g., SO₂, HCl) (Hartenstein, 1993a).

Three filter designs have been used in Europe: cross flow, counter current and counter-cross flow. The cross flow design has been applied to both utility boilers and waste combustors, the counter current design to waste combustors and the counter-cross flow design to utility boilers (Hartenstein, 1993a). Most of the information available on carbon filter beds pertains to the cross flow design. Therefore, this discussion focuses on cross flow design.

Figure 2-2 presents a basic diagram of the cross flow filters. The flue gas flows horizontally through the filter bed, while the adsorbent migrates through the filter from top to bottom. The bed is approximately 1 meter (m) (3 feet [ft]) thick and is divided into three layers. The thickness of each layer is approximately 100 to 150 millimeters (mm) (4 to 6 inches [in.]), 700 to 800 mm (28 to 31 in.), and 100 to 150 mm (4 to 6 in.), respectively. The first layer removes PM, heavy metals (including mercury), organic compounds and SO₂. Removal of HCl occurs primarily in the second layer. The third layer acts as a safety layer.

The three layers are separated by perforated metal sheets. On the outlet side, there is a slotted sheet designed to prevent particles from being carried out of the filter with the flue gas. Additionally, an impact separator is located downstream of the slotted sheet, functioning as a safety barrier against particle emissions (Hartenstein, 1993a; Hartenstein et al., 1991).

Fresh carbon is conveyed to and distributed within the bed by a screw conveyor on the top of the bed. Discharge cylinders at the bottom of each layer allow extraction of carbon from each layer. Pressure drop is the primary parameter for determining the rate of carbon removal from the bed. The designed pressure drop across the bed is 150 to 190 mm (6 to 7.5 in.) of water. The pressure drop across the whole system including fans and ducting is approximately 305 mm (12 in.) of water. Because of dust collection in and compaction of each layer, approximately 100 to 130 mm (4 to 5 in.) of material is periodically sloughed from each layer. Because of greater vibration of particles and, thus, greater settling of the particles within the first layer, carbon is sloughed from this layer on the shortest time interval, typically once every 6 to 8 hours. The second layer is sloughed once per day, and the third layer is sloughed once every 2 weeks. Based on these removal rates and bed thicknesses, the mass of carbon in the filter is fully replaced at an average rate of once per year. For MWCs, this equates to approximately 2 kilograms (kg) carbon/Mg (4 pounds [lb] carbon/ton) of MSW burned (Hartenstein, 1993a).

Spent carbon can be disposed of by combustion if the unit is equipped with a wet scrubbing (WS) system. The combustion process destroys the organic compounds captured in the carbon, and the wet scrubber collects the heavy metals and acid gases. Another disposal option more applicable to U.S. sources equipped with dry or semidry flue gas cleaning systems is to dispose of the carbon in a landfill. Because of the adsorbed pollutants, this waste may require disposal as a hazardous waste. Another option is to desorb the pollutants from the carbon. A pilot-scale desorption project is being constructed in Germany (Hartenstein et al., 1991; Hartenstein, 1990).

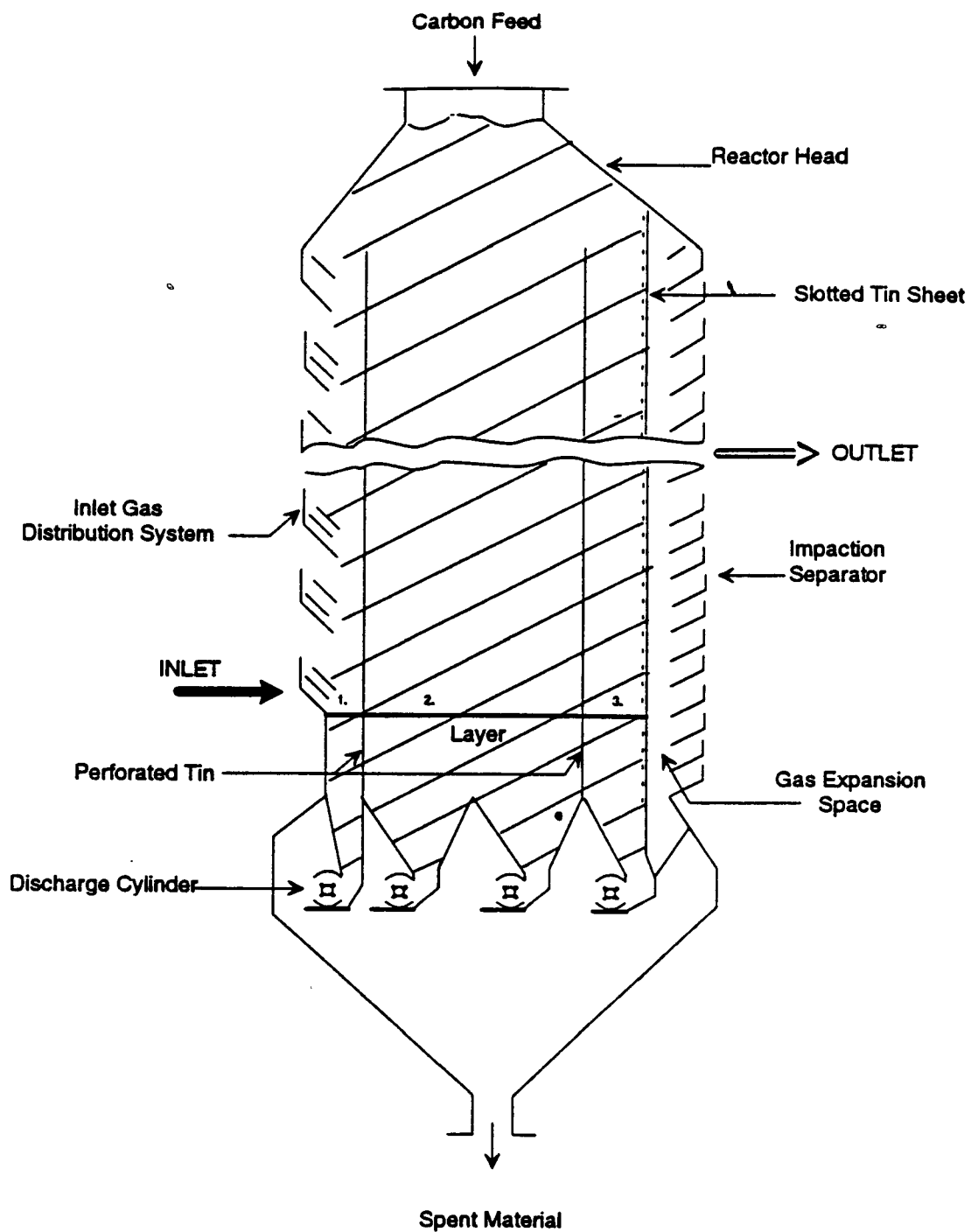
The size of a carbon filter bed is dependent on flue gas flow rate. The size of an individual carbon filter bed is limited by the height of the bed and the length of the screw conveyors and the discharge cylinders. To accommodate larger volumes of flue gas, a unit can include multiple beds, in which case each bed acts as a module within the filter. Filters with flow rates between 5,400 and 270,000 dry standard cubic meters per hour (dscm/hr) (190,000 and 9,500,000 dry standard cubic feet per hour [dscf/hr]),¹ containing one to eight modules, are in operation in Germany (Hartenstein, 1993a).

2.2.1.2 Commercial Status and Performance

Carbon filter beds have been used on power plants in Germany since the late 1980s. There are currently five full-scale applications of carbon filter beds on power plants in Germany. The

¹ At 11 percent oxygen (O₂), on a wet basis.

Figure 2-2
Carbon Filter Bed Design



Source: Hartenstein, 1993a

primary purpose of these filters is to remove residual SO₂ downstream of flue gas desulfurization (FGD) systems to prevent formation of ammonium sulfate in low-dust selective catalytic reduction systems. In one of the applications, a second filter module with catalytically active coke is used with ammonia injection for the catalytic reduction of nitrogen oxides (NO_x) (Hartenstein, 1993a). Although mercury control was not the primary purpose for installing the carbon filter beds on the boilers, reduction of mercury is inherent to the control system. A mercury level of under 1 microgram (µg)/dscm has been guaranteed by one vendor (Petersen, 1993).²

Application of carbon filter bed technology to MWCs in Europe was undertaken primarily for the purpose of reducing emissions of residual organic compounds, in addition to reducing heavy metals and acid gases. Mercury emissions from a pilot-scale MWC application in Germany were below the detection limits, corresponding to greater than 99 percent removal. Organic compounds were also removed by over 99 percent. No data were available on acid gas removal (Hartenstein, 1993). A full-scale application on a German industrial hazardous waste incinerator was installed in 1991, and low emissions of metals, organic compounds and acid gases were recorded (less than 1 µg/dscm [less than 0.44 grains (gr) per million dscf] for mercury, cadmium and lead; less than 0.6 nanograms [ng]/dscm [0.26 gr/billion dscf] for dioxins and furans; less than 1 ppm by volume (ppmv) for SO₂ and HCl). Inlet measurements, which would help characterize the hazardous waste stream, were not made (Hartenstein, 1993b). Without these measurements, the data cannot be used to directly assess the performance of the carbon filter beds. As noted, a mercury emission rate of under 1 µg/dscm has been quoted by a vendor (Petersen, 1993).

Future European installations of carbon filter beds on waste combustors include 5 units for hazardous waste incinerators and 11 for MWCs (Hartenstein, 1993a).³

2.2.1.3 Applicability

In Europe, carbon filter beds have been installed on waste incinerators and utility boilers. Carbon filter beds are also technically feasible applications to sources such as MWIs or smelters. This technology, however, has not been applied to these sources. Carbon filter beds have not been put into commercial practice in the U.S. for any of these source types.

2.2.1.4 Secondary Impacts and Benefits

In addition to mercury removal, carbon filter beds provide removal of residual organic compounds, other heavy metals and acid gases.

A potential negative impact associated with the carbon filter technology is disposing of the spent carbon. Options for disposal include burning the carbon or disposing of it in a hazardous waste landfill. Based on revolatilization data from ash samples collected during activated carbon injection testing, mercury revolatilization from the ash is not expected (Kilgroe et al., 1993). As a result, this potential negative impact may not be significant. Costs associated with hazardous waste disposal, however, must be considered.

² All concentrations in this report are corrected to 7 percent O₂ unless otherwise noted.

³ These totals are for the Steinmuller-Hugo Petersen cross flow filter. All of the units are in Europe. Additional units manufactured by other vendors may also be planned.

Another concern with this technology is the formation of "hot spots" in the bed that can result in bed fires. The cross flow filter is designed to evenly distribute the flue gas throughout the bed and to constantly remove excessive heat from the bed. To monitor hot spot formation, carbon monoxide concentrations in the flue gas are monitored upstream and downstream of the coke bed, and grate temperature measurements are taken. Gas probes are used to pinpoint any hot spot that forms. This system gives operators sufficient time to evaluate and troubleshoot problems (Hartenstein, 1990).

Another possible negative impact associated with this technology, or any carbon technology, is the potential release of mercury or other emissions during the coal-charring segment of the carbon activation processes. During the charring segment, when all volatiles are vaporized from the coal, mercury in the coal will be released. The level of mercury release is insignificant compared with the amount of mercury removed from the flue gas when using carbon filter beds. For example, if a mercury concentration in coal of 1 ppm and a 35 percent yield of activated carbon from a unit of bituminous coal is assumed (Rester, 1993), the mercury released during coal charring is less than 0.4 percent of the mercury captured using carbon filter beds. (Note that a 1 ppm level of mercury in coal is at least an order of magnitude higher than found in raw coal in the U.S. Thus, potential emissions would be commensurately lower.) Furthermore, the estimate of mercury emitted during charring assumes that all the mercury is released to the atmosphere. One activated carbon producer routes flue gas from the charring and activation process to a combustion unit and then through a wet scrubber (Rester, 1993). It can be assumed in this case that some of the mercury resulting from the charring is captured and not emitted to the atmosphere.

2.2.2 Wet Scrubbing

2.2.2.1 Process Description

Wet scrubbing (WS) systems can be used to control acid gases, metals, PM and dioxins and furans. Depending on the intended purpose of the WS system, different designs are available and include:

- Single-stage scrubbers designed primarily for control of acid gases;
- Two-stage scrubbers designed for control of acid gases and metals;
- Three-stage systems consisting of a two-stage scrubber followed by a high-efficiency wet particulate control system designed for improved control of fine particulates, metals and dioxins and furans; and
- Single-stage scrubbers, referred to as "polishing" scrubbers, that are installed following other APCD's and are designed for increased pollutant control (polishing).

Single-stage alkaline scrubbers are used on utility and industrial boilers for FGD, as well as on some small MWCs and MWIs. The effectiveness of such WS systems for mercury control on boilers is unclear at this time, but it is theorized that sulfites in the alkaline scrubber may act as a reducing agent to convert ionic mercury compounds into elemental mercury (Schager, 1990; Lerner, 1993a). Elemental mercury is not soluble in water and, therefore, is not retained by the scrubber solution.

This section focuses on single-stage polishing scrubbers, a type designed specifically for use as an add-on control for enhanced removal of mercury and other pollutants (PM, dioxins and furans).

The polishing WS system includes a single-tower water scrubber, which is installed at the outlet of an existing APCD such as a spray dryer/fabric filter (SD/FF). Soluble species of mercury compounds, such as HgCl_2 and mercuric oxide (HgO), can be effectively captured (greater than 90 percent) in the wet scrubber. If there is significant elemental mercury in the flue gas, however, collection efficiencies will be limited. The captured mercury is precipitated out during wastewater treatment through the use of any of several additives (e.g., sodium sulfide, trimercapto-s-triazine [TMT-15] and dithiocarbamate) (Reimann, 1993).

2.2.2.2 Commercial Status and Performance

Polishing WS systems for the purpose of enhanced mercury control and advanced two- and three-stage WS systems have not been applied to MWCs or boilers in the United States, although these WS polishing systems have been applied to MWCs in Europe. There is one application on an MWI in the United States (Lerner, 1992). Performance data are not available for a polishing wet scrubber on the MWI, but are available for three-stage systems and indicate mercury removal efficiencies of over 90 percent. Discussions with a vendor of WS systems indicated that a polishing scrubber on an MWC could reduce mercury by over 90 percent (Nebel et al., 1994).

2.2.2.3 Applicability

Polishing WS systems can be applied on MWCs, MWIs and boilers downstream of other acid gas and PM controls. As noted above, a factor determining effectiveness of this control is the amount of water-soluble mercury in the flue gas stream. For MWCs and MWIs, there is typically enough HCl in the flue gas to form HgCl_2 as the predominant mercury species.

2.2.2.4 Secondary Impacts and Benefits

As noted above, the use of WS systems requires treatment of wastewater before its disposal. The wastewater from WS systems on MWCs and MWIs generally contains more concentrated contaminants than other wastewater streams from the combustion of more homogenous fuels, such as coal. Even after standard neutralization and purification of the wastewater, excessive metal chloride complexes (including mercury) can remain. The use of TMT-15 or other similar precipitants can reduce these contaminants (Reimann, 1993). The manufacturer of TMT-15 (Degussa Corp.) claims that TMT-15 can reduce mercury levels in the wastewater to below 1 part per billion (Knivanek, 1993).

Zero water discharge is another alternative to handle the wastewater. Under this option, the scrubber water is evaporated using process heat, or in an upstream quench chamber or SD. It is still important to precipitate mercury out of the wastewater, so that revolatilization does not occur in the quench chamber or SD (Knivanek, 1993).

Another concern regarding WS systems is dechlorination of dioxins and furans isomers. Dechlorination raises concerns because the lesser-chlorinated dioxins and furans congeners are believed to be more toxic. Shifts in congener profiles have been observed at three wet-scrubber-equipped MWCs in the United States (Lerner, 1992), along with two wet-scrubber-equipped MWCs in Europe (Vogg, 1990; Marklund et al., 1993). Based on test data from other European facilities, however, this shift does not appear to occur in all WS systems (Jones, 1993). It is theorized that a reagent can be added to the scrubber water to control dioxin and furan shifts across all isomer classes (Knivanek, 1993).

2.2.3 Depleted Brine Scrubbing

2.2.3.1 Process Description

Depleted brine scrubbing is a WS process used to further reduce mercury emissions from mercury-cell chlor-alkali plants, specifically from the hydrogen and end-box ventilation gases. This scrubbing system is installed downstream of a primary and secondary cooling system, which includes a knockout drum and mist eliminator (U.S. EPA, 1973).

The depleted brine scrubbing technique uses discharged brine⁴ from the chlorine cell as a scrubbing liquor in a sieve plate tower or in a packed bed scrubber. Upon contact with the brine scrubbing solution, mercury vapor and mist form soluble mercury complexes. The mercury is subsequently recovered by electrolysis when the scrubbing solution is returned to the mercury chlor-alkali cell. Some systems are designed with an alkaline scrubber installed downstream of the brine scrubber to remove entrained chlorine and acid gases (U.S. EPA, 1973).

2.2.3.2 Commercial Status and Performance

Based on compliance test results provided in a 1984 U.S. EPA report, a brine scrubber installed to control mercury from the end-box stream of a chlorine plant with a 100 Mg/day (110-tons per day [tpd]) production capacity had mercury emissions of 48 g/day (0.11 lb/day) and 16 g/day (0.04 lb/day) during two tests (U.S. EPA, 1984). A 1973 study reported mercury emissions from a depleted brine scrubber followed by an alkaline scrubber to be 4.5 g/day (0.01 lb/day) for a 90-Mg/day (100-tpd) chlorine production facility (U.S. EPA, 1973). Without the alkaline scrubber, mercury emissions of 270 to 410 g/day (0.16 to 0.9 lb/day) were reported (U.S. EPA, 1973). More current information was not available.

2.2.3.3 Applicability

This technique is specific to chlor-alkali plants only. It cannot be used for other mercury source categories.

2.2.3.4 Secondary Impacts and Benefits

Very little information is available on this control technique. No data on secondary impacts or benefits from this system were identified.

2.2.4 Treated Activated Carbon Adsorption

2.2.4.1 Process Description

This mercury control technique uses a packed bed of sulfur- or iodine-impregnated carbon to reduce emissions of elemental and oxidized mercury compounds. This technique is similar in concept to the carbon filter bed (section 2.2.1) but is smaller in scale and does not involve the continual addition and sloughing of carbon. This technique has been applied to the hydrogen and end-box streams of chlor-alkali plants. The carbon adsorption bed is typically placed after the primary and

⁴ This depleted brine contains about 250 g (0.55 lb) per liter of NaCl and 0.6 to 0.9 g (0.001 to 0.002 lb) per liter of available chlorine. The pH of the brine solution is typically 2 to 4.

secondary cooling and mist elimination systems, and removes about 90 percent of the mercury content of the stream. The remaining mercury vapor is adsorbed by the carbon and chemically reacts with sulfur or iodine to form stable mercury compounds. Reduction of mercury to levels of 5 to 10 µg/cubic meter (m³) (at actual O₂ levels) was reported (U.S. EPA, 1973). The treated activated carbon can adsorb from 10 to 20 percent of its weight in mercury before it requires replacement (U.S. EPA, 1973). Several adsorber beds can be placed in series (U.S. EPA, 1984).

2.2.4.2 Commercial Status and Performance

Carbon filter beds have had widespread use in the chlor-alkali industry. Based on a 1984 mercury report (U.S. EPA, 1984), carbon bed systems were in use at 8 of the 20 chlor-alkali plants in operation at that time. Emissions of mercury reported for the seven plants equipped with the carbon beds on their hydrogen streams ranged from 1 to 264 g/day (0.002 to 0.58 lb/day). At the one plant with a carbon bed on the end-box stream, mercury emissions ranged from 5 to 120 g/day (0.01 to 0.26 lb/day) (U.S. EPA, 1984). These results are from plants of different capacities; therefore, the results cannot be compared directly. Additionally, there may be differences between the cooling systems at the plants and the designs of the carbon beds, which could affect emissions. In general, carbon bed systems should remove 90 percent of the mercury in the stream (U.S. EPA, 1973).

2.2.4.3 Applicability

This technology has been applied to hydrogen and end-box streams from chlor-alkali plants. It cannot be used for other mercury source categories.

2.2.4.4 Secondary Impacts and Benefits

Very little information is available on this control technique. No data on secondary impacts or benefits from this system for chlor-alkali plants were identified.

2.2.5 Selenium Filters

2.2.5.1 Process Description

Selenium filters have been developed to reduce elemental mercury emissions. The filters operate based on the affinity between mercury and metallic selenium. The mercury-laden flue gas passes through the filter, which is constructed of ceramic grains impregnated with metallic selenium. The gas pathway through the filter is tortuous, which increases the contact between the mercury and the selenium, forming mercury selenite (HgSe) (Edlund, 1993a). Standard filter sizes range from 535 to 53,500 standard cubic meters per hour (scm/hr) (18,900 to 1,890,000 standard cubic feet per hour [scf/hr]).⁵ For flows larger than 53,500 scm/hr (1,890,000 scf/hr), multiple filters in parallel can be used (Edlund, 1993b).

Selenium filters are effective on flue gas streams with inlet mercury concentrations of up to 9 milligrams (mg)/scm (3,900 gr/million scf). At higher mercury concentrations, the lifetime of the filter is short and an alternative control system, such as wet scrubbing, is recommended. At very low mercury concentrations, the removal efficiency decreases because of reduced mercury-selenium molecular collisions (Edlund, 1993a).

⁵ Oxygen levels not provided for flows or concentrations.

Based on vendor-provided information, a residence time of more than 2 seconds is recommended to achieve a constant degree of removal. Additionally, a flue gas temperature of less than 120°C (250°F) is recommended for maximum removal. Up to about 120°C (250°F), the removal efficiency remains fairly constant, but decreases at higher temperatures. With an inlet mercury concentration of less than 9 mg/scm (3,900 gr/million scf), a residence time of 4 seconds and a flue gas temperature of 120°C (250°F), an outlet mercury concentration of less than 40 µg/scm (17 gr/million scf)⁶ is claimed by the vendor (Edlund, 1993b).

Parameters to be considered when applying selenium filters include the dust content of the flue gas and the pressure drop through the filter. A maximum dust content of 9 mg/scm (3,900 gr/million scf) is recommended.⁶ Higher levels of dust necessitate frequent washing of the filter. The normal arrangement of the filter allows for a pressure drop of about 50 mm (2 in.) of water; however, it might be possible to accommodate higher pressure drops through alternate designs (Edlund, 1993a).

Because the removal of mercury in the filter is based on the formation of HgSe, the selenium in the filter is eventually exhausted. The selenium filter is designed to convert approximately 50 kg of mercury to HgSe per cubic meter of filter material (3 lb/cubic [ft³]) (Edlund, 1993a). The combination of pressure drop, mercury content in the flue gas and the mechanical construction of the filter determine the ultimate lifetime of the filter. On average, the filter lifetime is 5 years, after which the filter element is replaced (Edlund, 1993b). Once the lifetime of the filter mass has expired, the HgSe mass is landfilled (it is not combustible). Mercury selenite is a very stable compound, and the filter vendor indicated that laboratory leach tests showed negligible leaching, although no leaching data were provided. Also it is not clear whether mercury is recoverable through retorting (Edlund, 1993c).

2.2.5.2 Commercial Status and Performance

Selenium filters were first developed in Sweden in the 1970s for use on smelter flue gas streams, upstream of sulfuric acid plants (Lindquist, 1992). There is also one application on a crematory in Sweden. In total, there are four applications of selenium filters (Edlund, 1993c), all located in Sweden. Performance data were not available for any of the installations. Vendor information, however, indicates that mercury emissions less than 40 µg/scm (17 gr/million scf)⁷ are achievable (Edlund, 1993b). Since the application of selenium filters is limited and performance is based solely on vendor information, this technology should be viewed as an indicator of what may be feasible. This evaluation of this technology is not intended to be definitive.

2.2.5.3 Applicability

While use of this technology has been demonstrated in Sweden for metal smelters and crematories, it has not been demonstrated for a utility boiler. Use of these filters on MWCs and MWIs may also be technically feasible, although this application was not cited by the vendor.

⁶ Oxygen levels not provided for concentrations.

⁷ Oxygen levels not provided for concentrations.

2.2.5.4 Secondary Impacts and Benefits

A selenium filter consists of ceramic grains impregnated with metallic selenium. More information is needed about the possibility of selenium being emitted from the filter itself, particularly in the presence of hydrogen fluoride. Hydrogen fluoride gas from waste incineration could potentially generate volatile selenium compounds such as selenium hexafluoride. No data were located, however, to confirm this hypothesis.

2.2.6 Activated Carbon Injection

2.2.6.1 Process Description

Activated carbon injection involves the injection of powdered activated carbon into flue gas upstream of an APCD. Activated carbon is a specialized form of carbon produced by pyrolyzing coal or various hard, vegetative materials (e.g., wood) to remove volatile material. The resulting char then undergoes a steam or chemical activation process to produce an activated carbon that contains multiple internal pores and has a very high specific surface area. With this internal pore structure, the activated carbon can adsorb a broad range of trace contaminants, including mercury. After injection into the flue gas and adsorption of mercury and other contaminants, the activated carbon is captured in the PM control device.

The factors affecting the performance of activated carbon injection are the temperature of the flue gas, the amount of activated carbon injected, the concentration and species of mercury in the flue gas, the extent of contact between the carbon and mercury and the type of carbon used. Flue gas temperature, as noted above, is important because mercury is a vapor at temperatures above 150 to 200°C [300 to 400°F]. The flue gas temperature needs to be within, or preferably below, this range for the mercury to adsorb onto the carbon. The combustion device and the corresponding composition of the flue gas will affect this temperature range. In an MWC, where there is a substantial amount of HCl resulting in the formation of HgCl₂, temperatures within and below the noted range have proven to be effective when injecting carbon (U.S. EPA, 1992b; Kilgroe et al., 1993; Richman et al., 1993). Test data from an MWC retrofitted with activated carbon injection indicate mercury removals greater than 95 percent (Ogden Martin, 1994). Pilot testing on a coal-fired utility boiler indicated that a temperature under 90 to 120°C (200 to 250°F) was necessary for effective mercury removal (Chang et al., 1993).

With activated carbon injection, efficient distribution of the carbon in the flue gas is also important. The amount of carbon needed to achieve a specific level of mercury removal will vary depending on the fuel being burned, the amount of carbon inherent to the system and the type of PM control device. At a given carbon feed rate, a FF provides more mercury control than an electrostatic precipitator (ESP) because of the additional mercury adsorption that occurs on the bags of the FF (due to the increased gas contact time). As a result, an ESP-equipped facility may require a higher carbon feed rate to achieve the same level of control as a FF-equipped facility.

With chemically impregnated activated carbon, the contaminant reacts with the chemical that is bound to the carbon, and the carbon is removed by an APCD. For instance, with sulfur-impregnated carbon, mercury and the impregnated sulfur react to form mercury sulfide on the carbon. One pilot-scale study suggests that mercury removal efficiencies for sulfur- and iodide-impregnated carbons were 25 to 45 percent greater than those achieved with an equivalent amount of nonimpregnated carbon (Felsvang et al., 1993). Other studies however, have shown that the

effectiveness of different carbons is dependent on the mercury species, and in some cases, activated carbon had better mercury removal than iodated carbon (Miller, 1995; Krishnan, 1994).

2.2.6.2 Commercial Status and Performance

Activated carbon injection is being used commercially on one MWC in Europe and has been tested in the United States on three MWCs, two MWIs and a 1-megawatt (MW) slip stream from a coal-fired boiler. The U.S. test programs have shown mercury removals of 50 to over 95 percent, depending on the carbon feed rate (U.S. EPA, 1992b; Kilgroe et al., 1993; Richman et al., 1993; Chang et al., 1993; U.S. EPA, 1991c; U.S. EPA, 1992d). As of January 1995, there were four MWCs in the U.S. that had been retrofitted with activated carbon injection. Test results from the Hennepin County Energy Resource Company in Minnesota indicate that this technology is achieving mercury reductions greater than 95 percent. Currently, research is being done on activated carbon injection at two pilot-scale utility boilers. Tests on a full-size electric utility boiler are planned by the Electric Power Research Institute.

2.2.6.3 Applicability

Activated carbon injection is applicable to MWCs, MWIs and utility boilers. There are considerable data describing the efficacy of activated carbon injection for controlling mercury emissions from MWIs and MWCs. These data have been previously published by U.S. EPA and are not discussed in detail in this report. The reader is encouraged to review the Background Information Documents for MWIs and MWCs for activated carbon test results (U.S. EPA, 1989a; U.S. EPA, 1989b; U.S. EPA, 1994d; U.S. EPA, 1994e). Activated carbon injection test results for utility boilers have not been previously published by U.S. EPA. To inform the reader, therefore, a detailed discussion of the applicability of activated carbon injection to utility boilers is presented in Section 2.3. Activated carbon injection may also be technically feasible for smelters. No information, however, is available on the use of activated carbon injection for smelters.

2.2.6.4 Secondary Impacts and Benefits

In addition to removing mercury, injection of activated carbon will increase the removal of chlorinated dioxins and furans and potentially other semivolatile organics. Data from tests with activated carbon injection on an MWC, equipped with an SD/ESP, show 95 to 98 percent reduction of dioxin and furan emissions, versus a reduction of 78 to 80 percent without carbon injection (Richman et al., 1993). Similar dioxin and furan removal efficiencies were achieved during a test on an SD/FF-equipped MWI (U.S. EPA, 1991c).

The addition of carbon to MWCs and MWIs has a minimal impact on the quantity of collected PM requiring disposal. By comparison, the injection of activated carbon into a utility flue gas stream could have a significant impact on the quantity of PM requiring disposal. For example, as described in Appendix B, a 100 MW coal-fired boiler with an ESP could potentially inject almost 490 tons of activated carbon per year.

Testing conducted on ash samples collected during carbon injection testing at a U.S. MWC demonstrated the mercury collected on the carbon was stable at temperatures typical of landfills. As a result, there is a low potential for the mercury to be reemitted to the atmosphere (U.S. EPA, 1992b).

The issue of mercury released during the production of activated carbon, as discussed above, is also applicable to activated carbon injection. As with carbon filter beds, the amount of mercury

released during the charring is very small compared with the amount of mercury captured by the injected carbon. The amount of mercury released during charring is estimated to be less than 0.2 percent of the amount of mercury captured in injected carbon. This is slightly less than for carbon beds because a smaller quantity of carbon is used in activated carbon injection.

2.3 Activated Carbon Injection for Utility Boilers

The effectiveness of activated carbon injection in controlling mercury emissions from MWCs has been demonstrated (U.S. EPA, 1989a; U.S. EPA, 1989b). The application of activated carbon injection to utility flue gas, however, cannot be directly scaled from the application at MWCs due to differences in the amount and composition of flue gas at utility plants and MWCs. At utility plants, small concentrations of mercury are contained in a large volume of flue gas, and large amounts of activated carbon may be needed to provide adequate contact between the carbon particles and mercury. The differences in flue gas characteristics at MWCs and utility plants must be carefully examined before considering any technology transfer assumptions. Because fewer references are available on the application of activated carbon injection to utility boilers, a greater level of detail is presented here for this application than for MWCs.

The level of mercury control achieved in utility flue gas may depend upon flue gas characteristics such as volume, temperature, and chloride and mercury content. These properties are distinctly different from those in MWC flue gas.

As shown in Table 2-2, typical MWC flue gas is hotter than utility flue gas after leaving an air preheater. The air preheater cools the utility flue gas by transferring heat to the incoming combustion air. Moreover, the mercury concentration of the two gas streams differs significantly. Mercury concentrations in MWC flue gas streams may be up to several orders of magnitude greater than those seen in utility flue gas streams. Likewise, the chloride content of MWC flue gas may be from 1.4 to 400 times greater than the content seen in utility flue gas. Finally, with regard to the volume of flue gas, a utility boiler may have flow rates up to 30 times that of an MWC.

Table 2-2
Comparison of Typical Uncontrolled Flue Gas Parameters at Utilities and MWCs^{a,b}

Uncontrolled Flue Gas Parameters	Coal-Fired Utility Boilers ^{c,d}	Oil-Fired Utility Boilers ^{d,e,f}	MWC ^{g,h}
Temperature (°C)	121 - 177	121 - 177	177 - 299
Mercury Content (µg/dscm)	1 - 25	0.2 - 2 ⁱ	400 - 1,400
Chloride Content (µg/dscm)	1,000 - 140,000	1,000 - 3,000	200,000 - 400,000
Flow Rate (dscm/min) ^j	11,000 - 4,000,000	10,000 - 2,000,000	80,000 - 200,000

^a Standard conditions are 0°C and 1 atmosphere.

^b Moisture content in the MWC flue gas was assumed to be 13.2 percent.

^c Radian Corporation, 1993a.

^d Heath, 1994

^e Radian Corporation, 1994.

^f Radian Corporation, 1993b.

^g Brown and Felsvang

^h Nebel and White, 1991.

ⁱ It is not known if oil-fired utility boilers release less mercury overall than coal-fired boilers because the mercury release during oil refining is essentially unstudied.

^j Min = minute

Because of differences in the amount and composition of flue gas at utility plants and MWCs, pilot-scale studies of activated carbon injection were conducted on utility flue gas. Preliminary results from a limited number of pilot-scale tests on utility flue gas are summarized in Figure 2-3 and presented in greater detail in Appendix A. These data indicate that the effectiveness of activated carbon injection varies with several factors. The mercury removal efficiency for fabric filter and activated carbon systems ranged from a low of 14 to 47 percent with a median of 29 percent (107-121°C, low carbon injection) to a high of 95 to 99 percent with a median of 98 percent (88-107°C, high carbon injection). When activated carbon injection was used ahead of a spray dryer absorber, mercury removal efficiency ranged from 50 to 99 percent with a median of 60 percent when a fabric filter was used for particulate control, and from 75 to 91 percent with a median of 86 percent when an ESP was used for particulate control.

2.3.1 Utility Flue Gas Factors Affecting Mercury Removal

The level of mercury control achieved in utility flue gas may depend upon flue gas characteristics such as temperature, volume, form of the mercury present, the possible effect of chloride concentration in coal on mercury speciation and the type of activated carbon injected into the flue gas. These factors are reviewed below.

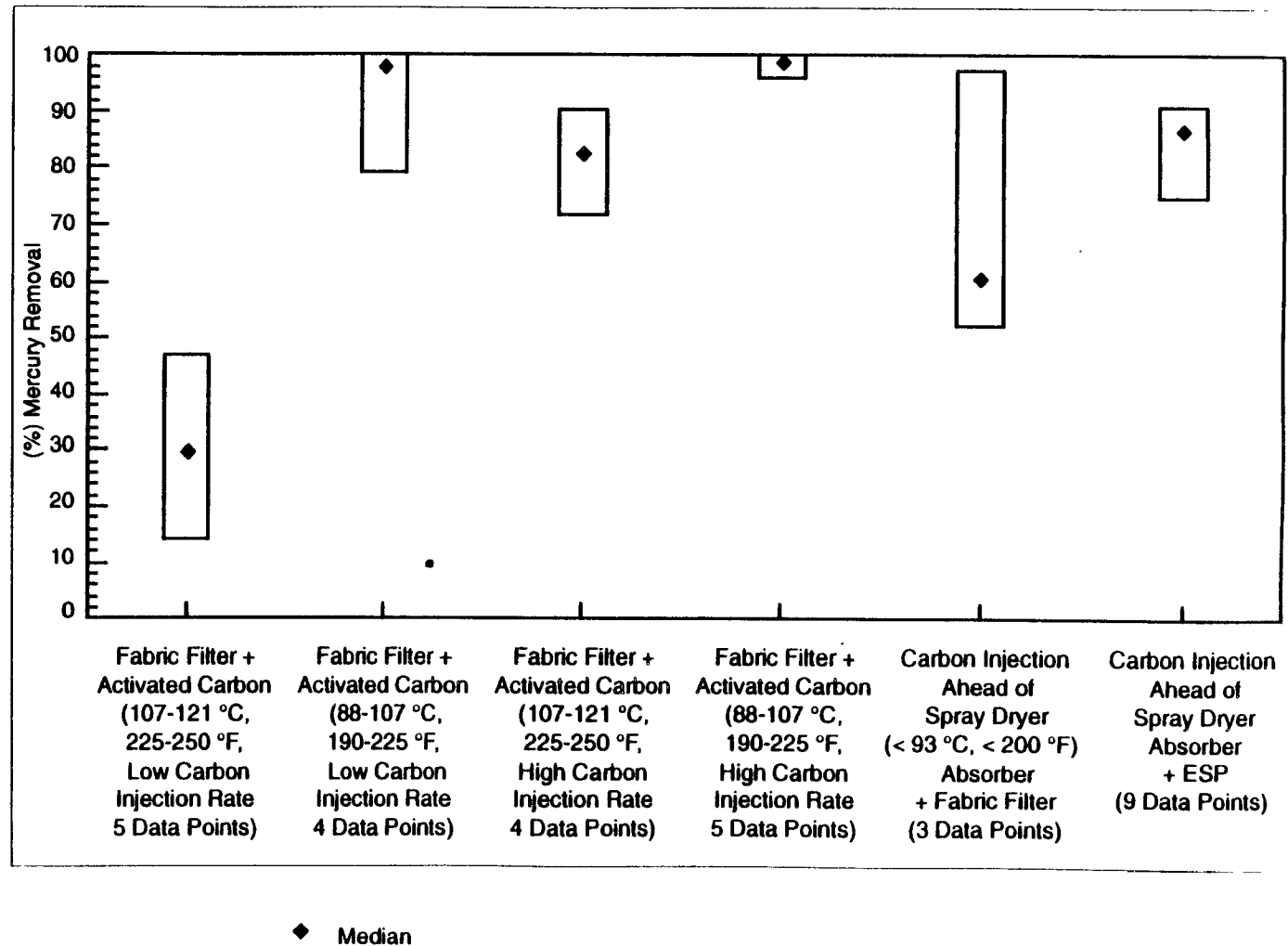
Temperature. Mercury is found predominantly in the vapor phase in utility flue gas (Clarke and Sloss, 1992). If the vapor-phase mercury were condensed onto PM, the PM could be removed with existing particulate control devices. Theoretically, cooler temperatures will increase mercury condensation onto PM (Clarke and Sloss, 1992) and, subsequently, increase mercury removal with existing PM control devices.

There is limited, preliminary evidence for the temperature dependence of mercury removal in a pilot-scale FF study. The pilot study suggests that mercury removal efficiencies apparently increase as the temperature of the flue gas decreases. Specifically, as the flue gas temperature decreased from 107 to 99 to 96°C (225 to 210 to 205°F), the mercury removal efficiency percentages for a pilot-scale FF correspondingly increased from 27 to 33 to 51 percent (Chang et al., 1993).

Mercury removal efficiencies and the required amount of activated carbon injection were apparently temperature dependent within a range of 88 to 121°C (190 to 250°F) in a pilot-scale study on the effect of reducing mercury levels in utility flue gas through activated carbon injection upstream of a FF (Chang et al., 1993). At the lower temperatures within this range (88 to 96°C [190 to 205°F]), mercury concentrations were reduced by 97.7 percent with an activated carbon injection rate of approximately 155 µg carbon/µg of inlet mercury, while at higher temperatures (110 to 121°C [230 to 250°F]) mercury concentrations were reduced by only 75 to 87 percent with an activated carbon injection rate of approximately 3,500 µg carbon/µg of inlet mercury.

The pilot-scale activated carbon injection study indicated that more mercury was removed and less carbon was needed at lower flue gas temperatures or, in other words, the ideal use of activated carbon is at lower flue gas temperatures. It may not be possible, however, to lower the flue gas temperature at a given utility plant because utility plants typically operate with a stack gas temperature between 121 and 177°C (250 to 351°F) upstream of any particulate control device to avoid acid condensation and, consequently, equipment corrosion. The stack gas temperature may be lowered below 96°C (205°F) and acid condensation be avoided provided low-sulfur coals (less than about 1 weight percent sulfur) are burned (McKenna and Turner, 1989). If a utility burns low sulfur coal and uses an ESP for particulate control, however, the flue gas will probably require conditioning to reduce

Figure 2-3
Comparison of Mercury Removal Efficiencies with Activated Carbon Injection



Source: Chang et al., 1993; Felsvang, 1993

the high resistivity of the fly ash because high resistivity makes the fly ash difficult to collect with an ESP.

Based upon the preliminary pilot-scale studies conducted at temperatures below 121°C (250°F), the least efficient use of carbon injection for mercury control is at higher temperatures with greater injection rates.

Volume. At utility plants, mercury control techniques must adequately treat the entire volume of gas in order to remove relatively small concentrations of mercury (0.2 to 21 µg/dscm, at 7 percent O₂).

Mercury Speciation and Type of Activated Carbon. The speciation of mercury will indicate potential ways to reduce mercury emissions. For example, the ionic mercury form (i.e., Hg⁺⁺) is water soluble and is less volatile than elemental mercury (i.e., Hg⁰). Thus, reducing the temperature of the flue gas and wet scrubbing of the flue gas may result in increased ionic mercury removal.

No validated methods are available at this time for quantifying mercury speciation in utility flue gas; however, preliminary results from an analytical technique that is under development by Bloom et al., (1993) are summarized in Table 2-3 for coal- and oil-fired utility flue gas streams.

As shown in Table 2-3, the distribution of ionic mercury, most likely HgCl₂ in coal-fired utility flue gas, ranged from 12 to 99 percent of the total mercury content and averaged 79 percent; the distribution of elemental mercury in coal-fired utility flue gas ranged from 0.8 to 87.5 percent of the total mercury content and averaged 21 percent. Analysis of two samples of flue gas taken from oil-fired boilers, however, suggests that mercury in oil-fired boiler flue gas is predominantly in the elemental form (see Table 2-3). The variability in the speciation of vapor-phase mercury in coal-fired flue gas may explain the variation in mercury removal that is seen with existing control devices (DeVito et al., 1993); however, validated methods are needed to establish the speciation of mercury before any relationship between mercury speciation and control device performance can be inferred.

The variability in the distribution of vapor-phase mercury species in coal-fired flue gas may depend upon the chloride concentration in coal. Using the analytical techniques developed by Bloom et al., (1993), it has been observed that higher concentrations of ionic mercury are obtained in utility flue gas when the combusted coal has a high chloride concentration (0.1 to 0.3 weight percent) (Felsvang et al., 1993; Noblett et al., 1993), but more data are needed to verify this association. The distribution of mercury species in coal-fired flue gas also appears to vary with the type of coal (e.g., bituminous, subbituminous, or lignite) (Chang, 1994; Boyce, 1994).

The association between chloride content of the fuel and the concentration of ionic mercury in the flue gas also may apply to fuel oil. This association, however, has not been examined.

Although the speciation of mercury is important in planning control strategies, it is still in the early stages of investigation. Studies of a pilot-scale wet FGD system treating coal-fired flue gas indicate that more than 90 percent of the ionic mercury was removed while hardly any of the elemental mercury was removed (Noblett et al., 1993). Similarly, studies at a pilot-scale SDA/ESP system treating coal-fired flue gas suggest that 95 percent of the ionic mercury and essentially none of the elemental mercury were removed (Felsvang et al., 1993). The effectiveness of activated carbon injection in recovering different forms of mercury is still being studied. Preliminary results are available, however, from two studies at three separate pilot-scale facilities.

Table 2-3
Mercury Speciation in 33 Samples of Coal-Fired and 3 Samples of Oil-Fired Utility Flue Gas

$\mu\text{g}/\text{Nm}^3^{\text{a}}$			Total ^b ($\mu\text{g}/\text{Nm}^3$)	Percent		Reference
Ionic	Other Hg Species ^c	Elemental		Total Ionic ^d	Elemental	
COAL-FIRED BOILERS						
5.17	0.29	1.6	7.06	77.34	22.66	e
4.24	0.41	0.59	5.24	88.74	11.26	e
2.96	0.41	0.31	3.68	91.57	8.42	e
2.84	0.56	0.15	3.55	95.77	4.23	e
8.7	---	0.29	8.99	96.77	3.23	f
8.6	---	0.2	8.8	97.73	2.27	f
10.8	---	0.22	11	98.00	2.00	f
7.86	---	0.06	7.92	99.24	0.76	f
7.7	---	0.07	7.77	99.1	0.90	f
9.4	---	0.1	9.5	98.95	1.05	f
7.7	---	0.08	7.7	98.97	1.03	f
6.03	---	1.94	7.97	75.66	24.34	g
6.46	---	1.29	7.75	83.35	16.65	g
3.03	---	1.52	4.55	66.59	33.41	g
5.36	---	1.54	6.9	77.68	22.32	g
5.98	---	2.54	8.52	70.19	29.81	g
3.5	---	1.67	5.17	67.7	32.30	g
3.1	---	1.72	4.82	64.32	35.68	g

Table 2-3 (continued)
Mercury Speciation in 33 Samples of Coal-Fired and 3 Samples of Oil-Fired Utility Flue Gas

$\mu\text{g}/\text{Nm}^3^{\text{a}}$			Total ^b ($\mu\text{g}/\text{Nm}^3$)	Percent		Reference
Ionic	Other Hg Species ^c	Elemental		Total Ionic ^d	Elemental	
2.99	---	1.51	4.5	66.44	33.56	g
2.83	---	0.88	3.71	76.28	23.72	g
3.33	---	1.26	4.59	72.55	27.45	g
3.83	---	1.52	5.35	71.59	28.41	g
2.2	---	1.57	3.77	58.36	41.64	g
---	---	---	---	12.5	87.5	h
7.37	0.0021	1.85	9.22	79.92	20.07	i
5.01 ^h	---	1.26	6.28	79.90	20.10	j
5.72	1.39	<0.1	7.10	98.61	1.39	j
4.49	0.40	0.89	5.78	84.60	15.40	j
2.93	0.07	0.16	3.15	94.94	5.06	j
2.79	0.32	1.02	4.13	75.30	24.70	j
5.71	1.37	2.99	10.22	70.31	29.69	j
7.01	1.02	0.11	8.13	98.65	1.35	j
4.11	0.62	2.32	7.05	67.09	32.91	j
1.14	0.30	3.12	4.56	31.58	68.42	j

Table 2-3 (continued)
Mercury Speciation in 33 Samples of Coal-Fired and 3 Samples of Oil-Fired Utility Flue Gas

µg/Nm ³ ^a			Total ^b (µg/Nm ³)	Percent		Reference
Ionic	Other Hg Species ^c	Elemental		Total Ionic ^d	Elemental	
COAL-FIRED BOILER DATA RANGES						
1.1 - 10.8	0.002 - 1.4	0.06 - 3.1	3.1 - 11	12 - 99	1.8 - 88	---
OIL-FIRED BOILERS						
0.03	0.01	0.15	0.19	21.05	78.95	j
<0.02	0.01	<0.02	---	---	---	j
0.02	0.01	0.09	0.12	25.00	75.00	j
OIL-FIRED BOILER DATA RANGES						
0.02 - 0.03	---	0.02 - 0.15	0.12 - 0.19	23	77	---

^a $\mu\text{g}/\text{Nm}^3$ = micrograms per normal cubic meter (at 0°C).

^b Value is a combination of ionic and other Hg species.

^c For other Hg species, the references list this value as an organic mercury species. According to Chang (1994), however, the organic mercury species values are invalid but could be added to the ionic species concentrations to give a total ionic mercury value.

^d The total ionic percentage represents the ionic and other Hg species concentrations.

^e Chang et al., 1993. Flue gas sampled at FF inlet.

^f Noblett et al., 1993. Flue gas sampled at wet scrubber inlet.

^g Felsvang et al., 1993. Flue gas sampled at inlet to pilot-scale spray dryer absorption (SDA)/ESP system.

^h Felsvang et al., 1993. Flue gas sampled at inlet to full- and pilot-scale SDA/FF system.

ⁱ Dismukes et al., 1993.

^j Bloom et al., 1993.

In testing at the first facility, a pilot-scale SDA/ESP system in Denmark (Felsvang et al., 1993), the flue gas contained from 66.6 to 83.4 percent ionic mercury, with an average of 75.2 percent ionic mercury, and elemental mercury comprised the remainder of the total mercury concentration in the flue gas. Without activated carbon injection, the pilot-scale SDA/ESP system removed 96.8 percent of the ionic mercury and essentially none of the elemental mercury from coal-fired flue gas or, in other words, the system removed 72.5 percent of the total mercury. During testing with activated carbon injection, the flue gas contained from 58.4 to 77.7 percent ionic mercury, with an average of 69.5 percent ionic mercury, and elemental mercury comprised the remainder of the total mercury concentration in the flue gas. Activated carbon injection ahead of the SDA/ESP system removed 46.4 percent of the elemental mercury and 84.3 percent of the total mercury (Felsvang et al., 1993).

In testing by Joy/Niro and Northern States Power at the second facility that had a full- and pilot-scale SDA/FF system, the flue gas contained 85 to 90 percent elemental mercury. Without activated carbon injection, the full- and pilot-scale SDA/FF systems removed 10 to 20 percent of the total mercury from the coal-fired flue gas (Felsvang et al., 1993), and the low removal of total mercury may be attributed to essentially complete removal of the ionic mercury and poor removal of the elemental mercury. Activated carbon injection ahead of the pilot-scale SDA/FF system increased the removal of total mercury to approximately 55 percent, and injection of iodide- and sulfur-impregnated activated carbon increased the removal of total mercury to approximately 90 percent (Felsvang et al., 1993). Thus, the studies at this SDA/FF system suggest that sulfur- and iodide-impregnated carbons are needed for total mercury removals of 90 percent, when elemental mercury is the predominant mercury species. Furthermore, the studies suggest that total mercury removal efficiencies are dependent upon mercury speciation.

Finally, recent tests conducted at a third facility, the pilot-scale combustor and FF at the University of North Dakota Energy and Environmental Research Center (UNDEERC), also suggest that mercury removal is dependent upon mercury speciation and the type of carbon used (Chang, 1994). The details of the tests conducted at UNDEERC were not available when this report was prepared.

2.3.2 Current Research on Activated Carbon Injection for Utilities

Currently, research is being done on activated carbon injection at a pilot-scale SDA/FF plant by Joy/Niro and Northern States Power Company (Felsvang et al., 1993), at a pilot-scale pulse-jet FF plant by the Electric Power Research Institute (EPRI) (Chang et al., 1993) and at a pilot combustor and FF plant by UNDEERC and EPRI (Chang, 1994). The EPRI plans to test activated carbon injection on a full-size electric utility boiler.

2.4 Emerging Technologies for Controlling Mercury Emissions from Utilities

Research continues on developing potential technologies for mercury emission reduction from utility plants. Although activated carbon injection has been shown to be a promising technology, research with impregnated activated carbons, sodium sulfide (Na_2S) injection and an activated carbon circulating fluidized bed (CFB) suggest that even greater mercury removal is possible.

With sulfur-impregnated activated carbon injection, the carbon-bound sulfur reacts with mercury to form mercuric sulfide (HgS) on the carbon, and the carbon is removed by a particulate control device. In a pilot-scale study, sulfur-impregnated carbon increased mercury removal to 80 percent, an increase of 25 percent over results achieved with an equal amount of nonimpregnated activated carbon (Felsvang et al., 1993).

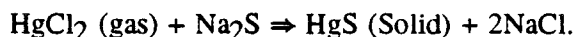
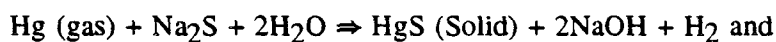
With iodide-impregnated activated carbon injection, the carbon-bound iodide reacts with mercury to form mercuric iodide (HgI_2) on the carbon, and the carbon is removed by a particulate control device. In a pilot-scale study, iodide-impregnated carbon increased mercury removal to nearly 100 percent, an increase of 45 percent over results achieved with an equal amount of nonimpregnated activated carbon (Felsvang et al., 1993).

Chloride-impregnated activated carbon injection has only been tested on MWCs in Europe. The chloride reacts with mercury to form HgCl_2 on the carbon, and the carbon is removed by a particulate control device. European MWC experiments have shown that impregnating activated carbon with chloride salts increases adsorptive capacity of the activated carbon 300-fold (New Jersey Department of Environmental Protection and Energy, 1992).

Another potential way for improving Hg collection efficiency is combining calcium hydroxide (Ca(OH)_2) with activated carbon. This reagent, consisting of approximately 95 to 97 percent lime and 3 to 5 percent activated carbon, is known under the product name Sorbalit® (Nebel and White, 1991). Sorbalit® has only been tested on European MWCs and MWIs.

Sulfur-, iodide-, chloride salt- and Ca(OH)_2 -impregnated activated carbons show promise for increasing the mercury removal efficiency, but further testing is needed. The cost of these modified carbons can be as much as 20 times higher than that of unmodified activated carbon (Maxwell, 1993).

Mercury reduction has been achieved at MWCs through the injection of Na_2S solution into the flue gas prior to the acid gas control device. The specific reactions of Na_2S and mercury (Hg) are not totally clear but appear to be (Nebel and White, 1991):



The resulting solid, HgS, can be collected by a FF.

There are several potential limitations to Na_2S injection. These include reaction of Na_2S with calcium (Ca) in the sorbent (as found in Sorbalit®) to form calcium sulfide (CaS), reduction of the amount of sulfur available to react with mercury (CaS can also cause scaling of the sorbent feed line), corrosion of ductwork (Na_2S is a corrosive material), clogging and plugging of the screw conveyor due to solidification of Na_2S and sludge formation due to the presence of inorganic salts in the mixing water (Nebel and White, 1991).

At present, full-scale operational injection of Na_2S has been done only in MWCs. No plans have been announced to test this technology on fossil fuel-fired electric steam-generating units.

Another potential process for the reduction of mercury emissions is the use of activated carbon in a CFB (Clarke and Sloss, 1992). In a CFB, the activated carbon is continuously fed to the reactor where it is mixed with the flue gas at a relatively high velocity, separated in the subsequent FF and recycled to the reactor. A small part of the used activated carbon is withdrawn from the process and replaced by fresh material (New Jersey Department of Environmental Protection and Energy, 1992).

The main advantages to CFB's over fixed carbon beds are the increased flue gas-to-carbon contact area and the smaller overall pressure drop. No pilot- or full-scale utility boiler testing has yet been done with this system, although it has been used in Germany for MWC operation.

Additional developing technologies for utility boilers also include a condensing heat exchanger, wet FGD additives, gold-impregnated monoliths, and advanced coal cleaning.

3. COST AND FINANCIAL IMPACTS OF MERCURY CONTROLS

Several technologies are available for the control of mercury emissions. This section summarizes the cost effectiveness of a variety of technologies (Section 3.1). It also presents an analysis of the potential financial impacts associated with the use of each technology (Section 3.2).

3.1 Cost Effectiveness

Cost information on each of the control technologies described in Chapter 2 was obtained from the literature and pollution control technology vendors. This information was used to estimate the cost of installing and operating applicable mercury control techniques at model plants within the six source categories studied. Each of the model plants considered in the cost evaluation is defined in the box on the next two pages. The cost estimates for these model plants reflect generalized costs and are not intended to be site-specific. Plant-to-plant variations can result in higher or lower technology performance and associated costs. Detailed documentation and analysis of model plant cost is provided in Appendix B.

The cost estimates were combined with the mercury removal efficiencies presented in Chapter 2 to develop cost effectiveness values for various mercury controls applied to model plants for each industry. Table 3-1 summarizes the results of the cost effectiveness analysis. This table presents, for each of the six emission sources, the applicable mercury controls and the estimated cost effectiveness, both in dollars per pound of mercury (\$/lb Hg) removed and other measures (e.g., \$/ton of medical waste for MWIs and \$/ton of copper produced from primary copper smelters).

To determine how the cost-effectiveness of carbon injection might vary with different chlorine levels in the coal, a sensitivity analysis was performed which assumed various levels of HgCl_2 in the flue gas. These results are shown in Table 3-1 and also described in Section B.3.2. (Note that utility deregulation could impact the competitiveness of certain utilities and affect their ability to pass costs to consumers.)

3.2 Financial Analysis

This section presents an analysis of the ability of the six industries studied (MWCs, MWIs, utility boilers, chlor-alkali plants, primary copper smelters and primary lead smelters) to finance the mercury controls for which the capital and annual costs have been estimated on a model plant basis. The methodology for conducting the financial analysis is described first followed by the results. The financial impact of installing polishing wet scrubbers to control mercury in MWC and MWI emissions was not analyzed because the activated carbon technology is less expensive and as effective, so it is more likely to be the technology of choice.

3.2.1 Methodology

MWCs, MWIs and utility boilers provide a service to consumers whom, in general, do not have other options for acquiring the same service. Consequently, all or part of the cost of control usually can be passed on directly to the consumer in the form of a cost increase for the service. For these industries, financial impact was determined from the potential increase in product or

Definition of Model Plants Considered in the Cost Evaluation

Municipal Waste Combustors (MWCs)

Two model plants, both assumed to operate at full capacity 90 percent of the year.

- (1) A small mass burn/waterwall (MB/WW) MWC with two units and a total plant capacity of 180 Mg/day (200 tpd); equipped with dry sorbent injection (DSI) and an ESP.
- (2) A large MB/WW MWC with three units and a total plant capacity of 2,045 Mg/day (2,250 tpd) of MSW; equipped with a SD/FF pollution control system.

Medical Waste Incinerators (MWIs)

Two model plants, both assumed to have an inlet mercury level of 3,000 µg/dscm (1,310 g/million dscf); no mercury control across the DSI/FF system; and DSI/FF systems that operate at a temperature of 160°C across the FF.

- (1) A small batch-run MWI with design capacity of 90 kg/hr (200 lb/hr), but operating at an average throughput of 60 kg/hr (130 lb/hr) because of the high heat content of the waste; assumed to operate 5.5 hr/day, 5 days/week (which equals a capacity factor of 20 percent); equipped with a DSI/FF system.
- (2) A larger, commercial MWI; design capacity of 680 kg/hr (1,500 lb/hr), but operates at 455 kg/hr (1,000 lb/hr); operates 24 hr/day, 7 days/week; operates 90 percent of the year; equipped with a DSI/FF system.

Utility Boilers

Three model plants; all three assumed to operate with a capacity factor of 65 percent (5,694 hr/yr); chloride levels in fuel assumed to be sufficiently high that all mercury in the flue gas is in the form of HgCl_2 ; inlet mercury level to the control systems associated with each coal-fired model plant is 10 µg/dscm (4.4 g/million dscf) at 20°C.

- (1) A 975-MW coal-fired boiler firing low-sulfur coal; has a flue gas volume of 4,050,000 dscm/hr and is equipped with a cold-side ESP; temperature ahead of the ESP is 157°C and the temperature exiting the ESP is 150°C; no mercury control across the ESP is assumed; four mercury control variations considered:
 - (a) activated carbon (AC) injection
 - (b) spray cooler, AC injection, FF
 - (c) spray cooler, AC injection
 - (d) carbon filter bed
- (2) A 975-MW coal-fired boiler firing high-sulfur coal; equipped with a cold-side ESP (150°C) and a FGD system; FGD system assumed to be 50 percent efficient for mercury control and the controlled mercury level is 5 µg/dscm (2.2 g/million dscf); also equipped with a carbon filter bed for mercury control.

Definition of Model Plants Considered in the Cost Evaluation (continued)

Utility Boilers (continued)

- (3) Identical to model plant (1), except that it has a capacity of 100 MW; assumed to have a flue gas volume of 411,000 dscm/hr; the gas temperature ahead of the ESP is 146°C and the ESP outlet temperature is 137°C; no mercury control across the ESP is assumed, but two variations of the model plant are equipped with the following for mercury control:
- (a) AC injection
 - (b) spray cooler, AC injection, FF

Chlor-alkali Plants

One model plant that produces 273 Mg (300 tons) of chlorine per day (represents the mid-range size of chlor-alkali plants in operation); assumed to have individual flow rates from the hydrogen and end-box streams of 4,080 dscm/hr (144,000 dscf/hr) each at 21 percent O₂ (combined to equal 8,160 dscm/hr); baseline control systems for both streams consist of a heat exchanger to cool the gas followed by a knockout drum to separate the condensed mercury from the hydrogen and end-box streams; a mercury level of 1,000 g/day after this baseline control is assumed; mercury control options considered were conversion to the membrane cell process and control of the hydrogen and end-box streams through the use of brine scrubbing and treated AC adsorption; no additional controls examined for the cell room.

Primary Copper Smelters

One model plant consisting of a flash furnace and three Pierce-Smith converters; off-gases from the furnace are combined with the off-gas from the converters to form the feed to a single-stage acid plant; the feed to the acid plant has a maximum flow rate of 124,000 scm/hr (4,370,000 scf/hr); the plant has a capacity to process 910 Mg/day (1,000 tpd) of ore consisting of 27 percent copper and 32 percent sulfur, which corresponds to a production capacity of 78,000 Mg/yr of copper; a capacity factor of 90 percent is assumed, corresponding to 236 Mg/day (260 tpd) copper production capacity.

Primary Lead Smelters

One model plant consisting of a sintering machine, which feeds to a blast furnace and a dross furnace; because the sintering machine is an updraft machine (which does not employ recirculation of the weak gas), there are two offgas streams -- one strong SO₂ stream and one weak SO₂ stream; the strong stream is fed to a single-stage acid plant at a flow rate of 31,600 scm/hr (1,120,000 scf/hr); the weak stream, with a flow rate of 138,000 scm/hr (4,860,000 scf/hr) is sent to an FF for particulate control and emitted to the atmosphere; the offgases from the dross furnace are fed to an FF at a flow rate of 72,000 scm/hr (2,550,000 scf/hr) for PM control and emitted to the atmosphere; the lead production capacity is assumed to be 90,900 Mg/yr (100,000 tons/yr) from concentrate containing 55 percent lead and 16 percent sulfur; the plant is assumed to operate 90 percent of the year, corresponding to a daily capacity of 276 Mg/day (304 tpd).

Table 3-1
Cost Effectiveness of Control Technologies

Source	Mercury Control Technique	Cost Effectiveness		Cost Comments
		\$/lb Hg Removed	Other Measures	
MWCs	Material separation (batteries)	1,450	\$0.37/ton MSW	Costs are very community specific; Results shown are based on one community's program
	Production substitution (e.g., batteries, fluorescent lights)	--	--	The potential for product substitutions requires that the specific circumstances of each situation be examined; general cost estimates are not possible
	Activated carbon injection	211-870	\$0.7-3.5/ton MSW	Costs assume an 85% reduction; range of costs cover the two model plants
	Carbon filter beds	513-1,083	\$5.44-9.39/ton MSW	Range of costs cover the two model plants
	Polishing wet scrubber	1,600-3,320	\$5.3-13.5/ton MSW	Costs assume an 85 percent reduction; range of costs cover the two model plants
MWIs	Material separation (batteries)	--	--	Costs vary on a site-specific basis; no costs were available; cost effectiveness for a hospital program would be assumed to be better than for a community program
	Activated carbon injection	228-955	\$10.3-43/ton medical waste	Costs assume an 85 percent mercury reduction; acid gas controls are assumed to be in place; range of costs cover the two model plants
	Polishing wet scrubber	310	\$12/ton medical waste	Costs developed for the large model plant only (see box on pages 3-2 and 3-3); acid gas controls are assumed to be in place

Table 3-1
Cost Effectiveness of Control Technologies
(continued)

Source	Mercury Control Technique	Cost Effectiveness		Cost Comments
		\$/lb Hg Removed	Other Measures	
Utility Boilers	Carbon filter beds	33,000-38,000	2.7-3.1 mills/kWh	Range of costs covers two model plants (Models 1d and 2 in the box on pages 3-1 and 3-2)
	Activated carbon injection (coal fired) 100% of Hg as HgCl ₂	4,970-27,700	0.41-2.26	Range of costs covers model plants burning coal (Models 1a, 3a, 1b, 3b, and 1c)
	80% of Hg as HgCl ₂	6,190-34,600	0.41-2.26	
	50% of Hg as HgCl ₂	9,840-55,400	0.40-2.25	
	30% of Hg as HgCl ₂	16,300-92,000	0.40-2.25	
Chlor-Alkali Plants Using Mercury Cell Process	Process modification	4,590	\$39.6/ton chlorine produced	Cost effectiveness calculated using capital and electrical costs only
	Depleted brine scrubbing	1,040	\$6.7/ton chlorine produced	Cost data were scaled from 1972 dollars
	Treated activated carbon adsorption	769	\$5.1/ton chlorine produced	Cost data were scaled from 1972 dollars
Primary Copper Smelters	Selenium filters	497	\$11.3/ton copper produced	Cost data and information on mercury emissions at copper smelters are very limited
Primary Lead Smelters				
Sintering Stream	Selenium filters	541	\$2.7/ton lead produced	Cost data and information on mercury emissions at lead smelters are very limited
Furnace Stream	Selenium filters	520	\$5.8/ton lead produced	Cost data and information on mercury emissions at lead smelters are very limited

service cost to the consumer arising from the cost of the mercury controls. This ratio, calculated as the ratio of annual control costs to revenues, reflects the potential percent cost increase necessary to recover the control costs.

For chlor-alkali plants, primary copper smelters and primary lead smelters, individual companies have little control over the price they are able to receive for their respective products. In each industry, individual companies are competing against other domestic producers and against foreign producers of the same products. Therefore, prices are often determined by international markets. In addition, primary copper smelters and primary lead smelters must also compete against domestic and foreign secondary producers that recover copper and lead from scrap materials, rather than from ores mined from the earth.

Because individual companies in these industries have little control over the price they receive for their products, mercury control costs cannot be passed on to the consumer. Therefore, the funds to purchase mercury controls must be financed with existing profits. The financial impact from installing and operating the mercury controls for these industries was determined from the ratio of total annual control costs to profits and the ratio of annual capital costs to annual expenditures. The annual capital costs of control equipment are the initial capital costs of equipment distributed over the lifetime of the equipment. These measures of financial impact represent the ability of the industries to absorb the annual control costs and to acquire the capital needed to purchase the controls while still remaining competitive.

Information on profits and expenditures was not available for the primary lead smelter industry because there are only two companies in the U.S. industry and this information is considered confidential. Consequently, potential increases in lead prices necessary to recover control costs were determined for this industry using the same methodology as was used for MWCs, MWIs, and utilities.

3.2.2 Results

The financial impact of installing mercury emission controls for the above-referenced industries is summarized in this section. Activated carbon injection costs were calculated for MWCs, MWIs and utility boilers. Costs of carbon filter beds were calculated for MWCs and utility boilers. The costs of membrane cell process conversion, depleted brine scrubbing and treated activated carbon adsorption were calculated for chlor-alkali plants. Selenium filter costs were calculated for primary copper smelters and primary lead smelters.

Municipal Waste Combustors. For the small and large MWC model plants, which are described in Appendix B, total revenues are estimated to be \$3.4 million and \$38.0 million, respectively (see Table 3-2). Revenues for MWCs are typically derived from energy production, tipping fees (the fee received by the MWC for each ton of MSW received), and sale of materials collected on site for recycling. Because information on revenues from tipping fees and recycling sales are not available, total revenues for the MWCs were estimated from electricity sales. An industry source estimates that electricity sales typically account for 35 to 50 percent of a facility's revenues (Kiser and Burton, 1992). Therefore, total revenues were estimated to be equal to twice the value of electricity sales. Electricity sales were estimated based on the average net kilowatt-hours generated per ton of MSW combusted (533 kWh/Mg [485 kWh/ton]), the annual amount of MSW combusted for the model plants (60,000 Mg/yr [66,000 tons/yr] for the small MWC and 672,000 Mg/yr [739,000 tons/yr] for the large MWC), and the average price paid for electricity (\$0.053/kWh) (Berenyi and Gould, 1993).

Table 3-2
Estimated Annual Revenues and Potential Cost Increases^a for MWCs, MWIs and Lead Smelters

Parameter	Municipal Waste Combustors	Medical Waste Incinerators	Primary Lead Smelters
Total Annual Revenues (millions of dollars)	Small ^b : \$3.4 Large ^c : \$38	Hospital: \$32.5 Commercial: \$2.0	\$66.8
Potential Cost Increases			
Activated Carbon Injection	Small ^b : 6.9% Large ^c : 1.3%	Hospital: 0.02% Commercial: 2.0%	N/A
Carbon Filter Beds	Small ^b : 83% Large ^c : 35%	-- ^e -- ^e	N/A
Selenium Filter	N/A ^d	N/A	1.3%

^a Potential cost increase = total annual operating cost divided by total annual revenue. Represents the potential cost increase in service or product to cover the cost of controls.

^b Capacity = 180 Mg/day.

^c Capacity = 2,045 Mg/day.

^d N/A = control technology is not applicable.

^e The costs of the carbon filter beds were not estimated for this model plant.

The MWC mercury control costs for activated carbon injection and carbon filter beds were estimated in Appendix B. The estimated annual control costs for activated carbon injection for the small MWC are \$232,000, indicating a potential 6.9 percent cost increase. For the large MWC, the estimated annual control costs for activated carbon injection are \$520,000, indicating a potential 1.3 percent cost increase. The estimated annual control cost for carbon filter beds are \$2.81 million or an 83 percent potential increase for the small MWC, and \$13.2 million or a 35 percent potential cost increase for the large MWC. Reliable cost data on battery separation programs as mercury control options were not available, so cost increases arising from these programs could not be estimated.

Medical Waste Incinerators. Control costs for activated carbon injection were estimated in Appendix B for a small batch-operated MWI and a large continuously operated MWI. The small MWI, which incinerates about 100 Mg/yr (90 tons/yr) of medical waste, is assumed to be operated by a hospital. The large MWI, which incinerates about 3,600 Mg/yr (3,300 tons/yr) of medical waste, is assumed to be part of a commercial MWI operation.

As summarized in Table 3-2, average hospital revenues are estimated to be about \$32.5 million annually and about 46 percent of all hospitals operate an MWI (U.S. EPA, 1991a; U.S. EPA, 1994d). The total annual control costs for activated carbon injection on the small MWI are estimated to be \$5,040, indicating a potential 0.02 percent increase in the cost of hospital services.

Average annual revenues for commercial MWIs are estimated to be about \$2 million, based on a typical commercial incineration fee of \$660/Mg (\$600/ton) and a throughput of 3,000 Mg/yr (3,300 tons/yr) of medical waste (U.S. EPA, 1991a). The estimated total annual control costs for activated carbon injection on the large commercial MWI are about \$40,600, indicating a potential 2.0 percent increase in the cost of services.

Primary Lead Smelters. Control costs for selenium filters were estimated in Appendix B for a model primary lead smelter. Because there are only two U.S. lead smelters, information on profits and expenditures is considered confidential and was not available. Therefore, the potential lead price increase necessary to recover control costs was calculated for this industry as a measure of financial impact. The actual financial impact of this control, however, should be lower than this measure would indicate because this industry operates at a very narrow profit margin. The average revenue for primary lead smelters was estimated from the model plant's annual lead production and the average North American producer price for lead from 1992 and 1993 of \$0.73/kg (\$0.33/lb). Average smelter revenue is estimated to be \$66.8 million.

The estimated annual costs for a selenium filter controlling the sintering flue gas stream upstream of the acid plant are \$266,000. The estimated annual costs of two selenium filters controlling the blast furnace and dross furnace flue gas streams after the FF on the model plant are \$584,000. The combined total annual costs of control on the model plant are estimated to be about \$850,000, indicating a 1.3 percent potential increase in lead prices necessary to recover control costs (see Table 3-2).

Chlor-Alkali Plants. Control costs were estimated for three different control scenarios: (1) conversion of the model plant to a membrane cell process; (2) control of emissions from the existing mercury cell plant with depleted brine scrubbing; and (3) control of emissions with treated activated carbon adsorption. Results of the analysis are summarized in Table 3-3.

Table 3-3
Estimated Annual Profits, Expenditures,
Revenues and Financial Impacts^a

Parameter	Chlor-Alkali Plants	Primary Copper Smelters
Total Annual Profits (millions of dollars)	\$12.9	\$22.0
Total Annual Expenditures (millions of dollars)	\$26.7	\$204
Financial Impact		
Membrane Cell Process	12% of Expenditures	N/A ^b
Depleted Brine Scrubbing	5.1% of Profits 0.7% of Expenditures	N/A
Treated Activated Carbon Adsorption	3.9% of Profits 0.5% of Expenditures	N/A
Selenium Filters	N/A	4.4% of Profits 0.2% of Expenditures

^a Financial impact = total annual control costs divided by profits and annual capital costs divided by total expenditures.

^b N/A = control technology is not applicable.

Note: The percentage of annual profits represents the amount of profit that would be needed to absorb the control costs. The percentage of annual expenditures provides a measure of the industry's ability to acquire the capital needed for the controls while still remaining competitive.

Information on profits and expenditures for the chlor-alkali industry were obtained from the 1987 Census of Manufacturers, which reported 27 companies operating 45 plants. Financial data specific to the 14 chlor-alkali plants that use the mercury cell process could not be obtained. Financial data were subsequently corrected to 1990 dollars using the Consumer Price Index. Total annual profits were \$581.2 million, or an estimated average of \$12.9 million per plant. Total annual expenditures for the industry were \$1.20 billion, or an estimated average of \$26.7 million per plant (U.S. Department of Commerce, 1990a).

The membrane cell process has lower electricity costs than the mercury cell process and, consequently, electricity savings were considered in estimating plant conversion costs. The estimated net annual capital cost of converting the model plant to a membrane cell process, after deducting estimated electricity savings, is about \$3.3 million, or about 12 percent of total annual expenditures. Comprehensive data on other annual operating expenses for the membrane cell process were not available, so the total annual operating expenses for the membrane cell process could not be determined. Consequently, the financial impact of this control, measured by annual costs as a percentage of profits, could not be determined.

The estimated total annual costs for depleted brine scrubbing on the combined hydrogen and end-box stream, including the costs for secondary cooling and mist elimination, are estimated to be

\$662,000, or 5.1 percent of annual profits. The estimated annual capital recovery cost for this control is \$177,000, or 0.7 percent of annual expenditures. These numbers provide a measure of the ability of the industry to absorb the control costs, and the ability of the industry to acquire the capital needed for the controls while still remaining competitive.

The total annual costs for treated activated carbon adsorption on the combined hydrogen and end-box stream, including the costs for secondary cooling and mist elimination, are estimated to be \$500,000, or 3.9 percent of profits. The estimated annual capital recovery cost for this control is \$134,000, or 0.5 percent of annual expenditures.

Primary Copper Smelters. Control costs for selenium filters were estimated for a model primary copper smelter (see Table 3-3). Information on profits and expenditures was obtained from the 1987 Census of Manufactures and subsequently corrected to 1990 dollars using the Consumer Price Index (U.S. Department of Commerce, 1990b). In 1987, there were 8 companies operating 13 plants. As of 1994, however, there are eight plants operated by six companies, which indicates a change in industry structure. The 1987 financial data were the most recent available for this analysis. Total annual profits in 1987 were \$285.8 million, or an average of \$22.0 million per smelter. Total primary copper smelter annual expenditures in 1987 were \$2.66 billion, or an average of \$204 million per smelter (U.S. Department of Commerce, 1990b).

Estimated annual costs for selenium filters for the model copper smelter are \$961,000, or 4.4 percent of profits. The estimated annual capital recovery cost for the selenium filters is \$355,000, or 0.2 percent of expenditures. These numbers provide a measure of the ability of the industry to absorb the control costs, and the ability of the industry to acquire the capital needed for the controls while still remaining competitive.

Utility Boilers. Control costs were estimated for activated carbon injection (with and without spray coolers and FFs) and carbon filter beds for three models of coal-fired utility boilers. Revenues were also estimated for each of the models and a potential cost increase for electricity was determined for each of the model boiler/control technology combinations.

The models included two large boilers (975-MW capacity). The first model boiler fired low-sulfur coal and was controlled by an ESP. The second model boiler fired high-sulfur coal and was controlled by an ESP and FGD. The third model was a small utility boiler (100-MW capacity) firing low-sulfur coal and controlled by an ESP. The different utility boiler models and the mercury controls applied to each are summarized in Table 3-4, along with estimated annual revenues for each model boiler, the estimated cost of control and the potential cost increase for each model/control technology combination.

Annual revenues were estimated for the model boilers based on revenues from electricity sales and electric generating capacity for 20 utility companies (Value Line Publishing, 1994a and 1994b). The median annual revenues per megawatt of generating capacity from these 20 companies was \$453,000; the range was \$268,000 to \$723,000/MW of generating capacity. Revenues were estimated for the model boilers from the revenue factor of \$453,000/MW of capacity and by applying a load factor of 65 percent (i.e., only 65 percent of generating capacity is utilized on an annual basis).

The estimated revenues for the 975-MW boiler are \$287 million; estimated revenues for the 100-MW boiler are \$29 million. Potential electricity cost increases for each model plant and control scenario are presented in the last column of Table 3-4.

Table 3-4
Model Utility Boilers: Estimated Annual Revenues and Potential Cost Increases^a

Model ^b	Size (MW)	Fuel	Pre-Existing Controls	Estimated Revenues (\$ Million)	Mercury Control	Annual Control Costs (\$ Million)	Potential Cost Increase %
1a	975	Low-sulfur coal	ESP	287	AC injection ^c	10.1	3.5
1b	975	Low-sulfur coal	ESP	287	Spray cooler, AC injection, FF	7.9	2.8
1c	975	Low-sulfur coal	ESP	287	Spray cooler, AC injection	2.4	0.8
1d	975	Low-sulfur coal	ESP	287	Carbon filter bed	59.9	21
2	975	High-sulfur coal	ESP/FGD	287	Carbon filter bed	60.7	21
3a	100	Low-sulfur coal	ESP	29	AC injection	0.64	2.2
3b	100	Low-sulfur coal	ESP	29	Spray cooler, AC injection, FF	1.3	4.4

a Potential cost increase = total annual operating costs divided by total annual revenues, which represents the potential increase in electricity costs to cover the costs of control.

b See model plant description in box on pages 3-2 and 3-3 as well as Appendix B.

c AC = activated carbon.

4. MERCURY CONTROL STRATEGIES

There are a number of laws, regulations and other initiatives designed to control mercury uses, releases to the environment and exposures among people and ecological receptors. Many of these programs have a direct bearing on the extent and effects of mercury emissions to the atmosphere.

This chapter summarizes existing and emerging authorities and activities that influence mercury emissions. This review is not intended to be comprehensive or complete, but rather focuses on major federal initiatives and relevant activities in selected states (principally states in the Great Lakes region). The chapter starts with an overview of ongoing federal and state activities in Section 4.1. Management alternatives and statutory authorities for controlling mercury emissions under the Clean Air Act are then presented in Section 4.2.

4.1 Ongoing Federal and State Activities

Mercury regulation and control spans multiple federal and state statutes. Ongoing programs to control mercury uses, releases and exposures under a spectrum of environmental laws are summarized below, first at the federal level and then at the state level.

4.1.1 Federal Activities

Several federal agencies have authority and responsibility for controlling mercury uses, releases and exposures. For example, U.S. EPA has addressed for many years and continues to address the risks posed by mercury through regulations designed to limit releases to air, water and land. The Food and Drug Administration (FDA) regulates mercury in cosmetics, food and dental products. The Occupational Safety and Health Administration (OSHA) regulates mercury air exposures in the workplace.

Table 4-1 summarizes major federal activities to control mercury uses, releases and exposures. In general terms, these activities can be grouped into environmental media standards, use- or release-related regulations and other standards and programs.

- Environmental media standards are numeric criteria that specify a maximum acceptable mercury concentration for different media, based on scientific or risk-based criteria. These standards have an indirect effect on individual sources. For instance, mercury standards shown in Table 4-1 for different media influence how much mercury different sources can release into the environment, both individually and cumulatively.
- Use- or release-related regulations have a direct effect on sources that use mercury or release mercury into the environment. These regulations specify, for individual sources or types of waste materials, the conditions associated with mercury disposal and release. Additionally, these regulations specify the conditions associated with using mercury in the manufacture of different products (batteries, paints, pesticides, etc.).
- Other standards and programs, for the purpose of this review, include regulations designed to limit mercury exposures in the workplace and during transportation, as well as special government initiatives to address mercury problems in an integrated manner (i.e., the Virtual Elimination Project and U.S. EPA's Mercury Task Force).

**Table 4-1
Federal Mercury Controls^a**

SPECIFIC SOURCES/FOCUS	CONTROL/ACTION
ENVIRONMENTAL MEDIA	
Drinking Water	<ul style="list-style-type: none"> Maximum contaminant level (MCL) = 0.002 mg/L (40 CFR 141.62, 21 CFR 103.35).
Surface Water	<ul style="list-style-type: none"> Ambient Water Quality Criteria; water and organisms = 0.012 µg/L (40 CFR 401, 403, Appendix B). Water Quality Guidance for the Great Lakes System: aquatic life = 1.44 µg/L (acute) and 0.77 µg/L (chronic); human health = 0.0018 µg/L; wildlife = 0.0013 µg/L (40 CFR 132).
Air	<ul style="list-style-type: none"> No ambient air standard.
Soil	<ul style="list-style-type: none"> No soil standard.
ENVIRONMENTAL SOURCES	
Air Point Sources	<ul style="list-style-type: none"> Emissions from mercury ore processing facilities and mercury cell chlor-alkali plants are limited to a maximum of 2,300 g/24 hours (40 CFR 61.01). Emissions from sludge incineration plants, sludge drying plants, or a combination of these that process wastewater treatment plant sludges are limited to a maximum of 3,200 g/24 hours (40 CFR 61.52). Industrial sources emitting mercury and mercury compounds may be subject to Maximum Achievable Control Technology (MACT) standards for major stationary sources and Generally Available Control Technology (GACT) standards for area sources.^b Emission guidelines on mercury emissions from municipal waste combustors under §§ 111 and 129 of the 1990 Clean Air Act Amendments. Regulations on mercury emissions from medical waste incinerators under §§ 111 and 129 of the 1990 Clean Air Act Amendments were proposed in 1995 (60 FR 10654, February 27, 1995). Airborne emissions of mercury and other substances from the burning of hazardous waste in boilers and industrial furnaces, including cement kilns, are regulated under the Resource Conservation and Recovery Act (40 CFR 266).
Water Point Sources	<ul style="list-style-type: none"> Effluents from industrial facilities and municipal wastewater treatment facilities are regulated through industry-specific pretreatment standards and effluent guidelines for existing and new sources of pollution and are based on the limits of the available control technology (40 CFR 401, 403, Appendix B). Groundwater at hazardous waste treatment, storage and disposal facilities must be monitored for the presence of mercury (40 CFR 302.4, 264.94).
Sewage Sludge	<ul style="list-style-type: none"> Permissible levels of mercury in municipal wastewater treatment sludges: 17 mg/kg dry wt. and cumulative load of 17 kg/hectare for agricultural land; 17 mg/kg dry wt. and annual load of 0.85 kg/hectare for home garden or lawn; 57 mg/kg dry wt. for other land applications; and 100 kg/hectare for surface disposal (CWA).

**Table 4-1
Federal Mercury Controls (continued)**

SPECIFIC SOURCES/FOCUS	CONTROL/ACTION
Mercury-Containing Wastes	<ul style="list-style-type: none"> Any solid waste (including soil that is being disposed) is considered a hazardous substance and prohibited from disposal in RCRA Subtitle D (non-hazardous) landfills if its leachate contains 0.2 mg/L mercury or greater (40 CFR 261.24); land disposal in RCRA Subtitle C (hazardous) landfills is allowed only after prescribed treatment to reduce mercury in extract to 0.2 mg/L (40 CFR 268). Certain wastes are listed as hazardous due, at least in part, to the presence of mercury (e.g., K071 = brine purification muds from the mercury cell process in chlorine production, and K106 = wastewater treatment sludge from the mercury cell process in chlorine production). The amount of mercury in a number of hazardous wastewaters (e.g., F039, K071, K106, P065, P092) must be treated down to specified levels to meet the land disposal restrictions.
Any Environmental Release	<ul style="list-style-type: none"> Any release of 1 pound or more of mercury into the environment in a 24-hour period (the reportable quantity) must be reported immediately to the National Response Center if the release is not federally permitted (40 CFR 302). Certain facilities that release more than a reportable quantity of mercury must immediately report the release to state and local entities. Any release or transfer of mercury by facilities that exceed use or manufacturing thresholds is reportable under the Toxic Release Inventory.
Foodstuffs or Feed	<ul style="list-style-type: none"> Action level for methylmercury in fish, shellfish and other aquatic animals = 1 ppm (FDA CPG 7180.07). The import of foods containing the residue of mercury-containing pesticides that are not registered for use in the U.S. is prohibited.
PRODUCTS	
Batteries	<ul style="list-style-type: none"> By early 1991, all U.S. manufacturers converted production so that the mercury content of batteries, except in button and coin cells, did not exceed 0.025% by weight. Federal legislation pending concerning the manufacture of only "non-mercury" formula batteries of all types by 1-1-97. Federal ban on mercury button cell batteries pending as of 1-1-95. Federal legislation permitting only the manufacture of "no mercury" formula zinc carbon batteries pending as of 1-1-95.
Paints and Pigments	<ul style="list-style-type: none"> All uses of mercury in paints have been discontinued.
Dental Uses	<ul style="list-style-type: none"> Dental mercury is classified as a Class I medical device, with extensive safety regulations on its use. Dental amalgam alloy is classified as a Class II device, subject to additional special controls. U.S. Public Health Service has recently studied risks from mercury amalgams and recommended tighter controls on dental uses of mercury and further research to reach more definitive conclusions on risk.

**Table 4-1
Federal Mercury Controls (continued)**

SPECIFIC SOURCES/FOCUS	CONTROL/ACTION
Lighting	<ul style="list-style-type: none"> Because many fluorescent lamps are classified as RCRA hazardous wastes under current test procedures, U.S. EPA is evaluating options for lamp disposal. Major options are 1) conditional exclusion of lamps from hazardous waste management requirements, and 2) handling lamps in a special collection system for other "low grade" and small quantity hazardous wastes (such as batteries and household pesticides).
Pesticides	<ul style="list-style-type: none"> No current production of mercury-containing pesticides; all former registrations have been cancelled or requests for voluntary cancellation have been received.
Special Paper Coatings	<ul style="list-style-type: none"> The only two companies that manufacture these products have announced that plans are being developed to phase out the use of mercury in the coatings. It is predicted that mercury will be eliminated entirely from this application by 1995.
Pharmaceuticals	<ul style="list-style-type: none"> Removal or restriction of mercury in "over-the-counter" (OTC) drugs such as anorectal products and topical antiseptics. Request for additional data on other OTC mercury antimicrobials.
Cosmetics	<ul style="list-style-type: none"> The use of mercury as a preservative or antimicrobial is limited to eye-area cosmetics or ointments in concentrations less than 60 ppm (21 CFR 700.13).
OTHER STANDARDS AND PROGRAMS	
Occupational Standards	<ul style="list-style-type: none"> OSHA Standards: ceiling limit of 0.1 mg/m³ for inorganic and elemental mercury, 0.01 mg/m³ as an 8-hr time weighted average for alkylmercury compounds, and a ceiling limit of 0.04 mg/m³ for alkylmercury compounds.^c All forms of mercury are assigned a skin notation, indicating that the substance is absorbed through the skin and therefore skin contact should be avoided. As an OSHA hazardous chemical, the presence of mercury at a facility requires submittal of a Material Safety Data Sheet.
Transportation Standards	<ul style="list-style-type: none"> Designated as hazardous substances by the Department of Transportation and subject to requirements for packaging, shipping and transportation (40 CFR 172.101).
Virtual Elimination Project	<ul style="list-style-type: none"> U.S. EPA and other mercury stakeholders are looking holistically at mercury sources and policies to identify and promote "cleaner, cheaper, smarter" ways of reducing mercury levels in the Great Lakes region. Efforts are designed to (1) reduce uses at the source through pollution prevention measures, (2) reduce releases through treatment or other management techniques and (3) clean up sites of past contamination.

^a Adapted from OECD, 1995.

^b Under the 1990 Clean Air Act Amendments, mercury and mercury compounds are regulated as hazardous air pollutants (HAPs). MACT standards will apply to major stationary sources emitting more than 10 ton/yr of mercury or any other one HAP, or 25 tons/yr of any combination of HAPs.

^c More stringent standards were recently overturned in U.S. court.

Environmental Media Standards. Different U.S. EPA program offices have established acceptable mercury concentration limits for environmental media. U.S. EPA's Office of Ground Water and Drinking Water, as part of the National Drinking Water Standards required under the Safe Drinking Water Act of 1986, has established a Maximum Contaminant Level (MCL) for inorganic mercury in drinking water. U.S. EPA's Office of Water also has set Ambient Water Quality Criteria that establish acceptable concentrations of mercury for the protection of human health and aquatic species. At present, there are no federal standards establishing mercury concentration limits for ambient air or soil, although U.S. EPA is in the process of developing clean-up levels for soil.

U.S. EPA recently published Final Water Quality Guidance for the Great Lakes System (60 FR 15366, March 23, 1995). The Guidance established water quality criteria for 29 pollutants -- including mercury -- to protect aquatic life, wildlife and human health. The Guidance also consists of detailed methodologies to develop criteria for additional pollutants; implementation procedures to develop more consistent, enforceable water quality-based effluent limits in discharge permits, as well as total maximum daily loads of pollutants that can be allowed to reach the Lakes and their tributaries from all sources; and antidegradation policies and procedures. Great Lakes States and tribes will use the water quality criteria, methodologies, policies and procedures in the Guidance to establish consistent, enforceable, long-term protection for fish and shellfish in the Great Lakes and their tributaries, as well as for the people and wildlife who consume them.

Environmental Source Controls. As shown in Table 4-1, regulations that control mercury releases into the environment include limits on air and water point sources, solid waste disposal restrictions and requirements for reporting and public disclosure of releases. These regulations have been established under the authority of various environmental statutes.

Under the 1990 Clean Air Act Amendments, mercury and mercury compounds are regulated as Hazardous Air Pollutants (HAPs). U.S. EPA has established National Emission Standards for three major point source categories of mercury emissions: ore processing facilities, mercury cell chlor-alkali plants and sewage sludge driers. Industrial sources emitting mercury and mercury compounds may be subject to Maximum Achievable Control Technology (MACT) standards for major stationary sources (emissions exceeding 10 ton/yr) and Generally Available Control Technology (GACT) standards for area sources. U.S. EPA also recently finalized regulations governing mercury emissions from municipal waste combustors under §§ 111 and 129 of the 1990 Clean Air Act Amendments and is in the process of developing similar rules for medical waste incinerators.

Under the Clean Water Act, mercury is listed as a toxic pollutant and mercury discharges from certain categories of industries are subject to technology-based effluent limits. The National Pollutant Discharge Elimination System (NPDES) implemented under the Clean Water Act regulates direct discharges to surface water bodies. Facilities are assigned a specific mercury discharge limit and/or are required to monitor their discharge for mercury levels. Discharge Monitoring Reports submitted by facilities serve as the basis for determining compliance with NPDES requirements. Indirect dischargers (facilities that discharge to publicly owned treatment works) are regulated by industry-specific pretreatment standards and effluent guidelines that are based on limits of the available control technology.

Under the Resource Conservation and Recovery Act (RCRA), U.S. EPA has established specific classification and disposal requirements for wastes that contain mercury. RCRA regulations are waste-specific, not source-specific, and thus may apply to any facility that generates mercury-containing wastes. Under current RCRA regulations, certain mercury-containing wastes are classified either as a "characteristic" or a "listed" waste. Wastes are considered characteristic hazardous wastes if

they exhibit any of four specified characteristics: ignitability, corrosivity, reactivity, or toxicity. A specific "D" waste code identifies the contaminant(s) for which a waste exhibits the toxicity characteristic. Waste code D009 identifies wastes that exhibit the toxicity characteristic for mercury (i.e., leaches mercury at or above 0.2 mg/l when analyzed using the Toxicity Characteristic Leaching Procedure). Listed wastes are specifically identified wastestreams or discarded products under RCRA, with each listed waste being assigned a different waste code. Six different mercury-containing wastes or discarded chemical products have been listed as hazardous wastes.¹ RCRA regulations prescribe specific treatment, storage and disposal requirements for individual waste codes. All mercury-bearing wastes are subject to land disposal restrictions, meaning that they have to be treated in accordance with RCRA regulations before they are allowed to be land disposed (such as in a landfill). Additionally, RCRA regulations limit the airborne emission of mercury and other substances from boilers and industrial furnaces, including cement kilns, that burn hazardous wastes.

The Superfund Amendments and Reauthorization Act of 1986 (SARA), which amended the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), provides broad Federal authority for responding to releases of listed hazardous substances, including mercury. The Emergency Planning and Community Right-to-Know Act also establishes emergency release, inventory and release reporting requirements. All facilities in the manufacturing sector (SIC 20-39) that meet the threshold reporting requirements are required to report releases to air, water and land for all listed chemicals, including mercury, in the Toxics Release Inventory (TRI).

Under section 302 of SARA, any facility that produces, uses, or stores "extremely hazardous" substances must notify the State Emergency Response Commission. Mercuric acetate, mercuric chloride and mercuric oxide are considered extremely hazardous substances (40 CFR 355). Additionally, any facility that releases a reportable quantity (RQ) or more of a hazardous substance into the environment must notify the National Response Center, State Emergency Response Commission and Local Emergency Planning Committee. The RQ for mercury is one pound. Finally, facilities that keep hazardous substances on-site in quantities greater than threshold levels must submit a chemical inventory to the State Emergency Response Commission, the Local Emergency Planning Committee and the local fire department. The threshold for mercury (a hazardous substance) is 10,000 pounds, and the threshold for mercury compounds listed as extremely hazardous substances is 500 pounds.

Product Controls. Mercury is a component of certain pesticides, special paper coatings, pharmaceuticals, cosmetics and a variety of household products, including batteries, paints, fluorescent light bulbs, electrical switches and thermometers. Such mercury-containing products are presently regulated under two federal statutes, the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) and the Federal Food, Drug, and Cosmetic Act (FFDCA). FIFRA regulates the sale and use of pesticides, including the registration of chemicals that meet health and safety tests. Until recently, several mercury compounds were registered as pesticides, bactericides and fungicides. By 1993, however, all registrations for mercury compounds in paints and pesticides had been either cancelled by U.S. EPA or voluntarily withdrawn by the manufacturer. The Food and Drug Administration regulates mercury in food, drugs, cosmetics and dental amalgam under FFDCA. Particular controls on these

¹ These wastes are: F039 (leachate resulting from the disposal of more than one restricted waste), K071 (brine purification muds from the mercury cell process in chlorine production where separately prepurified brine is not used), K106 (wastewater treatment sludge from the mercury cell process in chlorine production), P065 (mercury fulminate -- a mercury compound used in explosives -- is listed as an acute hazardous waste when discarded), P092 (phenylmercuric acetate -- a mercury compound used in paints -- is listed as an acute hazardous waste when discarded), and U151 (the chemical mercury is considered hazardous when discarded).

products include restrictions on the mercury content, specified conditions for use and labeling requirements. With regard to mercury-containing batteries, Federal legislation is pending that would eliminate the use of mercury in all types of batteries used in both industrial and household activities by January 1, 1997 (Ross & Associates, 1994). The battery industry already has converted production processes so that the mercury content of batteries, except in button and coin cells, does not exceed 0.025 percent by weight (National Electrical Manufacturers Association, 1995).

Other Standards and Programs. The Occupational Safety and Health Administration (OSHA) regulates exposures to mercury in the workplace by establishing Permissible Exposure Limits (PELs). The PELs for different mercury compounds are presented in Table 4-1. The National Institute for Occupational Safety and Health (NIOSH) defines an additional limit called the Immediately Dangerous to Life and Health (IDLH) level, defined as the maximum environmental concentration of a contaminant from which one could escape within 30 minutes without any escape-impairing symptoms or irreversible health effects. The IDLH is 28 mg/m³ for elemental mercury vapor and 10 mg/m³ for organo-mercury compounds.

Under the Hazardous Materials Transportation Act (HMTA), the Department of Transportation limits the potential for mercury releases during routine transportation and from transportation accidents. Mercury and mercury compounds are designated as hazardous substances and are subject to requirements for packaging, shipping and transportation under HMTA.

U.S. EPA and other organizations are actively developing strategies to achieve the Great Lakes Water Quality Agreement's goal that persistent toxic substances should be "virtually eliminated" from the Great Lakes. Because toxic substances enter the Great Lakes from ongoing economic activities, as well as from sites contaminated by past activities, eliminating toxic substances from the Great Lakes requires a three-pronged approach that:

- Reduces the use of toxic substances at the source, through pollution prevention efforts;
- Reduces toxic substance discharges, emissions and other ongoing releases through treatment or other management techniques; and
- Cleans up sites of past contamination, such as contaminated sediments or areas of concern, through remediation efforts.

U.S. EPA's Virtual Elimination Project has been studying mercury and polychlorinated biphenyls (PCBs). A systematic review of the sources has taken place under this effort. Currently, an analysis of options to further reduce and eliminate the use of these chemicals is being undertaken.

A central theme underlying the virtual elimination project is that opportunities may exist to alter the decisionmaking environment in which individuals and firms choose to use and release toxic chemicals in their ongoing activities. An individual or firm is likely to minimize its use and/or release of toxic substances when it understands:

- The full cost of environmental regulation pertaining to a particular chemical or waste (the cost of current practices), including costs such as liability, compliance, inputs, waste disposal and adverse public perception;
- Whether or not an alternative input, process, or product exists that serves the same purpose, but is less harmful to the environment (the availability of alternatives); and

- How much of the current cost associated with using or releasing a particular chemical can be avoided by adopting less costly alternative practices that are less harmful to the environment (the cost of alternatives).

In choosing to modify current activities, an individual or firm will evaluate the costs associated with its current practices against the cost of available alternatives.

Government actions can enhance pollution prevention opportunities and stimulate toxic substance reduction by (1) re-aligning the costs and/or regulatory structure to provide greater incentives and flexibility for individuals and/or companies to reduce their use and release of toxic substances and/or (2) reducing the cost of adopting alternatives that are less harmful to the environment.

The Virtual Elimination Project focuses on government actions -- or "signals" -- such as regulatory or voluntary programs that influence the economic and legal costs and benefits associated with using a particular chemical. These signals, which translate into costs for an affected entity, can motivate individuals and firms to choose pollution prevention based on their own economic interests.

4.1.2 State Activities

Mercury control regulations are increasing rapidly at the state level. Many states are developing new regulations that will control the release of mercury from different environmental sources. Minnesota, for example, is currently drafting management standards for facilities that recycle mercury-containing wastes, and has proposed new combustion rules. States also have developed new monitoring and reporting requirements on mercury release from air and water point sources. Table 4-2 summarizes examples of mercury control measures adopted by various states for different environmental sources.

Regulations on mercury-containing products differ by state. In addition to health-based concerns, states are focusing on waste disposal problems associated with mercury-containing products. Many states have regulations that ban or limit the amount of mercury in products, establish recycling requirements and impose disposal restrictions on products containing mercury. For example, certain types of batteries containing mercury are banned in a number of states, and 12 states have enacted laws that limit the amount of mercury in alkaline batteries to 0.025 percent by weight (Ross & Associates, 1994). Minnesota also requires removal of mercury from electric lamps and thermostats before they are disposed. Illinois restricts disposal of white goods containing mercury and limits the amount of mercury in packaging. Many states also have mercury waste collection programs in place to control the release of mercury into the environment. Table 4-2 summarizes the regulations and programs adopted by selected states for various products, including legislation and detailed regulations that apply to mercury-containing batteries.

In addition, states follow the reporting requirements and other standards developed and implemented by U.S. EPA and other federal agencies. Some states have regulations in addition to the federal regulations. For example, Michigan's Water Pollution Control Act requires businesses to report the amount of mercury used and released. This information is used to assist in permit development and compliance monitoring in the state water program (Ross and Associates, 1994).

Table 4-2
Examples of State Mercury Controls^a

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
ENVIRONMENTAL SOURCES		
Air Point Sources	Florida	<ul style="list-style-type: none"> Mercury emission standard for municipal solid waste incinerators (65 micrograms/m³).
	Minnesota	<ul style="list-style-type: none"> Proposed waste combustion rules include emission limits. New incinerator permits with mercury limits will require air monitoring systems and periodic stack testing.
	New Jersey	<ul style="list-style-type: none"> Mercury emission standard for municipal solid waste incinerators (65 micrograms/m³), with further reductions to be phased in.
	Ohio	<ul style="list-style-type: none"> Considering installing mercury-emission control equipment.
	Wisconsin	<ul style="list-style-type: none"> Medical waste incinerators with capacity greater than 5 tons per day must be tested for mercury during the first 90 day period of operation and once the following year.
Water Point Sources	Michigan	<ul style="list-style-type: none"> Businesses must report use and discharge information for mercury under the Water Pollution Control Act.
Hg-Containing Wastes	Minnesota	<ul style="list-style-type: none"> Management standards for facilities recycling mercury-containing hazardous wastes (currently being drafted). Mercury must be removed from products before disposal.
PRODUCTS		
White Goods	Illinois	<ul style="list-style-type: none"> Mercury components must be removed from discarded white goods before disposal.
	Minnesota	<ul style="list-style-type: none"> Products containing mercury must be labeled, and the labels must include any disposal restrictions. Mercury components must be removed prior to disposal. The distribution of mercury-containing fever thermometers is restricted. Manufacturers must provide information and incentives regarding recycling or proper management. HVAC dealers are required to properly manage or recycle used mercury thermostats. Mercury in repaired or replaced items must be recycled.
Batteries	Arkansas	<ul style="list-style-type: none"> Ban on the manufacture of alkaline batteries containing any mercury effective 1-1-96. Ban on the manufacture of mercury-containing zinc carbon and mercury button cell batteries effective 1-1-94. State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 1-1-94.

Table 4-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Batteries (continued)	California	<ul style="list-style-type: none"> Required manufacturer to reduce the level of mercury to 0.025% by weight in alkaline batteries effective 1-1-94, and ban on the manufacture of alkaline batteries containing any mercury effective 1-1-96. Ban on the manufacture of mercury-containing zinc carbon and mercury button cell batteries effective 1-1-94.
	Connecticut	<ul style="list-style-type: none"> Required manufacturer to reduce the level of mercury to 0.025% by weight in alkaline batteries effective 1-1-92. Ban on the manufacture of mercury-containing zinc carbon batteries effective 1-1-93. Collection of mercury button cell batteries required by retailer effective 1-1-92. State issued regulations defining responsibilities of manufacturer, supplier and user effective 1-1-92.
	Florida	<ul style="list-style-type: none"> Ban on the sale of alkaline batteries containing mercury greater than 0.025% by weight effective 7-1-95, and ban on the retail sale of alkaline and zinc carbon batteries containing any mercury effective 1-1-96. Ban on the sale of mercury button cell batteries effective 10-1-93. State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 1-1-94. Ban on the sale of mercury containing batteries if manufacturer fails to meet collection procedures and other responsibilities effective 1-1-94.
	Illinois	<ul style="list-style-type: none"> Task force to study storage, transport, disposal and recycling.
	Iowa	<ul style="list-style-type: none"> Ban on the sale of alkaline batteries containing mercury greater than 0.025% by weight effective 7-1-93, and ban on the retail sale of alkaline batteries containing any mercury effective 1-1-96. State issued regulations defining responsibilities of manufacturer, supplier and user effective 7-1-96. Ban on the sale of mercury-containing batteries (including button cells) if manufacturer fails to meet collection procedures and other responsibilities effective 7-1-96. Collection of mercury button cell batteries required by retailer effective 7-1-96.
	Maine	<ul style="list-style-type: none"> Ban on the sale of alkaline batteries containing mercury greater than 0.025% by weight effective 1-1-94, and ban on the manufacture of alkaline batteries containing any mercury effective 1-1-96. Ban on the sale of mercury containing zinc carbon batteries and mercury button cell batteries effective 1-1-93. State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 1-1-94.

Table 4-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Batteries (continued)	Maryland	<ul style="list-style-type: none"> • Collection of mercury button cell batteries required by retailer effective 7-1-94. • Ban on the sale of mercury button cell batteries if manufacturer fails to meet collection, transportation, disposal and consumer education responsibilities effective 7-1-94. • General ban on the sale of mercury containing batteries effective 7-1-94, but state authorized to grant exemptions if certain requirements are met. • State issued regulations defining responsibilities of manufacturer, supplier and user effective 7-1-94.
	Massachusetts	<ul style="list-style-type: none"> • Ban on the retail sale of mercury-containing alkaline batteries effective 1-1-95 is pending. • State regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries are pending. • Ban on the sale of mercury button cell batteries is pending.
	Michigan	<ul style="list-style-type: none"> • A new battery law signed on June 29, 1995. • This law bans the sale of alkaline batteries containing mercury (with the exception of alkaline manganese button cells containing less than 25 mg of mercury) and zinc carbon batteries containing mercury beginning January 1, 1996. • The sale of mercuric oxide batteries (with the exception of button cells) are also banned for sale after January 1, 1996, unless the manufacturer identifies a collection site for recycling, informs users of the locations and informs the purchasers of a telephone number that can be called to get information about returning mercuric oxide batteries for recycling or proper disposal.
	Minnesota	<ul style="list-style-type: none"> • Required manufacturer to sell alkaline batteries containing no more than 0.025% mercury by weight effective 2-1-92, and has banned manufacturer sale of alkaline batteries containing any mercury effective 1-1-96. • General ban on the sale of mercury-containing batteries effective 2-1-92, but state authorized to grant exemptions if certain requirements are met. • Mercury batteries may not contain more than 25 mg of mercury unless an exemption is granted. • Sale of dry cell batteries with mercuric oxide and electrode batteries prohibited without exemption. • Ban on mercury button cell batteries effective 2-1-92. • Manufacturers must set up collection, transport, recycling and consumer education programs.

Table 4-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Batteries (continued)	New Hampshire	<ul style="list-style-type: none"> • Required manufacturer to reduce the level of mercury in alkaline batteries to 0.025% by weight effective 1-1-93, and has banned the manufacturer sale of alkaline batteries containing any mercury effective 1-1-96. • Ban on the manufacture of mercury-containing zinc carbon batteries effective 1-1-93. • State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 1-1-93. • State issued regulations regarding the collection of mercury button cell batteries effective 1-1-93.
	New Jersey	<ul style="list-style-type: none"> • Required manufacturer to reduce the level of mercury to 0.025% by weight in alkaline batteries effective 1-1-92, and ban on the manufacture of alkaline batteries containing any mercury effective 1-1-96. • Ban on the manufacture of mercury-containing zinc carbon batteries effective 1-1-92. • Ban on mercury button cell batteries effective 1-1-94. • State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 1-20-93. • Sale of mercury batteries banned if manufacturer fails to meet collection and other responsibilities effective 1-20-93.
	New York	<ul style="list-style-type: none"> • Required manufacturer to reduce the level of mercury to 0.025% by weight in alkaline batteries effective 1-1-92. • Ban on the manufacture of mercury-containing zinc carbon batteries effective 1-1-93 • Mercury oxide battery ban pending.
	Oregon	<ul style="list-style-type: none"> • Required manufacturer to reduce the level of mercury to 0.025% by weight in alkaline batteries effective 1-1-92.
	Rhode Island	<ul style="list-style-type: none"> • Required manufacturer to reduce the level of mercury to 0.025% by weight in alkaline batteries effective 1-1-92. • State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 1-1-94. • Ban on mercury button cell batteries effective 1-1-93.

Table 4-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Batteries (continued)	Vermont	<ul style="list-style-type: none"> Required manufacturer to reduce the level of mercury in alkaline batteries by 0.025% by weight effective 2-1-92, and ban on the retail sale of alkaline batteries containing any mercury effective 1-1-96. Ban on mercury button cell batteries effective 1-1-93. State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 1-1-93.
	Wisconsin	<ul style="list-style-type: none"> Ban on the manufacture of mercury-containing alkaline batteries effective 1-1-96, and ban on the manufacture of mercury-containing zinc carbon batteries effective 7-1-94. State issued regulations defining collection procedures and responsibilities of the manufacturer and user of mercury batteries effective 7-1-94. Ban on the manufacture of mercury-containing batteries if manufacturer fails to meet collection procedures and other responsibilities effective 7-1-94.
Electrical Components	Pennsylvania	<ul style="list-style-type: none"> Mercury use in mining equipment (i.e., electrical machinery) is limited.
Paints and Pigments	Minnesota	<ul style="list-style-type: none"> No mercury can be deliberately introduced into products intended for use in Minnesota, except for art supplies.
Dental Uses	Minnesota	<ul style="list-style-type: none"> Disposal of dental equipment and supplies containing mercury is banned, unless the mercury is reused, recycled, or managed to ensure compliance. Purchaser must sign an agreement of use for medical or dental uses.
Toys	Michigan	<ul style="list-style-type: none"> No sale of toys containing mercury (pending).
	Minnesota Wisconsin	<ul style="list-style-type: none"> Toys with mercury are banned and fines are imposed on retail sales of toys containing mercury.
Lighting	Florida	<ul style="list-style-type: none"> Ban on incineration of lamps. Crushing, landfilling and recycling of lamps allowed with appropriate controls (proposed). Florida also will control management of residual mercury from recycling operations.
	Ohio	<ul style="list-style-type: none"> Lamps are viewed as by-products that can be recycled, and exempt from RCRA.

Table 4-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Lighting (continued)	Minnesota	<ul style="list-style-type: none"> Fluorescent lamps and high intensity lamps sold to managers of industrial, commercial, office, or multi-unit buildings must be labeled and building contractors must specify mercury management plans for removed lamps. Lamps in state-owned buildings must be recycled. Mercury must be removed from mercury vapor lights before disposal and lamp sellers and contractors must provide public education about mercury management requirements. The production and distribution of mercury vapor lights are limited.
	New York	<ul style="list-style-type: none"> Mercury vapor lights must be self-extinguishing or have protective shield.
Poisons	Ohio	<ul style="list-style-type: none"> The sale of mercury and mercury compounds is restricted.
	Pennsylvania	<ul style="list-style-type: none"> Levels established for mercury products considered to be safe.
Packaging	Florida	<ul style="list-style-type: none"> The mercury content in packaging and packaging components is restricted.
	Illinois Minnesota New York Wisconsin	<ul style="list-style-type: none"> Restrictions on mercury content in packaging are being phased in. (Dates and concentrations vary by state.)
	Pennsylvania	<ul style="list-style-type: none"> Intentionally introduced mercury in packaging is prohibited (pending).
Pharmaceuticals	Illinois Indiana New York Ohio Pennsylvania	<ul style="list-style-type: none"> Manufacturers must list the quantity of mercury in products.
Explosives/Fireworks	Minnesota Wisconsin	<ul style="list-style-type: none"> Explosives containing mercury are Class A maximum hazards. Permits required for fireworks with mercury (Minnesota only).

^a Adapted from Ross & Associates, 1994.

4.2 Management Alternatives and Statutory Authorities

4.2.1 Management Alternatives

Control of mercury emissions may require a mix of strategies including pollution prevention, materials separation and conventional regulatory strategies to control mercury emissions at the stack. Pollution prevention would be suitable for those processes or industries where a mercury substitute is demonstrated and available. Material separation is an appropriate approach for processes where mercury-containing products are disposed of by incineration, or where mercury can be reduced in the fuel prior to the fuel being combusted (e.g., coal cleaning). The third approach, conventional regulatory strategies, may be applicable when mercury is emitted to the environment as a result of trace contamination in fossil fuel or other essential feedstock in an industrial process (e.g., smelting). Other non-traditional approaches such as emissions trading or application of a use tax, or other market-based approaches may also prove feasible for mercury control. However, these options are not presented in detail in this Report as the control technology analyses focused on what might be achievable under the statutory language of sections 112 and 129 of the CAA.

The analyses on control technologies and costs presented in this Report are not intended to replace a thorough regulatory analyses as would be performed for a rulemaking. The information presented is intended to present the range of available options and provide a relative sense of the extent of mercury reductions achievable and the general magnitude of the cost of such reductions.

One possible means of achieving reductions in mercury emissions is through the use of pollution prevention or source reduction. Such approaches to achieving reductions involve changes in processes or inputs to reduce or eliminate emissions of mercury from a particular product or process. They could include, for example, the replacement of mercury with an appropriate substitute or the use of low-mercury content inputs.

In considering opportunities for pollution prevention or source reduction it is important to consider both the potential reductions achievable and the costs of these options. Any consideration of the potential reductions, should examine whether (and the extent to which) emission reductions from the particular sources in question will yield reductions in risk to public health and the environment. It is also essential to understand the costs associated with implementing a pollution prevention measure, including any changes in the quality of the end product.

Table 4-3 presents the six source categories for which a control technology and cost analysis was performed. The table presents the number of facilities in each category, the percent contribution of each to the national inventory and the relative potential for individual facilities within the source category to impact the area within 50 km of the facility. Potential national mercury reductions and potential national control costs are also presented. These estimates are based on the assumption that all plants within a source category will achieve the same reductions and incur the same costs as the model plants used in the analysis. Because this certainly would not be true in all circumstances, the estimates of potential reductions and costs should be used only for relative comparisons among the source categories to give an initial indication as to where mercury reductions would provide the most emission reduction for the least cost.

The estimates of cost for mercury reduction do not illustrate two important considerations. One is that all of the cost of control is attributed to mercury removal. As described previously in this Report, many of these controls achieve reductions of other pollutants as well (e.g., dioxin). The benefits of these additional reductions should also be considered. Second, the technologies available

Table 4-3
Potential Mercury Emission Reductions and Costs for Selected Source Categories

Mercury Source Category	Number of Facilities	% of U.S. Mercury Emission Inventory	Mercury Control Techniques	Potential National Reductions ^b	Potential National Annual Costs ^c	Cost-Effectiveness (\$/lb of mercury removed) ^d
Municipal waste combustors	149	23	Material separation Product substitution <u>Activated carbon injection</u> Carbon filter beds Polishing wet scrubber	50 tons (90% reduction)	\$56 million	\$211-870
Medical waste incinerators	~3,700	27	Material separation <u>Activated carbon injection</u> Polishing wet scrubber	60 tons (90% reduction)	\$24 million	\$228-955
Coal-fired utility boilers	426 (1,043 boilers)	21	Fuel switching Advanced coal cleaning Carbon filter beds <u>Activated carbon injection</u>	24-44 tons (50-90% reduction) ^e	\$2.9 billion	\$5,240-28,000
Chlor-alkali plants using the mercury cell process	14	2.7	<u>Process modification</u> Depleted brine scrubbing Treated activated carbon adsorption	6.5 tons (100% reduction)	\$70 million	\$4,590
Primary copper smelters	8	0.3 ^a	<u>Selenium filters</u>	>0.7 tons (90% reduction)	\$7.7 million	\$497
Primary lead smelters	3	3.7	<u>Selenium filters</u>	8 tons (90% reduction)	\$0.8 million	\$1,061
Total	~4,900	78			~\$3 billion	

NOTE: The underlined mercury control techniques are the techniques on which potential national reductions and potential national annual costs are based.

^a Reflects one smelter only; a national estimate would be higher.

^b Estimated reductions assuming every facility could achieve the reduction listed.

^c Potential national costs are estimates only and assume all facilities would incur the same costs as the model plants used in the analysis.

^d Where cost-effectiveness values are presented as a range, the values reflect the range across facilities of different sizes.

^e The range in potential national reductions reflects the variable efficiency of activated carbon injection to control mercury emissions from coal-fired utility boilers. Activated carbon injection has not been demonstrated for a full-scale utility boiler application. Control costs are based on the installation of spray cooler, fabric filter and carbon injection systems.

for mercury control represent relatively new applications of these technologies. Thus, in the future, it is possible that as new or emerging technologies develop, the cost-effectiveness of control may improve.

4.2.2 Clean Air Act Authorities

Mercury is a priority pollutant across numerous U.S. EPA programs including air, water, hazardous waste and pollution prevention. The focus of this section is the statutory authority under the CAA that could be used to control mercury emission sources. A brief summary of these authorities is presented below.

Section 112(a) Lesser Quantity Emission Rates (LQERs)

The U.S. EPA Administrator has the discretion to redefine major sources by setting an emissions cutoff lower than the 10 tons per year emission rate level for a single pollutant or 25 tons per year emission rate for a mixture of pollutants. This is referred to as a lesser quantity emission rate (LQER). The CAA states that LQERs are pollutant-specific and should be based on public health or environmental effects.

The major implications of setting an LQER are that all the requirements for a major source, including setting maximum achievable control technology (MACT) standards, mandatory residual risk analyses, calculation of the MACT floor, modification provisions and Title V permitting requirements become applicable to what was previously defined as an area source category.

Section 112(c)(6) List of Specific Pollutants

Section 112(c)(6) requires that by 1995, sources accounting for not less than 90 percent of the aggregate emissions of each of seven specific pollutants must be listed on the source category list, and be subject to standards under 112(d)(2) or (4) no later than 2005. The pollutants are: alkylated lead compounds; polycyclic organic matter; hexachlorobenzene; mercury; polychlorinated biphenyls; 2,3,7,8-tetrachlorodibenzo-p-dioxin; and 2,3,7,8-tetrachlorodibenzofuran. This provision makes a specific reference to utility boilers. It reads: "This paragraph shall not be construed to require the Administrator to promulgate standards for such pollutants emitted by electric steam generating units."

Section 112(d) Emission Standards

Section 112(d) requires that emission standards be established for each source category listed on the source category list. The emission standards are applicable to both new and existing sources and are based on the application of MACT. MACT is defined differently for new and existing sources as explained by 112(d)(2) and (3). Under 112(d)(4), if the pollutant is a threshold pollutant (i.e., noncarcinogen), the emission standard can be based on a health threshold with an ample margin of safety. A health threshold is a level where the risk of an adverse effect from exposure to the pollutant is negligible. Section 112(d)(5) allows the Administrator the discretion to apply generally available control technology (GACT) to area sources rather than MACT (or any other technologies that may be required of the source category on account of residual risk analyses under 112(f)).

Section 112(f) Residual Risk Program

Section 112(f) required the U.S. EPA to report to Congress on the methods that will be used to calculate the risk remaining after the promulgation of MACT emission standards under Section

112(d). This report should address the public health significance of the risk and the actual health effects experienced by persons living in the vicinity of emitting sources, and make recommendations on legislation regarding the risk. This report is due to Congress on November 15, 1996. If Congress does not accept any of the recommendations provided for reducing the residual risk, the Administrator has the authority to promulgate any additional standards required in order to protect public health with an ample margin of safety. The report is currently under development.

Section 112(k) Urban Area Source Program

By 1995, a national strategy to control emissions of hazardous air pollutants (HAPs) from area sources in urban areas must be transmitted to Congress. The strategy must identify not less than 30 HAPs which present the greatest threat to public health in the largest number of urban areas. Source categories accounting for at least 90 percent of the aggregate emissions of each HAP must be listed on the source category list and be subject to 112(d) standards. The strategy, when implemented, is to achieve a 75 percent reduction in cancer incidence attributable to these sources.

The urban area source program is a section 112 authority that does not require that a finding of adverse effects be made for the area source category (as required for listing by 112(c)(3)). Mercury is a likely candidate for the urban area source program.

112(m) Atmospheric Deposition to Great Lakes and Coastal Waters (Great Waters)

The Great Waters study is an ongoing study with periodic reports to Congress required. This program must identify and assess the extent of atmospheric deposition of HAPs to the Great Waters, the environmental and public health effects attributable to atmospheric deposition and the contributing sources. The first report was submitted in May 1994 and is to be submitted biennially hereafter. Mercury was identified as a priority pollutant under the Great Waters program. The Administrator must determine if other provisions under Section 112 will adequately control these sources. If not, by 1995, further emission standards to control these sources must be promulgated.

The recommendations of the first Great Water Report to Congress were (1) the U.S. EPA should strive to reduce emissions of the identified pollutants of concern through implementation of the CAA; (2) a comprehensive approach should be taken both within the U.S. EPA and between the U.S. EPA and other federal agencies to reduce and preferably prevent pollution in the air, water, and soil; and (3) the U.S. EPA should continue to support research for emissions inventories, risk assessment and regulatory benefits assessment.

112(n)(1)(A) Study of Hazardous Air Pollutants for Electric Utility Steam Generating Units

The Utility Study is required to address the hazards to public health that are reasonably anticipated to occur as a result of emissions by electric utility steam generating units of ... [hazardous air pollutants] ... after imposition of the requirements of the Act. The list of 189 HAPs is presented in section 112(b) of the CAA. In the study, the U.S. EPA must develop and describe alternative control strategies for HAPs that may require regulation under section 112, and, if appropriate and necessary, the U.S. EPA is to proceed with rulemaking to control HAP emissions from utility boilers. Mercury is one of the pollutants of concern for utilities.

Section 129 Solid Waste Combustion

Under this section, the Administrator must establish emission guidelines and standards for solid waste incineration units, including municipal waste combustors, medical waste incinerators and commercial and industrial waste incinerators. The performance standards must specify numerical emission limits for mercury as well as a number of other pollutants. The U.S. EPA has already issued final rules for municipal waste combustors (FR notice for final) and proposed rules for medical waste incinerators (60 FR 10654). Emission limits for hazardous waste combustors will be forthcoming under the Agency's Combustion Strategy.

5. 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.

- Conversion of mercury cell chlor-alkali plants to a mercury-free process is technically feasible and has been previously demonstrated.
- Energy conservation and switching to low-mercury fuels would reduce the amount of mercury being emitted by utility boilers.
- Injection of activated carbon into the flue gas of MWC's and MWI's can achieve mercury reductions of at least 85 percent. The addition of activated carbon to the flue gas of these source types would not have a significant impact on the amount of particulate matter requiring disposal.
- Numerous opportunities exist for replacing mercury in various products with other materials, such as solid state electronics for mercury switches, digital thermometers for mercury thermometers and zinc-air batteries for mercury batteries.
- Removing mercury-containing products such as batteries, fluorescent lights and thermostats from the waste stream can reduce the mercury input to waste combustors without lowering the energy content of the waste stream. The mercury removal efficiency would vary, however, depending on the extent of the separation.
- Selenium filters are a demonstrated technology in Sweden for control of mercury emissions from lead smelters. Carbon filter beds have been used successfully in Germany for mercury control on utility boilers and MWC's. These technologies have not been demonstrated in the U.S.
- Activated carbon injection provides variable control of mercury for utility boilers, based on limited pilot-scale testing. The most important factors affecting mercury control on utility boilers include the flue gas volume, flue gas temperature and chloride content, the mercury concentration and chemical form of mercury being emitted. The addition of activated carbon to utility flue gas for mercury control would significantly increase the amount of particulate matter requiring disposal.
- The available data on coal cleaning indicate that mercury reductions ranged from zero to 64 percent. The average reduction was 21 percent. This variation may be due to several factors including different cleaning methods, different mercury concentrations in the raw coal and different mercury analytical techniques. There are no data available to assess the potential for mercury emissions from coal-cleaning slurries.
- Control technologies designed for control of pollutants other than mercury (e.g., acid gases and particulate matter) vary in their mercury-removal capability, but in general achieve reductions no greater than 50 percent.

There are many uncertainties associated with the cost analysis for individual source categories due to assumptions inherent in a model plant approach. The impact of these uncertainties on the analyses include the following:

- The cost of mercury control incurred by any specific facility may be underestimated by the cost analysis presented in this Report because of variability inherent in the assumptions that were made in the analyses. These include the efficiency of the various control techniques for reducing mercury, the amount of mercury in the flue gas stream and other site-specific factors such as down-time and labor costs. In addition, costs for monitoring and record keeping were not included in the cost analyses.
- The financial impacts calculated in this Report may not represent the financial status of all facilities within a source category. The affordability of mercury control will depend on a given facility's profits and expenditures.
- In the control technology analysis, the cost of control was attributed to mercury reduction only. Because many of the controls also reduce emissions of other pollutants, apportioning the costs across all pollutants reduced would better characterize the cost of mercury control.

6. RESEARCH NEEDS

To improve the control technology, cost and impacts assessment, the U.S. EPA would need the following:

1. Data from full-scale testing of activated carbon injection at a coal-fired utility boiler.
2. Additional data are needed on the efficiency of activated carbon injection, and various impregnated carbons, in reducing the different chemical species of mercury present in flue gas.
3. Additional information on the efficiency and cost of other technologies for mercury control that are currently in the research stage. These include impregnated activated carbon, sodium sulfide injection and activated carbon fluidized bed.
4. More data are needed on both the ability of conventional or advanced coal cleaning techniques to remove mercury from raw coal and advanced coal cleaning techniques such as selective agglomeration and advanced column floatation. The potential for mercury emissions from coal-cleaning slurries needs to be characterized.
5. Additional analyses are required on the feasibility, cost effectiveness of other mercury emission prevention measures such as emissions trading, emissions averaging, energy conservation, renewable energy, and fuel switching.

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APPENDIX A TEST DATA ON THE EFFECTIVENESS OF ACTIVATED CARBON INJECTION FOR UTILITY BOILERS

Limited test data indicate that activated carbon (AC) injection effectively reduces mercury emissions when used in conjunction with existing control devices, such as fabric filters (FFs) and spray dryer absorbers (SDAs).

Table A-1 presents pilot-scale test data on the mercury removal efficiency of AC injection when used ahead of FFs. Such a configuration, with no prior PM control, has a median mercury removal efficiency that varies with temperature and AC injection rate. With a low AC injection rate (<1,000 wt C/wt inlet Hg) and an average flue gas temperature between 107°C (225°F) and 121°C (250°F), a median mercury removal efficiency of 29 percent was found, with a range from 14 percent to 47 percent removal. With a low AC injection rate (same as above) and an average flue gas temperature between 88°C and 107°C, a median mercury removal efficiency of 97 percent was found, with a range from 76 percent to 99 percent removal. A high AC injection rate (>1,000 wt C/wt inlet Hg) and an average flue gas temperature between 107°C (225°F) and 121°C (250°F) produced a median mercury removal efficiency of 82 percent, with a range from 69 percent to 91 percent removal. A high AC injection rate (same as above) and an average flue gas temperature between 88°C (190°F) and 107°C (225°F) produced a median mercury removal efficiency of 98 percent, with a range from 95 percent to 99 percent removal (Chang et al., 1993).

Table A-2 presents test data for AC injection when used before SDA systems. Tested SDA/ESP systems with AC injection had a median mercury removal efficiency of 85.9 percent, with a range from 74.5 percent to 90.9 percent removal (Felsvang, 1993). Pilot-scale testing of a SDA/FF system with AC injection had a median mercury removal efficiency of 60 percent, with a range from 50 percent to 99 percent removal (Felsvang, 1993).

Table A-1
Activated Carbon Injection Before Fabric Filter Data^a

Unit	Control Device	Hg removal %
Low temperature + low carbon injection rate (< 1000 wt C/wt Hg) based on inlet Hg		
Test #4, Run #1	AC + FF (88°C (190°F) and 216 wt C/wt inlet Hg; inlet Hg concentration of 5.35 µg/dscm)	97
Test #4, Run #2	AC + FF (88°C (190°F) and 126 wt C/wt inlet Hg; inlet Hg concentration of 8.19 µg/dscm)	99
Test #4, Run #3	AC + FF (91°C (196°F) and 123 wt C/wt inlet Hg; inlet Hg concentration of 8.62 µg/dscm)	97
Test #6, Run #3	AC + FF (102°C (216°F) and 727 wt C/wt inlet Hg; inlet Hg concentration of 1.94 µg/dscm)	76
High temperature + low carbon injection rate (< 1000 wt C/wt Hg) based on inlet Hg		
Test #5, Run #1	AC + FF (107°C (225°F) and 362 wt C/wt inlet Hg; inlet Hg concentration of 5.53 µg/dscm)	14
Test #5, Run #2	AC + FF (110°C (230°F) and 373 wt C/wt inlet Hg; inlet Hg concentration of 4.45 µg/dscm)	28
Test #5, Run #3	AC + FF (116°C (241°F) and 457 wt C/wt inlet Hg; inlet Hg concentration of 3.47 µg/dscm)	47
Test #6, Run #1	AC + FF (121°C (250°F) and 286 wt C/wt inlet Hg; inlet Hg concentration of 5.04 µg/dscm)	29
Test #6, Run #2	AC + FF (118°C (244°F) and 367 wt C/wt inlet Hg; inlet Hg concentration of 4.22 µg/dscm)	35
Low temperature + high carbon injection rate (> 1000 wt C/wt Hg) based on inlet Hg		
Test #2, Run #1	AC + FF (91°C (196°F) and 2843 wt C/wt inlet Hg; inlet Hg concentration not measured but assumed to be 7.00 µg/dscm)	95
Test #2, Run #2	AC + FF (96°C (205°F) and 3132 wt C/wt inlet Hg; inlet Hg concentration not measured but assumed to be 7.00 µg/dscm)	98
Test #2, Run #3	AC + FF (93°C (199°F) and 3121 wt C/wt inlet Hg; inlet Hg concentration not measured but assumed to be 7.00 µg/dscm)	98
Test #3, Run #2	AC + FF (93°C (199°F) and 4361 wt C/wt inlet Hg; inlet Hg concentration of 6.23 µg/dscm)	99

Table A-1
Activated Carbon Injection Before Fabric Filter Data
(continued)

Unit	Control Device	Hg removal %
Test #3, Run #3	AC + FF (96°C (205°F) and 3850 wt C/wt inlet Hg; inlet Hg concentration of 6.91 µg/dscm)	99
High temperature + high carbon injection rate (> 1000 wt C/wt Hg) based on inlet Hg		
Test #3, Run #1	AC + FF (110°C (230°F) and 3332 wt C/wt inlet Hg; inlet Hg concentration of 7.95 µg/dscm)	91
Test #7, Run #1	AC + FF (121°C (250°F) and 1296 wt C/wt inlet Hg; inlet Hg concentration of 4.66 µg/dscm)	69
Test #7, Run #2	AC + FF (121°C (250°F) and 1954 wt C/wt inlet Hg; inlet Hg concentration of 4.30 µg/dscm)	76
Test #7, Run #3	AC + FF (116°C (241°F) and 3649 wt C/wt inlet Hg; inlet Hg concentration of 2.09 µg/dscm)	87

^a Source: Chang et al., 1993

Table A-2
Activated Carbon Injection Before Spray Dryer Absorption Data^a

Unit	Control Device	Hg Removal %
SDA/ESP		
Denmark	AC + SDA/ESP (inlet Hg concentration ranges from 3.5 - 7.9 µg/dscm)	80.3, 85.8, 75.8, 74.5, 90.9, 89.5, 89.3, 86.7, 85.9
SDA/FF		
NSP Sherco 3 (pilot unit)	AC + SDA/FF (inlet Hg concentration unknown)	50-60
Plant D2	AC + SDA/FF (inlet Hg concentration of 3.9 µg/dscm)	>99

^a Source: Felsvang, 1993

APPENDIX B

MODEL PLANT COST EVALUATION

This appendix presents model plant cost analyses for installing and operating applicable mercury control techniques at each of the six source types described in Chapter 2. The cost estimates are based on information supplied by various vendors and taken from the literature, and may not reflect commercial considerations such as vendor guarantees. The cost estimates reflect generalized costs and are not intended to be site specific. Plant-to-plant variations can result in higher or lower technology performance and associated costs.

An effort was made to convert the dollars to reflect the currency rate in the early 1990s. For carbon filter beds and selenium filters, conversion from foreign currencies to U.S. dollars is based on a June 1993 exchange rate.

B.1 Municipal Waste Combustors

B.1.1 Model Plant Description

Two model plants were selected to represent MWCs based on the model plants used in the EPA's model plant cost report for MWCs (U.S. EPA, 1989a). The first is a small mass burn/waterwall (MB/WW) MWC with two units and a total plant capacity of 180 Mg/day (200 tpd). This model plant is equipped with dry sorbent injection (DSI) and an ESP. Few MWCs are currently equipped with this control device combination (Fenn and Nebel, 1992), but it is expected that a number of MWCs will be installing this technology in the future. The second model plant is a large MB/WW MWC with three units and a total plant capacity of 2,045 Mg/day (2,250 tpd) of MSW. This model plant is equipped with a SD/FF pollution control system. Both model plants are assumed to operate at full capacity 90 percent of the year.

Inlet mercury levels for the two model plants are assumed to be 700 $\mu\text{g}/\text{dscf}$ (306 gr/million dscf) based on typical inlet levels at MWCs (U.S. EPA, 1993). The model plant analyses assume a baseline DSI/ESP mercury reduction of 15 percent and a baseline SD/FF mercury reduction of 30 percent. This results in a mercury level before add-on controls of 595 $\mu\text{g}/\text{dscf}$ (260 gr/million dscf) for the small model plant and 490 $\mu\text{g}/\text{dscf}$ (214 gr/million dscf) for the large model plant. The assumed baseline control efficiency reflects typical MWC operation; however, actual values vary from plant to plant.

The following sections present the cost analyses of different mercury control options for MWCs. Section B.1.2 provides a qualitative cost discussion on material separation costs for batteries. Sections B.1.3 through B.1.5 provide quantitative cost information on applying activated carbon injection, CFB's and a polishing wet scrubber to the MWC model plants. A sensitivity analysis of the effect of lower mercury levels in the flue gas is also included.

B.1.2 Material Separation

Comprehensive cost data on battery separation programs in the United States are not available. Most programs are operated by local governments, and the expenses for administration, overhead and educational/promotional efforts typically are not reported. Hennepin County, Minnesota, reported that their voluntary drop-off program for household batteries costs an average of \$175,000 per year, which

is roughly equivalent to \$0.40/Mg (\$0.37/ton) of MSW combusted. This cost included collection, sorting, disposal and associated administration costs (Shepherd, 1993; White and Jackson, 1993). During the first 3 years of operation, this program collected an estimated 330 kg (730 lb) of mercury--equivalent to reducing the mercury content of the waste stream by approximately 13 percent (White and Jackson, 1993). Assuming a 50 percent mercury control efficiency by the SD/FF system at the Hennepin County MWC, this equates to a cost effectiveness of approximately \$3,180/kg¹ (\$1,450/lb) of reduced mercury emissions. Costs of other community household battery separation programs are not readily available, and they will vary depending on community-specific conditions.

Once household batteries that test hazardous have been collected, they must be disposed of at a hazardous waste facility or sent to a metals recycler. The only mercury-containing batteries that can be recycled currently in the United States are mercury-zinc batteries and mercuric oxide batteries that have been sorted by type. Mercury Refining Company in Lathem, New York, accepts sorted mercury-zinc button cells for a fee of \$3.30/kg (\$1.50/lb) of batteries (shipping not included) (Shepherd, 1993).

B.1.3 Activated Carbon Injection

Table B-1 presents estimated costs for using activated carbon injection on the 180-Mg/day (200-tpd) and the 2,050-Mg/day (2,250-tpd) MWCs. Carbon injection is assumed to achieve an average mercury reduction of 85 percent with a carbon feed rate of 320 mg/dscm (0.14 gr/dscf) for the DSI/ESP-equipped plant (Kilgroe et al., 1993) and 75 mg/dscm (0.033 gr/dscf) for the SD/FF-equipped plant (U.S. EPA, 1992c).² This results in outlet mercury levels after carbon injection of approximately 90 and 75 µg/dscm (35 and 26 gr/million dscf) for the two MWCs respectively. The DSI/ESP-equipped plant requires a higher feed rate because the ESP does not provide secondary reaction between the carbon and the flue gas, as occurs on the bags of the FF. Also, the flue gas temperature is higher with the DSI/ESP system than with the SD/FF system (177°C [350°F] vs. 135°C [275°F]).

The capital cost listed in Table B-1 includes purchased equipment (PE), installation, indirect and contingency costs for the mercury control system. The major equipment items include a carbon holding tank and metering system, a pneumatic feed system and injection ports into the ductwork. These costs are estimated to be approximately \$87,100 and \$372,000 for the small and large MWCs, respectively, and include the costs of installation (Guest and Knizek, 1991). Indirect costs are estimated to be 33 percent of PE costs, and contingency costs are estimated to be 20 percent of PE costs plus indirect costs. These costs are based on the cost factors used for a DSI system in the MWC cost procedures (U.S. EPA, 1989b). Installation is assumed to occur during a regular plant outage, so no cost associated with downtime is included in the capital costs.

The operating and maintenance (O&M) cost includes labor; maintenance materials; activated carbon costs; disposal; overhead; and taxes, insurance and administrative charges. With the exception of carbon, disposal costs and capital recovery, costs are based on the MWC

¹ \$175,000/yr divided by 110 kg/yr divided by 0.50 = \$3,180/kg.

² Mercury reduction can be increased with higher levels of carbon.

Table B-1
Cost Estimates for Activated Carbon Injection on Municipal Waste Combustors^a

Parameter	Small Model Plant	Large Model Plant
Unit Size (Mg/day)	90	682
Number of Units	2	3
Plant Size (Mg/day)	180	2,045
Flue Gas Flow (dscm/hr @ 7% O ₂)	30,500	343,000
Air Pollution Control Device	DSI/ESP	SD/FF
Hg Level Before Carbon (µg/dscm @ 7% O ₂)	595	490
Hg Level After Carbon (µg/dscm @ 7% O ₂)	90	75
Capacity Factor (%)	90	90
CAPITAL COST (\$)		
Purchased Equipment (PE) ^b	87,100	372,000
Installation ^c	0	0
Indirect ^d	28,700	123,000
Contingency ^e	23,200	99,000
Total Capital Cost (TCC)	139,000	594,000
OPERATING AND MAINTENANCE (O&M) (\$/yr)		
Operating Labor ^f	52,600	78,800
Supervision ^g	7,880	11,800
Maintenance Labor ^h	14,500	21,700
Maintenance Materials ⁱ	4,350	18,600
Power ^j	0	0
Carbon ^k	84,800	222,000
Disposal ^l	0	0
Overhead ^m	47,600	78,600
Taxes, Insurance, Administration ⁿ	5,560	23,700
Capital Recovery ^o	15,300	65,200
Total (\$/yr) ^p	232,000	520,000
\$/Mg MSW	3.90	0.77
\$/ton MSW	3.50	0.70
Mercury Reduction (kg/yr)	121	1,120
Mercury Reduction (lb/yr)	267	2,470
\$/kg Mercury	1,910	464
\$/lb Mercury	870	211

^a Cost numbers may not add exactly due to round-off.

^b Based on \$200,000 equipment costs for a 730-Mg/day MWC. Equipment costs scaled based on 0.6 rule.

^c Included in PE costs.

^d 33 percent of PE costs.

^e 20 percent of PE + indirect costs.

^f 2 hr/shift @ \$12/hr (per unit).

^g 15 percent of operating labor costs.

^h 0.5 hr/shift @ 10 percent wage rate premium over labor wage (per unit).

ⁱ 5 percent of operating labor costs.

^j Power costs associated with the active carbon injection system are assumed to be negligible.

^k Based on \$1.10/kg of carbon and a carbon feed rate of 320 and 75 mg/dscm for the two plants, respectively.

^l Increase in disposal costs due to carbon injection are assumed to be negligible.

^m 60 percent of labor and maintenance costs.

ⁿ 4 percent of TCC.

^o Using a capital recovery factor of 0.1098 (7 percent interest rate for 15 years).

^p Total costs equal capital recovery costs plus O&M costs.

cost procedures. The cost of activated carbon ranges from \$1.10 to \$2.20/kg (\$0.50 to 1.00/lb) depending on the raw material used to produce the carbon and the specific surface area of the carbon (Brown, 1991). Based on a carbon cost of \$1.10/kg (\$0.50/lb) and a carbon feed rate of 75 mg/dscm (0.033 gr/dscf), the carbon costs are roughly \$0.33/Mg (\$0.30/ton) of MSW. At a feed rate of 320 mg/dscm (0.14 gr/dscf), carbon costs are approximately \$1.40/Mg (\$1.30/ton) of MSW. The addition of carbon into the flue gas has a negligible impact on the quality of collected PM requiring disposal. Therefore, the ash disposal costs associated with an activated carbon injection system are assumed to be negligible. The capital recovery factor (CRF) is based on a 7 percent interest rate annualized over 15 years (CRF = 0.1098). Total costs include the capital recovery costs and O&M costs. The largest contributions to the cost of an activated carbon system are typically carbon and labor costs.

The overall estimated costs for an activated carbon injection system are approximately \$3.9/Mg (\$3.5/ton) of MSW for the 180-Mg/day (200-tpd) MWC and \$0.77/Mg (\$0.70/ton) of MSW for the 2,045-Mg/day (2,250-tpd) MWC. Of this amount, operating expenses account for approximately 80 percent of the total. On the basis of dollars per kilogram of mercury removed from the flue gas, the costs are \$1,910/kg (\$870/lb) and \$464/kg (\$211/lb) for the 180-Mg/day (200-tpd) and the 2,045 Mg/day (2,250-tpd) MWCs, respectively.

Table B-2 provides a sensitivity analysis showing the impact of mercury input level on the costs expressed in dollars per megagram of MSW burned and dollars per kilogram of mercury removed. For the purpose of the sensitivity analysis, the mercury input level was assumed to be reduced by 50 percent through materials separation or other means while the carbon feed rate is maintained at the same level. Under these assumptions, the outlet mercury levels are reduced to 45 µg/dscm (20 gr/million dscf) and 35 µg/dscm (15 gr/million dscf). The total annualized cost of the carbon system remains the same, but the cost per kilogram of mercury removed increases. For the 180-Mg/day (200-tpd) MWC, the cost increases to \$3,790/kg (\$1,720/lb) of mercury removed. For the 2,045-Mg/day (2,250-tpd) MWC, the cost increases to \$915/kg (\$416/lb) of mercury removed. Costs for a materials separation program are not included in the sensitivity analysis.

Table B-2
Sensitivity Analysis for
Activated Carbon Injection System on MWCs

Operating Assumptions	180-Mg/day MWC		2,045-Mg/day MWC	
	\$/Mg MSW	\$/kg Hg	\$/Mg MSW	\$/kg Hg
Base Case (Table A-1)	3.9	1,910	0.77	464
50% Lower Hg Input ^a	3.9	3,790	0.77	915

^a Inlet mercury level (upstream of APCD) is reduced to 350 µg/dscm. There is no change in capital or operating costs. The amount of mercury removed is the only factor affected. The outlet mercury level is 40 µg/dscm and 30 µg/dscm for the small and large plants, respectively.

B.1.4 Carbon Filter Beds

Table B-3 presents estimated costs for applying a carbon bed filter to the 180-Mg/day (200-tpd) and the 2,045-Mg/day (2,250-tpd) MWCs. The carbon filter bed is assumed to achieve greater than 99 percent reduction of mercury emissions, with outlet levels for both MWCs of 1 µg/dscm (0.44 gr/million dscf) (Hartenstein, 1993c).

The capital cost includes PE, installation, indirect and contingency costs. The major equipment items include the filter, the steel structure, carbon conveyors and storage and ducting. These costs are estimated to be \$10,500,000 for the 180-Mg/day (200-tpd) MWC and \$45,000,000 for the 2,045-Mg/day (2,250-tpd) MWC (Petersen, 1993). Indirect costs are estimated to be 28 percent of the PE costs based on the cost factor used for carbon adsorbers in the OAQPS Control Cost Manual (U.S. EPA, 1992d). Contingency costs are estimated to be 10 percent of PE costs. This contingency factor is limited to process uncertainties and reflects the technical and economic risks associated with a new process application (Electric Power Research Institute, 1986). For the small MWC, no cost associated with downtime for installing the filter bed is included. It is assumed that this retrofit will occur during other retrofits (e.g., of the DSI system) needed to comply with anticipated new source performance standard (NSPS) revisions under the Clean Air Act. For the large MWC, which is assumed to already have the SD/FF system in place, 1 month of downtime is included for installation of the filter system. Costs associated with the downtime are based on the EPA's MWC cost procedures (U.S. EPA, 1989b).

The O&M costs include labor; maintenance materials; carbon; power; disposal; overhead; taxes, insurance and administrative charges; and capital recovery. With the exception of carbon, power, disposal and capital recovery, these costs are based on the MWC cost procedures (U.S. EPA, 1989b). The annual carbon cost is based on the activated carbon price of \$1.10/kg (\$0.50/lb) and a usage rate of 2.20 kg carbon/Mg (4.4 lb carbon/ton) of MSW (Hartenstein, 1993a). Activated carbon is used in this estimate instead of the cheaper German Hearth Oven Coke (HOC) (~\$0.35/kg [\$0.15/lb]) because the availability of HOC in the United States is unknown.

The cost of power is based on a pressure drop of 305 mm (12-in. water) and a cost of 46 mills per kilowatt-hour (mills/kWh) (Hartenstein, 1990; U.S. EPA, 1989b). Disposal costs are based on a hazardous waste disposal rate of \$200/ton. A CRF of 0.1098 (7 percent over 15 years) was used to annualize the capital costs.

The estimated costs for a filter bed system are approximately \$10/Mg (\$9/ton) and \$6/Mg (\$5/ton) for the 180-Mg/day (200-tpd) and 2,045-Mg/day (2,250-tpd) MWCs, respectively. Of this amount, 75 to 80 percent is directly related to the capital cost of the system (including taxes, insurance and administration). On the basis of dollars per kilogram of mercury removed, the costs are \$2,378/kg (\$1,083/lb) and \$1,130/kg (\$513/lb) for the two unit sizes, respectively.

Table B-4 shows the results of a sensitivity analysis to determine the impact of reduced mercury input on the costs expressed in dollars per megagram of waste burned and dollars per kilogram of mercury removed. As discussed in section 2.2.1, the size of the filter and the amount of carbon used depend on flue gas flow rate and pressure drop, not mercury inlet level. As a result, when the mercury inlet level is reduced by 50 percent through materials separation or other means, the capital and operating costs do not change. The cost per kilogram of mercury removed, however, increases substantially because of the decreased amount of mercury removed annually by the filter bed. For the 180-Mg/day (200-tpd) MWC, the cost increases to \$4,756/kg (\$2,161/lb) of mercury removed. For the 2,045-Mg/day (2,250-tpd) MWC, the cost increases to \$2,260/Mg (\$1,027/lb) of mercury

Table B-3
Cost Estimate for a Carbon Filter Bed on MWC's^a

Parameter	Small Model Plant	Large Model Plant
Unit Size (Mg/day)	90	682
Number of Units	2	3
Plant Size (Mg/day)	180	2,045
Flue Gas Flow (dscm/hr @ 7% O ₂)	30,500	343,000
Air Pollution Control Device	DSI/ESP	SD/FF
Hg Level Before Filter (µg/dscm @ 7% O ₂)	595	490
Hg Level After Filter (µg/dscm @ 7% O ₂)	1	1
Capacity Factor (%)	90	90
CAPITAL COST (\$)		
Purchased Equipment (PE) ^b	330,834	2,315,284
Installation ^c	0	0
Indirect ^d	92,634	648,280
Contingency ^e	33,083	231,528
Total Capital Cost (TCC)	456,551	3,195,092
Downtime ^f	0	1,325,000
TCC + Downtime	456,551	4,520,092
OPERATING AND MAINTENANCE (O&M) (\$/yr)		
Operating Labor ^g	105,000	158,000
Supervision ^h	15,750	23,700
Maintenance Labor ⁱ	14,500	21,700
Maintenance Materials ^j	4,566	31,951
Carbon ^k	16,278	166,084
Power ^l	29,700	307,000
Disposal ^m	1,978	20,179
Overhead ⁿ	83,889	141,211
Taxes, Insurance, Administration ^o	18,262	127,804
Capital Recovery ^p	60,129	496,306
Total (\$/yr)^q	340,052	1,493,934
\$/Mg MSW	10.35	6.00
\$/ton MSW	9.39	5.44
Mercury Reduction (kg/yr)	143	1,322
Mercury Reduction (lb/yr)	314	2,910
\$/kg Mercury	2,378	1,130
\$/lb Mercury	1.083	513

^a Cost numbers may not add exactly due to round-off.

^b Based on order of magnitude estimates for carbon usage.

^c Included in PE costs.

^d 28 percent of PE costs.

^e 10 percent of PE costs.

^f Downtime of 15 days assumed for large MWC. For small MWC, installation assumed to occur during other retrofits, so downtime costs are zero.

^g 4 hr/shift @ \$12/hr (per unit).

^h 15 percent of operating labor costs.

ⁱ 0.5 hr/shift @ 10 percent wage rate premium over labor wage (per unit).

^j 1 percent of TCC.

^k Based on \$1.10/kg of carbon and 2.2 kg carbon/Mg MSW.

^l Using a pressure drop of 305 mm (water) and 46 mills per kilowatt-hour (mills/kWh).

^m Based on a hazardous waste disposal rate of \$220/Mg.

ⁿ 60 percent of labor and maintenance costs.

^o 4 percent of TCC.

^p Using a capital recovery factor of 0.1098 (7 percent interest rate for 15 years).

^q Total costs equal capital recovery costs plus O&M costs.

Table B-4
Sensitivity Analysis for a Carbon Filter Bed System on MWCs

Operating Assumptions	180-Mg/day MWC		2,045-Mg/day MWC	
	\$/Mg MSW	\$/kg Hg	\$/Mg MSW	\$/kg Hg
Base Case (Table B-3)	10	2,378	6	1,130
50% Lower Hg Input ^a	20	4,756	12	2,260

^a Inlet mercury level (upstream of APCD) is reduced to 325 µg/dscm. There is no change in capital or operating costs. The amount of mercury removed is the only factor affected. The outlet mercury level remains at 1 µg/dscm.

removed. Costs associated with a materials separation program are not included in the sensitivity analysis.

B.1.5 Wet Scrubbing

Table B-5 presents estimated costs for applying a polishing WS system on the 180-Mg/day (200-tpd) and the 2,045 Mg/day (2,250-tpd) MWCs. The polishing WS system is assumed to achieve 85 percent reduction of the mercury level after the APCD, resulting in outlet mercury levels of approximately 90 and 75 µg/dscm (35 and 26 gr/million dscf) for the two MWCs, respectively.

The capital cost includes PE, installation, indirect, contingency and downtime costs. The PE costs are estimated to be \$2,600,000 and \$13,000,000 for the small and large MWCs, respectively and include the costs of installation (Nebel et al., 1994). Indirect costs are estimated to be 33 percent of PE costs and contingency costs are estimated to be 20 percent of PE costs plus indirect costs. These costs are based on the cost factors used in the MWC cost procedures (U.S. EPA, 1989b). For the small MWC, no cost associated with downtime for installing the WS system is included. It is assumed that this retrofit will occur during other APCD retrofits. For the large MWC, which is assumed to already have the SD/FF system in place, 1 month of downtime is included for installation of the WS system. Costs associated with the downtime are based on the EPA's MWC cost procedures (U.S. EPA, 1989b).

The costs for operating and maintenance labor, electric power, lime, water, water treatment and residue disposal are based on information provided from a wet scrubber vendor (Nebel et al., 1994). Costs for overhead and taxes, insurance and administration charges are based on the EPA's MWC cost procedures (U.S. EPA, 1989b). A CRF of 0.1098 (7 percent over 15 years) was used to annualize the capital costs.

The overall estimated costs for a polishing WS system equate to approximately \$14.9/Mg (\$13.5/ton) and \$5.9/Mg (\$5.3/ton) of MSW for the small and large model plants, respectively. The cost effectiveness values are \$7,300/kg (\$3,320/lb) of mercury removed for the small MWC and \$3,520/kg (\$1,600/lb) of mercury removed for the large MWC.

Table B-5
Cost Estimate for a Polishing Wet Scrubbing
System on Municipal Waste Combustors^a

Parameter	Small Model Plant	Large Model Plant
Unit Size (Mg/day)	90	682
Number of Units	2	3
Plant Size (Mg/day)	180	2,045
Flue Gas Flow (dscm/hr @ 7% O ₂)	30,490	343,000
Air Pollution Control Device	DSI/ESP	SD/FF
Inlet Hg Level (µg/dscm @ 7% O ₂)	595	490
Outlet Hg Level (µg/dscm @ 7% O ₂)	90	75
Capacity Factor (%)	90	90
CAPITAL COST (\$)		
Purchased Equipment (PE) ^b	2,590,000	13,000,000
Installation ^c	0	0
Indirect ^d	854,000	4,290,000
Contingency ^e	688,000	3,460,000
Total Capital Cost (TCC)	4,130,000	20,750,000
Downtime ^f	0	270,000
TCC + Downtime	4,130,000	21,020,000
OPERATING AND MAINTENANCE (O&M) (\$/yr)^g		
Operating Labor	80,000	80,000
Supervision	12,000	12,000
Maintenance Labor	54,700	177,000
Electric Power	20,900	235,000
Reagent (Lime)	0	0
Water	3,300	37,100
Water Treatment	3,800	42,800
Residue Disposal	4,900	55,100
Overhead ^h	88,100	161,000
Taxes, Insurance, Administration ⁱ	165,000	830,000
Capital Recovery ^j	453,000	2,310,000
Total (\$/yr)^k	886,000	3,940,000
\$/Mg MSW	14.9	5.9
\$/ton MSW	13.5	5.3
Mercury Reduction (kg/yr)	121	1,120
Mercury Reduction (lb/yr)	267	2,470
\$/kg Mercury	7,300	3,520
\$/lb Mercury	3,320	1,600

^a Cost numbers may not add exactly due to round-off.

^b Based on cost estimates from Belco Technologies Corporation (35 percent less than PE cost for a two-stage system).

^c Included in PE costs.

^d 33 percent of PE costs.

^e 20 percent of PE + indirect costs.

^f Downtime of 1 month assumed for large MWC. For small MWC, installation assumed to occur during other retrofits, so downtime costs are zero.

^g Operating and maintenance costs are based on costs provided by Belco for a three-stage wet scrubbing system. Costs for electric power, water, water treatment and residue disposal were cut in half in attempt to account for the smaller system.

^h 60 percent of labor and maintenance costs.

ⁱ 4 percent of TCC.

^j Using a capital recovery factor of 0.1098 (7 percent interest for 15 years).

^k Total costs equal capital recovery costs plus O&M costs.

Table B-6 provides a sensitivity analysis showing the impact of mercury input level on the costs expressed in dollars per megagram of MSW burned and dollars per kilogram of mercury removed. Assuming the mercury input is reduced by 50 percent through materials separation or other means, the total annualized cost remains the same, since capital and operating costs of the WS system do not change. Outlet mercury levels are reduced to 45 µg/dscm (17 gr/million dscf) and 35 µg/dscm (13 gr/dscf) for the small and large MWCs, respectively. The cost per kilogram of mercury removed, however, increases substantially because of the decreased amount of mercury removed annually by the WS system. For the 180-Mg/day (200-tpd) MWC, the cost increases to \$14,570/kg (\$6,590/lb) of mercury removed. For the 2,045-Mg/day (2,250-tpd) MWC, the cost increases to \$6,930/kg (\$3,140/lb) of mercury removed. Costs for a materials separation program are not included in the sensitivity analysis.

Table B-6
Sensitivity Analysis for a
Polishing Wet Scrubbing System on MWCs

Operating Assumptions	180-Mg/day MWC		2,045-Mg/day MWC	
	\$/Mg MSW	\$/kg Hg	\$/Mg MSW	\$/kg Hg
Base Case (Table B-5)	14.9	7,300	5.9	3,520
50% Lower Hg Input ^a	14.9	14,570	5.9	6,930

^a Inlet mercury level (upstream of APCD) is reduced to 350 µg/dscm. There is no change in capital or operating costs. The amount of mercury removed is the only factor affected. The outlet mercury level is 45 µg/dscm and 35 µg/dscm for the small and large plants, respectively.

B.2 Medical Waste Incinerators

B.2.1 Model Plant Description

Two model plants have been selected to represent the MWI category based on the model plants used in the EPA's MWI study (U.S. EPA, 1991d; U.S. EPA, 1994e). The first is a small batch-run MWI with a design capacity of 90 kg/hr (200 lb/hr), but operating at an average throughput of 60 kg/hr (130 lb/hr) because of the high heat content of the waste. This plant is assumed to operate 5.5 hr/day, 5 days/week (which equals a capacity factor of 20 percent), and is equipped with a DSI/FF system. Currently, few MWIs have this level of control. For the purpose of evaluating mercury controls on MWIs, however, it was assumed that acid gas and PM controls would be installed to comply with other regulatory requirements. Costs for DSI/FF controls are not included in the mercury cost analysis since it is not appropriate to attribute the cost of acid gas and PM controls solely to the removal of mercury.

The second model plant is a larger, commercial MWI. It has a design capacity of 680 kg/hr (1,500 lb/hr), but operates at 455 kg/hr (1,000-lb/hr). It operates 24 hr/day, 7 days/wk, and is assumed to operate 90 percent of the year. This MWI is also assumed to be equipped with a DSI/FF system.

The inlet mercury level for the MWI model plants is assumed to be 3,000 µg/dscm (1,310 gr/million dscf). No control of mercury is assumed to occur across the DSI/FF system (U.S.

EPA, 1991b; U.S. EPA, 1992c; Lerner, 1992). Both DSI/FF systems operate at a temperature across the FF of 160°C (325°F).

The following sections present a cost analysis of different mercury control options for MWIs. Section B.2.2 provides a qualitative cost discussion on material separation costs and sections B.2.3 and B.2.4 provide more quantitative cost information on applying activated carbon injection to both MWI model plants and a polishing WS system to the large MWI model plant.

A sensitivity analysis presenting the effect that lower mercury levels in the flue gas (from materials separation or other means) would have on these costs is included.

B.2.2 Materials Separation

There is no report of costs of materials separation programs at medical facilities. The cost effectiveness of a battery separation program at a hospital may be better than in the general population if the hospital staff is well-trained and motivated. In general, the high-mercury-content instrument batteries used in hospitals are handled by a limited number of staff. Therefore, the administrative and educational costs associated with a separation program should be minimal. The collection and handling costs of separating a relatively small number of high-concentration mercury batteries would also be small.

B.2.3 Activated Carbon Injection

Table B-7 presents estimated costs for using activated carbon injection on the 60-kg/hr (130-lb/hr) and 455-kg/hr (1,000-lb/hr) MWIs. The carbon injection is assumed to achieve 85 percent reduction with a carbon feed rate of 676 mg/dscm (0.30 gr/dscf). This results in an outlet mercury concentration of 450 µg/dscm (197 gr/million dscf).

The cost components presented in Table B-7 are the same as those discussed in Section B.1.3. The total capital costs are estimated to be \$7,660 for the smaller batch MWI and \$25,700 for the larger commercial MWI (U.S. EPA, 1991d). The overall estimated costs for an activated carbon injection system are \$48/Mg (\$43/ton) of medical waste for the batch MWI and \$11.3/Mg (\$10.3/ton) of medical waste for the commercial MWI. This produces a cost effectiveness of \$2,110/kg (\$955/lb) of mercury removed and \$503/kg (\$228/lb) of mercury removed for the batch and commercial MWIs, respectively.

Table B-8 provides a sensitivity analysis showing the impact of mercury input level and capacity factor on the costs expressed in dollars per megagram of medical waste and dollars per kilogram of mercury removed. For the purpose of the sensitivity analysis, the mercury input level was assumed to be reduced by 50 percent through materials separation or other means while the carbon feed rate remained at 676 mg/dscm (0.30 gr/dscf). Under these assumptions, the outlet mercury level is 225 µg/dscm (98 gr/million dscf). The total annualized cost remains the same, since the capital and operating costs do not change. The cost per kilogram of mercury removed, however, increases substantially because less mercury is removed annually. For the 60-kg/hr (130-lb/hr) MWI, the cost increases to \$4,220/kg (\$1,910/lb) of mercury removed. For the 455-kg/hr (1,000-lb/hr) MWI, the cost increases to \$1,000/kg (\$452/lb) of mercury removed. If the capacity factor for the larger MWI is decreased to 20 percent (the same level as assumed for the small MWI), the cost increases to \$1,695/kg (\$767/lb).

Table B-7
Cost Estimate for Activated Carbon Injection
on Medical Waste Incinerators^a

Parameter	Small Model Plant	Large Model Plant
Unit Size (kg/hr)	60	455
Number of Units	1	1
Plant Size (kg/hr)	60	455
Flue Gas Flow (dscm/hr @ 7% O ₂)	537	4,033
Air Pollution Control Device	DSI/FF	DSI/FF
Hg Level Before Carbon (µg/dscm @ 7% O ₂)	3,000	3,000
Hg Level After Carbon (µg/dscm @ 7% O ₂)	450	450
Capacity Factor (%)	20	90
CAPITAL COST (\$)		
Purchased Equipment (PE) ^b	4,800	16,100
Installation ^c	0	0
Indirect ^d	1,580	5,310
Contingency ^e	1,280	4,280
Total Capital Cost (TCC)	7,660	25,700
OPERATING AND MAINTENANCE (O&M) (\$/yr)		
Operating Labor ^f	780	3,290
Supervision ^g	117	495
Maintenance Labor ^h	858	3,610
Maintenance Materials ⁱ	240	805
Power ^j	0	0
Carbon ^k	700	23,644
Disposal ^l	0	0
Overhead ^m	1,200	4,920
Taxes, Insurance, Administration ⁿ	305	1,030
Capital Recovery ^o	840	2,820
Total (\$/yr) ^p	5,040	40,600
\$/Mg Medical Waste	48	11.3
\$/ton Medical Waste	43	10.3
Mercury Reduction (kg/yr)	2.1	80
Mercury Reduction (lb/yr)	4.5	178
\$/kg Mercury	2,110	503
\$/lb Mercury	955	228

^a Cost numbers may not add exactly due to round-off.

^b Based on \$200,000 for an 730-Mg/day MWC. Scaled based on 0.6 rule.

^c Included in PE costs.

^d 33 percent of PE costs.

^e 20 percent of PE + indirect costs.

^f 0.25 hr/shift @ \$12/hr (per unit).

^g 15 percent of operating labor costs.

^h 0.25 hr/shift @ 10 percent wage rate premium over labor wage (per unit).

ⁱ 5 percent of PE costs.

^j Power costs associated with the active carbon injection system are assumed to be negligible.

^k Based on \$1.10/kg of carbon and a carbon feed rate of 676 mg/dscm.

^l Increase in disposal costs due to carbon injection are assumed to be negligible.

^m 60 percent of labor and maintenance costs.

ⁿ 4 percent of TCC.

^o Using a capital recovery factor of 0.1098 (7 percent interest rate for 15 years).

^p Total costs equal capital recovery costs plus O&M costs.

Table B-8
Sensitivity Analysis for an
Activated Carbon Injection System on MWIs

Operating Assumptions	60-kg/hr MWI		455-kg/hr MWI	
	\$/Mg MW ^a	\$/kg Hg	\$/Mg MW	\$/kg Hg
Base Case (Table A-7)	48	2,110	11.3	503
50% Lower Hg Input ^b	48	4,220	11.3	1,000
20% Capacity Factor ^c	--	--	38.5	1,695

^a MW = medical waste on this table only.

^b Inlet mercury level (upstream of APCD) is reduced to 1,500 µg/dscm. There is no change in capital or operating costs. The amount of mercury removed is the only factor affected. The outlet mercury level is 225 µg/dscm.

^c Base case capacity factor for 60-kg/hr MWI already 20 percent.

B.2.4 Wet Scrubbing

Table B-9 presents estimated costs for using a polishing WS system on the 455-kg/hr (1,000 lb/hr) MWI. The application of a WS system to the batch MWI was initially considered. Based on the small size of the MWI and the complexity of wet scrubbers, however, it was determined that this control option is not feasible. The WS system is assumed to achieve 85 percent reduction, resulting in an outlet mercury level of 450 µg/dscm (196 gr/million dscf).

The capital cost components presented in Table B-9 are the same as those discussed in section B.1.4. The PE costs are \$114,000 and the total capital costs are estimated to be \$182,000 (Lerner, 1993b).

The overall estimated costs for the WS system are \$13.2/Mg (\$12.0/ton) of medical waste. The cost effectiveness for the system is \$682/kg (\$310/lb) of mercury removed.

Table B-10 provides a sensitivity analysis showing the impact of mercury input level and capacity factor on the costs expressed in dollars per megagram of medical waste and dollars per kilogram of mercury removed. If the mercury input is reduced by 50 percent through materials separation or other means, the outlet mercury level would be 225 µg/dscm (98 gr/million dscf). The total annualized cost remains the same, since the capital and operating costs do not change. The cost per kilogram of mercury removed, however, increases substantially because of the decreased amount of mercury removed annually by the WS system. For the 455-kg/hr (1,000-lb/hr) commercial MWI, the cost increases to \$1,370/kg (\$621/lb) of mercury removed. If the capacity factor is decreased to 20 percent, the cost increases to \$3,067/kg (\$1,394/lb).

Table B-9
Cost Estimate for a Polishing Wet Scrubbing System
on Medical Waste Incinerators^a

Parameter	Commercial MWI
Unit Size (kg/hr)	455
Number of Units	1
Plant Size (kg/hr)	455
Flue Gas Flow (dscm/hr @ 7% O ₂)	3,460
Air Pollution Control Device	DSI/FF
Inlet Hg Level (µg/dscm @ 7% O ₂)	3,000
Outlet Hg Level (µg/dscm @ 7% O ₂)	450
Capacity Factor (%)	90
CAPITAL COST (\$)	
Purchased Equipment (PE) ^b	114,000
Installation ^c	0
Indirect ^d	37,700
Contingency ^e	30,400
Total Capital Cost (TCC)	182,000
OPERATING AND MAINTENANCE (O&M) (\$/yr)	
Operating Labor ^f	3,290
Supervision ^g	495
Maintenance Labor ^h	3,610
Maintenance Materials ⁱ	2,300
O&M Expenses ^j	4,600
Water/Wastewater Treatment ^k	115
Overhead ^l	5,800
Taxes, Insurance, Administration ^m	7,290
Capital Recovery ⁿ	20,000
Total (\$/yr) ^o	47,400
\$/Mg Medical Waste	13.2
\$/ton Medical Waste	12.0
Mercury Reduction (kg/yr)	70
Mercury Reduction (lb/yr)	153
\$/kg Mercury	682
\$/lb Mercury	310

^a Cost numbers may not add exactly due to round-off.

^b Based on cost estimates from Belco Engineering.

^c Included in PE costs.

^d 33 percent of PE costs.

^e 20 percent of PE + indirect costs.

^f 0.25 hr/shift @ \$12/hr (per unit).

^g 15 percent of operating labor costs.

^h 0.25 hr/shift @ 10 percent wage rate premium over labor wage (per unit).

ⁱ 2 percent of PE costs.

^j 4 percent of PE costs.

^k Using 2.3 liters/min @ \$0.11/1,000 liters.

^l 60 percent of labor and maintenance costs.

^m 4 percent of TCC.

ⁿ Using a capital recover factor 0.1098 (7 percent interest rate for 15 years).

^o Total costs equal capital recovery costs plus O&M costs.

Table B-10
Sensitivity Analysis for a Wet Scrubbing System on MWIs

Operating Assumptions	Commercial (445-kg/hr) MWI	
	\$/Mg MW ^a	\$/kg Hg
Base Case (Table B-9)	13.2	682
50% Lower Hg Input ^b	13.2	1,370
20% Capacity Factor	59.5	3,067

^a MW = medical waste on this table only.

^b Inlet mercury level (upstream of APCD) is reduced to 1,500 µg/dscm. There is no change in capital or operating costs. The amount of mercury removed is the only factor affected. The outlet mercury level is 225 µg/dscm.

B.3 Utility Boilers

B.3.1 Model Plant Description

Three model coal-fired utility plants were developed to represent different methods of controlling mercury emissions. All three operate with a capacity factor of 65 percent; that is 5,694 hr/yr. Fuel characteristics include chloride levels assumed to be sufficiently high that all the mercury in the flue gas is in the form of HgCl₂. (It should be noted that the assumption of mercury in the form of 100 percent HgCl₂ in the utility flue gas was made because the highest, or worst case, activated carbon injection rates were projected for the case of 100 percent HgCl₂ [Heath and Turner, 1994]). The inlet mercury level to the control systems associated with each coal-fired model plant is 10 µg/dscm (4.4 gr/million dscf) at 20°C (68°F).

Model plant 1 is a 975-megawatt (MW) coal-fired boiler firing low-sulfur coal with a chloride content of 0.1 percent. The model plant has a flue gas volume of 4,050,000 dscm/hr and is equipped with a cold-side ESP. The temperature ahead of the ESP is 157°C (314°F) and the temperature exiting the ESP is 150°C (300°F). No mercury control across the ESP is assumed.

Model plant 2 is a 975-MW coal-fired boiler firing high-sulfur coal (chloride content of 0.1 percent) and is equipped with a cold-side ESP (150°C [300°F]) and an FGD system. The FGD system is assumed to be 50 percent efficient for mercury control and the controlled mercury level is 5 µg/dscm (2.2 gr/million dscf) (Noblett et al., 1993).

Model plant 3 is identical to Model plant 1, except that it has a capacity of 100 MW (Noblett et al., 1993). The model plant has a flue gas volume of 411,000 dscm/hr. The gas temperature ahead of the ESP is 146°C (295°F) and the ESP outlet temperature is 137°C (280°F). Again, no mercury control across the ESP is assumed.

Seven control variations of the model plants were analyzed; five of the seven involved controlling emissions using activated carbon injection. Mercury control of the units is accomplished by one of the following methods:

- Direct injection of activated carbon ahead of the existing PM control device;

- Spray cooling of the flue gas after the existing PM control device (or after the air preheater for oil-fired boilers with no PM control device), followed by activated carbon injection and an FF to collect the mercury-laden carbon;
- Spray cooling of the flue gas after the air preheater, followed by activated carbon injection before the existing PM control device; or
- A carbon filter bed.

Table B-11 summarizes the model utility boilers and mercury controls used in the cost analysis.

Table B-11
Model Utility Boilers and Mercury Controls Used in Cost Analysis

Model	Size (MW)	Fuel	Pre-Existing Controls	Mercury Control
1a	975	Low-sulfur Coal	ESP	AC injection ^a
1b	975	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter
1c	975	Low-sulfur Coal	ESP	Spray cooler, AC injection
1d	975	Low-sulfur Coal	ESP	Carbon filter bed
2	975	High Sulfur Coal	ESP/FGD	Carbon filter bed
3a	100	Low-sulfur Coal	ESP	AC injection
3b	100	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter

^a AC = activated carbon.

The coal-fired boiler flue gas is at 150°C (302°F) and 5 percent moisture at the ESP outlet. It was assumed that the carbon injection would remove 90 percent of the mercury from coal-fired flue gas. Carbon with adsorbed mercury is assumed to be a hazardous waste if collected in a dedicated FF. When collected with fly ash in an existing ESP, the concentration of mercury is assumed to be too small to require the ash-carbon mixture to be classified as hazardous. Waste disposal costs are included in the cost estimates.

Simplified diagrams of the spray cooling and activated carbon injection systems are shown in Figures B-1 and B-2. As discussed in Section 2.2.7, each of these methods may not be applicable under certain conditions. Depending on the characteristics of the activated carbon, it may not be collected effectively in an existing ESP. High activated carbon injection rates were projected for the models that did not use spray cooling. The highest activated carbon injection rate may add a concentration of about 345 mg/actual cubic meter (acm) (0.15 gr/actual cubic feet [acf]), or about 5 percent of the typical fly ash loading in coal-fired utility flue gas. The increased carbon content of the fly ash may adversely affect ESP performance. Spray cooling the flue gas may cause corrosion and ash handling problems if the dew point is reached or if the spray is not completely evaporated. Alternately, spray cooling the particulate-laden flue gas may improve PM collection efficiency (especially if switching to low-sulfur coal) and the addition of small amounts of carbon may also improve collection efficiency. Whether the activated carbon has a positive or

Figure B-1
Spray Cooling System

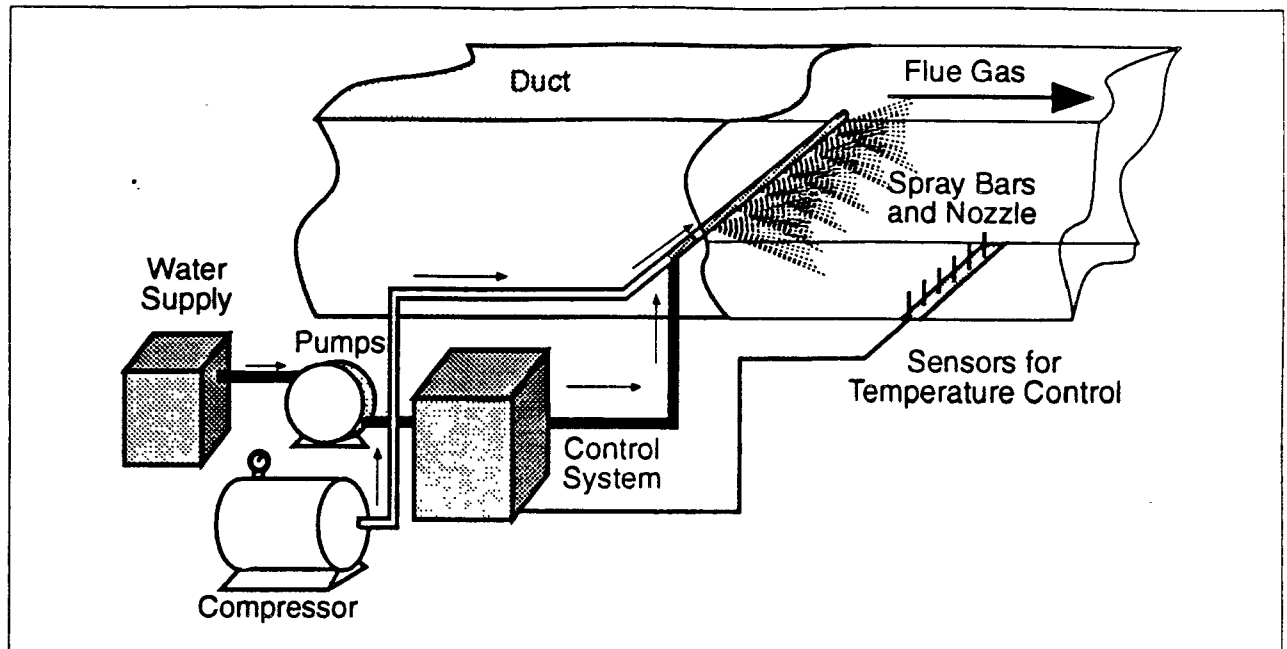
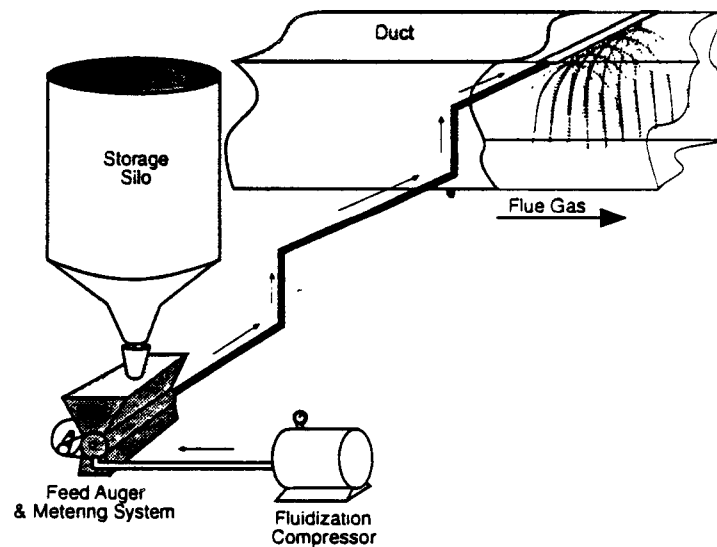


Figure B-2
Carbon Injection System



negative effect on ESP performance depends on site-specific operating and flue gas conditions. For model plants where a FF is added to collect mercury-laden carbon, the design is assumed to be appropriate for handling flue gas at 93°C (199°F) without "blinding" the FF bags.

The following sections present the cost analyses of different mercury control options for the model plant boilers. Cost information for the model plants using activated carbon injection alone, and in combination with spray cooling and additional PM control, is presented in section B.3.2. It should be stressed that the effectiveness of carbon injection in controlling mercury emissions from MWCs has been proven, but the technology is undergoing testing for application to utility boilers at only pilot-scale facilities. Section B.3.3 provides cost information on applying a carbon filter bed to the model plants.

B.3.2 Activated Carbon Injection

Activated carbon injection rates for model utility boilers used in costing carbon injection are shown in Table B-12. The table shows each model, its size, concentration of mercury in the flue-gas, temperature at the point of carbon injection and the mass ratio of carbon to mercury in the entering flue gas.

Table B-12
Carbon Injection Rates for Utility Boiler Models with Mercury Control

Model	Size (MW)	Flue Gas Hg Concentration at Point of Injection (µg/dscm)	Flue Gas Temperature at Point of Injection (°C)	Carbon Usage (g carbon/g Hg)
1a	975	10	157	34,200
1b	975	10	93	460
1c	975	10	93	460
3a	100	10	146	17,200
3b	100	10	93	460

Carbon to mercury ratios were selected by using two equations obtained from Calgon Corporation and described by Heath and Turner (1994). The equations depend on temperature as the only variable not attributable to adsorbate characteristics when used for a specific activated carbon. For this work, the carbon was assumed to be Calgon PCB 6x6.

As indicated in the table, only three flue-gas temperatures were used for the various models. For those with waste spray cooling, 93°C (199°F) was assumed to be the temperature at the point of carbon injection. For models not using water spray cooling, the temperatures were 157°C (315°F) and 146°C (295°F) for the 975 MW and 100 MW plants, respectively.

The carbon injection rates used for the cost models were based on information available at the time this Report was prepared. The rates seemed comparable (or conservative) compared to early EPRI work (Chang et al. 1993) for the low temperature injection, but were less certain for the higher temperatures. More current information may suggest using different injection rates.

Tables B-13 through B-16 present the costs of applying activated carbon injection to the model plants. The cost components presented in these tables are similar to those presented in section 2.1.3 with some exceptions. Purchased equipment costs for spray coolers and activated carbon injection systems are based on vendor contacts (Durham and Ebner, 1993; Hoagland, 1993) and FF costs are estimated from the EPA's OAQPS Control Cost Manual (U.S. EPA, 1992d). Costs from the OAQPS manual are based on average costs for baghouses in many industries and may not reflect the conservative nature of utility designs. This factor, and difficult retrofit installations, may increase baghouse costs by 50 percent or more. (No credit is taken for SO₂ or other pollutants such as dioxin that might be captured by the carbon nor were costs added for additional ductwork.) Cost factors in the EPA's OAQPS Control Cost Manual are also used to estimate installation costs and indirect costs. Contingency costs are included in the indirect costs. Labor usage for activated carbon injection systems is proportionally higher for models without spray cooling than for models with cooling because of significantly larger amounts of carbon that must be handled. Maintenance labor is calculated using a labor charge rate of \$13.20/hr, and maintenance materials are assumed to be equal to maintenance labor costs. The CRF is based on a 7 percent interest rate annualized over 20 years (CRF = 0.09439). The activated carbon injection system is assumed to reduce the mercury in the flue gas to less than 1 µg/dscm (0.44 gr/million dscf) equal to 90 percent control for coal-fired boilers and 50 percent control for oil-fired boilers.

Table B-13 presents the costs for applying activated carbon injection to the 975- and 100-MW ESP-equipped boilers firing low-sulfur coal (models 1a and 3a). The carbon injection system is installed in existing ductwork ahead of the ESP. This arrangement requires high carbon usage (1,385 kg/hr [3,050 lb/hr] for the 975-MW boiler and 70.9 kg/hr [156 lb/hr] for the 100-MW boiler) because of the elevated gas temperature at the injection location. A size increase from 100 to 975 MW increases the total capital cost for the control arrangement by a factor of 7.4 and increases the annualized cost by a factor of 6.1; the cost effectiveness value decreases by 38 percent and the annualized mills/kWh cost impact decreases by 32 percent. The total capital cost for the activated carbon injection system is \$1,260,000 for the 975-MW boiler and \$167,100 for the 100-MW boiler. The total annualized cost is \$10,110,000/yr for the 975-MW boiler and \$640,000/yr for the 100-MW boiler.

Tables B-14 and B-15 present the costs for the 975- and 100-MW ESP-equipped low-sulfur coal-fired boilers using activated carbon injection in conjunction with spray cooling and additional PM control using a reverse-air FF (models 1b and 3b). This arrangement requires accurate temperature control to prevent problems with excess acidic moisture condensation on ductwork and ash agglomeration, which leads to higher maintenance and operating costs. This combination is also expected to enhance the control of mercury. The spray cooling, which is upstream of the carbon injection and after the existing ESP, serves to reduce the temperature of the flue gas where the carbon is injected to 93°C (199°F). The FF is located after the injection system and before the stack. The FF enhances the capture of the injected carbon. As a result, the carbon injection rate is reduced to approximately 1 percent of the rate assumed in Table B-13. The activated carbon injection rate is 1.9 kg/hr (4.2 lb/hr) for the 100-MW coal-fired boiler (model 3b) and 18.8 kg/hr (41 lb/hr) for the 975-MW coal-fired boiler (model 1b). Greater than 80 percent of the total capital cost for this control arrangement is contributed by the FF and less than 3 percent is contributed by the activated carbon injection system. A size increase from 100- to 975-MW increases the total capital cost for the control arrangement by 86 percent and increases the annualized cost by 83 percent; the cost effectiveness value decreases by 37 percent and the annualized mills/per kilowatt-hour cost impact decreases by 30 percent.

Table B-13
Cost Estimates for Applying a Carbon Injection System on
975- and 100-MW Coal-Fired Utility Boilers

Parameter	Model 1a	Model 3a
Unit Size (MW)	975	100
Flue Gas Flow (dscm/hr)	4,050,000	411,000
Air Pollution Control Device	ESP ^a	ESP ^b
Hg Level Before Injector (µg/dscm, 68°F)	10	10
Hg Level After Injector (µg/dscm, 68°F)	1	1
Capacity Factor (%)	65	65
CAPITAL COST (\$)		
Purchased Equipment (PE)	869,830	115,226
Installation	130,474	17,284
Indirect	260,949	34,568
Total Capital Cost (TCC)	1,260,000	167,100
OPERATING AND MAINTENANCE (O&M) (\$/yr)		
Operating Labor ^c	103,680	38,880
Supervision Labor ^d	15,552	5,832
Maintenance Labor ^e	57,024	19,008
Maintenance Materials ^f	57,024	19,008
Carbon ^g	9,556,509	488,980
Power ^h	13,675	700
Disposal	316,068 ⁱ	16,173 ^j
Overhead ^k	139,968	49,637
Taxes, Insurance, Administration ^l	50,450	6,683
Capital Recovery ^m	119,050	15,770
Total (\$/yr) ⁿ	10,110,000	660,000
mills/kWh	1.82	1.16
Mercury Reduction (kg/yr)	208	21
Mercury Reduction (lb/yr)	457	46
\$/kg Mercury	48,700	31,000
\$/lb Mercury	22,100	14,200

^a ESP outlet temperature of 150°C, moisture content of 5 percent.

^b ESP outlet temperature of 137°C, moisture content of 11 percent.

^c \$12/hr.

^d 15 percent of operating labor costs.

^e \$13.20/hr.

^f Same as maintenance labor.

^g Based on \$1.10/kg of carbon, plus 10 percent for shipping.

^h Based on 46 mills/kWh.

ⁱ Based on a nonhazardous waste disposal rate of \$33/Mg.

^j Based on a nonhazardous waste disposal rate of \$36.36/Mg.

^k 60 percent of labor and maintenance costs.

^l 4 percent of TCC.

^m Using a capital recovery factor of 0.09439 (7 percent interest for 20 years).

ⁿ Total costs equal capital recovery costs plus O&M costs.

Table B-14
Cost Estimates for Applying a Carbon Injection System in
Conjunction with a Spray Cooler and a Fabric Filter on a
975-MW Coal-Fired Utility Boiler (Model 1b)

Parameter	Model Plant			
Unit Size (MW)	975			
Flue Gas Flow (dscm/hr)	4,050,000			
Air Pollution Control Device	ESP ^a			
Hg Level Before Injector (µg/dscm, 68°F)	10			
Hg Level After Filter (µg/dscm, 68°F)	1			
Capacity Factor (%)	65			
CAPITAL COST (\$)	<u>Cooling</u>	<u>Injection</u>	<u>Fabric Filter</u>	<u>Total</u>
Purchased Equipment (PE)	2,993,796	109,448	12,978,750	16,081,994
Installation	1,017,891	16,417	9,344,700	10,379,008
Indirect	1,347,208	32,835	5,840,438	7,220,480
Total Capital Cost (TCC)	5,358,894	158,700	28,163,888	33,700,000
OPERATING AND MAINTENANCE (O&M) (\$/yr)				
Operating Labor ^b	25,920	25,920	155,520	207,360
Supervision Labor ^c	3,888	3,888	23,328	31,104
Operating Materials	219,572	---	302,102	521,674
Maintenance Labor ^d	19,008	14,256	85,536	118,800
Maintenance Materials ^e	19,008	14,256	85,536	118,800
Carbon ^f	---	129,628	---	129,628
Power ^g	958,363	185	1,088,412	2,046,960
Disposal ^h	---	---	50,503	50,503
Overhead ⁱ	40,694	34,992	209,952	285,638
Taxes, Insurance, Administration ^j	214,356	6,348	1,126,556	1,347,259
Capital Recovery ^k	505,826	14,980	2,561,802	3,082,607
Total (\$/yr) ^l	2,006,635	244,453	5,689,245 ^m	7,940,000
mills/kWh	0.36	0.04	1.02	1.43
Mercury Reduction (kg/yr)	---	---	---	208
Mercury Reduction (lb/yr)	---	---	---	457
\$/kg Mercury	9,668	1,178	27,412	38,300
\$/lb Mercury	4,393	535	12,454	17,400

^a ESP outlet temperature of 150°C, moisture content of 5 percent.

^b \$12/hr.

^c 15 percent of operating labor costs.

^d \$13.20/hr.

^e Same as maintenance labor.

^f Based on \$1.10/kg of carbon, plus 10 percent for shipping.

^g Based on 46 mills/kWh.

^h Based on a hazardous waste disposal rate of \$220/Mg.

ⁱ 60 percent of labor and maintenance costs.

^j 4 percent of TCC.

^k Using a capital recovery factor of 0.09439 (7 percent interest for 20 years).

^l Total costs equal capital recovery costs plus O&M costs.

^m See text for factors that may increase fabric filter costs.

Table B-15
Cost Estimates for Applying a Carbon Injection System in
Conjunction with a Spray Cooler and a Fabric Filter on a
100-MW Coal-Fired Utility Boiler (Model 3b)

Parameter	Model Plant			
Unit Size (MW)	100			
Flue Gas Flow (dscm/hr)	411,000			
Air Pollution Control Device	ESP ^a			
Hg Level Before Injector (µg/dscm, 68°F)	10			
Hg Level After Filter (µg/dscm, 68°F)	1			
Capacity Factor (%)	65			
CAPITAL COST (\$)	<u>Cooling</u>	<u>Injection</u>	<u>Fabric Filter</u>	<u>Total</u>
Purchased Equipment (PE)	258,627	109,448	1,813,479	2,181,554
Installation	87,933	16,417	1,305,705	1,410,055
Indirect	116,382	32,835	816,066	965,282
Total Capital Cost (TCC)	462,941	158,700	3,935,249	4,560,000
OPERATING AND MAINTENANCE (O&M) (\$/yr)				
Operating Labor ^b	17,280	25,920	51,840	95,040
Supervision Labor ^c	2,592	3,888	7,776	14,256
Operating Materials	18,968	---	60,816	79,785
Maintenance Labor ^d	9,504	14,256	42,768	66,528
Maintenance Materials ^e	9,504	14,256	42,768	66,528
Carbon ^f	---	13,152	---	13,152
Power ^g	82,791	188	116,956	199,935
Disposal ^h	---	---	4,660	4,660
Overhead ⁱ	23,328	34,992	87,091	145,411
Taxes, Insurance, Administration ^j	18,518	6,348	157,410	182,276
Capital Recovery ^k	43,697	14,980	361,069	419,746
Total (\$/yr) ^l	226,182	127,980	933,155 ^m	1,290,000
mills/kWh	0.22	0.25	1.64	2.09
Mercury Reduction (kg/yr)	--	--	--	21
Mercury Reduction (lb/yr)	--	--	--	41
\$/kg Mercury	10,739	6,076	44,305	61,000
\$/lb Mercury	4,879	2,761	20.129	27,700

^a ESP outlet temperature of 137°C, moisture content of 11 percent.

^b \$12/hr.

^c 15 percent of operating labor costs.

^d \$13.20/hr.

^e Same as maintenance labor.

^f Based on \$1.10/kg of carbon, plus 10 percent for shipping.

^g Based on 46 mills/kWh.

^h Based on a hazardous waste disposal rate of \$220/Mg.

ⁱ 60 percent of labor and maintenance costs.

^j 4 percent of TCC.

^k Using a capital recovery factor of 0.09439 (7 percent interest for 20 years).

^l Total costs equal capital recovery costs plus O&M costs.

^m See text for factors that may increase fabric filter costs.

Table B-16
Cost Estimates for Applying a Carbon Injection System in
Conjunction with a Spray Cooler on a
975-MW Coal-Fired Utility Boiler (Model 1c)

Parameter	Model Plant		
Unit Size (MW)	975		
Flue Gas Flow (dscm/hr)	4,050,000		
Air Pollution Control Device	ESP ^a		
Hg Level Before Injector (µg/dscm, 68°F)	10		
Hg Level After Injection (µg/dscm, 68°F)	1		
Capacity Factor (%)	65		
CAPITAL COST (\$)	<u>Cooling</u>	<u>Injection</u>	<u>Total</u>
Purchased Equipment (PE)	2,993,796	109,448	3,103,244
Installation	1,017,891	16,417	1,034,308
Indirect	1,347,208	32,835	1,380,043
Total Capital Cost (TCC)	5,358,894	158,700	5,520,000
OPERATING AND MAINTENANCE (O&M) (\$/yr)			
Operating Labor ^b	25,920	25,920	51,840
Supervision Labor ^c	3,888	3,888	7,776
Operating Materials	219,572	---	219,572
Maintenance Labor ^d	19,008	14,256	33,264
Maintenance Materials ^e	19,008	14,256	33,264
Carbon ^f	---	129,488	129,488
Power ^g	958,363	185	958,548
Disposal ^h	---	4,288	4,288
Overhead ⁱ	40,694	34,992	75,686
Taxes, Insurance, Administration ^j	214,356	6,348	220,704
Capital Recovery ^k	505,826	14,980	520,806
Total (\$/yr) ^l	2,006,635	248,601	2,260,000
mills/kWh	0.36	0.04	0.40
Mercury Reduction (kg/yr)	---	---	208
Mercury Reduction (lb/yr)	---	---	457
\$/kg Mercury	9,668	1,198	10,900
\$/lb Mercury	4,393	544	4,940

^a ESP outlet temperature of 150°C, moisture content of 5 percent.

^b \$12/hr.

^c 15 percent of operating labor.

^d \$13.20/hr.

^e Same as maintenance labor.

^f Based on \$1.10/kg of carbon, plus 10 percent for shipping.

^g Based on 46 mills/kWh.

^h Based on a hazardous waste disposal rate of \$33/Mg.

ⁱ 60 percent of labor and maintenance costs.

^j 4 percent of TCC.

^k Using a capital recovery factor of 0.09439 (7 percent interest for 20 years).

^l Total costs equal capital recovery costs plus O&M costs.

Table B-14 shows that the total capital cost for a 975-MW boiler controlled with a spray cooler, activated carbon injection system and FF is \$33,700,000, the annualized cost is \$7,940,000/yr and the cost effectiveness is \$38,300/kg (\$17,400/lb) of mercury. Table B-15 shows that the total capital cost for a 100-MW boiler controlled with a spray cooler, activated carbon injection system and FF is \$4,560,000, the annualized cost is \$1,300,000/yr and the cost effectiveness is \$61,700/kg (\$28,000/lb) of mercury.

Table B-16 presents the cost for the 975-MW ESP-equipped coal-fired boiler using activated carbon injection in combination with spray cooling (model 1c). The activated carbon injection rate is 18.8 kg/hr (41 lb/hr). The temperature after the spray cooler, where the carbon is injected, is 93°C (200°F). The total capital cost for this control arrangement is \$5,520,000, with 98 percent of the cost contributed by the spray cooler and 2 percent contributed by the activated carbon injection system. The annualized cost is \$2,390,000, with 88 percent of the cost contributed by the spray cooler and 12 percent contributed by the activated carbon injection system.

Injection systems installed on units without cooling the flue gas stream have much lower capital investment costs than the more complex systems, but have increased annualized costs because of high activated carbon usage. The units that reduce temperature after the particulate control device to reduce carbon usage must spend more money for capital equipment, primarily for added PM control to capture the mercury-laden carbon. Their annualized costs are also increased because of operating costs for the cooling and PM collection systems.

Table B-17 presents the results of a sensitivity analysis for mercury species in the flue gas (elemental mercury vs mercuric chloride). As shown in the table, cost effectiveness as measured by additional cost of producing electricity (mills/kWh) does not change significantly as the HgCl_2 fraction of total mercury decreases. This lack of change occurs for two reasons: less carbon is used for lower amounts of HgCl_2 and equipment costs are relatively insensitive to small changes in particle concentration in the flue gas. However, cost effectiveness as measured by quantity removed from the flue gas stream (\$/lb of mercury), increases by a factor of about 3.3 in changing from 100 percent HgCl_2 to 30 percent HgCl_2 . This increase occurs because the costs of flue gas treatment decrease only slightly while the quantity of collectible mercury decreases by 70 percent.

Table B-17
Sensitivity Analysis for Carbon Injection on Coal-Fired Utility
Boilers with Different Ratios of Elemental to Oxidized Mercury

Oxidized Mercury in Flue Gas	Cost Effectiveness \$/lb Hg Removed	Cost Effectiveness mills/kWh
100% of Hg as HgCl_2	4,970-27,700	0.41-2.26
80% of Hg as HgCl_2	6,190-34,600	0.41-2.26
50% of Hg as HgCl_2	9,840-55,400	0.40-2.25
30% of Hg as HgCl_2	16,300-92,000	0.40-2.25

For the models operating at 200 °F, the small amounts of carbon injected may be insufficient for proper mixing with the gas stream. Table B-18 shows the results of a sensitivity analysis for carbon feed rates at the amount used for estimating cost effectiveness (3.5 mg carbon/m³ of flue gas) and a suggested value of 73 mg/m³. As shown in the table, increasing the carbon injection rate by a

factor of 20 increases the cost of operating the controls systems (mills/kWh) by about 400 to 120 percent depending on the model. Cost effectiveness values in \$/lb of Hg removed increase by similar percentages. The cost increases are caused by increased values for carbon usage and disposal. Carbon injection system costs for the 100 MW boiler have relatively modest increases because a minimum cost for the injection system is built into the model, i.e., the system will not cost less than about \$110,000 for any of the models.

Table B-18
Sensitivity Analysis for Carbon Injection
Feed Rates on Utility Boilers

Model	Feed Rate, mg carbon/actual m ³ flue gas	Cost Effectiveness, \$/lb of Hg Removed	Cost Effectiveness, mills/kWh
ob (975 MW)	3.5	17,300	1.43
	73	24,400	2.01
1c (975 MW)	3.5	4,970	0.41
	73	10,990	0.90
2b (100 MW)	3.5	27,700	2.26
	73	34,800	2.83

B.3.3 Carbon Filter Beds

Table B-19 presents the costs for applying the carbon filter bed to the two 975-MW boilers (models 1d and 2). As noted in Section 2.2.1.1, the size of the carbon filter bed is dependent on the flue gas flow rate. The flow rate (on a wet basis) for the FGD-equipped model plant (model plant 2) is slightly higher because of the higher moisture content of the flue gas from the FGD system. Therefore, a slightly larger carbon filter bed was assumed. The carbon filter bed is assumed to reduce the mercury in the flue gas to less than 1 µg/dscm (0.44 gr/million dscf) (Hartenstein, 1993a).

The total capital costs for the boilers are approximately \$40,000,000 for the ESP-equipped boiler and \$44,000,000 for the ESP/FGD-equipped boiler. The total annualized costs for the two plants are essentially equal, at about 3 mills/kWh. The cost per kilogram of mercury removed, however, is different. For the ESP-equipped boiler, the cost is approximately \$72,146/kg (\$32,679/lb) of mercury removed. For the ESP/FGD- equipped boiler, the cost is higher, at \$187,615/kg (\$37,769/lb) of mercury removed because of the lower mercury level. (Note that the cost estimates were derived from retrofit costs, which can easily be 50 percent higher than new installation costs.)

Table B-19
Cost Estimates for a Carbon Filter Bed on Utility Boilers^a

Parameter	Model 1d	Model 2
Unit Size (MW)	975	975
Number of Units	1	1
Plant Size (MW)	975	975
Air Pollution Control Device	ESP ^b	ESP + FGD ^c
Flue Gas Flow (Nm ³ /hr @ 11% O ₂ wet)	5,560,000	5,810,000
Flue Gas Flow (dscm/hr @ 7% O ₂ wet)	4,050,000	4,050,000
Hg Level Before Filter (µg/dscm @ 7% O ₂)	10	5
Hg Level After Filter (µg/dscm @ 7% O ₂)	1	1
Capacity Factor (%)	65	65
CAPITAL COST (\$)		
Purchased Equipment (PE) ^d	28,573,974	31,540,388
Installation ^e	0	0
Indirect ^f	8,000,713	8,831,309
Contingency ^g	2,857,397	3,154,039
Total Capital Cost (TCC)	39,432,084	43,525,735
OPERATING AND MAINTENANCE (O&M) (\$/yr)		
Operating Labor ^h	210,400	210,400
Supervision Labor ⁱ	31,520	31,520
Maintenance Labor ^j	28,920	28,920
Maintenance Materials ^k	394,321	435,257
Carbon ^l	5,245,319	6,543,801
Power ^m	1,670,000	1,670,000
Disposal ⁿ	1,047,804	1,396,879
Overhead ^o	399,097	423,658
Taxes, Insurance, Administration ^p	1,577,283	1,741,029
Capital Recovery ^q	4,329,643	4,779,126
Total (\$/yr) ^r	14,934,306	17,260,591
mills/kWh	2.7	3.1
Mercury Reduction (kg/yr)	207	92
Mercury Reduction (lb/yr)	457	457
\$/kg Mercury	72,146	187,615
\$/lb Mercury	32,679	37,769

^a Cost numbers may not add exactly due to round-off.

^b ESP outlet temperature of 150°C, moisture content of 5 percent.

^c FGD outlet temperature of 65°C, moisture content of 9 percent.

^d Based on order of magnitude estimates for carbon usage.

^e Included in PE costs.

^f 28 percent of PE costs.

^g 10 percent of PE costs.

^h 16 hr/shift @ \$12/hr.

ⁱ 15 percent of operating labor costs.

^j 2 hr/shift @ 10 percent wage rate premium over labor wage.

^k 1 percent of TCC.

^l Based on \$1.10/kg of carbon and 545 mg carbon/dscm.

^m Using a 305-mm (water) pressure drop and 46 mills/kWh.

ⁿ Based on a hazardous waste disposal rate of \$220/Mg.

^o 60 percent of labor and maintenance costs.

^p 4 percent of TCC.

^q Using a capital recovery factor of 0.1098 (7 percent interest for 15 years).

^r Total costs equal capital recovery costs plus O&M costs.

B.4 Chlor-Alkali Plants

B.4.1 Model Plant Description

One chlor-alkali model plant, which produces 273 Mg (300 tons) of chlorine per day, was used for the cost analysis. This model plant represents the mid-range size of chlor-alkali plants in operation (U.S. EPA, 1984). The model plant has individual flow rates from the hydrogen and end-box streams of 4,080 dscm/hr (144,000 dscf/hr) each at 21 percent O₂ (combined to equal 8,160 dscm/hr [288,000 dscf/hr]) (U.S. EPA, 1973).

Baseline control systems for both streams consist of a heat exchanger to cool the effluent gas, followed by a knockout drum to separate the condensed mercury from the hydrogen and end-box streams. A mercury level of 1,000 g/day (2.2 lb/day) after this baseline control is assumed for the purpose of the cost analysis. This is consistent with federally mandated mercury standards for the hydrogen and end-box streams at all chlor-alkali plants (U.S. EPA, 1984). The mercury control options considered were conversion to the membrane cell process and control of the hydrogen and end-box streams through the use of brine scrubbing and treated activated carbon adsorption. No additional controls were examined for the cell room, since housekeeping practices are in use at all of the plants (U.S. EPA, 1984). The different control options are discussed in the following sections.

B.4.2 Conversion to the Membrane Cell Process

Cost data on converting to the membrane cell process are limited. Estimates to convert a mercury cell plant to a membrane cell operation vary between \$110,000 and \$220,000/Mg (\$100,000 and \$200,000/ton) of chlorine produced per day (Curlin, 1992). Representatives of the European chlor-alkali industry cite higher costs on the order of \$285,000/Mg (\$260,000/ton) chlorine produced per day (Euro-Chlor, 1993). Using the mid-point of the first cost range, the capital cost of conversion for the 270-Mg/day (300-tpd) model plant would be on the order of \$45 million. This is an annualized cost of approximately \$5 million/yr. For the conversion, the cost of cells and the membranes accounts for 60 percent of the total investment (Curlin, 1992). Other costs include additional brine treatment systems (i.e., the mercury removal system and ion-exchanger), the caustic recycle and evaporation system and piping modifications to the cell room (Horvath, 1986).

The operating cost differential between mercury and membrane cell plants is due to power requirements. The energy consumption is estimated to be 3,500 to 4,000 kWh/Mg (3,200 to 3,600 kWh/ton) of chlorine for the mercury cell and 3,100 to 3,400 kWh/Mg (2,800 to 3,100 kWh/ton) of chlorine for the membrane cell (Curlin, 1992).

Information on other operating costs was not available. Therefore, annualized costs and cost effectiveness were calculated using only capital and electrical costs. If an estimated electricity savings of 400 kWh/Mg (440 kWh/ton) of chlorine and an energy cost of 46 mills/kWh are assumed, the energy savings is approximately \$1,630,000. This is subtracted from the annualized capital cost, leaving a net cost of \$3,310,000, which is roughly equivalent to \$43.5/Mg (\$39.6/ton) of chlorine produced. The cost effectiveness, assuming 1,000 g/day of mercury removed through conversion, is approximately \$10,100/kg (\$4,590/lb) of mercury removed. For plants with mercury controls already in place (e.g., effective cooling, carbon beds and scrubbing), the cost per kilogram of mercury removed would be much higher. Additionally, local energy costs are a determining factor when considering conversion from the mercury cell to the membrane cell process.

B.4.3 Hydrogen and End-Box Stream Controls

As noted, the two control options evaluated for these streams are depleted brine scrubbing and treated activated carbon. Costs for secondary cooling and mist elimination are included in the cost estimate, since these elements are necessary when employing both of the control options.

The capital and annual operating costs for the two control options are based on 1972 equipment cost estimates (U.S. EPA, 1973). The capital cost estimates were scaled to 1993 values using chemical engineering plant cost indices (Chemical Engineering, 1972; Chemical Engineering, 1993). As a result of scaling the costs over such a long time period, there is a high level of uncertainty associated with these costs.

Tables B-20 and B-21 present the capital and annualized costs for the depleted brine scrubbing system and the treated activated carbon adsorption system, respectively. The capital cost for the control equipment includes PE costs, installation, indirect costs and contingency (U.S. EPA, 1973). Installation costs equal PE costs, and indirect costs are 90 percent of PE costs. Contingency costs equal 30 percent of PE costs (U.S. EPA, 1973). Based on these additional capital requirements, the estimated total capital cost ranges from 320 to 380 percent of the equipment cost (U.S. EPA, 1973). Operating and maintenance costs are assumed to be 30 percent of the total capital cost (U.S. EPA, 1973). For the depleted brine scrubbing system, the total capital cost for the secondary cooling, mist elimination and scrubbing is approximately \$1,620,000 for the combined hydrogen and end-box streams. The total annualized cost, including O&M, is \$662,000. This equals into \$7.4/Mg (\$6.7/ton) of chlorine produced. Assuming a reduction in mercury to 90 µg/dscm (at 21 percent O₂) (U.S. EPA, 1973), the cost effectiveness is \$2,280/kg (\$1,040/lb) of mercury removed.

For the carbon adsorption system on both streams, including secondary cooling and mist elimination, the total capital cost is \$1,222,000. The annualized cost including O&M is \$500,000, which equals \$5.6/Mg (\$5.1/ton) of chlorine produced. Assuming an outlet mercury concentration of 10 µg/dscm (at 21 percent O₂), the cost effectiveness is \$1,690/kg (\$769/lb) of mercury removed.

More information is needed on the current level of controls on mercury cell plants and their respective emission levels to more accurately estimate the costs for these two technologies.

B.5 Primary Copper Smelters

B.5.1 Model Plant Description

The primary copper smelter model plant was selected from the model plants presented in the background information document for primary copper, zinc and lead smelters (U.S. EPA, 1974). It consists of a flash furnace and three Pierce-Smith converters. Off-gases from the furnace are combined with the off-gas from the converters to form the feed to a single-stage acid plant. The feed to the acid plant has a maximum flow rate of 124,000 scm/hr (4,370,000 scf/hr).³

³ At 10.25 percent O₂.

Table B-20
Cost Estimates for a Depleted Brine Scrubbing
on a Mercury Cell Chlor-Alkali Plant^{a,b}

Parameter	Model Plant		
Unit Size (Mg/day)	273		
Flue Gas Flue Hydrogen Stream (dscm/hr @ 21% O ₂)	4.080		
Flue Gas Flue End-Box Stream (dscm/hr @ 21% O ₂)	4.080		
Total Flue Gas Flow (dscm/hr @ 21% O ₂)	8.160		
Combined Hg Level Before Scrubbing (μg/dscm @ 21% O ₂)	4.600		
Combined Hg Level After Scrubbing (μg/dscm @ 21% O ₂) ^c	90		
Capacity Factor (%)	90		
CAPITAL COST (\$) ^d	<u>Hydrogen Stream</u>	<u>End-Box Stream</u>	<u>Combined Stream</u>
Secondary Cooler	117,000	106,000	---
Chiller	165,000	165,000	---
Mist Eliminator	133,000	120,000	---
Depleted Brine Scrubber with Alkaline Scrubber	425,000	386,000	---
Total Capital Costs (TCC)	839,000	777,000	1,620,000
COMBINED ANNUAL OPERATING COSTS (\$/yr)			
Operating Costs ^e	---	---	485,000
Capital Recovery ^f	---	---	177,000
Total (\$/yr) ^g	---	---	662,000
\$/Mg Chlorine	---	---	7.4
\$/ton Chlorine	---	---	6.7
Mercury Reduction (kg/yr)	---	---	296
Mercury Reduction (lb/yr)	---	---	639
\$/kg Mercury	---	---	2,280
\$/lb Mercury	---	---	1,040

^a Costs were scaled from 1972 dollars to 1993 dollars using chemical engineering plant cost indices.

^b Cost numbers may not add exactly due to round-off.

^c Includes a 90 percent reduction by the cooling and mist elimination system.

^d The fixed capital requirement for the control equipment includes purchase price (freight-on-board), installation and indirect and contingency costs.

^e 30 percent of TCC.

^f Using a capital recovery factor of 0.1098 (7 percent interest rate for 15 years).

^g Total costs equal capital recovery costs plus operating cost.

Table B-21
Cost Estimates for Treated Activated Carbon Adsorption
on a Mercury Cell Chlor-Alkali Plant^{a,b}

Parameter	Model Plant		
Unit Size (Mg/day)	273		
Flue Gas Flue Hydrogen Stream (dscm/hr @ 21% O ₂)	4,080		
Flue Gas Flue End-Box Stream (dscm/hr @ 21% O ₂)	4,080		
Total Flue Gas Flow (dscm/hr @ 21% O ₂)	8,160		
Combined Hg Level Before Carbon (µg/dscm @ 21% O ₂)	4,600		
Combined Hg Level After Carbon (µg/dscm @ 21% O ₂) ^c	10		
Capacity Factor (%)	90		
CAPITAL COST (\$)^d	Hydrogen Stream	End-Box Stream	Combined Stream
Secondary Cooler	117,000	106,000	---
Chiller	165,000	165,000	---
Mist Eliminator	133,000	120,000	---
Carbon Adsorption Bed	217,000	197,000	---
Total Capital Costs (TCC)	631,000	588,000	1,222,000
COMBINED ANNUAL OPERATING COSTS (\$/yr)			
Operating Costs ^e	---	---	366,000
Capital Recovery ^f	---	---	134,000
Total (\$/yr) ^g	---	---	500,000
\$/Mg Chlorine	---	---	5.6
\$/ton Chlorine	---	---	5.1
Mercury Reduction (kg/yr)	---	---	296
Mercury Reduction (lb/yr)	---	---	650
\$/kg Mercury	---	---	1,690
\$/lb Mercury	---	---	769

^a Costs were scaled from 1972 dollars to 1993 dollars using chemical engineering plant cost indices.

^b Cost numbers may not add exactly due to round-off.

^c Includes a 90 percent reduction by the cooling and mist elimination system.

^d The fixed capital requirement for the control equipment includes purchase price (freight-on-board), installation and indirect and contingency costs.

^e 30 percent of TCC.

^f Using a capital recovery factor of 0.1098 (7 percent interest rate for 15 years).

^g Total costs equal capital recovery cost plus operating cost.

The plant has a capacity to process 910 Mg/day (1,000 tpd) of ore consisting of 27 percent copper and 32 percent sulfur. This corresponds to a production capacity of 78,000 Mg/yr (86,000 tons/yr) of copper. A capacity factor of 90 percent is assumed for this plant. This corresponds to 236 Mg/day (260 tpd) copper production capacity. The cost of applying selenium filters on the stream entering the acid plant is evaluated below.

B.5.2 Selenium Filters

Table B-22 presents the cost estimate for applying three selenium filters to the flue gas stream upstream of the acid plant. Individual filters can handle flows up to 53,500 scm/hr (1,890,000 scf/hr) (Edlund, 1993b),⁴ and since the flow to the acid plant is 124,000 scm/hr (4,370,000 scf/hr), three selenium filters are needed with flows to each of 41,200 scm/hr. There are no data available on the mercury level upstream of the acid plant. For the purpose of this cost estimate, a mercury level of 1,000 µg/scm (436 gr/million scf) is assumed. Ninety percent reduction across the filter is assumed, so that the mercury outlet level is 100 µg/scm (44 gr/million scf). Vendor-quoted removal is typically between 95 and 99 percent, but in this case the 90 percent assumption was cited by the vendor since more detailed information on the mercury content of the stream was not available (Edlund, 1993c).

The PE cost (covering all three filters) is estimated at \$2,050,000 (Edlund, 1993b). This is a freight-on-board (FOB) cost and does not include installation and transportation. Information on the cost of installation was not available; it was assumed to be 30 percent of the PE cost (which includes transportation). The vendor indicated that erection of the filter is relatively simple (Edlund, 1993c). The vessel is delivered in one main piece (the hull), along with the top hatch and the baskets containing the selenium mass. A mobile crane places the baskets into the vessel and lifts the top hatch in place. The tie-in of the filter to existing duct work takes 1 to 2 days (Edlund, 1993c).

Indirect and contingency costs are based on the MWC cost procedures for SD/FF and DSI systems (U.S. EPA, 1989b). The total capital cost is estimated to be \$4,260,000.

According to the vendor, there are no labor costs associated with the filter. The filter may require periodic cleaning if the dust content is high. For cleaning, the baskets must be removed and the selenium mass is cleaned with water. Afterwards, the baskets are replaced in the vessel. It may be necessary to add filter mass (approximately 5 percent) after the cleaning (Edlund, 1993c). To account for periodic cleaning, 40 hours of maintenance labor are included. Costs for additional filter mass are not included, but costs for refilling the filter mass at the end of the filter life (approximately every 5 years in the model plant case) are included. Costs for refilling are 50 percent of the original filter cost (Edlund, 1993b). These costs have been annualized over the life of the filter, which is calculated assuming 50 kg (110 lb) of elemental mercury is captured per cubic meter (cubic foot) of the filter mass (Edlund, 1993b). The mercury content in the flue gas directly impacts filter life and, thereby, affects the economics of this technology.

Overhead and taxes, insurance and administrative costs are based on the MWC cost procedures (U.S. EPA, 1989b). A CRF of 0.1098 (7 percent over 15 years) was used to annualize the capital costs.

⁴ Oxygen level not specified.

Table B-22
Cost Estimates for Selenium Filters on a Copper Smelting Plant^a

Parameter	Model Plant
Production Capacity (Mg/day) ^b	236
Flue Gas Flow (scm/hr @ 10% O ₂)	124,000
Number of Filters	3
Flow Thru Each Filter (scm/hr)	41,200
Individual Filter Mass (m ³)	30
Hg Level Before Filter (μg/scm)	1,000
Hg Level After Filter (μg/scm)	100
Capacity Factor (%)	90
CAPITAL COST (\$)	
Purchased Equipment (PE) ^c	2,050,000
Installation ^d	616,000
Indirect ^e	881,000
Contingency ^f	710,000
Total Capital Cost (TCC)	4,260,000
OPERATING AND MAINTENANCE (O&M) COST (\$/yr)	
Operating Labor ^g	0
Maintenance Labor ^h	1,580
Maintenance Materials ⁱ	248,000
Power ^j	30,700
Disposal ^k	6,200
Overhead ^l	150,000
Taxes, Insurance, Administration ^m	170,000
Capital Recovery ⁿ	355,000
Total (\$/yr) ^o	961,000
\$/Mg Copper	12.4
\$/ton Copper	11.3
Mercury Reduction (kg/yr)	879
Mercury Reduction (lb/yr)	1,930
Filter Capability (kg of Hg)	4,450
Filter Capability (lb of Hg)	9,790
Filter Life (yr)	5.1
\$/kg Mercury	1,094
\$/lb Mercury	497

^a Cost numbers may not add exactly due to round-off.

^b Production capacity of copper.

^c Based on \$800,000 FOB cost for a 53,500 scm/hr filter scaled based on 0.6 rule; \$400,000 associated with initial filter mass.

^d 30 percent of PE costs.

^e 33 percent of PE and installation costs.

^f 20 percent of direct and indirect costs.

^g No labor required.

^h For yearly filter cleaning; 40 hours/yr @ \$13.2/hr.

ⁱ For replacing filter mass (annualized over life of filter).

^j Using 50-mm pressure drop (water) and 46 mills/kWh.

^k Assuming 1,600 kg/m³ of filter mass and disposal of \$220/Mg.

^l 60 percent of labor and maintenance costs.

^m 4 percent of TCC.

ⁿ Using a capital recovery factor of 0.1098 (7 percent interest for 15 years) applied to TCC excluding initial filter mass cost.

^o Total costs equal capital recovery cost plus O&M costs.

The annualized cost for the copper model plant is \$12.4/Mg (\$11.3/ton) of copper produced. With the assumed mercury concentrations, this cost equals \$1,094/kg (\$497/lb) of mercury removed.

Actual mercury emission data are necessary to provide a more realistic cost evaluation of this technology. Table B-23 presents a sensitivity analysis of the effect of mercury inlet level on the costs associated with the selenium filters. At a mercury inlet of 5,000 µg/scm (2,180 gr/million scf), the filter life decreases to 1 year. As a result, the annualized cost increases to \$30/Mg (\$27/ton) of copper produced. The cost effectiveness value, however, improves to \$529/kg (\$240/lb) of mercury removed because of the additional mercury collected annually. At an inlet of 10,000 µg/scm (4,360 gr/million scf), filter life decreases to 0.5 years, the annualized cost increases to \$52/Mg (\$47/ton) of copper and the cost effectiveness value improves to \$459/kg (\$209/lb) of mercury removed.

Table B-23
Sensitivity Analysis for Selenium Filters
on a Copper Smelting Plant

Operating Assumptions	\$/Mg Copper	\$/kg Hg
Base Case (Table B-20)	12.4	1,090
Inlet Hg Level ^a		
5,000 µg/dscm	29.9	529
10,000 µg/dscm	51.9	459
Installation Cost		
50% of Purchased Equipment	13.6	1,210
75% of Purchased Equipment	15.2	1,350

^a A mercury reduction of 90 percent is assumed for all cases.

Because of the lack of information on installation costs for selenium filters, the impact of the 30 percent of PE assumption was examined in a sensitivity analysis also. As shown in Table B-23, increasing the installation cost to 50 percent and 75 percent of PE costs has a relatively small impact on the annualized costs (increasing to \$13.6/Mg [\$12.4/ton] and \$15.2/Mg [\$13.8/ton] of copper, respectively). Cost effectiveness values increase to \$1,210/kg (\$550/pound) and \$1,350/kg (\$614/pound), respectively.

B.6 Primary Lead Smelters

B.6.1 Model Plant Description

The primary lead smelter model plant was selected from a U.S. EPA document on this industry (U.S. EPA, 1974). The model plant consists of a sintering machine, which feeds to a blast furnace and a dross furnace. The sintering machine is an updraft machine, which does not employ recirculation of the weak gas. Therefore, there are two off-gas streams, one strong SO₂ stream and one weak SO₂ stream. The strong stream is fed to a single-stage acid plant at a flow rate of

31,600 scm/hr (1,120,000 scf/hr);⁵ the weak stream, with a flow rate of 138,000 scm/hr (4,860,000 scf/hr),⁶ is sent to an FF for particulate control and emitted to the atmosphere. The off-gases from the dross furnace are fed to an FF, at a flow rate of 72,200 scm/hr (2,550,000 scf/hr)⁷ for PM control and emitted to the atmosphere. The lead production capacity for this facility is 90,900 Mg/yr (100,000 tons/yr) from concentrate containing 55 percent lead and 16 percent sulfur. The plant is assumed to operate 90 percent of the year. This corresponds to a daily capacity of 276 Mg/day (304 tpd).

Cost estimates are provided below for applying selenium filters on the sintering stream leading to the acid plant and on the blast and dross furnace flue gas stream (after the FF).

B.6.2 Selenium Filters

Tables B-24 and B-25 present the costs for applying selenium filters to the sintering flue gas stream upstream of the acid plant and the combined furnace flue gas stream after the FF, respectively. These streams are assumed to contain higher levels of mercury emissions than those from other plant processes because of the high temperatures in the sintering and furnace operations (U.S. EPA, 1994).

For the sintering stream (Table B-24), only one filter is required. There are no data on the mercury levels in this stream, so for the purpose of this estimate, 1,000 µg/scm (436 gr/million scf) is assumed. A 90 percent mercury reduction to 100 µg/scm (44 gr/million scf) is assumed (Edlund, 1993c).

The cost assumptions for applying the filter to this stream are the same as those discussed in Section B.5.2. The total capital costs are \$1,210,000 and the annualized costs are \$266,000/yr. This equals \$3.0/Mg (\$2.7/ton) of lead produced and a cost effectiveness of \$1,190/kg (\$541/lb) of mercury removed. As with the copper smelter, emissions data are needed to determine actual costs of applying this technology.

For the furnace flue gas stream following the FF (Table B-23), two filters are required. Because data were not available on mercury emissions, 1,000 µg/scm (435 gr/million scf)⁸ was assumed, with an outlet of 100 µg/scm (44 gr/million scf).

The total capital cost for applying the filters to this stream is \$2,620,000 and the annualized costs are 584,000/yr. This equals \$6.4/Mg (\$5.8/ton) of lead produced. Cost effectiveness is \$1,140/kg (\$520/lb) of mercury removed. Again, emissions data are needed to accurately reflect the costs of applying this control technology.

⁵ At 12 percent O₂.

⁶ Oxygen level not specified.

⁷ At 19 percent O₂.

⁸ At 19 percent O₂.

Table B-24
Cost Estimate for a Selenium Filter
on a Lead Smelting Plant (Sintering Stream)^a

Parameter	Model Plant
Production Capacity (Mg/day) ^b	276
Flue Gas Flow (scm/hr @ 10% O ₂)	31,500
Number of Filters	1
Flow Thru Each Filter (scm/hr)	31,500
Individual Filter Mass (m ³)	23
Hg Level Before Filter (µg/scm)	1,000
Hg Level After Filter (µg/scm)	100
Capacity Factor (%)	90
CAPITAL COST (\$)	
Purchased Equipment (PE) ^c	583,000
Installation ^d	175,000
Indirect ^e	250,000
Contingency ^f	201,000
Total Capital Cost (TCC)	1,210,000
OPERATING AND MAINTENANCE (O&M) COST (\$/yr)	
Operating Labor ^g	0
Maintenance Labor ^h	528
Maintenance Materials ⁱ	70,200
Power ^j	2,610
Disposal ^k	1,580
Overhead ^l	42,500
Taxes, Insurance, Administration ^m	48,300
Capital Recovery ⁿ	101,000
Total (\$/yr) ^o	266,000
\$/Mg Lead	3.0
\$/ton Lead	2.7
Mercury Reduction (kg/yr)	224
Mercury Reduction (lb/yr)	493
Filter Capability (kg of Hg)	1,130
Filter Capability (lb of Hg)	2,500
Filter Life (yr)	5.1
\$/kg Mercury	1,190
\$/lb Mercury	541

^a Cost numbers may not add exactly due to round-off.

^b Production capacity of copper.

^c Based on \$800,000 FOB cost for a 53,500 scm/hr filter scaled based on 0.6 rule; \$400,000 associated with initial filter mass.

^d 30 percent of PE costs.

^e 33 percent of PE and installation costs.

^f 20 percent of direct and indirect costs.

^g No labor required.

^h For yearly filter cleaning; 40 hr/yr @ \$13.2/hr.

ⁱ For replacing filter mass (annualized over life of filter).

^j Using 50-mm pressure drop (water) and 46 mulls/kWh.

^k Assuming 1,600 kg/m³ of filter mass and disposal of \$220/Mg.

^l 60 percent of labor and maintenance costs.

^m 4 percent of TCC.

ⁿ Using a capital recovery factor of 0.1098 (7 percent interest for 15 years) applied to TCC excluding initial filter mass cost.

^o Total costs equal capital recovery cost plus O&M cost.

Table B-25
Cost Estimate for a Selenium Filter
on a Lead Smelting Plant (Furnace Stream)^a

Parameter	Model Plant
Production Capacity (Mg/day) ^b	276
Flue Gas Flow (scm/hr @ 10% O ₂)	72,000
Number of Filters	2
Flow Thru Each Filter (scm/hr)	36,000
Individual Filter Mass (m ³)	26
Hg Level Before Filter (μg/scm)	1,000
Hg Level After Filter (μg/scm)	100
Capacity Factor (%)	90
CAPITAL COST (\$)	
Purchased Equipment (PE) ^c	1,260,000
Installation ^d	379,000
Indirect ^e	541,000
Contingency ^f	436,000
Total Capital Cost (TCC)	2,620,000
OPERATING AND MAINTENANCE (O&M) COST (\$/yr)	
Operating Labor ^g	0
Maintenance Labor ^h	1,060
Maintenance Materials ⁱ	152,000
Power ^j	11,900
Disposal ^k	3,600
Overhead ^l	92,000
Taxes, Insurance, Administration ^m	105,000
Capital Recovery ⁿ	218,000
Total (\$/yr) ^o	584,000
\$/Mg Lead	6.4
\$/ton Lead	5.8
Mercury Reduction (kg/yr)	512
Mercury Reduction (lb/yr)	1,130
Filter Capability (kg of Hg)	2,590
Filter Capability (lb of Hg)	5,700
Filter Life (yr)	5.1
\$/kg Mercury	1,140
\$/lb Mercury	520

^a Cost numbers may not add exactly due to round-off.

^b Production capacity of lead.

^c Based on \$800,000 FOB cost for a 53,500 scm/hr filter scaled based on 0.6 rule; \$400,000 associated with initial filter mass.

^d 30 percent of PE costs.

^e 33 percent of PE and installation costs.

^f 20 percent of direct and indirect costs.

^g No labor required.

^h For yearly filter cleaning; 40 hr/yr @ \$13.2/hr.

ⁱ For replacing filter mass (annualized over life of filter).

^j Using 50-mm pressure drop (water) and 46 mills/kWh.

^k Assuming 1,600 kg/m³ of filter mass and disposal of \$220/Mg.

^l 60 percent of labor and maintenance costs.

^m 4 percent of TCC.

ⁿ Using a capital recovery factor of 0.1098 (7 percent interest for 15 years) applied to TCC excluding initial filter mass cost.

^o Total costs equal capital recovery cost plus O&M costs.

Tables B-26 and B-27 present the sensitivity analyses examining the impact of mercury inlet levels on costs and the impact of installation cost assumptions on costs. As with the copper smelter sensitivity analysis, increased mercury levels increase the annualized costs but decrease the cost per kilogram of mercury removed. The effect of varying installation cost assumptions is also similar.

Table B-26
Sensitivity Analysis for Selenium Filters
on a Lead Smelter Sintering Stream

Operating Assumptions	\$/Mg Lead	\$/kg Hg
Base Case (Table B-22)	3.0	1,190
Inlet Hg Level ^a		
5,000 µg/dscm	7.2	583
10,000 µg/dscm	12.5	508
Installation Cost		
50% of Purchased Equipment	3.2	1,320
75% of Purchased Equipment	3.6	1,470

^a A mercury reduction of 90 percent is assumed for all cases.

Table B-27
Sensitivity Analysis for Selenium Filters
on a Lead Smelter Furnace Stream

Operating Assumptions	\$/Mg Lead	\$/kg Hg
Base Case (Table B-23)	6.4	1,140
Inlet Hg Level ^a		
5,000 µg/dscm	15.6	556
10,000 µg/dscm	27.2	483
Installation Cost		
50% of Purchased Equipment	7.1	1,260
75% of Purchased Equipment	7.9	1,470

^a A mercury reduction of 90 percent is assumed for all cases.

TECHNICAL REPORT DATA

(Please read Instructions on reverse before completing)

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16. ABSTRACT This volume of the draft Mercury Study Report to Congress focuses on mercury control technologies, costs, and financial impact estimates for six industries: municipal waste combustors, medical waste incinerators, utility boilers, chlor-alkali plants, primary copper smelters, and primary lead smelters. Control technologies reviewed for feasibility are pollution prevention measures (including product substitution and process modification), materials separation, and flue gas treatment methods. The estimated cost effectiveness values represent generalized costs and are not intended to be site-specific. Analysis of Clean Air Act Authorities that could be used to control mercury emission sources are summarized. Specific technological approaches to limiting mercury emissions for these six industries are described along with data needed to improve these assessments.					
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