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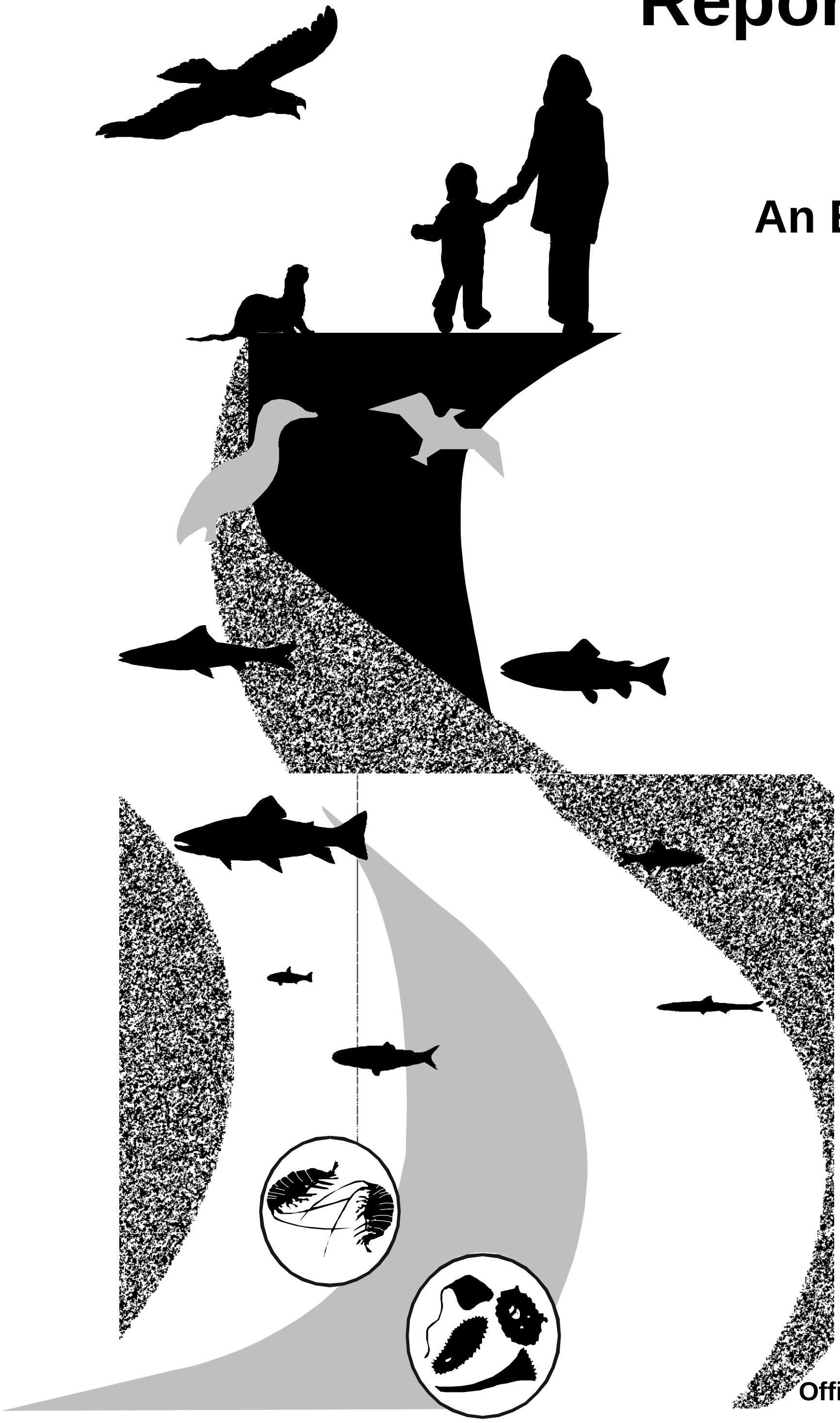
Volume VIII: An Evaluation of Mercury Control Technologies and Costs



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

MERCURY STUDY REPORT TO CONGRESS

VOLUME VIII:

AN EVALUATION OF MERCURY CONTROL TECHNOLOGIES AND COSTS

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Office of Air Quality Planning and Standards
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U.S. Environmental Protection Agency

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LIST OF SYMBOLS, UNITS AND ACRONYMS

AC	Activated carbon
APCD	Air pollution control device
ASME	American Society of Mechanical Engineers
ASTM	American Society for Testing and Materials
C	Carbon
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
ECTC	Environmental Control Test Center
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
M29	U.S. EPA Draft Multi-Metals Method or Method 29
MACT	Maximum achievable control technology
MB	Mass burn
MCL	Maximum contaminant level

LIST OF SYMBOLS, UNITS AND ACRONYMS
(continued)

MESA	Mercury Speciation Adsorption sampling train
Mg	Megagram or Metric ton (2000 pounds)
MSW	Municipal solid waste
MW	Megawatt
MWCs	Municipal waste combustors
MWIs	Medical waste incinerators
NAAQS	National Ambient Air Quality Standards
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
SO_2	Sulfur dioxide
SO_3	Sulfur trioxide
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
U.S.DOE FETC	United States Department of Energy Federal Energy Technology 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 an eight-volume Mercury Study Report to Congress. This Report -- Volume VIII 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 section 112 of the CAA.

Control Technologies and Associated Costs and Impacts

This Report focuses on mercury control technologies, costs and financial impact estimates for four industries: municipal waste combustors (MWCs), medical waste incinerators (MWIs), utility boilers and chlor-alkali plants. 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 potential 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 MWCs, MWIs, and chlor-alkali plants; results of the cost analysis for utility boilers are presented in Table ES-2. For each of the four 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).

The financial impact of mercury controls was determined for each of the model plants representing the four 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 the chlor-alkali industry, which has little control over the price it is able to receive for its products and thus cannot pass mercury control costs on to consumers, financial impact was determined based on both the ratio of 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.

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
	Good combustion, wet scrubber or dry scrubber with carbon injection Switching with waste segregation Switching without waste segregation	--	--	For cost-effectiveness estimates for individual facilites, the reader should consult *Hospital/Medical/Infectious Waste Incinerators: Background Information for Promulgated Standards and Guidelines - Regulatory Impact Analysis for New and Existing Facilities (EPA-453/R-97-009b).*

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

Source	Mercury Control Technique	Cost Effectiveness		Cost Comments
		$/lb Hg Removed	Other Measures	
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

Table ES-2
Cost Effectiveness of Control Technologies for Utility Boilers

Model	Model Definition				Sensitivity Analysis for Models			
	Size (MW)	Fuel	Existing Controls	Mercury Control	Source	Carbon Usage[a] (g C/g Hg)	Cost Effect. (mils/kWh)	Cost Effect. ($/lb Hg)
1a	975	Low-sulfur Coal	ESP	AC injection	EPA	34,200	1.82	22,100
					DOE	100,000	5.58	67,700
1b	975	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	1.43	17,400
					DOE	9,400	2.10	25,400
1c	975	Low-sulfur Coal	ESP	Spray cooler, AC injection	EPA	460	0.40	4,940
					DOE	30,000	2.19	26,500
1d	975	Low-sulfur Coal	ESP	Carbon filter bed	EPA	--[b]	2.70	32,700
					DOE	--	NA[c]	NA
2	975	High-sulfur Coal	ESP/FGD	Carbon filter bed	EPA	--	3.1	37,800
					DOE	--	NA	NA
3a	100	Low-sulfur Coal	ESP	AC injection	EPA	17,200	1.16	14,200
					DOE	100,000	5.71	70,000
3b	100	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	2.09	27,700
					DOE	12,600	3.15	38,600

[a] The lower carbon injection rates represent low-temperature flue gas while the higher rates are for high-temperature operations. Both the low and the high carbon injection rates would achieve a 90 percent reduction in mercury emissions under the given temperature scenario.
[b] The mercury control is a stationary bed that does not require carbon injection.
[c] NA = Not available.

The estimated financial impact of mercury controls for MWCs and MWIs is examined in Table ES-3. Estimated financial impacts to the chlor-alkali industry are summarized in Table ES-4. 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-5.

Table ES-3
Potential Cost Increases[a] for MWCs and MWIs

Control Option	Municipal Waste Combustors	Medical Waste Incinerators
Activated Carbon Injection	Small[b]: 6.9% Large[c]: 1.3%	
Hospitals, nursing homes, research laboratories:		
Switching with waste segregation		0.01 - 0.04%
Switching with no waste segregation		0.02 - 0.09%
Commercial Incineration		2.6 %

[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.

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.

Table ES-4
Estimated Annual Profits, Expenditures, Revenues and Financial Impacts[a] for Chlor-Alkali Plants

Parameter	Chlor-Alkali Plants
Total Annual Profits (millions of dollars)	$12.9
Total Annual Expenditures (millions of dollars)	$26.7
Financial Impact	
Membrane Cell Process	12% of Expenditures
Depleted Brine Scrubbing	5.1% of Profits 0.7% of Expenditures
Treated Activated Carbon Adsorption	3.9% of Profits 0.5% 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.

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, has drafted 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 at least 12 states have enacted laws that limit the amount of mercury in alkaline batteries to 0.025 percent by weight.

Table ES-5
Annual Revenues and Potential Cost Increases for Utility Boilers

Model	Model Definition				Sensitivity Analysis for Models				
	Size (MW)	Fuel	Existing Controls	Mercury Control	Source	Carbon Usage [a] (g C/g Hg)	Estimated Revenue ($ Million)	Annual Cost (10^6 $/yr)	Potential Cost Increases %
1a	975	Low-sulfur Coal	ESP	AC injection	EPA	34,200	287	10.1	3.5
					DOE	100,000	287	31.0	10.8
1b	975	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	287	7.94	2.8
					DOE	9,400	287	11.6	4.0
1c	975	Low-sulfur Coal	ESP	Spray cooler, AC injection	EPA	460	287	2.26	0.8
					DOE	30,000	287	12.1	4.2
1d	975	Low-sulfur Coal	ESP	Carbon filter bed	EPA	--[b]	287	14.9	5.2
					DOE	--	287	NA[c]	NA
2	975	High-sulfur Coal	ESP/FGD	Carbon filter bed	EPA	--	287	17.3	6.0
					DOE	--	287	NA	NA
3a	100	Low-sulfur Coal	ESP	AC injection	EPA	17,200	29	0.66	2.3
					DOE	100,000	29	3.25	11.2
3b	100	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	29	1.29	4.4
					DOE	12,600	29	1.79	6.2

[a] The lower carbon injection rates represent low-temperature flue gas while the higher rates are for high-temperature operations. Both the low and the high carbon injection rates would achieve a 90 percent reduction in mercury emissions under the given temperature scenario.

[b] The mercury control is a stationary bed that does not require carbon injection.

[c] NA = Not Available.

Management Alternatives

Effective control of mercury emissions may require a mix of strategies. The four major types of control techniques reviewed include:

- Pollution prevention measures, including product substitution, process modification and materials separation;
- Coal cleaning;
- Alternative approaches; and
- Flue gas treatment technologies.

Pollution prevention may be suitable for those processes or industries where a mercury substitute is demonstrated and available (e.g., mercury cell chlor-alkali plants). Another pollution prevention measure is material separation, which would be 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., medical waste incineration). 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. 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).

Cost-effective opportunities to deal with mercury during the product life-cycle, rather than just at the point of disposal, need to be pursued. A balanced strategy which integrates end-of-pipe control technologies with material substitution and separation, design-for-environment, and fundamental process change approaches is needed. In addition, international efforts to reduce mercury emissions as well as greenhouse gases will play an important role in reducing inputs to the global reservoir of mercury.

Because of the current, limited scientific understanding of the environmental fate and transport of this element, it is not possible to quantify the contribution of U.S. anthropogenic emissions relative to other sources of mercury, including natural sources and re-emissions from the global pool, on methylmercury levels in seafood and freshwater fish consumed by the U.S. population. Mercury methylation and subsequent uptake in fish is complex and not well understood. As a result, it cannot be assumed that a change in total mercury emissions will be linearly related to any resulting change in methylmercury in fish, nor over what time period these changes would occur. This is an area of ongoing study.

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.

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 constituents.

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.

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

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 less dense 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. For example, in one bench-scale study, five types of raw coal were washed by conventional cleaning methods followed by column froth floatation or selective agglomeration. Conventional cleaning and column froth flotation reduced mercury concentrations from the raw coals by 40 to greater than 57 percent, with an average of 55 percent. Conventional cleaning and selective agglomeration reduced mercury concentrations from the raw coals by greater than 63 percent to 82 percent, with an average of 68 percent. In a second bench-scale study in which three types of coals were cleaned with a heavy-media-cyclone (a conventional cleaning method) followed by a water-only-cyclone and a column froth flotation system, mercury concentrations in the raw coal were reduced by as much as 63 to 65 percent. Bench-scale testing is also being carried out by DOE to investigate the use of naturally occurring microbes to reduce 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.

Alternative Approaches

There are a variety of flexible approaches for reducing the emissions of hazardous air pollutants. These include incentive- or market-based systems, “co-control,” and energy conservation and renewable energy initiatives.

Incentive-based systems are tools that provide industry with more flexibility than traditional regulatory programs. In such a system, the regulatory agency generally sets a ceiling on allowable emissions (a cap) for each source along with clear and certain penalties for missing the target, but regulated entities have complete choice in how these targets will be met. The cost to industry is determined by the market and by the innovation used in meeting the cap. Emissions cap programs allow for increased incentives because sources that reduce emissions below their cap can sell the surplus reduction to sources that cannot achieve their cap. Trading is promising where sources have different compliance costs, or where local environmental impacts are minimal. Sources that reduce emissions before they are required to do so can “bank” the excess reductions and save them for later. Examples of existing market-based programs include the SO_2 allowance trading and NOx averaging programs implemented under Title IV of the CAA Amendments to reduce acid deposition; the Regional Clean Air Incentives Market Program and Rules developed in California to reduce emissions of NOx, SOx, and reactive organic compounds; and U.S. EPA’s Lead Trading Program designed to reduce the emissions of lead from gasoline in the mid-1980's.

Incentive-based systems to reduce mercury emissions, either through regulation or voluntary means, may be attractive to utilities and other facilities for several reasons: to reduce mercury emissions at a lower per unit cost, to insure against future regulation, to reduce the compliance costs of regulation, to bank credits toward future regulatory requirements, to build experience with technology and to demonstrate environmental leadership. Also, incentive-based programs could provide financing for the control of mercury among different industries (and potentially other countries) and may be a viable option for utilities and other sources where cost-effective technologies have yet to be identified.

Co-control refers to the control of mercury by control devices or other management measures that were designed or prescribed to limit the emissions of pollutants other than mercury. One example of co-control is fuel switching, in which one fuel is switched to another (e.g., high-sulfur coal to low-sulfur coal, or coal to natural gas) to achieve emission reductions in a more flexible or cost-effective way. Co-control can also be achieved through the implementation of the National Ambient Air Quality Standards (NAAQS) for ozone and particulate matter (PM). In support of the revised ozone and PM NAAQS, U.S. EPA conducted numerous detailed analyses to predict what control approaches industry might use to achieve the new standards.

U.S. EPA estimates that implementation of the New Fine Particle Standard for ambient air quality through a regional control strategy that significantly reduces SO_x below the CAA's Title IV requirements can indirectly lower forecasted mercury emissions in 2010 by about 11 tons from electric power generation by units burning fossil fuels. This reduction occurs from both the additions of flue gas desulfurization units (scrubbers) at coal-fired boilers to lower SO_x emissions and through greater reliance by the power industry on producing electricity from natural gas as another way to reduce SO_x. In the Regulatory Impact Analysis for the new NAAQS, U.S. EPA estimated that in 2010 a regional SO_x reduction strategy for the electric power industry to lower fine particle formation will lead to the installation of scrubbers on additional 60 gigawatts of coal-fired capacity (increasing forecasted scrubber capacity under Title IV by about two-thirds). U.S. EPA assumes that scrubbers remove close to 30 percent of the mercury contained in coal flue gas. U.S. EPA also estimated that electricity produced from natural gas would increase by 16 percent above baseline levels. Natural gas combustion produces negligible levels of mercury emissions.

Title IV of the CAA 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. Renewable energy is defined as energy that is derived from biomass, solar, geothermal or wind.

Flue Gas Treatment Technologies

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:

- 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, so can be efficiently scrubbed by a wet FGD system. 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.

- 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 (except for high removal efficiencies for $HgCl_2$ by wet scrubbers).

- 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. The cost analyses cover four source categories: municipal waste combustors (MWC), medical waste incinerators (MWI), chlor-alkali plants, and utility boilers.

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-6 presents the four 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 assumptions 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. These requirements would be specific to a regulatory action. 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 reductions do not illustrate two important considerations. One is that, as presented, all of the cost of control could mistakenly be attributed to mercury removal. As described in this Report, many of these controls achieve reductions of other pollutants as well (e.g., acid gases, dioxin, other metals). In some cases (e.g., the emission guidelines for MWI), the choice of control technology or control strategy is aimed at reducing pollutants other than mercury. In these cases, there is a co-control benefit of mercury reduction. The benefits of reducing other pollutants should be considered when interpreting the mercury control costs. Second, the technologies available for mercury control represent relatively new applications of these technologies. Thus, in the future, it is likely that as new or emerging technologies develop, the cost-effectiveness of control will improve. Air pollution control and prevention techniques are continuously under development and improvement. There is a fairly rapid pace of innovation in the air pollution control sector. The demand for cleaner products and cleaner processes that lower overall costs, combined with the necessity for improved air and water quality, create strong incentives for technological innovation and a growing market for such innovations. As the demand for more innovative, cost-effective and cost-saving technologies increase, new technologies will move from the research and development or pilot program phase to commercial availability.

Table ES-6
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[a]	Potential National Annual Costs[b]	Cost-Effectiveness ($/lb of mercury removed)[c]
Municipal waste combustors	129	18.6	Material separation Product substitution Activated carbon injection Carbon filter beds Polishing wet scrubber	27 tons	$11.4-47 million	$211-870
Medical waste incinerators	~2,400	10.1	Material separation Wet scrubber or dry scrubber with carbon Activated carbon injection	15 tons (95% reduction)	$60-120 million[d]	$2,000-$4,000[d]
Coal-fired utility boilers	426 (1,043 boilers)	32.5	Fuel switching Advanced coal cleaning Activated carbon injection Carbon filter beds	37 tons (90% reduction) [e]	$5 billion	$67,700-$70,000
			Co-control: ozone and PM NAAQS	11 tons[f]	No incremental mercury control costs	No incremental mercury control costs
Chlor-alkali plants using the mercury cell process	14	4.5	Process modification Depleted brine scrubbing Treated activated carbon adsorption	7.1 tons (100% reduction)	$65 million	$4,590
Total	~3,586	65.7			~$5.2 billion	

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

[a] Estimated reductions assuming every facility could achieve the reduction listed.

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

[c] Where cost-effectiveness values are presented as a range, the values reflect the range across facilities of different sizes.

[d] Cost of control should not be attributed to mercury control alone. Wet scrubbers efficiently remove nine other pollutants from the MWI flue gas as required by the emission guidelines for MWIs.

[e] The potential national reductions reflect sufficient amounts of activated carbon to control mercury emissions from coal-fired utility boilers by 90 percent. Activated carbon injection has not been demonstrated for a full-scale utility boiler application. Control costs are upper bound based on high temperature activated carbon injection. The 37 tons reduction is 90 percent of 41 tons, accounting for 11 ton reduction from the ozone and PM NAAQS.

[f] Assumes some fuel switching and additional installation of wet scrubbers which are assumed to remove 30 percent.

While existing technology will play a key role in reducing mercury from some sources, emerging technology may be more appropriate for others. Innovations in environmental policies may also play a key role in developing a national management strategy for mercury. These innovations could include multi-media approaches, greater emphasis on pollution prevention, regional control strategies and optimization of co-control opportunities.

Benefits Assessment Framework

The benefits assessment framework presented in Chapter 4 of this Volume discusses the theoretical background supporting a benefits assessment for reducing mercury contamination and raises relevant issues to be considered in future work on such a benefits assessment. The framework identifies and discusses the various steps and inputs necessary for such an assessment. It discusses the basic concepts and issues relevant to understanding and conducting an assessment of the economic benefits associated with reducing mercury contamination. An additional goal of the framework is to provide background on the theoretical and practical issues that need to be addressed in preparing a rigorous, comprehensive benefits assessment.

Performing a comprehensive benefits analysis for mercury contamination will require a coordinated effort across the Agency to take advantage of the knowledge and ongoing work on mercury and benefits assessment within various offices. Specific steps are identified for moving forward with a thorough benefits assessment, identifying readily available information on the effects of mercury contamination and possible approaches to assessing the benefits of reducing those effects. Relevant issues to be considered during this process are highlighted.

Ongoing U.S. EPA Activities to Reduce Mercury in the Environment

Mercury is a priority pollutant across numerous U.S. EPA programs including air, water, hazardous waste and pollution prevention. There are numerous activities currently underway to reduce mercury emissions and releases to the environment. A number of these activities are described below which reflect the broad scope U.S. EPA's approach to the mercury issue.

Clean Air Act Initiatives - The U.S. EPA already has efforts underway to reduce mercury emissions from industrial sources. Specific actions being taken under the Clean Air Act include the following:

- The U.S. EPA has promulgated final emission limits for municipal waste combustors and medical waste incinerators under the authority of section 129 of the CAA. Emission standards have also been proposed for hazardous waste incinerators.

- The U.S. EPA is evaluating the impacts of mercury reductions for the following source categories: commercial/ industrial boilers, chlor–alkali plants using the mercury cell process and portland cement kilns.

- The U.S. EPA plans to evaluate whether secondary mercury production should be added to the source category list under section 112(c) of the CAA and subsequently evaluated for regulation under the authority of section 112(c)(6).

- Numerous CAA requirements involve utilities either directly or indirectly. Section 112(n)(1)(B) which required this Mercury Study Report to Congress specified utility boilers for analysis as did section 112(n)(1)(A) which is referred to as the Utility Air Toxics Report to Congress (Utility Study). The Utility Study is charged with evaluating the hazards to public health reasonably anticipated to occur as a result of emissions by electric utility steam generating units of pollutants listed under Section 112(b), including mercury, and to evaluate the impact of other provisions of the CAA on these emissions. The other provisions of the CAA would include the Acid Rain program as well as provisions pertaining to National Ambient Air Quality Standards. The Utility Study is also required to offer a regulatory recommendation with respect to regulation of utility boilers under section 112 of the CAA.

- The "Great Waters" program (section 112(m)) is an ongoing study with biennial reports to Congress required. The program must identify and assess the extent of atmospheric deposition of hazardous air pollutants (including mercury) to the Great Lakes and other specified waters, the environmental and public health attributable to atmospheric deposition and the contributing sources. Two reports have been submitted to Congress which address these issues.

Mercury Task Force - U.S. EPA established this task force to consider strategies for coordinating various programs for use, management, and disposal of mercury.

Virtual Elimination Project - U.S. EPA and Environment Canada have created this joint project aimed at developing strategies to achieve the Great Lakes Water Quality Agreement's (GLWZA's) goal that persistent toxic substances should be "virtually eliminated" from the Great Lakes.

Other Pollution Prevention Programs - U.S. EPA is working with state and local governments to develop a national network of prevention programs that will assist regulators at all levels of government in promoting pollution prevention.

Conclusions

<u>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.</u>

- 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.

- 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.

- 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.

- Limited pilot-scale studies with the injection of activated carbon indicate variable control of mercury from 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 capture in utility flue gas streams include flue gas volume, flue gas temperature, flue gas vapor and particulate phase constituents (e.g., chlorine as HCl, nitrogen oxides, sulfur oxides, metal oxides on the surfaces of particulate matter, fly ash composition, percent carbon in fly ash, etc.), the mercury concentration and chemical species being formed, and the existing APCDs being augmented (e.g., fabric filters versus ESPs) for mercury capture by activated carbon. Mercury capture is mass transfer limited in utility flue gas streams due to the low mercury concentrations in the extremely high volumes of flue gas.

- The addition of activated carbon to utility flue gas for mercury control would increase the amount of particulate matter requiring disposal. Studies are just being conducted to assess the stability/leachability for possible re-emission of mercury in the near- and long-term from the carbons and sorbents captured in the gas phase and disposed of in the solid phase in landfills.

- The chemical species of mercury formed during the combustion process and post-combustion conditions vary significantly from one plant to another. While combustion conditions vary, the subsequent fly ash and vapor phase constituents can play a major role in the percentage of the chemical species of mercury formed. Understanding the rate controlling mechanisms (e.g., transport, equilibrium, and kinetics) will aid in predicting and possibly controlling the species formed in order to optimize existing APCDs for mercury removal. Removal effectiveness is dependent on the species of mercury present. There are promising technologies being investigated at the bench- and pilot-scales for the removal of mercury, but none have been tested at the full scale.

- Given the relative low maturity level of these technologies being tested, commercial deployment is still several years away. Deployment is strongly dependent on understanding the fundamental mechanisms of the flue gas and mercury chemistries, and the results of well designed bench- and larger pilot-scale studies. In addition, no single control technology has been tested at the pilot-scale or larger that removes all forms of mercury.

<u>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</u>:

- Data from full-scale testing of activated carbon injection with and without flue gas cooling at a variety of coal-fired utility boiler systems representative of the utility industry.

- Additional data on the efficiency of various sorbents including fly ash-based sorbents, activated carbon, impregnated carbons, noble metal sorption, and other types of sorbents, in reducing the different chemical species of mercury present in flue gas.

- Additional data on improving the mercury mass transfer limitation(s) and reactivity of activated carbon while decreasing the mass carbon-to-mercury ratio.

- Information on the cost-effectiveness and commercialization costs of other technologies for mercury control that are currently in the research stage. These include impregnated activated carbon, sodium sulfide injection, activated carbon fluidized bed, noble metal sorption, sorbent injection alone and with humidification, and other types of sorbents.

- Additional data/information on the impacts of flue gas cooling, through humidification on acid mist formation and the means to control the sulfur trioxide before the acid mist is formed. The range of the cost of control of sulfur trioxide under carbon injection scenarios with humidification needs to be determined.

- Additional data on the ability and cost of conventional or advanced coal cleaning techniques to remove mercury from raw coal. The potential for mercury emissions from coal-cleaning slurries needs to be characterized. In addition, the added costs for advanced coal cleaning in combination with post-combustion controls for mercury have not been fully developed.

- Additional data on the fundamental mechanisms responsible for conversion of mercury to other chemical species and the natural adsorption of mercury with the native fly ash as a result of combustion of certain coals or post-combustion conditions.

- Additional information on improving the capture of elemental mercury and the oxidized form of mercury in wet FGD systems.

- Improvement and development of modeling capabilities for predicting mercury speciation, conversion, and adsorption with fly ashes and sorbents in conjunction with laboratory and pilot-scale test results.

- Improvement and development of predictive models for the emission of elemental and non-elemental forms of mercury.

- Data on the impacts on waste and by-products as a result of mercury capture in fly ash, sorbents mixed with fly ash, FGD waste, and other aqueous and solid effluent streams associated with utilities. Determine the stability (identify the form of mercury in waste) and leaching properties, and potential near- and long-term re-emission of mercury from the various wastes.

- Improvements on development of real time mercury measurement techniques (CEMs) to measure and speciate at the 1 ppb level in order to possibly reduce the cost of control of mercury by optimization through advanced process control.

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 an eight-volume Mercury Study Report to Congress. The eight volumes are as follows:

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

This Report, Volume VIII, 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 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 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 VII). 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 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 four 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 six 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 presents a mercury benefits assessment framework and discussion.

- Chapter 5 summarizes mercury control strategies, including federal, international and state authorities and activities that impact mercury emissions.

- Chapter 6 presents <u>conclusions</u>.

- Chapter 7 identifies <u>research needs</u>.

All references cited in the volume are listed after Chapter 7. 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. Four major types of control techniques are described:

- Pollution prevention measures, including product substitution, process modification and materials separation;
- Coal cleaning;
- Alternative approaches, including incentive-based systems, co-control and conservation measures; 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

Faced with the increasing costs and liabilities associated with end-of-the-pipe waste management practices, many waste generators are turning to pollution prevention as a potentially cleaner, safer, and more cost-effective alternative. Pollution prevention (also known as source reduction) is defined as:

> Any practice which reduces the amount of any hazardous substance, pollutant, or contaminant entering the waste stream or otherwise released to the environment (including fugitive emissions) prior to recycling, treatment, or disposal; and reduces the hazards to public health and the environment associated with the release of such substances, pollutants, or contaminants (Pollution Prevention Act of 1990, 42 U.S.C. Section 13101, et. Seq.).

Pollution prevention includes such techniques as toxics use reduction, raw material substitution, process or equipment modification, product redesign, training, improved inventory control, production planning and sequencing, and better management practices.

Much of the pollution prevention idea revolves around “life-cycle assessment.” The concept of life-cycle assessment involves evaluating the environmental effects associated with any given activity from the initial gathering of raw material from the earth until the point at which all residuals are returned to the earth. This “cradle to grave” concept is not new, but only recently have industry, environmental interests and regulatory authorities begun to develop procedures that facilitate understanding overall processes, the underlying data and the inherent assumptions that make such an analysis possible. U.S. EPA and technical trade organizations have developed guidance on the specific details involved in conducting life-cycle assessments and the concept is gaining strength as a means of pollution prevention. Often life-cycle analysis and the resulting pollution prevention alternatives can reduce the cost associated with production or pollution controls.

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	• 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	• 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	• 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 • Material separation programs 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, $S0_2$, acid gases	• 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	MWCs, MWIs, boilers	Can be >90% for water-soluble species; limited for elemental mercury	Yes	Acid gases, metals, particulate matter, dioxins, furans (from MWCs & MWIs)	• Have not been applied to MWCs in the U.S., although they have been used at MWCs in Europe and MWIs in the U.S. • 25 percent of coal-fired boilers currently have wet scrubbers for sulfur dioxide removal • Requires treatment of wastewater prior to disposal • May form more toxic, lesser-chlorinated dioxin and furan congeners from MWCs & MWIs, but not from utility boilers
Depleted brine scrubbing	Chlor-alkali plants	98%	Yes	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	• 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	• Factors that influence performance include inlet mercury concentrations and its form(s), 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 plan; 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 and re-emission of mercury from the filter itself
Activated carbon injection	MWCs, MWIs, utility boilers	50-90+%	Yes	Chlorinated dioxins and furans, potentially other semi-volatile organics	• 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 volume & temperature, flue gas vapor and particulate phase constituents (e.g. chlorine as HCl, nitrogen & sulfur oxides, metal oxides on the surfaces of particulate matter, fly ash composition, percent carbon in fly ash, etc.), the mercury concentration and chemical species being formed, the existing APCDs being augmented (e.g. fabric filters verses ESPs) for mercury capture by activated carbon, and the type & amount of activated carbon being used. • Addition of carbon could have an impact on amount of particulate matter requiring disposal from utility boilers, but not from MWCs or MWIs

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[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.

According to the Society of Environmental Toxicology and Chemistry's guidance document entitled "A Technical Framework for Life-Cycle Assessments," the principles of life-cycle assessment are based on three basic components:

- Life-cycle inventory -- An objective, data-based process of quantifying energy and raw material requirements, air emissions, waterborne effluents, solid waste, and other environmental releases incurred throughout the life cycle of a product, process, or activity.
- Life-cycle impact analysis -- A technical, quantitative, and/or qualitative process to characterize and assess the effects of the environmental loadings identified in the inventory component. The assessment should address both ecological and human health considerations as well as other effects such as habitat modification and noise pollution.
- Life-cycle improvement analysis -- A systematic evaluation of the needs and opportunities to reduce the environmental burden associated with energy and raw materials use and waste emissions throughout the whole life cycle of a product, process, or activity. This analysis may include both quantitative and qualitative measures of improvements, such as changes in product design, raw material use, industrial processing, consumer use and waste management.

These three components comprise an integrated approach that, when combined with other appropriate information, can provide the information needed to maximize environmental improvement. U.S. EPA has compiled guidance on the life-cycle analysis inventory component (U.S. EPA, 1993d) and also has compiled a directory of resources for conducting life-cycle assessments and pollution prevention in general (U.S. EPA, 1993e).

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 discusses various other management measures that could be used to reduce mercury emissions from utilities, including incentive-based systems, co-control and conservation measures.

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. 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. This subsection discusses batteries and fluorescent lights and other products.

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.

2.1.1.2 Fluorescent Lights and Other Products

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, although Philips Lighting is now manufacturing lamps containing less than 10 mg of mercury. 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.

Other Products

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 also resist dissolution, wear better, and require a less precise technique during placement.

Mercury use by the dental profession decreases each year as a result of 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, decreasing the amount of amalgam discharge into the waste stream during amalgam replacement 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. Members of the auto industry are also in the process of replacing mercury containing switches used in their automobiles. These mercury switches have typically been used for lighting, anti-lock brake systems and active ride control. GM, Ford, and Chrysler have indicated that mercury switches will be replaced with suitable alternatives in most 1998 model year cars. (Freedman, 1996)

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. When economics justify it, a company will dismantle a mercury chlor-alkali operation, and replace it with the membrane technology (Lawrence, 1994).

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.

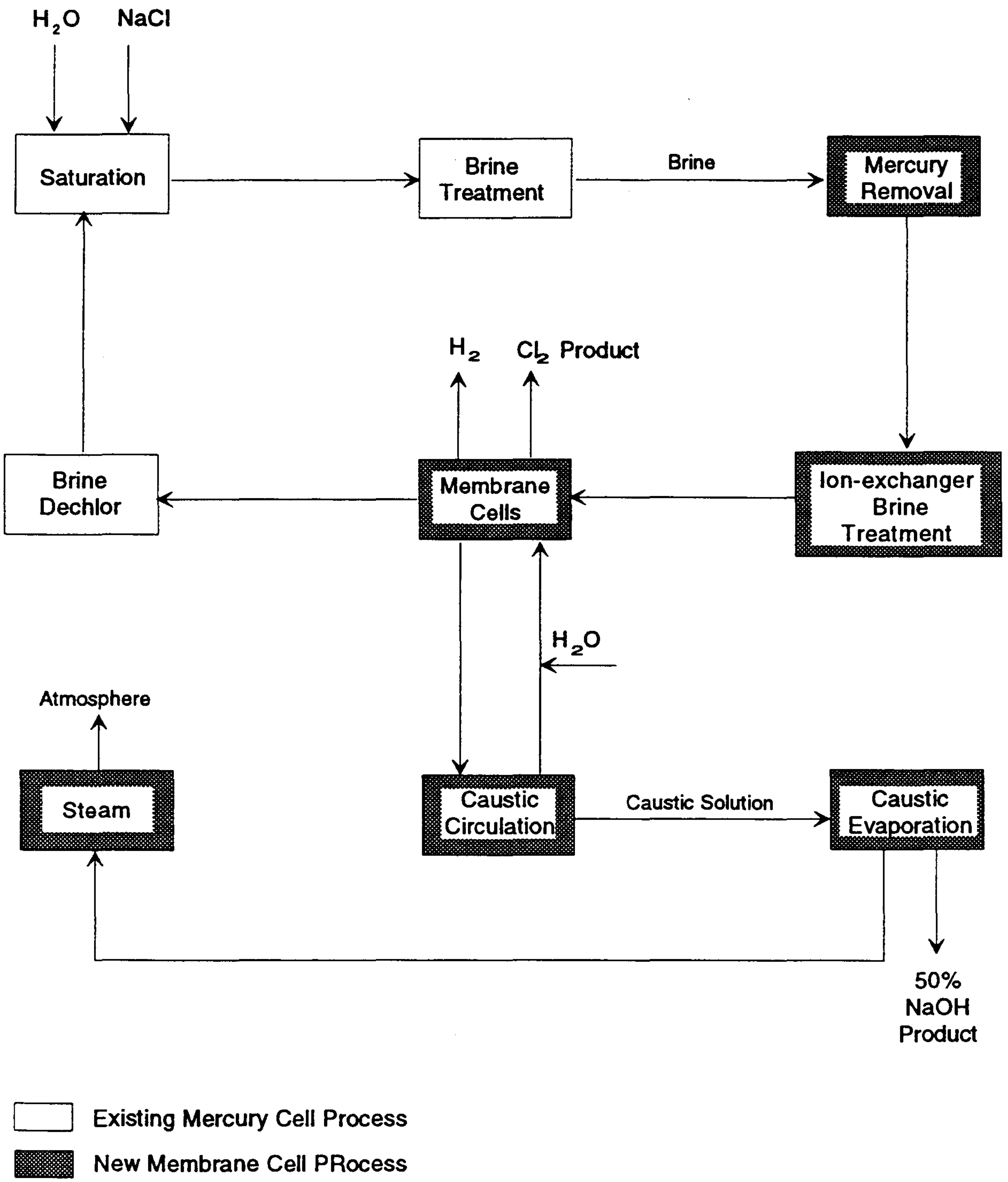
Other materials contain higher concentrations of mercury, but make up only a very small portion (less than 1 percent) of the waste stream. These materials include mercuric oxide batteries, fluorescent lights, thermostats and other electrical items. Separation of such materials can reduce mercury input 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.

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, have historically been 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.

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



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.

Many communities in the United States have implemented household battery separation programs in an effort to reduce mercury in the waste stream destined for MWCs (Lawrence, 1994). 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. In 1989, there were five mercury recycling facilities in the U.S. where the mercury from such lights can be recovered. However, 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.

2.1.3.4 Mercury Switches in Automobiles

Mercury containing switches are used in the auto industry for lighting, anti-lock brake systems and active ride control. The quantity of mercury contained in these switches is low. For example, switches used for light activation usually employ less than one gram of mercury. While that is not much for one switch, the number of switches in vehicles disposed of every year is noteworthy. About 9.8 metric tons of mercury go into an estimated 13 million switches supplied each year for auto use; according to the U.S. Council for Automotive Research (USCAR), Vehicle Recycling Partnership, 94 percent of automobiles that go out of registration are recovered for recycling, but the mercury switches are generally not recycled. The current fate of these switches is not well known, and at least some releases of mercury to air and land may be occurring in connection with the disposal of automobiles.

Even though mercury switches are being phased out of new vehicles, they will remain in existing vehicles for several years. Therefore, the American Automobile Manufacturers Association (AAMA), in conjuction with USCAR, is developing a procedure to identify, remove and dispose of, or recycle switches containing mercury. Their task force is drafting procedures for distribution to U.S., foreign and transplant automakers and to the recycling and dismantling industries.

2.1.4 Coal Cleaning

Approximately 77 percent of the eastern and midwestern bituminous coal shipments are cleaned to meet customer specifications for heating value, ash content and sulfur content (Akers et al., 1993). Along with removing ash-forming and sulfur-bearing minerals, coal cleaning can also reduce the concentration of many trace elements, including mercury.

Conventional coal cleaning methods are 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, dense media washing, uses heavy liquid solutions usually consisting of magnetite (finely ground particles of iron oxide) to separate coal from impurities.

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 and cleaned coal, as well as the percent reduction achieved by conventional coal cleaning methods. 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.

2.1.4.1 Advanced Coal Cleaning

Advanced coal cleaning methods such as selective agglomeration and column froth flotation have the potential to increase the amount of mercury removed by conventional cleaning alone. In one bench-scale study, five types of raw coal were washed by conventional cleaning methods followed by column froth flotation or selective agglomeration. Conventional cleaning and column froth flotation reduced mercury concentrations from the raw coals by 40 to greater than 57 percent, with an average of 55 percent (Smit, 1996). Column froth flotation reduced mercury concentrations remaining in the washed coals by 1 to greater than 51 percent, with an average of 26 percent (Smit, 1996). Conventional cleaning and selective agglomeration reduced mercury concentrations from the raw coals by greater than 63 percent to 82 percent, with an average of 68 percent (Smit, 1996). Selective agglomeration reduced mercury concentrations remaining in the washed coals by greater than 8 percent to 38 percent, with an average of 16 percent (Smit, 1996).

In a second bench-scale study, three types of coals were cleaned by a heavy-media-cyclone (a conventional cleaning method) followed by a water-only-cyclone and a column froth flotation system. The heavy-media-cyclone reduced mercury concentrations in the raw coal by 42 to 45 percent (ICF Kaiser Engineers, 1995). The water-only-cyclone and column froth flotation system reduced the concentrations of mercury remaining in the cleaned coals by 21 to 23 percent (ICF Kaiser Engineers, 1995). The combined reduction in mercury concentrations from the coals ranged from 63 to 65 percent (ICF Kaiser Engineers, 1995).

Bench-scale testing is also being carried out by DOE 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 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.

While advanced cleaning technologies can reduce mercury from the coal (30 to greater than 60 percent) the potential impact on post combustion form and control of the remaining mercury has not been thoroughly investigated. Mercury mass transfer limitations are encountered in emissions control systems on furnaces firing raw or conventionally cleaned coals. Advanced coal-cleaning may exacerbate this problem. In addition, chemical cleaning techniques being considered may provide a coal that yields a different form of mercury under combustion and post-combustion conditions. This could adversely impact the natural mercury capture of the fly ash and across wet/dry flue gas desulfurization (FGD) systems. There needs to be more laboratory, bench-, and pilot-scale combustion and subsequent post-combustion studies to evaluate these potential impacts. In addition, the added costs for advanced coal cleaning separately and in combination with post-combustion controls for mercury have not been fully developed.

2.1.4.2 Commercial Status

As mentioned above, approximately 77 percent of the eastern and midwestern bituminous coal is cleaned to meet customer specifications for heating value, ash content, and sulfur content. While most of this coal is cleaned by conventional cleaning methods, advanced cleaning methods, such as column froth flotation, are starting to emerge. Microcel™ is a type of column froth flotation available through ICF Kaiser Engineers and Control International. The company is the exclusive licensee for the technology in the coal fields East of the Mississippi River and has sold units for commercial operation in Virginia, West Virginia, and Kentucky, as well as in Australia under sub-license to Bulk Materials Coal Handling Ltd. Ken-Flote™ is another type of column froth flotation cell.

2.1.5 Alternative Approaches

In addition to the other measures discussed in this chapter, there are a variety of flexible approaches for reducing the emissions of hazardous air pollutants. For example, under Title IV of the CAA, U.S. EPA established a new approach as part of its Acid Rain Program for limiting SO_2 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 Incentive-Based Systems

Historically, environmental regulators have used command-and-control strategies to achieve reductions in air emissions of various pollutants. The resulting regulations often result in technology-based requirements determined by the cost of control of emissions on a dollars per mass basis. By using market incentives or market-based approaches to control pollution, emission reductions can potentially be achieved at lower cost and often with greater certainty.

Incentive-based systems are tools that provide industry with more flexibility than traditional regulatory programs. The flexibility and cost-savings provided by such approaches can allow for greater and faster reductions because incentives for innovations are created, and that innovation can be a driving force for technological change. Incentive-based systems do not eliminate the need for regulation, but stimulate the market to find new ways to reduce pollution through pollution prevention and new technologies. In general, these programs set a ceiling on allowable emissions (a cap) and allow flexibility in how the limits are met.

In an emissions cap program, the regulatory agency sets a cap for each source, along with clear and certain penalties for missing the target. Regulated entities have complete choice in how these targets will be met. In this scheme, environmental outcomes are known because they are not likely to exceed the cap. The cost to industry is determined by the market and by the innovation used in meeting the cap.

Emissions cap programs allow for increased incentives because sources that reduce emissions below their cap can sell the surplus reductions to sources that cannot achieve their cap. Trading is promising where sources have different compliance costs, or where local environmental impacts are minimal. Sources that reduce emissions before they are required to do so can "bank" the excess reductions and save them for later.

The following discussion highlights some existing market-based programs, followed by the applicability of such a strategy for mercury reductions.

Examples of Existing Market-Based Programs

SO_2 Allowance Trading. Sulfur dioxide (SO_2) emissions, a majority of which (70%) comes from coal-fired power plants, are the major cause of acid deposition (known also as "acid rain"). Under U.S. EPA's Title IV Acid Rain Program, the CAA established a goal of reducing the nations's annual SO_2 emissions by ten million tons from 1980 levels. The SO_2 program, which covers utility power plants in the continental U.S., utilizes a tradeable emission, permit system that caps the total allowable emissions, but allows flexibility to sources in meeting the cap limit.

The program works by distributing a stream of transferrable SO_2 emissions permits, each of which allows the holder to emit one ton of SO_2. The total annual issuance of these permits (called "allowances") is limited to a total number of tons that is well below the number of tons of SO_2 that were emitted in 1980. If a source's emissions exceed its allowance allocation, it must purchase additional allowances from other sources or through U.S. EPA auctions or direct sales, or pay substantial fines. Essentially, a source can choose the least costly method of complying with its emissions limits. Sources with high control costs may find it less expensive to pay another source to make its reductions by purchasing allowances from that source. Thus, utilities that reduce their emissions below their allocations can choose to sell their extra allowances or retain them by "banking" their emission reduction to use or sell in a future year.

As an adjunct to the SO_2 allowance program, sources emitting SO_2 that would otherwise not be affected by the Acid Rain Program may voluntarily "opt-in" based on their emissions in the baseline period and are then allowed to trade allowances with other affected sources under that program. This opt-in provision provides yet another opportunity for industry to pursue less costly means of reducing emissions.

U.S. EPA has estimated a 40 percent savings over the projected costs of the SO_2 program without the trade and cap component. As these cost savings are passed on as lower prices, they are expected to save purchasers of electricity an average of a half-percent or more on a typical bill, meaning that the advantages of the allowance program benefit not only the utilities but the entire economy. The allowance system contributes to savings in several ways. For example, it encourages the development of scrubber technologies that cut emissions below the rates required by the statute, since the excess reductions can be turned into saleable allowances. Similarly, it encourages the development and use of clean coal technologies or low-sulfur coals that might reduce emissions cheaply but not enough to comply with statutory limits if allowances could not be purchased. In addition, the allowance mechanism lets utilities switch fuels or apply scrubbers where they are most cost effective, and use the excess reductions to reduce the burden on their more expensive units.

NO_x Averaging. Like the SO_2 program, the NO_x Emissions Averaging program was designed to reduce acid deposition (of which NO_x is a constituent) and covers utility power plants in the continental U.S. This program sets an emission standard based on fuel burned and boiler type (e.g., 0.5 to 0.45 pounds of NO_x per million BTUs of fuel burned). Under an approved plan, the average NO_x emission rates of the pool's constituent boilers must meet an emissions standard -- individual boilers are not held to particular standards. Boilers unable to meet the applicable emissions rates using specified technologies, may opt to apply for a higher rate. Thus, the program employs a hybrid of performance- and technology-based standards. In addition, the NO_x program incorporates a flexible, market-like system that allows utilities to average the emissions of their boilers with those of other utilities within the same holding company, without regard to location. U.S. EPA weighs the emissions rate achieved by each boiler over the year (as measured by a continuous emissions monitor, or CEM) by the fuel used by the boiler. This procedure ensures that each averaging pool is limited to emitting no more tons of NO_x than its constituent boilers would have emitted if each one had held to its individual standard. Thus, the averaging provision allows some boilers to emit at rates above the nominal allowable rates if another boiler or boilers in the same pool emit at a rate low enough to compensate for the excess emissions.

Estimates of potential saving in the NO_x averaging program represent about 20 to 25 percent of the annual NO_x control costs to utilities per year of the NO_x controls without averaging. In the absence of the averaging program, utilities have no incentive to move beyond the minimum technology specified in the statute; the extra emission reductions provided by improved technology have no value to the utility, and are therefore unlikely to be sought.

RECLAIM. The Regional Clean Air Incentives Market (RECLAIM) Program and Rules were developed in the South Coast Air Quality Management District of California in conjunction with U.S. EPA, the California Air Resources Board, and the RECLAIM Steering and Advisory Committees. The program specifically targets emissions of NO_x, SO_x, and reactive organic compounds from participating facilities in the South Coast Air Basin, and replaces the emissions reductions requirements of more than 30 adopted rules and 12 future rules.

Facilities in the RECLAIM receive an annual emission cap (annual emission allocation) and are required to meet specific annual mass emission reduction targets. All companies with an allocation can buy, sell, trade, or otherwise transfer all or portions of their allocation, provided they follow the necessary protocols and reporting requirements. The RECLAIM rules require audits to ensure that program goals are being achieved and that improvements in air quality control technology are advancing. The cost of NO_x and SO_x programs under RECLAIM is estimated to be approximately 42 percent less than the cost of reducing these emissions under traditional U.S. EPA programs.

Lead Trading Program. U.S. EPA established the Lead Trading Program as an inter-refinery averaging mechanism designed to reduce the emissions of lead from gasoline in the mid-1980's. Averaging was achieved through the trading of lead credits equal to the difference between the actual average lead content and the current standard based on average quarterly lead content in gasoline.

Relative to other tradable permit markets, the lead credits market was very active throughout the life of the program, which ended in 1987. During the program, between one-fifth and one-third of the reporting refineries found it either necessary or desirable to purchase lead credits. U.S. EPA estimates that these trading provisions saved the refinery industry hundreds of millions of dollars during the transition from using lead, allowing companies to further refine the components of gasoline, use difference additives, or change the crude input used. This cost savings was passed on to retail consumers.

Applicability of Incentive-Based Programs to Mercury Reduction

Incentive-based programs to reduce mercury emissions, either through regulation or voluntary means, may be attractive to utilities and other industries for several reasons: to reduce mercury emissions at a lower per unit cost, to insure against future regulation, to reduce the compliance costs of regulation, to bank credits toward future regulatory requirements, to build experience with technology and to demonstrate environmental leadership.

Market-based programs could provide financing for the control of mercury among different industries (and potentially other countries) and may be a viable option for utilities and other sources where cost-effective technologies have yet to be identified. A market-based program may be applicable to mercury reductions for the following reasons:

- Because of mercury's persistence in the environment, any reductions in current emissions may be beneficial in the long term. A market-based program provides incentive to achieve lower-level reductions (often on the order of 10-50%, instead of 90% which is typical of regulatory programs). The high cost of higher-level controls may be offset by trades with lower-level controls. These lower-level reductions are often much less expensive on a cost per mass reduced basis and can be encouraged by making such reductions profitable for purchase by industries which have higher reduction costs.

- Mercury emissions do not always have a local environmental impact, but can be subject to long-range transport, contributing to both regional impacts and the "global" reservoir. This scenario is conducive to emission trades that balance overall emissions and reductions from different geographic locations.

- Currently, methods for measuring mercury emissions and some control technologies for mercury emissions reduction are expensive. If reducing mercury emissions is affordable, a new market incentive for companies that produce such technologies will be created.

Some State governments are already investigating the market-based approach. For example, Minnesota is embarking on an investigation of a comprehensive strategy for mercury control. The program is scheduled to begin operation in 2003 (see text box below). As part of that program, Minnesota is evaluating market-based incentives. In Michigan, a pollution prevention task force has recommended that utilities submit plans for reducing mercury emissions. The success of these state initiatives may influence future national mercury emissions reduction programs.

Minnesota Comprehensive Strategy for Mercury Control

The Minnesota Pollution Control Agency (MPCA) has embarked on a strategy to reduce mercury air emissions in a cost-effective manner. The state finds that mercury pollution, like acid rain, is a regional, national and even international problem, and that states with sensitive lakes must take the lead in developing solutions.

Although Minnesota has been a leader in reducing mercury pollution, these reduction efforts have been piecemeal. Now the state is developing a strategy to ensure that overall mercury pollution declines over time. Minnesota proposes that this strategy will be comprehensive (addressing all sources), effective (with reductions in the near term), fair (spreads the burden to all sources), economical (lowest cost, most effective approach), transferable (a model for others) and verifiable (measurable success).

The MPCA has recently received funding from U.S. EPA to develop the initiative. The MPCA is developing a number of possible approaches including the following:

- **Conventional pollution control standards** for industries limiting the allowable mercury emissions.
- **Mercury fees**, which would levy a tax on each pound of mercury released to the environment.
- **Special collection systems** for mercury-containing products.
- **Bans** prohibiting the sale of mercury-containing products.
- **Trading systems** based on market incentives.

2.1.5.2 Co-control

Co-control refers to the control of mercury by control devices or other management measures that were designed or prescribed to limit the emissions of pollutants other than mercury. Examples of co-control discussed below are fuel switching and implementation of the National Ambient Air Quality Standards (NAAQS) for ozone and PM.

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) to achieve required emission reductions in a more flexible or cost-effective way. For example, coal-fired utilities might switch to natural gas during the high ozone season in the Northeast, or to achieve reductions in greenhouse gases. This would also lower their mercury emissions. In addition, 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.

Attainment of Ozone and PM NAAQS. In support of the revised PM and ozone NAAQS, U.S. EPA conducted numerous detailed analyses to predict what control approaches industry might use to achieve the new standards. A number of analyses were performed to predict the impact of the standards on the utility industry. U.S. EPA examined additional NOx and SO_2 control measures for utilities. These included more stringent NOx reductions for the utility cap and trade program in the 37 states in the Ozone Transport Assessment Group, and more stringent SO_2 reductions for the nationwide Title IV utility cap and trade program (acid rain). The control options modeled included several approaches that utilities might take, including hybrid approaches (i.e., more than one approach implemented at the same time or staggered according to season). These actions could include wet scrubber installation, fuel switching, repowering (repowering existing coal-fired plants with cleaner fuel), natural gas replacement (replacing retired coal-fired units with natural gas units) and purchase of emission allowances. When modeling these various approaches, U.S. EPA also analyzed the potential reduction in mercury emissions based on the impact of implementing the national SOx strategy (to meet the PM NAAQS). Based on this analysis, it was estimated that the SOx strategy could achieve at least an 11 ton per year reduction in mercury emissions from utility boilers (U.S. EPA, 1997).

2.1.5.3 Energy Conservation and Renewable Energy

Title IV of the CAA 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 SO_2 emissions avoided through implementing energy conservation measures or by the use of renewable energy. For example, for each ton of SO_2 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.

2.2 Flue Gas Treatment for Sources Other Than Utility Boilers

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 ($HgCl_2$), which is readily adsorbed onto carbon-containing PM. 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 (Schager, 1990).

Add-on controls to reduce mercury emissions, primarily from sources other than utility boilers, are discussed in the sections below (controls specific to utility boilers are discussed in Section 2.3). 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., SO_2, 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.

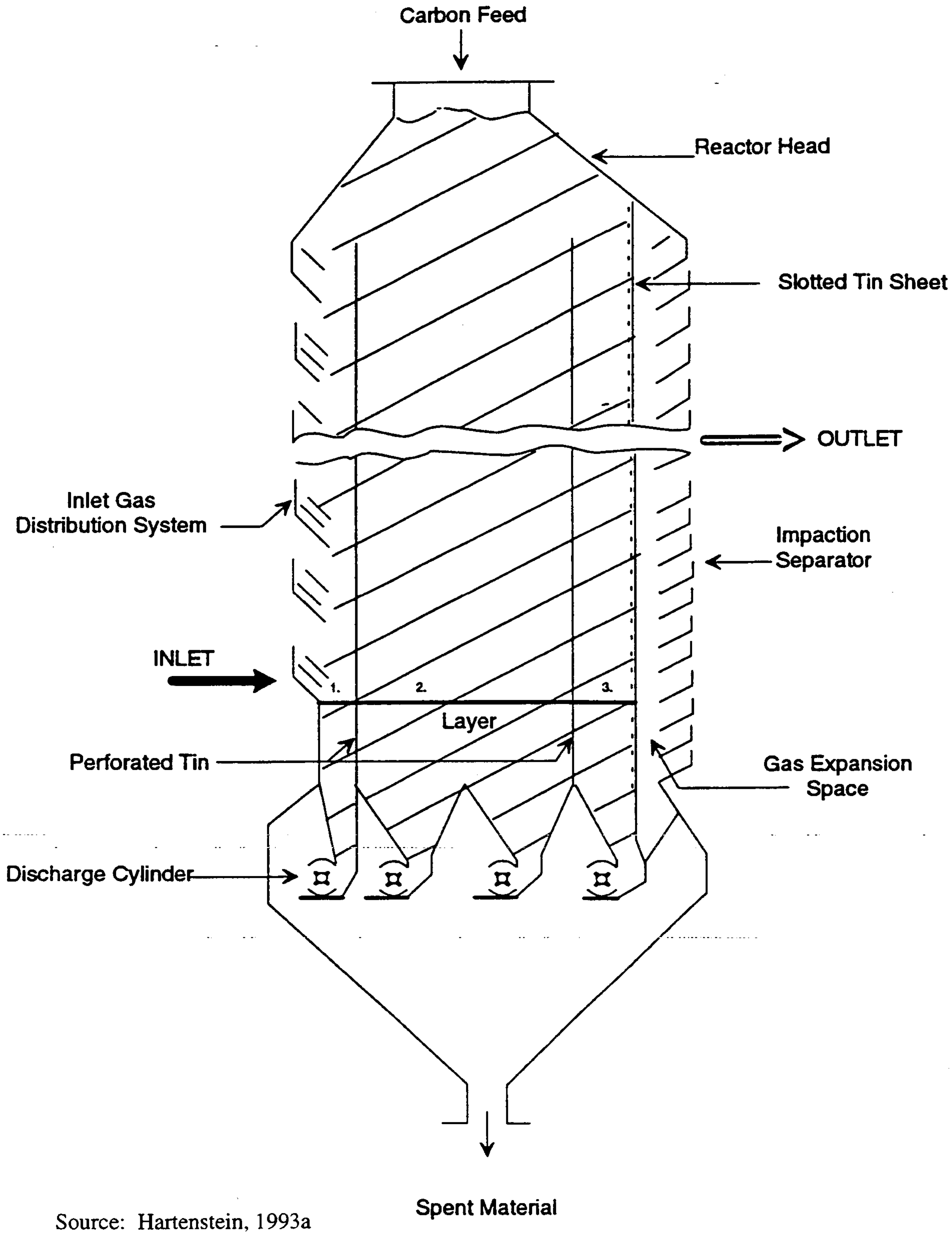
Figure 2-2 presents a basic diagram of the cross flow filters. The flue gas flows horizontally through the filter bed, while the adsorbent migrates through the filter from top to bottom. The bed is approximately 1 meter (m) (3 feet [ft]) thick and is divided into three layers. The thickness of each layer 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 SO_2. 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 190mm (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 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 has been 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

Figure 2-2
Carbon Filter Bed Design



Source: Hartenstein, 1993a

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. In 1994, there were five full-scale applications of carbon filter beds on power plants in Germany. The primary purpose of these filters is to remove residual SO_2 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 SO_2 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.

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[1] At 11 percent oxygen (O_2), on a wet basis.

[2] All concentrations in this report are corrected to 7 percent O_2 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.

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.

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 Use of Wet Scrubbers for Waste Combustors

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 that are installed following other APCD's and are designed for increased pollutant control (often referred to as "polishing" scrubbers).

2.2.2.2 Commercial Status and Performance

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 they have been applied to MWCs in Europe and to MWIs in the U.S. Soluble species of mercury compounds, such as $HgC1_2$ 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). Performance data available for three-stage systems on MWIs 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

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. Two-stage and three-stage wet scrubbers form the basis of the final emission standards for existing MWIs (standards were promulgated August 15, 1997). These scrubbers appear to be highly effective at removing mercury from MWI flue gas. The high chlorine content in medical waste may result in high concentrations of $HgCl_2$, which is efficiently collected by a wet scrubber in this type of application.

2.2.2.4 Secondary Impacts and Benefits

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 dioxin and furan isomers. Dechlorination raises concerns because the lesser-chlorinated dioxin and furan congeners are believed to be more toxic. Shifts in congener profiles have been observed at three wet-scrubber- equipped MWCs in the United States (Lerna, 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 (Krivanek, 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 brine[4] 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 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 (m^3) (at actual O_2 levels) was reported (U.S. EPA, 1973). The treated activated carbon can adsorb from 10 to 20 percent of

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[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.

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).

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[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 [ft^3]) (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.

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[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^{o}C [300 to 400^{o}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 $HgCl_2$, 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^{o}C (200 to 250^{o}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). Mercury is predominately removed upstream of an ESP-equipped facility where a nominal residence time of 1 second or less is available, limiting the capture. In addition, mercury is not effectively collected across the ESP further requiring substantially higher carbon feed rates than the FF-equipped facilities.

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 MWC's and MWI's in Europe and the U.S. It has also been tested on several pilot-scale facilities and slipstreams from utility boilers (0.5 MW_e to 1.0 Mw_e) while firing different coals. 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). 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 Department of Energy/Federal Energy Technology Center and 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 Flue Gas Treatment for Utility Boilers

This section discusses three types of flue gas treatment which have been evaluated to some extent for their effectiveness in removing mercury from utility boiler flue gases. The three technologies are activated carbon injection, wet flue gas desulfurization (FGD), or wet scrubbers, and FGD spray dryers. The effectiveness of these technologies for mercury control vary widely depending on a number of factors. These factors are described in the sections that follow. Current research into the improvment of mercury capture efficiency of these, and other, approaches is described in section 2.4 below.

2.3.1 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 are 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.

The level of mercury control achieved in utility flue gas may depend upon flue gas characteristics such as volume, temperature, fly ash, 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.

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 where the nominal concentration of mercury is one part per billion and may have a wide range of distribution between the different forms of mercury. 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.

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, UNDEERC, 1996, CONSOL INC, 1997.
[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

Recent results from a few pilot-scale studies under different flue gas conditions and APCD configurations are also summarized in this section of the report.

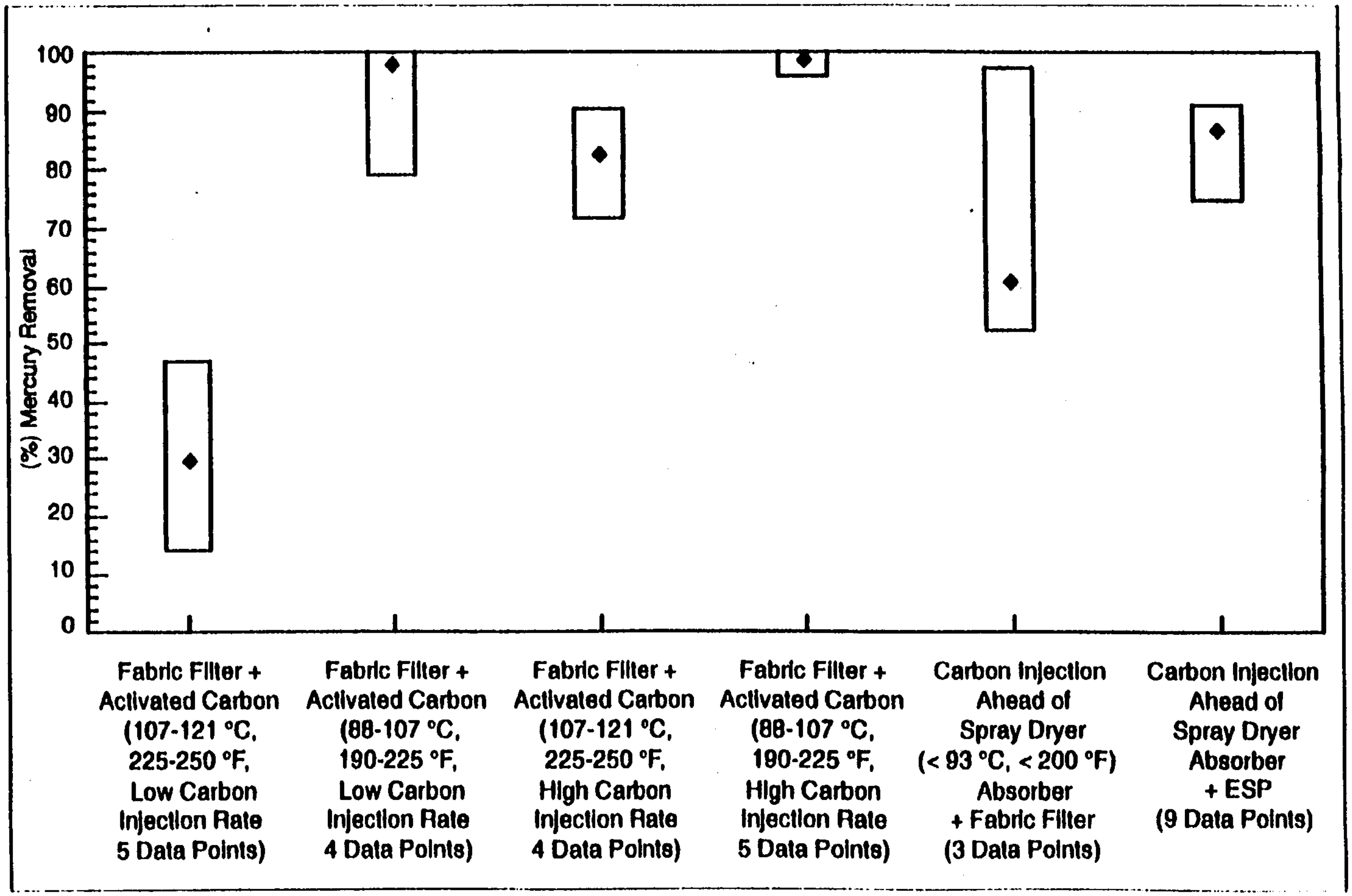
2.3.1.1 Utility Flue Gas Factors Affecting Mercury Removal by Activated Carbon Injection

The level of mercury control achieved in utility flue gas depends on the temperatures upstream and within the existing APCDs, residence time (e.g., extent of contact between the carbon and flue gas mercury) upstream and within the APCDs, volume of flue gas, flue gas vapor and particulate phase constituents (i.e., chlorine as HCl, nitrogen oxides, sulfur oxides, metal oxides on the surfaces of particulate matter, fly ash composition, percent carbon in fly ash, etc.), their interactions with the various types of carbon(s)/sorbent(s), and the mercury concentration and chemical species being formed.

Recent studies indicate mercury capture is mass transfer limited in utility flue gas streams and can be enhanced or suppressed depending on the temperature, flue gas composition and residence time within the flue gas. The reasons for this limitation are the low concentrations of mercury present (one ppb) in the relatively high volumes of flue gas (11,000 - 4,000,000 dscm/min). There are higher concentrations of competing species occupying the active sites of the carbon. In addition, the flue gas residence time upstream of an ESP is nominally one second or less with flue gas velocities in the range of 50 to 60 ft/sec at 149°C (300°F). Compounding the mass transfer limitations is the decrease in the carbon reactivity and capacity at this nominal, but high temperature. Particle size of the activated carbon can also impact mercury mass transfer (Vidic et al, 1996; Flora, et al, 1997; Korpiel, et al, 1997; Liu et al, 1997; Rostam-Abadi et al, 1997; PSCO/ADA et al, 1997; Radian, et al, 1997; Carey et al, June and August, 1997; Waugh et al, August and December, 1997; PSCO/ADA Technologies, Inc., et al, 1997; and Haythornthwaite et al, 1997). These factors are reviewed below.

Figure 2-3

Comparison of Mercury Removal Efficiencies with Activated Carbon Injection



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

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.

Earlier data provide some 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).

These studies indicate 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.

Recent data collected from some coal-fired facilities utilizing either pilot-scale FFs or ESPs further indicate an apparent temperature dependence on mercury removal. The FF and ESP pilot-scale studies indicate an increase of mercury removal with the native fly ash without carbon injection. Further increases of mercury removal with carbon injection during lower temperature operation were also indicated. The studies without carbon injection showed measured elemental mercury removals across a pilot-scale pulse-jet filter (air to cloth ratio of 4 ft/min) of 10 and 17% at 135°C (275°F) and 65% at 121°C (250°F); 67% at 93°C (200°F), across a pilot-scale reverse-gas baghouse of less than 20% for an average temperature of 143°C (289°F) , and upstream of a pilot-scale ESP of mean average of 30% at 93 - 109°C (200 - 228°F) for the native fly ash (nominal <0.5% carbon in ash) from the combustion of a PRB Belle Ayre coal (PSCO/ ADA Technologies, Inc.,et al, 1997, Sjostrum et al, 1997; and Haythornthwaite 1997).

In contrast to the higher mercury removals at lower temperatures are data collected from a full-scale utility boiler without carbon injection. The testing was conducted on a 70 MW_e unit firing a Powder River Basin coal from the Montana area in a Riley Stoker front-fired boiler. The only APCD is a reverse-gas baghouse for particulate control. Mercury measurements were taken at the inlet and outlet of the baghouse with triplicate samples being collected and analyzed for total mercury, including speciation. Draft U.S. EPA Method 29 and the Bloom or MESA method were utilized simultaneously at each location. Both methods measured total inlet mercury concentrations (three data points for each method) at the 6.4 and 6.5 $\mu g/m^3$ levels respectively, with approximately 60 percent of the total being measured as elemental mercury for each method. The elemental mercury was essentially removed across the baghouse due to the native fly ash (during the three test periods the percent carbon was 3.5, 2.9, and 2.9 with an average of 3.1 percent) with the outlet concentrations being 2.6 and 3.1 $\mu g/m^3$ of the ionic form as measured by the respective methods. The removals indicted by the two methods were 60 and 52 percent of the total, respectively at average temperatures (three data points each) at the air heater outlet of 189°C (372°F), baghouse inlet of 174°C (346°F), and baghouse outlet of 166°C (330°F). Approximately 40 percent of the total mercury was indicated on the filter catch of the Method 29 train [filter at 121°C ± 8°C (250°F± 15°F) which could capture the mercury as it comes in contact with the filtered fly ash] and the hopper ash samples indicated a high level of mercury comparable to the removals. The mercury capture during this testing was indicative of removals across the baghouse and not in-flight capture upstream of the baghouse. (Jackson et al, 1994)

As indicated, the mercury removals of the native fly ash at these conditions are not typical of the past and more recent field characterizations and pilot-scale mercury technology investigations. This utility site is proposed to be further characterized in mid-1998 with the more precise Ontario Hydro mercury speciation method. In addition, in-flight capture of mercury will be investigated upstream of the baghouse along with the baghouse removals (DOE/FETC et al, Phase II 1997). Currently, laboratory tests are being conducted on the fly ash under simulated flue gas conditions to provide some insight on the factors influencing high elemental mercury capture at nominal flue gas temperatures of 149℃ (300°F), (U.S. DOE/FETC R&D, 1997).

Typical removals of mercury by the fly ash for low-sulfur and medium- to high-sulfur bituminous coals under the above conditions is approximately 10% or less and can be influenced by the sampling method. The fly ash is captured on a filter of the sampling train at 121℃ (250℉ which is lower than the flue gas) before the chilled impinger based solutions being utilized for the collection of the vapor phase mercury. The passing of the flue gas through the captured fly ash on the filter can provide false indications of in-flight capture of mercury. As indicated, the removals of mercury assumed from the fly ash in-flight can be inflated due to the sampling method, but still in most cases are below 10% (Miller 1994 and 1995; EPRI, 1994; U.S. DOE Report, 1996; Laudal et al, 1996 and 11& 12, 1997; Hargis et al, 1996; Redinger, et al, 1997; Holmes, et al, 1997; Waugh, et al, 1997; and Devito et al, 1997).

The pilot-scale activated carbon injection studies 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 may be 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 may be avoided provided low-sulfur coals (less than about 1 weight percent sulfur) are burned, but it may depend on whether the coal is a subbituminous or a bituminous coal (McKenna and Turner, 1989; ABB et al,1996 & 1997; PSCO/ADA Technologies, Inc.,et al, 1996 & 1997, Sjostrum et al, 1997; and Haythornthwaite 1997; Radian et al, 1997; Carey et al, 1996 and 1997; Radian et al, 1997; Hargrove et al, 1997; Waugh et al, 1997). If a utility burns low-sulfur coal and uses an ESP for particulate control, however, the flue gas will probably require conditioning to reduce the high resistivity of the fly ash because high resistivity makes the fly ash difficult to collect with an ESP, but again, it is dependent on coal type.

Further research is needed to evaluate humidification in flue gas ducts while firing other low-sulfur coals and most importantly medium- to high-sulfur coals in the furnace. This is extremely important for the approximately 65% of the utility industry utilizing an ESP as the only APCD. Subsequent sulfuric acid mist formed from the condensation of sulfur trioxide below the acid dew point(s) can be extremely detrimental to ESP- and FF-equipped utilities, duct work, all downstream equipment, compliance for opacity, and plume effects (i.e., visibility - blue plume). In addition, it is desirable for utilities to minimize the amount of sulfuric acid being emitted as these emissions must be reported annually to the Toxics Release Inventory.

In some cases, lower temperatures do have an influence on the amount of mercury removed by certain native fly ashes alone and in combination with activated carbon, but this not typical of the utility population (e.g., majority of low- and medium-sulfur bituminous coals). The factors or mechanisms influencing the ability of the small percentage of coals and subsequent fly ash to adsorb mercury and/or convert mercury from one form to another in-flight and across fabric filters need to be further investigated in order to effectively capture the different forms mercury. These mechanisms can be associated with the type of activated carbon, fly ash components, the vapor phase chemical species of the flue gas, and all the possible interactions, along with the control device being augmented to remove mercury. These factors are not fully understood at this time, but many research organizations are performing fundamental and applied research studies to investigate and subsequently understand them.

Based upon the preliminary pilot-scale studies conducted at temperatures below 121°C (250°F), the least efficient and most costly 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 O_2). High mass carbon-to-mercury ratios will be required due to a nominal one ppb of mercury being in different forms and being in the high flue gas volumes with competing vapor phase compounds at many orders of magnitude higher. Currently, mercury mass transfer limitations are encountered regardless of the type of coal, operating conditions, and APCD.

Mercury Speciation and Type of Activated Carbon. With a few exceptions, the total mercury concentration in coal is relatively constant across the U.S. (20 ppb to 120 ppb). However, when the different coals are fired in a combustor there is substantial variation in the concentrations of elemental versus ionic mercury. The percentage of Hg^0 is from near zero percent to >70%. The speciation then is very dependent on coal type. The chemical species of mercury formed during the combustion process and post-combustion conditions vary significantly from one plant to another. While combustion conditions vary, the subsequent fly ash, carbon in the ash, and vapor phase constituents may play a major role in the percentage of the chemical species of mercury formed. Understanding the rate controlling mechanisms (i.e. transport, equilibrium, and kinetics) will aid in predicting the species formed and eventually to aid in optimizing existing APCDs for mercury removal. Kinetics may play more of a role on the form of mercury than anticipated. Depending on the type of coal utilized, effective removal maybe dependent on the species of mercury present in the flue gas.(Senior et al, June and November, 1997; and PSI et al, 1997) For example, the ionic mercury form (i.e., Hg^{++}) is water soluble and is less volatile than elemental mercury (i.e., Hg^0). Thus, reducing the temperature of the flue gas and wet scrubbing of the flue gas may result in increased ionic mercury removal.

In the early 1990s EPRI and DOE initiated very extensive electric utility air toxics characterization programs. As part of these programs, speciated mercury emissions were attempted to be measured from each plant. Because there was no validated mercury speciation sampling method, U.S. EPA Method 29 and the Bloom or Brooks Rand (to be referred to as the MESA) methods were used. The results from these characterizations strongly suggested that U.S. EPA Method 29 does not properly speciate mercury under certain conditions. In addition, there were questions as to the ability of the MESA method to speciate mercury in flue gas from coal combustion. Results from the MESA sampling method and unique analytical technique(s) are summarized in Table 2-3 for coal- and oil-fired utility flue gas (Bloom et al., 1993).

As shown in Table 2-3, the distribution of ionic mercury, most likely $HgCl_2$ 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

Table 2-3
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 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
2.99	---	1.51	4.5	66.44	33.56	g
2.83	---	0.88	3.71	76.28	23.72	g

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

μg/Nm^{3a}			Total[b] (μg/Nm^3)	Percent		Reference
Ionic	Other Hg Species[c]	Elemental		Total Ionic[d]	Elemental	
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
5.01h	---	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
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	--

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

µg/Nm^{3a}			Total[b] (µg/Nm^3)	Percent		Reference
Ionic	Other Hg Species[c]	Elemental		Total Ionic[d]	Elemental	
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	
OIL-FIRED BOILER DATA RANGES						
0.02 - 0.03	---	0.02 - 0.15	0.12 - 0.19	23	77	---

[a] µg/Nm3 = 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.

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).

Since that time a substantial amount of work has been done to develop sampling and analytical methods for determining mercury speciation in flue gas from fossil fuel combustion. In 1994 EPRI and DOE contracted with the University of North Dakota Energy & Environmental Research Center (UNDEERC) to complete a series of bench- and pilot-scale evaluations on mercury speciation measurement methods. Concurrently, work was also being conducted by CONSOL, Inc., Radian International, Advanced Technology Systems, and Babcock & Wilcox at the bench- and pilot-scales, along with full-scale coal-fired power plant studies and characterizations.

In the pilot-scale work conducted at EPRI's ECTC by Radian International and the pilot-scale work conducted by the UNDEERC for both EPRI and DOE, it was proven that U.S. EPA Method 29 does not properly speciate mercury under certain conditions (Hargrove et al, 1995; Laudal et al, 1996; Stouffer et al, 1996; Khosah, et al, 1996; and Laudal et al, December, 1997). Further studies at UNDEERC indicated this finding is related to high SO_2 concentrations with the method overestimating the ionic mercury up to 50%. Therefore, tests were conducted to evaluate a number of alternative sampling methods. Mercury speciation sampling methods that have been tested include the following:

- U.S. EPA Method 29
- Mercury Speciation Adsorption Method (Frontier Geosciences and Brooks Rand - the Bloom method)
- Ontario Hydro method (Ontario Hydro)
- Tris-buffer method (Radian International)
- U.S. EPA Draft Method 101B (Research Triangle Institute)

Bench- and pilot-scale studies also showed that the MESA method did not speciate mercury correctly when tested with coal-fired flue gas. The method is greatly affected by an interaction between SO_2 and NOx in the flue gas. When SO_2 is present in concentrations >500 ppm and NOx >250 ppm the MESA method can overestimate the ionic mercury fraction up to 75%, (Laudal et al, 1996). Based on the exploratory pilot-scale tests, the Ontario Hydro method and U.S. EPA Draft Method 101B were selected to be more formally evaluated using the protocol established in U.S. EPA Method 301. However, because there is no reference method to compare to U.S. EPA Method 301, the method only provides the precision and bias associated with the sampling procedures. To obtain the accuracy of the speciated mercury measurement methods, it was necessary to do dynamic spiking of the flue gas stream. Spiking was done first with elemental mercury then with $HgCl_2$. Results showed that both the Ontario Hydro and U.S. EPA Draft Method 101B passed the U.S. EPA Method 301 criteria, however the Ontario Hydro method showed much less variability than Method 101B. Therefore, the Ontario Hydro method is being recommended by DOE as the best method to speciate mercury in coal-fired systems. The method is being submitted to the American Society for Testing and Materials and U.S. EPA for approval.

Field tests comparing U.S. EPA Method 29 and/or the MESA method, with either or both the Ontario Hydro method and the tris-buffer method have been completed during 1995 through 1997. Results showed that U.S. EPA Method 29 and the MESA method gave a high bias for the ionic form of mercury compared to the Ontario Hydro and tris-buffer methods, which is in agreement with the Radian International and UNDEERC pilot-scale studies. DOE and EPRI are planning field studies and characterizations on mercury speciation with the Ontario Hydro method.

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; Laudal et al, 1996 and 1997; Redinger et al, 1996 and 1997; and Devito et al, 1997).

Low-sulfur bituminous coals and other subbituminous coals with low-sulfur content are very different regarding the mercury distribution between the elemental and oxidized forms in the flue gas. (Bloom et al, 1993; DeVito et al, 1993; EPRI, 1994; Prestbo et al, 1995; U.S. DOE Report, 1996; Laudal et al, 1996 and 11 & 12, 1997; Pavlish et al, 1997; Hargrove et al 1997; Senior et al, June and November 1997; PSI, et al., 1997; and Devito et al, 1997) The fly ash characteristics are extremely different and some of the subbituminous coals produce fly ash that are more reactive and adsorb mercury at higher rates than fly ash from the bituminous coals. In addition, the bituminous coals convert the elemental mercury at higher rates and levels as compared to the fly ash from subbituminous coals. The adsorption and/or conversion is impacted by temperature, but the composition of the fly ash and vapor phase compounds also play a major role in these effects (Miller 1994, and 1995; Laudal et al, 1996 and 11 & 12, 1997; Carey et al, 1996 and 1997; Radian International, et al, 1997; Senior, et al, June and November 1997; and Devito et al, 1997).

Radian International conducted both laboratory and field studies to investigate catalytic oxidation of vapor-phase elemental mercury in coal-fired utility flue gas streams. Catalytic oxidation of vapor-phase elemental mercury can potentially increase the total mercury removal in the two technologies with the most potential for removing mercury from flue gas: wet scrubbing and sorbent injection. To investigate this process, potential catalyst materials were tested using three different test configurations. These configurations included laboratory fixed beds tests, pilot-scale fabric filter tests, and sample filter tests using flue gas from a full-scale utility.

Oxidation of elemental mercury using catalyst materials was successfully demonstrated using each of the test configurations mentioned above. In the laboratory fixed bed tests, the effect of temperature and flue gas composition were investigated. In general, oxidation of elemental mercury decreased as the temperature increased. Flue gas composition also appears to be important to oxidation, with HCl and possibly NO_x affecting oxidation.

Based on the laboratory and pilot-scale tests, the most successful catalyst was a carbon-based material. After injecting about 20 pounds of this material into a pilot-scale fabric filter, greater than 75% of the inlet vapor-phase elemental mercury was oxidized across the fabric filter for 10 consecutive days. Similar results were obtained at a full-scale facility by measuring oxidation across a sample filter. These results confirmed the ability of the carbon-based material to oxidize elemental mercury under different flue gas conditions (with and without HCl and various levels of NO_x). Other catalyst materials that were identified and warrant further investigation, included several iron-based materials, a conventional SCR catalyst, and some fly ash samples (Carey et al, 1996 and 1997; Radian International, et al, 1997).

The speciation of mercury is extremely important in planning control strategies, but it is still in the early stages of investigation. Preliminary laboratory- and field pilot-scale studies indicate the form of mercury being removed is impacted by the type of carbon being injected. Both physical and chemical adsorption of the mercury can be achieved, but is dependent on the concentration and most importantly the form of mercury (elemental or ionic/oxidized). Limited studies have indicated simultaneous removal of both forms of mercury with one activated carbon, but at very low levels. A further complication is some activated and chemically impregnated activated carbon can, under certain conditions, convert the elemental mercury to an ionic form with either a net increase or decrease in mercury capture (Miller, 1994 and 1995; PSCO/ADA Technologies, Inc., 1997; and Radian et al, 1997).

Earlier studies with activated and chemically impregnated activated carbon utilized either U.S. EPA Method 101A (only total mercury) and either U.S. EPA Method 29 or the MESA method (both for speciated mercury as well as total) for the mercury measurements. As indicated from the studies conducted at the UNDEERC, these two speciated methods have overestimated the ionic form of mercury up to 50% and 75% respectively. The interactions of these carbons with the fly ash and vapor phase species in the flue gas can dramatically increase or decrease mercury capture of the carbon, and measuring the impacts are difficult and sometimes impossible to do. In addition, controlled laboratory studies were conducted with the injection of activated carbon(s) and elemental mercury or $HgCl_2$ in either nitrogen or simulated flue gas streams. The results indicated different and varying levels of mercury capture between the nitrogen and simulated flue gas streams. Promising results from these tests, in most cases, have not been repeated on actual flue gas streams of the pilot-scale and slipstream studies at the various coal-fired facilities.

More recent tests have been conducted on flue gas streams containing primarily elemental mercury that was often supplemented with additional elemental mercury during testing. The tests were designed to investigate elemental mercury capture with commercially available activated carbons. Limited studies have been conducted on chemically impregnated carbons, but they are being considered for future testing on both simulated and actual flue gas.

Several types of novel activated carbons for gas phase elemental mercury removal that have orders of magnitude higher saturation capacities when compared to virgin activated carbons are also available. These activated carbons are typically impregnated with sulfur or iodine lending to the enhanced capacity for mercury uptake due to the chemical reaction between the impregnated material and elemental mercury. However, many of the sorbents exhibited deteriorated performance at temperatures typical of coal-fired power plant operations.

Recently, researchers at the University of Pittsburgh developed a series of sulfur-impregnated carbons that exhibited high elemental mercury uptake efficiency at 140°C (284°F) when compared to commercially available activated carbons. Dynamic adsorption capacity of these carbons as high as 4000 μg Hg/g was measured using a fixed-bed absorber with an empty bed contact time of 0.011 second and influent mercury concentration of 55 $\mu g/m^3$. This capacity is almost three orders of magnitude greater than the capacity of virgin activated carbon and an order of magnitude greater than the capacity of commercially available impregnated activated carbon. The comparisons were conducted at identical operating conditions using nitrogen as a carrier gas.

The increased performance is attributed to the impregnation of the carbon(s) with sulfur at elevated temperatures of 400 - 600°C (752 - 1112°F). This promoted a more uniform distribution of short linear chains of sulfur allotropes (S_2 and S_6) on the carbon surface as opposed to having predominately S_8 rings condensed in the macropore region of commercially available sulfur impregnated carbons. In addition, the sulfur impregnated carbons prepared at elevated temperatures exhibited significantly better thermal stability since no sulfur loss was observed even after exposure at 400°C (752°F) (Vidic et al, 1996; Korpiel, et al, 1997; Flora, et al, 1997 and Liu et al, 1997).

These impregnated activated carbons exhibited orders of magnitude higher dynamic capacity as compared to virgin activated carbons. However, the key question remains as to whether this capacity can be utilized in a flue gas stream where residence times of one second or less are available for injection upstream of the ESP- equipped facility. These high capacity carbons may be limited to use on FF-equipped facilities or control strategies employing devices for higher flue gas and carbon contact or residence times. The costs associated with impregnated activated carbons may also limit their use to FF-equipped facilities.

Further investigation, development, and enhancement of activated carbons and chemically impregnated carbons for mercury capture in flue gas from coal-fired facilities is needed. The conditions of the chemical impregnation may be critical and commercially available impregnated activated carbons may not be highly effective in all the various flue gas produced from the combustion of coal. New virgin and chemically impregnated activated carbons may need to be developed for the highly variable and complex flue gas streams encountered in the utility industry and the extreme mercury mass transfer limitation(s).

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.

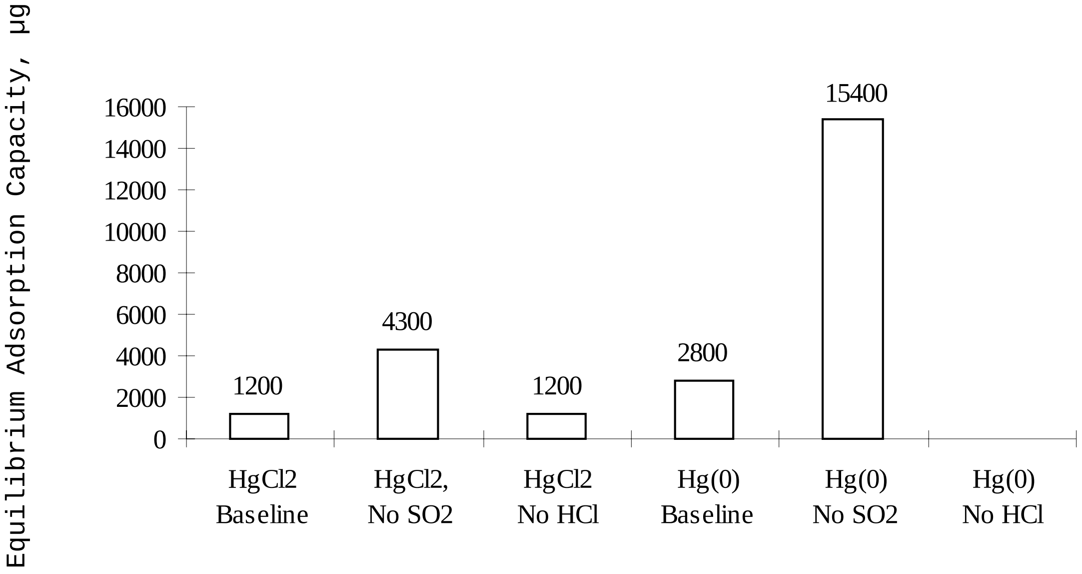
Studies of a pilot-scale wet FGD system treating coal-fired flue gas indicate that more than 90 percent of the ionic mercury was removed while hardly any of the elemental mercury was removed (Noblett et al., 1993; Redinger et al,1996 and 1997; Carey et al, (2) 1996; Evans et al, 1996; and Hargrove et al, 1995 and 1997). 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 from the studies described in Section 2.3.1.2, Current Research on Activated Carbon Injection for Utilities.

Flue Gas Composition. The temperature, volume of the flue gas, and type of activated carbon can have an impact on the form and subsequent capture of mercury in coal-fired produced flue gas streams. These factors are not independent of one another, but are synergistic with one another and are very dependent on the composition of flue gas. This includes both the vapor and particulate phases of the flue gas. As previously indicated, hydrogen chloride, sulfur and nitrogen oxides, oxygen, water, fly ash and its composition, and even carbon monoxide in the flue gas can either impede or enhance the form and subsequent capture of the mercury with fly ash and injected carbon. There are other flue gas constituents that could also impact mercury collection, but research is needed to determine what other constituents do and why.

A recent bench-scale study investigated the effects of SO_2 and HCl on the adsorption of elemental mercury and mercuric chloride ($HgCl_2$) by a lignite-based activated carbon (Carey et al., 1997). Equilibrium adsorption capacities were determined for fixed beds of the carbon at 275°F and three flue gas compositions: one containing 1600 ppm SO_2 and 50 ppm HCl (the baseline composition); a second containing no SO_2 and 50 ppm HCl; and a third containing 1600 ppm SO_2 and no HCl. (All three compositions of flue gas had the same concentration of elemental mercury, mercuric chloride, CO_2, water, and O_2).

Figure 2-4 illustrates the effect of SO_2 and HCl on the equilibrium adsorption capacity of the lignite-based activated carbon for elemental mercury and mercuric chloride. Removing SO_2 from the flue gas increased the equilibrium adsorption capacities for both kinds of mercury (compared to the baseline capacities). The increase was particularly notable for the adsorption of elemental mercury. For example,

Figure 2-4
Equilibrium Adsorption Capacity of Elemental Mercury (Hg(0)) and Mercuric Chloride (HgC12) by a Lignite-Based Activated Carbon



after removing SO_2 from the flue gas, the equilibrium adsorption capacity for elemental mercury increased by a factor of about 5.5 compared to 3.5 for mercuric chloride.

Removing HCl from the flue gas did not affect the equilibrium adsorption capacity of the carbon for mercuric chloride, however, it did prevent the carbon from adsorbing elemental mercury. The latter result suggests that HCl participates in the adsorption mechanism of elemental mercury when using a lignite-based activated carbon and that the adsorption mechanism is not purely physical, i.e., interactions between elemental mercury and HCl on the carbon surface may be important.

The results from Figure 2-4 indicate that flue gas composition affects carbon performance. With no HCl in the gas, the carbon adsorption capacity for mercuric chloride was larger than that for elemental mercury. This result is opposite to that observed at baseline conditions where the carbon adsorption capacity for elemental mercury was larger than that for mercuric chloride. The results from Figure 2-4 also indicate that performing carbon adsorption tests under realistic operating conditions is important. Many bench-scale carbon tests in the past have been conducted using nitrogen as the carrier gas. Tests conducted in nitrogen could produce different results than tests conducted in simulated flue gas; however, the effect of SO_2 and HCl on adsorption capacity could also be sorbent dependent. Other carbons may not be affected by the presence of HCl and SO_2 if the mercury adsorption mechanism is different.

Further details on the effects of flue gas components, including the interactions with fly ash, can be obtained from two references: (Laudal et al, November, 1996 and December, 1997). The flue gas and mercury chemistries and their subsequent interactions need to be fully understood at the various flue gas conditions encountered across the utility industry for effective low cost mercury strategies to be universally realized.

2.3.1.2 Current Research on Activated Carbon Injection for Utilities

Previously, research was conducted on activated carbon injection at a facility with a pilot-scale SDA/ESP system in Denmark (Felsvang et al, 1993); at a facility with both a pilot- and full-scale SDA/FF system by Joy/Niro and Northern States Power (Felsvang et al, 1993); at a pilot-scale coal combustor and FF by Miller et al., 1994 & 1995; and at a pilot-scale pulse-jet FF system at a utility power plant by EPRI (Chang et al., 1993). These results are presented in detail in Appendix A. Preliminary results are available from the first three studies as described below.

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, laboratory-scale tests at the University of North Dakota Energy & Environmental Research Center (EERC) found that for some conditions iodine-impregnated carbon is much more effective than lignite-based activated carbon in removing elemental mercury (Miller et al., 1994). Sorbent injection tests were conducted at flue gas temperatures ranging from 125 to 200°C (257 to 392°F). Iodine-impregnated carbon had a high removal efficiency of elemental mercury (greater than 95 percent removal) across the entire range of temperatures for one subbituminous coal. However, for a second subbituminous coal the iodine-impregnated carbon appeared to convert the elemental mercury to ionic mercury with little net total mercury removal. A reason for the difference is not obvious, but may be the result of differing concentrations of SO_2, HCl, NOx, HF, and possibly CO. Lignite-based activated carbon removed approximately 50 percent of elemental mercury at 130°C, however, it's removal efficiency for elemental mercury dropped dramatically as temperature increased. For both carbons, the removal efficiency of oxidized mercury was highly temperature dependent. At 125°C, the iodine-impregnated carbon was somewhat effective at removing oxidized mercury, while it removed no oxidized mercury at 175°C. The lignite-activated carbon showed a similar trend (Miller et al, 1994 and 1995).

The most recent studies have utilized American Norit Companies' commercially available Darco FGD activated carbon developed from a lignite coal. This carbon has been extensively utilized more than any other commercial activated carbon for the DOE and EPRI-funded mercury control studies investigating sorbent injection. (Miller et al, 1994 and 1995; Chen et al, 1996; Hunt, 1996; ABB et al, 1997; Carey et al, July, 1996 and June, 1997; Radian International, et al., 1997; Sjostrum, et al, 1997; Haythornthwaite et al, 1997; PSCO/ADA, et al, 1997; Rostam-Abadi et al, 1997; Waugh et al, August and December, 1997; and Brown, 1997.) The activated carbon typically has a mass mean diameter of 15 microns, a BET surface area of 600 m^2/g and a nominal equilibrium adsorption capacity of 500 μg Hg/g C. These parameters have been repeated by many research institutions and are in agreement with Norits' specifications (Carey et al, 1997; Radian international, et al., 1997; Haythornthwaite et al, 1997; Waugh et al, 1997; and Rostam-Abadi et al, 1997).

The equilibrium adsorption capacity of the activated carbon is important for fabric filter systems. For flue gas residence times of less than one second, typical upstream conditions prior to the inlet of an ESP, the equilibrium adsorption capacity of 500 μg Hg/g C may not be the most critical parameter. Reactivity may need to dominate, but can be suppressed at the nominal temperature of 149°C (300°F) of the flue gas upstream of utility ESPs. Chemically impregnated carbons may increase the reactivity and subsequent capture of mercury, but very few studies have indicated the effectiveness of chemically impregnated carbons for in-flight capture of mercury (especially at one second or less residence time) (Vidic, et al, 1996; Korpiel, et al, 1997; and Liu et al, 1997).

The chemically impregnated carbons may be cost prohibited and may be better suited for high mercury adsorption capacities corresponding to longer contact times (carbon and novel fluid beds or fabric filters - reverse-gas and pulse-jet with the pulse-jet also being downstream of an existing ESP). Examples of this technology are EPRI's **CO**mpact **H**ybrid **P**articulate **C**ollector (COHPAC) or TOXICON (a pulse-jet baghouse operating at a high air-to-cloth ratio downstream of the primary particulate control device with sorbent injection upstream of the baghouse for air toxics or in these cases mercury).

Recent studies further support the mercury mass transfer limitations since the removal of mercury above 50% to the 90% level for in-flight capture and above 75% to the 90% for extended contact times (>one half hour across a fabric filter) is dependent on near exponential increases in the carbon injection or carbon to mercury ratios. (Vidic et al, 1996; Flora, et al, 1997; PSCO/ADA et al, 1997; Carey et al, June and August, 1997; Korpiel, et al, 1997; Liu et al, 1997; Rostam-Abadi et al, 1997; and Waugh et al, August and December, 1997). The PSCO/ADA studies indicate a nominal 5000:1 carbon-(Norit or Darco FGD)to-mercury mass ratio at 106℃ (222℉) upstream of an pilot-scale ESP with a residence time ranging between 0.75 and 1.5 seconds to remove the mercury at a level of 48%. This 48% includes 30% of the mercury being removed by the native fly ash. Studies have indicated the fly ash from this PRB coal (Comanche or Belle Arye coal from Wyoming) has a high equilibrium adsorption capacity for mercury even at <0.5% carbon levels in the fly ash (Miller et al, 1994 and 1995; Laudal et al, 1996 and 11 & 12, 1997; Haythornthwaite et al, 1997; and PSCO/ADA et al, 1997). This mercury removal in-flight is high compared to other PRB and subbituminous coals. The overall mercury adsorption can be higher than bituminous coals for the same amount of carbon in the fly ash. The adsorption capacity or reactivity for both ranks of coal does increase with a decrease in temperature, but not at the same rate or level. In addition, tests were conducted with the re-injection of the Commanche fly ash upstream of the ESP configuration and indicated on average less than 10 percent mercury capture.

The pulse-jet pilot-scale FF tests at the PSCO facility also indicated a substantial increase in carbon injection or mass carbon-to-mercury ratio from 76% mercury removal at a ratio of >20,300:1 (C/Hg) to >90% mercury removal at a ratio of >36,600:1. Mercury concentrations were not constant at these ratios with nearly 18% mercury reductions being attributed to residual fly ash on the bags. These tests were conducted as “clean” tests, that is, no fly ash was in the flue gas stream (the flue gas was drawn downstream of the facility’s existing fabric filter). During the testing with fly ash present, different results were indicated. The mercury removal “by the fly ash” was dramatically impacted by temperature. At temperatures between 93°C (200°F) and 121°C (250°F) mercury removals due to the fly ash were at 66% while an increase to 135°C (275°F) indicated removals in the range of only 10% to 17%. In addition to the fly ash removals, the amount of carbon needed at even small increases in temperature was noticeable. Carbon to mercury ratios of 3400:1 were needed for mercury removals of 74% at only 109°C (228°F) while ratios of >8700:1 were needed to remove mercury at 87% for a temperature of 113°C (236°). The mercury concentrations were steady during these tests.

These data were collected at the same contact times (carbon exposed to flue gas across the fabric filter) and the QA/QC on the mercury sampling methods were indicative of the close mercury concentrations for all the tests at the close, but different temperatures. The adsorption of the mercury appears to be mass transfer limited even at high residence or contact times. In addition, the high mercury removals include the 66% mercury removed by the fly ash (Sjostrum, et al, 1997; Haythornthwaite et al, 1997; and PSCO/ADA et al, 1997). If this type of fly ash was not present, the mass carbon-to-mercury ratios could be much higher as indicated at the tests at the Public Service Electric and Gas Company’s Hudson station (Waugh et al, August and December, 1997).

These data indicate mercury removals at greater than 90%, but the mass of carbon-to-mercury was still between 20,000:1 and 50,000:1 (116°C or 240°F) for a pulse-jet at an air-to-cloth ratio of approximately 12 ft/min (in this case EPRI’s COHPAC or TOXICON). ESP pilot-scale tests indicated mercury removals of 83% at 105°C (221°F) and a mercury removal of 35% at 133°C (272°F) at the same mass carbon-to-mercury ratio of 45,000:1. Low-sulfur Eastern bituminous coal was fired at the utility and the fly ash mercury removals across the range of temperatures was a nominal 15% (Waugh et al, August and December, 1997).

Mercury mass transfer limitation(s) may be dominant under these most recent field pilot-scale studies. Small deviations in the temperature indicate an increase in carbon needed to maintain even low levels of removal with fabric filters and most indicative, upstream of an ESP with or without flue gas cooling. Optimizing is not the issue at this time. Research is needed and the high mass carbon-to-mercury ratios may not be cost effective, based on the recent data on carbon injection for mercury removal. The data presented in 1993 by EPRI (Chang et al, 1993) were extremely innovative, but since then many improvements have been made to aid in the collection and interpretation of the data. The methods to measure mercury were not at the level of todays standards and the fly ash, based on the recent tests at the Comanche Station can account for close to 65% of the mercury removal. Data have been presented that the fly ash alone can remove >90% of the mercury across the Station's existing reverse-gas baghouse. This is not typical of the majority of the fly ashes collected in the utility industry. The recent PSCO data is collected at the same facility as the 1993 data. The mass carbon-to-mercury ratios are higher than indicated in the 1993 work.

Mass carbon-to-mercury ratios of >100,000:1 maybe required at one second or less residence time upstream of an ESP at 149°C (300°F) in order to achieve 90% mercury removal. The scenarios for the ESPs may require fabric filters downstream. The fabric filter of choice would probably be a pulse-jet filter operating at a high air-to-cloth ratio.

A reverse gas fabric filter is an option in the cost of control models in Appendix B being utilized downstream of an ESP for mercury capturing the injected carbon being used for mercury removal. A more compact pulse-jet filter could be utilized for mercury removal and this option would also be effective for collecting the fine particulate escaping the upstream ESP (e.g., EPRI's COHPAC or TOXICON). Further research is needed to verify this. If the ESP is 98.5 to 99 % efficient (greater than the 0.03 lb/MMbtu NSPS limit), then a considerable amount of particulate (less than 5 microns) will accumulate or be collected with the injected activated carbon. This is a benefit, but it could have an impact on pressure drop and cleaning frequency of the pulse-jet. This could limit the utilization of the carbon for mercury capture and the increase of pressure drop would require additional fan power. If the size of the pulse-jet is at the levels requiring a higher air to cloth ratios between 6 and 8 ft/min or higher, the pressure drop would increase in a shorter period of time requiring more frequent cleaning and subsequently the mercury capture would decrease per unit mass of carbon injected due to less contact time. There are currently problems with pulse-jet filters as a polishing device while cleaning on line for the fine particulate (reentrainment of the fine fly ash) since there is not an adequate dust cake formed. Humidification may help, but it has just been tested under this type of application (Waugh et al, December 1997). The reentrainment issue could further complicate the problem and demand additional costs for taking the filter off-line. A design could be provided to recirculate the under utilized carbon and fly ash mixture, which would require an additional cost of handling of the solids and re-injection. If there is no recirculation of the carbon collected in the hoppers, then more carbon would be needed than anticipated. These concepts or designs are in their infancy and data still needs to be collected and carefully interpreted.

The Department of Energy Federal Energy Technology Center and the Electric Power Research Institute are planning to conduct several pilot-scale field studies at different utility sites, with possible full-scale demonstrations. Before the use of activated carbon for mercury removal is cost effective in the coal-fired electric utility, a large collaborative effort, the collection of the data and its interpretation from all the fundamental, laboratory-, bench-, and pilot-scale being performed must be realized.

2.3.2 Flue Gas Desulfurization (FGD) Scrubbers

Wet FGD systems are currently installed on about 25% of the coal-fired utility generating capacity in the U.S. (Redinger et al., 1997). Although their primary function is to remove SO_2 emissions from boiler flue gas, wet FGD systems can also be effective in removing mercury emissions from boiler flue gas. The mercury removal efficiencies of wet FGD systems can vary widely depending on the mercury species in the incoming flue gas, the design and operation of the wet FGD system, and reactions of mercury species in the scrubbing solution.

Mercury Speciation of Incoming Flue Gas. The mercury removal efficiency of a wet FGD system varies depending on the form or species of mercury vapor in the incoming flue gas. Mercury in flue gas is either associated with particulate matter or in the gas phase. In the U.S., most commercial wet FGD systems are used downstream of ESPs (Redinger et al., 1997). An ESP removes most of the particulate-bound mercury from the boiler flue gas before it reaches the wet FGD system; thus, most of the mercury that enters a wet FGD system is in the gas/vapor phase. The vapor phase mercury in boiler flue gas is generally present as elemental mercury (Hg^0) or oxidized mercury ($HgCl_2$) (Redinger et al., 1997). The proportion of elemental mercury to oxidized mercury in the flue gas is influenced by a number of factors such as the type of coal fired in the boiler, fly ash composition, flue gas temperature, and the presence of other compounds in the flue gas such as HCl, SO_2, and NOx. Because oxidized mercury is much more soluble in the aqueous solution present in a wet FGD system than elemental mercury, it is more likely to be removed from the flue gas.

Recent studies indicate fly ash and its subsequent interaction(s) with the vapor phase compounds in the post-combustion zone can influence a higher proportion of oxidized mercury as compared the elemental mercury (Carey et al., 1996 and 1997; Hargrove et al., 1997; Laudal et al., 1996 and 11/97 &12/97; and Senior et al., 6/97 & 11/97). The fly ash from the combustion of certain Northern Appalachian bituminous coals can have a significant impact, resulting in high levels of the oxidized form of mercury entering the wet FGD systems. A high conversion (>75%) of spiked elemental mercury into a particle laden flue gas upstream of highly efficient pilot-scale pulse-jet FFs was observed at two coal-fired facilities. The conversion was measured with the Tris-Buffer and Ontario Hydro speciation measurement methods. There was no apparent conversion of the spiked elemental mercury measured in the particle free flue gas at the outlet of the pulse-jet FFs (the FFs particulate control efficiencies were measured at 99.99%) by the Tris-Buffer and Ontario Hydro methods.

The coals fired during the separate tests were both N. Appalachian coals (Pittsburgh Seam/Blacksville and a blend of Ohio No. 5 and No. 6) that provide a high percentage of natural occurring oxidized mercury. Bench-scale tests conducted by Radian International and UNDEERC have indicated that the fly ash from the combustion of Blacksville coal has the ability to convert elemental mercury to an oxidized form. The exact vapor phase compounds and subsequent mechanisms responsible for the conversion are being investigated with this and other fly ashes. The conversion is less pronounced or not indicated with PRB and other subbituminous coal fly ashes (Carey et al., 1996 and 1997; Hargrove et al., 1997; Laudal et al., 1996 and 11/97 &12/97; and Senior et al., 6/97 & 11/97).

EPRI has reported pilot-scale experience showing significant capture of oxidized mercury in an ESP/wet FGD system (Chow and Owens, 1994). Approximately 60 percent of the total 10 $\mu g/m^3$ of mercury in the flue gas was in the oxidized form. The ESP/wet FGD system captured all of the oxidized mercury while allowing the elemental mercury to pass through the scrubber.

Radian conducted a series of pilot scale tests that showed significant capture of oxidized mercury by a wet FGD system (Noblett, 1993). In these tests, more than 95 percent of the mercury in the inlet flue gas to the scrubber was in the oxidized form. The scrubber system removed over 90 percent of the oxidized mercury from the flue gas while removing little elemental mercury.

FGD pilot testing by Babcock & Wilcox (B&W) with three Eastern bituminous coals has demonstrated a range of total mercury emissions reductions across the scrubber with the scrubber operating at constant conditions (Redinger et al., 1997). With a baghouse/FGD emissions control configuration, total FGD system mercury emissions control ranged from 88% to 92% for the three coals. For the same coals, with an ESP/FGD system configuration, mercury emissions reduction across the FGD ranged from 23% to 80%.

Coal Type. EPRI has published data which show distinct differences between the forms of mercury in the vapor phase and the distribution of mercury between the particulate and vapor phases for bituminous and sub-bituminous coals (Chang, 1994). In general, a higher level of elemental mercury was observed for sub-bituminous coal versus bituminous coal at typical wet FGD system inlet temperatures. The EPRI data indicated that at 300°F, 68 percent of the total vapor phase mercury was present as elemental mercury for the sub-bituminous coal compared to 6 percent as elemental mercury for the bituminous coal. This difference in mercury speciation suggests that a wet FGD system will have a low mercury removal efficiency if it treats flue gas from a boiler that fires sub-bituminous coal and a high mercury removal efficiency if it treats flue gas from a boiler that fires bituminous coal.

Design and Operation of the Wet FGD System. The liquid-to-gas (L/G) ratio of a wet FGD system impacts the removal efficiency of oxidized mercury. The L/G ratio of a wet limestone FGD system is dictated by the desired removal efficiency of SO_2. In general, high efficiency (95 percent SO_2 removal) systems are designed with L/G ratios of 120 gal/1000 acf to 150 gal/1000 acf. In an EPRI pilot study, increasing the L/G ratio from 45 gal/1000 acf to 133 gal/1000 acf increased the removal efficiency of oxidized mercury from 90 percent to 99 percent (EPRI, 1994). In another pilot study by B&W, increasing the L/G ratio from 37 gal/1000 acf to 121 gal/1000 acf increased the removal efficiency of oxidized mercury from 91 to 98 percent; increasing the L/G ratio did not affect the removal efficiency of elemental mercury, which was close to zero percent (Redinger et al., 1997).

Configuration of the Wet FGD System. Most of the existing US wet FGD systems have open spray tower or tray tower designs (Redinger et al., 1997). Recent research has shown that tray tower designs are more effective in removing oxidized mercury from boiler flue gas than open spray tower designs at the same operating conditions. In one study where the composition of the flue gas was mostly oxidized mercury, total mercury removal efficiencies from a wet FGD system with a tray tower design ranged from 85 to 95 percent, whereas total mercury removal efficiencies from a wet FGD system with an open spray tower design ranged from 70 to 85 percent (removal efficiencies for both systems increased as their L/G ratios increased from 39 to 122 gal/1000 acf) (Redinger et al., 1997).

Measurement Limitations and Reduction of Oxidized Mercury. A high proportion of oxidized mercury in the inlet flue gas to a wet FGD system does not guarantee that the scrubber will have a high total mercury removal efficiency. Evidence exists that elemental mercury can be generated in a wet FGD system by reduction of a portion of the oxidized mercury absorbed in the scrubbing solution. Radian evaluated mercury removal across a wet FGD system, in which 67 to 95 percent of the inlet mercury to the scrubber was present in the oxidized form (Hargrove, 1994). Despite these relatively high levels of oxidized mercury, the average removal efficiency of total mercury from the scrubber was only 50 percent. Radian noted possible generation of elemental mercury across the scrubber. Recent tests by B&W using the Ontario Hydro method have also noted higher concentrations of elemental mercury in the outlet of a wet FGD system compared to the inlet concentrations of elemental mercury. Pilot-scale testing using the Ontario Hydro method to measure mercury upstream and downstream of the scrubber has demonstrated the conversion of oxidized mercury species at the scrubber inlet to elemental mercury across the scrubber can be minimized by control of the dissolved species in the scrubbing system slurry (Redinger et al., 1997).

Previous field studies conducted by EPRI and DOE did indicate higher levels of elemental mercury (Hg^o) at the outlet of wet FGD scrubbers relative to the inlet. In addition, the removals indicated higher than 95 percent of the reported oxidized mercury at the inlet. These measurements were reported from separate U.S. EPA Draft Method 29 (M29) samples and in combination with the MESA Method samples. Two questions were raised: “Was the U.S. EPA M29 capable of accurately measuring the oxidized form of mercury?” or “Was the oxidized form of mercury being captured in the wet FGD scrubber solutions being released as an “alternate” form not capable of being collected in the appropriate impinger solutions?”

Innovative pilot-scale studies were conducted by Radian International at the EPRI ECTC to address these two questions. Extensive flue gas and intra-train mercury spiking tests were conducted to investigate the acidified peroxide solutions of M29 (solutions for collecting the oxidized form of mercury). The first series of tests had Hg^o and $HgCl_2$ injected separately into the flue gas stream at the inlet of the wet FGD. Results indicated 96 percent of the $HgCl_2$ (naturally occurring and spiked) was collected across the wet FGD and the increase in Hg^o across the FGD was from 0.66 to 0.96 µg/m^3. The results for the Hg^o spiking indicated 37% of spike was measured in the acidified peroxide solutions and the total Hg removal was only 29%. These results indicated the injected $HgCl_2$ was being effectively collected in the scrubber solutions and not being reduced and subsequently re-emitted as Hg^o. In addition, M29 was not effective in speciating the mercury at the inlet of this wet FGD system when Hg^o was spiked.

The intra train-spiking of either forms of mercury into the flue gas further indicated the inability of M29 to accurately measure the distribution of the speciated and elemental mercury in the flue gas at typical conditions upstream of a wet FGD. Radian conducted all of these initial tests in 1994 and repeated them in 1995, which are summarized in an EPRI and DOE report (Laudal et al, 1996).

Studies at the UNDEERC have duplicated the results of Radian. Recent studies at the UNDEERC indicated an overestimation of the oxidized mercury of up to 50 percent for M29 and up to 70 percent for the MESA method. The UNDEERC work has indicated the conditions at the inlet of wet FGD systems (e.g., high SO_2 concentrations and moderate to high concentrations of NO_x) have an impact on the overestimation of the oxidized form of mercury - SO_2 for the U.S. EPA M29 and the combination of SO_2 and NO_x for the MESA. These findings are also detailed in the EPRI and DOE report (Laudal et al, 1996).

After two years of evaluating and developing mercury speciation measurement methods, the UNDEERC has identified the Ontario Hydro Method as one of the most promising mercury speciation measurement methods. To obtain the accuracy of the speciated mercury measurement method, it was necessary to perform an U.S. EPA Method 301 validation procedures with dynamic spiking of mercury in the flue gas stream. Spiking was done first with elemental mercury then with $HgCl_2$. Results showed the Ontario Hydro method passed the U.S. EPA Method 301 criteria and was able to collect the form(s) of mercury correctly from the flue gas. The testing was conducted at the same and higher levels of SO_2 in the flue gas as compared to the previous validation studies for M29. The Ontario Hydro method was not impacted by the SO concentrations as indicated for M29 and the MESA method. The Ontario Hydro method is being recommended as the best method to measure mercury speciation in coal-fired systems. The method is being submitted to the American Society for Testing and Materials (ASTM) and U.S. EPA for approval (Laudal et al, December, 1997).

The recent pilot-scale speciation measurement evaluation/development studies and field results with the promising methods indicate less of an increase in the apparent re-emission of the captured oxidized mercury. Under certain conditions there has been an increase of the outlet elemental mercury as compared to the inlet of a wet FGD system (possible re-emission of the captured oxidized mercury) while utilizing the Ontario Hydro method (Redinger et al, 1997). Further testing at the McDermott facility will be conducted to determine at what wet FGD conditions the possible re-emission occurs.

2.3.3 Spray Dryer FGD Systems

In 1990, spray dryer FGD systems were installed on approximately one percent of coal-fired units in the US (UDI, 1992). The primary function of spray dryer FGD systems is to remove SO_2 emissions from boiler flue gas, however, they can also be effective in removing mercury emissions from boiler flue gas.

The effectiveness of a spray dryer FGD system to remove mercury emissions from boiler flue gas depends on the form or species of mercury vapor present in the incoming flue gas. In one study, the removal efficiencies of SO_2 and total mercury from a spray dryer FGD system were 82 percent and 63 percent respectively; oxidized mercury represented 73 percent of the total mercury at the scrubber inlet. In another study, the removal efficiencies of SO_2 and total mercury from a spray dryer FGD system were 68 percent and 64 percent respectively; oxidized mercury represented 68 percent of the total mercury at the scrubber inlet (Redinger et al., 1997).

2.4 Research and Emerging Technologies for Controlling Mercury Emissions from Utilities

Considerable research continues to develop efficient and cost-effective technologies for mercury emission reductions from utility plants. This section describes ongoing research and summarizes the results to date. Much of the research is being sponsored by three organizations: U.S. EPA, DOE and EPRI. Table 2-4 lists the areas of research currently being funded by these groups.

Eleven Phase I mercury control projects have been completed as part of DOE’s Advanced Emissions Control Technology “MegaPRDA Program.” These Phase I efforts began in October 1995 and encompassed two years of laboratory and bench scale testing and evaluation of several approaches for controlling the emission of mercury from coal-fired utility boilers. The approaches included those listed in Table 2-4. DOE has selected six Phase II proposals (two to three year efforts) to further investigate and develop fine particulate and mercury control technologies and concepts. Given the relative low maturity level of these technologies, commercial deployment is still at least several years away, and will be strongly dependent on the results of the Phase II efforts.

Research continues on developing potential technologies for mercury emission reduction from utility plants. This research is aimed at either the addition of some type of sorbent technology to adsorb the mercury, improving the mercury capture effectiveness of existing pollution control technology, or using new technology for mercury control. Before any of the technologies are fully realized for utility application, the fundamental mechanisms of the flue gas and mercury chemistries during the combustion and post-combustion conditions, along with the various interactions with the different types of fly ash must be understood (Brown, T.D., 1997).

Table 2-4
Current Mercury Control Research for Utility Boilers

Sponsor	Research Area
U.S. Environmental Protection Agency	Mercury speciation/High temperature control
	Fundamental reactions/Low temperature control
	Combined SO_2/Mercury control
U.S. Department of Energy	Fundamental and bench-scale investigation of adsorption and conversion of mercury by fly ash
	Fundamental studies & model development to predict mercury speciation, partitioning, and fate in coal-based power systems
	Fundamental and bench-scale studies on enhanced sorbents for mercury adsorption
	Pilot-scale field studies on sorbent injection for conventional APCDs
	Enhanced removal of oxidized and elemental mercury in wet FGD systems
	Capture of total mercury with regenerable sorbents
	Coal cleaning (physical, biological, mild chemical)
Electric Power Research Institute	Bench-scale: adsorption of mercury onto fly ash
	Fundamental studies & model development to predict mercury speciation, partitioning, and fate in coal-based power systems
	Field scale: pilot tests (two sites) of sorbent injection with ESP's and fabric filters
	Bench scale studies of mass transfer
	Wet scrubber controls for mercury
	Absorption of mercury in aqueous solution

Research at the fundamental level is being conducted by Physical Sciences, Inc., to determine the mechanisms involved with both gas-phase mercury transformations and the gas-solid interactions.

Attempts have been made to use thermochemical equilibrium calculations to predict the mercury species in coal combustion flue gas by using equilibrium calculations (see, for example, the review by Galbreath and Zygarlicke, 1996). The results of equilibrium calculations for mercury speciation in flue gas as a function of temperature can be summarized briefly. Above about 975 K (700°C) 99% of the Hg is predicted to exist as gaseous Hg. The rest (1%) is predicted to be gaseous HgO. Below 725 K (450° C) all the Hg is predicted to exist as $HgCl_2$. Between 725 and 975 K, the split between $HgCl_2$ and Hg is determined by the chlorine content of the coal (via the HCl content of the gas). HCl concentrations in flue gas from U.S. coals are typically in the range of 1 to 100 ppm. Even at these low concentrations, the reaction between Hg and HCl dominates the equilibrium chemistry. At temperatures representative of the inlet to the APCD, therefore, all the mercury should exist in the gas phase as $HgCl_{2(g)}$, if equilibrium is attained in the flue gas.

However, there are strong arguments against the existence of chemical equilibrium in the flue gas of a coal-fired power plant. The flue gas cools rapidly as heat is transferred to water and steam; typical cooling rates are on the order of 500 K/s. Minor species in the flue gas such as CO and SO_2 do not have time to equilibrate as the gas cools. For example, the oxidation of SO_2 to SO_3 in coal combustion flue gas does not proceed at a fast rate below about 1500 K (Flagan and Seinfeld, 1988) and thus the SO_3 concentration is effectively frozen below this temperature in the flue gas. Similarly for trace species, present in ppm or ppb amounts, equilibrium may not be attained as the flue gas cools. Recent kinetic calculations also indicate that the conversion of another trace species, HCl, to Cl_2 is frozen as the flue gas cools (Senior, et al., 1997).

The evidence from pilot-scale and full-scale combustion systems is not consistent with the assumption of equilibrium for mercury species in flue gas at the temperatures corresponding to the location of the air pollution control devices (APCD). At the inlet to the APCD, measurements in large scale combustion systems indicate that only about 75% of the gas-phase mercury is found as Hg^{+2} (Prestbo and Bloom, 1990; Fahlke and Bursik, 1995; Meij, 1994). The range of observed values is broad: one study consisting of mercury speciation measurements from fourteen different coal combustion systems reported anywhere from 30% Hg^{+2} to 95% Hg^{+2} upstream of the APCD (Prestbo and Bloom, 1990). There is some evidence from laboratory and pilot data that the kinetics of Hg oxidation are slow at low temperatures. Based on pilot data, the addition of HCl at temperatures below 450°K (180° C) did not increase the amount of $HgCl_2$ in coal combustion flue gas, indicating no reaction at those temperatures (Galbreath and Zygarlicke, 1996). In laboratory experiments (Nordin et al., 1990) using simulated flue gas (in the presence of activated carbon), equilibrium was not attained for Hg at temperatures below 473 K (200°C).

The assumption of gas-phase equilibrium for mercury-containing species in coal-fired power plant exhaust is not valid. Preliminary evidence suggests that the oxidation of elemental mercury to mercury chloride in the gas is frozen when the gas cools below 750-900°K. Kinetic calculations on the formation of Cl_2, which is highly reactive with elemental mercury, indicate that the conversion of HCl to Cl_2 does not attain equilibrium given the time temperature-history in a power plant which lends support to the conclusion of frozen equilibrium for mercury oxidation.

Understanding gas-phase speciation of mercury in coal fired power plant flue gas is not sufficient to describe the transformations of mercury in the combustion system. In order to understand the capture of mercury in APCDs and the effectiveness of sorbents for mercury capture, better understanding of the gas-to-particle conversion is also needed, particularly the relationship between fly ash properties and oxidation and/or adsorption of mercury.

Two key questions can be posed: first, what is the process by which fly ash (and certain other solids) seem to catalyze the transformation of gaseous elemental mercury to oxidized forms; second, what are the mercury species adsorbed on fly ash? Answering both these questions will require a detailed look at the constituents of the fly ash and how they interact with mercury at temperatures characteristic of the flue gas (400-600°K) as it enters the APCD.

Gas-phase oxidized mercury is readily captured by activated carbon, while elemental mercury has a much lower affinity for carbon. The surface of the carbon is crucial to mercury sorption; adding sulfur or iodine can dramatically increase the capacity of activated carbon for elemental mercury (Dunham and Miller, 1996; Krishnan et al., 1994; Vidic and McLaughlin, 1996). Residual carbon from coal combustion is not the same as activated carbon. The pore structure, surface properties, and inorganic content may be strikingly different. Nonetheless, coal char does have some capacity for adsorbing mercury. Based on the recent experimental work (Senior et al., 1997), it can be concluded that the mechanisms for adsorption of elemental and oxidized mercury on coal char are very different. Properties of the coal char (surface area, sulfur content, and forms of sulfur) have been shown to determine the amount of mercury adsorption. In addition to carbon, there is evidence for the adsorption of mercury on coal fly ash (Carey et al., 1996) although the specific species which are adsorbed is not known.

In addition to adsorption, laboratory and pilot scale evidence suggest that solids such as activated carbon and fly ash can act as catalysts for oxidation of elemental mercury. Kinetic experiments in a continuous flow reactor showed that the oxidation of elemental mercury by oxygen only occurred in the presence of activated carbon (Hall et al., 1991). A series of bench-scale experiments explored the catalytic effect of solids, including traditional metal catalysts, activated carbon, and coal fly ash, on the oxidation of elemental mercury in simulated flue gas in a fixed bed reactor (Carey et al., 1996). The results showed that coal fly ash converted gaseous elemental mercury to a mixture of gaseous oxidized mercury and adsorbed mercury at temperatures from 420°K to 640°K (300ºF to 700ºF). Fly ash from five different coals was tested. At 420°K, 20-50% of the elemental mercury was converted to a gaseous oxidized form, probably $HgCl_2$ based on equilibrium considerations, while 0-80% was converted to an adsorbed form on the solids. The adsorbed species might be $HgCl_2$, HgO, or $HgSO_4$. There was a wide variation in the amount of adsorbed mercury depending on coal type. At 640°K, less elemental mercury was typically converted.

Information on the reactions of mercury species with fly ash can be obtained by identifying specific mercury species on the surface of char or carbon and then inferring the reaction pathway. Preliminary analysis of the forms of mercury on four carbon-based sorbents as described in PSI et al, 1997 was recently completed (Huggins et al., 1997). These samples were treated with a simulated flue gas containing N_2, O_2, CO_2, SO_2, H_2O, HCl, and elemental mercury. In order to better understand the forms of adsorbed mercury, X-ray absorption fine structure (XAFS) spectra were collected at the mercury L_{III} edge at approximately 12,284 eV at the Stanford Synchrotron Radiation Laboratory. By combining both the XANES and EXAFS evidence, one could speculate that the Hg bonding in the three different mercury sorbents is different. In the iodine-impregnated activated carbon, the mercury bonding appears consistent with Hg-I. In the sulfur-impregnated carbon and the lignite-based activated carbon, the bonding is more consistent with Hg-Cl or Hg-S. Further study, particularly of the Cl-edge XAFS spectra in the SAC and LAC samples is required.

Thus, particulate matter can promote oxidation of elemental mercury and can collect a significant amount of mercury in flue gas. The amount retained in the particulate matter seems to depend on the following factors:

- carbon content
- properties of the carbon surface
- inorganic constituents in carbon particles
- Hg speciation in the flue gas

2.4.1 Sorbent Technology

Research continues on developing potential technologies for mercury emission reduction from utility plants. Although sorbent injection with activated carbon has been shown to be a promising technology, even greater mercury removal may be possible with impregnated activated carbons, sodium sulfide, and other types of sorbents. The application of an activated carbon circulating fluidized bed (CFB) also shows promise in removing mercury.

With sulfur-impregnated activated carbon injection, the carbon-bound sulfur reacts with mercury to form mercuric sulfide (HgS) on the carbon, which is then 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).

Sulfur-impregnated carbons can potentially be enhanced for mercury sorption by the impregnation of the carbon(s) with sulfur at elevated temperatures of 400 - 600°C (752 - 1112°F). This has promoted a more uniform distribution of short linear chains of sulfur allotropes (S_2 and S_6) on the carbon surface as opposed to having predominately S_8 rings condensed in the macropore region of commercially available sulfur impregnated carbons. In addition, the sulfur impregnated carbons prepared at elevated temperatures have exhibited significantly better thermal stability since no sulfur loss was observed even after exposure at 400°C (752°F). The sulfur impregnated carbons exhibited high elemental mercury uptake efficiency at 140°C (284°F) when compared to commercially available activated carbons. Dynamic adsorption capacity of these carbons were measure as high as 4000 μg Hg/g C. This capacity is almost three orders of magnitude greater then the capacity of virgin activated carbon and an order of magnitude greater than the capacity of commercially available impregnated activated carbon (Vidic et al., 1996; Korpiel et al., 1997; and Liu et al., 1997).

With iodide-impregnated activated carbon injection, the carbon-bound iodide reacts with mercury to form mercuric iodide (HgI_2) on the carbon, which is then 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 non-impregnated activated carbon (Felsvang et al., 1993).

A study by the UNDEERC, as part of a Cooperative Agreement with the DOE-FETC, found that iodide-impregnated activated carbon was effective at removing mercury in a test combustor. Removal effectiveness using the iodide-impregnated activated carbon exceeded 99 percent. Other sorbents tested were steam-activated lignite, thermal-activated bituminous coal, chemical-activated hardwood, iodine impregnated, steam-activated coconut shell, and sulfur-impregnated steam-activated bituminous coal (UNDEERC, 1995).

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. Experiments have shown that impregnating activated carbon with chloride salts increases adsorptive capacity of the activated carbon by a factor of 300 (Teller and Quimby, 1991).

Public Services Company of Colorado (PSCo) has investigated the application of dry-sorbent injection for controlling mercury emitted from coal-fired boilers. A number of sorbents, including activated carbon, sulfur- and iodine-impregnated carbons, several proprietary sorbents, and high-carbon fly ash, were screened in the laboratory prior to pilot-scale testing. Two activated carbons have been tested on a pilot-scale facility drawing flue gas from PSCo's Comanche Station in Pueblo, Colorado under pulse-jet and reverse-gas FF-, and ESP-configurations. American Norit Companies' Darco FGD, an activated carbon derived from lignite which has been utilized in the control of mercury from municipal solid waste combustors, was be tested. The second sorbent is an activated carbon prepared from a bituminous coal (Feeley, 1997).

Parameters of flue gas temperature and carbon residence time were varied to cover a wide range of utility conditions. The effects of fly ash were also evaluated by pulling flue gas from the upstream and downstream side of the existing reverse gas baghouse with carbon injected in the slipstream prior to the inlet of the pilot-scale configuration being tested. Elemental mercury had to be spiked upstream of the pilot-scale unit due to low mercury concentrations of the native flue gas stream.

The results indicate a high level of carbon is needed to remove the mercury, but deceasing the temperature (either by heat exchangers or spray cooling with water) had a net increase of the mercury captured by both the injected carbon and the native fly ash. The fabric filter configurations had the greatest removals up to 90%, but at high carbon injection rates. The ESP results indicate removals of 50% with approximately 30% of the total removal due to the native fly ash with the mass carbon-to-mercury ratios greater than 5000:1. The test results for all the configurations are summarized under Section 2.3.1.2, "Current Research on Activated Carbon Injection for Utilities" (Sjostrum et al., 1996; Haythornthwaite et al, 1997; and PSCO/ADA et al, 1997).

Other innovative activated carbon injection studies have been conducted by ADA Technologies for EPRI at Public Service Electric and Gas Company's (PSE&G) Hudson Unit 2 located in Jersey City, New Jersey. The results also indicate a high level of carbon is needed to remove the mercury, but deceasing the temperature had a net increase of the mercury captured by the injected carbon, but not for the native fly ash. EPRI's COHPAC or TOXICON configurations and a pilot-scale ESP were tested with the Darco FGD activated carbon. The test results for the different configurations are also summarized under Section 2.3.1.2, "Current Research on Activated Carbon Injection for Utilities" (Waugh et al., 1997).

All the current work indicates the removal of mercury is mass transfer limited in the various flue gases produced from the combustion of coal. The reasons for this limitation are the low concentrations of mercury present in the relatively high volumes of flue gas. There are higher concentrations of other species competing and occupying the active sites of the carbon. In addition, the flue gas residence time upstream of an ESP is nominally one second or less with flue gas velocities in the range of 50 to 60 ft/sec at 149°C (300°F). Compounding the mercury mass transfer limitation(s) is the decrease in the carbon reactivity and capacity at this nominal, but high temperature. Fundamental studies have been performed in the past two years designed to understand the mechanisms impacting the mercury mass transfer limitation(s) (Carey et al., 1996 & 1997; Vidic et al., 1996; Rostam-Abadi et al., 1997; Korpiel et al., 1997; and Liu et al., 1997).

Another technology with potential for improving mercury collection efficiency combines 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.

While sulfur-, iodide-, chloride salt- and $Ca(OH)_2$-impregnated activated carbons show promise for increasing the mercury removal efficiency, the cost of these modified carbons can be as much as 20 times higher than that of unmodified activated carbon (Maxwell, 1993). In addition, chemically impregnated carbons may increase the reactivity and subsequent capture of mercury, but very few studies have indicated the effectiveness of chemically impregnated carbons for in-flight capture of mercury (especially at one second or less residence time) (Vidic et al., 1996; Korpiel et al., 1997; and Liu et al., 1997). These carbons, while being cost prohibited for in-flight mercury removal, can possibly be designed for high mercury adsorption capacities indicative of long contact times (carbon beds or fabric filters - pulse-jet, if installed downstream of an existing ESP). The effectiveness of FF-configurations downstream of an ESP must be further investigated.

Argonne National Laboratory is investigating potentially low-cost, chemically treated, solid sorbents, such as volcanic pumice, as an economical alternative to activated-carbon injection. In addition, Argonne is planning to assess several key, ancillary issues that may impact the potential use of these sorbents to control mercury, including the effect of the sorbents on particulate control equipment performance, fly-ash marketability, and by-product disposal (Feeley, 1997).

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 Hg are not totally clear but appear to be (Nebel and White, 1991):

$Hg\ (gas) + Na_2S + 2H_2O \Rightarrow HgS\ (Solid) + 2NaOH + H_2$ and

$HgCl_2\ (gas) + Na_2S \Rightarrow HgS\ (Solid) + 2NaCl$.

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.

Sorbent Technologies is marketing a sorbent called Mercsorbent (Nelson et al., 1997). The company claims that the sorbent is effective in removing elemental mercury at high temperatures typical of utility flue gas, and is unaffected by common co-existing flue gases, such as SO_2, HCl, and H_2O. Mercsorbent can be used for sorbent injection or it can be used as a coating on a FF. A bench-scale duct-injection system at Sorbent Technologies facilities is now being used to test Mersorbents with this approach. The company is also scheduled to demonstrate the sorbent at the refuse incinerator in Fort Dix, New Jersey, in 1997; prior compliance sampling at this facility suggests that a significant amount of its mercury is in the elemental form. A coal-fired boiler or slipstream is also being sought for a test of the new sorbent material.

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 (Riley, 1991). 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. This system has been used in Germany for MWC operation.

In the U.S., Environmental Elements Corporation has been developing and testing a CFB promoting agglomeration of fine particulate matter, allowing for their capture in an ESP. In addition, a single injection of iodine-impregnated activated carbon was added to the fluid bed to adsorb mercury vapor. High residence time, due to the recirculation of the particles, allows for effective utilization of the carbon and high collection of the fine particles. Results from the laboratory-scale testing indicate spiked elemental mercury was significantly reduced when passed through the fluidized bed of fly ash (50% mercuiry removed) and further reduced to esentially to zero when the activated carbon was injected into the bed (25 $\mu g/m^3$ to zero) at 110^o C (230^o F). The iodine-impregnated activated carbon was fully utilized after >2 hours within the bed. An adsorption capacity was calculated to be 770gm/gm for the carbon and 480 gm/gm for the bed of ash. The ash still was able to remove 30 $\mu g/m^3$ after 100% breakthrough (carbon fully utilized) was indicated for the carbon. The unit needs to be tested on actual flue gas from coal combustion and there are plans to install a pilot unit and conduct testing at Public Service Electric and Gas's Mercer Station (Feeley, 1997).

2.4.2 Improving the Mercury Capture Efficiency of Existing Pollution Control Technology

Research on improving the mercury capture efficiency of existing pollution control technology can be categorized as an investigation of either mercury removal with wet FGD systems or particulate control technology for capturing mercury.

Enhancing Mercury Removal by Wet FGD Systems. Argonne National Laboratory is investigating several additives that combine strong oxidizing properties with relatively high vapor pressures to enhance the capture of mercury in a wet scrubber. Due to a much higher solubility compared to elemental mercury, oxidized mercury is readily removed in a wet scrubber. Experimentation is continuing on the effect of solutions of chlorine, bromine, and iodine on the conversion and removal of elemental mercury in a laboratory-scale reactor. Of the three halogen species tested to date, the chlorine solution appears to remove the most elemental mercury in the presence of SO_2 and NO. Further testing of these and possibly other oxidizing reagents is planned (Feeley, 1997).

Radian International LLC has also investigated the conversion of vapor-phase elemental mercury to more soluble Hg^{++} at the bench- and pilot-scales. Radian screened a number of catalysts and coal-based fly ashes for their ability to oxidize elemental mercury, including the effect of flue gas temperature, flue gas vapor phase compounds, and residence time on the oxidation potential of the materials. Bench- and pilot-scale testing of iron-based catalysts, various carbons, bituminous, subbituminous, and lignite fly ash have been performed on a slipstream of flue gas at the EPRI Environmental Control Test Center (ECTC) in Barker, New York. In addition, bench-scale testing has been conducted at an utility firing a coal producing a higher percentage of elemental mercury in the flue gas as compared to the ECTC.

To date, the pilot-scale tests have shown the carbon-based catalyst to be the most effective in converting elemental mercury to Hg^{++}. Further testing of the carbon catalysts is being planned at three utility sites at the bench-scale. Flue gas composition, interaction with the fly ash, and temperature will be the variables. Deactivation of the catalysts will be investigated with reactivation concepts being initiated. The tests will be designed to determine the long-term capabilities of the catalysts, with testing being conducted over a six month period of performance for all the catalysts. The influence of sulfur and nitrogen oxides, HCL, and other vapor phase compounds will be investigated. Converting elemental mercury into an oxidized form could be advantageous in reducing mercury emissions with existing technologies (Carey, et al., 1996 & 1997; Hargrove, et al, 1997; and Radian International, et al, 1997).

Improving Particulate Control Technology for Capturing Mercury. Research into improving the existing particulate control technology for capturing mercury is being conducted by several companies.

ABB Power Plant Laboratories is developing retrofittable enhancements to existing ESPs to increase their efficiency in capturing fine particles and air toxics. Several approaches to improving the capture of fine particulates have been investigated. The most significant results were through flue gas cooling (humidification and heat exchange) and in combination with pulsed energization. The pulsed energization was accomplished through an ABB proprietary transformer rectifier set - Switched Integrated Rectifier (SIR). Flue gas cooling in combination with the SIR provided particle reductions from 45 mg/m^3 to less than 5 mg/m^3 (<0.005 lbs/MMBtu) at a gas temperature of 150^oC (300^oF) . The particles in the 2.5 micron range and less were effectively reduced by a factor of 10 to 20. Preliminary tests indicated a reduction between 40 and 50 percent of the mercury in the flue gas by the native fly ash, which is encouraging for both the low-sulfur bituminous and subbituminous coals. This approach shows promise in improving the collection of particulate-bound mercury, and may also cause vapor-phase mercury to condense on particulate matter and be captured in the ESP. Future work entails scaling the technology and testing under a variety of coals and further investigating activated carbon injection with flue gas cooling. Potential impacts on fine particle collection will be monitored during all phases of testing (Feeley, 1997; Srinivasachar and Porle,1997; and ABB, et al, 1997).

The performance of conventional control technology in reducing the emissions of mercury from coal-fired boilers is being evaluated in pilot-scale studies as part of Babcock & Wilcox's Advanced Emissions Control Development Program (AECDP). Phase I of the AECDP involved benchmarking the mercury capture performance of an ESP, a baghouse, and a wet scrubber installed at B&W’s Clean Environment Development Facility (CEDF). The focus of Phase II was to optimize the mercury removal capability of the conventional pollution control technologies. The results of the work conducted in 1996 and 1997 were detailed in the sections under “2.3.2, Flue Gas Desulfurization (FGD) Scrubbers” (Feeley, 1997; Redinger, et al, 1997; and Holmes, et al, 1997).

Phase III of the program will be directed at the development of new air toxics emissions control strategies and devices. Further testing at the McDermott facility will be conducted to determine at what wet FGD conditions the possible re-emission of captured oxidized mercury occurs.

Under DOE funding, the Energy and Environmental Research Center together with W.L. Gore and Associates is developing a new technology for ultrahigh collection of fine particles, including the difficult-to-collect trace element enriched submicron fraction. The concept utilizes electrostatics and filtration in a unique manner that provides over 99.99% fine particle collection in a device that is up to 75% smaller than conventional technologies. The approach also shows promise for collecting vapor-phase trace elements such as mercury and selenium when combined with an effective sorbent. The concept will be scaled up for testing on a variety of coals under various operating conditions (Miller et al, 1997; and UNDEERC et al, 1997).

2.4.3 New Technology for Controlling Mercury

A new technology for controlling all forms mercury from coal-fired electric utility units has been investigated at the laboratory- and bench-scales on simulated and on actual flue from coal combustion. ADA Technologies has been developing a technology utilizing a regenerable sorbent allowing for the recovery of liquid elemental mercury from the flue gas and appropriately called the Mercu-RE process. The process takes mercury from flue gases and produces liquid, elemental mercury with no secondary wastes. Noble metals are used to adsorb mercury at typical flue gas temperatures. The mercury is then thermally desorbed.

Results from laboratory tests indicate that a gold-coated monolith captured virtually all of the elemental mercury injected into a simulated flue gas. Bench-scale tests on actual flue gas from the combustion of four different coals showed the regenerable sorbent is capable of removing 95% of both elemental and oxidized forms of the merury at temperatures between 150^{o}C (300^{o}F) and 204^{o}C (400^{o}F). The unit ran for more than 700 hours and consistently reduced the mercury (both forms) in the flue gas from inlet concentrations averaging 10 ug/m^3 to less then 1 ug/m^3 at the outlet after more than 20 sorption-desorption cycles at Consol's research facility in Library, Pennsylvania. Further testing of the gold monoliths will include repeated sorption and desorption cycles over longer-term testing periods at different operating conditions and at a larger scale (Feeley, 1997; Roberts and Stewart, 1996; Roberts and Stewart, 1997; ADA Technologies, Inc., et al, 1997).

Based on condensing heat exchanger technology, Babcock & Wilcox is developing an integrated flue gas treatment system for recovering waste heat and removing SO_2, SO_3, particulates, and trace elements from coal combustion flue gas. The condensing heat exchanger is a two-pass, counter-flow shell and tube heat exchanger. The hot flue gas enters the top and flows downward through the first cooling stage, across a horizontal transition region, and then upward through the second cooling stage. An alkali reagent is sprayed from the top of the second stage to aid in the removal of SO_2. Testing of the technology was conducted at B&W's research facility in Alliance, Ohio. Preliminary results indicate that total mercury removal across both stages of the condensing heat exchanger is about 62 percent when firing a blend of Ohio coals. Testing has been conducted on two other bituminous coals with similar or higher mercury removals (Feeley, 1997).

The Enhanced Limestone Injection Dry Scrubbing (E-LIDS™) process combines furnace limestone injection with dry scrubbing to achieve high efficiency SO_2 particulate, and trace element emissions control. Dry, pulverized limestone is injected into the upper furnace region of the boiler. The limestone is calcined to lime and a portion of the sorbent reacts with SO_2 in the flue gas. The flue gas passes through a particulate collector ahead of the dry scrubber to remove some of the solids from the gas stream. The solids are mixed with material collected in the baghouse to produce the SO_2 scrubbing reagent for the spray dryer.

Application of the E-LIDS™ system when firing an Ohio bituminous coal in the Clean Environment Development Facility (CEDF) at the Alliance Research Center of McDermott Technology, Incorporated has shown efficient emissions control performance. Sulfur dioxide emissions generated from firing the nominal 3 percent sulfur coal were reduced by more than 99 percent to less than 0.10 lbs $SO_2/10^6$ Btu. Total mercury emissions were reduced from an uncontrolled level of 17.6 μg/dscm to less than 0.2 μg/dscm for an average total removal efficiency of greater than 95 percent from the as-fired coal mercury. The measured performance confirmed earlier results obtained in the $5x10^6$ Btu/hr small boiler simulator (SBS) facility. Mercury measurements upstream of the dry scrubber indicated that both the limestone injection and operation of the spray dryer/baghouse system at close to the saturation temperature contributed to the observed total mercury emissions reduction. The furnace limestone injection alone reduced mercury emissions to an average of 3.1 μg/dscm (Redinger et al., 1997).

Environmental Elements Corporation is developing a process for mercury control through DOE's Small Business Innovative Research program. The first concept utilizes an intense corona discharge to convert Hg^0 to mercuric oxide. The process also produces SO_3 to serve as a conditioner for high-resistivity fly ash. A corona discharge in coal combustion flue gas will produce oxidizing radicals, such as OH and atomic oxygen. Bench-scale results indicate that the corona reactor, operating at relatively low power levels and short residence time, yielded high elemental mercury vapor oxidation. The mercuric oxide, in the form of a solid particle, was removed using conventional particulate control technology. The corona reactor may also convert mercuric chloride to mercuric oxide, allowing for its capture as well. The system is currently being tested on a slipstream at Alabama Power's Plant Miller (Feeley, 1997).

The capture of mercury on solid surfaces such as fly ash is being studied by UNDEERC and DOE-FETC. Data have shown wide variation in the amount of mercury that can be collected on fly ash associated with particulate control devices. On occasion, very high levels of capture have been observed in the presence of HCl separately and in combination with nitrogen oxides. A number of possible interactions between vapor-phase mercury and solid surfaces can occur, including chemical adsorption, physical adsorption, and condensation. However, the exact mechanisms of capture remain unknown. Research is being conducted by UNDEERC to elucidate these mechanisms in order to better define control strategies for mercury in coal combustion flue gases (Brown, 1997).

There are plans to investigate the interaction of mercury with metals such as zinc, silver, tin, and cadmium. Mercury has been shown to amalgamate, rather than adsorb, when in contact with certain metals. Both experimental and modeling efforts are planned to determine the suitability of metals for the capture of mercury (Feeley, 1997).

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 some of these technologies (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 four 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. In addition, it should be noted that for the combustion sources, the control devices described are also effective in controlling other pollutants. These might include acid gases and other metals for utility boilers, MWCs, and MWIs with the addition of dioxin compounds for the MWCs and MWIs. Therefore, although the costs are presented in terms of the cost of mercury reductions, it would be incorrect to attribute all of the control costs to mercury control. This is particularly the case for MWI. The emission guidelines finalized for MWI include emission limits for ten pollutants including mercury. The controls described here for mercury are also designed to control the other pollutants as well. If mercury alone were to be controlled it is possible that many MWIs could meet the emission limit for mercury by enacting an aggresive source separation program aimed at removing mercury products from the waste stream at a much lower cost than installing a control device.

With the exception of MWIs, detailed documentation and analysis of model plant cost is provided in Appendix B. For detailed information on the MWI cost analyses the reader is encouraged to consult *Hospital/Medical/Infectious Waste Incinerators: Background Information for Promulgated Standards and Guidelines - Regulatory Impact Analysis for New and Existing Facilities (EPA-453/R-97-009b).*

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 for MWCs, MWIs, and chlor-alkali plants. It shows the applicable mercury controls and the estimated cost effectiveness in dollars per pound of mercury ($/lb Hg) removed as well as other measures of cost (e.g., dollars per ton of municipal waste). For MWIs, cost-effectiveness values for individual facilities are not presented in this document because they have been recently published separately; only national impacts are presented in this chapter.

To determine how the cost-effectiveness of carbon injection might vary with different chlorine levels in the coal, a sensitivity analysis was performed for the utility boiler model plants which assumed various levels of $HgCl_2$ in the flue gas. These results are shown in Table 3-2 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.)

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)

Three model plants, representing small ($\leq$ 200 lb waste incinerated per hour (lb/hr)), medium (201-500 lb/hr), and large (> 500 lb/hr). These plants represent commercial medical waste incinerators, hospitals, nursing homes and research laboratories. Based on the MWI emission guidelines (promulgated August 15, 1997), the level of air pollution control for these model plants would have to result in either an outlet concentration of 0.55 mg/dscm or an 85 percent reduction in mercury. The controls which would achieve these levels are good combustion practices with wet scrubbers (or dry scrubbers with carbon injection). However, the emissions guidelines do not specify these controls. The emission limits can be achieved by any means or any technology. There are two other likely scenarios for the non-commercial MWI. One is that the facility would segregate it's infectious waste, switch to a commercial incinerator and landfill the remaining waste. This is referred to as "switching with segregation" in the cost analyses. The second scenario is that the facility already segregates it's infectious waste, but now switches to a commercial incinerator rather than on-site incineration. This is referred to as "switching without incineration" in the cost analyses. For the commercial incinerators, installation and operation of a control device is considered in the cost analyses.

Details of the MWI cost analyses are not presented in this document, but may be found in *Hospital/Medical/Infectious Waste Incinerators: Background Information for Promulgated Standards and Guidelines - Regulatory Impact Analysis for New and Existing Facilities (EPA-453/R-97-009b).*

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 0_2 (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.

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

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.

(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

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
	Good combustion, wet scrubber or dry scrubber with carbon injection Switching with waste segregation Switching without waste segregation	--	--	For cost-effectiveness estimates for individual facilites, the reader should consult *Hospital/Medical/Infectious Waste Incinerators: Background Information for Promulgated Standards and Guidelines - Regulatory Impact Analysis for New and Existing Facilities (EPA-453/R-97-009b).*

Cost Effectiveness of Control Technologies (continued)

Source	Mercury Control Technique	Cost Effectiveness		Cost Comments
		$/lb Hg Removed	Other Measures	
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

Table 3-2
Cost Effectiveness of Control Technologies for Utility Boilers

Model	Model Definition				Sensitivity Analysis for Models			
	Size (MW)	Fuel	Existing Controls	Mercury Control	Source	Carbon Usage[a] (g C/g Hg)	Cost Effect. (mils/kWh)	Cost Effect. ($/lb Hg)
1a	975	Low-sulfur Coal	ESP	AC injection	EPA	34,200	1.82	22,100
					DOE	100,000	5.58	67,700
1b	975	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	1.43	17,400
					DOE	9,400	2.10	25,400
1c	975	Low-sulfur Coal	ESP	Spray cooler, AC injection	EPA	460	0.40	4,940
					DOE	30,000	2.19	26,500
1d	975	Low-sulfur Coal	ESP	Carbon filter bed	EPA	--[b]	2.70	32,700
					DOE	--	NA[c]	NA
2	975	High-sulfur Coal	ESP/FGD	Carbon filter bed	EPA	--	3.1	37,800
					DOE	--	NA	NA
3a	100	Low-sulfur Coal	ESP	AC injection	EPA	17,200	1.16	14,200
					DOE	100,000	5.71	70,000
3b	100	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	2.09	27,700
					DOE	12,600	3.15	38,600

[a] The lower carbon injection rates represent low-temperature flue gas while the higher rates are for high-temperature operations. Both the low and the high carbon injection rates would achieve a 90 percent reduction in mercury emissions under the given temperature scenario.
[b] The mercury control is a stationary bed that does not require carbon injection.
[c] NA = Not available.

3.2 Financial Analysis

This section presents an analysis of the ability of the four industries studied (MWCs, MWIs, utility boilers and chlor-alkali plants) 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 wet scrubbers to control mercury emitted from MWCs 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 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, individual companies have little control over the price they are able to receive for their respective products. In this 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. As a result, mercury control costs cannot be passed on to the consumer and the funds to purchase mercury controls must be financed with existing profits. The financial impact from installing and operating the mercury controls for chlor-alkali plants using the mercury cell process 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 industry to absorb the annual control costs and to acquire the capital needed to purchase the controls while still remaining competitive.

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.

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 (see Table 3-3). 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 3-3
Potential Cost Increases[a] for MWCs and MWIs

Control Option	Municipal Waste Combustors	Medical Waste Incinerators
Activated Carbon Injection	Small[b]: 6.9% Large[c]: 1.3%	
Hospitals, nursing homes, research laboratories: Switching with waste segregation Switching with no waste segregation		0.01 - 0.04% 0.02 - 0.09%
Commercial Incineration		2.6 %

[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.

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. The national annual control costs (i.e, for the entire industry) to meet the mercury emission guidelines for MWI are estimated to range from $59.2 million/year to $120 million/year depending om whether the facilites choose to segregate their infectious waste prior to incineration. These options would results in potential cost increases ranging from 0.01 to 0.09 percent for the non-commercial facilities (e.g., hospitals and nursing homes), and 2.6 percent for commercial incinerators.

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 3-4.

Table 3-4
Estimated Annual Profits, Expenditures,
Revenues and Financial Impacts[a] for Chlor-Alkali Plants

Parameter	Chlor-Alkali Plants
Total Annual Profits (millions of dollars)	$12.9
Total Annual Expenditures (millions of dollars)	$26.7
Financial Impact	
Membrane Cell Process	12% of Expenditures
Depleted Brine Scrubbing	5.1% of Profits 0.7% of Expenditures
Treated Activated Carbon Adsorption	3.9% of Profits 0.5% of Expenditures

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

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.

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 3-5, 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 3-5. Two sets of results are presented, reflecting different values for carbon usage.

Table 3-5
Annual Revenues and Potential Cost Increases for Utility Boilers

Model	Model Definition				Sensitivity Analysis for Models				
	Size (MW)	Fuel	Existing Controls	Mercury Control	Source	Carbon Usage [a] (g C/g Hg)	Estimated Revenue ($ Million)	Annual Cost (10^6 $/yr)	Potential Cost Increases %
1a	975	Low-sulfur Coal	ESP	AC injection	EPA	34,200	287	10.1	3.5
					DOE	100,000	287	31.0	10.8
1b	975	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	287	7.94	2.8
					DOE	9,400	287	11.6	4.0
1c	975	Low-sulfur Coal	ESP	Spray cooler, AC injection	EPA	460	287	2.26	0.8
					DOE	30,000	287	12.1	4.2
1d	975	Low-sulfur Coal	ESP	Carbon filter bed	EPA	--[b]	287	14.9	5.2
					DOE	--	287	NA[c]	NA
2	975	High-sulfur Coal	ESP/FGD	Carbon filter bed	EPA	--	287	17.3	6.0
					DOE	--	287	NA	NA
3a	100	Low-sulfur Coal	ESP	AC injection	EPA	17,200	29	0.66	2.3
					DOE	100,000	29	3.25	11.2
3b	100	Low-sulfur Coal	ESP	Spray cooler, AC injection, fabric filter	EPA	460	29	1.29	4.4
					DOE	12,600	29	1.79	6.2

[a] The lower carbon injection rates represent low-temperature flue gas while the higher rates are for high-temperature operations. Both the low and the high carbon injection rates would achieve a 90 percent reduction in mercury emissions under the given temperature scenario.

[b] The mercury control is a stationary bed that does not require carbon injection.

[c] NA = Not Available.

4. MERCURY BENEFITS ASSESSMENT FRAMEWORK AND DISCUSSION

This chapter presents a recommended framework for estimating the benefits of reducing mercury emissions. The intent is to recommend a framework that is realistic (i.e., feasible within a reasonable time frame and budget), and consistent with current practices within the Agency.

The chapter is organized into two main sections. Section 4.1 discusses the theoretical background supporting a benefits assessment for reducing mercury contamination and raises relevant issues to be considered in future work on such a benefits assessment. Section 4.2 identifies and discusses the various steps envisioned as part of such an assessment for reducing mercury contamination.

4.1 The Economics of Benefits Assessment

This section discusses some of the basic concepts and issues relevant to understanding and conducting an assessment of the economic benefits associated with reducing mercury contamination. An additional goal of this section is to provide background on the theoretical and practical issues that need to be addressed in preparing a rigorous, comprehensive benefits assessment.

4.1.1 Definition of "Economic Benefits" and Goal of the Benefits Assessment

The general term "benefits" refers to any and all outcomes of a regulation that are considered positive; that is, that contribute to an enhanced level of social welfare. Social welfare is dependent on the quantity and quality of the goods and services that society has access to and are used or valued by the individuals that make up society. Environmental pollution, such as mercury contamination, can reduce both the quantity and quality of goods and services available to society, and thus, reduce social welfare (i.e., leave society worse off then without the environmental pollution). The benefits of a regulation that reduces the level of pollution in the environment is thus measured by the change in social welfare associated with the reduction in pollution levels, where the change in social welfare depends on how society values the changes in the quantity and quality of the goods and services affected by environmental pollution. Example goods and services affected by environmental quality or pollution levels include human health, fish consumed as food, and recreational activities, such as fishing and animal viewing. A comprehensive discussion of the potential benefit endpoints of mercury contamination is provided in Section 4.2.

An important distinction is being drawn here regarding how "environmental quality" is assumed to be valued by society for the purposes of a benefits assessment. As described here, environmental quality is valued by society because of the impact it has on various goods and services that are consumed or used by society. This framework for thinking about the benefits of improved environmental quality affects how one estimates the value of improved quality because the focus is on the change in the value of goods and services affected by the level of environmental quality, not the value of improving environmental quality explicitly. Some of the goods and services affected by changes in environmental quality that are consumed or used by society are readily apparent, while others are not. For example, people clearly value forests for the timber they supply and for the recreational opportunities they supply, but they may not realize that they also value forests for providing oxygen in the air they breath, for providing erosion control that protects water quality, and for providing habitat for many animals the individual values (these are all "services" provided by the forest that are affected by environmental quality). Some of these later services can, however, be connected to other activities that humans value and therefore valued through that relationship. Individuals may also get "value" from simply knowing that the forest exists - thus, the "existence value" of the forest is also a "service" provided by the forest and part of the benefits of reducing pollution levels is the value to all individuals of reducing effects that negatively impact the existence of the forest. Sections 4.1.4 and 4.2.2. discuss these concepts further in identifying categories of benefits and specific benefit endpoints associated with reducing mercury contamination.[1]

The goal of a benefits analysis for an environmental regulation is to identify all of the goods and services that are affected by a change in the level of environmental pollution (both directly and indirectly), estimate the change in the quality and/or quantity of those goods and services consumed by society resulting from the change in the level of environmental pollution, and then estimate the value to society of that change in the quality and/or quantity of goods and services consumed.

4.1.2 Measuring Benefits - The Concept of Willingness-to-Pay

Willingness-to-pay (WTP) is the concept used by economists to measure the value of different goods and services to society. WTP is defined as the maximum amount of money an individual would pay such that the individual would be indifferent between having the good or service and having kept the money. WTP values reflect individuals' preferences. Because preferences are likely to vary from one individual to another, WTP values for different goods and services will vary from one individual to another. The total social value of a good or service is the sum of the WTP values of all individuals who consume the good or service.

For consistency with how goods and services are traded through markets, economists measure the benefits of a regulation, or the change in social welfare associated with a regulation, in dollar terms using WTP. In the case of an environmental regulation, the benefits are measured in terms of the WTP for the improvement in the quality or quantity of the good or service (or alternatively, the change in the WTP for the good or service with and without the improvement). For both market and non-market goods and services, the total value of improvements in the quality or quantity of those goods and services is the sum of the change in the dollar amount that each individual is willing to pay for each good and service across all individuals (i.e., the change in WTP).

The monetary value or WTP for goods and services sold through markets can be approximated by the sum of predicted changes in "consumer and producer surpluses."[2] These "surplus" measures are standard and widely accepted terms of applied welfare economics, and reflect the degree of well-being enjoyed by people given different levels of goods and prices. For goods and services that are not sold through markets (such as many recreational activities), other methods are used to estimate the WTP for these goods and services.

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[1] There are several questions within this framework, however, that require additional consideration in a developing a rigorous framework for assessing the benefits of regulation. One issue is the fact that improvements in environmental quality will affect many goods and services used or consumed by society in multiple ways (e.g., through direct and indirect pathways). A future comprehensive benefit analysis, therefore, will need to develop a framework for analyzing simultaneous changes to multiple goods and services that accounts for the fact that a particular good or service may be affected through multiple pathways and that there may be interactive effects between changes in particular goods and services that arise from a change in environmental quality. Resolution of these issues and the development of a comprehensive framework for analyzing the benefits of regulations that have nationwide effects requires input on both a theoretical and an applied level.

[2] A comprehensive benefit analysis will need to evaluate the appropriateness of using changes in consumer and producer surplus to approximate the true change in welfare associated with a quality change.

4.1.3 Introduction to Techniques Used to Estimate WTP

There are a wide variety of economic valuation techniques that can be used to estimate WTP for a change to a specific good or service. This section is intended to introduce the different types of approaches that are available for estimating WTP and provide a general sense of what data might be required to implement each type of analysis for use in planning future benefits analyses. More detailed information on the specific techniques is not provided here because the current purpose of this chapter is to identify a framework for a more thorough benefits analysis and not to implement a specific approach for estimating the benefits of reducing mercury emissions. Furthermore, there are numerous sources of information on each of these techniques, including many textbooks as well as U.S. EPA materials.[3]

The techniques available for estimating WTP differ by their approach for eliciting the value that an individual (an ultimately, society) places on the good or service provided, their data requirements, and their relative advantages and disadvantages (e.g., accuracy, ease of use, acceptability). Table 4-1 lists the variety of techniques used to measure WTP and divides them into four categories based on two characteristics:

(1) Does the technique use data or observations of people acting in real-world situations (i.e., revealed preferences) or of people responding to hypothetical situations (i.e., stated preferences)?

(2) Does the technique yield monetary values directly (i.e., direct estimation of WTP) or must monetary values be inferred based on a model of individual behavior (i.e., indirect estimation of WTP)?

Table 4-1
Categorization of Approaches for Estimating WTP

Approach	Direct Estimation of WTP	Indirect Estimation of WTP
Revealed Preferences Approach	Market Price/Quantity (Estimated Supply/Demand) Cost of Illness User Fees Simulated Markets Replacement Costs	Value of a Statistical Life Travel Cost Studies Hedonic Studies Random Utility Models Avoidance/Defensive Expenditures Referendum Voting
Stated Preferences Approach	Contingent Valuation Studies	Contingent Ranking Contingent Activity Contingent Referendum Conjoint Analysis

Direct, revealed preference approaches require data on real-life choices made by individuals regarding their consumption or use of a particular good or service. These approaches assume that an

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[3] For a comprehensive summary of approaches available to measure market and non-market ecological benefits, see U.S. EPA, Ecological Benefits assessment Framework, draft, prepared for EPA Social Sciences Discussion Group, EPA Science Policy Council, 1996.

individual who is free to choose the quantity of the good or service they desire at a specific price will choose the quantity that maximizes their welfare (or benefit) given the constraints placed upon them by the market (e.g., limited individual income, availability of substitutes and other goods, limited availability of specific goods or services). Thus, these types of approaches can only be applied for goods and services bought and sold through markets. Estimating market supply and demand for a good or service requires time series or cross-sectional data on the price of the good or service, the quantity sold and consumed, and detailed cost and revenue information for representative producers.

Indirect, revealed preference approaches rely on the relationships between the value placed on a good or service affected by environmental quality that is not traded through markets and the other real-world choices that individuals make. These approaches typically require modeling of these relationships to infer values for the non-marketed good or service. Travel cost studies, for example, have been used to estimate the value of a particular recreational activity, such as fishing, based on the time and expense required to partake in that activity. Similarly, in using the avoidance/defensive expenditures approach, the cost of a particular event (or benefits of preventing an event), such as flooding, is estimated based on current expenditures to prevent or reduce the negative impact of an event. Because of the need to model complex relationships in order to infer values for a specific good or service, these techniques tend to have fairly significant data needs, which may include: price and quantity information for consumption of related market goods and services; use or consumption information for the good or service one wants to value; characteristics of the good or services as well as substitute goods and services; and characteristics of users.

Direct, stated preference approaches, or contingent valuation approaches, involve asking people directly about the values they place on certain effects or changes. Some direct approaches used to determine an individual’s willingness to pay for a specific improvement include:

- Asking each individual in a sample directly how much they would be willing to pay to ensure/prevent a change;

- Asking each individual in a sample whether they would be willing to pay some specific amount of money to ensure/prevent a change, varying the amount of money across the sample; and

- Conducting a bidding game with each individual in a sample to determine the maximum amount each would be willing to pay to ensure/prevent a change.

By aggregating over the sample, an analyst can estimate a demand curve for the specific change, which can then be used to estimate total WTP.

Indirect, stated preference approaches are also contingent valuation studies, except that the individuals questioned are not asked directly about the value they place on a specific change, but rather are asked to make a decision about another situation that depends or otherwise relates to the value they would place on the specific change to be valued. The responses to these questions are then used to draw inferences about the value of changes to the non-market good or service of interest. For example, individuals may be asked:

- To rank combinations of varying quantities or qualities of goods, including both market goods, which have prices associated with their use, and non-market goods, for which the analyst wants to estimate the value; or

- To estimate the change in their current level of activity or use of a specific good or service under alternative scenarios in which the availability and quality of the good or service is varied.

Benefits transfer analysis can often be used to estimate the value of a particular change when it is not possible to use one of the above techniques. This approach involves identifying other valuation studies of similar changes or effects and using, or transferring, the value from the previous study(ies) to the new change or effect of concern. In some instances, additional data might be used to adjust the value estimate to better suit the new situation or to correct for errors introduced in the original study. This technique can be used to develop rough or "order of magnitude" estimates of the potential benefits from an action, with only slight adjustments in the application of a value from a previous study to the new situation, as well as more accurate estimates of the benefits, with more sophisticated adjustments to better represent the new situation.

4.1.4 Types of Benefit Endpoints - Goods and Services Affected by Environmental Quality

To conduct a benefits analysis, the types or categories of benefits that apply need to be defined. There are numerous types of goods and services affected by the level of environmental quality that have value to some or all individuals in society. The purpose of the benefits analysis, as discussed above, is to quantify and, when possible, monetize the increase in the value of these goods and services as a result of an action that improves environmental quality. The benefits typology shown in Table 4-2 summarizes the various categories of benefits in terms of the types of services that can be protected or improved by protecting or improving environmental quality.

Table 4-2
Typology and Examples of Uses/Services Affected by Changes in Environmental Quality

Direct, Market Uses Food Products (fish, crops, animals) Building Materials (timber, stone) Fuel (timber, coal, oil) Drinking Water Chemicals/Minerals Medicine	Direct, Non-Market Uses Human Health Recreational Activities Recreational/Subsistence Fishing Recreational/Subsistence Hunting Animal Viewing (bird watching) Boating, Swimming, Beach Use Hiking/Camping
Indirect, Non-Market Uses Water Filtration Flood Control Pollution Mitigation Soil Generation Nutrient Cycling Wave Buffering Habitat Value/Biodiversity	Non-Market, Non-Use Values Existence/Intrinsic Value Cultural/Historical Value Bequest/Philanthropic Value Option Value

As reflected in Table 4-2, benefits are typically categorized according to whether they involve some form of direct or indirect use by mankind. In addition, for the purposes of valuing improvements to each benefits category, it is useful to think about whether a market exists to provide each good or service that is affected.

Use benefit endpoints can embody both direct and indirect uses and include both consumptive and nonconsumptive activities. Direct use endpoints include those goods and services which are sold through open markets, such as commercial fish and timber, as well as non-market goods and services, such as recreational fishing, swimming, and boating. Human health also represents a direct service used or valued by mankind that can be impaired by reductions in environmental quality. Use benefit endpoints may also include some “passive use” or “indirect, non-market use” services, which include services and ecological processes that indirectly benefits humans, such as providing pollination services, wildlife habitat, flood control, or filtering run-off waters. In most applications to pollutant reduction scenarios, the use benefit endpoints most often examined are those related to human health risk reductions; recreational fishing, boating, and swimming; visibility; material damages; effects on crops; and effects on endangered species.

Benefits analyses often are unable to quantify the effects on indirect use benefit endpoints, although they may represent a potentially significant quantity of benefits. This omission is due to the difficulty in measuring (and attributing) changes to these endpoints to the action being taken to improve environmental quality as well as the difficulty in valuing these types of uses and services. One approach for valuing these types of services is to link them to direct uses that can be valued. For example, the value of recreational fishing activities (a direct use) may increase due to improvements in the quality of fish nursery habitat (an indirect service) provided by nearby wetlands. This requires fairly detailed information on the ecological relationships and interactive processes by which different functions and services are related.

Non-use (intrinsic) benefit endpoints are values an individual may have that are unrelated to his or her own exposure or use of any good or service. Improved environmental quality can be valued by individuals apart from any past, present, or anticipated future use of the goods and services affected by changes in environmental quality. Non-use values may be related to the desire to ensure that a clean environment be available for the use of others now and in the future, benevolence toward friends and relatives, sympathy for people and animals adversely affected by environmental degradation, or a sense of environmental responsibility or stewardship. Such non-use values may be of highly significant magnitude in terms of the total value of preserving ecological resources, but the magnitude of changes in non-use values and the ability to measure changes in such values from improvements in environmental quality has been highly debated.

4.1.5 Issues Associated with Benefits Analysis

This section discusses several relevant issues and potential limitations for a benefits analysis of an environmental regulation.

4.1.5.1 Benefits are Anthropocentric

This conceptual economic foundation for measuring benefits is anthropocentric -- all benefits arise from how environmental changes are perceived and valued by people in present-day values. Thus, all near-term and temporally distant future physical outcomes associated with reduced pollutant loadings need to be predicted and then translated into the framework of present-day human activities and concerns. Furthermore, the assessment of benefits relating to changes in ecosystem function or health and to effects on lower-level species is severely limited in this context because of the lack of clear relationships and information to model the relationships between these effects and changes in the values held by mankind for related environmental goods and services.

4.1.5.2 Aggregation Issues

There are numerous issues that arise in aggregating individual WTP estimates to develop a national-level benefits estimate. Although many of these issues require additional theoretical deliberation to determine the most appropriate approach, below are three aggregation issues that warrant consideration in the applied context of estimating benefits:

- How to sum benefits across benefit endpoints;
- How to address potential double-counting when using multiple techniques to measure the WTP for changes in related benefit endpoints or overlapping effects; and
- How to determine the affected population for calculating social WTP based on estimates of individual WTP and how to sum WTP over the affected population.

4.1.5.3 Effect of Changes in Real Income on Valuation Estimates

A thorough benefits assessment should consider how the benefits estimates would be affected by growth in real income. Economic theory argues that WTP for most goods will increase if real income increases. Furthermore, a benefits estimate based on the change in consumer surplus is a better measure of the change in social welfare when the demand for a particular good or service being examined does not vary with income level. Thus, the effect of changes in real income may also be important to the consistency and accuracy of the benefits estimates. Accounting for these effects requires information on the income elasticity of the WTP estimates for specific benefit endpoints. Therefore, future valuation studies conducted to estimate WTP for specific benefit endpoints should include a methodology for estimating the income elasticity associated with those WTP estimates or otherwise address the potential impact of changes in income levels on the benefits estimates derived by that analysis.

4.1.5.4 Cumulative and Lagged Effects and the Role of Discounting

Because mercury is known to bioaccumulate over time and up through the food chain, reductions in mercury pollution levels will have cumulative and lagged effects. As a result, a given reduction in pollution concentrations in one year will confer benefits not only in that year, but in future years as well as environmental concentrations (e.g., sediment concentrations, concentrations in predator fish species) fall over time. Furthermore, the benefits of a reduction occurring in any single year will not be fully realized until long after the year in which the exposure occurs. Even though mercury emissions are reduced, elevated levels of mercury will remain in the sediments (although concentrations will decline over time) and continue to contribute to fish uptake and exposures up the food chain to larger species, including humans who consume fish. A thorough benefits assessment needs to consider the role of lagged or future effects and determine how best to account for these types of effects. This may include better characterizing the stream of benefits based on scientific information on changes in environmental concentrations over time from a specific reduction in emissions and determining an appropriate discounting scheme for comparing changes in future effects against changes in current effects.[4]

The simplest approach for a benefits analysis is to examine the total change in each benefit endpoint if all adverse effects of exposure to elevated mercury levels are eliminated without consideration for when, in time, these benefits will be realized. That is, the analysis assumes that any reductions in mercury emissions and deposition result in immediate reductions in sediment concentrations and exposures of higher level species, including humans (i.e., not attempt to account for delays in the realization of benefits). Another approach might be to model the delay in the effect of emissions reduction on exposure levels and estimate benefits as they accrue over time.

When the benefits of an action accrue over time, such as with lagged and/or cumulative effects, the role and importance of discounting needs to be considered in the context of the benefits assessment. The discount rate used and the time period for comparison can have significant effects on the magnitude of the benefits estimate and the conclusions of the benefits assessment, especially if the benefits and costs occur in different points in time, as with lagged and cumulative benefits. Traditionally, present value costs and benefits have been calculated using the shadow price of capital or the consumption rate of interest as the discount rate. These may be appropriate or inappropriate discount rates, however, depending on the assumptions made regarding the flow of capital and the value of future consequences (e.g., are future values adjusted upward to reflect increased value due to increased scarcity). Furthermore, a different discount rate (or even no discounting) might be appropriate for intergenerational effects. With respect to the time period of comparison, the analysis might choose to translate future values into present ones - the traditional approach - or alternatively, annualize the costs and benefits or accumulate benefits (and costs) forward to some future time period. Discounting, and other issues including baselines, uncertainty, non-monetized effects, equity, and valuing lethal risks, are currently being examined by U.S. EPA's Office of Policy, Planing, and Evaluation (OPPE) in support of the preparation of a revised guidance for preparing Economic Impact Analyses and Regulatory Impact Analyses. These guidelines, therefore, should be evaluated in the course of a comprehensive benefits assessment for reducing mercury contamination.

4.1.5.5 Variability and Uncertainty

The variability and uncertainty associated with specific estimates is an important consideration in a thorough benefits assessments. Variability and uncertainty are introduced in estimating emissions changes, modeling the fate and transport of emissions (e.g., air modeling), estimating effects, and valuing the effects (or changes in the effects). Variability and uncertainty arise from the inherent variation of natural processes as well as from limited knowledge about the many relationships between emissions and exposures and effects. Distributional information from both the risk assessment and the economic valuation study should be carried through in the final benefits assessment.

There are several treatments of variability and uncertainty available that can be applied in the estimation of benefits. The appropriate approach to characterize and quantify the degree of variability and uncertainty associated with a specific estimate will depend on the objectives of the analysis and the needs of the decision-makers. Depending on the particular valuation approach being used to develop a

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[4] A RCG/Hagler-Bailly (1994) report says that little is understood about mercury decay rates, but cites an analysis by NJ DEP that suggests that it will require about 20 years to cover contaminated sediments with new sediments to a depth of 5 cm (but offers no equations that quantify impacts during the decay period), p. IX-12.

benefits estimate, the uncertainty and variability associated with the results of that approach might be addressed by:

- Presenting the benefits estimates as ranges based on a plausible set of input values (e.g., estimated risk, economic value estimates);

- Conducting sensitivity analyses to examine the potential variation in the benefits estimates under different assumptions regarding the level of effects;

- Using Monte Carlo analyses or other probabilistic techniques using probability distributions for the inputs to the analysis (e.g., risks, values) to estimate a probability distribution for the output (e.g., benefits);

- Discussing and/or incorporating expert judgement regarding the potential range of effects and/or benefits (e.g., Delphi methods); and/or

- Using meta-analysis to combine estimates of inputs (e.g., risks, values) or outputs (e.g., benefits estimates) from multiple studies.

Accounting for uncertainty and variability can provide a more complete characterization of the distribution of benefits than point-estimates. Nonetheless, many sources of uncertainty will likely remain unquantified. Thus, qualitative descriptions of the limitations and known omissions, biases, and data gaps are also an important component of a thorough benefits analysis.

4.2 Benefits Assessment for Reducing Mercury Contamination

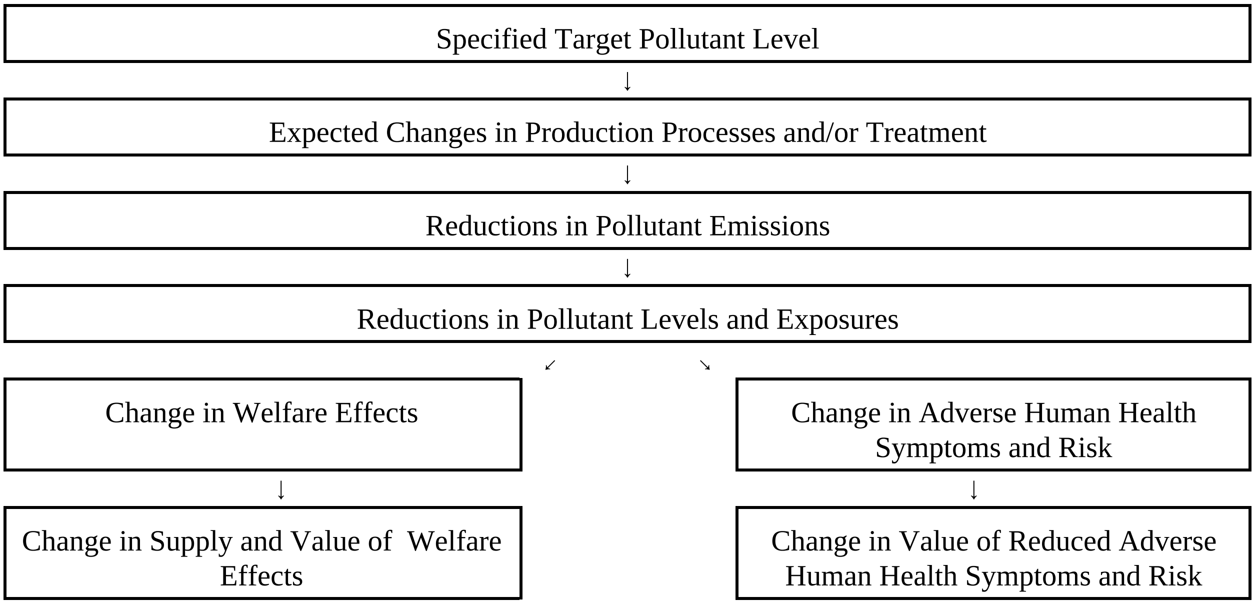
This section discusses the specifics of assessing the benefits of reducing mercury contamination applying the concepts discussed above. The goal is to establish a clear framework for conducting a comprehensive economic benefits assessment. Performing a comprehensive benefits analysis for mercury contamination will require a coordinated effort across the Agency to take advantage of the knowledge and ongoing work on mercury and benefits assessment within various offices. This section discusses the specific steps for moving forward with a thorough benefits assessment, identifies readily available information on the effects of mercury contamination and possible approaches to assessing the benefits of reducing those effects, and highlights relevant issues to be considered during this process.

4.2.1 Steps to a Benefits Assessment

Conducting a benefits analysis for anticipated changes in air emissions is a challenging exercise. Assessing the benefits of a regulatory action requires a chain of events to be specified and understood. As shown in Figure 4-1, which illustrates the causality for air quality related benefits, these relationships span the spectrum of: (1) institutional relationships and policy-making; (2) the technical feasibility of pollution abatement; (3) the physical-chemical properties of air pollutants and their consequent linkages to biologic/ ecologic responses in the environment, and (4) human responses and values associated with these changes.

The first two steps of Figure 4-1 reflect the institutional and technical aspects of implementing improved process changes or pollutant abatement. The benefits assessment framework presented in this document begins by assuming reductions in mercury concentrations. The estimated changes in these concentrations are directly linked to the estimated changes in precursor pollutant emission reductions. The health and welfare benefits to be estimated represent the identifiable benefits expected to result from the application of control measures.

Figure 4-1
Example Methodology of a Benefits Analysis



Other information necessary for the analysis are the physical and chemical parameters and the consequent improvement in the environment (e.g., concentration response data). Finally, the analysis reaches the stage at which anthropocentric benefits concepts begin to apply, such as reductions in human health risk and reductions in fish advisories. These final steps reflect the focal point of the benefits analyses and are defined by the benefit endpoints described below. Potentially relevant benefit endpoints are described qualitatively, and where possible, quantitatively.

4.2.2 Identification of Benefit Endpoints for a Mercury Benefits Analysis

As discussed in detail in other portions of this report, mercury contamination in the environment presents a number of potential risks to human health and ecological resources. The value to society of reducing the potential risks to human health and ecological resources represents the potential benefits of regulatory action to reduce mercury pollution.

Benefit endpoints are defined as the specific human health or ecological resource measures by which reductions in risk, and thus benefits, will be realized. Tables 4-3 and 4-4 present a fairly comprehensive list of potential benefit endpoints based on current scientific information on the (potential) effects of mercury on human health and ecological resources. For many endpoints, the benefit accruing to an individual may be the reduction in the potential risk of such an adverse effect occurring, rather than the actual reduction in the incidence of the effect. This is particularly true for benefit endpoints for which there is currently limited information regarding the expected extent or magnitude of current or future effects (e.g., developmental impairment in children, reduced ecosystem integrity, lost agricultural productivity). There may be additional health and ecological benefits of reducing mercury exposures, in addition to those listed here, that U.S. EPA is not currently aware of, and thus, are not included in these exhibits.

Tables 4-3 and 4-4, in addition to identifying specific benefit endpoints that may be examined in future benefit analysis, provide information on possible measures of changes in each benefit endpoint and on the probable magnitude of the effect associated with each benefit endpoint (e.g., size of the at-risk population, existence of studies finding significant impacts). In addition to starting to draw the link between the risk assessment and the economic benefit analysis, the column on "probable measures of change" included in Tables 4-3 and 4-4 may also serve as a starting point for identifying future data and research needs for conducting a comprehensive benefit analysis.

4.2.2.1 Discussion of Health Effects

As discussed in more detail in Volume V of this Report, the most significant demonstrated health effects of mercury are kidney and neurological damage and developmental effects.

Exposure to all forms of mercury, including organic mercury compounds, can damage the kidney. The primary mechanism of damage appears to be a toxic effect on the renal tubules. Symptoms include blood or elevated levels of protein or cholesterol in the urine, edema, and inability to concentrate urine. The effects of mercury toxicity are generally reversible, but in severe cases acute kidney failure has occurred (HHS 1994).

MAJOR HEALTH EFFECTS OF MERCURY

- **Kidney Damage**
- **Neurological Damage**
- **Developmental Effects**

The nervous system is highly sensitive to mercury. Neurological effects have been observed following exposures to all forms of mercury through inhalation, ingestion, and dermal absorption. Specific symptoms include the following:

- Prickling or tingling of the skin, numbness, hyperactive reflexes, and slowed nerve conduction velocities;
- Tremors of the hands and sometimes other parts of the body;
- Neuromuscular effects, such as weakness, twitching, and muscular atrophy;
- Loss of cognitive and motor function;
- Sensory effects, including hearing loss and loss of visual field;
- Emotional instability, including irritability and loss of confidence; and
- Insomnia, memory loss, headaches, and hallucinations.

Table 4-3
Summary of Human Health Benefit Endpoints

Effect Category	Target Population(s)	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Developmental/ Neurological	Children exposed *in utero* Young Children	Delayed motor and verbal development	Number of incidences with population description Potential number of individuals at risk	One to three percent of women of child bearing age (15-44) are estimated to regularly consume more than 100 g fish/day. There are approximately 54 million children under age 15 in the U.S. (based on 1990 Census data); approximately 665,000 are estimated to consume an average of 100 g fish/day.
		Physical and mental disability resulting from damage to the central nervous system		
		Acrodyma syndrome		

Table 4-3 (continued)
Summary of Human Health Benefit Endpoints

Effect Category	Target Population(s)	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Neurological	Native American populations Subsistence Fishers Young Children General Population	Loss of cognitive and motor function	Number of incidences with population description Potential number of individuals at risk	Evidence from a study of Wisconsin tribes shows significant risk of adverse effects (EPA 1992b). No national estimates of risk or exposure of Native American populations.
		Hearing and vision loss		
		Memory loss		
		Emotional instability		
		Insomnia		
		Altered reflexes, hand tremors, weakness, twitching		

Table 4-3 (continued)
Summary of Human Health Benefit Endpoints

Effect Category	Target Population(s)	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Kidney Toxicity	Native American populations Subsistence Fishers Young Children General Population	Temporary disability due to kidney damage Permanent disability due to kidney failure Death due to kidney failure	Number of cases with population description Potential number of individuals at risk	Evidence from a study of Wisconsin tribes shows significant risk of adverse effects (U.S. EPA 1992b). No national estimates of risk or exposure of Native American populations. One to three percent of women of child bearing age (15-44) are estimated to regularly consume more than 100 g fish/day.
Cancer	General population	Cancer (incidence or probability/risk)	Number of cases with population description Potential number of individuals at risk	Low probability of incidence given mercury is Class D carcinogen.

Table 4-4
Summary of Ecological/Welfare Benefit Endpoints

Effect Category	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Direct, Use Values			
Recreational Fishing	Reduced number of fishing trips	Number of fish advisories per state Percentage of fishing area per state covered by advisory Number of fishermen Changes in expenditures with and without advisory	Currently 39 states and some Tribes have fish advisories; 10 states have statewide advisories.
	Lost value per trip due to fish advisory		
	Lost value due to inability to consume fish		
Commercial Fishing	Lost value of fish exceeding maximum allowable concentration	Quantity of fish exceeding allowable max concentrations Quantity fish demanded in state/area with and without fish advisory	Currently 39 states and some Tribes have fish advisories; 10 states have statewide advisories.
	Reduced demand for all fish due to perceived health threat		

Table 4-4 (continued)
Summary of Ecological/Welfare Benefit Endpoints

Effect Category	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Subsistence Fishing	Lost value of fish no longer consumed	Estimated number of subsistence fishers in areas covered by fish advisories Reduction in fish consumption levels with and without advisories	Evidence from a study of Wisconsin tribes shows significant levels of subsistence fishing (U.S. EPA 1992b). No national estimates of subsistence fisher populations.
	Lost nutritional value		
	Lost cultural value associated with subsistence fishing activity		
Timber	Reduce growth/productivity of commercial forests	Change in growth/ productivity with different levels of exposure	Unknown. Evidence of adverse effect of mercury on growth rates of plants.
Forest Recreational Activities	Reduced number and/or value of recreational activities due to reduced quality of surrounding plants	Change in health of forests/plants with different levels of exposure	Unknown. Evidence of adverse effect of mercury on growth rates of plants.
Agricultural	Reduced growth/productivity of crops	Change in growth/ productivity with different levels of exposure	Unknown. Evidence of adverse effect of mercury on growth rates of plants.

Table 4-4 (continued)
Summary of Ecological/Welfare Benefit Endpoints

Effect Category	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Recreational/Commercial Hunting/Trapping	Reduced or lost commercial value of target species	Change in population and/or catch rate for target species Change in participation rates	Unknown. Evidence of effects on furbearers and larger mammals, but lack hard evidence linking changes in population numbers with mercury exposure levels. Approximately 10 million mink live in the North American continent and 300 thousand river otters live in the U.S.
	Reduced value of recreational hunting/trapping activity with reduced population of target species		
Recreational Bird Hunting	Reduced number of trips for target species	Change in population and/or catch rate for target species. Change in participation rates.	Unknown. Evidence of effects on some bird species, but lack hard evidence linking changes in population numbers for target species with mercury exposure levels. An estimated 10 to 12 thousand eagles, 10 to 20 thousand osprey, and 170 thousand belted kingfishers inhabit the lower 48 United States.
	Lost value per trip due to reduction in target species		

Table 4-4 (continued)
Summary of Ecological/Welfare Benefit Endpoints

Effect Category	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Bird/Animal Viewing	Reduced value of recreational activities with lower probability of viewing target species: - Florida panther - Wood stork - Loons - Eagles, osprey, kingfisher - Furbearers (e.g., minks, otters) - Other	Change in population and/or view rate for target species. Change in participation rates.	Unknown. Evidence of effects on some bird and animal species, but lack hard evidence linking changes in population numbers for target species with mercury exposure levels. See above population estimates for some specific species.
Indirect Use and Non-Use Values			
Cultural/Religious Value	Reduced cultural/religious value due to fish advisories, not associated with use	Contamination levels of “traditional” waters and/or species	Unknown. Recent studies of Wisconsin tribes indicate cultural/religious impacts may be significant.
Existence Value of Specific Species (includes option value, bequest value, scarcity value, as well as existence value)	Adverse effects on specific species (individuals and populations): - Florida panther - Wood stork - Loons - Eagles, osprey, kingfisher - Furbearers (e.g., minks, otters) - Other	Change in populations Change in reproductive success (survivorship of young) Change in mortality rates	Unknown. Some evidence of negative effect of mercury exposure in certain species, on an individual level. Little known on population effects. See above population estimates for some specific species.

Table 4-4 (continued)
Summary of Ecological/Welfare Benefit Endpoints

Effect Category	Benefit Endpoint (Adverse Effect Reduced)	Possible Measures of Magnitude and Change in Endpoint	Magnitude of Effect (Size of At-Risk Population)
Biodiversity	Adverse effect on biodiversity	Change in number of viable species in an area	Unknown. Some evidence of negative effect of mercury exposure in certain species, on an individual level. Little known on population effects. See above population estimates for some specific species.
Stewardship Value (includes moral obligation to reduce harm to ecological resources)	Adverse effects on specific species or in general	Change in atmospheric/ deposition levels and/or water concentrations Change in body concentration levels	Unknown. Some evidence of negative effect of mercury exposure in certain species, on an individual level. Little known on population effects. See above population estimates for some specific species.
Preservation of Ecosystem Health (includes maintaining the integrity of predator/prey relationships)	Adverse effect on ecosystem health	Change in predator/prey populations Change in other measures of ecosystem health.	Unknown.

Some of these neurological effects may be reversible if the source of mercury exposure is removed. However, some changes may be permanent; autopsies have shown degenerative changes to the brains of some patients poisoned by mercury.

Mercury, especially ingested organic mercury compounds, is particularly toxic to developing nervous systems. Effects on exposed fetuses and children can range from delayed motor and verbal development to severe brain damage. Infants born to mothers exposed to mercury over long periods or at high levels show permanent damage to the central nervous system, including mental retardation and symptoms similar to cerebral palsy (HHS 1994). A study of a population in New Zealand observed an inverse correlation between IQ in children and hair mercury levels in their mothers (Kjellstrom, et. al., 1989).

Mercury has *not* been demonstrated to cause cancer in humans. Some animal studies have suggested that mercury may cause tumors in mice or rats; however, no similar effects have been documented in people. U.S. EPA's cancer classification for mercury is D ("not classifiable") (HHS 1994). Mercury exposure generally does not lead to death from other causes either. Infrequently, inhalation or ingestion of very high doses may lead to death from respiratory failure, kidney failure, cardiovascular collapse, severe gastrointestinal damage, or central nervous system toxicity (HHS 1994). However, such effects are not expected to occur from exposure to the levels of mercury currently present in the environment in the United States.

4.2.2.2 Discussion of Ecological Effects

Mercury can adversely affect ecological systems at various levels of organization: the individual organism level, population level, and community or ecosystem level. While the effects to populations, communities, and ecosystems are of primary concern for most species, individual effects are also of interest because they may cause effects at higher levels of organization, especially population effects in vulnerable or reduced populations such as threatened and endangered species, raptors, and furbearers.

Effects of mercury on individual organisms are the effects most commonly tested and reported in the literature. Exposure to mercury has been found to cause adverse effects in plants, birds, fish, and mammals. These effects vary among different types of organisms. Individual effects on fish include death and sublethal effects, such as reduced reproductive success, impaired growth and development, altered blood chemistry, osmoregulatory effects, effects on oxygen exchange, and behavioral abnormalities that can reduce ingestion rates and predatory success. In birds, mercury can cause death or sublethal effects, including liver damage, kidney damage, neurobehavioral effects, effects on enzyme systems, reduced cardiovascular function, impaired immune response, reduced muscular coordination, impaired growth and development, altered blood and serum chemistry, and reproductive effects. Reproductive effects are of primary concern in birds (especially in raptors) and can occur at mercury levels in the diet one-fifth of the lethal dose. Effects of mercury on mammals include death, reduced reproductive success, renal damage, and neurological effects, such as paresthesia, visual disturbances, mental disturbances, hallucinations, ataxia, hearing defects, stupor, and coma (Roelke et al. 1991, Eisler 1987, Klaassen et al. 1986). Mercury effects on plants include decreased growth, root damage, and decreased photosynthetic activity (Eisler 1987, Lindqvist et al. 1991).

Population effects are less studied, but can include increased mortality rates (in any exposed populations) and decreased reproductive rates (particularly in bird and mammal populations). For example, methylmercury can cause embryo death in bird eggs (Lindqvist et al. 1991).

Very little information is available on the effects of mercury on communities and ecosystems. In general, mercury may be expected to have a significant effect on communities and ecosystems because there is evidence that mercury can have adverse effects on all components of an ecosystem and because mercury biomagnifies up the food chain. Thus, mercury may have greater impacts on top-level predators, which may distort community or ecosystem relationships.

Volume VI presents the results of the ecological risk assessment for anthropogenic mercury emissions. Volume VII presents estimates of the size of the piscivorous wildlife populations (for select species) potentially affected by mercury contamination.

Effects on Fisheries

Economic and social values associated with fisheries can be grouped into five general categories: commercial, recreational, subsistence, cultural, and non-use values. Mercury and other toxic contaminants can decrease these values by reducing U.S. fishing opportunities and reducing the quality of the fishing experience.

One concrete way in which fishing opportunities may be reduced is through fish consumption bans or advisories, enacted by the Food and Drug Administration (FDA) (bans on sale of fish with mercury concentrations in excess of 1 ppm) or by state governments (advisories and bans on recreational and subsistence fishing). Mercury fishing bans can reduce dollar values associated with commercial fisheries directly, when fish are illegal to sell, or indirectly, when public perceptions that fish are undesirable to buy and eat lowers the market value of fish not covered under a ban (NWF 1993). Both of these effects lead to a decline in the economic benefits derived from a commercial fishery.

Mercury contamination and advisories can reduce the recreational value associated with fishing because anglers may react by:

(1) Ceasing fishing, thus relinquishing the total value of the fishing experience;

(2) Changing to new fishing locations or new species of fish, presumably reducing the value of the fishing experience;

(3) Catching and releasing fish, presumably reducing the value of the fishing experience; or

(4) Ignoring the advisory, potentially resulting in increased health risks (see Section 4.2.2.1).

Mercury contamination and advisories can decrease the value of subsistence fisheries by causing subsistence fishers to switch to substitute foods, which cost more or are less enjoyable, or forgo consumption (representing nutritional losses if substitute foods are not purchased). Cultural values associated with fishing can be reduced by mercury contamination and advisories by diminishing or banning traditional fishing and fishing-related activities. Non-use values may be reduced by the willingness of non-users to pay simply to know that waters are fishable (and/or less polluted in general).

<u>Fishing Bans and Advisories Due to Mercury Contamination</u>. Mercury is the only metal that the FDA regulates in fish. FDA presently bans the commercial sale of finfish or shellfish when mercury levels in tissue exceed 1 μg mercury/g tissue (i.e., 1 ppm).

Some states use the FDA's limit of 1 ppm mercury in fish tissue as a trigger for issuing consumption advisories, but many other states have lower trigger levels, commonly 0.5 ppm. Though fish consumption advisories and bans vary from state to state, they generally:

(1) Provide information to recreational and subsistence anglers about contaminant levels in finfish and shellfish caught from specific water bodies; and

(2) Recommend how much fish tissue from particular fish species the general population and specific subpopulations should consume.

State mercury fish advisories often provide more stringent consumption recommendations for sensitive subpopulations, such as children. Because mercury can pass through the placenta or mother's milk to sensitive fetuses or infants, stringent consumption recommendations are often provided for pregnant women, nursing women, and women who plan to have children.

There are currently 39 states that have at least one waterbody under mercury advisory; 10 of which have state-wide advisories in place. Chapter 3 of Volume 3 provides a detailed discussion of fish advisories and mercury concentrations in various types of fish.

Effects on Threatened and Endangered Species

Effects of mercury on individual members of threatened and endangered species are of particular concern. Population numbers of these species are so low that effects on an individual animal or plant can translate into an important effect on the species population and even the community as a whole. For example, the death of one reproductive female in a population including only three reproductive females could be a major step toward extinction from the region or global extinction of that species. Extinction of the species (e.g., a top predator species) could in turn change the community's species composition dramatically.

Benefits of preserving species from extinction fall into four categories (FWS 1993b):

- Protection of ecosystem integrity. Protecting natural diversity protects the integrity of ecosystems. Moreover, the decline of a sensitive species can alert us to the decline of the entire system. For example, the rapid decline in raptors such as bald eagles and peregrine falcons served to alert humans to previously unforeseen adverse effects of the pesticide DDT.

- Preservation of human uses. Plants and animals can provide a wide variety of known and as-yet-undiscovered uses for humans, including agricultural, medicinal, and industrial uses. Some plants and animals also provide the service of cleaning our environment. For example, filter feeders, such as the endangered Higgins' eye pearly mussel, improve water quality.

- Maintenance of human aesthetic pleasure. Many people prize threatened and endangered species for their aesthetic values, to be enjoyed by themselves as well as generations to come.

- Moral and responsible stewardship. Some believe that humans have a stewardship obligation to prevent our activities from exterminating other forms of life.

Effects on Other Species

Many of the benefits of reducing adverse effects described above for threatened and endangered species are expected to also apply to other related species (e.g., other bird and mammal species). At this time, however, there is little known about the actual individual or population effects of mercury contamination on other non-threatened or endangered wildlife and plant species. With respect to non-threatened or endangered wildlife species, individual effects are less important than population effects. For example, if one bird in a population of 1,000 is adversely affected by mercury contamination, no population effects would be expected. However, if 500 out of 1,000 birds experienced an adverse effect such as decreased reproductive success, the reproductive rate of the population as a whole would decline, and the population could be substantially affected.

Ecosystem and Community Effects

Mercury effects on higher levels of organization (i.e., communities and ecosystems) are not well studied. The few studies that have examined indicators of ecological health in mercury-contaminated ecosystems provide no strong evidence of mercury damage to community and ecosystem parameters (e.g., productivity, species diversity).

The CAA and other statutes, through the requirements to protect natural and ecological systems, indicate that these are scarce and highly valued resources. In a recent attempt to estimate the “marginal” value (changes in quantity or quality) of ecosystem services, Costanza *et al*. (Nature, 1997) state that policy decisions often give little weight to the value of ecosystem services because their value cannot be fully monetized or quantified in commercial market terms. Costanza *et al*. warn that “this neglect may ultimately compromise the sustainability of humans in the biosphere.” Lack of comprehensive information, insufficient valuation tools, and significant uncertainties result in understated ecological benefits estimates. However, a number of expert biologists, ecologists, and economists (Costanza, 1997) argue that the benefits of protecting natural resources are enormous and increasing as ecosystems become more stressed and scarce in the future. Just the value of the cultural services (i.e., aesthetics, artistic, educational, spiritual and scientific) may be considered infinite by some, albeit in the realm of moral considerations.

4.2.2.3 Ancillary Benefits of Reducing Mercury Contamination

The ancillary benefits associated with reducing mercury contamination will depend on the types of controls used to reduce mercury emissions and exposures and how and where those controls are implemented. Because this analysis does not examine a specific control strategy for reducing mercury contamination, this benefits analysis does not consider the role of ancillary benefits, which potentially could be quite significant. A thorough benefits assessment conducted to help evaluate alternative control strategies should, however, thoroughly examine, quantify, and, when possible, monetize the ancillary benefits associated with each control strategy. These ancillary benefits can then be added to the benefits of the reduction in mercury achieved by the control strategy and weighed against the cost of the alternatives in selecting the most cost-effective approach.

4.2.3 Measuring Improvements to Benefit Endpoints

The extent to which improvements resulting from reduced mercury exposure for each specific benefit endpoint (i.e., reduction in the negative effect for each benefit endpoint described in Tables 4-3 and 4-4 can be measured and monetized will vary significantly. As a result, only some of the relevant human health and welfare effects will be quantified (expressed in terms of incidences reduced) and monetized (expressed in terms of dollars) in the benefits analysis.

The ability to measure the benefits of reducing the risks to a particular benefit endpoint depends on the availability of appropriate scientific information to discern the relationship between changes in mercury exposure and subsequent effects on the endpoint. Although current scientific information may be useful in identifying a specific benefit endpoint, the lack of more detailed scientific (e.g., specific concentration-response relationships, values associated with specific injuries) often prevents quantification of the benefits for a specific endpoint. The benefits analysis should push to quantify the potential range and magnitude of the effects whenever possible. Elicitation of expert opinion is one possible approach for identifying the potential magnitude of a specific effect. In such situations, however, the thorough treatment of the variability and uncertainty surrounding any assumptions used in the analysis becomes an integral part of the benefits assessment (see Section 4.1.5). Any benefit categories that remain unquantified should also be included in the benefits assessment when evidence in the scientific literature supports a reasonable connection between mercury exposure and the effect. Some cultural and religious values, for example, cannot typically be adequately addressed using currently available methods for quantifying and monetizing such values (see text box below).

Effects of Mercury Contamination on Cultural Values

In 1991 and 1992, U.S. EPA and the eleven Native American Tribes in the State of Wisconsin conducted a comparative risk project to evaluate environmental risks faced by the Tribes (U.S. EPA 1992b). The Agency attested that the common methods for evaluating economic and social damages by converting them to dollar values were inadequate when dealing with non-market and difficult-to-price activities such as damages to subsistence fishing and associated cultural losses. Instead, a qualitative approach was used to evaluate losses. U.S. EPA (U.S. EPA 1994b) made two important observations relevant to mercury contamination from air emissions:

(1) Nonpoint source contamination was the most important source of social and economic damages; and

(2) Increases in fish consumption advisories cause substantial damage to cultural values. (Note that the majority of the fish consumption advisories in Wisconsin are for mercury).

U.S. EPA also noted that the top three social and economic damages to the tribes were (1) diminishment of cultural and religious values; (2) damage to subsistence activities (e.g., subsistence fishing); and (3) damage to natural resources in commercial use. Fishing plays a role in all three of these areas. With respect to cultural values, for centuries the Wisconsin Native Americans have built traditions around spearing fish and sharing the catch. Growing concerns about limiting fish consumption and limiting the locations where fish may be caught seriously impede the Tribe's traditions.

Monetization of the quantified benefits depends on the availability of the necessary data and an appropriate economic valuation technique. Because many pollution effects (e.g., adverse health or ecological effects) traditionally have not been traded as market commodities, economists and analysts cannot look to readily available information on changes in prices and quantities to estimate the value of these effects. For non-marketed effects, the analysis can use any number of available economic approaches (discussed in Section 4.1.3) for estimating monetary value of reducing adverse effects to specific benefit endpoints. The benefits analysis is often limited, however, in terms of the number of non-marketed effects that can be studied in detail. Thus, the benefits assessment will need to focus on those effects that are expected to be the most significant in terms of magnitude and value. Also, concerns about double-counting or an overlapping of effect categories might lead to a decision to omit a particular benefits category from the aggregation scheme.

The result is that the benefits assessment may have potentially significant gaps in the benefits calculations, due to the omission of many benefits categories from the estimation of monetized benefits, which leads to an underestimation of the monetized benefits presented. The effect of the underestimation is to limit the conclusions that can be reached regarding the monetized benefits and net benefits estimates of mercury reductions.

4.2.3.1 Establishing a Baseline for Measuring Benefits

An integral step in conducting a benefits assessment is selecting the baseline from which the benefits are measured. An appropriate baseline for the assessment of the benefits of additional controls on mercury emissions might be full or current state of compliance with all current regulations. U.S. EPA will need to determine to what extent the baseline for measuring benefits should change over time to account for those regulations which are not currently fully effective, such as the Acid Rain Program, new MACT standards, and the new ozone and particulate matter (PM) NAAQS. Assumptions regarding how society will respond to these regulations can have a significant impact on the magnitude of the estimated costs and benefits of requiring additional mercury controls. For example, SO_x emissions controls expected to be applied for the purpose of implementing the $PM_{2.5}$ standard are expected to result in considerable reductions of mercury emissions (by at least 16 percent). The magnitude of the potential benefits will also depend on the growth rate assumptions for the major source and area categories for mercury as well as the mercury emissions inventory estimates used in establishing the baseline and estimating changes in emissions under alternative control strategies.

The benefits analysis might initially consider the total potential social benefits of reducing mercury exposures without accounting for how those reductions may be achieved (i.e., not measuring from a baseline which nets out reductions from existing regulations as described above). Such an analysis would examine currently observed effects without considering future reductions that may occur as a result of recently promulgated regulations. Under that approach, however, it is not appropriate to consider the total reduction of the effects and the estimated monetized benefits of reducing those effects as the potential benefits of additional controls to reduce mercury emissions; some of those benefits are likely to be realized from full compliance with existing and recently promulgated regulations. Alternatively, a more detailed benefits analysis can be conducted that focuses on the estimated benefits of specific approaches for reducing mercury emissions (e.g., specific regulatory requirements).

4.2.3.2 Use of Modeling to Predict Baseline Effects and Changes in Effects

Predicting baseline mercury emissions and exposures and changes in emissions and exposures under alternative control strategies will require the use of a model that can trace out the effect of a change in mercury emissions on atmospheric methylmercury concentrations (and concentrations of other mercury compounds) and the resulting deposition and loadings to water bodies. Preferably, this model will also estimate the resulting changes in exposure levels (e.g., changes in fish tissue concentrations) over time, and possibly also changes in effects as measured by the specific benefit endpoints. Accomplishing this last step, estimating changes in exposures and effects, most likely will require that an atmospheric model be linked to an available or developed fate and transport and effects model(s). A model developed for estimating the benefits of reducing mercury emissions will need to account for biogenic and geogenic levels of mercury as well as anthropogenic contributions in determining atmospheric and water body concentrations of methylmercury (and other mercury compounds). In addition, the atmospheric, exposure, and effects model(s) should be based on verifiable linkages between emissions, methylmercury concentrations (and concentrations of other mercury compounds), exposures, and effects.

A baseline mercury emissions inventory, as well as alternative inventories under a specific control strategy(ies), will also need to be developed to support the modeling effort. The development and verification of an appropriate atmospheric, exposure, and effects model(s) and emissions inventory is a time and resource intensive activity. As a result, it may require several years to establish a defensible modeling approach for use in a thorough benefits analysis for reducing mercury emissions. It may be possible, however, to use currently available models, with some additional validation for mercury emissions, exposures, and effects, to develop preliminary estimates or bounds for a rough "order of magnitude" benefits assessment. Using current models may be acceptable if the potential error associated with those models can be quantified and represented in the benefits assessment. Furthermore, the uncertainty introduced through the modeling needs to be considered in relation to the potentially large amount of uncertainty associated with the estimated magnitude of the effects on the benefit endpoints and any estimated values associated with those effects.

4.2.4 Selecting an Appropriate Valuation Approach/Technique for Each Benefit Endpoint

The scope of the future benefits analysis is to estimate national-level benefits associated with reducing mercury contamination. The national-level social benefits associated with a change in the environment is the sum of the change in each individual's willingness to pay (WTP) for those goods and services affected by the change across all individuals and all goods and services (see discussion in Section 4.1). As previously discussed, there are a wide variety of economic valuation techniques that can be used to estimate WTP for a change to a specific benefit endpoint (i.e., a specific good or service). The analyst must select the most appropriate technique or approach for estimating WTP for each endpoint given the type of benefit endpoint being valued, the data available regarding the change in the use of or demand for the endpoint, and the time and resource constraints imposed on the benefits assessment.

For some endpoints, a combination of techniques might be applied to estimate the WTP to protect or improve that endpoint. For example, different techniques can often be applied to the same problem with each offering slightly different information and introducing different uncertainties or errors. Thus, when applicable, the use of multiple techniques may provide better information to quantify the range of possible benefits.

The remainder of this section discusses some of the approaches and techniques that can be used to estimate monetary benefits for specific human health and ecological benefit endpoints.

4.2.4.1 Human Health Related Benefits

The economic value of changes in human health effects from mercury exposure should reflect the full costs to the affected individual and society. Health effects can lead to lost wages, medical expenses, and lost productivity, which can be readily measured in dollars, but also to pain and suffering and inconvenience to others, which is more difficult to measure.

There are two primary approaches that are used to measure the economic value of health effects. One is to measure the effects of illness that are directly observed in the marketplace, such as lost wages and health care costs. This can be called the "cost of illness" (COI) approach. An advantage of the COI approach is that economists can rely on observed human behavior. In addition, the data are not usually difficult to collect. This method is commonly accepted by many researchers in the health care industry because it provides estimates for the value of a wide range of health effects.

The other approach is to measure the total value of health effects, by estimating people's willingness to pay (WTP) to avoid them. The WTP measure should include the cost of illness[5], but also includes less tangible values such as pain and suffering. WTP provides a more complete estimate than COI of the economic value of health effects, but it is more difficult to measure. Because of this, several studies have estimated WTP/COI ratios that can be used to estimate the social WTP for human health effects from COI information. One report, which reviewed three of these studies, recommended a WTP/COI ratio of 2.0 for non-cancer adverse health effects associated with exposure to ozone and PM.[6]

As discussed earlier, there are many approaches to measuring WTP. Two techniques that are used for estimating WTP for reducing adverse health effects are contingent valuation and revealed preference. Contingent valuation (CV) approaches use sophisticated survey techniques to elicit people's WTP to avoid health effects. Correct application of CV techniques is required, however, to obtain valid and reliable WTP values. Although CV has been increasingly accepted in recent years, its application remains controversial. Potential biases in willingness to pay estimates derived by CV include: hypothetical bias, strategic bias, starting point bias, vehicle bias, and information bias.

Because environmental mercury exposure is unlikely to have fatal effects, values for reducing the risk of death are not useful for estimating the social cost of mercury pollution. Less information is available on the value of nonfatal effects (or morbidity), though some studies have been conducted on health effects like nonfatal injuries, bronchitis, hospital visits, and respiratory symptom days. Values for these effects can range from a few dollars for a day of respiratory symptoms to several million dollars for severe, disabling injuries.[7]

Thus, in terms of developing estimates of the value of reducing the incidence and severity of the adverse health effects associated with mercury exposure (e.g., kidney damage, neurological damage, and developmental effects), future benefits analyses might involve:

- Conducting original analyses, such as contingent valuation surveys, to estimate the value of reducing the risk of the major health effects associated with mercury;
- Identifying existing studies of the WTP to avoid these types of health effects and conducting a benefits transfer analysis to estimate the benefits associated with reducing the incidence or severity of such effects by reducing mercury contamination (studies of the benefits of reducing health effects associated with exposure to high levels of lead, for example, might provide good estimates of the value of reducing similar adverse health effects);

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[5] To the extent that an individual pays it: if some of the costs are borne by society but not by the individual (e.g., they are paid by insurance) then the total cost to society may exceed the individual's WTP.

[6] Hagler Bailly Consulting, Inc., November 1994. *The New York State Externalities Cost Study,* p. X-30.

[7] See for example: Loehmann et al. (1979) for respiratory symptoms, and Miller (1989) for other health effects, including injuries.

- Gathering detailed COI information on the health effects associated with mercury exposure and other data necessary to determine the extent to which those effects can be attributed to elevated mercury concentrations;
- Adapting existing models, which currently estimate and value human health effects on a local or regional level, to estimate national level reductions in associated human health effects and the associated benefits. For example, a model developed to estimate the value of human health effects from mercury emissions from utility boilers in New York could potentially be generalized and linked to other existing emissions, transport, and deposition models.[8]

Table 4-5 presents some possible approaches, using the techniques described above, for estimating the benefits of reducing mercury contamination levels.

Example Health Risk Valuation Study

One study of the benefits of the Great Lake Water Quality Initiative did not specifically address mercury, but did discuss potential values of reducing toxic contamination in general (NWF 1993). One study examined the effects on the intelligence and development of children, which could be related to mercury. The study suggested that higher IQS and fewer developmental problems would lead to greater happiness for children and their families, as well as savings in health and educational expenditures. One methodology for measuring the benefits of reduced effects on children is suggested: the "human capital" approach. The underlying premise of this approach is that, if eliminating toxics eliminates developmental and learning deficits, then it will lead to increased productivity in later life and associated benefits to the individual and to society. The study cites a 1991 U.S. EPA analysis of the effects of lead poisoning that showed a lifetime loss in wages of 1.76 percent for each IQ point lost.

4.2.4.2 Welfare or Ecological Benefits

The economic value of changes to a specific welfare or ecological benefit endpoint should include the full social value (i.e., both private and public benefits) of improvements or reduced potential for adverse effects resulting from the reduction in mercury contamination levels. As discussed previously, estimating individual willingness-to-pay (WTP) for a change is the approach used by economists to value the potential changes to welfare or ecological benefit endpoints. As previously noted, there are many different techniques and approaches used to estimate WTP, which vary by their level of accuracy, ease in estimation, and data needs.

In previous analyses, U.S. EPA has used several approaches to measure the economic value of welfare and ecological effects that may be applicable to future mercury benefits assessments. These approaches include:

- Benefits transfer analyses to develop national benefits estimates using estimated WTP from previous studies that looked at similar effects for a specific location;

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[8] The model referenced here was developed by Hagler Bailly to estimate the human health externality costs associated with mercury emissions from electric utility boilers in New York. See Hagler Bailley, November 1994. New York State Environmental Externalities Cost Study, Report 2: Methodology., EP 91-50, Final Report, November 1994.

Table 4-5
Examples of Potential Approaches for Future Benefits Analyses

Benefit Endpoint	Technique Used	Possible Approach
Human Health - Child Development and IQ	Benefits Transfer	Use benefit analyses (i.e., benefit functions or WTP estimates) conducted in support of lead regulations that analyzed the value of reducing the adverse effects of elevated lead exposure on child development and IQ and neurological effects. To the extent that lead and mercury exposure result in similar adverse health effects in similar at-risk populations, the benefit estimates for reducing lead exposures may be applicable in a benefits transfer analysis estimating the benefits of reducing mercury exposures. At a minimum, information would be needed on the at-risk populations and incidence rates or risks for both lead and mercury exposure, and WTP estimates to reduce incidence and/or the risk of incidence of those adverse effects from lead exposure that are comparable to mercury exposure.
Variable - May Encompass Various Ecological, Welfare, and Human Health Benefit Endpoints	Revealed preference, for assigning site-specific benefits, and benefits transfer, to develop a national-level estimate of benefits	Future benefits analyses could possibly use the cost of remedial actions (if the actions are in fact taken) to reduce mercury exposures as a minimum measure of the benefits expected to accrue to society as a result of reducing mercury contamination. The assumption in using this technique is that people will only choose to undertake an expensive remedial action if the expected benefits of doing so, in terms of reduced human health and ecological effects, are greater than the expected costs of the action. In this way, society “reveals” its preferences and its willingness to pay to reduce adverse effects. One possibility for applying this approach to a mercury benefits analysis is to use Superfund cleanup cost information for sites with significant mercury contamination levels. Application of this approach requires detailed information on the potential reduction in adverse effects with the remedial action and possibly other data to use the site specific benefits to develop a national-level benefits estimate.

Table 4-5 (continued)
Examples of Potential Approaches for Future Benefits Analyses

Benefit Endpoint	Technique Used	Possible Approach
Lost Value of Recreational Fishing Activity due to Inability to Consume Catch under a Fish Advisory	Original contingent valuation analysis	Future benefits analysis might involve an original contingent valuation study to estimate the reduction in the value of a fishing trip experienced by recreational fishermen due to the fact that they cannot consume (or have to reduce their consumption) the fish they catch in areas affected by mercury fish advisories.
Recreational Fishing	Benefits Transfer	Future benefits analysis could develop national benefits estimates based on the estimated change in the value of a fishing day associated with reducing contaminant levels from other analyses. Previous U.S. EPA analyses have used results from an analysis by Lyke (1992), which estimated that a "contaminant free" fishery in Wisconsin would result in an increase in the value to recreational anglers of 11 to 31 percent. Such an analysis would require an assumption regarding what percentage of this increase in value should be associated with reducing mercury fish advisories, information on the total value of U.S. fisheries under current (or baseline) conditions, and additional data to develop a national estimate, such as the percentage of fish advisories in each state due to mercury contamination, percentage of number or acreage of fishing areas affected by mercury advisories, and number of registered fisherman per state.

- Original analyses using models to predict changes in yield and value of timber and agricultural products;

- Revealed preference approaches looking at current expenditures to reduce pollution effects as an estimate of the minimum value of reducing pollution effects; and

- Estimated cost savings associated with avoiding damages (this approach is similar to the cost of illness approach for estimating WTP for health effects).

Table 4-5 presents some possible approaches, using the techniques described above, for estimating the benefits of reducing mercury contamination levels. For many of the ecological/welfare benefit endpoints identified in Table 4-4, the potential for significant benefits associated with reducing mercury contamination levels is clear. With respect to fishing-related activities, in particular, the potential for benefits (i.e., increased value of fishing activities and values) has been demonstrated for specific states. Currently, however, the national magnitude of the economic and social benefits associated with reducing mercury contamination in fisheries, as well as other benefit endpoints, has not been estimated. In fact, very little information exists on the potential fisheries and other ecological/welfare related benefits that could be specifically attributed to reducing mercury contamination.

Example Welfare or Ecological Effects Valuation Studies

<u>Arkansas Recreational Fisheries</u>. The Arkansas Game and Fish Commission is the only agency contacted (of six different states and three federal agencies contacted for this report) that had attempted to quantify fishing-related dollar losses due to mercury contamination as of 1994. The Commission estimated a loss of fishing expenditures due to mercury fish consumption advisories of over $5 million dollars from 1991 to 1992. This loss was estimated based on decreases in fishing license purchases in counties where mercury advisories were issued, multiplied by the average number of trips an angler takes per year, and by the average per-trip expenditures. The Commission has not published its findings (Armstrong 1994). Changes in expenditures, however, represent changes in welfare.

<u>New Jersey Quality of Life</u>. The New Jersey Department of Environmental Protection and Energy concluded in its 1994 report *Task Force on Mercury Emissions Standard Setting: Final Report on Municipal Solid Waste Incineration* that reducing municipal solid waste mercury emissions will improve the "quality of life" in New Jersey by decreasing the accumulation of mercury in aquatic systems. The Department, however, did not attempt to quantify these benefits. Changes in expenditures do not, however, represent changes in welfare.

4.2.4.3 Conclusions

As in previous benefits analyses, the benefits assessment for mercury is expected to need to focus on a few of the benefit endpoints listed in Tables 4-3 and 4-4, based on the expected magnitude and value of the potential benefits associated with each endpoint. It is expected that future analyses to assess the benefits of reducing mercury, in terms of reduced effects on human health and welfare/ecological benefit endpoints, would rely primarily on benefits transfer analysis. As feasible, the benefits transfer analyses may be supplemented with alternative analyses, possibly involving original analysis, estimated cost savings, and/or revealed preference approaches, for certain benefit endpoints.

In considering the ideas described here, particularly those examples discussed in Table 4-5, it is important to note that there are certainly many other possible analyses that can be conducted as part of future benefits assessments for reducing mercury. Thus, an early step in a future comprehensive benefits assessment should be the identification and evaluation of all possible approaches for estimating the value of improvements or reduced risks to the specific benefit endpoints on which the analysis chooses to focus.

5. 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, international activities and relevant activities in selected states (principally states in the Great Lakes region). The chapter starts with an overview of ongoing federal, international and state activities in Section 5.1. Management alternatives and statutory authorities for controlling mercury emissions under the Clean Air Act are then presented in Section 5.2.

5.1 Ongoing Federal, International 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.

5.1.1 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 5-1 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 5-1 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 regulations designed to limit mercury exposures in the workplace and during transportation.

Table 5-1
Federal Mercury Controls[a]

SPECIFIC SOURCES/FOCUS	CONTROL/ACTION
ENVIRONMENTAL MEDIA	
Drinking Water	• Maximum contaminant level (MCL) = 0.002 mg/L (40 CFR 141.62, 21 CFR 103.35).
Surface Water	• 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	• No ambient air standard.
Soil	• No soil standard.
ENVIRONMENTAL SOURCES	
Air Point Sources	• 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 promulgated on August 15, 1997. • Airborne emissions of mercury on 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	• 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	• 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 5-1
Federal Mercury Controls (continued)

SPECIFIC SOURCES/FOCUS	CONTROL/ACTION
Mercury-Containing Wastes	• 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	• 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	• 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	• 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	• All uses of mercury in paints have been discontinued.
Dental Uses	• 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.
Lighting	• 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).

Table 5-1
Federal Mercury Controls (continued)

SPECIFIC SOURCES/FOCUS	CONTROL/ACTION
Pesticides	- No current production of mercury-containing pesticides; all former registrations have been cancelled or requests for voluntary cancellation have been received.
Special Paper Coatings	- 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	- 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	- 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	- OSHA Standards: ceiling limit of 0.1 mg/m^3 for inorganic and elemental mercury, 0.01 mg/m^3 as an 8-hr time weighted average for alkylmercury compounds, and a ceiling limit of 0.04 mg/m^3 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	- 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	- 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.

- Other government programs to address mercury problems in an integrated manner (i.e., the Virtual Elimination Project and pollution prevention initiatives).

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 con-sists of detailed methodologies to develop criteria for additional pollutants; implementation procedures to develop more consistent, enforceable water quality-based effluent limits in dis-charge 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 5-1, 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 and medical waste incinerators under §§ 111 and 129 of the 1990 Clean Air Act Amendments.

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.[9] 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

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[9] 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).

mercury in food, drugs, cosmetics and dental amalgam under FFDCA. Particular controls on these 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 coil cells, does not exceed 0.025 percent by weight (National Electrical Manufacturers Association, 1995).

Other Regulations. 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 5-1. 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/m^3 for elemental mercury vapor and 10 mg/m^3 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.

Virtual Elimination Project. U.S. EPA and Environment Canada are actively developing strategies to achieve the Great Lakes Water Quality Agreement's (GLWQA'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 began the Virtual Elimination Pilot Project in 1993. The project was designed to answer the following question: "What options exist for improving the current regulatory and non-regulatory framework to encourage continuing reductions towards zero in the use, generation, and release of selected toxic substances?" The aim of the project was to identify barriers to achieving virtual elimination and to develop strategies to overcome these barriers.

The pilot 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.

Other Pollution Prevention Programs. U.S. EPA is working with state and local governments to develop a national network of prevention programs that will assist regulators at all levels of government in promoting pollution prevention. To that end, U.S. EPA is providing funding support, technical assistance, information dissemination and forming federal/state/local government partnerships to focus efforts on pollution prevention as the national goal for environmental management.

State and federal partnerships have already led to actions that will reduce mercury loadings to the environment. For example, the National Wildlife Federation, funded in part by U.S. EPA, has recently released a report detailing how hospitals in Detroit, Michigan; Grand Rapids, Michigan; Boston, Massachusetts; and Duluth, Minnesota have successfully reduced mercury releases by applying pollution prevention principles. This Report contains practical and cost effective suggestions for improving the environmental performance of hospitals and to help them meet increasingly stringent limits in regulatory permits. Industry groups have also made advances in pollution prevention (see text box below on Chrysler Corporation).

U.S. EPA is working continually to incorporate pollution prevention into the mainstream of its work. Over the last six years, the agency has undertaken a concerted effort to find the best ways to incorporate prevention into regulations and permitting. For example, in 1992, U.S. EPA began an effort to evaluate pollution prevention options for a number of new regulations under development. This effort, called the Source Reduction Review Project, required U.S. EPA’s media offices to identify multi-media approaches to addressing air, water and solid waste regulations. Another program, U.S. EPA’s Common Sense Initiative (CSI), created a pollution prevention framework for environmental protection on an industry-by-industry basis by focusing on opportunities to change complicated or inconsistent environmental requirements into comprehensive strategies. The goal of each of these programs is a cleaner environment at less costs to taxpayers.

In addition, and on a broader scale, U.S. EPA is currently developing a long-term plan to mitigate the risks associated with mercury and other chemicals of concern under its Persistent, Bioaccumulative and Toxic (PBT) Chemicals strategy by using pollution prevention principles. Through its current efforts on the PBT strategy, U.S. EPA will focus these activities more intensively on the key persistent, bioaccumulative pollutants, especially mercury. U.S. EPA expects that through partnership with states and local organizations, and in collaboration with industry, there will be more opportunities to use pollution prevention as a means to mitigate the potential risk to human health and the environment associated with exposure to mercury.

Life Cycle Cost Management in the Auto Industry

Industry remains at the center of pollution prevention activities. Studies have shown that the economic benefits can be compelling arguments in favor of pollution prevention, but only when managers are able to see the cost savings that pollution prevention would bring. Environmental accounting is the key factor in demonstrating to businesses the value of prevention. The following serves as a specific example of pollution prevention in practice to reduce mercury loadings.

The Chrysler Corporation is now removing or replacing all mercury switches that have been traditionally used in its underhood convenience light applications. Chrysler has done so as a result of the application of life cycle cost management methodologies that are advocated by U.S. EPA Pollution Prevention Division's Environmental Accounting Project. This Project is a cooperative effort with business, academia and others to promote sound management accounting and capital budgeting practices which better address environmental costs. The project encourages and motivates business to understand the full spectrum of environmental costs and integrate these costs into decision making. Chrysler is partnering with the Project to share its environmental accounting experience and case studies with the 800-plus members of the Project-facilitated Environmental Accounting Network.

By applying the principles of environmental accounting, the Chrysler Corporation determined that it could cost-effectively replace the mercury switches with a rolling ball switch or remove the switches altogether. For the first group of cars on which Chrysler tested the feasibility of substitution and removal, it determined that it could avoid $40,000 in costs. Most of those costs were associated with the documentation of the removal of mercury switches from the vehicle before disposal, and with the potential liability for any mercury that enters the environment following vehicle disposal. After conducting their own total cost analyses, other auto manufacturers are now following suit and are actively removing mercury switches from their own automobiles.

5.1.2 International Activities

On an international level, mercury is being addressed as part of the Great Lakes Binational Toxics Strategy, and the North American Regional Action Plan, among other efforts. These two initiatives are summarized below, followed by a description of a proposed international collaborative effort to examine the fate of mercury in the Lake Superior Basin.

Binational Strategy. The Great Lakes Binational Toxics Strategy, which was signed between Canada and the United States on April 7, 1997 (U.S. EPA and Environment Canada, 1997), was developed to help achieve the objectives of the 1987 GLWQA. Although both Canada and the United States have domestic virtual elimination strategies as described above, a coordinated strategy is necessary for the greatest reduction in toxic substances throughout the Great Lakes Basin.

The Binational Strategy provides the framework to achieve quantifiable goals in a specified time frame (1997 to 2006) for targeted persistent toxic substances, especially those which bioaccumulate. The Strategy recommends that goals be accomplished through a four-step process:

- Gather information on generation, uses, and sources of the pollutant within and outside the Great Lakes Basin;
- Analyze current regulatory and non-regulatory programs and initiatives that manage or control the pollutants and identify the gaps in these regulations that offer opportunities for reductions;
- Develop cost-effective options and provide recommendations for increasing the pace and level of reductions; and
- Recommend and implement actions to achieve goals.

Mercury and mercury compounds are considered immediate priorities and are targeted for reduction and eventual virtual elimination through pollution prevention and other incentive-based actions.

Both the United States and Canada have set “challenge” goals to achieve reductions through implementation of voluntary efforts and regulatory actions. One of these challenges is the commitment of these countries to work together to assess atmospheric inputs of persistent toxic substances to the Great Lakes, with the goal of evaluating and reporting jointly on the contribution and significance of long-range transport of these substances from worldwide sources. Efforts will be made to work within the existing international framework to reduce releases of such pollutants from remaining long-range sources.

<u>North American Regional Action Plan</u>. The North American Regional Action Plan (NARAP) is one of a number of regional undertakings that stem from the North American agreement on Environmental Cooperation between the governments of Canada, the united Mexican states and the United States of America (Parties). The NARAP calls for the development of regional action plans for selected persistent and toxic substances as a first priority in the Parties’ common desire to address national and regional concerns associated with the sound management of chemicals.

The action plans are designed to reflect a long-term commitment to regional action. The sharing and transfer of information and best practices are seen as an important means of enhancing national capacity for the sound management of chemicals. Other important elements and outcomes of these cooperative initiatives include collaboration and cooperation in the measurement, monitoring, modeling, research and assessment of selected persistent and toxic substances in environmental media. Such cooperation will improve the quality, availability and relevance of the “environmental information” needed to make informed and responsible decisions throughout the implementation of the action plans.

Mercury is one of the targeted chemicals and has its own action plan designed to unite the Parties in their joint and differentiated efforts to reduce the exposure of North American ecosystems, fish and wildlife, and especially humans, to mercury through the prevention and reduction of anthropogenic releases of mercury to the North American environment. The objectives of the action plan are to reduce mercury levels in and fluxes among environmental media in order to prevent or minimize exposure to ecosystems, fish and wildlife, and humans.

Implementation on the mercury action plan is predicated on the following objectives:

- Building on existing initiatives. Examples include the Great Lakes Binational Toxics Strategy, described above.
- Promoting North American regional and global activities. The mercury action plan will promote regional actions to reduce mercury emissions and serve as an example for initiatives under development throughout the region and globally.
- Best practices. The action plan will promote the sharing, transfer, and general adoption of policies, programs, technologies, and other measures that have proven to be cost-effective and environmentally appropriate.
- Challenging stakeholders to take cooperative action on mercury. The action plan promotes stakeholder partnerships in information and technology exchanges.
- Improving scientific understanding. The action plan will use government and private-sector partnerships to fund research and monitoring, and to advance the science and technology state-of-knowledge for mercury.
- Capacity building in Mexico. The Parties are committed to working cooperatively to build Mexico's capacity with respect to the prevention and reduction of anthropogenic releases of mercury and the sound management of mercury.
- Extended Americas. The Parties agree to actively promote cooperation with other countries to promote pertinent initiatives.

Specific actions outlined for mercury include a series of workgroups and workshops to assess the current knowledge on mercury issues and compile information into shared databases. An implementation committee will provide oversight of the action plan.

The Fate of Mercury in the Lake Superior Basin Proposal. The U.S. EPA, the U.S. Department of Energy, the Electric Power Research Institute, and Ontario Hydro are proposing a project to develop models to accurately predict the relationship between source mercury emissions (four utilities firing coal -- two in the United States and two in Canada) and bioaccumulation of mercury in fish in the Great Lakes Region. A team of leading experts in mercury measurement, transport, deposition, methylation rates, and bioaccumulation been assembled to plan and conduct the research. The team includes the University of North Dakota Energy and Environmental Research Center (UNDEERC), University of Michigan Air Quality Laboratory, Michigan Department of Environmental Quality, Academy of Natural Sciences, Wisconsin Department of Natural Resources, Minnesota Pollution Control Agency, Roy F. Weston, Inc., CONSOL INC, Tetra Tech, Inc., and Frontier Geosciences.

5.1.3 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, has drafted 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 5-2 summarizes examples of mercury control measures adopted by various states for different environmental sources.

Table 5-2
Examples of State Mercury Controls[a]

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
ENVIRONMENTAL SOURCES		
Air Point Sources	Florida	• Mercury emission standard for municipal solid waste incinerators (65 micrograms/m^3).
	Minnesota	• Proposed waste combustion rules include emission limits. • New incinerator permits with mercury limits will require air monitoring systems and periodic stack testing.
	New Jersey	• Mercury emission standard for municipal solid waste incinerators (65 micrograms/$m^{3)}$, with further reductions to be phased in.
	Ohio	• Considering installing mercury-emission control equipment.
	Wisconsin	• 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	• Businesses must report use and discharge information for mercury under the Water Pollution Control Act.
Hg-Containing Wastes	Minnesota	• Management standards for facilities recycling mercury-containing hazardous wastes (currently being drafted). • Mercury must be removed from products before disposal.
PRODUCTS		
White Goods	Illinois	• Mercury components must be removed from discarded white goods before disposal.
	Minnesota	• 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	• 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 5-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Batteries (continued)	California	• 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	• 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	• 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	• Task force to study storage, transport, disposal and recycling.
	Iowa	• 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	• 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 5-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Batteries (continued)	Maryland	• 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	• 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	• 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	• 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.
Batteries (continued)	New Hampshire	• 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.

Table 5-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
	New Jersey	• 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	• 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	• Required manufacturer to reduce the level of mercury to 0.025% by weight in alkaline batteries effective 1-1-92.
	Rhode Island	• 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.
Batteries (continued)	Vermont	• 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	• 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.

Table 5-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
Electrical Components	Pennsylvania	• Mercury use in mining equipment (i.e., electrical machinery) is limited.
Paints and Pigments	Minnesota	• No mercury can be deliberately introduced into products intended for use in Minnesota, except for art supplies.
Dental Uses	Minnesota	• 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	• No sale of toys containing mercury (pending).
	Minnesota Wisconsin	• Toys with mercury are banned and fines are imposed on retail sales of toys containing mercury.
Lighting	Florida	• 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	• Lamps are viewed as by-products that can be recycled, and exempt from RCRA.
Lighting (continued)	Minnesota	• 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	• Mercury vapor lights must be self-extinguishing or have protective shield.
Poisons	Ohio	• The sale of mercury and mercury compounds is restricted.
	Pennsylvania	• Levels established for mercury products considered to be safe.
Packaging	Florida	• The mercury content in packaging and packaging components is restricted.
	Illinois Minnesota New York Wisconsin	• Restrictions on mercury content in packaging are being phased in. (Dates and concentrations vary by state.)

Table 5-2
Examples of State Mercury Controls (continued)

MERCURY SOURCES/TARGETS	STATES WITH CONTROLS	CONTROL/ACTION
	Pennsylvania	• Intentionally introduced mercury in packaging is prohibited (pending).
Pharmaceuticals	Illinois Indiana New York Ohio Pennsylvania	• Manufacturers must list the quantity of mercury in products.
Explosives/Fireworks	Minnesota Wisconsin	• Explosives containing mercury are Class A maximum hazards. • Permits required for fireworks with mercury (Minnesota only).

[a] Adapted from Ross & Associates, 1994.

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 5-2 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).

5.2 Management Alternatives and Statutory Authorities

5.2.1 Management Alternatives

Control of mercury emissions may require a mix of strategies including pollution prevention, materials separation and conventional and non-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. Other non-conventional approaches such as emissions trading or application of a use tax, or other market-based approaches may also prove feasible for mercury control.

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 5-3 presents the four 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 reductions do not illustrate two important considerations. One is that, as presented, all of the cost of control could mistakenly be attributed to mercury removal. As described previously in this Report, many of these controls achieve reductions of other pollutants as well (e.g., acid gases, dioxin, other metals). In some cases (e.g., the emission guidelines for MWI and the ozone and PM NAAQS), the choice of control technology or control strategy is aimed at reducing pollutants other than mercury. In these cases, there is a co-control benefit of mercury reduction. The benefits of reducing other pollutants should be considered when interpreting the mercury control costs. Second, the technologies available for mercury control represent relatively new applications of these technologies. Thus, in the future, it is likely that as new or emerging technologies develop, the cost-effectiveness of control will improve. Air pollution control and prevention techniques are continuously under development and improvement. There is a fairly rapid pace of innovation in the air pollution control sector. The demand for cleaner products and cleaner processes that lower overall costs, combined with the necessity for improved air and water quality, create strong incentives for technological innovation and a growing market for such innovations. As the demand for more innovative, cost-effective and cost-saving technologies increase, new technologies will move from the research and development or pilot program phase to commercial availability.

While existing technology will play a key role in reducing mercury from some sources, emerging technology may be more appropriate for others. Innovations in environmental policies may also play a key role in developing a national management strategy for mercury. These innovations could include multi-media approaches, greater emphasis on pollution prevention, regional control strategies and optimization of co-control opportunities.

5.2.2 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.

Table 5-3
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[a]	Potential National Annual Costs[b]	Cost-Effectiveness ($/lb of mercury removed)[c]
Municipal waste combustors	129	18.6	Material separation Product substitution Activated carbon injection Carbon filter beds Polishing wet scrubber	27 tons	$11.4-47 million	$211-870
Medical waste incinerators	~2,400	10.1	Material separation Wet scrubber or dry scrubber with carbon Activated carbon injection	15 tons (95% reduction)	$60-120 million[d]	$2,000-$4,000[d]
Coal-fired utility boilers	426 (1,043 boilers)	32.5	Fuel switching Advanced coal cleaning Activated carbon injection Carbon filter beds	37 tons (90% reduction) [e]	$5 billion	$67,700-70,000
			Co-control: ozone and PM NAAQS	11 tons[f]	No incremental mercury control costs	No incremental mercury control costs
Chlor-alkali plants using the mercury cell process	14	4.5	Process modification Depleted brine scrubbing Treated activated carbon adsorption	7.1 tons (100% reduction)	$65 million	$4,590
Total	~3,586	65.7			~$5.2 billion	

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

[a] Estimated reductions assuming every facility could achieve the reduction listed.

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

[c] Where cost-effectiveness values are presented as a range, the values reflect the range across facilities of different sizes.

[d] Cost of control should not be attributed to mercury control alone. Wet scrubbers efficiently remove nine other pollutants from the MWI flue gas as required by the emission guidelines for MWIs.

[e] The potential national reductions reflects sufficient amounts of activated carbon to control mercury emissions from coal-fired utility boilers by 90 percent. Activated carbon injection has not been demonstrated for a full-scale utility boiler application. Control costs are upper bound based on high temperature activated carbon injection. The 37 tons reduction is 90 percent of 41 tons, accounting for 11 ton reduction from the ozone and PM NAAQS.

[f] Assumes some fuel switching and additional installation of wet scrubbers which are assumed to remove 30 percent.

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 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) 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 U.S. EPA and between U.S. EPA and other federal agencies to reduce and preferably prevent pollution in the air, water, and soil; and (3) 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, U.S. EPA must develop and describe alternative control strategies for HAPs that may require regulation under section 112, and, if appropriate and necessary, 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 on October 31, 1995 and final rules for medical waste incinerators on August 15, 1997. Emission limits for hazardous waste combustors will be forthcoming under the Agency’s Combustion Strategy.

6. CONCLUSIONS

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

- 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.

- 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.

- 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.

- Limited pilot-scale studies with the injection of activated carbon indicate variable control of mercury from 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 capture in utility flue gas streams include flue gas volume, flue gas temperature, flue gas vapor and particulate phase constituents (e.g., chlorine as HCl, nitrogen oxides, sulfur oxides, metal oxides on the surfaces of particulate matter, fly ash composition, percent carbon in fly ash, etc.), the mercury concentration and chemical species being formed, and the existing APCDs being augmented (e.g., fabric filters versus ESPs) for mercury capture by activated carbon. Mercury capture is mass transfer limited in utility flue gas streams due to the low mercury concentrations in the extremely high volumes of flue gas.

- The addition of activated carbon to utility flue gas for mercury control would increase the amount of particulate matter requiring disposal. Studies are just being conducted to assess the stability/leachability for possible re-emission of mercury in the near- and long-term from the carbons and sorbents captured in the gas phase and disposed of in the solid phase in landfills.

- The chemical species of mercury formed during the combustion process and post-combustion conditions vary significantly from one plant to another. While combustion conditions vary, the subsequent fly ash and vapor phase constituents can play a major role in the percentage of the chemical species of mercury formed. Understanding the rate controlling mechanisms (e.g., transport, equilibrium, and kinetics) will aid in predicting and possibly controlling the species formed in order to optimize existing APCDs for mercury removal. Removal effectiveness is dependent on the species of mercury present. There are promising technologies being investigated at the bench- and pilot-scales for the removal of mercury, but none have been tested at the full scale.

- Given the relative low maturity level of these technologies being tested, commercial deployment is still several years away. Deployment is strongly dependent on understanding the fundamental mechanisms of the flue gas and mercury chemistries, and the results of well designed bench- and larger pilot-scale studies. In addition, no single control technology has been tested at the pilot-scale or larger that removes all forms of mercury.

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 and its form(s) in the flue gas stream, the natural mercury sorption and conversion properties of fly ash, 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. In addition, the impacts of the natural mercury sorption and conversion properties of fly ash in conjunction with vapor phase portion of the flue gas were not considered.

7. RESEARCH NEEDS

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

1. Data from full-scale testing of activated carbon injection with and without flue gas cooling at a variety of coal-fired utility boiler systems representative of the utility industry.

2. Additional data on the efficiency of various sorbents including fly ash-based sorbents, activated carbon, impregnated carbons, noble metal sorption, and other types of sorbents, in reducing the different chemical species of mercury present in flue gas.

3. Additional data on improving the mercury mass transfer limitation(s) and reactivity of activated carbon while decreasing the mass carbon-to-mercury ratio.

4. Information on the cost-effectiveness and commercialization costs of other technologies for mercury control that are currently in the research stage. These include impregnated activated carbon, sodium sulfide injection, activated carbon fluidized bed, noble metal sorption, sorbent injection alone and with humidification, and other types of sorbents.

5. Additional data/information on the impacts of flue gas cooling, through humidification on acid mist formation and the means to control the sulfur trioxide before the acid mist is formed. The range of the cost of control of sulfur trioxide under carbon injection scenarios with humidification needs to be determined.

6. Additional data on the ability and cost of conventional or advanced coal cleaning techniques to remove mercury from raw coal. The potential for mercury emissions from coal-cleaning slurries needs to be characterized. In addition, the added costs for advanced coal cleaning in combination with post-combustion controls for mercury have not been fully developed.

7. Additional data on the fundamental mechanisms responsible for conversion of mercury to other chemical species and the natural adsorption of mercury with the native fly ash as a result of combustion of certain coals or post-combustion conditions.

8. Additional information on improving the capture of elemental mercury and the oxidized form of mercury in wet FGD systems.

9. Improvement and development of modeling capabilities for predicting mercury speciation, conversion, and adsorption with fly ashes and sorbents in conjunction with laboratory and pilot-scale test results.

10. Improvement and development of predictive models for the emission of elemental and non-elemental forms of mercury.

11. Data on the impacts on waste and by-products as a result of mercury capture in fly ash, sorbents mixed with fly ash, FGD waste, and other aqueous and solid effluent streams associated with utilities. Determine the stability (identify the form of mercury in waste) and leaching properties, and potential near- and long-term re-emission of mercury from the various wastes.

12. Improvements on development of real time mercury measurement techniques (CEMs) to measure and speciate at the 1 ppb level in order to possibly reduce the cost of control of mercury by optimization through advanced process control.

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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 (200°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 (200°F) and 4361 wt C/wt inlet Hg; inlet Hg concentration of 6.23 μg/dscm)	99

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

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 four 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 µg/dscm (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 µg/dscm (260 gr/million dscf) for the small model plant and 490 µg/dscm (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/kg[1] ($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

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[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
Plant Size (Mg/day)	180	2,045
Number of Units	2	3
Unit Size (Mg/day)	90	680
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

	180-Mg/day MWC		2,045-Mg/day MWC	
Operating Assumptions	$/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 removed. Costs associated with a materials separation program are not included in the sensitivity analysis.

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

Parameter	Small Model Plant	Large Model Plant
Plant Size (Mg/day)	180	2,045
Number of Units	2	3
Unit Size (Mg/day)	90	682
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.

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 2,0450 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
Plant Size (Mg/day)	180	2,045
Number of Units	2	3
Unit Size (Mg/day)	90	682
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

	180-Mg/day MWC		2,045-Mg/day MWC	
Operating Assumptions	$/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 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.2.2 Wet Scrubbing

For detailed cost estimates of the technolgies needed to meet the MWI emission guidelines for mercury and other pollutants, the reader is encouraged to consult *Hospital/Medical/Infectious Waste Incinerators: Background Information for Promulgated Standards and Guidelines - Regulatory Impact Analysis for New and Existing Facilities (EPA-453/R-97-009b).* National cost estimates and a discussion of the MWI emission guidelines are presented in Chapter 3 of this volume.

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 $HgCl_2$. (It should be noted that the assumption of mercury in the form of 100 percent $HgCl_2$ 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 $HgCl_2$ [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-7 summarizes the model utility boilers and mercury controls used in the cost analysis.

Table B-7
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 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 (200°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.

Figure B-1
Spray Cooling System

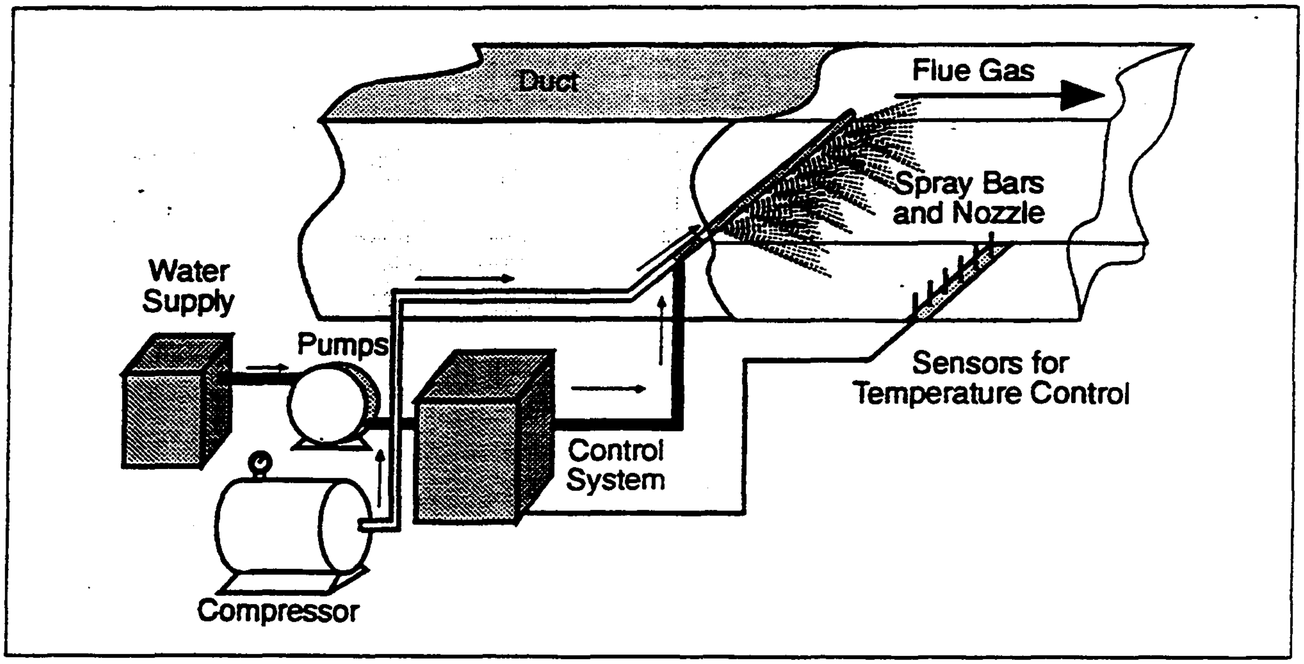
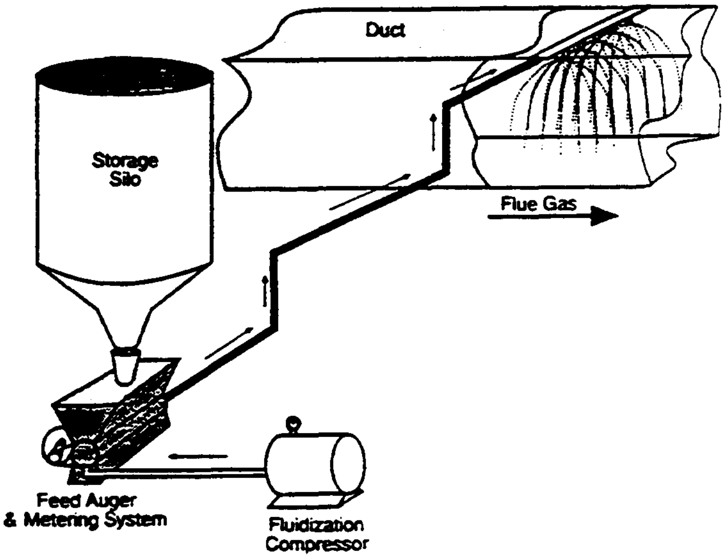


Figure B-2
Carbon Injection System



B.3.2 Activated Carbon Injection

Activated carbon injection rates for model utility boilers used in costing carbon injection are shown in Table B-8. 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-8
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 indicted in the table, only three flue-gas temperatures were used for the various models. For those with waste spray cooling, 93°C (200°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. For some conditions, rates to 100,000 g carbon/g Hg may be required.

Tables B-9 through B-12 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 SO_2 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

Table B-9
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-10
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)				
Capacity Factor (%)	1			
	65			
CAPITAL COST ($)	Cooling	Injection	Fabric Filter	Total
Purchased Equipment (PE)	2,993,796	109,448	12,978,750	16,081,994
Installation	1,017,891	16,417	9,344,700	10,379,008
Indirect	1,347,208	32,835	5,840,438	7,220,480
Total Capital Cost (TCC)	5,358,894	158,700	28,163,888	33,700,000
OPERATING AND MAINTENANCE (O&M) ($/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]	---	---	28,573	28,573
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	5,667,316[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-11
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)				
Capacity Factor (%)	1			
	65			
CAPITAL COST ($)	Cooling	Injection	Fabric Filter	Total
Purchased Equipment (PE)	258,627	109,448	1,813,479	2,181,554
Installation	87,933	16,417	1,305,705	1,410,055
Indirect	116,382	32,835	816,066	965,282
Total Capital Cost (TCC)	462,941	158,700	3,935,249	4,560,000
OPERATING AND MAINTENANCE (O&M) ($/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]	---	---	2,902	2,902
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	931,396[m]	1,290,000
mills/kWh	0.40	0.22	1.64	2.26
Mercury Reduction (kg/yr)	--	--	--	21
Mercury Reduction (lb/yr)	--	--	--	46
$/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-12
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 ($)	Cooling	Injection	Total
Purchased Equipment (PE)	2,993,796	109,448	3,103,244
Installation	1,017,891	16,417	1,034,308
Indirect	1,347,208	32,835	1,380,043
Total Capital Cost (TCC)	5,358,894	158,700	5,520,000
OPERATING AND MAINTENANCE (O&M) ($/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.

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-9 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 7.5 and increases the annualized cost by a factor of 15.3; the cost effectiveness value decreases by 57 percent and the annualized mills/kWh cost impact also decreases by 57 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 $660,000/yr for the 100-MW boiler.

Tables B-10 and B-11 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 (200°F). The FF is located after the injection system and before the stack. The FF is required for capture of the injected carbon. As a result of the cooling, the carbon injection rate is reduced to approximately 1 to 3 percent of the rates assumed in Table B-9. 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 4 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 a factor of 7.4 and increases the annualized cost by a factor of 6.2; the cost effectiveness value decreases by 37 percent and the annualized mills/per kilowatt-hour cost impact also decreases by 37 percent.

Table B-10 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-11 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 $1,290,000/yr and the cost effectiveness is $61,000/kg ($27,700/lb) of mercury.

Table B-12 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 97 percent of the cost contributed by the spray cooler and 3 percent contributed by the activated carbon injection system. The annualized cost is $2,260,000, with 89 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-13 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. This analysis applies to all the models in Tables B-9 through B-12. However, as described below, the cost effectiveness values may be understated when the models are revised to include different conditions and additional equipment.

Table B-13
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

Several items have been named that would increase overall system costs for the five model plants. Included were utility company tendencies toward conservative design, high retrofit costs, additional ductwork requirements, and increased carbon requirements due to incomplete conversion of mercury to its oxidized form. Other items are insufficient carbon quantities to obtain adequate mixing in the gas stream or to provide proper coverage of FF cloth surface, effects on sales of waste ash, adverse coal and ash characteristics, and insufficient allowance for auxiliary equipment. Choice of carbon injection rate and its impact on baghouse type are of particular interest and are described below.

Carbon injection rates used for the models are based on Calgon PCB 6x6 and the equations for its use. However, much experimental work has been done for the DOE and EPRI with a lignite-based carbon (American Norit Company's Darco FGD). This carbon appears to have a mercury adsorption capacity about 25 percent that of the Calgon carbon at 200 °F, indicating that more of the lignite-based carbon would be required for cases where sufficient time is available for equilibrium removal. Some of the experimental work indicates that mercury removal is mass transfer limited, i.e. insufficient time exists for the mercury and injected carbon to mix and come in intimate contact with each other for equilibrium adsorption to take place in the ductwork ahead of the particle removal device (an FF for these models). For this case, the adsorption capacity is not a critical parameter. Under these conditions, higher carbon injection rates are required to achieve high mercury removal rates.

If insufficient mixing time is available for complete adsorption in the flue-gas duct, remaining adsorption must take place on and in the filter cake that builds on the baghouse fabric. Where low carbon injection rates are used, insufficient mixing time coupled with the large cloth area in a reverse-air baghouse indicate that higher carbon injection rates are required than are specified for the costing models. This condition also suggests that a pulse-jet baghouse, with its smaller cloth area for the same quantity of flue gas, may be more effective than a reverse-air baghouse, but may still require higher carbon injection rates than used for the costing models. The reduced cloth area becomes covered with carbon more quickly than would occur in the reverse-air baghouse. While the DOE and EPRI work has been done largely on pulse-jet systems, no data have been presented to show the capabilities of a reverse-air baghouse.

Where coal characteristics provide flue gas and ash that interfere with the adsorption process and cause poor adsorption, higher carbon injection rates are required. The effects of these characteristics may include high SO_2 or HCl levels for which low injection temperatures are not appropriate, high elemental mercury levels that require high carbon rates, and ash that has no mercury adsorption capability. For cases in which ash has high mercury adsorption capability, low injection rates may be appropriate.

To account for the factors described above, costs (prepared by DOE) are shown in Table B-14 for comparison with costs taken from Tables B-9 through B-12. The DOE costs include increased carbon injection rates and additional system equipment.

Table B-14
Sensitivity Analysis for Model Boiler Characteristics[a]

Characteristic	Model 1a	Model 1b	Model 1c	Model 3a	Model 3b
Carbon usage (g carbon/g Hg)	34,200 100,000	460 9,400	460 30,000	17,200 100,000	460 12,600
Capital Cost (10^6 $)	1.26 6.14	33.7 41.6	5.52 7.76	0.167 0.708	4.56 5.63
Annual Cost (10^6 $/yr)	10.1 31.0	7.94 11.6	2.26 12.1	0.66 3.25	1.29 1.79
Cost Effectiveness (mils/kWh)	1.82 5.58	1.43 2.10	0.40 2.19	1.16 5.71	2.09 3.15
Cost Effectiveness ($/lb Hg)	22,100 67,700	17,400 25,400	4,940 26,500	14,200 70,000	27,700 38,600

[a] Boilers costed using Calgon equations are given first.

The sensitivity analysis in Table B-14 shows that cost effectiveness values increase by factors of about 1.4 to 5.5, depending on the model, for mercury removal systems with higher injection rates and higher equipment costs than used for Tables B-9 through B-12. The largest increases are for models not using the combination of cooling, carbon injection, and FF. Annual costs increase by factors of about 1.2 to 3.2, with the larger increases for models that use only carbon injection. It is likely that actual costs would lie between the two extremes, with the higher end being favored.

B.3.3 Carbon Filter Beds

Table B-15 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.)

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 O_2 (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

Table B-15
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.

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 evaporization 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-16 and B-17 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 O_2) (U.S. EPA, 1973), the cost effectiveness is $2,280/kg ($1,040/lb) of mercury removed.

Table B-16
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]			
Capacity Factor (%)	90		
	90		
CAPITAL COST ($)[d]	Hydrogen Stream	End-Box Stream	Combined Stream
Secondary Cooler	117,000	106,000	---
Chiller	165,000	165,000	---
Mist Eliminator	133,000	120,000	---
Depleted Brine Scrubber with Alkaline Scrubber	425,000	386,000	---
Total Capital Costs (TCC)	839,000	777,000	1,620,000
COMBINED ANNUAL OPERATING COSTS ($/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-17
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.

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 O_2), 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.