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Study of Hazardous Air Pollutant Emissions from Electric Utility Steam Generating Units -- Interim Final Report

Volume 1.

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Glossary

HAPs	Hazardous Air Pollutants
MW	Megawatt
lb/MMBtu	Pounds per Million British Thermal Units
μm	Micron (Micrometer)
μg	Microgram
lb/trillion Btu	Pounds per Trillion British Thermal Units
Nm^3	Dry Normal Cubic Meter (1 atm, 0°C)
dscm	Dry Standard Cubic Meter (1 atm, 20°C)
m/s	Meters per Second
MEI	Maximum Exposed Individual
HEM	Human Exposure Model
WOE	Weight of Evidence
IURE	Inhalation Unit Risk Estimate
RfC	Inhalation Reference Concentration
TEQ	Toxic Equivalent Emissions
HQ	Hazard Quotient
CNS	Central Nervous System
AALG	Ambient Air Level Guidelines
PEL	Permissible Exposure Limit
MIR	Maximum Individual Risk
ng/L	Nanograms per Liter
ng/m^3	Nanograms per Cubic Meter
pg/m^3	Picograms per Cubic Meter
the Act	Clean Air Act as amended in 1990
a utility	a fossil-fuel-fired electric utility steam generating plant

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PREFACE

Overview of Report

This interim final report on hazardous air pollutant (HAP) emissions from fossil fuel-fired electric utility steam generating units (i.e., utilities) has been prepared by the United States Environmental Protection Agency (EPA) pursuant to section 112(n)(1)(A) of the Clean Air Act, as amended in 1990 (the Act). This report provides the Congress and the public with information regarding the emissions, fate, and transport of utility HAPs.

The primary components of this interim report are the following: (1) a description of the industry; (2) an analysis of emissions data; (3) an assessment of hazards and risks due to inhalation exposures to numerous HAPs (e.g., arsenic, nickel, cadmium, chromium, beryllium, and others [but excluding mercury]); (4) an assessment of risks due to multipathway (inhalation plus non-inhalation) exposure to one class of HAPs (radionuclides); (5) a general assessment of the fate and transport of mercury through various environmental media; and (6) a discussion of alternative control strategies.

The assessment for mercury in this interim report includes a description of mercury emissions, deposition estimates, control technologies, and a dispersion and fate modeling assessment which includes predicted levels of mercury in various media (including soil, water, and freshwater fish). These predicted levels are based on modeling of mercury emissions from four representative utility plants using hypothetical scenarios. The EPA has not evaluated exposures to mercury emissions from utilities for humans or wildlife in this interim report. If appropriate and feasible, the EPA may include these analyses in the final report.

To provide general information regarding potential background levels of several HAPs (i.e., mercury, arsenic, cadmium, lead, and dioxins) in the environment due to all sources (natural and anthropogenic), this interim report presents measured levels in various media (e.g., soil, air, water, and food products) as reported by various studies.

Assessments of human exposures to mercury and the associated risks of health effects were included in previous drafts of this report and in a related draft EPA report (Mercury Study Report to Congress [i.e., mercury study]). However, during external review of these draft reports, several critical issues related to the mercury risk assessment, including the impending release of new mercury health data, were raised. As a result of that review, the Agency plans to complete the mercury study when two important on-going human health studies are published and reviewed. At this time, the EPA believes that it is appropriate to exclude such assessments for mercury until after the mercury study is issued. However, this issue is still under consideration and negotiation, and may be dependent on results of additional peer review and other factors.

With regard to non-inhalation exposures (e.g., ingestion) to other HAPs, this report presents a limited qualitative discussion of arsenic, cadmium, dioxins, and lead. However, non-inhalation exposures were not estimated for these four HAPs because of the complexity and the intensive data requirements of such analyses. The EPA recognizes that non-inhalation exposures could be important for these HAPs. Therefore, the EPA has initiated a multipathway assessment for arsenic, and may consider conducting assessments for additional HAPs in the future.

This report is not a final report because the assessment of impacts to public health is not yet complete. For example, as indicated above, the evaluation of risks due to non-inhalation exposures was limited. In addition, conclusions regarding the significance of the risks, as well as the regulatory determination required in section 112(n)(1)(A), are not provided.

The EPA plans to publish a final utility HAP report at a later date which will include a more complete assessment of the exposures, hazards, and risks due to utility HAP emissions, and will include conclusions, as appropriate and feasible, regarding the significance of the risks and impacts to public health. In addition, the EPA plans to include in the final report a determination as to whether regulation of HAPs from utilities under section 112 is appropriate and necessary, as required by section 112(n)(1)(A) of the Act and a court order. This court order was issued pursuant to litigation filed against the EPA for failing to meet the statutory deadline for the utility report. The EPA intends that this regulatory determination would be a decision, based on the estimated impacts to public health, whether or not to pursue a regulatory development program under section 112. During any regulatory development process, the EPA would evaluate a range of potential control technologies and emission reduction options and their associated costs.

There are uncertainties, data gaps, and limitations to the current analyses, which are discussed throughout this interim report. If new data become available or improvements are made to the analyses, these changes will be included in the final report.

Peer Review

Draft versions of this report were reviewed during the summer of 1995 by numerous non-EPA scientists representing industry, environmental groups, academia, and other organizations. In the Spring of 1996, the draft report underwent additional review by EPA, State and local air pollution agencies, and other Federal agencies. In addition, a revised draft interim report underwent an expedited review (1 week) by State and local air pollution agencies and other Federal agencies during September 1996.

The EPA has revised the report, as appropriate, based on the reviewers' comments. However, there were several comments that could not be fully addressed because of limitations in data, methods, and resources. At the end of each Chapter, the EPA has included comments received from other Federal Agencies (e.g., Department of Energy, Food and Drug Administration, National Marine Fisheries Service) that were not fully addressed, along with relevant explanations, as appropriate.

Draft versions of this report, along with all the comments received, have been submitted to the docket (A-92-55) and are available for public inspection.

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EXECUTIVE SUMMARY

ES.1 BACKGROUND

This interim final report on emissions of hazardous air pollutants (HAP) from fossil fuel-fired electric utility steam generating units (i.e., utilities) was prepared by the United States (U.S.) Environmental Protection Agency (EPA) pursuant to section 112(n)(1)(A) of the Clean Air Act (the Act), as amended in 1990. The primary components of this interim report are: (1) a description of the industry; (2) an analysis of emissions data; (3) an assessment of hazards and risks due to inhalation exposures to numerous HAPs (e.g., arsenic, nickel, chromium); (4) an assessment of risks due to multipathway (inhalation plus non-inhalation) exposure to one class of HAPs (i.e., radionuclides); (5) a general assessment of the fate and transport of mercury through environmental media; and (6) a discussion of alternative control strategies.

The study was based on two scenarios: (1) 1990 base year emissions; and (2) 2010 emissions. The 1990 scenario was chosen since that was the year the Amendments to the Act were passed and was the latest year for which utility operational data were available. The 2010 scenario was selected to meet the section 112(n)(1)(A) mandate to evaluate hazards "after imposition of the requirements of the Act." Primarily, this meant assessing the hazards after the on-going and future regulatory activities under other provisions of the Act (e.g., ambient air quality and acid rain programs) are in place. The 2010 scenario also included estimated changes in HAP emissions resulting from projected trends in fuel choices and electric power demands.

ES.2 DESCRIPTION OF INDUSTRY

A total of 684 utility plants were identified in the U.S. These utilities are fueled primarily by coal (59 percent of total units), oil (12 percent), or natural gas (29 percent). Many plants have two or more units (i.e., boilers) and several plants burn more than one type of fuel (e.g., contain both coal- and oil-fired boilers). There are 426 plants that burn coal as one of their fuels, 137 plants that burn oil, and 267 plants that burn natural gas.

There are many different types of facilities, varying in boiler type, emission control devices (controls), and other characteristics. Based on data for 1990, all coal-fired units and about one-third of oil-fired units use some form of particulate matter (PM) control. Approximately 15 percent of coal-fired units utilize add-on controls for sulfur dioxide (SO_2). Approximately 70 percent of oil- and gas-fired units employ controls for nitrogen oxides (NO_x); and 80 percent of coal-fired units have NO_x controls.

ES.3 EMISSIONS DATA ANALYSIS

Emission estimates for the years 1990 and 2010 were based on emissions test data from 52 units obtained from extensive emission tests by the Electric Power Research Institute (EPRI), the Department of Energy (DOE), the Northern States Power Company, and the EPA. The testing program was designed to test a wide range of facility types with a variety of control scenarios; therefore, the data are considered generally representative of the industry. However, there are uncertainties in the data because of the small sample sizes for specific boiler types and control scenarios.

These test data provided the basis for estimating average annual emissions for each of the 684 plants. A total of 67 HAPs were identified in the emissions testing program as potentially being emitted by utilities. Tables ES-1 and ES-2 present estimated emissions for a subset of HAPs.

The average annual emissions estimates are considered appropriate for assessing long-term exposures on a national basis. However, since the EPA did not have emissions test data for each utility in the U.S., there may be individual plants for which the EPA either underestimated or overestimated emissions. Based on an uncertainty analysis, the average annual emissions estimates are predicted to be roughly within a factor of plus or minus three of actual annual emissions. However, this analysis had limitations. For example, the analysis did not include data on potential upsets or unusual operating conditions; therefore, the range of uncertainty could be greater. The range of uncertainty for short-term emissions has not been determined.

ES.4 GENERAL APPROACH TO EXPOSURE AND RISK ASSESSMENT

Most of the risk assessment focused on inhalation exposure. All 67 HAPs were assessed for inhalation exposures, at least at a screening level. Non-inhalation exposures are presented for one class of HAP (radionuclides).

For many of the 67 HAPs, inhalation exposure is believed to be the dominant exposure pathway. However, for HAPs that are persistent, bioaccumulate, and are toxic by ingestion, the non-inhalation exposure pathways are likely to be more important. In addition to radionuclides, the EPA also identified five other

Table ES-1. Nationwide Utility Emissions for a Subset of HAPs

HAP	Nationwide HAP emission estimates (tons per year)[a]					
	Coal (426 plants)		Oil (137 plants)		Natural gas (267 plants)	
	1990	2010	1990	2010	1990	2010
Arsenic	54	54	5	3	0.16	0.25
Cadmium	1.9	2.3	1.7	0.9	0.054	0.086
Chromium	70	83	4.7	2.4	1.2	1.9
Lead	72	83	11	5.6	0.44	0.68
Mercury	51	65	0.25	0.13	0.0016	0.0024
Nickel	48	57	390	200	2.3	3.5
Hydrogen chloride	140,000	150,000	2,900	1,500	NM[b]	NM
Hydrogen fluoride	20,000	26,000	140	73	NM	NM
Dioxins[c]	0.00015	0.00020	1×10^{-5}	5×10^{-6}	NM	NM

[a] The emissions estimates in this table are derived from model projections based on a limited sample of specific boiler types and control scenarios. Therefore, there are uncertainties in these numbers. Based on an uncertainty analysis conducted for this study, the EPA predicts that the emissions estimates for individual plants are generally within a factor of roughly three of actual emissions.

[b] NM = Not measured.

[c] These emissions estimates were calculated using the toxic equivalency (TEQ) approach, which is based on the summation of the emissions of each congener after adjusting for toxicity relative to 2,3,7,8-tetrachlorodibenzo-p-dioxin (i.e., 2,3,7,8-TCDD).

HAPs (mercury, arsenic, dioxins, cadmium, and lead), that could present additional impacts due to non-inhalation exposures. The dispersion, fate, and environmental concentrations of mercury were evaluated; however, exposures and risks were not estimated. The other four HAPs (arsenic, dioxins, lead, and cadmium) were examined qualitatively for their potential for multipathway hazards. However, multipathway exposure assessments were not conducted for these four HAPs. The EPA recognizes that for mercury, as well as these other four HAPs, non-inhalation exposures could be important. Quantitative analyses were not performed for arsenic, cadmium, dioxins, and lead because of the complexity of such analyses, the intensive data requirements of such analyses, and because of the limited chemical-specific data available (e.g., chemical-specific air-to-plant biotransfer factors, bioconcentration factors, chemical-specific plant uptake rates) for conducting such analyses. The EPA plans to continue assessing the multipathway exposures and hazards for mercury. The EPA has initiated a multipathway assessment for arsenic.

Table ES-2. Estimated Emissions From Characteristic Utility Units (1990; tons per year)

Fuel:	Coal	Oil	Natural gas
Unit size (MWe):	325	160	240
Arsenic	0.081	0.016	0.0003
Cadmium	0.00051	0.0077	NC[b]
Chromium	0.086	0.018	NC
Lead	0.075	0.053	NC
Mercury	0.05	0.0012	NC
Hydrogen chloride	190	9.4	NC
Hydrogen fluoride	14	NC	NC
Dioxins[c]	0.00000014	0.000000035	NC
Nickel	NC	2.1	0.004

[a] There are uncertainties in these numbers. Based on an uncertainty analysis conducted for this study, the EPA predicts that the emissions estimates are generally within a factor of roughly three of actual emissions.

[b] NC = Not calculated.

[c] See footnote b of Table ES-1.

Multipathway analyses may be undertaken for some of the other HAPs (e.g., dioxins) in the future should the EPA determine that such analyses are feasible and warranted, and as resources allow.

ES.5 SCREENING ASSESSMENT

Initially, the EPA conducted a screening assessment that considered inhalation and non-inhalation exposure routes for all 67 HAPs to identify priority HAPs for more detailed assessment. To screen for inhalation exposures, the EPA used the Human Exposure Model (HEM) to model the 67 HAPs from all 684 utility plants utilizing generally conservative assumptions (i.e., assumptions that are more likely to overestimate rather than underestimate risks) to estimate inhalation risks for maximally exposed individuals (MEIs). If the MEI risk was above a minimum measure (e.g., exposure greater than one-tenth the inhalation reference concentration [RfC] or cancer risk greater than 1 chance in 10 million), then the HAP was chosen for more study. For non-inhalation exposures, the 67 HAPs were prioritized by considering four criteria: (1) persistence; (2) tendency to bioaccumulate; (3) toxicity by ingestion; and (4) quantity of emissions.

Based on this assessment, 15 HAPs (arsenic, beryllium, cadmium, chromium, lead, manganese, mercury, nickel, hydrogen chloride [HCl], hydrogen fluoride [HF], acrolein, dioxins, formaldehyde, n-nitrosodimethylamine, and radionuclides) were identified as priority based on their potential to pose impacts to public health due to inhalation or non-inhalation exposures. The other 52 HAPs were not evaluated beyond the screening assessment.

ES.6 INHALATION RISK ASSESSMENT – LOCAL ANALYSIS

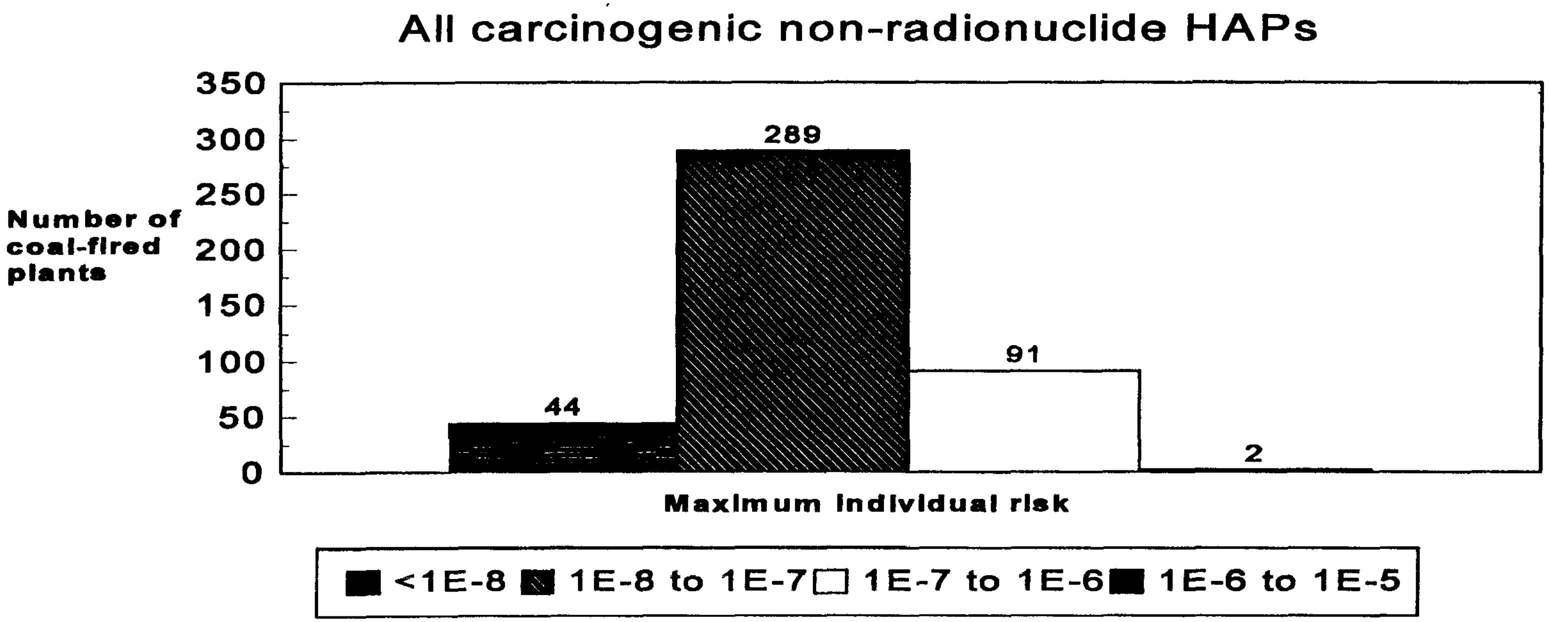
The EPA estimated inhalation exposures and risks due to dispersion of HAP emissions within 50 kilometers (km) of each of the 684 plants individually (i.e., local analysis). For 14 of the 15 HAPs, the HEM was used; for radionuclides, the Clean Air Act Assessment Package-1993 (CAP-93) model was used. The cancer risks for gas-fired plants were less than one chance in one million (i.e., 1×10^{-6}) and no noncancer hazards were identified; therefore, gas-fired plants are omitted from the following discussions.

In cases where data were missing or incomplete, the EPA had to make various assumptions. A few of these assumptions are more likely to overestimate risks. Other assumptions used are likely to underestimate risks. Based on an uncertainty analysis conducted for this study, it is estimated that these assumptions taken together lead to a reasonable high-end (i.e., conservative, but not overly conservative) estimate of the risks due to inhalation exposure within 50 km of plants. That is, the risk estimates from the local analysis are estimated to represent approximately the 90th to 95th percentile. Conservative estimates are considered appropriate so that errors are on the side of public health protection.

ES.6.1 Inhalation Cancer Risks for Coal-fired Utilities Based on Local Analysis

The large majority of coal-fired plants (424 of the 426 plants) are estimated to pose lifetime cancer risks (i.e., increased probability of an exposed person getting cancer during a lifetime) of less than 1 chance in 1 million (i.e., 1×10^{-6}) due to inhalation exposure. Only two of the 426 plants are estimated to pose inhalation risks greater than 1×10^{-6} (see Figure ES-1).

Figure ES-1. Distribution of Maximum Individual Risk Posed by HAPs Emitted from Coal-Fired Utilities



Note: The maximum individual risk (MIR) is often presented in either scientific notation or as an exponent. For example, an increased cancer risk of one chance in one million can be expressed as 1 x 10^{-6} or as 1E-6.

The increased lifetime cancer risk due to inhalation exposure to HAP emissions for the highest MEI, based on the local analysis, is estimated to be 5 x 10^{-6}. Arsenic and chromium are the HAPs contributing most to the inhalation cancer risks (Table ES-3). All other HAPs, including radionuclides, were estimated to present inhalation risks less than 1 x 10^{-6}.

The cancer incidence in the U.S. due to inhalation exposure to HAP emissions (including radionuclides) from all 426 coal-fired utility plants based on the local analysis is estimated to be approximately 0.2 cancer case per year (cases/yr), or 1 case every 5 years.

ES.6.2 Inhalation Cancer Risks for Oil-fired Utilities Based on Local Analysis

The majority of the oil-fired plants (more than 114 of the 137 plants) are estimated to pose inhalation cancer risks less than 1 x 10^{-6}. However, up to 22 of the 137 oil-fired plants are estimated to present inhalation risks above 1 x 10^{-6} (see Figure ES-2). Nickel, arsenic, radionuclides, and chromium are the primary contributors to these cancer risks.

The highest contribution to the MEI risk is nickel. The range in MEI risk (see Table ES-4) reflects a range in assumptions regarding the form of nickel being emitted and the associated cancer potency. Nickel subsulfide is a known human carcinogen and appears to be the most carcinogenic form based on available data. Several other nickel species are also potentially carcinogenic but the potencies are not known.

To evaluate the range of potential risks due to nickel emissions, the EPA estimated risks due to nickel emissions using various assumptions for nickel cancer potency. For example, assuming the nickel mix is 100 percent as carcinogenic as nickel subsulfide, the highest MEI inhalation cancer risk due to the aggregate of HAP emissions from the highest risk oil-fired utility plant is estimated to be 1 x 10^{-4}. Assuming the nickel mix is 10 percent as carcinogenic as nickel subsulfide, the highest MEI inhalation risk is approximately 3 x 10^{-5}. The values in Figure ES-2 are based on the assumption that the nickel mix is 100 percent as carcinogenic as nickel subsulfide.

Estimated risks due to inhalation exposure for a subset of HAPs based on the local analysis are presented in Table ES-4. All other HAPs analyzed were estimated to pose inhalation cancer risks below 1 x 10^{-6} for all 137 oil-fired plants.

Table ES-3. Summary of 1990 Inhalation Cancer Risk Estimates from Local Analysis for Coal-fired Utilities

HAP	MEI lifetime risk[a]	Population with lifetime risk $> 1 \times 10^{-6}$	Number plants with MEI lifetime risk $> 1 \times 10^{-6}$
Arsenic	3×10^{-6}	2,400	2
Chromium	2×10^{-6}	110	1
Total[b] (Aggregate of HAPs)	5×10^{-6}	2,400	2

[a] *Estimated MEI risk due to inhalation exposure for the "highest risk" coal-fired plant. Based on an* uncertainty analysis, these estimates are considered reasonable high-end estimates (roughly the 90th to 95th percentile) of the risks for the MEI due to inhalation exposure (see section ES.6.3).

[b] Estimated risk due to inhalation of the aggregate of HAPs assuming additivity of risk for 26 individual carcinogenic HAPs.

Table ES-4. Summary of 1990 Inhalation Cancer Risk Estimates Based on Local Analysis for Oil-fired Utilities

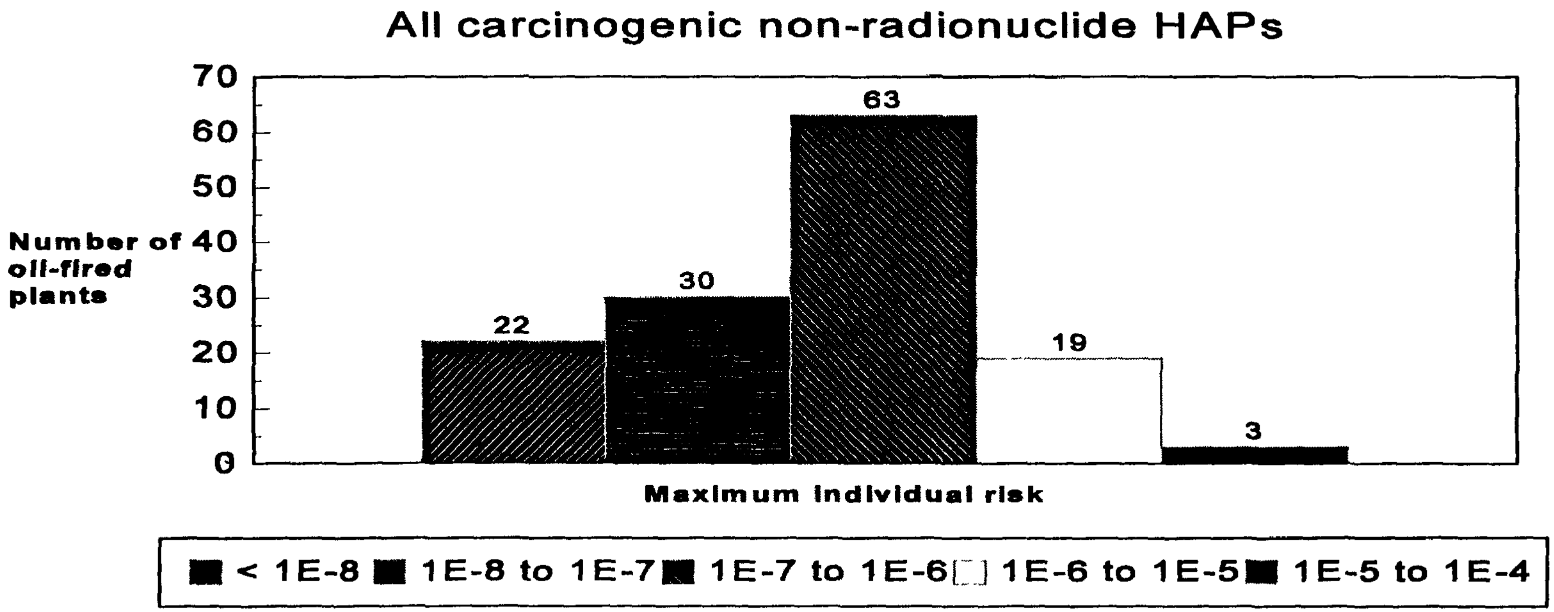
HAP	Highest MEI lifetime risk[a]	Population with lifetime risk $> 1 \times 10^{-6}$	Number plants with MEI lifetime risk $> 1 \times 10^{-6}$
Nickel[b]	1×10^{-5} to 9×10^{-5}	2,400 to 1,600,000	2 to 20
Arsenic	1×10^{-5}	2,400	2
Radionuclides	1×10^{-5}	2,400	2
Chromium	5×10^{-6}	2,300	1
Cadmium	2×10^{-6}	45	1
Total[c] (aggregate)	3×10^{-5} to 1×10^{-4}	2,400 to 1,600,000	2 to 20

[a] Estimated MEI risk due to inhalation exposure to HAPs for the "highest risk" oil-fired plant. Based on *an uncertainty analysis, these estimates are considered reasonable high-end estimates* (roughly the 90th to 95th percentile) of the risks for the MEI due to inhalation exposure. See section ES.6.3 for discussion.

[b] These estimates are presented as a range because of the uncertainties associated with the nickel risk assessment. If the nickel mix is assumed to be 10% as carcinogenic as nickel subsulfide, then the MEI risk for nickel is estimated to be 1×10^{-5}. If the nickel mix is assumed to be 100% as carcinogenic as nickel subsulfide, the estimated MEI risk for nickel is 9×10^{-5}.

[c] Estimated risk due to inhalation of the aggregate of HAPs assuming additivity of risk for 14 individual carcinogenic HAPs. The low end of the range is based on assumption that the mix of nickel compounds is 10% as carcinogenic as nickel subsulfide. The high-end of the range is based on assumption that the mix of nickel compounds is 100% as carcinogenic as nickel subsulfide.

Figure ES-2. Distribution of Maximum Individual Risk Posed by HAPs Emitted from All Oil-Fired Utilities



Note: The maximum individual risk (MIR) is often presented in either scientific notation or as an exponent. For example, an increased cancer risk of one chance in one million can be expressed as 1 x 10^{-6} or as 1E-6.

The cancer incidence in the U.S. due to inhalation exposure to HAP emissions (including radionuclides) from all 137 oil-fired utilities, based on the local analysis, is estimated to be between 0.3 and 0.7 cancer cases/yr. The high end of this range is based on the assumption that the nickel mix is as carcinogenic as nickel subsulfide. The low end of the range assumes that the mix of nickel is 10 percent as carcinogenic as nickel subsulfide.

ES.6.3 Inhalation Cancer Risks Based on Long-Range Transport Analysis

In addition to the above analyses, the EPA conducted long-range transport analyses to assess emissions dispersion and exposures on a national scale. The Regional Lagrangian Model of Air Pollution (RELMAP) was used to estimate the dispersion of HAP emissions from the facility stack out to the borders of the continental U.S. This is in contrast to the HEM, which estimates dispersion and air concentrations within 50 km of the source.

The RELMAP modeling was conducted for all coal- and oil-fired utilities, but was limited to mercury and arsenic. Only arsenic is discussed in this section; the modeling for mercury is presented in section 7. The long-range transport modeling of arsenic indicates that the local HEM analysis alone does not account for a substantial percentage of the population exposures due to utility emissions. A comparison of the HEM results to the RELMAP results for arsenic indicates that a significant portion of emissions disperse further than 50 km, apparently due to the tall stack heights and other dispersion factors. Based on the RELMAP analysis, the nationwide dispersion of arsenic emissions leads to an estimate of population exposure and cancer incidence that is approximately seven-fold greater than the population exposures and cancer incidence predicted by the HEM when only local dispersion is considered (see Table ES-5).

The RELMAP results for arsenic (which is emitted mainly as PM) were used to estimate the potential long-range transport inhalation exposures for cadmium, chromium, nickel, and radionuclides since it is believed that these other HAPs are also emitted as PM and exhibit proportional emission rates and atmospheric dispersion behavior similar to that of arsenic. Because the estimated population exposures resulting from the long-range transport analysis for arsenic were about seven times greater than the population exposures predicted by the local analysis alone, it was also assumed that this ratio also could hold true for nickel, chromium, cadmium, and radionuclides. Using this methodology, the cancer incidence for coal-fired utilities considering both local and long-range transport is estimated to be roughly 1.4 cases/yr (i.e., 0.2 x 7 = 1.4). The

Table ES-5. Summary of Inhalation Risk Estimates Due to Local and Long-range Transport

LOCAL IMPACTS (dispersion within 50 km of each utility plant)[d]

	OIL-FIRED PLANTS		COAL-FIRED PLANTS	
Pollutant	Maximum exposed individual (MEI)	Annual increased cancer Incidence	Maximum exposed individual (MEI)	Annual increased cancer incidence
Radionuclides	1×10^{-5}	0.2	2×10^{-8}	0.1
Nickel[a]	9×10^{-5}	0.4	7×10^{-7}	0.005
Chromium	5×10^{-6}	0.02	2×10^{-6}	0.02
Arsenic	1×10^{-5}	0.04	3×10^{-6}	0.05
Cadmium	2×10^{-6}	0.005	2×10^{-7}	0.0006
All Others[b]	8×10^{-7}	0.005	8×10^{-7}	0.004
Total[c]	1×10^{-4}	0.7	5×10^{-6}	0.2

LOCAL PLUS LONG-RANGE IMPACTS (dispersion from utility emission points to borders of continental U.S.)[e]

	OIL-FIRED PLANTS		COAL-FIRED PLANTS	
Pollutant	Maximum exposed individual (MEI)	Annual increased cancer incidence	Maximum exposed individual (MEI)	Annual increased cancer incidence
Radionuclides	Not estimated	1.4	Not estimated	0.7
Nickel[a]	9×10^{-5}	2.8	9×10^{-7}	0.035
Chromium	5×10^{-6}	0.14	3×10^{-6}	0.14
Arsenic	1×10^{-5}	0.28	4×10^{-6}	0.35
Cadmium	2×10^{-6}	0.035	3×10^{-7}	0.04
All Others[b]	8×10^{-7}	0.035	1×10^{-6}	0.03
Total[c]	1×10^{-4}	4.8	7×10^{-6}	1.3

[a] Assumes that the nickel mixture is as carcinogenic as nickel subsulfide.

[b] Estimated risks due to exposure to all remaining HAPs analyzed (i.e., excluding nickel, arsenic, chromium, cadmium, and radionuclides).

[c] This is the aggregate risk (i.e., the risk due to inhalation exposure to all carcinogenic HAPs, assuming additivity of risks).

[d] There are uncertainties associated with these risk estimates. See sections 6.4 for discussion.

[e] These risk estimates are based on an extrapolation of RELMAP modeling results for arsenic to other HAPs. Therefore, there are considerable uncertainties associated with these results. See sections 6.3 and 6.4 for discussion.

cancer incidence for oil-fired utilities (assuming the nickel mix is 100 percent as carcinogenic as nickel subsulfide) is estimated to be as high as 5 cases/yr (i.e., 0.7 x 7 = 4.9). These estimates should be viewed as highly uncertain high-end estimates (particularly the estimate of five cases/yr for oil-fired utilities) because of modeling uncertainties and extrapolations (e.g., using the modeling results for arsenic to predict dispersion and exposure for the other HAPs) and because of the assumption for nickel carcinogenicity.

For risks to the MEI, a comparison between the HEM local dispersion results and the long-range transport modeling results indicates that long-range transport is not as important for the MEI risks as it is for cancer incidence. For example, the MEI risk from the local analyses for coal-fired utilities (i.e., inhalation risk of 5 x 10^{-6}) is increased by approximately 40 percent to roughly 7 x 10^{-6} when ambient concentrations are added from long-range transport of arsenic from all other utilities in the continental U.S. For oil-fired utilities, the long-range transport of HAPs has no impact on the highest MEI inhalation risks because of the remote location of the two highest risk oil-fired plants. Table ES-5 presents a comparison of results from the local versus long-range transport analyses.

ES.6.3 Uncertainties with the Inhalation Cancer Risk Assessment

There are several areas of uncertainty in the inhalation risk assessment including: (1) the impacts of long-range transport; (2) the emissions and health effects of different forms of chromium and nickel; (3) the use of a linear non-threshold high-to-low dose extrapolation model for estimating cancer risks at low exposure concentrations; (4) the impacts of episodic releases resulting from upsets or unusual operating conditions; (5) how residence times and activity patterns impact the exposures; (6) the impacts on sensitive subpopulations; (7) the impacts of background exposures; and (8) the risk of complex pollutant mixtures.

The quantitative uncertainty analysis indicates that the MEI inhalation cancer risk estimates presented above from the local analysis are reasonable high-end estimates of the risks due to inhalation exposure within 50 km of each plant. That is, the estimates are considered generally conservative (i.e., roughly the 90th to 95th percentile). Conservative assumptions are considered appropriate so that errors are on the side of public health protection. The uncertainty analysis suggests that the most likely estimated inhalation risks for MEIs (i.e., central tendency MEI risk estimates) may be roughly 5 to 10 times lower than the MEI estimates presented above.

ES.6.4 Summary of the Inhalation Cancer Risks

For the majority of utility plants (approximately 662 of the 684 plants), the estimated inhalation cancer risks due to HAP emissions are less than 1×10^{-6}. However, several plants (2 coal-fired plants and between 2 and 22 oil-fired plants) are estimated to pose inhalation cancer risks above 1×10^{-6}, and one oil-fired plant is estimated to pose an MEI inhalation cancer risk between 3×10^{-5} and 1×10^{-4}. The cancer incidence in the U.S. due to inhalation exposure to HAP emissions from all utilities (coal-, oil- and gas-fired combined) is estimated to be between 0.5 and 6 cases/yr. Further research and evaluation is needed to more comprehensively assess the inhalation cancer risks, especially the impacts of long-range transport of HAPs and speciation of nickel.

ES.6.5 Inhalation Noncancer Risks

The EPA also assessed noncancer risks (i.e., health effects other than cancer) due to short- and long-term inhalation exposure. Manganese, HCl, HF, and acrolein were found to be the four HAPs of highest potential concern for noncancer effects.

Based on modeling HAPs for all 684 plants with the HEM, the estimated long-term ambient HAP concentrations were generally 100 to 10,000 times below the RfC or similar benchmark. The highest estimated long-term ambient HAP concentration was 10 times below the RfC. The RfC is an estimate (with uncertainty spanning perhaps an order of magnitude) of the daily inhalation exposure of the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime.

Using a short-term air dispersion model that considers all reasonable meteorological conditions, the EPA modeled maximum one-hour concentrations for three HAPs (HCl, HF, and acrolein). The highest short-term exposure was 140 times below the acute reference level.

ES.7 MULTIPATHWAY ASSESSMENT

The utility HAPs were prioritized for potential non-inhalation exposures. The following characteristics were considered: (1) persistence, (2) toxicity, and (3) potential to bioaccumulate. Mercury, radionuclides, arsenic, dioxins, cadmium, and lead were selected as priority for multipathway assessment.

ES.7.1 Mercury Modeling Assessment

To assess the transport and deposition of mercury emissions from utilities and to estimate concentrations in environmental media and biota, three modeling efforts were undertaken: (1) long-range fate and transport modeling, (2) local scale dispersion modeling, and (3) modeling of environmental concentrations. The RELMAP was used to predict long-range dispersion and deposition across the continental U.S. For the local analysis, a model designed to predict deposition of HAPs within 50 km was used. The Indirect Exposure Model (IEM) was used to estimate environmental concentrations.

Three types of hypothetical locations were considered in the modeling analyses: (1) agricultural, (2) lacustrine (near lakes), and (3) urban. Using four model utility plants, and various assumptions and scenarios, mercury concentrations in various environmental media were estimated.

There are significant uncertainties in the models, data inputs, assumptions, and the quantitative results. However, the analyses were useful for gaining a better understanding of the fate and transport of mercury in the environment, and for estimating plausible levels in environmental media.

The modeling also provided information on whether local and/or long-range transport of mercury is important in a variety of scenarios. The models indicate that most of the mercury from utilities is transported further than 50 km from the source.

ES.7.1.1 General Findings for Mercury. Mercury emissions disperse in the atmosphere and deposit to land and water bodies. Deposition is of potential concern because mercury persists in the environment, and bioaccumulates in the food web (especially in the aquatic food web). The form of mercury found in fish tissue is predominantly methylmercury. Of all the media and biota studied, fish have the highest concentrations of mercury in the environment.

ES.7.1.2 Summary of Mercury Assessment Results for Utilities. For the year 1990, coal-fired utilities were estimated to emit approximately 51 tons per year (tpy) of mercury nationwide, which is approximately 21 percent of the 248 tpy of anthropogenic emissions of mercury estimated to be emitted in the U.S. for the years 1990 to 1992. The EPA also estimates that utility mercury emissions will increase to 65 tpy by the year 2010. If one assumes that current anthropogenic activity represents between 40 and 75 percent of the total emissions (anthropogenic plus other emissions [e.g., natural emissions]), one can calculate that U.S. utilities emit roughly 8 to 15 percent of the total emissions of mercury in the U.S.

Recent estimates of global anthropogenic mercury emissions are about 4,400 tpy. Point sources such as fuel combustion; waste incineration; industrial processes (e.g., chlor-alkali plants); and metal ore roasting, refining, and processing are the largest point source categories on a world-wide basis. Given this global estimate, U.S. anthropogenic mercury emissions could account for about 6 percent of the global total, and U.S. electric utilities would account for about 1 percent of global anthropogenic emissions (using 1990 emission estimates).

Based on the RELMAP modeling analysis, approximately 30 percent (i.e., 15 tpy) of utility mercury emissions deposit in the continental U.S. The estimated annual deposition rates resulting from utility mercury emissions range from 0.5 to greater than 10 micrograms per square meter. The highest deposition appears to occur in the eastern half of the U.S., particularly areas such as southeastern Great Lakes and Ohio River Valley, central and western Pennsylvania, large urban areas in the eastern U.S. (e.g., Washington, D.C., New York City) and various locations in the vicinity of large coal-fired utilities. Based on the limited available data, the RELMAP model seems to over- and underestimate mercury values within a factor of two and appears to be relatively unbiased in its predictions.

Although the amount of mercury being emitted from any single utility may seem relatively small, these emissions are of potential concern for a number of reasons. First, mercury is persistent. It is not degraded, but continually accumulates in the environment. Consequently, over time there is potential for concentrations in the environment to build up. Second, mercury bioaccumulates in the food web. Third, current scientific evidence indicates that most of the mercury emitted to the atmosphere from sources such as utilities, which have tall stacks, does not deposit near the source but is deposited farther away. As a result, even though the ambient concentration of mercury around a single source may not be elevated, there are sufficient data from which to conclude that the cumulative impact of many small sources may lead to the accumulation of mercury in the soils and sediments, and bioaccumulation in freshwater fish. Therefore, the incremental emissions of mercury from utilities, added to the mercury emissions from all of the other sources, contribute to overall environmental loadings, and thus, may contribute, to some degree, to the mercury levels in freshwater fish.

The modeling assessment in conjunction with available scientific knowledge, suggests that there is a plausible link between emissions of mercury from utilities and the mercury found in soil, water, air, and freshwater fish. As noted above, there are many sources of mercury emissions worldwide, both natural and anthropogenic. The fish methylmercury levels are probably due, in part, to mercury emissions from all of these various sources over time. The coal-fired utilities are one category of the mercury sources. The EPA has not yet determined whether the mercury emissions from utilities are a concern for public health.

The EPA recognizes that there are significant uncertainties regarding the extent of the exposures and risks due to utility mercury emissions, and that further research and evaluation is needed to reduce uncertainties and to characterize the exposures and risks. Areas of uncertainty include the following: (1) what exposure levels are likely to result in adverse health effects; (2) what percent of mercury emissions are elemental versus divalent mercury; (3) how much mercury is emitted from natural sources; (4) how much mercury is removed during coal cleaning; and (5) what affects the bioaccumulation of methylmercury in fish. The EPA plans to continue evaluating the exposures and public health impacts due to mercury emissions. In addition, the EPA plans to review new data (e.g., health and exposure data) as they become available and will consider the new data, as appropriate, in future assessments.

Regarding potential methods for reducing mercury emissions, the EPA has not identified any demonstrated add-on control technologies currently in use in the U.S. that effectively remove mercury from utility emissions. (However, there may be add-on control technologies used in other source categories that effectively reduce mercury emissions.) Based on available data, mercury removal by existing PM control devices on utilities varies considerably, ranging from 0 to 82 percent removal, with a median efficiency of 16 percent removal. Existing flue gas desulfurization (FGD) units exhibit poor mercury control, ranging from 0 to 59 percent removal, with a median removal of 17 percent. Pilot-scale studies have shown that mercury removal can be enhanced through the use of activated carbon injection. However, the limited results to date utilizing carbon injection are inconsistent and more data and research are needed. Other various pollution prevention strategies, such as coal cleaning, have shown some effectiveness in reducing utility emissions of mercury. Conventional coal cleaning removes, on average, approximately 21 percent of the mercury contained in the coal. Also, fuel switching, such as switching from coal to natural gas, would result in decreased emissions of mercury.

ES.7.2 Multipathway Assessment for Radionuclides

Radionuclide emissions from utilities may result in human exposure from multiple pathways including: (1) external radiation exposure from radionuclides suspended in air or deposited on the ground, and (2) internal exposure from the inhalation of airborne contaminants or ingestion of contaminated food. The CAP-93 model was used to estimate multipathway exposures and risks due to radionuclide emissions to humans within 50 km of all 684 utilities. However, this assessment did not use site-specific data for the non-inhalation exposure analysis, but rather relied on various generic assumptions and general input data.

Based on the CAP-93 modeling, 667 of the 684 plants are estimated to pose multipathway risks less than 1×10^{-5}. The highest estimated MEI cancer risk due to multipathway exposure to radionuclide emissions from utilities is 3×10^{-5}. Seventeen plants (13 coal- and 4 oil-fired plants) were estimated to pose multipathway risks between 1×10^{-5} and 3×10^{-5}. The estimated cancer incidence in the U.S., due to emissions and dispersion of radionuclides within 50 km of each utility, is estimated to be 0.3 cancer deaths/yr. Including consideration of long-range transport (based on extrapolation from the arsenic RELMAP results), the cancer incidence is estimated to be roughly as high as 2 cancer deaths/yr. The cancer incidence appears to be mostly due to inhalation exposure. The non-inhalation exposures contribute only slightly to the incidence. The non-inhalation exposure pathways have a greater impact on the MEIs, especially for coal-fired plants.

ES.7.3 Qualitative Multipathway Exposure Assessment

Other than radionuclides, the EPA has not assessed the non-inhalation exposures of HAPs emitted from utilities. The EPA recognizes that non-inhalation exposure pathways could be important for other HAPs (e.g., mercury, arsenic, dioxins, cadmium, lead) that are persistent and tend to bioaccumulate. As indicated above, further evaluation of mercury is planned. The other four HAPs are discussed below.

ES.7.3.1 Arsenic. Multipathway exposures potentially could increase the total arsenic risks. Inhalation cancer risks are estimated to be above 1×10^{-6} for arsenic for 4 plants (2 coal and 2 oil). Arsenic is persistent and has a tendency to bioaccumulate. Ingestion of arsenic can pose a cancer risk, and utilities emit approximately 59 tpy of arsenic nationwide. For these reasons, the EPA has initiated a multipathway assessment for arsenic.

ES.7.3.2 Dioxins. The EPA estimates that coal-fired utilities emit 0.4 pounds per year (lb/yr) of dioxin (toxic equivalents, TEQ) and that oil-fired utilities emit 0.02 lb/yr. These estimates combined are roughly 1 to 2 percent of the nationwide anthropogenic dioxin emissions. However, dioxin emissions data were only available for eight test utility plants; therefore, the emissions data for dioxins from utilities are considered more uncertain than the emissions data for many of the other HAPs.

The highest MEI inhalation cancer risk due to dioxin emissions from any utility was estimated to be 1×10^{-7}. The qualitative multipathway exposure assessment indicates that dioxins are highly persistent, tend to bioaccumulate in the food chain, are highly toxic by low-dose ingestion exposure, and present the greatest exposure through ingestion of contaminated foods. Thus, although the inhalation risks are low, the EPA believes that further evaluation of multipathway exposure for dioxins may be needed to more comprehensively evaluate the risks.

ES.7.3.3 Cadmium and Lead. Cadmium emissions from the vast majority of plants (i.e., 683 of the 684 plants) are estimated to pose inhalation risks less than 1×10^{-6}, and the highest modeled air concentration of lead was 200 times below the national ambient air quality standard (NAAQS) for lead. Yet, cadmium and lead are persistent, may bioaccumulate, and are toxic by ingestion. Therefore, the EPA may consider conducting further evaluations of multipathway exposures of cadmium and lead emissions from utilities in future analyses.

ES.7.3.4 Nickel and Chromium. Nickel and chromium were not considered to be priority for non-inhalation exposures. At relatively high oral doses, nickel and chromium do cause noncancer toxicity. However, at relatively low ingestion doses (below the toxic threshold), nickel and chromium are considered to be relatively nontoxic. Also, it is highly uncertain whether they pose a carcinogenic risk by ingestion. Therefore, these metals appear to be mainly a concern from inhalation exposure. Hence, the EPA does not plan to assess multipathway exposures for nickel and chromium for utilities.

ES.8 ALTERNATIVE CONTROL STRATEGIES

There are numerous potential alternative control technologies and strategies for HAPs. These include precombustion controls (e.g., fuel switching, coal switching, coal cleaning, coal gasification), combustion controls, post combustion controls (e.g., PM controls, SO_2 controls), and nontraditional controls (e.g., demand side management [DSM], pollution prevention, energy conservation). The degree of feasibility, costs, and effectiveness of each of these potential control technologies varies. For example, coal cleaning tends to remove at least some of all the trace metals, with lead concentrations being removed to the greatest extent (averaging approximately 55 percent removal) and mercury being removed the least (averaging approximately 21 percent). Existing PM controls tend to effectively remove the trace metals (with the exception of mercury) while FGD units remove trace metals less effectively and exhibit more variability. Fuel switching (e.g., switching from coal to natural gas) could result in substantial reductions in HAP emissions. There are few existing data that show the HAP reduction effectiveness of DSM, pollution prevention, and energy conservation. These control strategies need to be examined further for technical and economic considerations.

ES.9 OTHER ISSUES AND FINDINGS

ES.9.1 Emissions and Risks for the Year 2010

In addition to the 1990 analysis, the EPA also estimated emissions and inhalation risks for the year 2010. There are substantial data gaps and uncertainties in the projections to the year 2010. However, the approach utilized is reasonable given the limitations of data to complete such projections.

Based on EPA's assessment for this interim report, HAP emissions from coal-fired utilities are predicted to increase by 10 to 30 percent by the year 2010. However, based on EPA's analysis, the inhalation risks in 2010 for coal-fired utilities are estimated to be roughly equivalent to the 1990 inhalation risks. For oil-fired plants, emissions and inhalation risks are estimated to decrease by 30 to 50 percent by the year 2010. Multipathway risks for 2010 were not assessed. Utilization of add-on controls to comply with other provisions of the Act are not expected to significantly impact on HAP emissions due to their limited numbers and limited HAP control efficiency improvement. However, if additional actions are taken to reduce emissions of criteria pollutants and acid rain precursors (e.g., add-on controls to reduce SO_x and NO_x emissions), these actions could result in reductions in HAP emissions. Other potential (but unknown) actions (e.g., fuel switching, repowering) may have a significant impact on HAP emissions; however, these unknowns were not included in the 2010 projection.

The approach EPA utilized to estimate emissions for the year 2010 is one of several possible approaches for making such projections. Other organizations have made projections that differ from EPA's. For this interim report, the EPA did not conduct alternative approaches and did not compare its results with projections made by other organizations. However, if feasible, the EPA will consider evaluating alternative approaches and comparing the EPA's results with those from other organizations in future analyses.

ES.9.2 Peer Review

Draft versions of Chapters 1 through 10 of this report (not including the Executive Summary) and draft technical support documents were reviewed by numerous non-EPA scientists representing industry, environmental groups, academia, and other parties. The EPA held a scientific peer review meeting and also a public meeting in July 1995 to obtain comments from reviewers. In February, April, and September 1996, all sections of the draft report underwent additional review by EPA, State and local Agencies, and other Federal Agencies. The EPA has revised the report, as appropriate, based on the reviewers' comments. However, there were several comments that could not be fully addressed because of limitations in data, methods, and resources. Comments received by other Federal agencies that could not be substantially addressed are presented at the end of each Chapter. Draft versions of this report, along with all the comments received, have been submitted to the public docket (A-92-55) at the following address: U.S. EPA, Air and Radiation Docket and Information Center, mail code 6102, 401 M Street, S.W., Washington, D.C. 20460; telephone number (202) 260-7548. Materials are available for public review at the docket center or copies may be mailed (for a reasonable fee) on request by calling the above number.

ES.9.3 Industry Report

If alternative methods and assumptions were used to study the HAP emissions from utilities, the results would likely be somewhat different. To assess the impact of using alternative assumptions and methods, it is useful to compare the EPA study with a similar study completed by the EPRI.

The EPRI prepared a report, entitled "Electric Utility Trace Substances Synthesis Report," (November 1994) that paralleled the EPA's study. Many of the same emissions data were used and similar risk assessment methods were utilized. The EPRI study concluded that cancer inhalation risks are below 1×10^{-6} for all utilities, and noncancer inhalation risks are well below Federal threshold levels for all utilities. Population inhalation risks were determined by the EPRI to be insignificant (less than 0.1 cancer case/year). Case studies at four plants found that multimedia risks, including mercury, are below levels of concern.

However, it should be noted that in the EPRI analysis, exposures to mercury through fish consumption were only considered for two of the four plants studied.

The EPRI risk estimates are generally similar to, but in several cases lower than, those of EPA. Differences between the two studies include: (1) EPA's use of a higher unit risk factor for arsenic; (2) EPA's assumption that nickel emissions were carcinogenic (EPRI assumed nickel was not carcinogenic); (3) EPA's evaluation of exposure beyond 50 km to all locations in the U.S. (EPRI did not attempt this analysis); (4) the EPRI radionuclide analysis was based on several model plants, while the EPA evaluated every plant in the U.S.; and (5) the EPRI assumed that chromium emissions were five percent chromium VI, while EPA assumed that 11 percent (for coal-fired plants) and 18 percent (for oil-fired plants) were chromium VI. In addition, the EPRI mercury multimedia study considered only the local impact from four plants (not worst-case) and did not include potential impacts of total nationwide utility mercury emissions and contributions to total environmental loadings.

ES.9.4 Potential Environmental Impacts Not Included in Study

There are other potential environmental issues associated with utilities not assessed in this report. First, this study did not assess the impacts of criteria pollutants (SO_2, NO_x, PM, carbon monoxide, and ozone) or acid rain precursors (SO_2 and NO_x), which are studied and regulated under other sections of the Act. Second, this study did not include an assessment of ecological impacts. Third, this study did not assess the impacts of carbon dioxide emissions. Fourth, this study did not assess the impacts resulting from mining, drilling, solid waste disposal, transmission, transportation, or other activities associated with electric power generation. These issues and potential impacts were not assessed because they were considered beyond the scope of this study as mandated by the Act.

ES.9.5 Link to Particulate Matter (PM)

Arsenic, cadmium, chromium, lead, nickel, and radionuclides are emitted primarily as PM. Consequently, these HAPs may contribute to PM emissions and PM health concerns, especially from poorly controlled coal-fired units and uncontrolled oil-fired units (roughly two-thirds of oil-fired units are uncontrolled for PM). The impacts for PM were not addressed in this study, but are being studied under Title I of the Act. However, if additional controls of PM emissions are utilized, this could result in reductions in HAP emissions.

ES.10 OVERALL SUMMARY

Based on this study, cancer risks due to inhalation exposure to HAP emissions from the large majority of utility plants are less than 1 x 10^{-6}. However, 2 coal-fired plants and up to 22 oil-fired plants are estimated to present inhalation cancer risks above 1 x 10^{-6} (primarily due to nickel, arsenic, radionuclides, chromium, and cadmium). The inhalation cancer risks due to exposure to the remaining HAPs emitted from utilities are estimated to be less than 1 x 10^{-6}. The EPA estimates that between 0.5 and 6 cancer cases/yr occur in the U.S. each year due to inhalation exposure to HAP emissions from utilities.

With regards to noncancer effects from inhalation exposure, the modeling assessment indicates that HAP emissions from utilities are not expected to result in any exceedances of the RfCs or similar inhalation benchmarks.

Further evaluation of the impacts of the long-range transport of HAPs and the speciation of nickel, and also the potential impacts of short-term peak emissions of certain HAPs (e.g., HCl, HF), may be needed to more comprehensively evaluate the inhalation exposures and risks.

Available information indicates that mercury emissions from utilities may contribute to the mercury levels in the environment, including the levels in freshwater fish. However, at this time, the EPA has not yet determined whether the mercury emissions from utilities are a concern for public health. The EPA plans to continue evaluating the potential exposures and potential public health concerns due to mercury emissions from utilities. In addition, the EPA plans to evaluate information on the various potential control technologies for mercury, including pollution prevention options, and the costs, technical feasibility of such measures, and resulting economic impacts. The EPA plans to issue a final Report to Congress at a later date which will include a more complete assessment of the exposures, hazards, and risks due to utility HAP emissions, and will include conclusions, as appropriate, regarding the significance of the risks and impacts to public health. In addition, the EPA plans to include in the final report a determination as to whether regulation of HAPs from utilities under section 112 is appropriate and necessary.

1.0 INTRODUCTION

This chapter presents an introduction to the study of hazardous air pollutant (HAP) emissions from electric utility steam-generating units (utilities). The chapter is divided into three main sections: the legislative mandate that requires this report, the provisions of the 1990 amendments to the Clean Air Act (the Act) related to this study, and an overview of the utility study and its approach to meeting the provisions of the Act.

1.1 LEGISLATIVE MANDATE

In section 112(n)(1)(A) of the Act, Congress mandated that the Environmental Protection Agency (EPA) "... perform a study of the hazards to public health reasonably anticipated to occur as a result of emissions by electric utility steam generating units of ... [HAPs] ... after imposition of the requirements of this Act." The list of HAPs is presented in section 112(b) of the Act. Section 112(n)(1)(A) also requires that:

- Results of this study be presented in a report to Congress by November 1993
- The EPA develop and describe alternative control strategies for HAPs that may warrant regulation under section 112
- The EPA is to proceed with rule-making activities under section 112 to control HAP emissions from utilities if it determines from the study that such regulation is appropriate and necessary.

The Act, in section 112(a)(8), defines an electric utility steam-generating unit as any fossil-fuel-fired combustion unit of more than 25 megawatts electric (MWe) that serves a generator that produces electricity for sale. A unit that cogenerates steam and electricity and supplies more than one-third of its potential electric output capacity and more than 25 MW output to any utility power distribution system for sale is also considered an electric utility steam-generating unit under section 112(a)(8).

The Code of Federal Regulations (CFR), chapter 40, part 60.41a, defines fossil fuels as natural gas, petroleum, coal, and any form of solid, liquid, or gaseous fuel derived from such material for the purpose of creating useful heat. Fossil fuels include coal (bituminous, subbituminous, anthracite, lignite), oil (Nos. 2, 4, and 6), and natural gas.

The wording of section 112(n)(1)(A) did not include a mandate to include an analysis of the cost(s) of alternative controls in the study. Therefore, no cost analyses (e.g., control costs, economic, cost-benefit) were performed as a part of this interim study. These analyses would be conducted during any regulatory effort should regulation be determined to be appropriate and necessary in determining the level of any standard(s).

When this study began, only a small amount of reliable data on HAP emissions from utilities were available; most of the new data did not become available until the beginning of 1994. Because of the lack of data, the submission of this interim final report for this study was delayed until October 1996. The EPA plans to issue a final report at a later date.

This study addresses the impact of pollution controls mandated by other sections of the Act, determines which HAPs are present in utility unit emissions, and partially estimates exposures and risk to humans from the emission of these HAPs.

1.2 ACT PROVISIONS AND STUDIES RELATED TO THIS STUDY

There are several other provisions in the Act that relate to the utility industry and may have impact in the future. This section describes these provisions and their relevance to the study.

1.2.1 Section 112 of the Act - Hazardous Air Pollutants

The 1990 amendments to the Act also mandated five other related studies: (1) the mercury study, (2) the National Institute of Environmental Health Sciences (NIEHS) health effects of mercury study, (3) the National Academy of Sciences (NAS) risk assessment methodologies study, (4) Great Waters study, and (5) the Presidential Risk Commission.

1.2.1.1 Mercury Study. Section 112(n)(1)(B) required that the EPA complete a study of mercury emissions from utilities, municipal waste combustion units, and other sources, including area sources. The study was to consider the rate and mass of mercury emissions, the health and environmental effects of such emissions, technologies that are available to control such emissions, and the costs of such technologies. Section 112(n)(1)(B) mandated that the EPA submit a report to Congress by November 15, 1994, reporting the results of the study. However, the EPA has decided to delay significantly the release of the mercury study report to allow time to incorporate the results of two major studies on the impact of methylmercury on children in fish-eating populations in the Faroe Islands and Seychelles Islands. The delay is necessary so that the EPA can completely fulfill the mandate to evaluate the health effects of mercury.

1.2.1.2 NIEHS Health Effects of Mercury Study. In section 112(n)(1)(c), Congress gave the NIEHS the task of identifying the threshold level of mercury exposure that would not adversely affect human health. A report on the NIEHS study was published in 1993.[1]

1.2.1.3 NAS Risk Assessment Methodologies Study. In January 1995, the National Academy of Sciences finalized a report[2] on the risk assessment methodologies used by the EPA. The results of the NAS study were used to help develop the methodologies for the risk assessment portions of this study.

1.2.1.4 The Great Waters Study. In response to section 112(m), the EPA finalized a report in May 1994 on the atmospheric deposition of pollutants to the "Great Waters," namely, the Great Lakes, Chesapeake Bay, Lake Champlain, and coastal waters.[3] The pollutants of concern to the Great Waters study that are emitted from utilities include lead, cadmium, dioxins, and, in particular, mercury. The report discussed the following:

- The contribution of atmospheric deposition to pollutant loadings in these waters
- Environmental and public health effects of atmospheric pollution that is deposited to these waters
- Sources of pollutants deposited to these waters.

The May 1994 report noted that the Great Waters are polluted by HAPs that originate from local and distant sources; however, more data are needed to identify sources of the pollutants. The recommendations of the May 1994 Great Waters report were: (1) the EPA should strive to reduce emissions of the pollutants of concern through implementation of the Act; (2) a comprehensive approach should be taken, both within the EPA and with other agencies, to reduce and preferably prevent pollution in air, water, and soil; and (3) the EPA should continue to support research for emissions inventories, risk assessment, and regulatory benefits assessment. The May 1994 report was based on a limited amount of existing data and will be updated in the future.

1.2.1.5 Presidential Risk Commission. In section 303 of Title III of the 1990 amendments to the Act, Congress directed that the President form a Commission whose mandate would be to "...make a full investigation of the policy implications and appropriate uses of risk assessment and risk management in regulatory programs under various Federal laws to prevent cancer and other chronic human health effects which may result from exposure to hazardous substances." This Commission meets monthly and is currently working on the draft of its report. Because the report is not due until November 1996, the results were not available for this report. However, the results may be considered in the final utility report to be published at a later date.

1.2.2 Title I - Nonattainment Provisions

Title I includes requirements for attaining and maintaining the national ambient air quality standards (NAAQS). The NAAQS are designed to protect public health and welfare and have been established for six criteria pollutants (ozone, carbon monoxide [CO], particulate matter [PM], lead, sulfur dioxide [SO_2], and nitrogen oxides [NO_x]). Sources in ozone nonattainment areas, including utilities, may need to install NO_x controls to reduce NO_x emissions. These new NO_x controls may affect HAP emissions. Future ozone NAAQS may set even lower ozone concentration limits, and the lower limits could result in the need for additional utility NO_x reductions. Changes in the ambient PM standard (e.g., from PM_{10} to $PM_{2.5}$ or PM_1) may also affect HAP emissions.

1.2.3 Title IV - Acid Deposition Control

Title IV of the Act addresses control of the pollutants associated with acid rain in two phases. The pollutants covered by Title IV are SO_2 and NO_x.

Phase I and Phase II requirements of Title IV grant utility units "allowances" to emit SO_2. Emission allowances are allocated to existing utility units based upon historical operating conditions. One allowance equals the right to emit 1 ton of SO_2. Affected units are required to turn in to the EPA one allowance for each ton SO_2 emitted in a calendar year. Unused or "excess" allowances may be sold on the open market. To comply with the requirements, utilities may do many things, such as: (1) install flue gas scrubbers, (2) switch to a fuel that contains less sulfur ash, or (3) purchase emission allowances. The control option a utility selects to comply with the SO_2 reduction requirements may also have an effect on HAP emissions.

The Phase I requirements affect 202 boiler units at 110 utility plants. These high-SO_2-emitting, coal-fired utility units must comply with the Phase I requirements by January 1995. Under Phase II, all utility units will be covered by 2000. Both

Phase I and Phase II require facilities to install continuous emission monitoring systems for SO_2, NO_x, and volumetric flow to ensure compliance and provide an accurate basis for allowance trading.

Under Phase I, the EPA required that tangential-fired and dry-bottom wall-fired boilers meet annual average NO_x emission limits of 0.45 pound per million British thermal units (lb/MMBtu) and 0.50 lb/MMBtu, respectively, by January 1, 1996. Utilities could meet the limits by installing low-NO_x burner technology or other combustion control technology or by averaging emissions among several units. This rule was issued as a direct final rule on April 13, 1995 (60 FR 18751).

Under Phase II, the EPA will establish NO_x emission limits for all other boilers, including wet-bottom wall-fired boilers and cyclones, by January 1, 1997, and the affected units must be in compliance by 2000. The EPA will also reevaluate and revise, if necessary, the standards established under Phase I to implement any new technologies that could meet more stringent NO_x emissions limits (57 FR 55633). Units that do not meet the NO_x emission limits may install controls or average emissions among several units. The effects of Title IV on utility HAP emissions were estimated in this study.

1.2.4 New Source Performance Standards (NSPS)

Emissions of SO_2, NO_x, and PM from utilities are subject to NSPS for new or modified sources, pursuant to section 111 of the Act. Units greater than 73 MW heat input that commenced construction or modification after August 17, 1971, are subject to requirements of the NSPS (40 CFR part 60, subparts D or Da).

Under section 407 of the Act, the EPA must revise the NSPS requirements for NO_x emissions from utility and nonutility units to reflect improvements in emission reduction methods. Furthermore, future NO_x emission limits could be set to minimize the multiple environmental effects of NO_x on ground-level ozone formation, ozone and nitrous oxide formation in the atmosphere, nitrogen enrichment in water, and acid rain.

The NSPS are technology-based standards and are designed to reflect the degree of emission limitation achievable through application of the best demonstrated technology that is also cost-effective. The NSPS program also results indirectly in the control of some HAPs. For example, NSPS that limit emissions of PM will also control HAPs that are PM or that condense onto the PM in the affected gas streams. Furthermore, installation of

SO_2 scrubbers will also control some vapor-phase HAPs such as hydrogen chloride (HCl) and hydrogen fluoride (HF).

1.3 OVERVIEW AND APPROACH OF ELECTRIC UTILITY HAP STUDY

This report is the result of the work of government and nongovernment personnel. Emissions testing and emission estimation issues were discussed among numerous branches within the EPA and among representatives of industry, the Electric Power Research Institute (EPRI), and the Department of Energy (DOE). In particular, EPRI, DOE, and the EPA coordinated their utility emissions testing to cover more plant configurations and obtain as much information as possible for the assessment. This report, and the data and methodologies utilized, was reviewed by numerous scientific experts within and outside the Agency. Outside reviewers included representatives from industry, other Federal Agencies, State and local agencies, academia, and environmental organizations.[a]

The report is organized as follows. The electric utility industry is described in Chapter 2, including the types of fossil fuels, boilers, and air pollution control devices in use in the year 1990, as well as changes in control devices and fuel usage expected for the year 2010. Chapter 3 describes emissions testing conducted since 1990, the determination of emission modification factors (EMFs) from test reports, and the estimation of emissions for several characteristic units using a computer emission factor program. Chapter 4 introduces the health hazard assessment. The screening risk assessment used to determine the priority HAPs is described in Chapter 5. Chapter 6 discusses the inhalation route for HAP exposure, while Chapters 7, 8, and 9 address mercury, arsenic, dioxins, lead, cadmium, and radionuclides noninhalation exposure. Alternative control strategies for HAP emissions reductions are given in Chapter 10. Chapter 11 presents the conclusions of the study. Additional supporting material is provided in the appendices.

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[a] Reviewers provided comment through a variety of venues (e.g., EPA Work Group, scientific peer review, Federal interagency review, public comment period). However, participation by a reviewer did not imply agreement with the methodology or conclusions presented by the EPA. All comments were considered during revision of the document and will continue to be considered during preparation of the final report.

1.4 REFERENCES

1. U.S. Department of Health and Human Services, National Institutes of Health. Report to Congress on Methylmercury. NIEHS, Research Triangle Park, NC. 1993.

2. National Academy of Sciences, National Research Council. *Science and Judgement in Risk Assessment*. Washington DC. 1994.

3. U.S. Environmental Protection Agency. *Deposition of Air Pollutants to the Great Waters, First Report to Congress*. EPA-453/R-93-055. Office of Air Quality Planning and Standards, Research Triangle Park, NC. May 1994.

2.0 CHARACTERIZATION OF THE INDUSTRY

This chapter presents a characterization of the utility industry. This is provided as a brief background for those who may not be familiar with the industry. In addition, some components of the process itself (e.g., type of boiler, method of firing, type of emission control) may impact on the generation or emissions of HAPs. These process components are introduced in this chapter and their impact on HAPs is discussed in chapter 10. The chapter is divided into seven main sections: background of the industry, types and ownership of utilities, utility furnace design, PM control, SO_2 control, NO_x control, and a projected characterization of the utility industry after implementation of the 1990 amendments to the Act. All of the sections except the last describe the utility industry as it existed in 1990. The last section projects conditions that are expected to exist in 2010, after the amendments are fully implemented.

2.1 INDUSTRY BACKGROUND

An electric utility steam-generating unit (i.e., utility) is defined (section 112(a)(8) of the Act) as any fossil-fuel-fired combustion unit of more than 25 MW that serves a generator producing electricity for sale. It can also be defined as a unit that cogenerates steam and electricity and supplies more than one-third of its potential electric output capacity and more than 25 MW electrical (MWe) output to any utility power distribution system for sale.

Utilities are fueled primarily by coal, oil, or natural gas. Figure 2-1 shows the 1990 distribution of fossil fuels burned by the electric utility industry by unit (boiler) and by total megawatts.[1] Coal-fired boilers account for the largest portion of the industry by number of units (1,097 units, 56 percent), representing 66 percent of the industry's total megawatts. Gas-fired boilers make up 33 percent of the industry's units (663 units) and account for 24 percent of the total megawatts. Oil-fired boilers account for 11 percent of the units (227 units) and represent 10 percent of the megawatts. This characterization excludes 151 units that are effectively shut down but that still retain operating permits.

2.2 FOSSIL-FUEL-FIRED ELECTRIC UTILITY STEAM-GENERATING UNITS

This section describes the two basic types of utility facilities and the types of ownership in the industry.

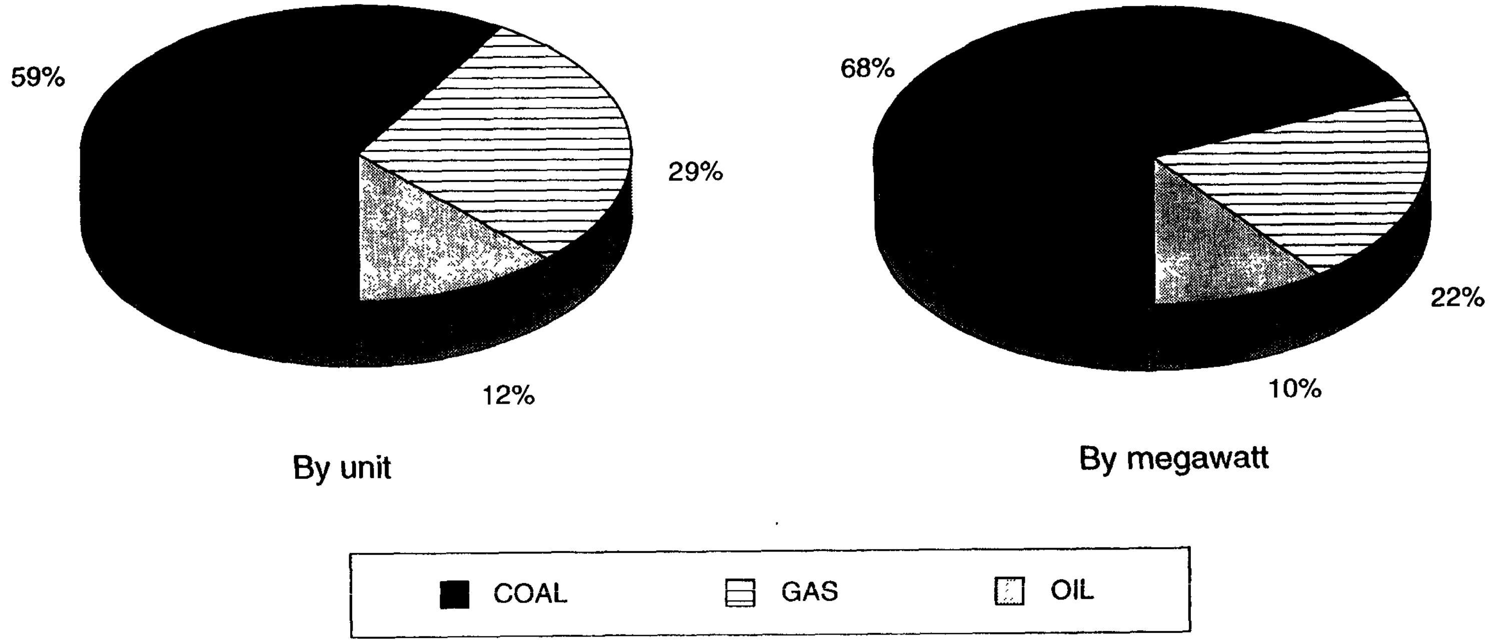

Figure 2-1. Fossil fuel use in the utility industry in 1990.[1]

2.2.1 Types of Electric Utility Facilities

There are two basic types of facilities in the utility industry: conventional utility power facilities and cogeneration facilities. Although both types of facilities share similar designs, their major difference is that conventional utility power facilities produce their power solely for commercial power production while cogeneration facilities produce their power primarily for an industrial purpose and sell excess steam or electricity equal to more than one-third of their potential electric output capacity and more than 25 MWe output to any utility power distribution system.

Conventional facilities consist of units that produce heat in a boiler to make high-pressure steam, which in turn powers units that produce electricity through a combined cycle turbine system or a steam turbine (see section 2.3.4). In both systems, the steam is recycled without being used for any other purpose. Conventional facilities account for most of the utility steam-generating units in the United States (U.S.). In 1990, there were 1,893 conventional utility steam-generating units in the U.S., with 1,043 burning coal of some type. The total output was 493.1 gigawatts (GW) electrical.

Cogeneration is defined as the simultaneous production of power (usually electricity) and another form of useful thermal energy (usually steam or hot water) from a single fuel-consuming process.[2] Cogeneration facilities can also consist of units that produce heat in a boiler to make high-pressure steam that powers a steam turbine to produce electricity or units that produce electricity through a combined-cycle turbine system. Because of their primary uses as industrial power and steam sources, however, they normally are too small to fit the regulatory definition of a utility boiler. There were 218 fossil-fuel-fired cogeneration facilities rated at 25 MWe or greater that provided at least one-third of their excess power to a grid operating in the U.S. as of 1990. These cogeneration facilities consist of coal-, oil-, and gas-fired steam turbines and combined-cycle turbines that provide 21,053 MWe of capacity. This megawatt capacity was less than 5 percent of the total conventional utility capacity in 1990 and was made up of only 54 coal-fired plants (providing 5,098 MWe of capacity) and 12 oil-fired plants (providing 756 MWe of capacity). Thus, the electrical capacity of the coal- and oil-fired cogeneration facilities represented less than 1.2 percent of total utility capacity in 1990.[3]

2.2.2 Types of Ownership

There are four basic types of electric power ownership in the utility industry: publicly owned utility companies, Federal power agencies, rural electric cooperatives, and investor-owned utility generating companies. Publicly owned utilities are not-for-profit and are operated by municipalities, counties, States, or other bodies such as public utility districts. Federal power agencies are Federal government agencies that provide electric power, usually to rural or remote areas. Rural electric cooperatives are private, not-for-profit corporations owned by their members who are also the customers they serve; the cooperatives are not a part of the municipal government. Investor-owned utility generating companies are owned by their investors and sell electricity to make a profit.[4]

The oldest and largest companies (based on total megawatts electric capacity) are the investor-owned utilities. Although numbering only approximately 260 separate companies, investor-owned utilities provide 75.4 percent of kilowatt hour (kWh) generation of electric power to the Nation. Publicly owned utility companies, which consist of approximately 2,017 separate companies, represent 10.6 percent of the Nation's electric power supply. The 10 Federal power agencies generate 8.6 percent of the Nation's electric power supply. Rural electric cooperatives, numbering approximately 939 separate companies, serve 5.4 percent of the Nation's electric power supply. These utilities maintain jointly owned electric power grids to which electric power is supplied and then sold to other utilities, industries, and individual customers.[5]

The fastest growth in the production of electricity (by unit) for the 1990s has been projected for nonutility generators.[6] Ownership of nonutility generators can be further divided into ownership by:

- Units that cogenerate steam and electricity (qualifying facilities[7])[a]
- Small power producers (<80 MWe) that generate electricity primarily from a renewable source
- Other nonutility generators (e.g., independent power producers [IPPs], units that cogenerate steam and

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[a] A qualifying facility, under the Public Utility Regulatory Policies Act (PURPA), Sections 292.303 and 292.305, may buy or sell energy to the local utility or indirectly to other utilities. The local utility is obligated to purchase or sell the energy at a price that is "just, reasonable, and in the public interest" and does not "discriminate against any qualifying facility in comparison to rates for sales to other customers served by the electric utility."

electricity [nonqualifying facilities], and other commercial and industrial units).

Nonutility generating units are generally smaller than other utility units, of newer design technology, and built to fill a specific need for power in their geographic area(s). Nonutility generating units are usually privately owned (although some are sponsored by larger publicly or investor-owned utilities) and sell their power to private customers and the jointly owned electric power grids.

2.3 DESIGN OF ELECTRIC UTILITY UNITS

This section contains a summary of unit designs used in the utility industry. Hazardous air pollutants are either formed during combustion or introduced into the combustion unit (e.g., trace constituents in the fuel). Thus, the design and operation of a unit may impact on the generation and emission of HAPs.

2.3.1 Furnace Types

Utility furnace-fired boilers can be divided into five basic firing types: stoker-, cyclone-, tangential-, and wall-fired boilers and fluidized-bed combustors (FBC).

2.3.1.1 Stoker-fired. Stoker firing is one of the oldest furnace firing methods still in use. In this process, fuel is deposited on a moving or stationary grate or spread mechanically or pneumatically from points usually 10 to 20 feet above the grate.[8] The process utilizes both the combustion of fine coal powder in air and the combustion of larger particles that fall and burn in the fuel bed on the grate.[9] Because of their design, stokers are used only for smaller furnaces firing coal.

2.3.1.2 Cyclone-fired. Cyclone firing uses several water-cooled horizontal burners that produce high-temperature flames that circulate in a cyclonic pattern. The burner design and placement cause the ash to become a molten slag that is collected below the furnace. Because of this slagging system, cyclone-firing furnaces are almost exclusively coal-fired; however, some units can fire oil.[10]

2.3.1.3 Tangential-fired. Tangential-fired boilers are based on the concept of a single flame envelope and project both fuel and combustion air from the corners of the furnace. The flames are directed on a line tangent to a small circle lying in a horizontal plane at the center of the furnace. This action produces a fireball that moves in a cyclonic motion and expands to fill the furnace.[11] Tangential-fired boilers can fire coal, gas, or oil.

2.3.1.4 Wall-fired. Wall-fired boilers are characterized by rows of burners on the wall(s) of the furnace. The two basic forms of wall-fired furnaces are single wall (having burners on one wall) or opposed (having burners on more than one wall that face each other). Circular register burners and cell burners are types of burner configurations found in single-wall or opposed-fired units. A circular register burner is a single burner mounted in the furnace wall, separated from other burners so that it has a separate, distinct flame zone. Cell burners are several circular register burners grouped closely together to concentrate their distinct flame zones. This use of a distinct flame zone is in contrast to the fireball effect created by the tangentially fired furnace.[12] Wall-fired boilers can fire coal, gas, or oil.

2.3.1.5 Fluidized-bed Combustors. In a typical FBC, combustion occurs when coal, together with inert material (e.g., sand, silica, alumina, or ash) and/or a sorbent such as limestone, are suspended through the action of primary combustion air distributed below the combustor floor.[13] "Fluidized" refers to the state of the bed of material (fuel or fuel and inert material [or sorbent]) as gas passes through the bed. As the gas flow rate is increased, the forces on the particles become just sufficient to cause buoyancy. The gas cushion between the solids allows the particles to move freely, giving the bed a liquid-like characteristic.[14]

Fluidized bed combustors can be further divided into circulating fluidized-bed (CFB) and bubbling fluidized-bed (BFB) steam generators. The main difference between these two types is the state of fluidization, which in turn depends mainly on the bed particle diameter and fluidizing velocity. The CFB combustors have relatively high velocities and fine bed particle size, while the BFB combustors have relatively low velocities and coarse bed-particle size.[15,16]

Most FBCs are of the atmospheric fluidized-bed (AFBC) type, which, as the name suggests, operate at atmospheric pressure. A newer and potentially promising type of FBC is the pressurized fluidized-bed combustor (PFBC). These combustors are physically smaller (yet maintain the same megawatt capacity as equivalent AFBCs), operate at 10 to 20 times atmospheric pressure, and incorporate a gas turbine in their power production cycle. Because of these features, PFBCs offer a potentially significant gain in overall thermal efficiency over AFBCs.[17]

2.3.1.6 Distribution of Furnace Types. Figure 2-2 shows the 1990 distribution of furnace types by fuel in the utility industry by unit and by total megawatts.[1] Wall-fired designs account for the largest portion of the coal-fired units by number of units (50.2 percent), which represents 47.8 percent of the coal-fired units' total megawatts. The second and third most common designs are the tangential-fired and cyclone-fired units. Tangential firing is used in 39.8 percent of the units (43.6 percent of the total megawatts), and cyclone firing is used in 8.4 percent of the units (8.3 percent of the total megawatts). Stoker-fired boilers and FBCs account for about 1.6 percent of designs among the coal-fired units (0.3 percent of the total coal-fired megawatts). Wall-fired designs represent the largest portion of gas- and oil-fired units by number of units (68 percent), which represents 64 percent of the total megawatts. The second most common design is the tangential-fired unit. Tangential-fired units represent 30 percent (34 percent of the total megawatts) of the gas- and oil-fired units, and combined-cycle gas turbine units account for about 2 percent (2 percent of the total megawatts) of designs for gas- and oil-fired units.

2.3.1.7 Effects of Furnace Type on HAP Emissions. Many of the organic HAPs leaving a furnace in the gas stream are produced in the combustion zone and succeeding parts of the gas path. Factors expected to affect the types and quantities of HAPs produced and emitted include temperature, residence time, fuel characteristics, firing scheme, bottom-ash and/or fly-ash partitioning, and adsorption onto ash. In comparison, essentially all elemental HAPs leaving the furnace enter into the fuel. The proportion of elemental HAPs in the gas stream depends primarily on the bottom-ash and/or fly-ash partitioning and adsorption onto ash. For both cases, furnace type appears to influence the HAPs that leave the furnace and continue to a control device or stack. Chapter 10 provides a discussion, from limited data, suggesting that, for example, organic HAP emissions are increased as furnace conditions are changed. Similarly for elemental HAPs, chapter 10 shows data suggesting that, for example, cyclone boilers emit some elemental HAPs at lower rates than tangential boilers, and tangential boilers emit at lower rates than cyclone boilers for other HAPs. Although tentative, furnace type characterizations as related to HAP emissions are used for the modeling described later in this report. Appendix D describes the construction of the models and the manner in which HAP emissions are assigned to each furnace type.

2.3.2 Bottom Types

There are two types of furnace bottoms, wet and dry. The type of bottom used depends on the type of fuel to be burned and

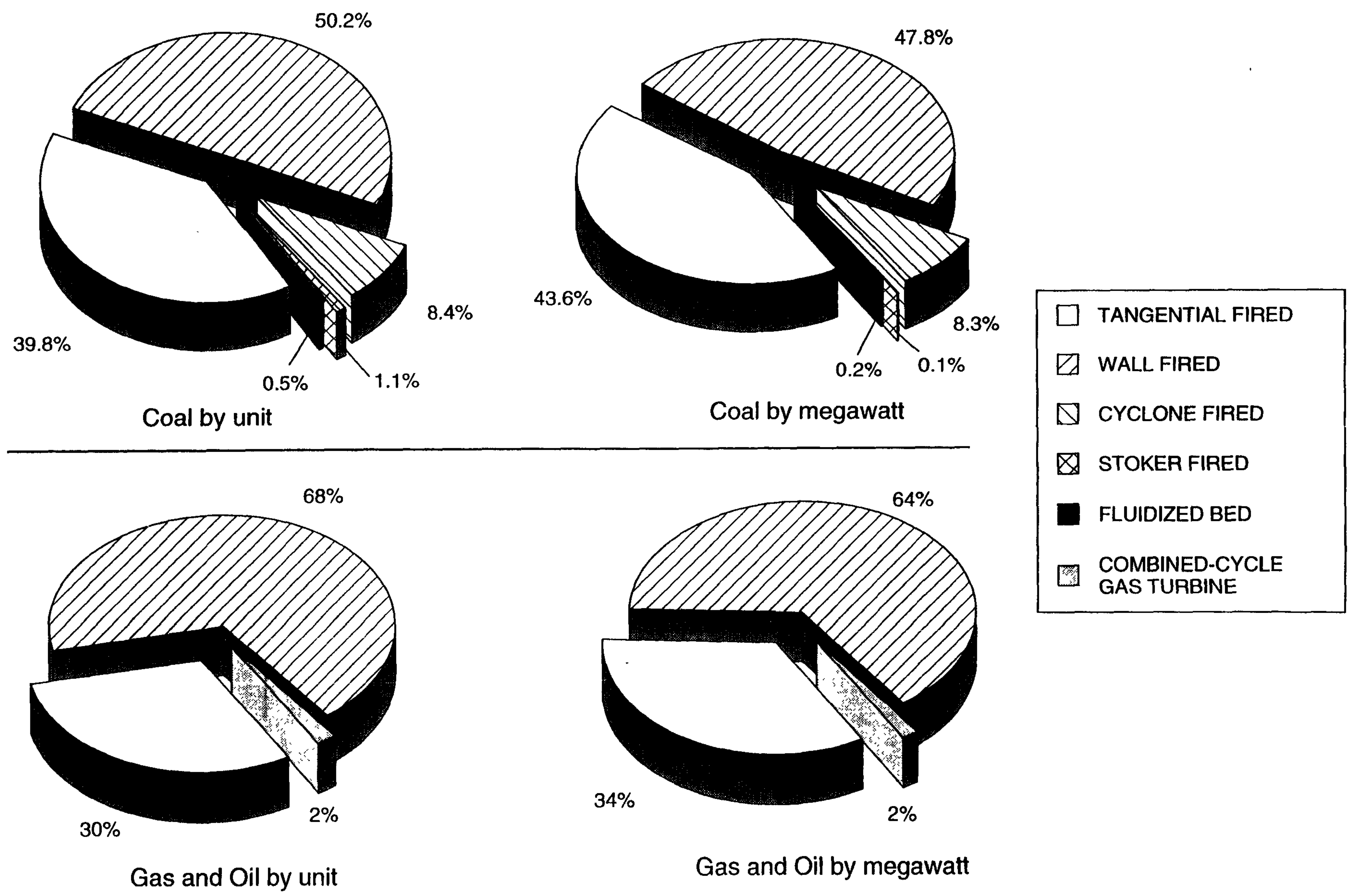


Figure 2-2. Unit types in the utility industry by fuel type in 1990.[1]

on the engineering requirements of the furnace. Wet-bottom furnaces sweep the flame across the furnace floor at all firing rates to maintain the ash in a molten state. Because of the ash handling and temperature limitations of wet-bottom boilers, dry-bottom furnaces are the only type currently used in new furnace construction.

In dry-bottom boilers, the ash reaches the melting point but cools when deposited on the furnace walls; thus, it can be removed in a dry state. This type of bottom is used in furnaces with tilting fuel nozzles. It can provide a wider steam temperature control range and can burn coals with widely varying ash characteristics.[18]

2.3.3 Cogeneration

Units that cogenerate steam and electricity can be classified as topping or bottoming systems. Topping systems produce electricity first, and all or part of their exhaust heat is subsequently used in an industrial process. A bottoming system uses waste heat from a boiler (or other high-temperature thermal process) to run a steam turbine and/or generating unit to produce electricity.[19-21]

2.3.4 Combined-Cycle Systems

The use of one source of hot gas to produce electricity by the means of two separate thermal cycles and associated turbines is known as combined cycle, such as a combustion gas turbine with exhaust gas used to create steam for a steam turbine. Only systems that incorporate a steam turbine as one of the two cycles are considered in this study. Simple-cycle gas turbines with waste gas vented directly to the atmosphere are not considered. Combined-cycle systems consisting of a gas turbine with exhaust gases serving a heat recovery steam generator are considered if they otherwise meet the definition of an electric utility steam generating unit.

2.4 PARTICULATE MATTER CONTROL

This section describes the four major types of PM controls used on utility boilers: mechanical collectors, electrostatic precipitators (ESPs), particle scrubbers, and fabric filters (FFs). Figure 2-3 illustrates the 1990 distribution of PM control by fuel in the utility industry by unit and by total MW.[1]

In 1990, ESPs accounted for the largest portion of the PM control technology used on coal-fired units by number of units (91 percent) and by total megawatts (91 percent). The second

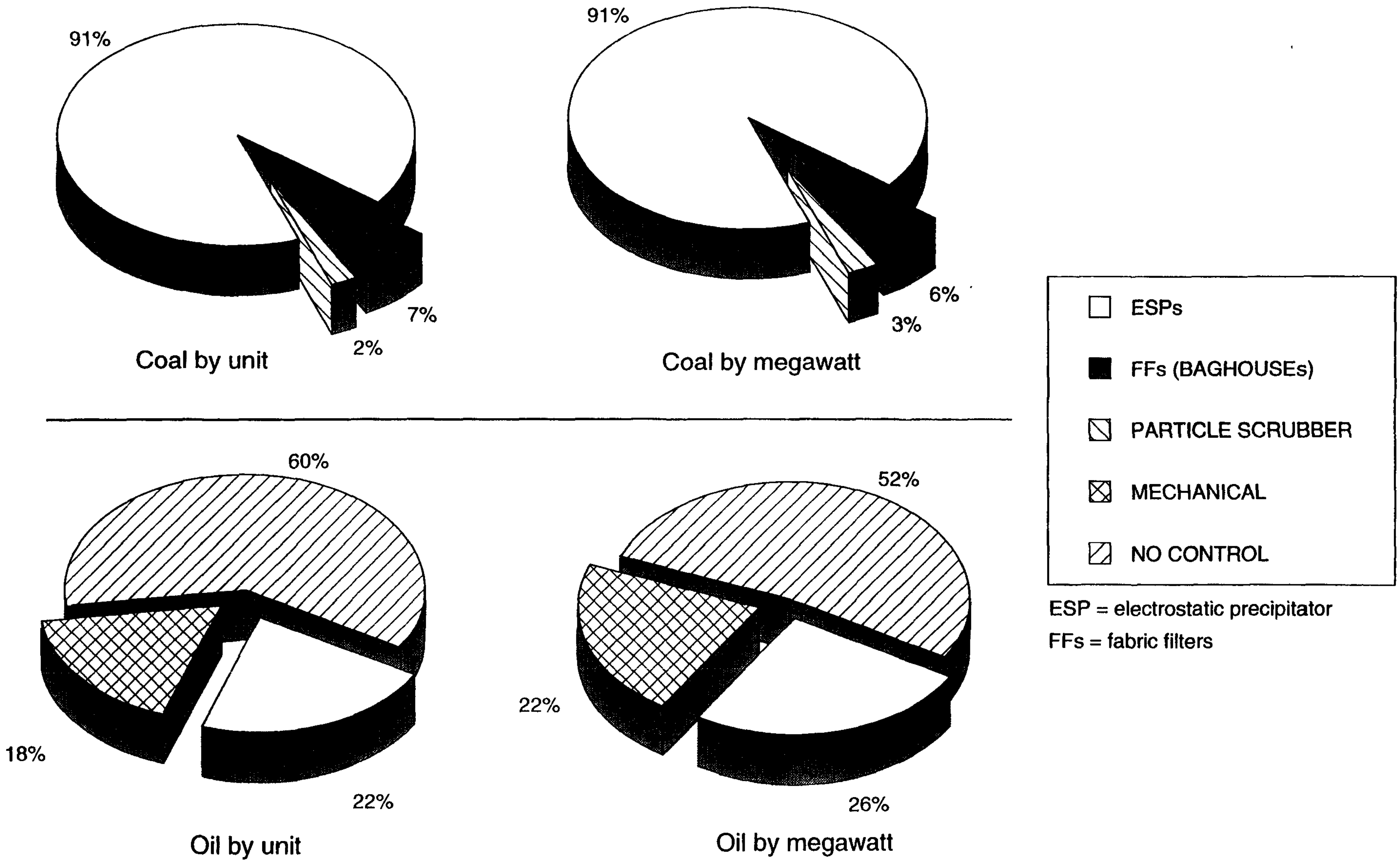


Figure 2-3. Particulate control in the utility industry by fuel type in 1990.[1]

most common control technology was the FF (also referred to as a baghouse). Fabric filters were used on 7 percent of the coal-fired units (6 percent of total megawatts). Particle scrubbers were used on 2 percent of the coal-fired units (approximately 3 percent of the total megawatts).

Uncontrolled units represented (1990) the largest portion of the oil-fired units (60 percent) and accounted for 52 percent of the oil-fired industry's total MWs. In 1990, ESPs were used on 22 percent of the oil-fired units or at 26 percent of the capacity of the oil-fired industry. Mechanical controls (cyclones) were used on 18 percent of the oil-fired units (22 percent of the total MWs). Gas-fired units had no PM controls.[1]

As PM is formed during the combustion process and moves through the boiler system, HAPs can be condensed or adsorbed on particle surfaces. Although most particles are formed in the 3-µm to 50-µm range[22] (on a mass basis), HAPs tend to concentrate preferentially on particles smaller than about 7 µm, and especially on those around 0.3 µm.[23] Because of this preferential concentration, high collection efficiency for fine particles is an important factor in evaluating HAP control from PM collection devices. Each of the four major control devices is described here, along with its method of operation and collection efficiency by particle size. Many of the efficiency data by particle size originate from extensive studies performed by the EPA expressly for the purpose of comparing field performance of FFs, ESPs, and particle scrubbers applied to combustion sources. Special care was taken to provide accurate measurements for particles smaller than about 10 µm.

2.4.1 Mechanical Collectors

Mechanical collectors are the oldest, simplest, and least efficient of the four types of PM control devices. The collectors used for utility boilers are generally in the form of groups of cylinders with conical bottoms (multicyclones). Particles in the entering gas stream are hurled to the outside of the cylinder by centrifugal force and are discharged at the bottom of the cone. Collection efficiency for a typical multicyclone may be about 70 to 75 percent for 10-µm particles, but may drop to less than 20 percent for 1-µm particles.[24] Thus, the multicyclone would be the least effective of the four devices discussed here for reducing HAPs emitted into the atmosphere as small particles or attached to small particles.[25]

2.4.2 Electrostatic Precipitators

Electrostatic precipitators have been used on boilers for about 80 years, can be designed for high efficiencies (>99 percent, but at the cost of increased unit size), and are the most frequently used PM control devices on utility boilers. They operate by imparting an electrical charge to incoming particles, then attracting the particles to oppositely charged plates for collection. The collected particles are periodically dislodged in sheets or agglomerates by rapping the plates. Particle removal in an ESP depends largely on the electrical resistivity of the particles being collected. An optimum value exists for any ash, above and below which particles become less effectively charged and collected. Coal that contains a moderate to high amount of sulfur (more than about 3 percent) produces an easily collected fly ash. Low-sulfur coal produces a high-resistivity fly ash that is difficult to collect. Resistivity of the fly ash can be changed by operating the boiler at a different temperature or by conditioning the particles upstream of the ESP with sulfur trioxide, sulfuric acid, water, sodium, or ammonia. In addition, efficiency is not uniform for all particle sizes. For coal fly ash, particles larger than about 1 to 8 µm and smaller than about 0.3 µm are typically collected with efficiencies from 95 to 99.9 percent.[26] Particles near the 0.3 µm size are in a charging transition region that reduces collection efficiency.[27] These particles have been shown to have lower collection efficiency (about 50 to 95 percent). However, for particles in the 1- to 8-µm size range, the reasons for poorer collection efficiency are not as well understood. There is often a penetration peak in this size range.[28] If these particles escape capture by the ESP, boiler emissions are likely to show an increase in smaller particles that may be enriched with HAPs.[29] As mentioned above, ESPs can be designed to control particulate emissions to high efficiencies. On a total mass basis, these efficiencies can be equivalent to FFs. However, on a fine particulate basis, the ESP may not be quite as effective as an FF. Because designing for higher overall efficiencies in an ESP requires increasing the size (and cost) of the device, past practice has been to design to meet regulatory requirements. Further study is required to determine the capabilities of ESPs for higher overall HAP removal compared to other control systems.

2.4.3 Particulate Matter Scrubbers

The use of wet scrubbers for PM collection has three distinct disadvantages: high energy consumption when high efficiency is required, the presence of a wet effluent to be disposed of, and difficulty in obtaining high collection efficiencies for fine particles. Scrubbers operate by shattering streams of water into small droplets that collide with and trap

PM contained in the flue gas or by forcing the flue gas into intimate contact with water films. The particle-laden droplets or water films coalesce and are collected in a sump at the bottom of the scrubber. The three common types of scrubbers for fly ash control are venturi, preformed spray, and moving bed. Venturi scrubbers, the type most commonly used for utility systems, transport particle-laden flue gas through a constriction at which violent mixing takes place. Water is introduced either at, or upstream of, the constriction. Preformed spray scrubbers are usually vertical cylinders with flue gas passing upward through droplets sprayed from nozzles near the top of the unit. Moving-bed scrubbers have an upper chamber in which a bed of low-density spheres (often plastic) is irrigated by streams of water from above. Gas passing upward through the bed agitates the wetted spheres, which continually expose fresh liquid surfaces for particle transfer. Particle collection efficiency in scrubbers is generally size and energy dependent. Although some scrubbers collect particles at high efficiency with low energy consumption, venturi scrubbers are normally energy intensive compared to ESPs or FFs. Particles larger than a few micrometers can be collected with efficiencies greater than 99 percent, but, at sizes smaller than about 1 or 2 μm, efficiency may be reduced to less than 50 percent.[30] Because of this low collection efficiency, the emission of HAP-laden particles from scrubbers is expected to be greater than for ESPs. However, water in the scrubber may remove water-soluble HAPs.[31]

2.4.4 Fabric Filters

Fabric filters have been used on utility boilers for about 20 years. They are inherently efficient and are effective when high-efficiency PM collection is required. Unlike ESPs, their size is not a strong function of desired efficiency. They must be designed and operated carefully to ensure that the fabric tubes (bags) inside the collector are not damaged or destroyed by adverse operating conditions. Fabric filters collect PM by placing a fabric barrier in the flue gas path. Gas passes freely through the fabric, but particles are trapped and retained for periodic removal. Data from a small utility boiler show collection efficiencies not lower than 99.6 percent across all particle sizes from 0.3 μm to about 10 μm (the range of the measuring equipment).[32] Because of its high collection efficiency for small particles, the baghouse should be particularly effective for removing particles that have been enriched with HAPs.[33,34] However, further study is required to determine if baghouses can remove significantly greater quantities of HAPs than are removed by other control systems.

2.4.5 *Comparison of Particle Collectors*

Table 2-1 compares the characteristics and capabilities of the four particle collection devices described. Fabric filters and ESPs appear to provide the highest mass collection efficiency for fly ash. Fabric filters appear to be the best of the four devices for capturing small particles that may be enriched with HAPs. Examination of Tables 10-6 and 10-9, which compare HAP removal by cold-side ESPs and FFs on utility boilers, also suggests that FFs may be more effective. However, further study is required for confirmation.

2.5 SULFUR DIOXIDE CONTROL

Sulfur dioxide emissions are controlled through either (1) precombustion measures, namely, the combustion of fuels that contain lesser amounts of sulfur; (2) combustion measures, such as an FBC system that combusts coal and limestone (or an inert material); and (3) postcombustion measures, such as the use of flue gas scrubbing devices.

Precombustion measures may include the use of compliance fuels (fuels having characteristics, such as low sulfur content, that allow the user to comply with emission limitations solely by switching to the fuel) to meet State implementation plans (SIPs) or NSPS. The use of SIP or NSPS fuels means that the sulfur content in the fuel is sufficiently low that add-on controls or postcombustion controls are not required. As of 1990, all oil- and gas-fired units burn compliance fuel, while approximately 85 percent of the coal-fired units burn compliance coal.[1]

Compliance coal may be obtained through the mining of lower-sulfur coals, coal washing, and/or coal blending. (Because coal washing is reviewed in section 2.5.1, it is only briefly mentioned here.) Most bituminous coals are cleaned in order to meet customer specifications on sulfur, ash, and heating content. In the process of cleaning, the sulfur and ash content of the coal are reduced, while the heating content may be increased. Consequently, less of the cleaned coal, containing less sulfur, is needed to achieve a given heating rate. Compliance coal may also be obtained through coal blending, in which higher-sulfur coals are blended with lower-sulfur coals.

Combustion measures control emissions of SO_2 from five coal-fired units, representing a total capacity of 610 MWe. These units are FBCs and control SO_2 in the combustion zone by using limestone as a sorbent.

Table 2-1. Comparison of Particulate Matter Collection Systems[35]

Collector	Typical mass efficiency, %	Efficiency at 0.3 μm, %	Energy consumption, in. H_2O	
			Collector	System
Multicyclone	70 - 90	0 - 15	4 - 10	7 - 13
ESPs	99 - 99.7	80 - 95	0.5 - 1	3.5 - 4
Particle scrubber	95 - 99	30 - 85	2 - 70	5 - 73
FFs	99 - 99.9	99 - 99.8	5 - 10	8 - 13

ESPs = Electrostatic precipitators.
FFs = Fabric filters.

Because of the 1990 amendments, oil- and natural-gas-fired units now burn compliance fuels that combust with lower SO_2 emissions. Using compliance fuels allows the units to avoid postcombustion scrubbing.[1] However, approximately 15 percent of the coal-fired units use postcombustion control of SO_2 emissions.[1] The rest of this section describes precombustion techniques and postcombustion SO_2 control devices, namely scrubbers, that are used in the coal-fired utility industry.

Figure 2-4 shows SO_2 control devices used in coal-fired utilities in 1990 based on the number of units and total MW capacity.[1] As shown in Figure 2-4, 15 percent of the units, representing about 22 percent of the coal-fired generation capacity, used postcombustion flue gas scrubbing to comply with SO_2 regulations. A wet scrubber was used at approximately 14 percent of the units (approximately 21 percent of the coal-fired total electric capacity), while a spray dryer adsorber/FF (SDA/FF, also called a dry scrubber) system was used at approximately 1 percent of the coal-fired units (approximately 1 percent of the coal-fired total electric capacity).

Sulfur dioxide emission standards for utility steam generators vary according to the size, age, and location of a facility. Existing boilers are regulated by SIPs. Plants built after 1971 are subject to NSPS SO_2 emission limits of 1.2 lb SO_2/MMBtu. Plants built after 1978 are additionally required to reduce their SO_2 emissions by 70 to 90 percent.

The extent of postcombustion SO_2 control used by the utility industry will increase in response to Title IV of the 1990 amendments to the Act, which require SO_2 reduction in two phases. The likely mix of SO_2 control approaches that will be used to

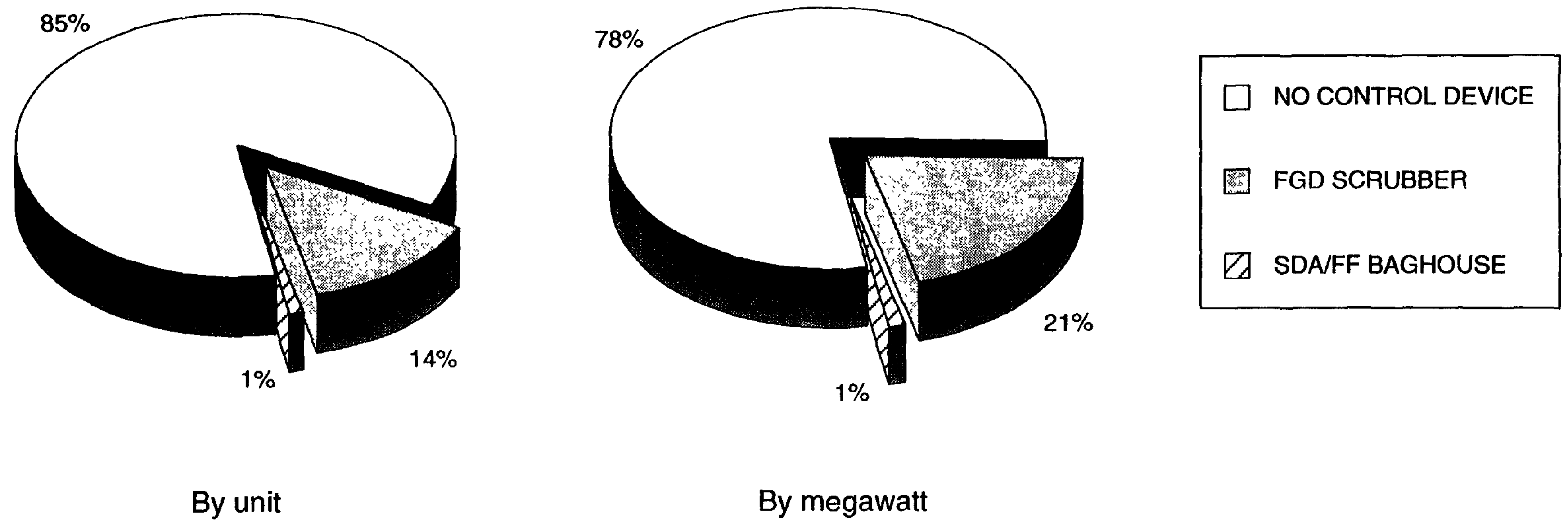


Figure 2-4. SO_2 control in the utility industry in 1990 (coal-fired boilers only).[1]

comply with the Phase I and Phase II requirements is discussed in section 2.7.

2.5.1 Precombustion Control: Fuel Options

By using coal with an appropriately low sulfur content, 85 percent of the coal-based utility units currently comply with SO_2. Compliance coals may be mined from the ground or may be obtained by cleaning or blending mined coal.

Physical coal cleaning typically involves (1) size reduction and screening, (2) gravity separation of coal from sulfur-bearing mineral impurities, and (3) dewatering and drying.[36] Approximately 77 percent of the eastern and midwestern bituminous coal shipments are subjected to some physical cleaning process.[37] Subbituminous and lignite coals are not routinely cleaned.[38,39] The primary purpose of physical cleaning has been to remove ash; coal cleaning has the consequence of increasing the heating value of the coal and reducing the sulfur content in the coal.[40] Bituminous coals from the eastern U.S., cleaned with a 1.6 specific gravity separation, were found to provide reductions of 48 percent ash, 65 percent pyritic sulfur, 43 percent total sulfur, and 48 percent SO_2 emissions at a Btu recovery rate of 94 percent.[41]

As with sulfur, many trace elements may be both organically bound and present as a part of a mineral in the same coal. Thus, physical coal cleaning has the potential to remove some of the trace elements associated with the mineral matter. Recent experimental studies showed significant reductions of a number of trace elements.[37,42] The reduction percentages were found to depend on the type of coal and the trace element's nature within the coal. For a few trace elements, an enrichment effect was observed for some of the coal samples; however, when expressed on a Btu basis, physical cleaning will always reduce, to some extent, the amount of trace elements present in coal. The effectiveness of coal cleaning in reducing concentrations of trace elements in coal is discussed in section 10.1.2.

2.5.2 Postcombustion Control: Flue Gas Scrubbing for SO_2 Control

According to the 1992 compilation of the Edison Electric Institute's (EEI) Power Statistics database (examining 1990 data), scrubbers were installed on 152 boiler units (out of about 1,043 coal-fired units in the U.S.) with a total rated capacity of 68,695 MWe.[1] Table 2-2 lists the different types of scrubbing installations used in U.S. utility power plants. As shown in Table 2-2, wet limestone/lime slurry scrubbing represents the

Table 2-2. Distribution of SO_2 Control Technologies in 1990[1]

	No. of boiler units	Installed FGD capacity, MWe	Total percent of installed FGD capacity, %
Wet limestone	69	34,521	50.3
Wet lime	45	19,977	29.0
Dry lime/SDA	15	5,626	8.2
Sodium carbonate	9	3,181	4.6
Dual-alkali	6	2,267	3.3
Wellman-Lord	4	1,779	2.6
Mag-Ox	3	895	1.3
Dry aqueous carbonate	1	450	0.7
Total			100.0

FGD = Fluidized gas desulfurization.
SDA = Spray dryer adsorber.

most prevalent scrubber type with almost 80 percent of the total flue gas scrubbing capacity.[1]

2.5.2.1 Wet Limestone. In a wet limestone scrubber, flue gas containing SO_2 is brought into contact with a limestone-water slurry. The SO_2 is absorbed into the slurry and reacts with limestone to form an insoluble sludge. The sludge, mostly calcium sulfite hemihydrate and gypsum, is usually disposed of in a pond.[43]

The two common absorber designs include fixed packing and horizontal or vertical spray towers, with spray towers being the most prevalent. The absorber must be constructed of materials that resist corrosion, erosion, and scaling. To reduce corrosion and erosion problems, a scrubber is located downstream of a PM collection device. A flue gas cooler and humidifier are used to cool the flue gases, generally to 50° C (122° F), prior to absorption. The size and number of scrubber modules are directly related to boiler size, load fluctuations, and system availability and compliance requirements.

Auxiliary equipment includes a demister to remove entrained droplets from the scrubber outlet gas, a heat exchanger system to reheat the outlet gas prior to exhaust, a slurry preparation system, and a disposal system for the large quantities of sludge produced. Sludge disposal needs can be very site specific and depend upon the local climate and soil conditions.[44]

The basic wet limestone scrubbing process is simple and well established. Limestone sorbent is cheap and generally locally available in the U.S. The SO_2 removal efficiencies of existing wet limestone scrubbers range from 52 to 95 percent, with an average of 85 percent.[1] Operating parameters affecting SO_2 removal efficiency include liquid-to-gas ratio, pH of the scrubbing medium, and the ratio of calcium sorbent to SO_2. Periodic maintenance is needed because of scaling, erosion, and plugging problems.

Recent advancements include the use of additives or design changes to promote SO_2 absorption or to reduce scaling and precipitation problems. Gypsum can now be recovered as a salable byproduct. Extensive operating experience has increased industry confidence in designing larger, more reliable limestone scrubber modules. In 1990, wet limestone scrubbers were used at 69 units, or at 34,521 MWe of the total scrubbing capacity.

2.5.2.2 Wet Lime. In a wet lime scrubber, flue gas containing SO_2 is contacted with hydrated lime-water slurry; the SO_2 is absorbed into the slurry and reacts with hydrated lime to form an insoluble sludge. The hydrated lime provides greater alkalinity (higher pH) and reactivity than limestone.[45]

Wet lime scrubbing is a proven technology; considerable operating experience has been gained in 45 utility units.[1] These units represented 19,977 MWe of the total scrubbing capacity in 1990. The SO_2 removal efficiencies of existing wet lime scrubbers range from 72 to 99 percent. Recent advances include the use of additives to improve performance, reduce scaling problems, and produce a salable gypsum byproduct. Lime scrubbing processes require consideration of appropriate disposal of large quantities of waste sludge.

2.5.2.3 Dry Lime/Spray Dryer Adsorber. This process produces dry reaction waste products for easy disposal. In this process, flue gas at air preheater outlet temperatures of 121° to 177° C (250° to 350° F) is contacted with fine spray droplets of hydrated lime slurry in a spray dryer vessel. The SO_2 is absorbed in the slurry and reacts with the hydrated lime reagent to form solid calcium sulfite and calcium sulfate as in a wet lime scrubber.[46] The water is evaporated by the heat of the flue gas. The dried solids are entrained in the flue gas, along with fly ash, and are collected in a PM collection device. Most of the SO_2 removal occurs in the spray dryer vessel itself, although some additional SO_2 capture has also been observed in downstream PM collection devices, especially baghouses.

The primary operating parameters affecting SO_2 removal are the calcium-reagent-to-sulfur stoichiometric ratio and the approach to saturation in the spray dryer. To increase overall sorbent utilization, the solids collected in the spray dryer and the PM collection device may be recycled. The SO_2 removal efficiencies of the existing lime spray dryer systems range from 60 to 90 percent.[1] Spray dryers were used at 15 units and constituted 5,626 MWe of scrubbing capacity in 1990.

2.5.2.4 Wet Sodium Carbonate. Flue gas scrubbing with sodium carbonate solution minimizes the operation and maintenance problems related to lime and/or limestone slurry scrubbers. However, the process uses a reagent that is relatively expensive unless it can be found as a byproduct from another process or as a locally mined material (trona). There were nine units (in 1990) using wet sodium carbonate scrubbing in the U.S., representing 3,181 MWe of the total scrubbing capacity.[1] Waste products of this process include sodium sulfite and sodium sulfate.

Due to the higher solubility and greater reactivity of the sodium carbonate compared to lime and/or limestone, a smaller size scrubber can be used. The primary operating parameters are liquid-to-gas ratio and the reagent stoichiometric ratios used. Sorbent utilizations are high. The SO_2 removal efficiencies reported for this process range from 53 to 91 percent.[1] The soluble reaction products must be treated before disposal. The treated flue gas is demisted and reheated before exhausting through a stack.

2.5.2.5 Dual Alkali. A dual alkali system combines the operational advantage of a sodium-based solution scrubbing system with the economic advantage of a lime and/or limestone-based system. As practiced in the U.S., a dual (or double) alkali system uses a sodium sulfite solution to absorb SO_2 from flue gas and to form sodium bisulfite. The spent sorbent is reacted with lime to precipitate calcium sulfite and to regenerate the active sodium sulfite sorbent.[47] The precipitated calcium salts are separated and dewatered for disposal. The treated flue gas is demisted and reheated before it is exhausted through a stack.

The dual alkali process has been installed (1990) on six boiler units in the U.S. with a combined capacity of 2,267 MWe.

The SO_2 removal efficiencies at these plants range from 89 to 95 percent.[1] This process also requires appropriate disposal of large quantities of waste calcium salts. Recent advances in this process include forced oxidation of calcium sulfite to a salable gypsum byproduct, which reduces the waste disposal load.

2.5.2.6 Wellman-Lord. In the Wellman-Lord process, SO_2 from the flue gas is absorbed in a sodium sulfite solution to form sodium bisulfite as in the dual-alkali process. The spent sorbent is, however, thermally regenerated by reversing the absorption reaction. Regenerated sodium sulfite crystals are dissolved and returned to the absorber. The concentrated, stripped SO_2 stream is converted to salable sulfuric acid, elemental sulfur, or liquid SO_2.[48] The treated flue gas is demisted and reheated before it is exhausted through a stack. The Wellman-Lord process has been installed on four U.S. boiler units with a combined capacity of 1,779 MWe (1990), with SO_2 removal efficiencies ranging from 85 to 90 percent.[1]

2.5.2.7 Magnesium Oxide. Similar to Wellman-Lord, the magnesium oxide (MAG-OX) fluidized gas desulfurization (FGD) process is regenerable. The SO_2 in the flue gas is absorbed by a magnesium oxide slurry, and the resulting magnesium sulfite is calcined to regenerate magnesium oxide that is slurried and recycled back to the absorber. The SO_2-rich gas produced in the regeneration step is processed further to produce a salable product such as sulfuric acid or elemental sulfur.[49]

Because of the high-temperature regeneration step at 800° to 1,000° C (1,472° to 1,832° F), energy requirements for this process are high. However, due to the regenerative nature of the process, reagent and disposal costs are small. Scrubber plugging and scaling problems are reduced compared to a limestone scrubbing system. The corrosion and/or erosion problems related to a slurry operation are still significant. The magnesium oxide process has been installed on three boiler units in the U.S. with a combined capacity of 895 MWe (1990). The SO_2 removal efficiencies at these plants range from 92 to 95 percent.[1]

2.5.2.8 Dry Aqueous Carbonate. In the dry aqueous carbonate process, the flue gas is contacted with an aqueous sodium carbonate solution in a spray dryer. The sodium carbonate reacts with and removes SO_2 from the flue gases, then the solution is evaporated to dryness by the hot flue gases. The dry reaction products, consisting of sodium sulfite, sodium sulfate, and unreacted sodium carbonate, are removed from the flue gases by passage through multicyclones and an ESP. Subsequent processing of the reaction products with crushed coal yields regenerated sodium carbonate and hydrogen sulfide gas. The sodium carbonate is recycled to the spray absorber, and hydrogen sulfide gas is converted to salable sulfur.[50] Only one unit, of 450 MWe capacity, uses the dry aqueous carbonate system for FGD (1990); it has a 70 percent SO_2 removal efficiency.[1]

2.6 NO_x CONTROL

This section provides a brief review of the formation and control of NO_x emissions, as well as the general types of NO_x control used in the utility industry. Detailed information on the formation and control of NO_x can be found in four major technical documents.[51-54]

Figure 2-5 shows NO_x control approaches used in 1990 based on the number of units and total MW capacity.[1] Around 81 percent of coal-fired plants, representing about 67 percent of the coal-fired MW capacity, had no NO_x control, while around 19 percent of the units, representing about 33 percent of the coal-fired MW capacity, used some kind of NO_x control. Approximately 73 percent of the gas- and oil-fired units, with about 62 percent of the MW capacity, did not use NO_x control, while approximately 27 percent of the units, representing about 38 percent of the gas- and oil-fired MW capacity, used some kind of NO_x control.

The chemical species nitrogen dioxide (NO_2) and nitrogen oxide (NO) are collectively called NO_x. In general, NO_x from combustion consists of about 95 percent NO and 5 percent NO_2; however, NO_x is reported as NO_2.[55] Nitrogen oxides are primarily formed during fossil fuel combustion in one of two ways: (1) oxidation of nitrogen in the combustion air to give thermal NO_x, or (2) oxidation of nitrogen contained in the fuel to give fuel NO_x. There is a third form of NO_x, namely prompt NO_x, that is formed by the reaction of nitrogen and hydrocarbons in the fuel, but prompt NO_x has a lifetime of several microseconds.[56] Thermal NO_x is the predominant form during the combustion of fuels that contain relatively little fuel-bound nitrogen (such as natural gas and distillate oil). Both thermal and fuel NO_x are formed during the combustion of fuels that contain fuel-bound nitrogen (such as residual oil and coal).[57] Fuel switching, then, may yield reduced NO_x emissions.

The formation of NO_x for coal-fired units depends on factors such as the type of boiler, type of burner, and facility operation.[58] Any of these factors that increase temperature or residence time at high temperature will promote NO_x formation.[59] In general, cyclone and other wet-bottom boilers have relatively higher NO_x emissions, with an approximate range of 1 to 2

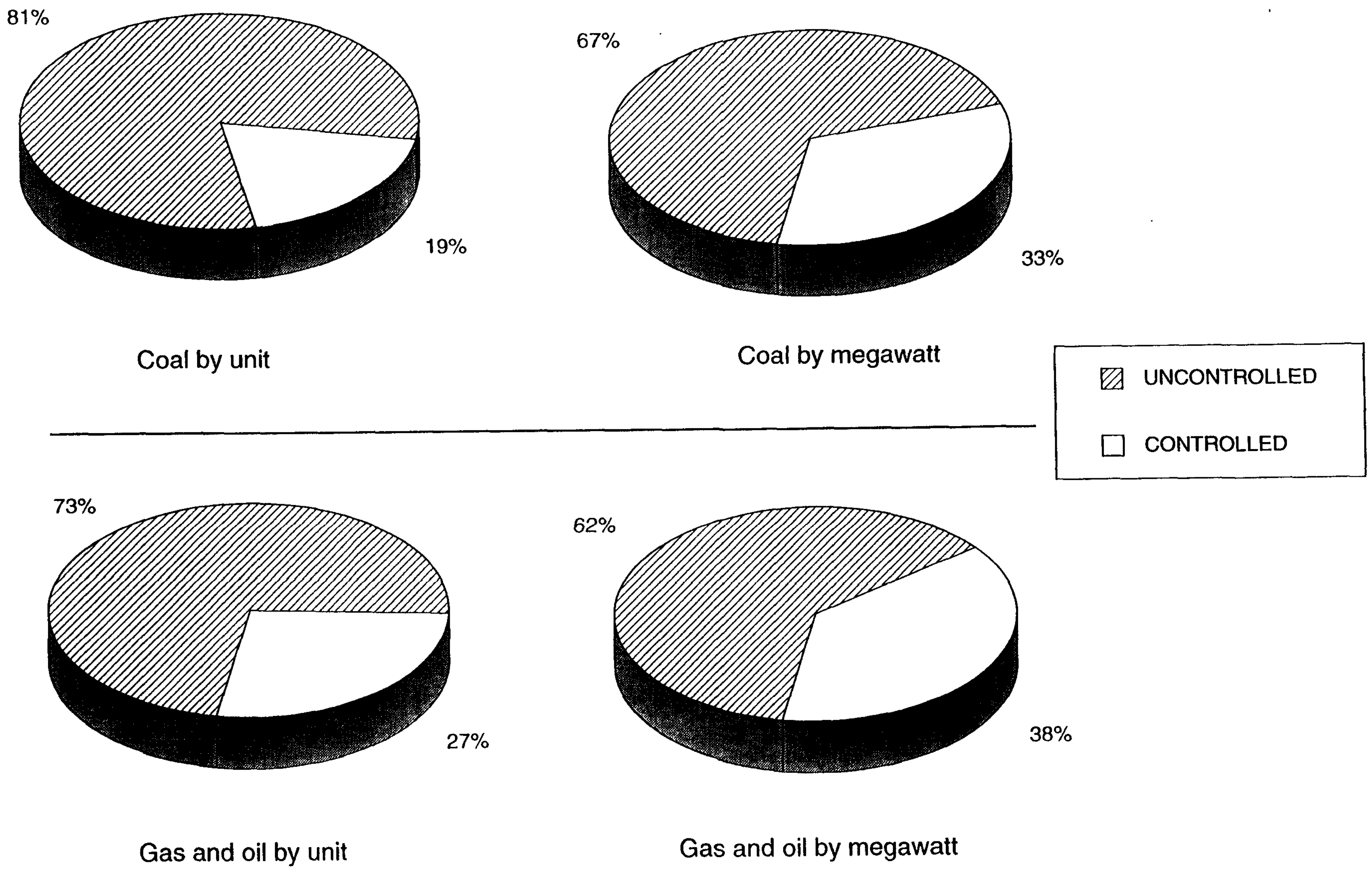


Figure 2-5. Nitrogen oxide control in the utility industry by fuel type in 1990.[1]

lb/MMBtu, than do dry-bottom boilers, which have an approximate range of 0.4 to 1.5 lb/MMBtu.[60] With regard to the type of burner, wall-fired wet-bottom boilers have relatively higher NO_x emissions with an approximate range of 1.6 to 2 lb/10^6 Btu, wall-fired dry-bottom boilers have moderate NO_x emissions with an approximate range of 0.5 to 1.45 lb/MMBtu, and tangential-fired dry-bottom boilers have relatively lower NO_x emissions at approximately 0.4 to 0.9 lb/MMBtu.[61] Because of their low combustion temperatures, an FBC's thermal NO_x is essentially zero. Design features such as staged combustion can significantly reduce fuel NO_x, leading to low NO_x emissions.[62]

The reduction of NO_x emissions is important for controlling acid rain and ozone formation.[63] Techniques used to reduce NO_x formation include those for combustion and postcombustion control. Combustion control techniques regulate the amount of combustion air and may also control the flame temperature at different stages of the combustion process; postcombustion control involves the removal of NO_x from the flue gas.[64] More than one form of combustion control may be used for a given unit.

2.6.1 Combustion Control

Control can be achieved through staged combustion (also called air staging). With staged combustion, the primary combustion zone is fired with most of the air needed for complete combustion. The remaining air needed is introduced into the products of the incomplete combustion in a second combustion zone. Air staging lowers the peak flame temperature, thereby reducing thermal NO_x, and reduces the production of fuel NO_x by reducing the oxygen available for combination with the fuel nitrogen.[65] Staged combustion may be achieved through low NO_x burners, overfire air (OFA), off-stoichiometric firing (OSF), selective or biased burner firing (BBF), and burners-out-of-service (BOOS).[66] Each of these methods requires modifying equipment or operating conditions so that a fuel-rich condition exists near the burners. In cyclone boilers, combustion occurs with a molten ash layer and the combustion gases flow to the main furnace; this design precludes the use of low NO_x burners and air staging.[67]

Low NO_x burners may be used in coal-, oil-, and gas-fired boilers to lower NO_x emissions by about 25 to 55 percent.[68] Overfire air may be used as a single NO_x control technique, with NO_x reductions of 15 to 50 percent.[69,70] When OFA is combined with low NO_x burners, reductions of up to 60 percent may result.[71] The actual NO_x reduction achieved with a given control technique may vary from site to site.[72]

Just as the combustion air to the primary combustion zone may be reduced, part of the fuel may be diverted to create a secondary flame with fuel-rich conditions downstream of the primary combustion zone. This combustion technique is termed reburn and involves injecting 10 to 20 percent of the fuel after the primary combustion zone and completing the combustion with OFA.[73] The fuel injected downstream is not necessarily the same as that used in the preliminary combustion zone. In most applications of reburn, the primary fuel is coal and the reburn fuel is natural gas. Natural gas reburn has been successfully demonstrated in several field tests in the United States and abroad.[74,75] Reburn with other fuels, primarily coal, is currently under development, as are improvements in the process.[76]

Other ways to reduce NO_x formation by reducing peak flame temperature include using flue gas recirculation (FGR), reducing amounts of OFA, injecting steam or water into the primary combustion zone, and increasing spacing between burners.[77] By returning part of the flue gas to the primary combustion zone (FGR), the flame temperature and the concentration of oxygen in the primary combustion zone are reduced. Flue gas recirculation is usually used with natural gas and distillate oil combustion. The peak temperature may also be reduced in natural gas and distillate fuel oil combustion units by reducing the amount of combustion air that is preheated; however, the unit efficiency will also be reduced.

To cool combustion gases, steam or water may be injected with the fuel, with the air, or directly into the combustion zone. This technique is used for gas-fired turbines due to the relatively low efficiency penalty (typically 1 percent).[78] However, this technique is not used for utility boilers because it has a high efficiency penalty (about 10 percent).[78] As shown in Table 2-3, approximately 36 percent of the combined-cycle turbine units used steam or water injection for NO_x control in 1990, while only approximately 2 percent of the boilers reported using this technique. Temperatures may also be reduced in the primary combustion zone by increasing the spacing between burners for greater heat transfer to heat-absorbing surfaces.[79] Another combustion control technique involves reducing the boiler load. In this case, the formation of thermal NO_x generally decreases directly with decreases in heat release rate; however, reducing the load may cause poor air and fuel mixing and increase carbon monoxide and soot emissions.[80]

2.6.2 Postcombustion Control

Postcombustion control involves the removal of NO_x from the flue gas downstream of the combustion zone and is achieved either

Table 2-3. Distribution of NO_x Control by Fuel Burned, by Unit, in 1990 [1,a]

	Percent NO_x control			
Fuel	None	Staged combustion	Boiler design	Staged combustion with injection of water or steam
Coal, Boiler bottom				
Dry	79 (64)	18 (32)	3 (4)	--
Wet	97 (97)	3 (3)	--	--
Oil	73 (66)	26 (33)	--	--
Gas	73 (60)	25 (38)	< 1	2 (2)
Combined-cycle turbine	64 (63)	--	--	36 (37)[b]

[a] Values listed in parentheses are the percent distribution by MWe for each type of fuel.
[b] Only steam or water injection.

by reducing NO_x emissions only (selective noncatalytic reduction [SNCR]) or by reducing combined emissions of CO, hydrocarbons, and NO_x (selective catalytic reduction [SCR]).[81] Postcombustion control has, to date, seen limited use in new coal-fired units with the application concentrated in California, where SCR is used at cogeneration plants and with gas-fired turbines and where SNCR is used at FBCs, two pulverized coal-fired units, and a gas-fired unit boiler.[82] Selective catalytic reduction has also been implemented at an IPP Cogen plant in New Jersey.[3]

With SCR, ammonia or another reducing agent is diluted with air or steam, and the mixture is injected into the flue gas upstream of a vanadium, titanium, platinum, or zeolite catalyst bed. The NO_x is reduced to molecular nitrogen on the catalyst surface.[83] Selective catalytic reduction units provide up to 70 to 90 percent NO_x reduction[84] and are usually located between the economizer outlet and air heater flue-gas inlet, where temperatures are 230° to 400° C (450° to 750° F).[85]

Selective noncatalytic reduction is currently achieved commercially in one of two ways: THERMAL $DeNO_x$®, an Exxon process, or NO_xOUT®, an EPRI process. THERMAL $DeNO_x$® reduces NO_x to nitrogen through injection of ammonia into the air-rich flue gas. NO_xOUT® achieves NO_x reduction by injecting urea into the oxygen-rich and/or high-temperature convection part of the boiler.[86]

The necessity of using nitrogen-based reagents requires SCR and SNCR systems to closely monitor and control the rate of reagent injection. If injection rates are too high, NO_x emissions may increase (in SNCR systems), and stack emissions of ammonia may also occur in concentrations of 10 to 50 ppm. A portion (usually around 5 percent) of the NO reduction by SNCR systems is due to transformation of NO to N_2O, which is a global warming gas.

Table 2-3 presents a general breakdown of utility industry NO_x control usage according to the 1990 EEI power statistics database.[1] As shown in Table 2-3, most of the utility industry has no NO_x control; 79 percent of the dry-bottom coal-fired boiler units, 97 percent of the wet-bottom coal-fired boiler units, 73 percent of the oil- and gas-fired boiler units, and 64 percent of the combined-cycle turbine units had no NO_x control in 1990. Units that had NO_x control equipment used various types of staged combustion techniques, including low-NO_x burners, OFA, OSF, BBF, and BOOS. Staged combustion control was used in 18 percent of the dry-bottom coal-fired units, 3 percent of the wet-bottom coal-fired units, 26 percent of the oil-fired units, and 25 percent of the gas-fired boiler units. As previously noted, steam or water injection was used for NO_x control in approximately 36 percent of the combined cycle units. Table 2-3 also shows that approximately 3 percent of the dry-bottom coal-fired units had boiler design as a NO_x control method.

2.7 UTILITY INDUSTRY AFTER IMPLEMENTATION OF 1990 AMENDMENTS

This section describes the changes in the utility industry expected during the 1990-2010 time frame. The effect of planned generation capacity growth on the fuel use and technologies that will be used for steam and power generation is discussed in section 2.7.1. Title IV of the 1990 amendments requires the utility industry to reduce SO_2 emissions in two phases. The effect of SO_2 control measures likely to be used to comply with the Phase I and Phase II requirements on the overall mix of utility SO_2 control technology is discussed in section 2.7.2. For the purposes of this analysis, the projected compliance date for Phase II was determined to be 2010. This year was chosen after discussions with Agency, nonagency, and industry sources concerning possible delays written into the Acid Rain Provision of the Act.

Title IV also contains other provisions that will affect utility responses to regulations. These revisions include topics such as permitting, monitoring, enforcement, re-powering, and penalties. Although these provisions affect the manner in which the utility industry will respond to regulations, they are generally subsidiary to emissions estimates based on fuel usage. These provisions are not discussed further here.

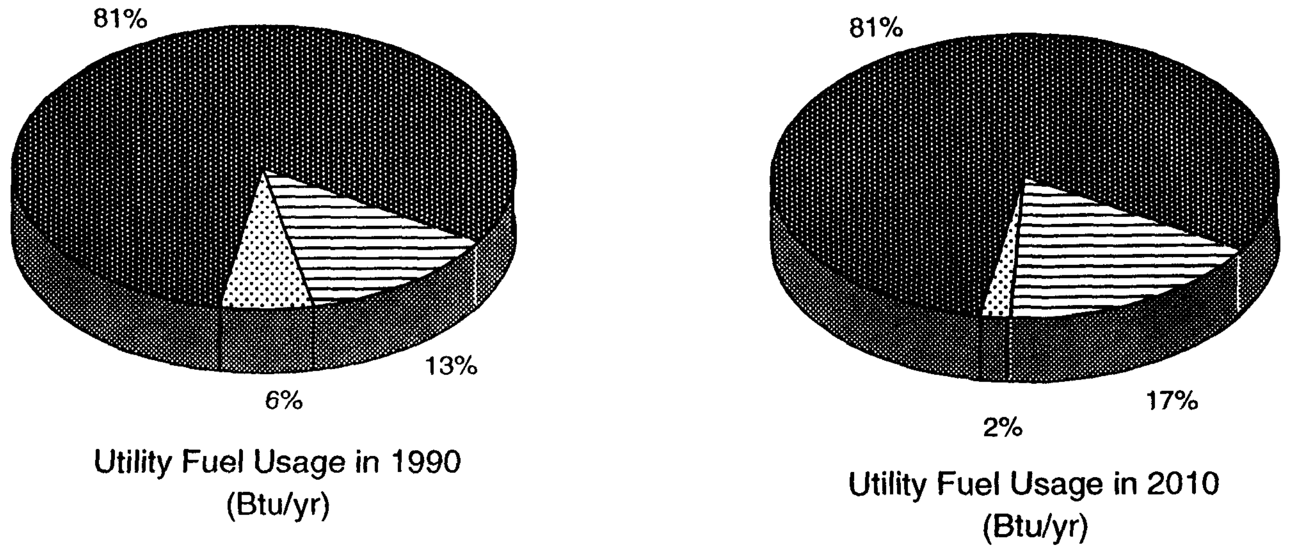
2.7.1 Industry Growth

The publicly owned utility companies, Federal power agencies, rural electric cooperatives, and investor-owned utility generating companies are projected to increase their new generating capacity in service or scheduled for service in the 1990-2010 time frame by 750 billion kWh, from 1,940 to 2,690 billion kWh.[87] These and other projections for utility industry configuration and growth were taken from a study titled "Economic Analysis of The Title IV Requirements of The 1990 Clean Air Act Amendments" produced for the Acid Rain Division of the EPA's Office of Air and Radiation by ICF Resources Incorporated. This single projection was used by the Office of Air Quality Planning and Standards (OAQPS) to maintain consistency with the Acid Rain Division. This projection may be compared with others for future analyses.

Figure 2-6 compares utility fossil fuel consumption, on a Btu basis, for 1990 and projected use for 2010 (publicly owned utility companies, rural electric cooperatives, investor-owned utility generating companies only). On this basis, the predominant fuel both in 1990 and projected for 2010 is coal, at approximately 81 percent of the total industry fuel usage. Oil and gas consumption in 1990 were, respectively, 6 and 13 percent of the total industry fuel usage on a Btu/yr basis. For 2010, oil consumption was projected to decrease to 2 percent, and gas consumption was projected to increase to 17 percent on a Btu/yr basis for the total industry fuel usage.[88] Projected coal consumption in 2010 is expected to be the same percentage of the total utility fossil fuel usage as in 1990 (81 percent).

Figure 2-7 shows the projected growth of each utility fuel between 1990 and 2010.[88] Between 1990 and 2010, the consumption of coal, oil, and natural gas is projected to increase by 32 percent, decrease by 50 percent, and increase by 59 percent, respectively.

The projected increase or decrease in nationwide fuel consumption noted above has been apportioned to only those units projected to be in existence in 2010. The actual increased consumption (coal and oil) would, in most cases, be distributed among new units (existing units not being able to increase their capacity factors to account for the majority of the growth). These new units could be of various sizes and be located at new or existing sites. However, since the Agency cannot estimate the



1990

Fuel	Btu / yr
Coal	1.7×10^{16}
Oil	1.2×10^{15}
Gas	2.8×10^{15}

▩ COAL ▤ GAS ⬚ OIL

2010

Fuel	Btu / yr
Coal	2.2×10^{16}
Oil	6.2×10^{14}
Gas	4.5×10^{15}

Figure 2-6. Fuel use in the utility industry by fuel type in 1990 and projections for the year 2010.[1, 88]

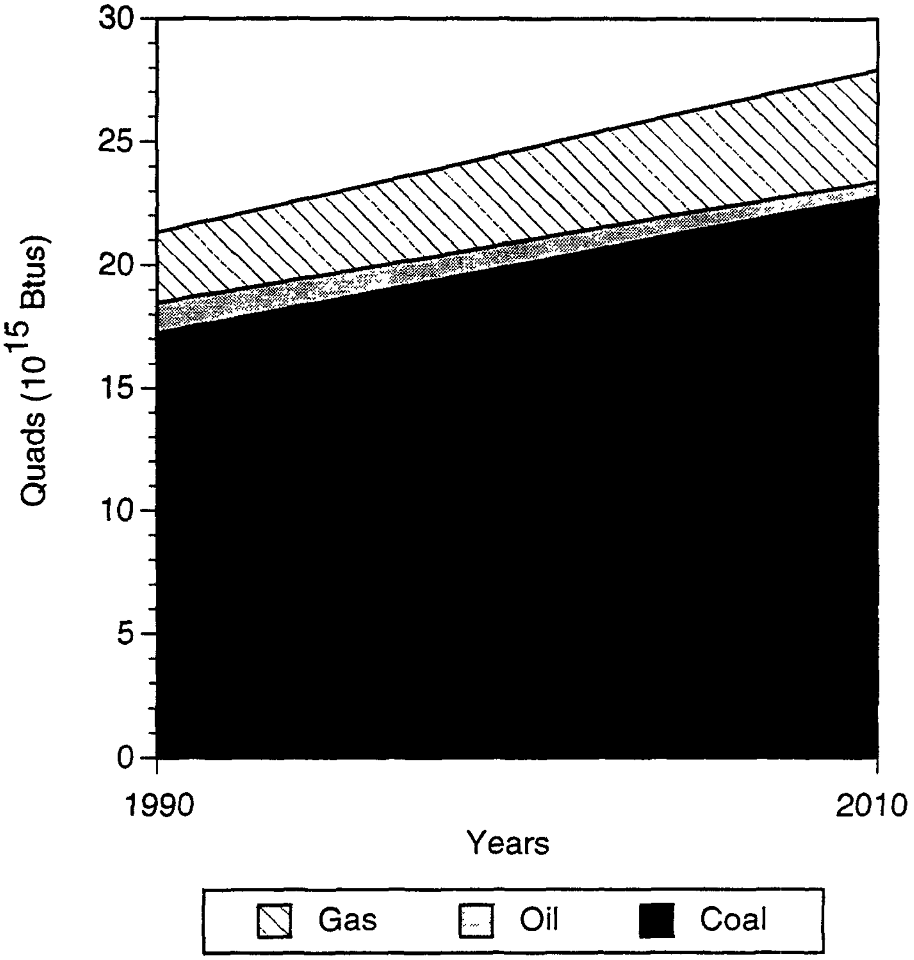


Figure 2-7. Projected use of fuels by 2010 for utility industry.[88]

size nor the location of the new units, the increased consumption has been allocated to existing units (in 2010) for the analyses. The decrease in oil consumption could result in units being retired or in a decrease in capacity factor, or a mix. The decreased consumption has been allocated among those oil-fired units believed to still be operating in 2010. Any new units built to accommodate the increased consumption would be required to comply with all applicable NSPS and State and local regulations. However, for the purposes of the analyses, the controls currently in use on the unit were considered to be in place for the same unit with increased fuel consumption.

The Agency projects that 135 units will be retired during the period between 1990 and 2010. These units have been removed from the 2010 analyses.

2.7.2 Title I and Title IV, Phase I and Phase II, Compliance Strategy Impact

Phase I and Phase II requirements of the Acid Rain Provisions, listed in Title IV, establish a cap on the national, annual SO_2 emissions. To achieve compliance with the requirements, utilities may do one or any combination of the following strategies (among others) at any given unit: (1) install flue gas scrubbers, (2) switch to a fuel that contains less sulfur, and (3) reduce the capacity factor of the Phase I unit to the extent that the unit is in compliance and provide plans for replacing the reduced capacity. This reduction can be accomplished by either: (1) energy conservation, (2) improved boiler efficiency, (3) use of a designated sulfur-free (nuclear or hydro not natural gas-fired) replacement, (4) use of a Phase II compensating unit, or (5) purchase of emission allowances. Emission allowances are allocated to existing utility units based upon historical operating conditions. One allowance equals the right to emit 1 ton SO_2. Affected units are required to turn in to the EPA one allowance for each ton SO_2 emitted in a calendar year. Unused or "excess" allowances may be sold on the open market.

The Phase I requirements affect 202 units at 110 plants, which must comply with the Phase I requirements by January 1996. To date, only 27 units at 16 plants (total generation capacity of about 14,058 MWe) have announced plans to install scrubbers to meet the Phase I requirements. The remaining 175 units (94 plants) will comply with Phase I requirements by (as mentioned above) either fuel switching (to low-sulfur coal), by buying more allowances than allocated, or by otherwise having enough allowances at the end of 1995 to cover their emissions.[89] Nearly 50 percent of the Phase I units plan to fuel switch or to blend to obtain low-sulfur coal. Each of the 27 units known to be installing scrubber units was modeled with the scrubber unit for the 2010 scenario. In the absence of specific plans for the remaining 175 units in Phase I, EPA modeled these units by increasing their coal consumption in proportion to ICF Resources, Inc. (2010) projections. These increases were also weighted by the expected increased use of western, low-sulfur coal.

Under Phase II of the Acid Rain Provisions, all utility units will be covered by 2000. Although industry projections suggest an additional 25 units (at 10 plants) will install scrubbers to comply with Phase II, the EPA believes that these units will comply with Phase II requirements using alternate methods.[90] This assumption was based on several factors including: (1) the increased availability of low-sulfur coal at favorable prices; (2) the introduction of processes that reduce sulfur emissions by 20 to 50 percent through partial cleaning of higher sulfur coal, which allows for a variety of coal types to be utilized (although the impact of these processes as Title IV control options is uncertain at this time); and (3) the increased age and small size of the affected units, giving the utility companies little incentive to spend large amounts of capital on installing scrubbers.[91]

Many utility units will be implementing NO_x controls to comply with both Title I and IV requirements. This control may involve switching from coal- or oil-firing to natural gas-firing (for at least a portion of the year), improved combustion controls, or installation of low-NO_x burners, among other activities. If a fuel switch was known, that switch was accounted for in the 1990 vs. 2010 analysis. No change in a unit's burner configuration (i.e., "old" vs. new low-NO_x) was included in the 2010 scenario. The impact of low-NO_x burner installation is discussed in chapter 10.

Under the acid rain program, the rules for NO_x control require that tangential-fired and dry-bottom wall-fired boilers subject to Phase I SO_2 reduction requirements also meet annual average NO_x emission limits of 0.45 lb/MMBtu and 0.50 lb/MMBtu, respectively, by January 1, 1996. Utilities that did not meet the limits were to comply with the regulation by installing low-NO_x burner technology or by averaging emissions among several units. This rule was issued as a direct final rule on April 13, 1995 (60 FR 18751).

Under Phase II of the acid rain program, the EPA will establish NO_x emission limits for all other boilers, including wet-bottom wall-fired boilers and cyclones, by January 1, 1997, and the affected units must be in compliance by 2000.[92] Also by 1997, the EPA will determine if more stringent emission limits should be established based on technology developments for dry-bottom wall-fired boilers or tangential-fired Phase II units.

Particulate control devices may also need to be upgraded at individual utility units to account for the different ash qualities of any new coal being utilized to comply with SO_2 requirements or to account for installation of low-NO_x burners. In late 1993, the Utility Data Institute (UDI) conducted a particulate control equipment survey to identify those utility facilities that were either in the process of upgrading their PM removal equipment or had definite plans to do so in the near future.[90] The survey was mailed to 286 utilities and received a 68 percent response. No information was received for 831 units; 1,215 units indicated that no PM control equipment modifications were planned. Modification plans were received for 132 units. The data received were analyzed for any potential impact on HAP emissions.[93] From the data, it appears that the modifications are being made strictly to account for differences in ash quality as coals are switched and not to effect an overall increase in PM control efficiency. Therefore, for the 2010 scenario analysis, it has been assumed that no change in PM control efficiency will occur since the actual reported values do not vary significantly. In addition, this assumption will account for any future degradation in PM control performance. The validity of this assumption is borne out by indications that some utility units are experiencing emissions increases (as evidenced by continuous emission monitor excess emission reports) following switches to lower sulfur coal and/or installation of low-NO_x burners.[94] It is not known how transient these excess emissions will be.

2.8 DISCUSSION OF FEDERAL INTERAGENCY REVIEW COMMENTS

Previous drafts of chapters 1 through 10, along with the appendices, were reviewed by numerous non-EPA scientists representing industry, environmental groups, academia, and other Federal Agencies during the summer of 1995. In February, April, and September 1996, all sections of the draft report underwent additional review by EPA, State and local agencies, and other Federal agencies. The EPA has revised the report, as appropriate, based on the reviewers' comments. The EPA revised the report to incorporate the majority of the comments received. However, there were several comments that could not be fully addressed because of limitations in data, methods, and resources. This section presents comments received by other Federal agencies that could not be substantially addressed in this interim report.

2.8.1 Comment

The Department of Energy (DOE) commented that "[o]ther 2010 utility industry scenario forecasts should be considered..." and the utility industry scenario forecasts used by the EPA should be compared to those of other groups (e.g., Edison Electric, EIA, GRI, The WEFA Group, Data Resources, Inc., etc.).[95]

2.8.2 Response

The EPA added discussions of the uncertainties and limitations to the approach used to estimate emissions for 2010. Also, the EPA acknowledges that other methods and other projections exist. However, the EPA did not evaluate or present the alternative approaches or projections in this report. To the extent feasible, the EPA plans to review other industry growth scenarios and projections of what the industry will look like in 2010 before issuing the final report.

2.8.3 Comments

The DOE also commented that "...EPA has chosen to describe utility sector emissions in a manner which misrepresents to Congress both the present and the future emissions...and thus overstates the argument for regulation." In addition, the DOE commented that the EPA had not captured "...the effects of fuel switching, fuel cleaning, combustion controls, and post-combustion controls implemented since [1991]" which have reduced air toxics. This omission leads to an overestimate of air toxic emissions.[96]

Similarly, the Council of Economic Advisors (CEA) commented on the EPA's growth estimates and noted that "[n]o justification is provided...to explain the assumed lack of improvement in either generating efficiency or abatement technology that would make it possible to meet the increased demand for electricity without causing as much pollution. In particular, the report seems to underestimate the potential of biomass as fuel. The predicted growth in demand for electricity for the 1990 to 1995 period exceeds actual industry experience, raising doubts about the accuracy of the predicted growth in emission from 1990 to 2010. Also, the report does not adequately examine the possibility of substituting demand side management for expansion of output." [97]

2.8.4 Response

As noted in this chapter, the baseline was chosen to characterize the industry as it stood, and to quantify the risk to human health, as of 1990, the most recent year for which facility data were available when the project started. This approach is believed to be appropriate and consistent with the mandate of section 112(n)(1)(A).

As to future emissions, the EPA has incorporated into its 2010 analysis those changes announced by utilities for compliance with Title IV of the Act. The EPA is unaware of any significant changes in the industry over the past 5 years (or planned for the next 10 years) that would increase generating efficiency or HAP emission removal efficiency to any great extent. The EPA notes that fuel use has increased in the past 5 years which may have had the effect of negating any potential HAP emission reductions accruing from increased use of control devices. The use of the control measures noted by the DOE and CEA have been addressed in the interim report (see chapter 10) to the extent possible using current information. Further analyses may be performed for the final Report to Congress should additional data become available.

No change in a unit's combustion control efficiency was made to account for increased use of NO_x controls in 2010. Existing data indicate that combustion controls would not significantly affect HAP emissions. Where new controls for SO_2 were known to be installed, these controls were employed in the 2010 analyses. Where EPA has estimated the impact of new facilities, the emission reductions are based on compliance with the new source performance standards (NSPS) for both PM and SO_2 control.

Although the EPA believes its approach is reasonable, the EPA also recognizes that there are uncertainties in the assessments. The EPA intends to address these uncertainties and consider the above comments, to the extent feasible and appropriate, before issuing a final report. In addition, the EPA intends to address other factors that may impact on emission projections, including improvements in HAP emission removal by FGD units and the impact of any actions taken under Title I of the Act (e.g., any tightening of the NAAQS for PM) before issuing a final report.

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3.0 EMISSION DATA GATHERING AND ANALYSIS

3.1 LITERATURE REVIEW AND BACKGROUND

Prior to the beginning of this study, the Agency conducted a literature search of available nonradionuclide HAP emission and control information and assessed the usefulness of these data. Much of the data had been gathered over an extended time period using a wide variety of dated, and sometimes ill-defined, sampling and analytical techniques. Many of these techniques, including the method for mercury, have since been replaced with more accurate methods. The literature data exhibited extensive variability in the reported concentrations of HAPs in emissions (sometimes varying by several orders of magnitude). There was often insufficient documentation of the techniques and assumptions used to distinguish the reliable data from the unreliable data.

In addition, many of these literature data were gathered at laboratory or pilot-scale installations or from utility units that did not reflect the configurations of the current utility unit population. Again, there was often insufficient documentation of the design parameters or process operating conditions to assess the validity of the data or the impact of the process operating conditions on the nonradionuclide emissions.

Because of these deficiencies, the Agency was not able to use the prior existing data (prior to 1990) in control strategy analysis or to project the data for nationwide application in the health hazard assessment. The EPRI and DOE conducted major test programs during the period from 1990 to 1994 to obtain HAP data from the utility industry and coordinated these programs and test methodology processes with the EPA. These new data from field testing became available for this report during 1993 and 1994.

To obtain the necessary nonradionuclide chemical HAP emission test data, two avenues were followed. The first was to pursue cooperation with industry and DOE test programs, and the second involved Agency testing for HAP organics. The EPRI tested emissions at locations of about 25 of their member companies for approximately 25 of the 189 HAPs listed in section 112(b) of the Act. Of these locations, test reports were available for 29 sites (and particulate control data for one additional site) in time for inclusion in the health hazard assessment. These tests encompassed coal-, oil-, and gas-fired boilers of several firing types and emission control technologies. Emission test sites were selected based on industry utilization (e.g., the largest percentage of coal-fired units are dry bottom, use bituminous coal, with ESPs for PM control). This approach allowed the acquisition of data for the broadest spectrum of the utility industry in the most cost-effective manner. Some of the EPRI emission test sites were DOE Clean Coal Technology (CCT) sites, which provided for the acquisition of HAP data before and after installation of controls for NO_x, an important element in the acid rain program (under Title IV of the Act). This information will be helpful in determining the implications of the acid rain program on HAP emissions. In the test programs, samples were collected before and after each emission control device when feasible. The Northern States Power Company (NSPC) also provided test reports from eight of its coal-fired plants for testing performed from 1990 to 1992.

The DOE, through its Pittsburgh Energy Technology Center (PETC), initiated contract activities in mid-1992 for a HAP emission sampling program at eight coal-fired utility units. The contracts were awarded in early 1993 and the sampling was completed at seven sites over the period from June to December of 1993. The DOE program was similar in nature and scope to that of EPRI, although the number of facilities evaluated was much smaller under the DOE program. The timing of the DOE program was such that the data were available concurrently with those from the EPRI studies and could be analyzed for this report.

The EPA was involved with the design and test method selection for both the EPRI and the DOE test programs. The Agency also cofunded a field validation of several mercury emission test methods at a coal-fired utility boiler with EPRI, including those methods that measure the various species of mercury that may be emitted from a utility boiler.[1]

For the EPRI program, the Agency independently developed a matrix of the industry and established that the types of plants selected for the EPRI program were the same types that would have been selected for an EPA emission test program. For the DOE program, the Agency had the opportunity to provide input into the type of plants that should be selected. All emission test reports from both programs, and from individual company tests, were reviewed by the EPA for completeness, adherence to accepted sampling and analytical techniques, and proper unit operations (typical information missing from the existing literature-based database). The Agency provided support for the onsite quality assurance/quality control activities performed during the DOE program.

The EPA also completed the initial development of the Fourier transform infrared spectrometry (FTIR) field testing system and system validation for real-time, simultaneous measurement of approximately 120 gaseous organic HAPs. Validation tests for the FTIR at a coal-fired unit were conducted in February 1993.[2] The FTIR system was utilized in emission testing by the EPA at five utility sites. The FTIR system is a lower-cost and much more flexible measurement technology than those currently available for sources of organic HAPs. To examine the magnitude of HAP emissions from utility units, the EPA conducted emissions testing with FTIR as a screening level analysis. Since few HAPs were detected with FTIR testing at these five utility sites, the EPA decided not to use the FTIR test results to estimate HAP emissions from utility units at this time.

For inclusion in this report, a total of 52 tests were conducted at 48 sites using FTIR and conventional sampling and analysis methods from the EPRI, the DOE, the NSPC, and the EPA. Although 52 test reports were received by EPA in time for inclusion in this study, 3 contained data that could not be used in the emission factor program (EFP) described in section 3.4. These reports were excluded because measurements were not made between the boiler and the PM control device. This exclusion resulted in a test containing only a fuel analysis and stack emission numbers, which did not allow analysis of control system effectiveness.

The data reliability and the precision and accuracy of the sampling techniques were addressed by the individual test contractors in their test reports. If a contractor had concerns about the quality of the data or about the precision or accuracy of a particular test sample, the EPA did not use the data in its computations.

3.2 POLLUTANTS STUDIED

As many of the 189 HAPs listed in section 112(b) as possible were included in this study. Table A-1 (Appendix A) lists the HAPs that were detected at least once in the utility test data (excluding FTIR-detected data), their estimated nationwide emissions in 1990, and projected nationwide emissions for 2010.

The Agency's Office of Radiation and Indoor Air (ORIA) has completed a report on radionuclide emissions from the utility industry. The results of this study, along with their impact on public health, are included in chapter 9.

3.3 DESCRIPTION OF EMISSION TEST PROGRAMS

At the beginning of this study, the utility industry was characterized. Through the use of the EEI Power Statistics Database (1991) from the UDI,[3] a matrix was developed showing the ranking of utility unit configurations from the most to the least prevalent. Table B-1 (Appendix B) shows these unit configurations down to a unit type that accounts for only 1.05 percent of the fossil-fuel-fired unit megawatts in the U.S. (plus any additional unit types tested that were below this cutoff). The matrix was then used as a guide to gather data on the largest number of unit configurations possible with the available resources by targeting the most prevalent unit types.

The emission test reports used in this study were produced for various government agencies as well as for nongovernment and industry groups (discussed above). Although various test contractors performed this emission testing, certain specific testing protocols were followed. Table B-2 (Appendix B) provides a list of all the sites that were available for this report and were tested under the DOE, the NSPC, the EPRI, and the EPA test programs. The table also shows the type of fuel burned and the emission controls applied to the boiler system. In some cases, the controls are pilot-scale units applied to a slipstream from the boiler flue gas system. The contractor who tested the boiler and the date of the test report are also given.

3.4 DEVELOPMENT OF HAP EMISSION TOTALS

To estimate emissions of HAPs from fossil-fuel-fired electric utility units (≥25 MWe), the EPA developed the EFP. This program incorporates unit configuration data from individual units as well as emission testing data to compute estimated emissions. An explanation of the program and several assumptions about the data and how they were used are described here.

3.4.1 Program Operation

Emissions of HAPs considered in this study consist of two types: trace elements and organic compounds. Trace elements exist in the fuel when fired, while the organic HAPs are mostly formed during combustion and postcombustion processes. Different programming methods are required for handling the two types of HAPs. Program diagrams for modeling trace element emissions are shown in Figure 3-1 for coal and Figure 3-2 for oil and gas. The two figures differ only in treatment of the fuel before trace elements reach the boiler. Figure 3-3 shows the program diagrams for modeling organic HAP emissions.

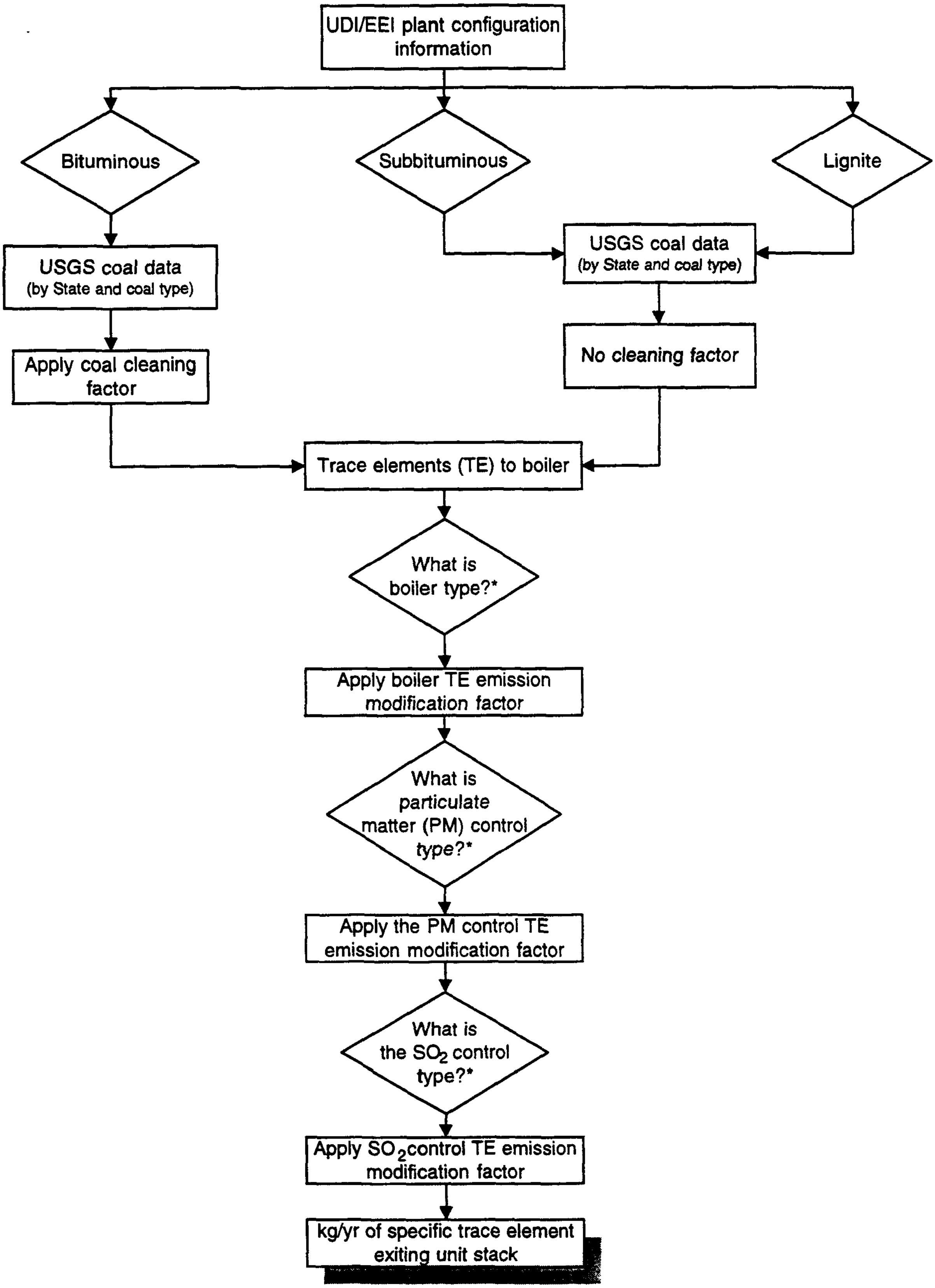


Figure 3-1. Trace elements in coal.

*Taken from UDI/EEI data.

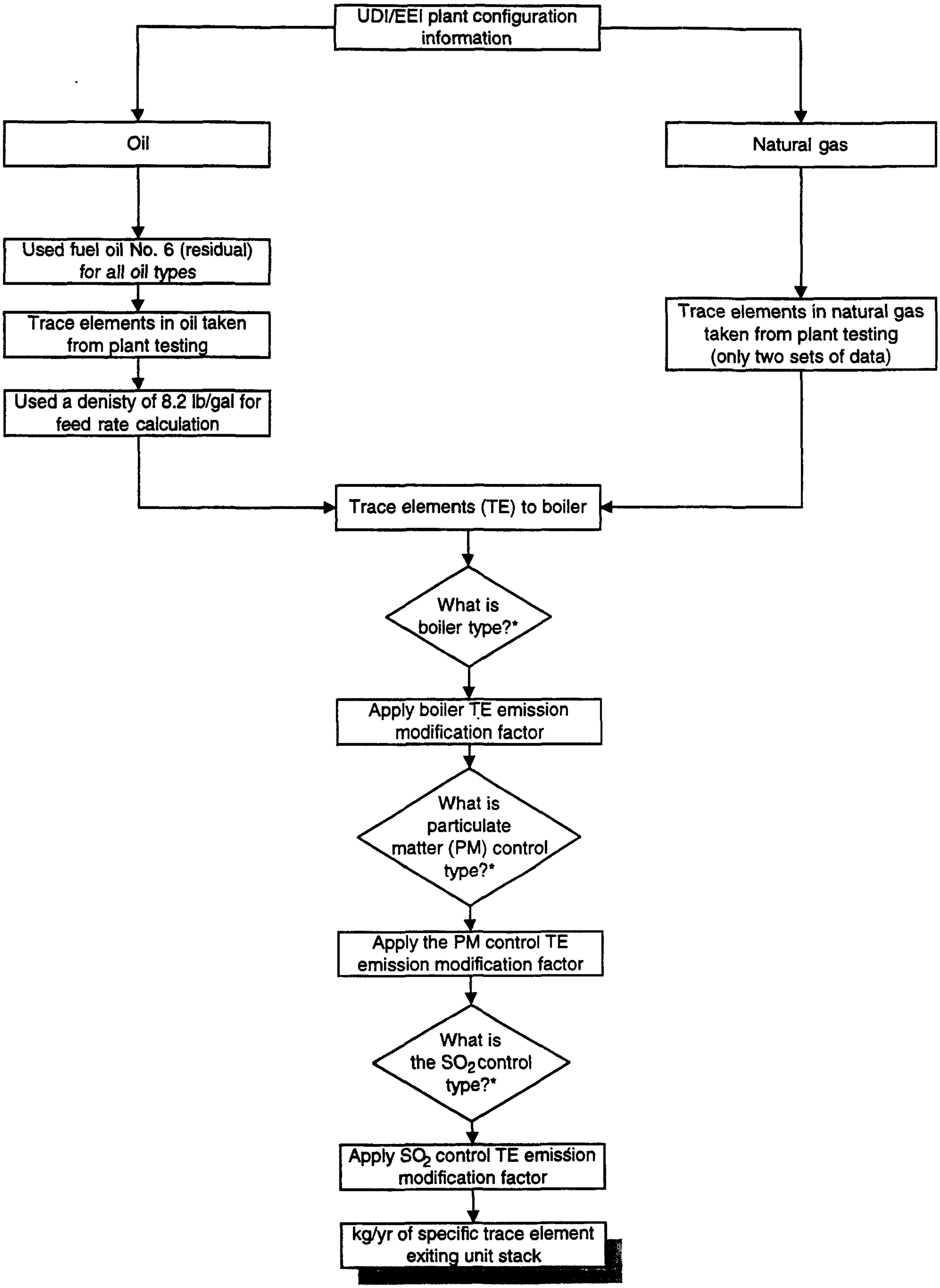
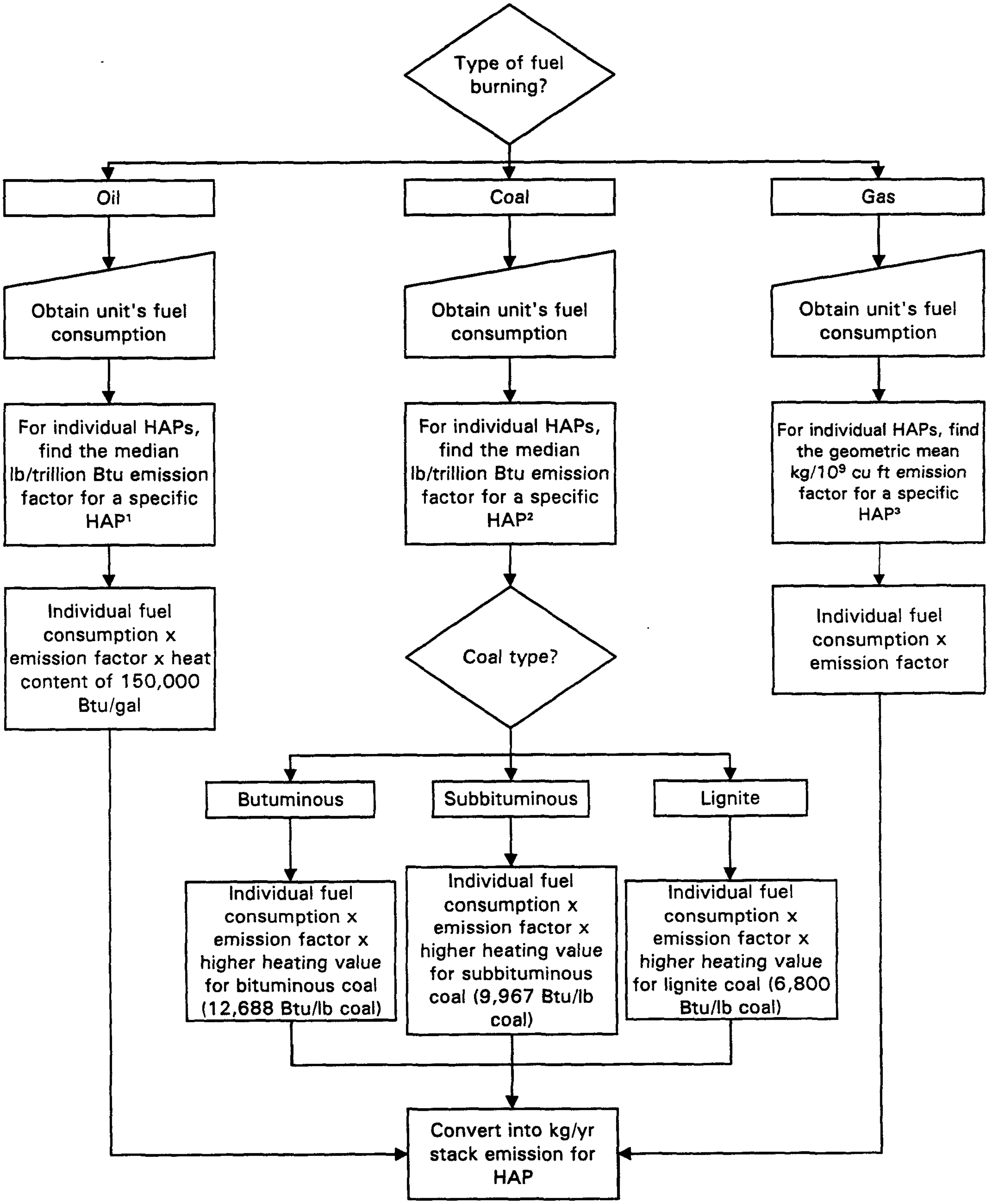


Figure 3-2. Trace elements in oil and natural gas.

*Taken from UDI/EEI data.



[1] *Only oil-fired units were used to obtain these emission factors.*
[2] *Only coal-fired units were used to obtain these emission factors.*
[3] *Only gas-fired units were used to obtain these emission factors.*

Figure 3-3. Organic emissions.

3.4.2 Data Sources

The EFP was built to accept data from two sources. The first source of data is a data input file containing plant configurations, unit fuel usage, and stack parameters. This input file was based on UDI/EEI Power Statistics database (1991 edition). The UDI/EEI database is composed of responses from electric utilities to the yearly updated DOE Energy Information Administration (EIA) Form EIA-767.

The second data file is an emissions modification factor database. This database contains information from emissions tests conducted by EPRI, DOE, and the electric utility industry.

The program first searches the input file for the type of fuel burned and the amount of fuel consumed per year in an individual unit. If the fuel type is coal, the EFP then looks for the coal's State of origin. Origin is important because the trace elements in coal are addressed by coal type (bituminous, subbituminous, and lignite) and State of origin in the U.S. Geological Survey (USGS) database, which analyzed core and channel samples (3,331 samples) of coal from the top 50 (1990 or later) economically feasible coal seams in the U.S.

3.4.3 Operational Status of Boilers

The operational status of units was taken from the UNIT_90.dbf file of the EEI/UDI Power Statistics database (1991 edition addressing 1990 data). Only units that were listed as either operational or on standby were used in the EFP. It was found that 151 units were listed as being on standby in the EEI/UDI Power Statistics database but were actually on indefinite standby and thus did not emit any HAPs. These units were excluded from the nationwide emissions totals in Appendix A.

Only coal-fired, oil-fired, and natural gas-fired units were included in the EFP. This decision was made because units using these fuels make up an overwhelming majority of the fossil-fuel-fired electric utility units with a capacity ≥25 MWe.

Anthracite was disregarded as a fuel because of the limited number of units burning this type of coal.[4] Four units burning anthracite coal were assigned to burn bituminous coal for program computations.

Coal-fired boiler concentrations were modified for different heating values, depending on the type of coal, before being converted to a rate basis (kilograms per year [kg/yr] of individual HAP). This procedure was necessary because different coal ranks have different heating values. For example, it would require burning more lignite to achieve the same heat input to the boiler as burning bituminous coal. These values were determined as averages for each type of coal (see Table 3-1).[5]

The EEI/UDI database had a number of gaps in the fuel consumption data. Some of these gaps were filled by data supplied voluntarily by the industry. To address the remaining gaps, the available data were plotted and point-slope equations were fit to estimate fuel consumption.[6] These equations involved plotting nameplate megawatts (modified to take into account the unit's capacity factor) against fuel usage. If the fuel usage and the unit capacity factor in 1990 were not given, 1989 fuel consumption data were used. If 1989 data were not available, the geometric mean of the 1980-1988 EEI fuel consumption data was used. When all other options had been tried unsuccessfully, an average fuel consumption of units rated within ±5 MW of the unit with unknown fuel usage was used.

Utility units may burn coal that originated from several States; however, in the EFP each coal-fired unit was assigned a single State of coal origin.[7] The State of origin used in the EFP was the State that contributed the highest percentage of the unit's coal.

3.4.4 Trace Element Concentration in Fuel

The USGS database contains concentrations of trace elements in coal that were extracted from the ground but does not include analyses of coal shipments. The concentrations of trace elements in coal in the ground and in coal shipments to utilities may differ because, in the process of preparing a coal shipment, some of the mineral matter in coal may be removed. Since approximately 77 percent of the eastern and midwestern bituminous coal shipments are cleaned[8] to meet customer specifications on heat, ash, and sulfur content, a coal cleaning factor was applied to most bituminous coals in the EFP.[9]

For a unit that burned bituminous coal, the kg/yr feed rate of trace elements to the boiler was determined from the average trace element concentration in the coal, a coal cleaning factor, and the annual fuel consumption rate. No coal cleaning factors were applied to lignite and subbituminous coals (see Equations No. 1 and No. 2 in Table D-2, Appendix D).

Oil-fired organic HAP exit concentration calculations included a 150,000-Btu/gallon (gal) heating value for oil. An oil density of 8.2 lb/gal was also used.

An emission rate for each organic HAP emitted from gas-fired units was extracted from the test reports. There were only two

Table 3-1. Average Higher Heating Values of Coal[10]

Class and group[a]	Agglomerating character	Fixed carbon limits, % (dry, mineral-matter-free basis)		Volatile matter limits, % (dry, mineral-matter-free basis)		Calorific value limits, Btu/lb (moist,[b] mineral-matter-free basis)		Average
		Equal or greater than	Less than	Equal or greater than	Less than	Equal or greater than	Less than	
I. Bituminous								
1. Low-volatile bituminous coal	commonly agglomerating[c]	78	86	14	22	—	—	
2. Medium-volatile bituminous coal	"	69	78	22	31	—	—	
3. High-volatile A bituminous coal	"	—	69	31	—	14,000[d]	—	14,000
4. High-volatile B bituminous coal	"	—	—	—	—	13,000[d]	14,000	13,500
5. High-volatile C bituminous coal	"	—	—	—	—	11,500	13,000	12,250
High-volatile C bituminous coal	agglomerating	—	—	—	—	10,500	11,500	11,000
Average of Averages (Value used in EFP for bituminous coal)								***12,688***
II. Subbituminous								
1. Subbituminous A Coal	nonagglomerating	—	—	—	—	10,500	11,500	11,000
2. Subbituminous B Coal	"	—	—		—	9,500	10,500	10,000
3. Subbituminous C Coal	"	—	—	—	—	8,300	9,500	8,900
Average of Averages (Value used in EFP for subbituminous coal)								***9,967***
III. Lignitic								
1. Lignite A	nonagglomerating	—	—	—	—	6,300	8,300	7,300
2. Lignite B	"	—	—	—	—	—	6,300	6,300
Average of Averages (Value used in EFP for lignite coal)								***6,800***

[a] This classification does not include a few coals, principally nonbanded varieties, which have unusual physical and chemical properties and which come within the limits of fixed carbon or calorific value for high-volatile and subbituminous ranks. All of these coals either contain less than 48 percent dry, mineral-matter-free fixed carbon or have more than 15,500 moist, mineral-matter-free Btu per pound.

[b] Moist refers to coal containing its natural inherent moisture but not including visible water on the surface of the coal.

[c] It is recognized that there may be nonagglomerating varieties in these groups of the bituminous class, and there are notable exceptions in high-volatile C bituminous group.

[d] Coals having 69 percent or more fixed carbon on the dry, mineral-matter-free basis shall be classified by fixed carbon, regardless of calorific value.

test reports that analyzed organic HAPs, and a geometric mean emission rate of each observed organic HAP was used. This rate in kilogram HAP/10^9 cubic feet was then multiplied by the unit's gas consumption to obtain a kilogram HAP/year stack emission rate of each specific HAP.

3.4.5 HCl and HF Concentration in Fuel

To obtain hydrogen chloride (HCl) or hydrogen fluoride (HF) emissions from the boiler, emission factors were derived by performing mass balances for chloride and fluoride, then converting these balances to the equivalent levels of HCl or HF throughout the boiler system.[11] For example, for each part per million of chloride in the feed coal at one of the test sites, 0.61 lb/h of HCl was found in the gas stream leaving the boiler and 0.00145 lb/h in the stack gas. Similarly for HF, the boiler emissions were 0.56 lb/h for each part per million of fluoride in the coal and 0.00448 lb/h in the stack. For ease of programming, the HCl and HF emissions were addressed starting in the fuel. This programming was done by multiplying the chloride and fluoride concentrations in the fuel constituents by 0.61 or 0.56, respectively. The resulting numbers allowed direct conversion into boiler emissions that could be further modified for systems with PM control or SO_2 control.

The chloride concentrations were not available for coals from the following States: Alaska, Illinois, Indiana, Iowa, Missouri, Utah, and Washington. Chloride concentrations were assigned, as shown in Table 3-2, for coals originating from these States.[12]

3.4.6 Emission Modification Factors for Inorganic HAPs

To address the partitioning of the HAP stream through the combustion and pollution control process, partitioning factors known as emission modification factors (EMFs) were developed from inorganic HAP testing data. The EMFs are fractions of the amount of a HAP compound exiting a device (boiler or air pollution control device [APCD]) divided by the amount of the same HAP compound entering that device.[13] These EMFs were averaged by taking the geometric mean of similar devices (e.g., all oil-fired tangential boilers, all cold-side ESPs). Geometric means were used because of the presence of outlying data points, the small amount of data, and the general fit of the data to a log-normal curve. These geometric means were then applied to the kg/yr feed rates entering the boiler, the effect of which either reduced or left unchanged the emissions that passed through them. Those EMFs calculated as being greater than 1.0 (i.e., more material exiting a device than entering it) are set to equal 1.0.

Table 3-2. Assigned Chloride ppmw and HCl ppmw Concentrations in coal, by State of Coal Origin [12]

State	**Conversion of assigned ppmw chloride to assigned HCl ppmw**	**Assigned ppmw HCl in coal**
Alaska	54 x 0.61 =	32.9
Illinois	1,136 x 0.61 =	693.0
Indiana	1,033 x 0.61 =	630.0
Iowa	1,498 x 0.61 =	914.0
Missouri	1,701 x 0.61 =	1,038.0
Utah	220 x 0.61 =	134.0
Washington	104 x 0.61 =	63.0

Nearly all EMFs were computed from three data samples before and three data samples after the particular device. When all six data samples for a particular EMF computation were nondetects, the EPA decided to disregard the EMF. As such, EMFs were computed when there was at least one detected sample among the six measured samples. Appendix D discusses in more detail the methodology used to develop emission totals.

The EMFs were computed with data from different test reports but for similar devices (i.e., cold-side ESPs, front-fired boilers in oil-fired units). The data from coal-fired units were not segregated by State of coal origin. The EMFs from devices are segregated into only coal-, oil-, or gas-fired bins.

The EFP itself uses EMFs to partition the emissions as they proceed from the fuel through the unit to the stack exit as follows. The average concentrations of metallic HAPs in an individual fuel by State (based on USGS data) were multiplied by the amount of fuel that the unit burned in 1990. After accounting for variables such as coal cleaning (bituminous coal only) and coal type (higher heating value), the emission concentration of an inorganic HAP was thus converted into an emission rate in kg/yr entering the boiler. The emission rate entering the boiler was then modified by EMFs for the boiler, particulate control device (when applicable), and the SO_2 control device (when applicable).

As stated above, these geometric mean EMFs were then applied to the fuel HAP concentration estimates and the kilogram/year fuel feed rates entering the boiler, the effect of which either reduced or left unchanged the emissions that passed through it, depending on the value of the EMF.

Appendix C contains all of the EMFs used to develop the unit emission estimates for inorganic HAPs.

3.4.7 Organic and Mineral Acid HAPs

Organic and mineral acid HAP emissions were handled in one of two ways. The first method was used only with HCl or HF emissions. The numbers resulting from the method allowed direct conversion into boiler emissions that could be further modified for systems with PM control or SO_2 control.

Hydrochloric acid and HF EMFs for PM and SO_2 control devices were developed with data from four test reports where contractors conducted tests individually for HCl and chlorine as well as HF and fluorine, before and after each control device. The rest of the available reports showed tests only for chlorine and fluorine and estimated the fractions that were HCl or HF. In developing the HCl and HF EMFs for wet FGDs and dry scrubbers, the EPA decided to address the effect of flue gas bypass. After analyzing test data and having discussions with industry representatives, it was decided to assume an industry average flue gas bypass of 17 percent for wet FGDs and 14 percent for dry scrubber systems. This assumption was used only in the development of HCl and HF EMFs.[14] Because each of the four test sites was different than the others regarding SO_2 and PM control, the emission factors for chlorine and fluorine were maintained separately for the four system types rather than averaging them.

The second method of handling organic and mineral acid HAPs was for organics. Because organic HAPs were not always tested at the entrance and exit of each control device in the emissions testing, all organic HAP emissions were addressed by examining the test data and determining the concentration of a particular HAP exiting the stack. Organic HAP concentrations were obtained from emission test reports.

If stack emission or APCD exit emission data were unavailable or reported as nondetected, and, if at least one-third of the data samples at the inlet of the APCD were detected concentrations, EPA used organic emissions at the inlet of the APCD and accounted for the effect of the APCD with EMFs. For each individual organic HAP observed in testing, a median concentration was obtained. This fuel-specific median concentration was then individually multiplied by each utility unit's fuel consumption. The result was a fuel-specific emission rate for all organic HAPs that were observed at least once during testing.

3.4.8 Model Estimates for the Year 2010

Emission estimates for 2010 were derived from the same basic model described above. However, changes to input files were made to accommodate expected changes in fuel usage (by fuel type), generating capacity, and responses to Phases I and II of the 1990 amendments under Title IV. The details of these expected changes are described in section 2.7.

3.5 SELECTED ESTIMATED NATIONWIDE HAP EMISSIONS

Based on the screening assessment presented in chapter 5, a subset of HAPs was determined to be of potential concern to public health and was, therefore, given a priority label for further analysis. Table 3-3 presents estimated 1990 and 2010 nationwide emissions of this subset of HAPs from utility units (see also Table A-1, Appendix A).

3.6 COMPARISON OF EFP ESTIMATES WITH TEST DATA

Comparisons were made between test data from 19 utility boiler stacks and predicted emissions for the same plants using the EFP.[15] Results suggest that the EFP performs as expected, i.e., across a range of boilers and constituents the average of the predicted values agrees well with the average of the reported values. This close prediction occurs even with large differences between predicted and reported values for individual boilers and constituents.

For three elements and 19 boilers, the average of predicted emissions (as represented by pounds emitted per trillion Btu) was about 1 percent different from the average of reported values. Averages for estimates of three individual elements were different from the test values by +38, -28, and -6 percent. The highest individual difference between predicted and reported values was represented by a factor of 5,000.

Table 3-4 presents comparisons for the individual plants for arsenic, chromium, and nickel. Averages for each element and for the combination of all three elements are also given. The reported values are ratios of EFP estimates to measured values in terms of pounds of element emitted per trillion Btu heat input. The test data represent only a few hours of operation at each plant, while the EFP estimates are extrapolated to annual emissions. Plants 1 through 14 fire coal, plants 18 and 19 fire

Table 3-3. Selected Nationwide HAP Emissions (estimated) in tons/year for 1990 and 2010

	Selected nationwide HAP emissions (estimated) in tons/year for 1990	Selected nationwide HAP emissions (estimated) in tons/year for 2010
Coal-fired electric utility plants		
Arsenic	54	62
Beryllium	6.6	7.6
Cadmium	1.9	2.3
Chromium	70	83
Lead	72	83
Manganese	180	232
Mercury	51	65
Hydrogen chloride	137,000	150,000
Hydrogen fluoride	19,500	25,600
2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)	0.000154	0.000203
n-nitrosodimethylamine	5.9	7.7
Oil-fired electric utility plants		
Arsenic	5	3
Beryllium	0.45	0.23
Cadmium	1.7	0.87
Chromium	4.7	2.4
Lead	10.6	5.3
Manganese	9.5	4.8
Mercury	0.25	0.13
Nickel	389	197
Hydrogen chloride	2,870	1,456
2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)	0.000011	0.000005
Natural-gas-fired electric utility plants		
Arsenic	0.16	0.25
Nickel	2.3	3.5
Formaldehyde	55	83

Table 3-4. Comparison of Utility Boiler Emissions from EFP Estimates and from Tests[a]

Plant number	Arsenic EFP/ test	Chromium EFP/ test	Nickel EFP/ test
1	0.12	0.15	0.0030
2	0.26	0.11	0.27
3	1.25	0.35	0.08
4	1.41	1.19	1.30
5	7.67	1.76	0.79
6	0.99	0.25	0.37
7	1.54	0.59	0.79
8	0.16	1.12	7.33
9	0.0004	0.0040	0.0002
10	0.72	0.33	0.11
11	2.25	0.18	0.18
12	0.20	0.0042	0.0008
13	0.01	1.09	0.03
14	5.49	3.13	0.29
15	0.05	0.16	0.17
16	1.92	0.54	0.63
17	1.23	0.75	0.40
18	0.59	1.11	2.44
19	0.30	0.89	2.70
Average: EFP/test	1.38	0.72	0.94
Average for all 3 elements		1.01	

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[a] Values presented are the ratio of emission factor program estimates to test data in terms of lb/10^{12} Btu.

a combination of coal and petroleum coke, and plants 16 and 17 fire oil.

Possible reasons were examined for large differences between projected and actual emissions. In the EFP, only one fuel was assumed to be burned. However, some of the plants burned combinations of coal and petroleum coke, but the EFP recognizes only coal from one State. The petroleum coke used by one plant had nickel concentrations that may be more than 100 times higher than that found in the Montana coal used for that plant by the EFP, and concentrations in ash on the order of 1,000 times higher than that found in coal ash. In this case, the EFP underpredicted actual nickel emissions by factors of up to 5,000 as mentioned above.

3.7 CHARACTERISTIC PLANT EMISSIONS

To give the reader a better grasp of the potential emissions of selected HAPs from an individual utility unit, a set of characteristic units was chosen (one for each fuel type). The EFP and organic HAP stack emission factors were then used to determine the units' projected HAP emissions of concern (according to the health hazard assessment).

In presenting the characteristic coal-fired unit, the EPA looked for an existing utility unit that had the characteristics of a typical coal-fired unit in the U.S. Once the specific plant was chosen, its 1990 HAP emissions of concern (projected by the EFP and organic HAP stack emission factors) were listed (see Table 3-5).

The most important parameter of the characteristic oil and gas-fired plants (see Tables 3-6 and 3-7) is their fuel consumption, as there are usually no control devices to reduce emissions. The fuel consumptions chosen are the averages of each fuel type (oil or gas).

The characteristic unit emissions in these three tables are actually projected emission outputs from the EFP for three existing units. They are chosen for having the most prevalent fuel, boiler/furnace, and control device type in their fuel class (coal, oil, or gas). They are also chosen for having megawatt capacities that are the average for their fuel class.

It should be noted, however, that characteristic emissions are based on 1990 fuel consumption values, and the emission testing (on which the EFP is based) was performed under

Table 3-5. **Emissions from an Characteristic Coal-fired Electric Utility Plant (1990)**

	1990
Megawatts	approximately 325 MWe
Coal source	Kentucky (bituminous, sulfur content of 1.5 - 3.5 %)
Fuel consumption	approximately 600,000 tons/year
Particulate matter control device	Electrostatic precipitator (cold-side)
Sulfur dioxide control device	Compliance fuel/State implementation plan (SIP)
	Selected HAP emissions (estimated) in tons/year for 1990
Arsenic	0.081
Beryllium	0.012
Cadmium	0.00051
Chromium	0.086
Lead	0.075
Manganese	0.13
Mercury	0.05
Hydrogen chloride	194
Hydrogen fluoride	14.4
2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)	0.00000014
n-nitrosodimethylamine	0.00515

Table 3-6. Emissions from an Characteristic Oil-fired Electric Utility Plant (1990)

	1990
Megawatts	approximately 160 MWe
Fuel	Residual oil (sulfur content < 1.5 %)
Fuel consumption	approximately 640,000 barrels / year
Particulate matter control device	None
Sulfur dioxide control device	Compliance fuel/State implementation plan (SIP)
	Selected HAP emissions (estimated) in tons/year for 1990
Arsenic	0.0158
Beryllium	0.0029
Cadmium	0.0077
Chromium	0.0182
Lead	0.053
Manganese	0.0377
Mercury	0.0012
Nickel	2.1
Hydrogen chloride	9.4
2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)	0.000000035

Table 3-7. Emissions from an Characteristic Gas-fired Electric Utility Plant (1990)

	1990
Megawatts	approximately 240 MWe
Fuel consumption	approximately 4,600,000,000 cubic feet / year
Particulate matter control device	N/A
Sulfur dioxide control device	N/A
	Selected HAP emissions (estimated) in tons/year for 1990
Arsenic	0.0003
Nickel	0.004
Formaldehyde	0.09

essentially steady-state conditions (with little or no variation from the baseline operating condition). Therefore, the characteristic emissions are a snapshot in time. In reality, emissions of HAPs are not constant, steady-state values, but fluctuate with operating conditions as well as changes in fuel.

3.8 UNCERTAINTY ANALYSIS OF EMISSION FACTOR PROGRAM

In order to quantify the uncertainty in the EFP output, the EFP was subjected to an uncertainty analysis using the statistical method known as a Monte Carlo analysis. The results of this analysis are presented in section 6-14.

3.9 REFERENCES

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4.0 INTRODUCTION FOR THE HEALTH HAZARD RISK ANALYSIS

4.1 INTRODUCTION AND BACKGROUND

The EPA partially evaluated the potential hazards and risks for the year 1990 and for the year 2010. A significant portion of the analyses focuses on inhalation risks due to utility emissions within a 50 km radius of each facility (i.e., local analysis). The analyses of long-range transport or regional analysis (i.e., emissions dispersion and exposure outside of 50 km) and multipathway assessment (e.g., risk due to ingestion and dermal exposure) were limited, mostly qualitative, and considered only a few pollutants. This situation does not necessarily mean that inhalation exposure within 50 km is the most important route of exposure. For some of the HAPs emitted from utilities (e.g., mercury and dioxins), noninhalation exposure through ingestion is likely to be the dominant route of human exposure.[1,2]

The estimates of risks due to inhalation exposure presented in this report are the incremental increased risks due to utility emissions only. For the most part, this assessment does not consider exposure to emissions from other sources and does not consider background levels of the HAPs in the environment. However, background concentrations were evaluated to a limited extent and are discussed briefly in later sections of this report.

This chapter begins with a summary of risk assessment principles and guidelines as used by the EPA and discussions of pertinent reports such as the NAS recent report *Science and Judgement in Risk Assessment*[3] and the EPA Science Policy Council's (SPC's) *Guidance for Risk Characterization.*[4] Section 4.2 presents the general approach and methods for this health hazard risk assessment. Section 4.3 discusses health effects data. Section 4.4 describes the methodology to the inhalation exposure assessment, and section 4.5 describes the methodology for estimating inhalation risks.

4.1.1 Principles of Risk Assessment

Risk assessment is a multidisciplinary evaluation of factual information as a basis for estimating and evaluating the potential health effects that individuals or populations may experience as a result of exposure to hazardous substances. Risk assessments typically involve both qualitative and quantitative information.

Risk estimates describe the nature and likelihood of adverse effects and the probabilities that these health effects will occur in an exposed population. Numerical risk estimates can be calculated for two categories of adverse health effects:

- Risk of developing cancer
- The likelihood of developing adverse health effects other than cancer (e.g., asthma).

To derive statements of risk or the likelihood of adverse health effects, quantitative information on exposure is combined with information on toxicity. This process is different for carcinogens and noncarcinogens due to the underlying assumptions that cancer is a nonthreshold phenomenon and that thresholds exist for adverse health effects other than cancer (i.e., noncancer effects).

In 1983, the National Academy of Sciences established a framework to guide risk assessments by Federal agencies.[5] As defined by the NAS, risk assessment consists of four steps:

- Hazard assessment, or hazard identification
- Dose-response assessment
- Exposure assessment
- Risk characterization.

Hazard identification is the review of relevant toxicologic, biological, and chemical information to determine whether or not a pollutant may cause adverse health effects. It is a qualitative assessment of the potential of a pollutant to increase the incidence of an adverse health effect if exposure to the pollutant occurs.[6,7]

Dose-response assessment defines the relationship between the degree of exposure (or amount of dose) observed in animal or human studies and the magnitude of the observed adverse health effects. This usually includes a quantitative measure of adverse health effects for a range of doses. For carcinogens, dose-response data are used to calculate quantitative estimates of the increased risk of developing cancer per unit of exposure (e.g., inhalation unit risk estimates [IUREs]). For noncarcinogens, dose-response data are used to calculate "safe" levels (e.g., inhalation reference concentrations [RfCs]).

Exposure assessment estimates the extent of pollutant exposure via various routes (e.g., oral, inhalation, dermal) to individuals or populations. For air pollutants, this often involves the application of exposure models.

Risk characterization is the integration of the hazard identification, dose-response, and exposure assessments to describe the nature, and often to estimate the magnitude, of the health risk in a given population.[6] The risk characterization also includes a presentation of the qualitative and quantitative uncertainties in the assessment.

Risk assessment should not be confused with risk management. Risk management is the process of developing and weighing policy options and selecting appropriate actions. Risk management integrates the results of the risk assessment with other information such as economic, engineering, political and social factors and uses this integrated information to make policy and regulatory decisions.

4.1.2 U.S. EPA Risk Assessment Guidelines

Several publications were used to establish the methods for this risk assessment. The methods generally follow the risk assessment guidelines published by the U.S. EPA in 1986.[7] Other sources consulted for preparation of this assessment are discussed briefly below.

4.1.3 Risk Assessment Council (RAC) Guidance

The RAC of the EPA evaluated EPA risk assessment practices in 1992 and recommended guidance on risk assessment focusing on the risk assessment-risk management interface and risk characterization.[8] Major elements relevant to this study are summarized below:

- Complete presentation of risk is needed including discussions of uncertainty and statements of confidence about data and methods used. The assessment should clearly identify all assumptions, their rationale, and the effect of reasonable alternative assumptions on the conclusions and estimates.

- Assessors should use consistent and comparable risk descriptors. For example, assessments should include: descriptions of risk to individuals and to populations, presentations of central tendency and worst-case portions of the range of risk, and, if feasible, highly exposed or highly susceptible groups should be identified.

4.1.4 NAS Report *Science and Judgement in Risk Assessment*

In 1994, the National Research Council (NRC) of the NAS released a report *Science and Judgement in Risk Assessment*, which contains a critique of existing EPA methods and several recommendations for improvements.[3] A few of the recommendations important for the utility assessment are described briefly here.

The NRC stated that default options are a reasonable way to deal with uncertainty about underlying mechanisms in selecting methods and models. However, default options should be explicitly identified, and the basis explained fully.

The NRC believes the EPA should undertake an iterative approach to risk assessment. An iterative approach starts with relatively inexpensive screening techniques to estimate chemicals without health concerns followed by more resource-intensive levels of data gathering and model application.

It is appropriate to use "bounding" estimates for screening assessments to determine whether further levels of analysis are necessary. For example, if there are no health impacts even in a worst-case assumption scenario, then it may not be necessary or desirable to proceed with further analysis.

4.1.5 SPC's Guidance for Risk Characterization

In 1995, the SPC of the EPA provided guidance for characterizing risk.[4] A few points are briefly summarized here.

Risk assessors should be sensitive to distinctions between risk assessment and risk management. Risk assessors are charged with (1) generating a credible, objective, realistic, and scientifically balanced analysis; (2) presenting information on hazard, dose response, exposure, and risk; and (3) clearly describing confidence, strengths, uncertainties, and assumptions.

The risk characterization should include qualitative and quantitative descriptions of risk. Both high-end and central tendency descriptors should be used to convey the variability in risk levels experienced by different individuals in the population. The assessment should identify and discuss important strengths, limitations and uncertainties, and degree of confidence in the estimates and conclusions. The assessment should also include discussions of data quality and variability.

4.2 GENERAL APPROACH AND METHODS FOR THE UTILITY HEALTH HAZARD RISK ASSESSMENT

Emissions of HAPs can be a threat to public health if sufficient exposure occurs. For many HAPs, exposure through inhalation is the major concern. However, humans can also be exposed to HAPs via indirect pathways (multipathway) such as through ingestion or dermal exposure to HAPs through other media such as food, water, or soil that has been contaminated by the deposition of the HAPs. Indirect exposure is primarily a concern for HAPs that are persistent and bioaccumulate.

To assess the public health concerns due to emissions of HAPs from utilities, the EPA conducted inhalation and multipathway exposure and risk analyses. First, to be consistent with the NAS recommendations, a screening assessment was conducted on 67 HAPs potentially emitted from utilities to determine priority HAPs. After the screening assessment was completed, further analyses were conducted for the priority HAPs. In addition to the inhalation risk assessment, the EPA conducted a multipathway analysis of radionuclides, a long-range transport modeling analysis for mercury and arsenic, and a limited qualitative assessment of the potential hazards due to multipathway exposure to a few other persistent, bioaccumulative HAPs.

Chapter 5 presents the screening assessment. Chapter 6 presents the inhalation risk assessment for 14 priority HAPs. Chapter 7 presents an assessment of mercury. Chapter 8 presents an evaluation for dioxins, lead, cadmium, and arsenic, and chapter 9 presents the assessment for radionuclides.

4.3 HEALTH EFFECTS DATA: HAZARD IDENTIFICATION AND DOSE RESPONSE

Health effects data include qualitative and quantitative data on hazard identification and dose response. These data are closely related and evaluated concurrently in toxicologic studies. Therefore, this section of the report includes summary discussions of both. For detailed information on health effects data for seven of the priority HAPs emitted from utilities, the reader is referred to Appendix E.

Most of the health effects data used were obtained from the EPA's Integrated Risk Information System (IRIS). The IRIS is an online database maintained by the EPA, containing chemical-specific health risk information. The data provided in IRIS have been reviewed by EPA work groups and represent Agency consensus.[9] Primarily, EPA-verified risk values were used in this study. However, for HAPs without IRIS data, health data from other toxicologic data sources were utilized. If other data sources were used, they are indicated by footnotes in tables or discussed in the text.

4.3.1 Hazard Identification for Carcinogens

Animal and human cancer studies are evaluated to determine the likelihood that a chemical causes cancer in humans. The evidence for each of these is determined to be sufficient, inadequate, or limited. Other types of experimental evidence (e.g., in vitro genotoxicity studies) may be used to support the epidemiological or animal bioassay results.[7,10] The EPA uses a weight-of-evidence, three-step procedure to classify the likelihood that the chemical causes cancer in humans. In the first step, the evidence is characterized separately for human studies and for animal studies. The human studies are examined considering the validity and representativeness of the populations studied, any possible confounding factors, and the statistical significance of the results. The animal studies are evaluated to decide whether biologically significant responses have occurred and whether the responses are statistically significant. Second, the human and animal evidence is combined into an overall classification. In the third step, the classification is adjusted upward or downward, based on an analysis of other supporting evidence. Supporting evidence includes structure-activity relationships (i.e., the structural similarity of a chemical to another chemical with known carcinogenic potential), studies on the metabolism and pharmacokinetics of a chemical, and short-term genetic toxicity tests.[6,7] The result is that each chemical is placed into one of the five categories listed in Table 4-1.

4.3.2 General Discussion of Dose Response

The NAS[5] defined dose-response assessment as:

> "...the process of characterizing the relation between dose of a chemical administered or received and the incidence of adverse health effects in exposed populations and estimating the incidence of the effect as a function of human exposure to the agent. It takes account of intensity of exposure, age pattern of exposure, and possibly other variables that may affect response, such as sex, lifestyle, and other modifying factors."

In general, as dose increases, so does the probability that an adverse effect will occur. Critical to a dose-response assessment is the basic assumption that thresholds exist for particular compounds and particular health effects, and thus doses below the threshold would not result in adverse effects. Thresholds may exist if the body has the ability to detoxify or compensate for exposures to pollutants or if multiple numbers of cells perform the same function. When doses increase to the point that the body can no longer accommodate or compensate for the exposure to pollutants, adverse health effects can be observed and the likelihood of effect increases with increased dose. For "nonthreshold" toxicants, it is assumed that

Table 4-1. Weight-of-Evidence (WOE) Classification

Group	Description
A	Known Human Carcinogen
B1	Probable Human Carcinogen, Limited Human Data Are Available
B2	Probable Human Carcinogen, Sufficient Evidence in Animals and Inadequate or No Evidence in Humans
C	Possible Human Carcinogen
D	Not Classifiable as to Human Carcinogenicity
E	Evidence of Noncarcinogenicity for Humans

there is no threshold concentration or dose below which health effects do not appear and that any exposure means an increase in risk.

The EPA assumes that cancer is a nonthreshold disease, that is, any exposure to a chemical carcinogen, no matter how low, contributes to an increased lifetime probability (i.e., risk) of developing cancer. In contrast, chemicals causing health effects other than cancer are typically defined as having a threshold exposure concentration or dose below which adverse health effects are not expected to occur. The threshold concept influences the way in which dose-response modeling or dose-response assessment is done. Assessments of carcinogens and noncarcinogens are conducted separately and are based on different assumptions and methods. Information for carcinogens and noncarcinogens is discussed separately in this section.

4.3.3 Dose-Response Evaluation for Carcinogens

For chemicals that have been classified as carcinogens (WOE = A, B, or C), the dose-response data are evaluated, and, if data are adequate, then the EPA calculates quantitative estimates of the increased risk of developing cancer per unit of exposure. For example, for air pollutants, an IURE is calculated. The IURE for a pollutant is the estimated increased risk (upper limit probability) of a person developing cancer from breathing air containing a concentration of 1 microgram of the pollutant per cubic meter ($\mu g/m^3$) of air for a lifetime (70 years). The EPA also calculates oral unit risk estimates for assessing cancer risks from ingestion exposure.[6,7,9]

Since risks at low exposure levels cannot be measured directly, a number of mathematical models have been developed to extrapolate from high to low dose to calculate the unit risk estimates. The linearized multistage model, which is the default model generally used by the EPA, leads to a plausible upper limit to the risk that is consistent with some proposed mechanisms of carcinogenesis. The true risk is unlikely to exceed the value predicted by the linear multistage model and may be lower; as low as zero is a possibility.[7] For most HAPs included in this assessment, the EPA has used the linear multistage low-dose extrapolation model. However, there are a few important HAPs, with WOE ratings of "A", (e.g., chromium VI, arsenic) for which the EPA used other linear extrapolation models. The IUREs for these HAPs are also considered upper limit estimates of the risks at low concentration because of the use of linear high- to low-dose extrapolation and other factors. Table 5-1 presents a summary of the EPA-verified cancer health effects data for HAPs emitted from utilities. Table 5-1 also contains some health effects data that are not EPA-verified.

The EPA assumes that, for carcinogenesis, no threshold for dose-response relationships exists or that, if one does exist, it is very low and cannot be reliably identified. As a result, any increase in dose is associated with an increase in risk of developing cancer. Although a number of theories exist to explain the process of carcinogenesis, the multistage process is the most widely accepted. The multistage process consists of three distinct stages: initiation, promotion, and progression.[11] One reason the multistage process is so well accepted is that it has been demonstrated experimentally for a number of carcinogens and has been shown to adequately describe carcinogenesis in the cells of some animal tissues, including the skin, lung, liver, and bladder.[12] Individual carcinogens can affect one or more of these stages.

The method for deriving IUREs based on animal data is different than the method used for deriving IUREs based on human data. When animal data are used, EPA typically determines the 95th percentile confidence limit of the dose-response curve, then extrapolates linearly down to zero. When human data are used, EPA typically determines the "maximum likelihood" estimate of the dose-response curve, then extrapolates linearly down to zero.

There are factors involved with the human occupational data that may result in high- or low-biasing effects, including uncertainties in the estimation of individual exposures and the assumption that the susceptibility of the exposed workers in the epidemiology studies is equivalent to the susceptibility of the general population.

4.3.4 Long-term Noncancer Health Effects Data

Pollutants can cause a variety of noncancer effects including neurological, reproductive, developmental, and immunological toxicity. Noncancer effects can be reversible or irreversible and can occur following acute (short-term) exposure or chronic (long-term) exposure.[6]

Subchronic and chronic animal and human studies are evaluated to determine potential adverse noncancer effects and the estimated doses or exposure concentrations that cause those effects. If data are sufficient, the EPA calculates an inhalation RfC, which is an estimate (with uncertainty spanning perhaps an order of magnitude) of the daily inhalation exposure of the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime. The RfC is derived based on the assumption that thresholds exist for certain toxic effects such as cellular necrosis but may not exist for other toxic effects such as carcinogenicity. The RfC is calculated as follows: EPA reviews many human and/or animal studies to determine the highest dose level tested at which the critical adverse effect does not occur—i.e., the no observed adverse effect level (NOAEL)—or the lowest dose level at which the critical adverse effect is observed, the lowest observed adverse effect level (LOAEL). The NOAEL from an animal study is adjusted for exposure duration and respiratory tract differences between animals and humans. EPA then applies uncertainty factors to adjust for the uncertainties in extrapolating from animal data to humans (10), and for protecting sensitive subpopulations (10). Also, a modifying factor is applied to reflect professional judgment of the entire database. The inhalation RfC considers toxic effects for both the respiratory system (portal-of-entry) and for effects peripheral to the respiratory system (extrarespiratory effects). Exposures below the RfC are not likely to be associated with adverse noncancer health effects including respiratory, neurologic, reproductive, developmental, and other effects. In this report the RfC is expressed in micrograms of pollutant per cubic meter of air ($\mu g/m^3$). The EPA also calculates a similar value, called the reference dose for assessing ingestion exposure and risks. The RfD is expressed in units of milligrams/kilograms/day (mg/kg/d). Doses below the RfD are not expected to result in adverse noncancer health effects. The EPA considers reproductive and developmental effects when establishing RfCs and RfDs. If data are absent, an uncertainty factor is often added to adjust the RfC or RfD downward. Doses or concentrations above the RfD or RfC do not necessarily indicate that adverse health effects will occur.[9,13,14] As the amount and frequency of exposures exceeding the RfC or RfD increases, the possible occurrence of adverse effects in the human population also increases. When exceedances of the RfD or RfC are predicted, the data on exposure and health effects should be evaluated further to determine the data quality, uncertainties, degree of exceedance, and the likelihood, frequency, and severity of potential adverse effects. Evaluating this information helps to characterize the public health concerns.

The EPA ranks each RfC as either low, medium, or high in three areas: (1) confidence in the study on which the RfC was based, (2) confidence in the database, and (3) overall confidence in the RfC.[13,14] Table 5-2 presents the EPA-verified RfCs for HAPs listed in chapter 3 and other health effects information.

4.3.5 Short-term Noncancer Health Effects Data

Short-term exposure to HAPs can also cause adverse noncancer health effects. There are no EPA-verified acute health effects benchmarks available for the priority HAPs. However, reference exposure levels (RELs) for acute exposures were obtained from the California Air Resources Board's *Risk Assessment Guidelines for the Hot Spots Program.*[15]

4.3.6 Summary of Health Effects Data Sources

As mentioned, IRIS was the primary source of information on health effects. However, other sources were also consulted such as the Toxicological Profiles published by the Agency for Toxic Substances and Disease Registry (ATSDR), the monographs published by the International Agency for Research on Cancer (IARC), and various EPA documents.

4.4 METHODOLOGY FOR ESTIMATING INHALATION EXPOSURE FOR LOCAL ANALYSIS

Exposure assessment is the determination or estimation (qualitative or quantitative) of the magnitude, frequency, duration, and route of exposure. An exposure assessment has four major components:

- Emissions characterization
- Environmental fate and transport

- Characterization of the study population
- Exposure calculation.

This section summarizes the local inhalation exposure assessment approach including discussions of the Human Exposure Model (HEM), data, default options, and limitations. The long-range transport exposure analysis is explained in section 6.6.

4.4.1 Emissions Characterization

The emissions data gathered from 52 utility units (described in chapter 3) were used as the basis for estimating emissions of HAPs from 684 utility plants in the U.S. As described in chapter 3, a computer program was developed to estimate emissions from each utility unit based on boiler type, electric output, fuel type, and APCDs. This resulted in average annual emissions estimates for each HAP from all 684 utility plants.

4.4.2 Atmospheric Fate and Transport

To arrive at long-term (annual) average ambient air concentrations within 50 km of the facility, air dispersion modeling was conducted using the HEM, which utilizes the Industrial Source Complex Long-Term, version 2 (ISCLT2) dispersion model (see Appendix F for details). The ISCLT2 was used to estimate atmospheric fate and transport of HAPs from the point of emission to the location of exposure. The ISCLT2 uses emissions source characterizations and meteorological data to estimate the transport and dispersion of HAPs in the atmosphere and to estimate the ambient HAP concentrations within 50 km of each source (i.e., local analysis). Plant-specific parameters needed for modeling (e.g., stack heights, stack temperature, stack exit velocity, stack diameter, latitude, and longitude) were obtained from the UDI/EEI database. Emissions estimates, also needed as input to the model, were obtained from the analysis described in chapter 3. Long-range transport (beyond 50 km) was also addressed and is described in chapters 6 and 7.

The ISCLT2 uses meteorological data in the form of stability arrays, or STAR data summaries. The STAR summaries contain joint frequencies of occurrence of windspeed, wind direction, and atmospheric stability. These factors are combined into an overall frequency distribution. The meteorological database is based on hourly surface observations obtained mostly from the Office of Air Quality Planning and Standard's (OAQPS's) Technology Transfer Network (TTN). The TTN contains data files of surface observations from National Weather Service locations (primarily airports) across the U.S. and its Territories. The STAR summaries combine several available years (typically 6 years) into one long-term estimate of the location's dispersion characteristics. In all cases, the meteorological data from the site (out of a possible 349 sites) nearest each plant were used in modeling each utility plant's emissions. In addition, there are two smaller databases that provide average mixing height and temperature by atmospheric stability class. Every STAR site has a matching temperature database. However, the mixing height database contains information for only 74 sites; although, as with the STAR summaries, the nearest site is always selected.

4.4.3 Characterization of Study Population

Census data from 1990, which are the most current and comprehensive data available, were used in estimating population exposures. The data were available on the "block" level, containing 6.9 million records. For each plant, all census blocks within 50 km were identified and used to estimate local exposure.

4.4.4 Exposure Calculations

Exposure is calculated by multiplying the population (i.e., number of people) by the estimated air concentration to which that population is exposed. The HEM exposure algorithms pair the air concentration estimates produced by ISCLT2 with the census information contained in the population database.

Within the HEM, the ISCLT2 calculates air concentrations at numerous grid points within 50 km of each source. For this study, grid points were placed around the source along 16 radials, spaced every 22.5 degrees, at distances of 0.2, 0.5, 1.0, 2.0, 5.0, 10.0, 20.0, 30.0, 40.0, and 50.0 km from the source, for a total of 160 grid points (which is the default setting). Except for receptors that are very close to the stack, HEM calculates the air concentration at the population centroid (the population center of the census block) by interpolating between the values at the receptors surrounding the centroid. Exposures were calculated by multiplying the number of people living within a census block and the modeled air concentration at the centroid of the census block. When the population blocks are within 0.5 km of the plant, the population is distributed to each grid point within 0.5 km to more realistically account for actual locations of people. (In this region, the areas associated with census blocks are larger than the sections in the polar grid and thus it is logical to spread people out by assigning people to grid points rather than block centroids). Exposure is calculated by multiplying the grid point concentrations by the number of people assigned to the grid point. For a more detailed description of the HEM, see Appendix F.

4.5 METHODOLOGY FOR ESTIMATING QUANTITATIVE INHALATION RISKS

Numerical estimates can be calculated for two categories of adverse health effects:

- Risk of developing cancer
- Likelihood of noncancer health effects.

To derive statements of risk, quantitative information on exposure was combined with information on dose response.

4.5.1 Estimating Cancer Inhalation Risks

For this analysis, the HEM calculated the cancer risk from inhalation using standard EPA risk equations and assumptions. The risk equation, which is a linear, nonthreshold model, defines the exposure-response relationship. The estimate of the inhalation exposure concentration (µg/m^3) is multiplied by the IURE to calculate risks for exposed persons who are assumed to be exposed on average to the modeled ambient concentration of the carcinogen for a lifetime. Risks are generally expressed as either individual risk or population risk. By the nature of the exposure and risk assessment models, the estimated risks are expressions of the risks associated only with exposure to utility emissions.

4.5.2 Individual Risk

Individual risk is commonly used to express risk and is defined as the increased probability that an exposed individual would develop cancer following exposure to a pollutant. Individual cancer risks can be calculated by multiplying the estimated long-term ambient air concentration (µg/m^3) of a HAP (i.e., exposure estimate) by the IURE. The IURE generally represents an upper bound estimate of the increased risk of developing cancer for an individual exposed continuously for a lifetime (70 years) to a specific concentration (e.g., 1 µg/m^3) of a pollutant in the air.

If the highest modeled ambient air concentration occurs in an area (e.g., census block) where no people are known to reside, it is assumed that, theoretically, a person could be exposed to this concentration (e.g., someone could move to this location); therefore, the EPA calculates an estimated risk assuming that someone is exposed to the concentration. The risk calculated in this situation is termed the MEI risk. Hence, the MEI risk is the estimated risk to a theoretical individual exposed to the highest estimated long-term ambient concentration associated with an emission source. If the highest modeled ambient air concentration occurs in an area where people are known to reside, the EPA again calculates an estimated risk. The risk calculated in this situation is termed the Maximum Individual Risk (MIR) risk. Hence, the MIR represents the increased cancer risk to an individual exposed at the highest estimated long-term ambient concentration in the area in which people are assumed to reside. In this report, both the MEI risk and the MIR are calculated assuming that a person is exposed to the modeled long-term ambient pollutant concentration for a lifetime (i.e., 70 years). By the nature of the assumption, the MEI estimates must be equal to or greater than the MIR. For this study, the MEI and MIRs were either the same or very similar. For oil-fired utilities, the MEI and MIR were exactly the same because the highest concentration occurred in an area where people are known to reside. For coal-fired utilities, the MEI risk was slightly higher than the MIR risk. For example, the MEI risk for arsenic from coal-fired utilities was 3 x 10^{-6}, and the MIR for arsenic from coal-fired utilities was 2 x 10^{-6}.

4.5.3 Population Cancer Risk

Population risk is an estimate that applies to the entire population within the given area of analysis. Two population risk descriptors are:

- The probabilistic number of health effects cases estimated in the population of interest over a specified time period (e.g., number of cancer cases per year) or cancer incidence
- The percentage of the population, or the number of persons, exposed above a specified level of lifetime risk (e.g., 10^{-6}).

Each modeled ambient HAP concentration level is multiplied by the estimated number of people exposed to that level and by the IURE, providing an estimate of cancer incidence for a 70-year lifetime exposure. These risk values are summed to give aggregate risks for the population within the study area, i.e., the total estimated excess cancer cases in the exposed population. This lifetime risk estimate is divided by 70 years to calculate annual incidence in units of cancer cases per year.

4.5.4 Distribution of Individual Risk within a Population

The HEM estimates a distribution of individual risks throughout the exposed population. The risk distribution presents an estimate of the number of people exposed to various levels of risk, e.g., the number of people who are exposed to individual risk levels above 10^{-7}, 10^{-6}, 10^{-5}, or 10^{-4}.

4.5.5 Aggregate Inhalation Cancer Risk

The HEM calculates risk from individual HAPs and does not calculate total risk for the mixture of pollutants from a single source. To calculate total risks from the emissions of the mixture of HAPs, the MIR and cancer incidence attributed to each individual HAP was identified for each power plant. The MIR and cancer incidence were then added across HAPs for each plant. This addition is consistent with the EPA's default procedure for assessing mixtures.[16] The highest total MIR across all plants was identified and the total cancer incidence was summed across all plants. Given the structure of the HEM output, it is not possible to calculate total risk (summed across all carcinogens) for the entire exposed population. Therefore, the population distribution by total risk is not presented in this report.

4.5.6 Estimating Noncancer Inhalation Risks

The concepts of individual and population risks also apply to noncancer risks. However, there are differences between how cancer risks and noncancer risks are estimated. The noncancer result is not a measure of risk, but rather indicates the possibility for an adverse health effect. To assess potential noncancer health effects, the EPA evaluated exposure to the individuals predicted to receive the maximum modeled concentration.

Unlike cancer risk characterization, noncancer risks are not expressed as a probability of an individual suffering an adverse effect. Instead, the potential for noncancer effects is evaluated by comparing an exposure concentration over a specified period of time (e.g., a lifetime) with a toxicity benchmark called the inhalation RfC.

4.5.7 Inhalation Hazard Quotient (HQ)

The HQ, a ratio of exposure (E) to the RfC, is commonly calculated. The HQ indicates whether the concentration or dose to an individual has the potential to cause an adverse effect. HQ values at or less than 1 imply that exposures are at or below the RfC and not likely to cause adverse effects. An HQ value exceeding 1 implies that the RfC is exceeded and the likelihood of adverse effects increases as the amount and frequency of exposures exceeding the RfC increase.

In risk assessments in which RfCs are used and exposures are approaching or exceeding the RfC, information about its derivation, data, assumptions, and uncertainties should be evaluated along with the HQ values to determine the concerns for public health and likelihood for adverse effects. For example, the critical health effect associated with the RfC, the type of epidemiologic or toxicologic studies considered, the degree of exceedance, the uncertainty and modifying factors used in deriving the RfC, and the uncertainties and degree of confidence in the RfC should be evaluated to characterize the potential concerns for public health.

4.5.8 Total Risk for Noncancer Effects

The Hazard Index (HI) is used to address total risks from multiple chemicals and is the sum of HQ values for individual pollutants to which an individual is exposed. As an initial screen, the individual HQ values are added within a power plant and the highest HI across all plants is identified. Similar to the HQ, hazard indexes at or less than 1 indicate that adverse noncancer health effects are not expected to occur. As the HI increases, approaching or exceeding 1 (unity), concern for the potential hazard of the chemical mixture increases. If the HI exceeds unity, the mixture has exceeded the equivalent of the RfC for the mixture. The HI should not be interpreted as a probability of risk nor as a strict delineation of safe and unsafe levels.[7,17]

The HI approach assumes that simultaneous exposures to several chemicals (even at subthreshold levels) could, in combination, result in an adverse health effect. Even if no single compound exceeds its RfC, the HI for the overall mixture may exceed 1. Should the HI exceed unity, the HI should be reevaluated with HQ values summed only for noncarcinogens with similar target organs based on U.S. EPA Risk Assessment Guidelines[7] and the assumption that each target organ has a threshold that must be exceeded before adverse effects can occur and that toxicity among target organs is independent. In addition, the mixture of pollutants should be assessed for potential synergistic or antagonistic effect if the HI is near or at unity and if sufficient data are available. The EPA has produced a database called Mixtox[18] that contains information regarding potential effects of mixtures of pollutants. If the HI is at or near unity, then Mixtox can be used to evaluate the mixture. For this study, only the maximum HI values associated with a power plant were calculated.

4.5.9 Direct Inhalation Exposure and Risk Default Options

The EPA's risk assessment guidelines contain a number of "default options." These options are used in the absence of convincing scientific knowledge about which ones of several competing models and theories are correct. Several of the defaults are generally conservative (i.e., they represent a choice that, although scientifically plausible given the existing uncertainty, is more likely to result in overestimating rather than underestimating risk). However, some of the default options are not necessarily conservative, and may actually lead to an underestimate of the risks. A number of default options were incorporated into the HEM inhalation exposure assessment. These include:

- The HEM only estimates exposure within 50 km of each plant. Exposure due to long-range transport is not considered in the HEM analysis. (Long-range transport is analyzed in section 6.6.)
- Dispersion occurs as predicted by a Gaussian plume model in flat terrain.
- Exposure is based on centroids of census blocks since locations of actual residences are not in the database.
- For MEIs and MIRs, people are assumed to reside at the same location for their entire lifetimes (assumed to be 70 years)
- Indoor concentrations are assumed to be the same as outdoor concentrations.
- The average lifetime exposure is based on all exposures occurring at home; exposure due to movement between home, school, work, etc., is assumed to be, on average, equal to exposure at home.
- Utilities emit HAPs at rates predicted by the emissions factor program described in chapter 3 at the same level for a 70-year lifetime of exposure. Only stack emissions were considered. Fugitive dust from coal piles was not included.
- The HEM only estimates exposure due to direct inhalation. The HEM does not estimate exposure from indirect pathways (i.e., multipathway exposure).
- The population database is not adjusted for population growth.
- Varying exposures as a result of differences such as age, sex, health status, and activity are not considered.

The impact of using these default values is evaluated in sections 6.12 and 6.13 and Appendix G.

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5.0 SCREENING RISK ASSESSMENT TO DETERMINE PRIORITY HAPS

The EPA conducted a screening assessment for the 67 HAPs identified in the emissions database to determine priority HAPs for further analyses. The HAPs were prioritized based on their potential to pose hazards and risks through inhalation or multipathway exposure.

5.1 MODELING DESCRIPTION

To screen for inhalation risks, the EPA conducted exposure modeling, using the Human Exposure Model (HEM), to estimate direct inhalation exposure within 50 km of each utility for 66 of the 67 HAPs identified in the emissions database. Radionuclides could not be modeled adequately using the HEM. Therefore, radionuclides were screened based on previous studies conducted in the 1980s.[1] The screening for radionuclides is discussed in section 5.6. A general description of the HEM, input data, and default options is presented in chapter 4. The HEM is also described in detail in Appendix F.

Using the average annual emission estimates (discussed in chapter 3) for each HAP for all 684 plants along with site-specific parameters as input (e.g., location, stack height, stack exit velocity, stack temperature, and population data), the HEM was utilized to estimate inhalation exposures for the maximally exposed individuals (MEIs). The exposure estimates were then combined with health effects data to estimate risks due to inhalation exposure for the MEIs. Based on these MEI risks, priority HAPs were selected for inhalation risk assessment. As described below, the screening for multipathway assessment was based on factors such as persistence of the HAP, bioaccumulation potential, and toxicity by ingestion exposure.

5.2 SCREENING CRITERIA

First, HAPs were screened based on cancer risk effects due to inhalation exposure. The maximum modeled air concentrations for each HAP were multiplied by the IUREs to estimate upper limit increased lifetime cancer risks to the MEIs. If the highest MEI risk was greater than 1 in 10 million (i.e., 1×10^{-7}), the HAP was considered a priority for further analysis.

Second, HAPs were screened for noncancer effects due to long-term (chronic) inhalation exposure. The maximum modeled air concentrations were compared to RfCs. Hazard quotients (HQ) were calculated by dividing the maximum modeled concentrations by the RfCs. If the highest HQ was greater than 0.1, then the HAP was considered a priority for further analysis.

Third, in addition to the inhalation screening assessment, HAPs were prioritized for potential multipathway exposure and risks. The nonradionuclide HAPs were prioritized for multipathway assessment based on persistence of the HAP, tendency to bioaccumulate, toxicity by ingestion exposure route, and quantity of emissions. This resulted in five nonradionuclide HAPs being identified as a priority for multipathway assessment.

After HAPs were prioritized based on the above criteria, additional HAPs were prioritized because of potential concerns for noncancer effects due to short-term inhalation exposures. Also, radionuclides were identified as a priority for multipathway assessment based on results of previous studies. The following sections present more details of each of the screening analyses.

5.3 INHALATION SCREENING ASSESSMENT FOR COAL-FIRED UTILITIES

Emissions data were available for 66 nonradionuclide HAPs from coal-fired utilities. Cancer and noncancer quantitative health effects data were available from IRIS and various EPA documents for 50 of the 66 nonradionuclide HAPs. Table 5-1 presents the results for HAPs that are considered carcinogens and for which a quantitative cancer risk estimate was available. Table 5-2 presents results for the noncancer screening assessment using EPA-verified RfCs. Table 5-3 presents the HAPs for which no EPA-approved quantitative health data were available for assessment.

5.4 INHALATION SCREENING ASSESSMENT FOR OIL- AND GAS-FIRED UTILITIES

Emissions data were available for 28 HAPs from oil-fired utilities. EPA cancer and noncancer quantitative health effects data were available for 22 of the 28 HAPs. Table 5-4 presents the results for HAPs from oil-fired utilities that are considered carcinogens and for which quantitative cancer risk estimates were available. Table 5-5 presents results for the noncancer screening assessment for HAPs from oil-fired utilities for which EPA-approved RfCs were available. Table 5-6 presents HAPs from oil-fired utilities for which no EPA-verified quantitative health data were available for assessment.

For gas-fired utilities, emissions data were available for 14 HAPs. Table 5-7 presents the screening results for gas utilities.

Table 5-1. Inhalation Screening Assessment for Carcinogenic HAPs from Coal-Fired Utilities for Which Quantitative Cancer Risk Estimates Were Available

Hazardous air pollutant	**Highest MEI conc.[a] ($\mu g/m^3$)**	**EPA WOE[b]**	**IURE[c] per $\mu g/m^3$**	**MEI cancer risk[d]**	**Primary type of cancer assoc. w/ inhalation[e]**	**MEI cancer risk > 10^{-7}**
Arsenic compounds	0.0014	A	0.0043	6×10^{-6}	Lung	Yes
Chromium (11 percent VI)[f]	0.0023	A	0.0016[f]	4×10^{-6}	Lung	Yes
Beryllium	0.00025	B2	0.0024	6×10^{-7}	Lung	Yes
Cadmium	0.00009	B2	0.0018	2×10^{-7}	Lung	Yes
Nickel compounds[g]	0.0027	A[g]	0.00048	1×10^{-6}	Lung & nasal	Yes
Dioxins[h]	2×10^{-9}	B2	30.0[h]	7×10^{-8}	Tongue, lung, nasal, liver	No
PAHs[i]	0.00002	B2	0.0021[i]	4×10^{-8}	Lung (BAP)	No
Naphthalene	0.00009	C	4×10^{-6}	4×10^{-10}		No
Hexaclorobenzene	9×10^{-6}	B2	0.00046	4×10^{-9}	NA	No
Carbon tetrachloride	0.00038	B2	0.000015	6×10^{-9}	Liver	No
Quinoline[j]	0.000006	C	0.0035[j]	2×10^{-8}	NA	No
Vinylidene chloride	0.0011	C	5×10^{-5}	6×10^{-8}		No
Formaldehyde	0.00047	B1	1×10^{-5}	6×10^{-9}	Nasal, lung	No
n-Nitrosodimethylamine[k]	0.00008	B1	0.014	1×10^{-6}	Liver & other	Yes[k]
1,1,2-Trichloroethane	0.00054	C	2×10^{-5}	9×10^{-9}	NA	No
Acetaldehyde	0.00078	B2	2×10^{-6}	2×10^{-9}	Nose & larynx	No
Benzene	0.00029	A	8×10^{-6}	2×10^{-9}	Leukemia	No
Benzyl chloride	6×10^{-7}	B2	5×10^{-5}	3×10^{-11}	NA	No
Bis(2-e-h)phthalate[j]	0.00047	B2	4×10^{-6}[j]	2×10^{-9}	NA	No
Bromoform	0.00077	B2	1×10^{-6}	9×10^{-10}	NA	No
Chloroform	0.00037	B2	2×10^{-5}	9×10^{-9}	Kidney & liver	No
Ethylene dichloride	0.00036	B2	3×10^{-5}	9×10^{-9}		No
Isophorone[j]	0.003	C	3×10^{-7}[j]	9×10^{-10}		No
Methyl chloride[j]	0.0007	C	2×10^{-6}[j]	1×10^{-9}	Kidney	No
Methylene chloride	0.0015	B2	5×10^{-7}	7×10^{-10}	Liver & lung	No

Table 5-1. (continued)

Hazardous air pollutant	Highest MEI conc.[a] ($\mu g/m^3$)	EPA WOE[b]	IURE[c] per $\mu g/m^3$	MEI cancer risk[d]	Primary type of cancer assoc. w/ inhalation[e]	MEI cancer risk > 10^{-7}
Trichloroethylene[j]	0.00036	B2/C	2×10^{-6} [j]	6×10^{-10}	Lung, liver, & testicular	No
Pentachlorophenol[j]	1×10^{-6}	B2	3×10^{-5} [j]	3×10^{-11}	NA	No
Tetrachloroethylene[j]	0.00036	B2/C	6×10^{-7} [j]	2×10^{-10}	Liver	No

NA = Not available.

[a] This is the highest estimated ambient concentration (annual average) due to emissions from the one highest risk coal-fired utility based on HEM modeling of all coal-fired utilities in the U.S. The MEI = the Maximally Exposed Individual.

[b] WOE = Weight of evidence, for carcinogenicity. See section 4.3.1 and Table 4-1. for explanation of WOE.

[c] IURE = Inhalation Unit Risk Estimate. The IURE is the estimated increased risk of cancer from breathing 1 μg of pollutant per cubic meter of air for 70 years. The IUREs were obtained from EPA's Integrated Risk Information System (IRIS),[2] unless indicated otherwise by footnotes.

[d] This is the estimated increased lifetime cancer risk to the highest MEI due to inhalation exposure.

[e] This column presents the type of cancer observed in experimental animal studies or human studies. For more details see Appendix E and/or various references.

[f] For coal-fired utilities it is assumed that 11 percent of chromium is chromium VI and that the remainder is chromium III. For oil-fired utilities it is assumed that 18 percent of chromium is chromium VI. This is based on the limited speciation data described in Appendix H. It is assumed that the carcinogenic effects are caused only by the Cr VI fraction. The IURE was calculated by multiplying the IURE on IRIS for Cr VI (1.2×10^{-2}) by 0.11 (11 percent).

[g] For this screening assessment all nickel was assumed to be as carcinogenic as nickel subsulfide. This assumption is considered an "upper bound" conservative assumption. Nickel risk uncertainty issues are discussed more thoroughly in Chapter 6.

[h] The emissions were estimated using the toxic equivalency (TEQ) approach described in the draft EPA Dioxin Reassessment Report.[3] Exposure was estimated by modeling the TEQ emissions with HEM. The IURE is for 2,3,7,8-tetrachlorodibenzodioxin (TCDD) and was obtained from the draft EPA Dioxin Reassessment Report.

[i] To estimate the potential risk from polycyclic aromatic hydrocarbon (PAH) emissions, first the EPA summed the emissions of the 7 PAHs that are classified as B2 carcinogens (WOE = B2).[4] (These are listed in Appendix H). Second, exposure was estimated by using the HEM and modeling the sum of the 7 PAHs. Third, the estimated exposure to the 7 B2 PAHs were multiplied by the IURE for benzo[a]pyrene (BAP) (2.1×10^{-3}). However, this IURE has not been verified by the EPA and has not been peer reviewed. It is an interim value with significant uncertainties and is intended for screening assessment only. This IURE was calculated by converting the oral unit risk estimate of 2.1×10^{-4} per $\mu g/L$ to inhalation units.[5] The conversion assumes equal absorption and metabolism and assumes equal risk from the different routes of exposure, which may not be the case.

[j] The IUREs for these HAPs are not EPA-verified and are intended for screening assessment only. Readers must exercise caution interpreting the results using these numbers. These IUREs were obtained from *Documentation of De Minimis Emission Rates - Proposed 40 CFR Part 63, Subpart B, Background Document.*[5] This document was developed to support the proposed rulemaking pursuant to 112(g) of the Clean Air Act (*Federal Register*, Volume 59, No. 63, April 1, 1994). There are significant uncertainties associated with these IUREs. They are not EPA-verified. They are interim screening values intended for the screening assessment only. For further discussion of the health data and uncertainties, see the de minimis document cited above.

[k] The risk estimate for n-nitrosodimethylamine is highly suspect and uncertain because the emissions estimates were based on one measured value and several nondetect values.

Table 5-2. Inhalation Screening Assessment for Noncancer Effects of HAPs Emitted From Coal-Fired Utilities for Which Inhalation Reference Concentrations Are Available

Hazardous air pollutant	RfC^{a,b} (µg/m3)	Noncancer health effect on which RfC is based^{b,c}	Confidence^{b,d} in RfC	Highest MEI^{e} conc. (µg/m^3)	Max. HQ^{f}	Is the highest HQ above 0.1^{g}
2-Chloro-Acetophenone	0.03	Hyperplasia of nasal resp. epith. in rats	low	3 x 10^{-5}	0.001	No
Acrolein	0.02	Metaplasia and inflammation rat nasal epithel.	med	4 x 10^{-4}	0.02	No
Cumene^{h}	9.0^{h}	--	NA	0.00003	3 x 10^{-6}	No
Ethyl benzene	1000	Developmental effects	low	0.00005	5 x 10^{-8}	No
Ethyl chloride	10000	Delayed fetal ossification	med	0.0003	2 x 10^{-8}	No
Hexane	200	CNS & nasal epith. lesions humans	med	0.00009	5 x 10^{-6}	No
Hydrogen chloride	20	Hyperplasia of nasal mucosa & larynx in rats	low	2.3	0.115	Yes
Hydrogen cyanide	3.0	CNS symptoms and thyroid effects	low	0.0033	0.001	No
Lead^{i,j}	1.5^{i}	CNS & devel. humans	NA	0.007	0.0057	No
Manganese	0.05	CNS, humans	med	0.02	0.4	Yes
Mercury^{j}	--	--	--	0.001	--	No
Methyl bromide	5.0	Lesions of olfactory epithelium	high	0.0001	2 x 10^{-5}	No
Methyl chloroform^{h}	1000^{h}	Hepatotoxicity	NA	0.0004	4 x 10^{-7}	No
Methyl ethyl ketone	1000	Decreased fetal birth weight (mice)	low	0.0009	9 x 10^{-7}	No
MTBE	3000	Increased liver & kidney weight (rat)	med	0.0002	7 x 10^{-8}	No
Styrene	1000	CNS in humans	med	0.00036	4 x 10^{-7}	No
Toluene	400	Neurological effects; degeneration of nasal epithelium	med	0.0004	1 x 10^{-6}	No
1,3-Dichloropropene	20	Hypertrophy/hyperplasia of nasal respiratory epithelium	high	0.00054	3 x 10^{-5}	No
Vinyl acetate	200	Nasal epithelium lesions	high	0.00005	3 x 10^{-7}	No

[See Footnotes on next page]

FOOTNOTES for Table 5-2

[a] RfC = Inhalation reference concentration.

[b] See chapter 4, Appendix E, and references for more information.

[c] This is the critical adverse noncancer health effect that was observed in the animal or human studies.[2,6] CNS = central nervous system.

[d] This is the overall confidence in the RfC as reported on IRIS.

[e] MEI = Maximally Exposed Individual. This is the highest estimated ambient concentration (annual average) due to coal-fired utility emissions based on HEM modeling within 50 km of all coal-fired utilities in the United States.

[f] HQ = Hazard Quotient. The ratio of exposure concentration/RfC. An HQ < 1.0 indicates that no adverse health effects are expected to occur (see Chapter 4 for discussion of HQ).

[g] If HQ > 0.1, this means that the highest modeled concentration is greater than 1/10 of the RfC. This value (0.1) was used as criteria in screening assessment. This is not considered a level of concern, but rather it is a conservative level to ensure that potentially important HAPs are not missed by screen. See text for explanation.

[h] The RfC was obtained from the 1992 EPA Health Effects Summary Tables. It has not been verified by the EPA's RfC/RfD workgroup.

i There is no RfC available for lead compounds. Therefore, as a substitute, the lead National Ambient Air Quality Standard[7] (1.5 $\mu g/m^3$) was used in this assessment. However, the lead NAAQS is not considered equivalent to an "RfC." The lead NAAQS is based on a quarterly average, but the exposure estimates here represent annual averages. The reader should exercise caution when interpreting the HQ for lead. Lead has also been classified as a B2 carcinogen.[2,5]

[j] These compounds may also be a health concern from multipathway exposure. The assessment here considers only inhalation exposure. Considering multipathway exposure may increase the risk estimates for these pollutants. Multipathway screening assessment is presented in section 5.5.

Table 5-3. Inhalation Screening Assessment for HAPS Emitted From Coal-Fired Utilities for Which No EPA-Verified Health Benchmarks Are Available. Comparison of Highest Modeled Air Concentration to Various Non-EPA Health Benchmarks

Hazardous air pollutant	NIOSH REL/420 ($\mu g/m^3$)[a]	OSHA PEL/420 ($\mu g/m^3$)[a]	ACGIH TLV/420 ($\mu g/m^3$)[a]	Highest MEI conc. ($\mu g/m^3$)[b]	Maximum HQ	Is the HQ > 0.1
Acetophenone	NA	NA	NA	0.00008	NA	NA
Antimony compounds	1.2	1.2	1.2	0.0005	4×10^{-4}	No
Carbon disulfide	7.1	29	74	0.0005	7×10^{-5}	No
Chlorobenzene	NA	833	830	0.00037	4×10^{-7}	No
Cobalt compounds	0.12	0.12	0.12	0.0017	0.014	No
o & p-Cresols	24	52		0.0003	1×10^{-5}	No
Cumene	580	580	580	0.00003	3×10^{-6}	No
Dibutyl phthalate	11.9	11.9	11.9	0.00033	3×10^{-5}	No
Hydrogen fluoride	6.0	6.0	6.0[c]	0.365	0.06	No
Methyl methacrylate	NA	980	980[c]	0.00013	1×10^{-7}	No
MIBK	490	980	490	0.00058	1×10^{-6}	No
Phenol	48	45	45[c]	7×10^{-4}	2×10^{-5}	No
Phthalic anhydride	14	14	14	6×10^{-4}	4×10^{-5}	No
Phosphorus	0.24	0.24		0.0036	0.015	No
Propion-aldehyde	NA	NA	NA	0.0012	NA	NA
Selenium compounds	0.48	0.48	0.48[c]	0.0056	0.012	No
m,o,p-Xylenes	1000	1000	1000	0.0005	5×10^{-7}	No
2,4-Dinitrotoluene	3.6	3.6	0.36	1×10^{-6}	3×10^{-6}	No
Methyl iodide	24	67	29	0.00005	2×10^{-6}	No

[a] NIOSH = National Institute for Occupational Safety and Health, a U.S. government organization that focuses on research.

OSHA = Occupational Safety and Health Agency, a U.S. Government Agency

ACGIH = American Conference of Government Industrial Hygienists, which is a professional society, not a government agency.

FOOTNOTES for Table 5-3 (continued)

The NIOSH, OSHA, and ACGIH are primarily involved with the safety and health of workers.

REL = Recommended Exposure Levels are developed by NIOSH and are recommended levels to protect workers.

TLV = Threshold Limit Values. The TLV are established by ACGIH and are used by industrial hygienists in the work place to assess the potential concerns for worker exposure.

PEL = Permissible Exposure Levels, legal limits established by OSHA.

The RELs, PELs, and TLVs are relatively similar. Breathing concentrations below these levels are expected to be reasonably protective of healthy workers, exposed for 8 hours per day, 5 days per week (~40 hours). However, there are uncertainties and often the data are less than complete. Also, for some of these values (especially the PELs), measurement techniques and economic factors are sometimes factored in.[8,9,10]

The use of Occupational Exposure Limits (OELs) are being used in this study for screening assessment purposes only. For this screening assessment, the REL, PEL, and TLV were divided by 420 (4.2 x 10 x 10). The 4.2 is the conversion factor to extrapolate from 40 hr/week to 168 hr/week. A factor of 10 is used to adjust for sensitive subpopulations. Another factor of 10 is used to account for additional uncertainties associated with these values. A similar method was used by the California Air Resources Board (CARB) in the Air Toxics "Hot Spots" Program.[11] CARB also divides the TLV by 420 to calculate some of their noncancer reference exposure levels (4.2 to account for exposure time adjustment, 10X to account for sensitive individuals, and another 10X because health effects are sometimes observed at the TLV level).

[b] MEI = Maximally Exposed Individual. This is the highest estimated ambient concentration due to coal-fired utility emissions based on HEM modeling within 50 km of all coal-fired utilities in the United States.

[c] These values are the same as the CARB Noncancer Reference Exposure Levels used in the "Hot Spots Program".[11]

Table 5-4. Inhalation Screening Assessment for Carcinogenic HAPS From Oil-Fired Utilities for Which Quantitative Cancer Risk Estimates Were Available

Hazardous air pollutant	Highest MEI conc. ($\mu g/m^3$)[a]	EPA WOE[b]	IURE^3 per $\mu g/m^c$	Cancer MEI Risk[d]	Primary Type of Cancer associated w/ Inhalation.[e]	MEI cancer risk > 10^{-7}
Arsenic	0.0032	A	0.0043	1×10^{-5}	Lung	Yes
Chromium (18 percent Cr VI)[f]	0.0025	A	0.0028^6	7×10^{-6}	Lung	Yes
Beryllium	0.0003	B2	0.0024	7×10^{-7}	Lung	Yes
Cadmium	0.0009	B2	0.0018	2×10^{-6}	Lung	Yes
Nickel compounds[g]	0.21	A^2	0.00048	1×10^{-4}	Lung & nasal	Yes
Dioxins[h]	4×10^{-9}	B2	30.0	1×10^{-7}	Tongue, lung, nasal, liver, thyroid	Yes
PAHs[i]	0.00003	B2	0.0021	6×10^{-8}	Lung (BAP)	No
Formaldehyde	0.007	B1	1.3×10^{-5}	9×10^{-8}	Nasal, lung	No
Acetaldehyde	0.0019	B2	2.2×10^{-6}	4×10^{-9}	Nasal & laryngeal	No
Benzene	0.0003	A	8.3×10^{-6}	3×10^{-9}	Leukemia	No
Methylene chloride	0.008	B2	4.7×10^{-7}	4×10^{-9}	Liver & lung	No
Naphthalene	0.00008	C	4.2×10^{-6}	3×10^{-10}	--	No
Tetrachloroethylene[j]	0.00013	B2/C	5.8×10^{-7}	8×10^{-11}	Liver	No

[SEE FOOTNOTES FOR Table 5-1]

Table 5-5. Inhalation Screening Assessment for Noncancer Effects of HAPS Emitted from Oil-Fired Utilities for Which EPA-Verified Inhalation Reference Concentrations Are Available

Hazardous air pollutant	RfC[a] ($\mu g/m^3$)	*Critical noncancer health effect that RfC is based on.*[b]	Overall *confidence* in RfC[c]	Highest MEI[d] conc. ($\mu g/m^3$)	Highest HQ[e]	Is HQ > 0.1
Ethyl benzene	1000	Developmental effects	Low	1×10^{-4}	1×10^{-7}	No
Hydrogen chloride	20	Hyperplasia of nasal mucosa, larynx, and trachea in rats	Low	1.1	0.16	Yes
Lead[i]	1.5	Neurotoxicity and developmental in humans	NA	0.005	0.003	No
Manganese	0.05	Neurobehavioral effects in humans	Medium	0.002	0.04	No
Mercury	--	--	--	0.00014	--	No
Methyl chloroform[h]	1000[h]	Hepatotoxicity[h]	NA	0.0018	2×10^{-6}	No
Toluene	400	Neurological effects	Medium	0.002	5×10^{-6}	No
Vinyl acetate	200	Nasal lesions	High	0.0012	6×10^{-6}	No

[SEE FOOTNOTES FOR Table 5-2]

Table 5-6. Inhalation Screening Assessment for HAPS Emitted from Oil-Fired Utilities for Which No EPA-Verified Health Benchmarks Are Available—Comparison of Highest Modeled Concentration to Various Non-EPA Health Benchmarks

Pollutant	NIOSH REL/420 ($\mu g/m^3$)[b]	OSHA PEL/420 ($\mu g/m^3$)[b]	ACGIH TLV/420 ($\mu g/m^3$)[b]	Highest MEI conc. ($\mu g/m^3$)[c]	Max HQ	Is Max HQ >0.1
Cobalt compounds	0.12	0.12	0.12	0.0096	0.08	No
Hydrogen fluoride	6.0	6.0	NA	0.03	0.005	No
Phenol	48	45	45	0.006	0.0001	No
Phosphorus	0.24	0.24	0.24	0.026	0.1[a]	No
Selenium compounds	0.48	0.48	NA	0.001	0.002	No
m,o,p-Xylenes	1040	1040	1040	0.0005	5×10^{-7}	No

[SEE FOOTNOTES FOR Table 5-3]

Table 5-7. Inhalation Screening Assessment for HAPS Emitted from Gas-Fired Utilities

Hazardous air pollutant	Highest MEI conc. (μg/m^3)	IURE per μg/m^3	HEM Cancer MEI Risk	RfC μg/m^3	Highest HQ
Arsenic	2×10^{-5}	0.0043	1×10^{-7}	NA	NA
Nickel compounds[b]	0.0003	0.00048	2×10^{-7}	NA	NA
Naphthalene	0.0001	4×10^{-6}	4×10^{-10}	NA	NA
Toluene	0.0018	NA	NA	400	4.5×10^{-6}
Lead	0.00006	NA	NA	1.5	4×10^{-5}
Formaldehyde	0.008	1.3×10^{-5}	1×10^{-7}	NA	NA
Mercury	0.0000002	NA	NA	--	--
Benzene	0.0003	8.3×10^{-6}	2×10^{-9}	NA	NA
Phosphorus	0.0002	NA	NA	0.24[a]	0.0008
Cobalt	0.00002	NA	NA	0.12[a]	0.0002

[a] These values are not RfCs. They are TLV/420. See Tables 5-3 and 5-6.

[SEE FOOTNOTES FOR TABLES 5-1 to 5-6]

5.5 MULTIPATHWAY SCREENING ANALYSIS FOR NONRADIONUCLIDE HAPS

5.5.1 Overview

In past years, most analyses of human health risk associated with atmospheric emissions of nonradionuclide HAPs from combustion sources have focused only on exposures occurring by inhalation. The inhalation exposure pathway is generally the significant pathway for human exposure to air pollutants. In the past decade, though, studies have linked elevated levels of pollutants in soils, lake sediments, and cow's milk to atmospheric transport and deposition of pollutants from combustion sources.[12] Scientists have collected convincing evidence showing that toxic chemicals released to air can travel long distances and be deposited to land and water at locations both near and far from their original emission sources.[13] Many studies indicate that deposition of atmospherically emitted pollutants can result in indirect avenues of exposure for humans.[14] For some HAPs, these noninhalation routes of exposure can be as significant, or more significant, than inhalation.

Certain HAPs have been associated with significant adverse effects on human health and wildlife from noninhalation exposure pathways.[13] HAPs that pose a concern for noninhalation exposure generally have common characteristics. They are persistent in the environment, have the potential to bioaccumulate, and exhibit toxicity via ingestion. For lipophilic contaminants such as dioxins, furans, polychlorinated biphenyls, and certain pesticides and for metals such as lead and mercury, exposures through food consumption have been demonstrated to be dominant contributors to total dose for nonoccupationally exposed populations.[15] It is also likely that atmospheric pollution from combustors and other thermal processes significantly contributes to the ubiquitous presence of some of the highly persistent lipophilic compounds.[15]

Multipathway exposure to HAPs can potentially occur through the following exposure routes:

- Soil ingestion
- Soil dermal contact
- Water ingestion
- Inhalation
- Fish and meat ingestion
- Fruit ingestion
- Vegetable ingestion
- Ingestion of animal fats
- Milk ingestion
- Ingestion of other food products.

The following section presents the screening assessment to prioritize the nonradionuclide HAPs for further analysis of multipathway exposures and risks. Chapters 7 and 8 present the multipathway assessments for the selected priority nonradionuclide HAPs.

5.5.2 Prioritization of HAPs for Multipathway Exposure Assessment

The 66 nonradionuclide HAPs potentially emitted by utilities were evaluated for their potential to cause health effects through noninhalation exposure pathways. To select the highest priority HAPs for multipathway exposure assessment, a four-step process was followed. This process involved assessing the HAPs for their potential to be of concern for exposure through noninhalation pathways, evaluating their toxicity, and considering the emission levels from utilities. First, a subset of HAPs was selected from the list of 66 nonradionuclide HAPs by using the HAP ranking presented in Attachment A (draft Focus Chemicals Report) of the EPA document, Schedule for Standards: Methodology and Results for Ranking Source Categories Based on Environmental Effects Data.[16] The four criteria evaluated and used in this ranking were human toxicity, aquatic toxicity, bioconcentration potential, and environmental persistence. Environmental partitioning was not used as a ranking criterion but was used as a "qualifying" criterion. The HAP ranking method is a modified version of the Inerts Ranking Program (IRP) methodology developed by EPA's Office of Research and Development, Environmental Research Laboratory - Duluth, for evaluating pesticide ingredients. The IRP scoring method was modified for scoring the environmental criteria and for determining overall scores for the HAPs. For the environmental criteria modification, acute aquatic toxicity and chronic aquatic toxicity were combined into a single aquatic toxicity criterion that is based strictly on chronic toxicity data when such data are available. Each criterion, except environmental persistence, allowed a possible score of 0, 1, 2, or 3. A score of 0 indicates that no data are available, and scores of 1, 2, and 3 indicate low, medium, and high concern, respectively. For environmental persistence, substances were assigned a score of 1 or 3 only, because data did not support finer discrimination. The method for deriving the overall score was similar to that for deriving the original IRP score. For each substance, the overall score was derived by adding the scores for the four criteria, dividing by the number of criteria for which there were data, and then multiplying by 10 to produce an overall score on a scale of 10 to 30.

The HAP ranking in the Focus Chemicals report is a ranking of all of the HAPs based on the overall score for each HAP. Of the 66 HAPs potentially emitted to air by utilities, those that ranked the highest, with overall ranking "scores" of greater than 23, were selected for further evaluation. The cutoff score of 23 was selected because, at this level and below, a HAP would have scores of 1 or 2, indicating low and medium concern, respectively, for at least two of the four criteria. Thirteen HAPs were selected on these criteria. The high ranking reflects that these 13 HAPs are more likely to be highly persistent in the environment and/or to bioaccumulate, as well as to potentially be toxic to humans. The 13 HAPs selected and their ranking "scores" are listed in Table 5-8. This approach to ranking the HAPs is a screening-level, hazard-based ranking of chemicals. This approach yielded a subset of 13 HAPs from which to select five HAPs for further evaluation.

In the second step of the process, additional information was gathered for each of the 13 selected HAPs to determine the HAPs that are most important for multipathway assessment for the utilities. Where available and applicable, the RfD, the oral unit risk estimate (OURE), the EPA WOE classification, and the emissions estimate were obtained for each of the 13 HAPs. This information is presented in Table 5-9.

Several criteria were used to further prioritize HAPs for multipathway exposure assessment. The six HAPs with the highest noncancer toxicity (i.e., lowest RfDs [less than 1 x 10-3]), as well as the HAPs with EPA-verified OUREs and a WOE classification of A or B, were selected. Also, due to their extremely

Table 5-8. Thirteen HAPs Selected from the Hazard-based Multipathway Ranking (shown in order of ranking), and the Overall and Individual Criterion Scores Assigned to Each

HAP	Overall score	Human toxicity score	Aquatic toxicity score	Bioconcentration potential score	Environmental persistence score
2,3,7,8-TCDD (dioxins)	30	3	3	3	3
Cadmium compounds	30	3	3	3	3
Mercury compounds	30	3	3	3	3
Hexachlorobenzene	28.75	2.5	3	3	3
Selenium compounds	28.75	2.5	3	3	3
Lead compounds	27.5	3	3	2	3
Cobalt compounds	26.67	3	2	0	3
Pentachlorophenol	25	2	3	2	3
Arsenic compounds	25	3	2	2	3
POM (PAH)	25	3	3	3	1
Beryllium compounds	23.75	2.5	2	3	3
Cyanide compounds	23.75	2.5	3	3	1
Manganese compounds	23.75	1.5	2	3	3

PAH = Polycyclic aromatic hydrocarbons.
POM = Polycyclic organic matter.

high toxicity and the concern that they are "no threshold" or extremely low threshold chemicals, 2,3,7,8-TCDD and lead compounds were also included. Mercury was also selected because mercury is persistent, tends to bioaccumulate, and is relatively toxic by ingestion exposure. This second step in the prioritization process resulted in eight HAPs being selected: 2,3,7,8-TCDD, lead, mercury, arsenic, cadmium, hexachlorobenzene, beryllium, and particulate organic matter (POM). In the next step in the selection process (the third step), emission estimates from utilities were examined for each of the eight HAPs. For two of the HAPs, POM and hexachlorobenzene, the emissions data for utilities did not support their inclusion in further assessments. For hexachlorobenzene, emissions were not considered high (0.7 ton/yr). Also, this estimate was highly uncertain because of the very limited emissions data for hexachlorobenzene. For POM, the emission levels of 1.9 ton/yr from coal-fired utilities and less than 1 ton/yr for oil-fired utilities are low relative to other anthropogenic sources of POM.

Table 5-9. Comparison of Cancer and Noncancer Effects Benchmarks and Emissions Estimates for 13 Selected HAPs

HAP	RfD (mg/kg/day)	OURE (per µg/L)	WOE	Coal-fired emissions estimates[d] (ton/yr)	Oil-fired emissions estimates[d] (ton/yr)
2,3,7,8-TCDD (dioxins)(TEQ)	NA	3 x 10^{+0}[a]	B2	1.5 x 10^{-4}	1.0 x 10^{-5}
Lead compounds	no threshold[b]	--	B2	7.2 x 10^{+1}	11
Mercury compounds	--	--	--	5.1 x 10^{+1}	0.3
Arsenic compounds[e]	3 x 10^{-4}	5.0 x 10^{-5}	A	5.4 x 10^{+1}	5
Cadmium compounds	5 x 10^{-4}	--	B2	1.9	2
Hexachlorobenzene	8 x 10^{-4}	4.6 x 10^{-5}	B2	0.7	NA
Selenium compounds	5 x 10^{-3}	--	--	1.9 x 10^{+2}	2
Beryllium compounds	5 x 10^{-3}	1.2 x 10^{-4}	B2	6.6	0.5
Cyanide compounds	5 x 10^{-3}	--	--	2.4 x 10^{+2}	NA
Manganese compounds	5 x 10^{-3}	--	--	1.8 x 10^{+2}	10
Pentachlorophenol	NA	--	B2	7.0 x 10^{-2}	NA
Cobalt compounds	NA	--	--	2.1 x 10^{+1}	20
POM (PAH)[c]	NA	2.1 x 10^{-4}	B2	1.9	< 1

NA = not available.
PAH = Polyarometic hydrocarbons.
POM = Particulate organic matter.

[a] This is an unverified oral unit risk estimate.[3]

[b] The Agency has determined that some of the effects of lead, particularly changes in the levels of certain blood enzymes and in aspects of children's neurobehavioral development, may occur at blood lead levels so low as to be essentially without a threshold.

[c] POM emissions were estimated by summing the emissions estimates for each individual PAH listed in Appendix H.

[d] This is an estimate of total nationwide emissions from the source category.

[e] RfD is for inorganic arsenic. There was not a clear consensus for developing this value. See the IRIS database for details.

For the final step, the two lowest-ranking of the six remaining HAPs (cadmium and beryllium) were compared with each other in terms of toxicity, emissions, and the original "ranking" scores they were assigned. Cadmium was selected for further assessment, rather than beryllium, because of its higher ranking scores for human and aquatic toxicity and its lower RfD, representing higher noncancer toxicity. This resulted in the selection of five highest priority HAPs for multipathway exposure assessment. The five HAPs selected to be highest priority for further analysis were 2,3,7,8-TCDD, lead compounds, mercury compounds, arsenic compounds, and cadmium compounds.

5.6 SELECTION OF HAPS FOR FURTHER ANALYSIS

In the initial phase of the screening assessment 12 HAPs were selected as priority. Pollutants were selected as priority in the initial phase if they met one of the following three criteria: (1) the MEI inhalation cancer risk was estimated to be greater than 1 in 10 million (i.e., 1×10^{-7}); (2) maximum inhalation exposure concentration is greater than one-tenth the RfC (i.e., if the HQ was greater than 0.1); or (3) the emitted HAP is persistent in the environment, tends to bioaccumulate, and emissions are significant enough that there are potential concerns for human health from multipathway exposure. The risk levels chosen for the first two criteria (i.e., 1×10^{-7} and 1/10 the RfC) are lower than levels that have been considered historically as levels for regulatory and policy decisions (e.g., 1×10^{-6} for cancer and RfC for noncancer). These lower values were chosen for screening purposes so that it is unlikely that potentially important HAPs would not be identified by screen. That is, these conservative levels were chosen to ensure that all potentially important HAPs would pass the screen. The third criterion was primarily chosen to identify HAPs that are considered a potential concern from multipathway exposure. Based on these three criteria, 12 HAPs (arsenic, beryllium, cadmium, chromium, dioxin/furans, nickel, n-nitrosodimethylamine, hydrogen chloride, manganese, lead, mercury, and formaldehyde) were chosen to be priorities for further assessment.

Radionuclides were also chosen as a priority for multipathway assessment because previous risk assessments indicate that radionuclides from utilities could potentially cause cancer risks greater than 1×10^{-6} for MEIs.[1]

In addition, three HAPs (HCl, HF, and acrolein) were chosen as priority for assessment of potential noncancer effects due to short-term (acute) exposures. The prioritization of HAPs for short-term exposure analysis was based on review of health effects data,[2,10,11] emissions estimates, and recommendations from the peer review panel. Hydrogen chloride, HF, and acrolein were the three HAPs considered to be of highest potential concern for health effects due to short-term exposures. Table 5-10 presents the 15 HAPs that were selected as priority based on the screening assessment.

5.7 LIMITATIONS OF SCREENING ASSESSMENT

The HAPs that were not chosen for further analysis were below the screening level and not considered priority for this report. These HAPs are considered lower priority and less likely to present significant risks to public health. However, due to uncertainties and limitations in the data, it is not possible to conclusively determine that they do not pose a threat to public health. Although these HAPs are not analyzed further in this report, it is possible that future data, such as revised emissions data or new toxicologic data, could warrant further evaluation of these HAPs in the future.

Table 5-10. Pollutants Considered Priority for Further Analysis Based on Results of Screening Assessment

Pollutant	Priority for coal	Priority for oil	Priority for gas	Inhalation MEI cancer risk > 10-7	Noncancer Inhalation HQ > 0.1	Priority for multi-pathway assessment
Acrolein[a]	Yes	No	No	No	No	No
Arsenic	Yes	Yes	Yes	Yes	NA	Yes
Beryllium	Yes	Yes	No	Yes	NA	No
Cadmium	Yes	Yes	No	Yes	NA	Yes
Chromium	Yes	Yes	No	Yes	NA	No
Dioxins/furans	Yes	Yes	No	Yes (oil)	NA	Yes
Nickel	Yes	Yes	Yes	Yes	NA	No
Radionuclides[b]	Yes	Yes	Yes	NA	NA	Yes[b]
n-Nitroso-dimethylamine	Yes	No	No	Yes	NA	No
Hydrogen chloride	Yes	Yes	No	No	Yes	No
Hydrogen flouride[a]	Yes	Yes	No	No	No	No
Manganese	Yes	Yes	No	No	Yes	No
Lead	Yes	Yes	No	No	No	Yes
Mercury	Yes	Yes	No	No	No	Yes
Formaldehyde	No	No	Yes	Yes (gas)	NA	No

[a] Acrolein and hydrogen fluoride did not pass screen based on RfC analysis. However, these two HAPs were identified as priority because of potential concern for acute exposure.

[b] Radionuclides were considered priority based on previous risk assessments conducted in the 1980s on radionuclides from utilities.[1]

5.8 REFERENCES

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6.0 INHALATION RISK ASSESSMENT

6.1 BASELINE ASSESSMENT OF INHALATION EXPOSURES AND RISKS FOR 14 PRIORITY POLLUTANTS

This chapter presents estimates of risks due to inhalation exposure to 14 of the 15 priority HAPs identified in the screening assessment (chapter 5). Radionuclides were the one priority HAP not included in this chapter because the analysis for radionuclides requires an air dispersion model that predicts the impacts of the radioactive decay process. The radionuclide analysis is presented in chapter 9. In the baseline assessment, for the 14 priority HAPs, risks have been calculated using the HEM for HAP emissions from all 684 utilities utilizing the standard HEM default options and assumptions described in chapter 4. The HEM estimates ambient air concentrations within 50 km of each utility. Therefore, the baseline risk estimates reflect only inhalation exposure within 50 km of each utility (i.e., local analysis). In addition, the baseline risks do not account for background levels, long-range transport, complex terrain, indirect exposures, or overlapping plumes. These issues are analyzed and discussed in later sections of the report.

Not incorporating the above factors may lead to underestimating risks. However, there are several important assumptions that were incorporated into the baseline assessment that are generally conservative (i.e., more likely to overestimate rather than underestimate risks). For example, the baseline assessment assumes that MEIs are exposed to the modeled concentrations for 70 years. Also, the cancer potency values (i.e., IUREs) that were used in this assessment are considered "upper limit" estimates.[1] These are just a few of the assumptions and uncertainties associated with the baseline assessment. Later sections describe many of the data inputs and default assumptions and discuss various issues and uncertainties.

The HEM exposure modeling conducted for the screening assessment (chapter 5) was very similar to the HEM exposure modeling conducted for the baseline assessment (this section). The same default options described in chapter 4 and same input data were used. However, there is one important difference. For the baseline assessment, a distinction was made between urban and rural locations. If a utility plant is located in an urban area, it was modeled using the urban mode (i.e., dispersion is assumed to be characteristic of emissions emitted by a facility in an urban location where there are buildings nearby). If a utility plant is located in a rural location, it was modeled using the rural mode (i.e., dispersion is assumed to be characteristic of a facility located in a rural location). In the screening assessment, all plants were modeled using the urban default because using the urban default typically leads to more conservative estimates of human exposures. However, using the urban and rural distinction is believed to reflect more realistic conditions; therefore, it was considered appropriate to use the urban versus rural distinction in the baseline assessment, and in subsequent HEM modeling analyses presented throughout chapter 6. The urban and rural options (which differ in the assumptions for surface roughness) and their impact on the risk estimates are discussed in detail in section 6.2.

The uncertainty analysis (presented later) suggests that the baseline risk estimates are generally conservative, but not overly conservative. Therefore, the results presented in this section (baseline risk estimates) are generally considered reasonably high-end estimates of the risks due to inhalation exposure of utility HAP emissions within 50 km of the utility plants. This conservatism is considered appropriate given EPA's mandate of public health protection.

6.1.1 Baseline Inhalation Risks for Coal-Fired Utilities for Priority HAPs

A total of 426 coal-fired units were modeled with the HEM using 1990 emissions and population data. Two of the plants resulted in individual risks less than 1 x 10^{-9} and were excluded from the presentation of results, thus reducing the total number of plants to 424. Table 6-1 summarizes the MEI risks, the number of persons exposed above individual cancer risk levels of 1 chance in 1 million (i.e., 1 x 10^{-6}), the number of plants whose emissions result in those risk levels, and the maximum HQs.

6.1.1.1 Individual Cancer Risk. Table 6-1 presents the estimated risks due to inhalation exposure within 50 km for each HAP across all coal-fired plants. As stated previously, the MEI is calculated based on the maximum modeled ambient concentration even though a person may or may not reside in the vicinity of the maximum concentration. The MEI risk was highest for arsenic (a Class A, human carcinogen) at 3 x 10^{-6} for the "highest-risk" coal-fired plant. The highest estimated MIR at a single plant was 2 x 10^{-6} for arsenic. Table 6-1 shows that arsenic emissions from two plants resulted in MIRs greater than or equal to 10^{-6}. The MIRs for the remaining 424 coal-fired plants were lower than 1 x 10^{-6}. Figures 6-1a and 6-1b show that most inhalation risks were considerably lower than 1 x 10^{-6}. The risk for chromium assumes that 11 percent of total chromium is hexavalent chromium,

Table 6-1. Summary of Baseline Risk Estimates from Chronic Inhalation Exposure by HAP for 424 U.S. Coal-fired Utilities

	Carcinogens				Noncarcinogen
Pollutant	**Highest MEI cancer risk**[e]	**Population with risk $> 10^{-6}$**	**# Plants with MIR $> 10^{-6}$**	**Cancer incidence (cases/yr)**[c]	**Max. HQ**
Arsenic	3 x10^{-6}	852	2	0.05	NA
Beryllium	3 x10^{-7}	0	0	0.004	NA
Cadmium	2 x10^{-7}	0	0	0.0006	NA
Chromium[a]	2 x10^{-6}	107	1	0.02	NA
Dioxin/Furans	5 x10^{-8}	0	0	0.001	NA
Hydrogen Chloride	NA	NA	NA	NA	0.12
Lead	NA	NA	NA	NA	0.001
Manganese	NA	NA	NA	NA	0.046
Mercury	NA	0	0	NA	--
Nickel[b]	7 x10^{-7}	0	0	0.005	NA
n-Nitrosodi-methylamine [d]	8 x10^{-7}	0	0	0.02	NA
Total	4 x10^{-6}	NA	2	0.1	NA

HQ =	Hazard quotient, the ratio of exposure concentration to the reference concentration (RfC). HQ values below 1 are not expected to result in adverse effects.
MEI =	Maximum exposed individual, which is calculated using the highest annual average concentration predicted with the HEM. An individual may or may not be exposed at that point. This value may be greater than the MIR.
MIR =	Maximum individual risk is the highest risk identified at the centroid of a census tract to which a population is assigned. *See chapter 4 for description of MEI and MIR.*
NA =	Not available.
Total =	Total MEI are the sum of MEI for individual HAPs within a plant. The total HQ (=HI) is the sum of the HQs within a plant.

[a] Assumes that 11 percent of total chromium emitted is hexavalent chromium, the species of chromium responsible for carcinogenic potential. Trivalent chromium, which would also be present, is assumed not to have carcinogenic effects.

[b] The nickel emitted is a mixture of various nickel compounds such as soluble nickel. This analysis assumes that all nickel emitted has the same carcinogenic potency as nickel subsulfide.

[c] The cancer incidence could be up to roughly 7 times greater when considering the impacts of long-range transport (i.e., exposure outside of 50 km) from all coal-fired utilities combined. See section 6.8 for discussion of long-range transport.

[d] The risk assessments for n-nitrosodimethylamine are highly uncertain because of the very limited emissions data available for n-nitrosodimethylamine. The emissions estimates were based on one measured data value and several nondetects. Therefore, the risk estimates for n-nitrosodimethylamine are considered conservative and considerably uncertain.

[e] Of all 424 coal-fired plants modeled with the HEM, this is the estimated increased inhalation cancer risk for a person assumed to be exposed for 70 years to the highest modeled HAP ambient air concentration around any of the 424 coal-fired plants.

Figure 6-1a. Maximum Individual Risk Posed by HAPs Emitted from All U.S. Coal-Fired Electric Utilities
(Number of coal-fired plants posing various levels fo risk, by HAP)

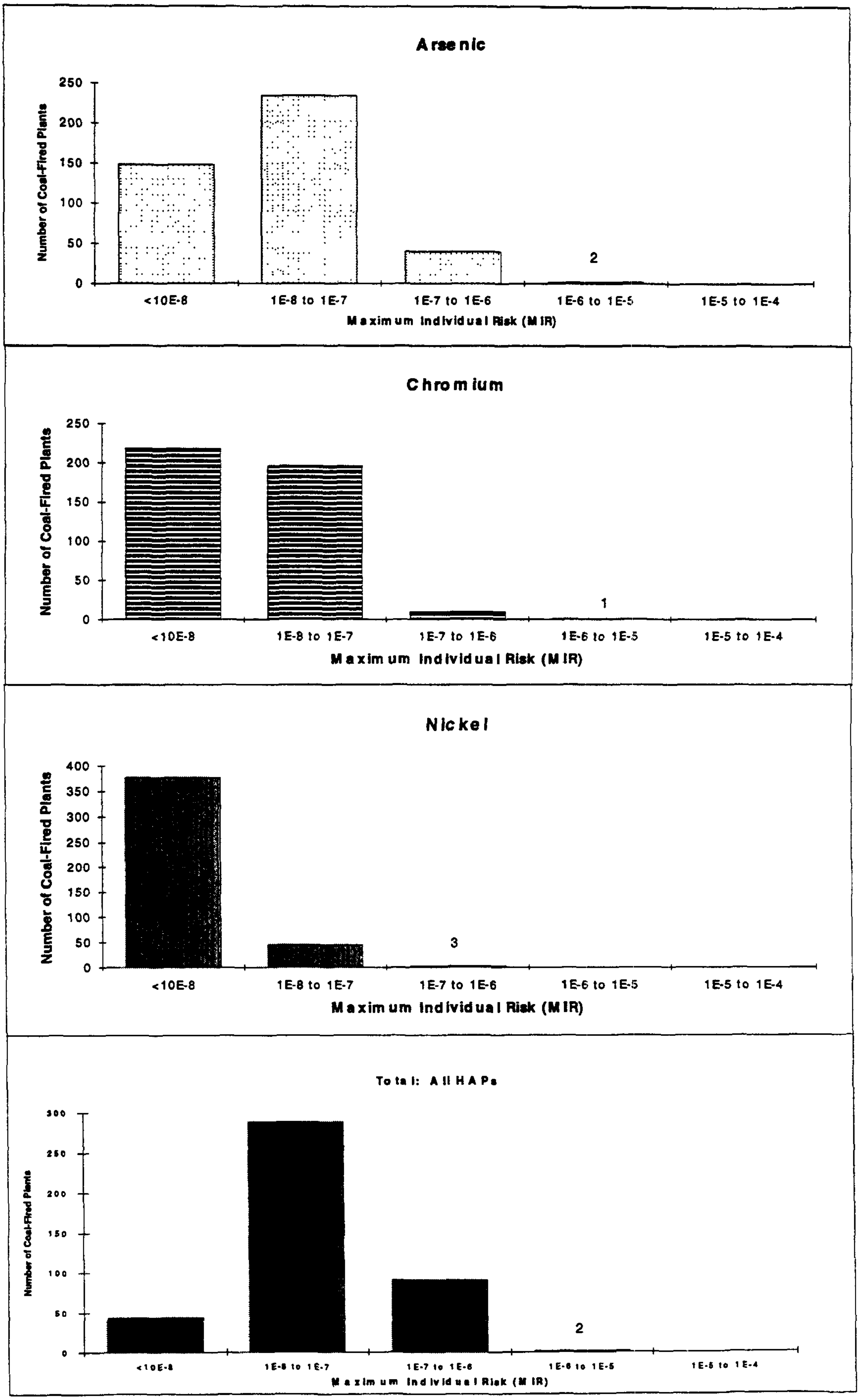
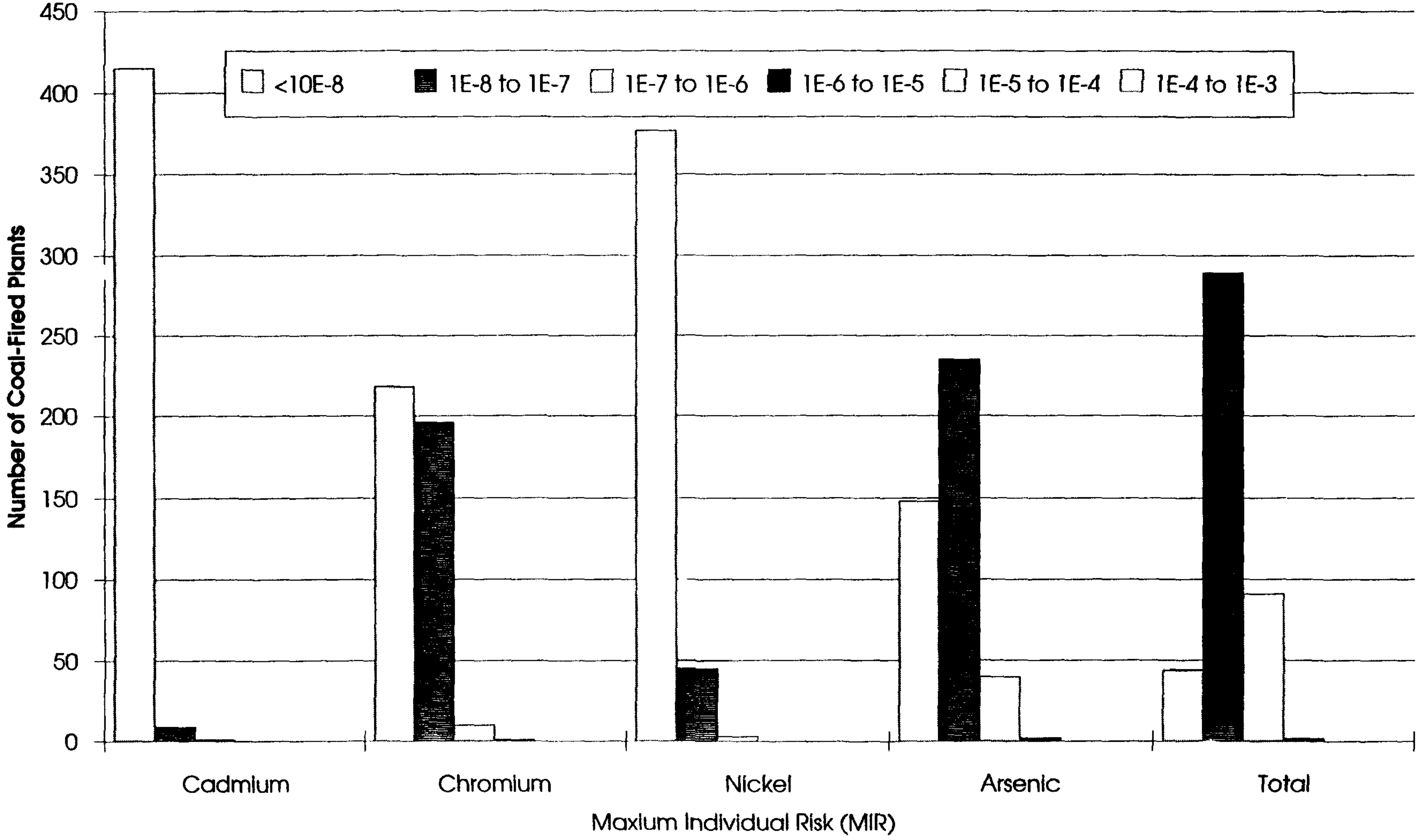


Figure 6-1b. Maximum Individual Risk Posed by HAPs Emitted from All U.S. Coal-Fired Electric Utilities (Number of coal-fired plants posing various levels of risk)



(a Class A, human carcinogen). The limited emissions speciation data (described in Appendix H) found hexavalent chromium between 0.3 and 34 percent of total chromium. The average percent chromium VI based on limited speciation data was 11 percent. The other HAPs do not appear to make a significant contribution to the MIR from coal-fired plants. Figures 6-1a and 6-1b present the distribution of plants at different MIR levels for the major carcinogens. Arsenic and chromium are the major contributors of inhalation cancer risks from coal-fired utilities. Of the 424 coal-fired plants, the median MIR is 2 x 10^{-8} for arsenic and 2 x 10^{-9} for chromium. The 90th percentile MIR of all 424 plants modeled is 1 x 10^{-7} for arsenic and 4 x 10^{-8} for chromium (that is, 10 percent of the plants have MIR levels above, and 90 percent of the plants have MIR levels below, these risk levels).

The total MIR due to inhalation exposure to the aggregate of HAPs for each plant was calculated by summing the MIR for each HAP for each plant. There are two coal-fired plants with total MIRs greater than 1 x 10^{-6}. The highest total MIR for a single coal-fired plant is 4 x 10^{-6}. Of the 424 coal-fired plants modeled, the median total MIR is 5 x 10^{-8}, and the 90th percentile is 2 x 10^{-7} (that is, 10 percent of plants pose an MIR greater than 2 x 10^{-7}).

6.1.1.2 Population Cancer Risk. The population distribution at various risk levels is shown in Table 6-2 for each of the six major carcinogenic HAPs. As with the MIR, arsenic and chromium are the major contributors. The number of people estimated to be exposed to risks of 1 x 10^{-6} or greater from exposure to arsenic is 850 and from exposure to chromium is about 107.

The HEM also calculated the annual incidence of cancer expected for each of the HAPs due to inhalation exposure within 50 km. The total cancer incidence from the carcinogenic HAPs was estimated, using the HEM, as 0.1 cancer case per year for coal-fired plants. Arsenic and chromium are again the major contributors and account for almost 90 percent of the estimated cancer incidences.

6.1.1.3 Noncancer Risk. The maximum HQ estimated for noncarcinogenic HAPs emitted from coal-fired power plants was 0.12 for HCl. The next highest was 0.046 for manganese. HQ values for all other HAPs were at least an order of magnitude lower. This assessment does not include background concentrations due to other sources.

Table 6-2. Summary of Population Exposed at Various Levels of Inhalation Risk or Greater by HAP: Coal-Fired Utilities

Risk level	Arsenic	Chromium	Nickel	Beryllium	Cadmium	Dioxins/ furans
5×10^{-6}	0	0	0	0	0	0
2.5×10^{-6}	0	0	0	0	0	0
1×10^{-6}	852	107	0	0	0	0
5×10^{-7}	5,990	2,160	0	0	0	0
2.5×10^{-7}	88,800	8,630	947	0	0	0
1×10^{-7}	1,710,000	80,500	5,100	1,280	107	0

6.1.2 Baseline Inhalation Risks for Oil-fired Utilities

A total of 137 oil-fired plants were modeled using 1990 HAP emissions and population data. The HEM estimated the individual and population risks for each of the HAPs evaluated. Eight plants had risks less than 1 x 10^{-9} and were excluded from the presentation. Table 6-3 presents the results.

6.1.2.1 Individual Cancer Risk. For oil-fired utilities, the HEM predicts that people live in the location of highest modeled ambient air concentration, therefore the MEI and the MIR are equal. The maximum MEI/MIR estimated for a single carcinogenic HAP across all plants was 9 x 10^{-5} from inhalation of nickel compounds.

There are numerous uncertainties that are discussed and analyzed in later sections, but the EPA believes that the uncertainties associated with nickel speciation are worth discussing here. There are substantial uncertainties associated with nickel speciation. In this analysis, as a conservative assumption, all nickel was assumed to be equipotent to nickel subsulfide, which is a Class A human carcinogen and has the highest cancer potency of all nickel compounds evaluated by the EPA. The limited speciation data indicate that less than 10 percent of nickel emissions (from oil-fired utilities) are nickel subsulfide (see appendix H). The remainder of the nickel is a combination of various nickel compounds for which the EPA has not yet determined the carcinogenic potential. Many nickel compounds are thought to have some carcinogenic potential via inhalation exposure although the potency is not known. This issue is discussed further in section 6.10.

Table 6-3. Summary of Baseline Risk Estimates from Inhalation Exposure for Priority HAPs for 137 U.S. Oil-Fired Utilities

	Carcinogens				Noncarcinogen
Pollutant	**Highest MEI Cancer Risk**	**Population with risk $> 10^{-6}$**	**# Plants with MIR $> 10^{-6}$**	**Cancer Incidence[c] (cases/yr)**	**MAX HQ**
Arsenic	1×10^{-5}	2,400	2	0.04	NA
Beryllium	7×10^{-7}	0	0	0.002	NA
Cadmium	2×10^{-6}	45	1	0.005	NA
Chromium[a]	5×10^{-6}	2,300	1	0.02	NA
Dioxin/Furans	1×10^{-7}	0	0	0.0007	NA
Hydrogen Chloride	NA	0	0	NA	0.06
Lead	NA	0	0	NA	0.004
Manganese	NA	0	0	NA	0.04
Mercury	NA	0	0	NA	--
Nickel[b]	9×10^{-5}	1.65M	20	0.4	NA
Total	1×10^{-4}	NA	22	0.5	NA

MEI = Maximum exposed individual, which is calculated using the highest concentration. An individual may or may not be exposed at that point. This value may be greater than the MIR, which is calculated at the centroid of a census block.
MIR = Maximum individual risk is the highest risk identified at the centroid of a census tract to which a population is assigned. See chapter 4 for description of MIR and MEI.
NA = Not available.
HQ = Hazard quotient, which is the ratio of exposure concentration to the reference concentration (RfC). HQ values below 1 are not expected to result in adverse effects.
Total = Total MEI is the sum of the MEI for individual HAPs within a plant. The total HQ (=HI) is the sum of the HQs within a plant.

[a] Assumes that 18 percent of total chromium emitted is hexavalent chromium, the species of chromium responsible for carcinogenic potential. Trivalent chromium, which would also be present, is assumed to be noncarcinogenic.

[b] This analysis conservatively assumes that all nickel emitted from utilities has the same carcinogenic potency as nickel subsulfide (the highest potency of nickel compounds tested). However, the nickel emitted is a mixture of various nickel compounds such as soluble nickel. Emissions tests indicate nickel subsulfide to be present as less than 10 percent of total nickel emitted. Many nickel compounds are thought to have carcinogenic potential although the potency is not known. See section 6.10 for further discussion and analysis of nickel speciation uncertainty.

[c] The cancer incidence could be up to roughly 7 times greater when considering the impacts of long-range transport (i.e., exposure outside of 50 km) from all utilities combined. See section 6.8 for discussion of long-range transport.

Figures 6-2a and 6-2b show the distribution of plants at different MIR levels for the major carcinogenic HAPs. The median MIR across all plants is 2 x 10^{-7} for nickel and 1 x 10^{-8} for arsenic. The 90th percentile for MIR among plants is 1 x 10^{-6} for nickel (that is, 90 percent of plants pose risks less than 1 x 10^{-6} due to nickel emissions) and 1 x 10^{-7} for arsenic.

The total MIR was calculated for each facility by summing the MIRs for individual HAPs. The highest total MIR from the sum of risks for each carcinogen is 1 in 10,000 (1 x 10^{-4}) at only one plant. The second and third highest-risk oil-fired plants pose MEI inhalation risks of 3 x 10^{-5} and 2 x 10^{-5}, respectively. The total MIR exceeded 1 x 10^{-6} as a result of HAP emissions from 22 oil-fired plants. The median total MIR for all plants is 7 x 10^{-7}, and the 90th percentile is 4 x 10^{-6}. However, these estimates are considered conservative, high-end estimates because they are mainly due to nickel emissions and the assumption that the mix of nickel compounds is as carcinogenic as nickel subsulfide (see section 6.10).

6.1.2.2 Population Risk. The population distribution at various risk levels is shown in Table 6-4 for each of six carcinogens. As with the MIR, nickel, arsenic, and chromium are the major contributors to the total population exposed to risk levels of 1 in 1,000,000 (1 x 10^{-6}) or more. The number of people estimated to have a risk from exposure greater than 1 x 10^{-6} is 1.65 million for nickel, and about 2,400 for arsenic and chromium.

Incidences from each HAP were summed to estimate total cancer incidence, which was estimated as 0.5 case per year from these 137 oil-fired plants. Nickel accounts for over 86 percent of the total annual incidence and arsenic contributes about 9 percent.

As with individual risk estimates for oil-fired plants, there are significant uncertainties associated with these population risk estimates because of the uncertainties associated with nickel speciation.

6.1.2.3 Alternative Analysis for Estimating Population Risks. Figure 6-3 summarizes the impact of using alternative IURE values for nickel (as a percent of the nickel subsulfide IURE) on annual cancer incidence. The estimated annual cancer incidence due to oil-fired utilities would be 0.3 case per year if the potency (IURE) of the mixture of nickel compounds emitted from oil-fired utilities was about 50 percent nickel subsulfide,

Figure 6-2a. Maximum Individual Risk Posed by HAPs Emitted from All U.S. Oil-Fired Electric Utilities (Number of Oil-Fired Utilities Posing Various Levels of Risk, by HAP)

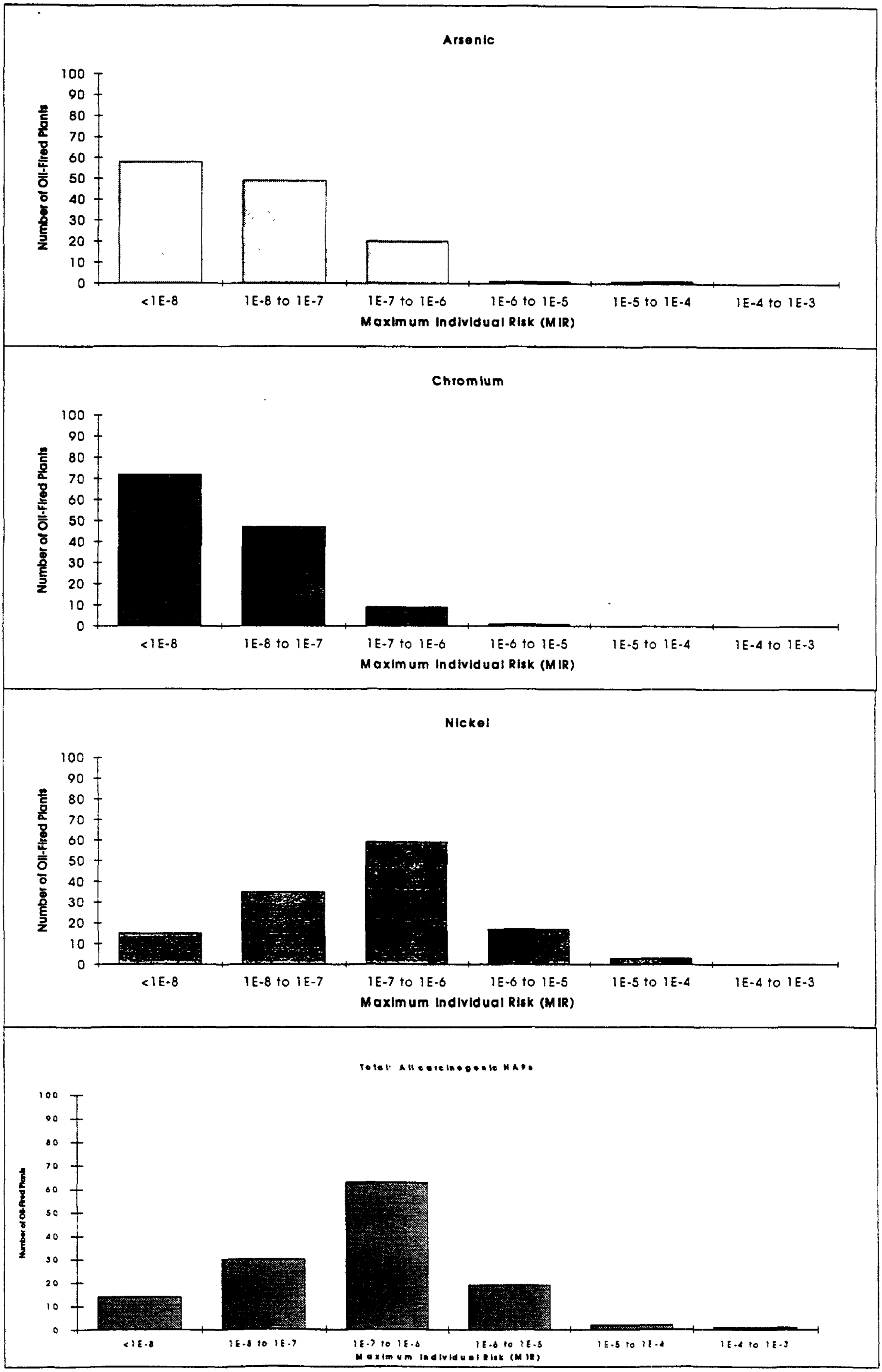


Figure 6-2b. Number of Oil-Fired Utilities Posing Various Levels of Maximum Individual Risk (by Levels of MIR)

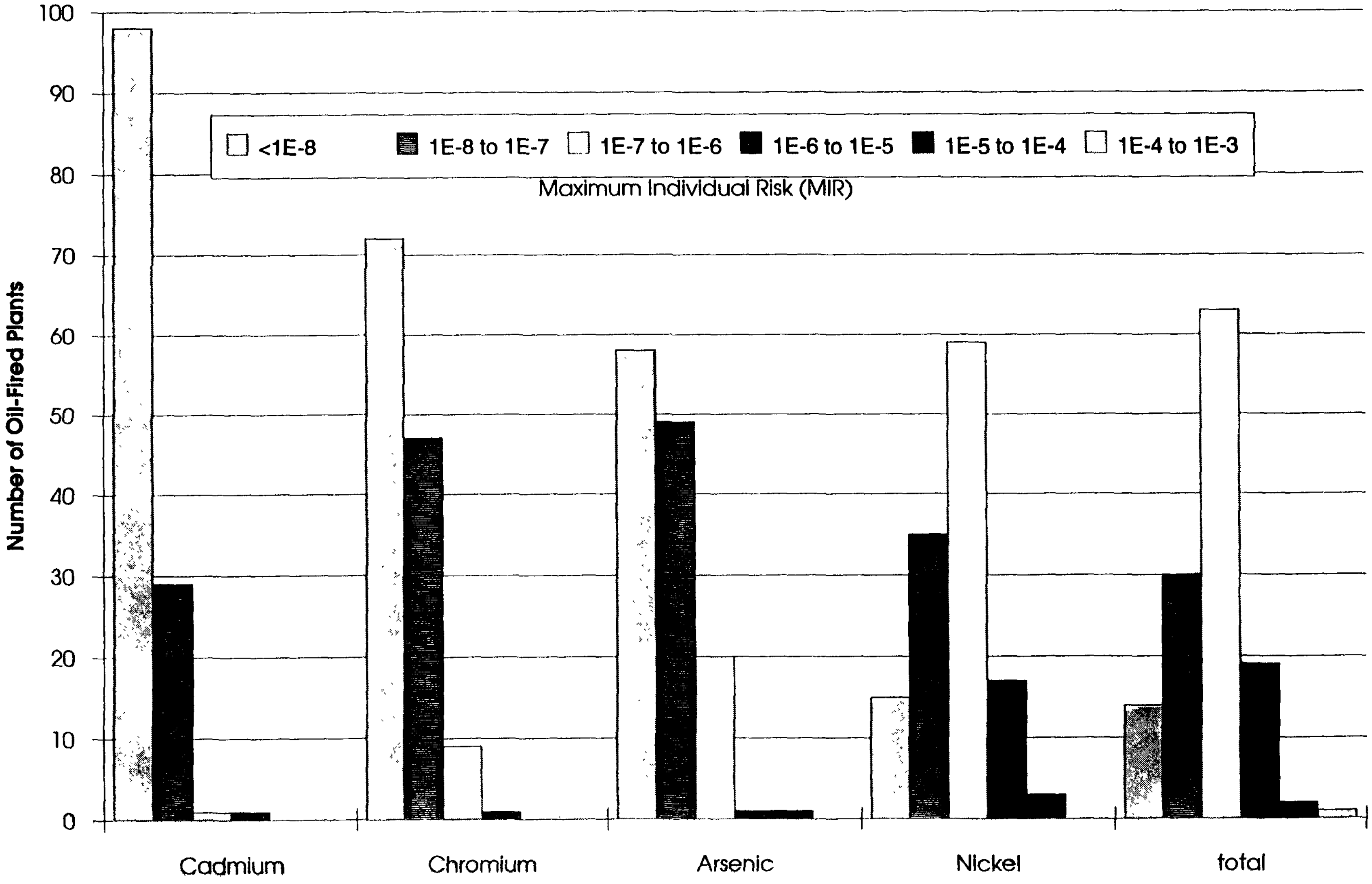


Table 6-4. Summary of Population Exposed at Various Levels of Risk or Greater from Oil-Fired Utilities

Risk Level	Nickel	Arsenic	Chromium	Cadmium	Beryllium	Dioxins/ furans
5×10^{-5}	89	0	0	0	0	0
2.5×10^{-5}	2,240	0	0	0	0	0
1×10^{-5}	2,310	45	0	0	0	0
5×10^{-6}	9,930	89	45	0	0	0
2.5×10^{-6}	100,000	2,280	89	0	0	0
1×10^{-6}	1,650,000	2,370	2,280	45	0	0
5×10^{-7}	7,460,000	32,600	2,280	89	45	0
2.5×10^{-7}	23,100,000	287,000	9,490	2,280	89	0
1×10^{-7}	73,300,000	2,140,000	257,000	3,040	2,280	45

Note: Double counting of population around facilities within 50 km of each other may occur. Exposed individuals are included in the statistics for each plant within 50 km, presumably at different risk levels. See Section 6.5 and Appendix F for further discussion of double counting and related issues.

about 0.15 case/yr if the IURE was 20 percent nickel subsulfide, and about 0.1 case per year if the IURE was 10 percent nickel subsulfide. Likewise, there would be changes in the risk levels to which people were exposed. If the nickel mixture IURE were 50 percent as potent as nickel subsulfide about 100,000 people would be exposed at an MIR > 10^{-6}. If the IURE were 20 percent nickel subsulfide, about 9,930 persons would be exposed at an MIR > 10^{-6}. Nickel speciation uncertainty is discussed in more detail in section 6.10.

6.1.2.4 Noncancer Risks Due to Chronic Exposures. The highest HQ resulting from oil-fired utility emissions was 0.04 for manganese.

6.1.3 Baseline Risks from Gas-Fired Utilities

Risks were estimated from 267 gas-fired facilities. Table 6-5 summarizes the results. The HAP emissions from only one plant resulted in risks greater than 1 in 10 million (10^{-7}) with 23 persons exposed above that level. For noncarcinogens, the maximum HQ was 1×10^{-7}.

Figure 6-3. Estimates of Annual Cancer Incidence Due to Inhalation Exposure to HAP Emissions from Oil-fired Electric Utilities Based on the Local Analysis Using Alternative UREs for Nickel (as % of Nickel Subsulfide)

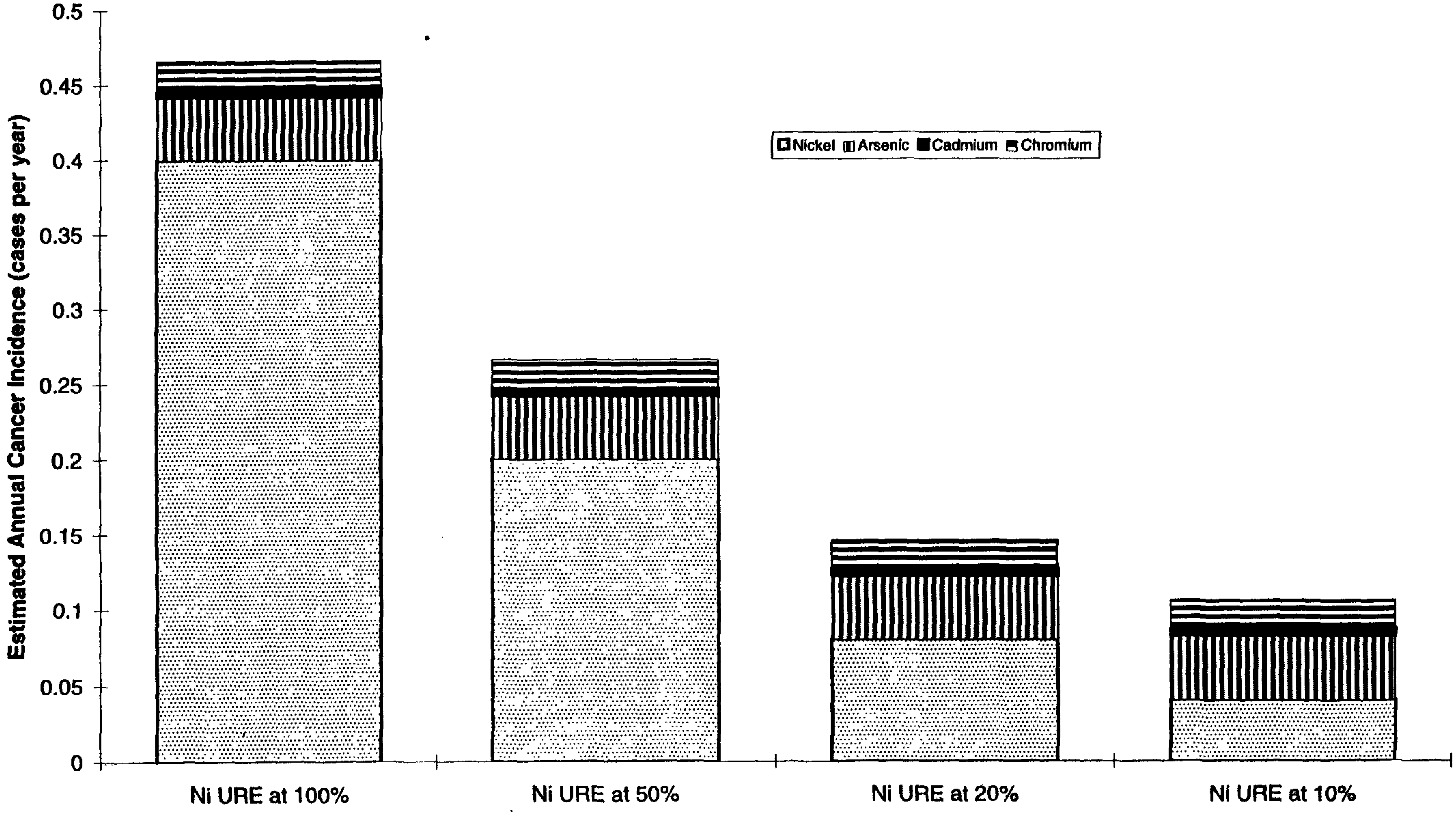


Table 6-5. Summary of Baseline Inhalation Risk for Gas-Fired Utilities

	Carcinogens			Noncarcinogen
Pollutant	MEI risk	Population MIR > 10^{-6}	# Plants MIR > 10^{-6}	HQ_{max}
Arsenic	2×10^{-7}	0	0	NA
Lead	NA	NA	NA	1×10^{-7}
Mercury	NA	NA	NA	NA
Nickel[a]	2×10^{-7}	0	0	NA

MEI = Maximum exposed individual, which is calculated using the highest annual average concentration. An individual may or may not be exposed at that point. This value may be greater than the MIR, which is calculated at the centroid of a census block.
MIR = Maximum individual risk is the highest risk identified at the centroid of a census tract to which a population is assigned.
NA = Not available
HQ = Hazard quotient, which is the ratio of exposure concentration to the reference concentration (RfC). HQ values below 1 are not expected to result in adverse effects.

[a] The nickel emitted is a mixture of various nickel compounds such as soluble nickel. This analysis assumes that all nickel emitted has the same carcinogenic potency as nickel subsulfide.

6.2 DISTINGUISHING BETWEEN URBAN VERSUS RURAL LOCATIONS

The HEM has two distinct modeling options (urban or rural) intended to simulate atmospheric dispersion behavior of gases via different surface roughness.[2] The urban option assumes that there are buildings near the emission source and that turbulence results because of these surfaces and other urban effects such as heat transfer from buildings and roadways. The rural option assumes that there are not any major buildings nearby, and therefore emissions dispersion plumes are not as turbulent. Typically, for tall stacks, the urban option predicts higher exposure concentrations, and therefore higher risks to nearby populations.

In the screening assessment, the urban option was used in all modeling runs. However, to assess the impact of this default option on the risk assessment results, all of the priority HAPs were modeled distinguishing between urban and rural locations. As an option provided by the U.S. EPA Guidelines on Air Quality Models (40 CFR, Appendix W to Part 51), it was assumed that, if 21,000 people lived within a 3-km radius of the plant (i.e., density = 750 people/km^2), then the area was urban and was modeled using the urban modeling option. If less than 21,000 people lived within a 3-km radius, then the area was considered rural and the rural modeling option was chosen.[2] Tables 6-6 and 6-7 present the results. There were some differences in site-by-site estimates. As Tables 6-6 and 6-7 show, choosing the default option versus a more refined selection of surface roughness options has some impact on the overall results. There were only slight changes in the results for oil-fired utilities. Many oil-fired facilities were estimated to be located in urban areas. The differences in the risk estimates from coal-fired utilities were greater. Generally, risk estimates are lower when urban and rural modeling distinctions are used.

The use of the refined analysis, whereby surface roughness distinction was made for urban and rural locations, was considered appropriate for the remainder of the inhalation exposure modeling analyses since it is believed to more realistically reflect the location of utilities and the impacts of rural and urban conditions on the dispersion of pollutants. Although the EPA believes using this distinction is appropriate, there are still uncertainties and limitations to this approach, which are discussed in later sections of this report.

6.3 INHALATION RISK ESTIMATES FOR THE YEAR 2010

The EPA analyzed potential inhalation risks from utility emissions for the year 2010. This analysis was conducted to estimate hazards and risks to public health after imposition of the requirements of the Clean Air Act. The primary differences between the 1990 and 2010 scenario are increased emissions from coal-fired utilities and decreased emissions from oil-fired utilities. Other predicted changes include the installation of scrubbers for a small number of facilities, the closing of a few facilities, and an increase in production of other facilities. The details of the expected changes are explained in chapters 2 and 3. Similar to any analyses that predict future events, significant uncertainties are associated with the method used for projecting risks of HAP emissions to the year 2010. The method used by EPA is considered reasonable given the available data. The exposures and risks were estimated using the HEM, utilizing the same modeling assumptions, defaults, and inputs used in the 1990 risk estimates, except that the emissions inputs were changed to 2010 estimates. Instead of modeling all 15 priority HAPs a second time, the EPA modeled a subset of HAPs that appear to present the majority of the risks from utility emissions. The analysis of this subset of priority HAPs provides information regarding the anticipated potential public health risks for the year 2010.

Table 6-6. Comparison of Inhalation Cancer Risk Estimates Based on (1) HEM Modeling Using Urban Default Assumption and (2) HEM Modeling Using Urban vs. Rural Distinction

	Urban default			Rural vs. urban		
HAP & fuel	**MEI risk**	**Cancer incidence (cases/year)**	**Population[a] with cancer risk> 10^{-6}**	**MEI risk**	**Cancer incidence (cases/year)**	**Population[a] with cancer risk> 10^{-6}**
As, from Coal	6×10^{-6}	0.08	21,000	3×10^{-6}	0.05	850
Cr, Coal (assuming 11% Cr VI)	3×10^{-6}	0.03	890	2×10^{-6}	0.02	110
Be, Coal	6×10^{-7}	0.006	0.0	3×10^{-7}	0.004	0.0
Cr, Oil (assuming 18% Cr VI)	5×10^{-6}	0.02	2,300	5×10^{-6}	0.02	2,300
Be, Oil	7×10^{-7}	0.002	0.0	7×10^{-7}	0.002	0
Cd, Oil	1.6×10^{-6}	0.007	45	1.6×10^{-6}	0.005	45
Ni, Oil	9×10^{-5}	0.5	2,300,000	9×10^{-5}	0.4	1,600,000
As, Oil	1×10^{-5}	0.05	4,600	1×10^{-5}	0.04	2,400

HAP = Hazardous air pollutant.
MEI = Maximum exposed individual.

[a] The number of people estimated to be exposed to ambient air concentrations causing an estimated increased risk of cancer of 1 in 1 million or greater.

Table 6-7. Comparison of Inhalation Noncancer Risk Estimates Based on (1) HEM Modeling Using Urban Default Assumption and (2) HEM Modeling Using Urban vs. Rural Distinction

	Urban default		Selection of appropriate setting (rural vs. urban)	
HAP & Fuel	**MEI HQ**	**# People above an HQ of 0.01**	**MEI HQ**	**# people above an HQ of 0.01**
HCl from Coal	2.3/20 = 0.12	157,000	2.3/20 = 0.12	15,100
Mn from Coal	0.02/0.05 = 0.4	104,000	0.002/0.05 = 0.04	27,900

HAP = Hazardous air pollutant.
HQ = Hazard quotient.
MEI = Maximum exposed individual.

The results (Tables 6-8 and 6-9) indicate that, based on the expected changes between 1990 and the year 2010, the inhalation risks from coal-fired utilities will not change substantially, and the risks from oil-fired utilities will decrease by roughly a factor of 2.

6.4 ASSESSMENT OF RISKS DUE TO SHORT-TERM EXPOSURE

The potential for exceeding short-term reference exposure levels (RELs)[3] was evaluated for compounds emitted from coal- and oil-fired utilities. The RELs (1-hour averages) are set to prevent adverse acute responses in the exposed population. The pollutants of highest concern were acrolein, HCl, and HF because these pollutants are potentially emitted in significant quantities and are toxic due to short-term (acute) exposures. Although the Agency has not determined RELs for these compounds, REL values were obtained from the California Air Pollution Control Officers Association (CAPCOA) *Air Toxics 'Hot Spots' Program Risk Assessment Guidelines*, October 1993.[3] The CAPCOA RELs are listed in Table 6-10.

The utilities modeled included the coal-fired and the oil-fired utility that presented the highest predicted long-term concentrations as determined from the earlier HEM screening analysis. In addition, the largest emitter of each compound from a coal-fired and oil-fired utility was modeled. Note that acrolein was not detected in the emissions tests for oil-fired utilities.

6.4.1 Methodology. The EPA used a short-term air dispersion model that considers all reasonable meteorological conditions (called TSCREEN) to estimate the maximum 1 hour concentration of the three compounds in the vicinity of selected coal- and oil-fired utilities. The TSCREEN provides estimates of 1-hour concentrations at various distances from the stack being analyzed. The user specifies the minimum distance to the stack at which concentrations will be predicted. For all utilities modeled, 100 meters from the stack was selected.

The reported concentrations are the maximum predicted from a range of atmospheric stability classes and windspeeds. The modeler must also specify whether urban or rural meteorological conditions exist at the utility site. Urban was selected to maximize the predicted concentrations.

Each of the selected plants emitted the HAPs from several stacks at the site. Because the TSCREEN model can evaluate only

Table 6-8. Estimated Inhalation Cancer Risks for the Year 2010 Compared to 1990 for Coal- and Oil-fired Utilities

	Cancer risk 2010			Cancer risk 1990		
Pollutant and fuel	**MEI risk**	**Cancer incidence (cases/year)**	**Population w/ MIR > 10^{-6}**	**MEI risk[a]**	**Cancer incidence (cases/year)**	**Population w/ MIR > 10^{-6}**
As from Coal	3×10^{-6}	0.051	590	3×10^{-6}	0.045	852
Be from Coal	3×10^{-7}	0.004	0.0	3×10^{-7}	0.0035	0.0
Cd from Coal	3×10^{-8}	0.0007	0.0	2×10^{-7}	0.0006	0.0
Cr from Coal (11% Cr VI)	1×10^{-6}	0.021	399	2×10^{-6}	0.02	107
Dioxins from Coal	2×10^{-9}	0.0012	0.0	5×10^{-8}	0.001	0.0
Ni from Coal	3×10^{-7}	0.006	0.0	7×10^{-7}	0.005	0.0
n-Nitroso-dimethylamine from Coal	8×10^{-7}	0.011	0.0	8×10^{-7}	0.016	0.0
Cr from Oil (18% Cr VI)	3×10^{-6}	0.009	89	5×10^{-6}	0.02	2,300
Be from Oil	4×10^{-7}	0.0008	0.0	7×10^{-7}	0.0017	0.0
Cd from Oil	8×10^{-7}	0.0026	0.0	2×10^{-6}	0.0053	45
Ni from Oil	5×10^{-5}	0.2	240,000	1×10^{-4}	0.40	1,600,000
Dioxins from Oil	7×10^{-8}	0.0004	0.0	1×10^{-7}	0.0007	0.0
As from Oil	7×10^{-6}	0.026	2,300	1×10^{-5}	0.042	2,400

Note: The EPA used Urban vs. Rural modeling data distinction in this analysis.

[a] *These MEI risk estimates are for the "highest risk" plant.*

[b] This is the estimated cases of cancer predicted to occur in the U.S. due to emissions of this HAP from all utilities of that fuel type based on the HEM analysis.

Table 6-9. Estimated Inhalation Noncancer Risks for Coal-fired Utilities for the Year 2010 Compared to the Year 1990

HAP	**RfC ($\mu g/m^3$)**	**Highest MEI Conc. for 2010**	**Maximum HQ for 2010**	**Highest MEI Conc. for 1990**	**Maximum HQ for 1990**
HCl	20	2.6 $\mu g/m^3$	0.1	2.3 $\mu g/m^3$	0.1
Manganese	0.05	0.003 $\mu g/m^3$	0.06	0.002 $\mu g/m^3$	0.05

Table 6-10. Noncancer Reference Exposure Levels[3] (Acute) from CAPCOA

Pollutant	REL -- Hourly average concentration ($\mu g/m^3$)
Acrolein	2.5
Hydrochloric acid	3,000
Hydrogen fluoride	580

one emission point at a time, some adjustments were required for each utility's emission parameters. The concept was to select one stack and one emission rate with one set of stack parameters that would represent the multiple stacks and their corresponding emissions and stack parameters. If the stacks at each utility varied in height or other release characteristics (e.g., stack temperature, stack gas exit velocity), emissions were assumed to be emitted under conditions to maximize downwind concentrations: from the shortest stack present, the lowest temperature among the stack characteristics, and the lowest exit velocity (see Table 6-11). The emissions rate was calculated by summing the emissions from each stack.

To illustrate this methodology, a sample utility is presented in Table 6-12. The resulting inputs to the dispersion model for this sample utility would have been one stack with a stack height of 70 m, exit velocity of 12 m/s, and temperature of 390 K. An average of the inside stack diameters for the four stacks would be used (see Table 6-12).

6.4.2 Results

As shown in Table 6-13, for all scenarios and all pollutants modeled, the predicted maximum concentrations were more than 100 times lower than the RELs. The emission rate used for each compound represents an average. The analysis does not address peak short-term emissions that may result from upsets or other atypical operations. Peak emission episodes would reduce the gap between predicted maximum concentrations and REL, but the peak hourly emission rates are not expected to be 100-fold higher than the average.

The TSCREEN can also incorporate terrain characteristics. Terrain was not considered an important factor in the analysis since the utilities that caused the highest individual risk in the cancer analysis were located in relatively flat terrain. (The

Table 6-11. Sample Stack Parameters for Typical Utility Plant

Stack	Stack height (m)	Exit velocity (m/s)	Stack temperature (K)
1	75	15	400
2	75	15	400
3	70	22	390
4	75	12	410

Table 6-12. Stack and Emission Values Input to TSCREEN

Pollutant	Stack height (m)	Stack gas exit velocity (m/s)	Stack diameter (m)	Stack gas temperature (K)	Emission rate (g/s)
COAL					
HF	49	45.7	2.5	395	0.42
HCl	49	47.5	2.5	395	9.07
Acrolein	49	45.7	2.5	395	0.01
OIL					
HF	42	12.3	3.0	396	0.06
HCl	42	12.3	3.0	396	1.24

Table 6-13. Results of the TSCREEN Model

Pollutant	Reference exposure levels (hourly avg $\mu g/m^3$)	Coal-fired maximum predicted concentration (hourly avg $\mu g/m^3$)	Oil-fired maximum predicted concentration (hourly avg $\mu g/m^3$)	How much lower? AAC/Pred	
				coal	oil
Acrolein	2.5	0.016	not emitted	150	
HCl	3,000	21.5	5.5	140	1,200
HF	580	1.0	0.3	580	2,100

Note: Since the largest emissions are generally associated with taller stacks, other analyses indicated that the estimated concentrations were generally a factor of 2 lower than that presented.

effects of terrain are analyzed in Appendix G.) Although hilly terrain can cause an estimated 15-fold higher predicted long-term concentration than flat terrain, this increase would still not result in exceedances of RELs for the three compounds.

6.5 OVERLAPPING PLUMES/DOUBLE COUNTING

In general, the default standard mode of operation for the HEM is to evaluate exposure to each source, one at a time, out to 50 km from the plant. Each source's exposure is independently estimated, and detailed exposure estimates are not saved for the next source's exposure analysis. Summary information, such as the total numbers of people who are exposed, is saved. Thus, if two plants are located very close together, the HEM would independently estimate the total number of people exposed to each plant's emission and sum the two totals even though the same people are being exposed to both plant's emissions. In this mode, the HEM will most likely overestimate the number of people who are exposed when two or more plants are within 50 km of each other. This effect has been called "double-counting." Although not intuitive, experience has shown that this effect is not of great concern when estimating the risks to the MEI and to the population as a whole. Because of the linear nature of the exposure and risk models, the population risks (cancer cases per year) are the same whether one calculates the exposure one plant at a time or calculates the exposure from nearby plants together; only the number of people who are estimated to be in the exposed group will differ. In the case of the risk to the MEI, nearby plants can only significantly change the estimated maximum concentration when plants of equal emission rates are located very close to each other, perhaps within several hundred meters. This is very unlikely for the utility industry.

There is an option to the HEM, called single-count, which can provide further insight into this potential problem of double-counting. This option still evaluates exposure on a source-by-source basis, but exposure is calculated for each population census block within 50 km and this detailed information is saved (stored in the computer memory). As each source is considered, the exposure estimate for each census block is added to the previous source's exposure estimates at the same census block. At the end of the computer run, the computer has a total exposure estimate for each census block in the U.S. and, by adding the census block exposure estimates together, provides a national level estimate of total exposure. For this study, single-count HEM runs have indicated that individuals may be living within 50 km of up to 12 coal-fired plants or 17 oil-fired plants; thus, a concern has arisen over multiple exposures to many plants. However, the single-count analyses conducted for arsenic emissions indicate that overlapping effects from nearby sources do not significantly change the estimated risks (see Table 6-14).

6.6 ASSESSMENT OF EXPOSURE DUE TO LONG-RANGE TRANSPORT

6.6.1 History and Background Information

During the mid-1970s, SRI International developed a Lagrangian puff air pollution model called the EUROPEAN Regional Model of Air Pollution (EURMAP) for the Federal Environment Office of the Federal Republic of Germany.[4] This regional model simulated monthly SO_2 and sulfate (SO_4^{2-}) concentrations, wet and dry deposition patterns, and generated matrices of international exchanges of sulfur for 13 countries of western and central Europe. In the late-1970s, the EPA sponsored SRI International to adapt and apply EURMAP to eastern North America. The adapted version of this model, called Eastern North American Model of Air Pollution (ENAMAP), also calculated monthly SO_2 and SO_4^{2-} concentrations and wet and dry deposition patterns and generated matrices of interregional exchanges of sulfur for a user-defined configuration of regions.[4,5], In the early-1980s, EPA modified and improved the ENAMAP model to increase its flexibility and scientific credibility.

By 1985, simple parameterizations of processes involving fine (diameters < 2.5 μm) and coarse (2.5 μm < diameters < 10.0 μm) PM were incorporated into the model. This version of the model, renamed the Regional Lagrangian Model of Air Pollution (RELMAP), is capable of simulating concentrations and wet and dry deposition patterns of SO_2, SO_4^{2-}, and fine and coarse PM and can also generate source-receptor matrices for user-defined regions. In addition to the main model program, the complete RELMAP modeling system includes 19 preprocessing programs that prepare gridded meteorological and emissions data for use in the main program. A complete scientific specification of the RELMAP as used at EPA for sulfur modeling is provided in *RELMAP: a Regional Lagrangian Model of Air Pollution - User's Guide.*[6] The next section discusses modifications made to the original sulfur version of RELMAP to enable the simulation of atmospheric arsenic.

6.6.2 RELMAP Modeling Strategy for Atmospheric Arsenic

6.6.2.1 Introduction. Previous versions of RELMAP have been described by Eder et al.[6] and Clark et al.[7] The goal of the current effort was to model the emission, transport, and fate of airborne arsenic from utilities in the continental U.S. for the

Table 6-14. Comparison of Risk Estimates for Single-Count Versus Double-Count Runs to Assess the Impact of Overlapping Plumes

HAP, fuel, year	Single-count runs			Double-count runs		
	MEI risk	Incidence	Population w/ risk > 10^{-6}	MEI risk	Incidence	Population w/ risk > 10^{-6}
As, Coal, 1990	3×10^{-6}	0.05	850	3×10^{-6}	0.05	850
As, Oil, 1990	1×10^{-5}	0.04	2,200	1×10^{-5}	0.04	2,400
As, Coal, 2010	3×10^{-6}	0.05	590	3×10^{-6}	0.05	590
As, Oil, 2010	7×10^{-6}	0.03	2,200	7×10^{-6}	0.03	2,300

year 1989. Modifications to the RELMAP for atmospheric arsenic simulation were based on the assumption that all arsenic emissions are in particulate form.

The RELMAP may be run in either of two modes. In the field mode, wet deposition, dry deposition, and air concentrations are computed at user-defined time intervals. In the source-receptor mode, RELMAP also computes the contribution of each source cell to the deposition and concentration at each receptor cell. For this study, only the field mode of RELMAP operation was used. With over 10,000 model cells in the high-resolution receptor grid and a significant fraction of these cells also emitting arsenic, the data accounting task of a source-receptor run for all electric utility sources could not be performed with the computing resources and time available.

Unless specified otherwise in the following sections, the modeling concepts and parameterizations described by Eder et al.[6] were preserved for the RELMAP arsenic modeling study.

6.6.2.2 Physical Model Structure. Because of the long atmospheric residence time of fine PM, significant long-range transport of arsenic was expected. For this study, RELMAP simulations were limited to the area bounded by 25 and 55 degrees north latitude and 60 and 130 degrees west longitude and with a minimum spatial resolution of one-half degree longitude by one-third degree latitude (approximately 40 km^2) to provide high-resolution coverage over the entire continental U.S.

Since the descriptive document by Eder et al.[6] was produced, the original three-layer puff structure of the RELMAP has been replaced by a four-layer structure. The following model layer definitions were used for the RELMAP arsenic simulations:

Layer 1 top - 30 to 50 m above the surface (season-dependent)

Layer 2 top - 200 m above the surface

Layer 3 top - 700 m above the surface

Layer 4 top - 700 to 1,500 m above the surface (month-dependent).

6.6.2.3 Treatment of Emissions. All of the utilities within each high-resolution RELMAP grid cell were treated as a single integrated point source located at the center of the grid cell. As mentioned in earlier sections of this report, the utility database contained the necessary information to satisfy the RELMAP data needs, including long-term arsenic emission rates, stack parameters, and plant location. All point source emissions (assumed to be in steady state) were introduced into model layer 2 to account for the effective stack height of the point source type in question. Effective stack height is the actual stack height plus the estimated plume rise. The layer of emission is inconsequential during the daytime when complete vertical mixing is imposed throughout the four layers. At night, since there is no vertical mixing, source emissions to layer 1 are subject to dry deposition while point source emissions to layer 2 are not. Large industrial emission sources and sources with very hot stack emissions tend to have a larger plume rise, and their effective stack heights might actually be larger than the 700-m top of layer 2. However, since the layers of the pollutant puffs remain vertically aligned during advection, the only significant process affected by the layer of emission is nighttime dry deposition.

6.6.2.4 Lagrangian Transport and Deposition. In the model, each pollutant puff begins with an initial mass equal to the total emission rate of all sources in the source cell multiplied by the model time-step length. For arsenic, as for most other pollutants, emission rates for each source cell were defined from input data and a time step of 3 hours was used. The initial horizontal area of each puff was set to 1,200 km^2, instead of the standard initial size of 2,500 km^2, in order to accommodate the finer grid resolution used for the modeling study; however, the standard horizontal expansion rate of 339 km^2/h was not changed. Although each puff was defined with four separate vertical layers, each layer of an individual puff was advected through the model cell array by the same wind velocity field. Thus, the layers of each puff always remained vertically stacked. Wind field initialization data for a National Weather

Service prognostic model, the Nested Grid Model (NGM), were obtained from the National Oceanographic and Atmospheric Administration's (NOAA's) Atmospheric Research Laboratory for the entire year of 1989. Wind analyses for the vertical level of approximately 1,000 meters above ground level of the NGM were used to define translation of puffs across the model grid, except during the months of January, February, and December when the 600-m vertical level was used to reflect a more shallow mixed layer.

Pollutant mass was removed from each puff by the processes of wet deposition and dry deposition. The model parameterizations for these processes are discussed in section 6.6.3. Precipitation data for the entire year of 1989, obtained from the National Climatic Data Center, were used to estimate the wet removal of all pollutant species modeled. Wet and dry deposition mass totals are accumulated and average surface-level concentrations are calculated monthly for each model cell designated as a receptor. Except for cells in the far southwest and eastern corners of the model domain where there were no wind data, all cells were designated as a receptor for the arsenic simulation. When the mass of pollutant on a puff declines to a user-defined minimum value, or when a puff moves out of the model grid, the puff and its pollutant load is no longer tracked. The amount of pollutant in the terminated puff is taken into account in monthly mass balance calculations so that the integrity of the model simulation is assured. Output data from the model include monthly wet and dry deposition totals and monthly average air concentrations for each modeled pollutant in every receptor cell.

6.6.3 Model Parameterizations

6.6.3.1 Chemical Transformation. The simplest type of pollutant to model with RELMAP is the inert type. To model inert pollutants, one can simply omit chemical transformation calculations for them and not be concerned with chemical interactions with other chemical species. Arsenic was treated as an inert pollutant species.

6.6.3.2 Dry Deposition. All atmospheric arsenic was assumed to be in particulate form. Since arsenic and its compounds make up only a small fraction of total PM loading of the atmosphere, it was treated as a minor component of the general population of conglomerate aerosol particles. Heavy metals have been generally associated with fine particle sizes (<1 μm diameter), but there is evidence larger particles may play a significant role in dry deposition in urban areas.[5,6] Therefore, arsenic particles were modeled in five sizes; 0.1, 0.3, 1.0, 3.0, and 10.0 μm diameter. The results of the RELMAP simulation for each particle size were then used in a postprocessing operation to estimate effects of dry deposition on a particle size distribution appropriate for regional-scale air masses with urban influences. The following percent fractions of particle mass were assumed to be in each size class: 20 percent in 0.1 μm, 50 percent in 0.3 μm, 20 percent in 1.0 μm, 5 percent in 3.0 μm, and 5 percent in 10.0 μm.

The semi-empirical dry deposition model as described in Sehmel[8] was used to estimate dry deposition velocity for conglomerate particles in the 5-μm size class. This model requires as input the particle density, the particle diameter, the friction velocity, the Monin-Obhukov length, the surface roughness length, and the air temperature. Assuming sulfate, nitrate, and organic compounds make up most of the particulate mass for particles less than 10 μm in diameter, a density value of 2 g/cm^3 was used to represent all particles containing arsenic. Although arsenic and most of its compounds have densities of over 2 g/cm^3, it was assumed that they make up only a small part of the conglomerate aerosol particles in the modeled size range. Dry deposition velocities for particulate arsenic were calculated using a FORTRAN subroutine developed by the CARB.[9] Table 6-15 shows the windspeed (μm)(m/s) used for each Pasquill stability category in the calculation of deposition velocity from the CARB subroutine, and Table 6-16 shows the roughness length used for each land-use category.

6.6.3.3 Wet Deposition. Alcamo et al.[10] used a scavenging ratio of 0.5 x 10^6 for both arsenic and cadmium, noting that these values are close to the average values reported by Chan et al.[11] They also note that Chan et al. did not address As, but used values based on measurements in Canada, not Europe. Schroeder et al.[12] show a range of measured values for the scavenging ratio of arsenic, cadmium, and lead.

Some of the parameters used for dispersion and deposition modeling are shown in Tables 6-15 and 6-16. These parameters include windspeed vs. stability category and roughness length vs. land-use category. As a compromise, a scavenging ratio of 0.25 x 10^6 was used for the spring and summer seasons and a value of 0.40 x 10^6 was used for the autumn and winter seasons.

6.6.4 Exposure and Risk Estimates

The RELMAP analysis produced an average annual air arsenic concentration for each grid cell in the continental U.S. The maximum annual arsenic RELMAP concentration was 0.28 ng/m^3. Of the 12,600 grid cells in the study area, 33 grid cell

Table 6-15. Windspeeds Used for Each Pasquill Stability Category in CARB Subroutine Calculations

Stability category	Windspeed (m/s)
A	10.0
B	5.0
C	5.0
D	2.5
E	2.5
F	1.0

Table 6-16. Roughness Length Used for Each Land-Use Category in CARB Subroutine Calculations

	Roughness length (ms)	
Land-use category	Autumn-winter	Spring-summer
Urban	0.5	0.5
Agricultural	0.15	0.05
Range	0.12	0.1
Deciduous Forest	0.5	0.5
Coniferous Forest	0.5	0.5
Mixed Forest/Wetland	0.4	0.4
Water	10^{-6}	10^{-6}
Barren Land	0.1	0.1
Nonforested Wetland	0.2	0.2
Mixed Agricultural/Range	0.135	0.075
Rocky Open Areas	0.1	0.1

concentrations were greater than 0.100 ng/m^3. The 50th percentile of the grid cell values was 0.0034 ng/m^3 and the average cell value was 0.01 ng/m^3. The figure indicates that, typically, the largest concentrations occur in the eastern part of the U.S.

Once the grid cell concentrations are known, public exposure and risks can be calculated by applying the population database used in the HEM. This population database contains the location of and number of people living within each census block. By overlaying the population database onto the grid cells, exposure can be estimated for each group of people within the continental U.S. By multiplying the grid cell arsenic concentration by the number of people within that grid cell and summing these products over all the U.S. grid cells, one can estimate total population exposure. To estimate annual population risks (cancer incidence), the model multiplied the total exposure product by the arsenic IURE and divided by 70 years. The results are shown in Figure 6-4 and Table 6-17.

To evaluate potential impacts due to long-range transport, the coal, oil, and gas emissions were modeled together. By applying the algorithm described above, 0.6 cancer case/year was estimated, for all three fuels, for arsenic emissions from utilities in the continental U.S. This estimate is about seven times greater than the population risks estimated modeling arsenic emissions within 50 km of each facility using the HEM (i.e., 0.05 case/year for coal and 0.04 case/year for oil).

The potential impacts to the MEIs appear to be considerably less than for population exposures. The maximum RELMAP concentration of 0.27 ng/m^3 is about 40 percent of the highest HEM arsenic concentration for coal-fired utilities. The modelers expect that the other metals of potential concern (e.g., chromium, nickel, cadmium) would show similar results. These trace metals are also associated with fine particulate matter in the utility emissions and probably act in a similar manner in the atmosphere. In addition, these other HAPs are generally emitted in roughly proportional quantities for each fuel type and are emitted from the same set of plant locations. Therefore, for a screening exercise, the factor of 7 can be applied to these other HAPs to roughly estimate the potential impact of long-range transport of HAPs on the overall cancer incidence. In the HEM analysis (i.e., within 50 km) the total cancer incidence (not including radionuclides) was estimated to be up to 0.6 case per year (0.1 case per year for coal-fired utilities and 0.47 case per year for oil-fired utilities). Multiplying the 0.6 case per year by the factor of 7 results in a cancer incidence estimate

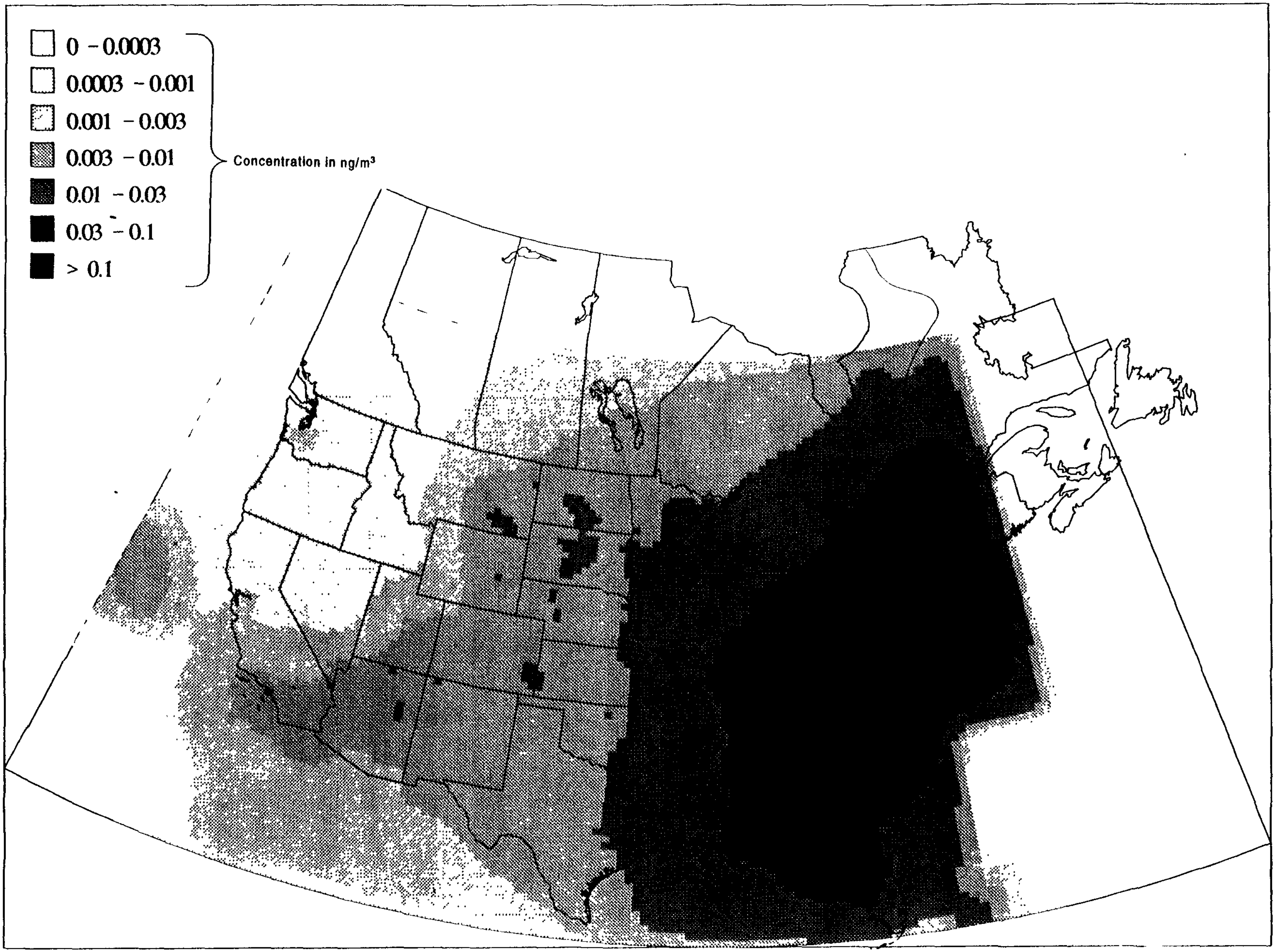


Figure 6-4. Results of the RELMAP Analysis for Arsenic for all Utilities: Air Concentration of Arsenic, Units: ng/m^3

Table 6-17. Exposure and Risk Estimates Based on RELMAP Modeling of Arsenic Emissions from All Oil- and Coal-fired Utilities in the U.S.

Exposure Concentration ($\mu g/m^3$)	Inhalation Risk[a]	Number of People Exposed to this Level of Risk
2.5×10^{-4}	1×10^{-6}	42,000
1×10^{-4}	4×10^{-7}	11,700,000
5×10^{-5}	2×10^{-7}	96,400,000
2.5×10^{-5}	1×10^{-7}	147,000,000
1×10^{-5}	4×10^{-8}	183,000,000
5×10^{-6}	2×10^{-8}	204,000,000
2.5×10^{-6}	1×10^{-8}	221,000,000
1×10^{-6}	4×10^{-9}	230,000,000

[a] Based on standard cancer risk equation, assuming 70-year exposure for people living in exposure grids (described in chapters 4 and 6).

of roughly four cancer cases/year due to emissions of nonradionuclide HAPs from coal- and oil-fired utilities (including radionuclides, which are analyzed in chapter 9, cancer incidence is estimated to be as high as 6 cases/year). Assuming that the factor of 7 applies equally to oil and coal utilities, the cancer incidence for coal-fired utilities for nonradionuclide HAPs is estimated to be roughly 0.7 case/year (i.e., 0.1 multiplied by 7) and that the cancer incidence for oil-fired utilities is roughly three cases/year (i.e., 0.47 x 7).

However, there are numerous uncertainties in the modeling, the assumptions, the extrapolations, and the resulting cancer incidence estimates. For example, the long-range transport of emissions from oil-fired utilities may be different than the long-range transport of emissions from coal-fired utilities. Also, since the exposure concentrations for much of the exposed population are quite low, this analysis relies heavily on the assumption of cancer being a nonthreshold phenomenon and the assumption that the dose-response curve for these carcinogens is linear at very low doses. Also, since nickel is the HAP contributing most to the cancer risks in the HEM analysis, the cancer incidence estimate for oil-fired utilities (i.e., three cases per year) and the overall cancer estimate (i.e., four cases per year) is heavily dependent on the assumption that the mix of nickel compounds is as carcinogenic as nickel subsulfide. Because of these and other uncertainties, the cancer incidence estimates and the extrapolation factor of 7 should be viewed with caution. The resulting cancer incidence estimates are considered high-end, conservative estimates. Further evaluation of the data, models, and methods is needed to reduce the uncertainties and to fully evaluate the impacts of long-range transport.

6.7 DISCUSSION OF BACKGROUND EXPOSURES

6.7.1 Arsenic

There are over 250 sites that reported ambient arsenic data to the EPA's Aerometric Information Retrieval System. Up to the year 1987, arsenic was measured by performing an analysis of the filter catch from 24-hour high-volume total-suspended-particulate (TSP) sampling devices. The Agency compared the results of the dispersion modeling to available data in 1987 (latest available data) and attempted to provide insight into typical arsenic concentrations in areas away from utilities and to provide a check on the credibility of the predicted concentrations.

A review of 1987 ambient arsenic data indicated that the minimum concentration that could be detected was about 3 ng/m^3. Much of the reported data were at or below the minimum detectable level (MDL); for instance, 145 of the 261 total sites reported no values above the MDL. At sites not located near known, large arsenic emitters, such as copper smelters, the largest annual concentration reported was about 8 ng/m^3. Further analysis indicated the large majority (about 75 percent) of monitors were located within 50 km of at least one coal- or oil-fired utility plant, and six sites were located within 50 km of at least 10 plants. On the other hand, there were 59 sites that were not within 50 km of any coal- or oil-fired utility plant.

Typical arsenic concentrations can be determined by reviewing the data from the 59 sites not near utility plants. Of the 59 sites, 8 were known to be near large arsenic sources and were not representative of typical sites. Only 13 of the remaining 51 sites recorded annual arsenic concentrations above the MDL. The highest concentration reported was about 8 ng/m^3. Thus, based on these data, typical concentrations are probably not much higher than 8 ng/m^3 and are most likely to be lower (or much lower) than the MDL of 3 ng/m^3. In fact, as seen from review of the data collected near utilities, this result is typical of all the available arsenic data, when the monitors are not located near large arsenic-emitting sources.

Next, a comparison was made between the predicted arsenic concentrations and the measured values near the plants. The highest arsenic long-term concentration estimated for any utility plant using the HEM was about 3 ng/m^3. The estimated maximum concentrations predicted with the HEM for all the other utility plants were lower or much lower than 3 ng/m^3. The monitor nearest the plant that caused the maximum arsenic concentration was about 12 km away and that monitor did not register any concentrations above the MDL of 3 ng/m^3. The air dispersion analysis using the HEM predicted an arsenic concentration of 0.05 ng/m^3 at that monitoring site, so concentrations due to utility emissions were not expected to register on this monitor. At the site where the highest arsenic concentration was reported, the air dispersion analysis predicted arsenic concentrations well below 0.01 ng/m^3.

However, direct comparisons between estimated and measured values can be misleading. As suggested by the analysis of sites away from where arsenic concentrations were detected, there are confounding factors. One confounding factor occurs because arsenic is a naturally occurring element in the earth's crust. Some arsenic is expected to be in every TSP filter catch (i.e., a natural background concentration that would be present even without nearby anthropogenic sources). There is a second confounding factor because any other PM-arsenic source in the area will also have an impact on the monitor. So, for arsenic, the monitored concentrations are measuring a combination of concentrations: (1) from natural background, (2) from other arsenic sources, and (3) from nearby utilities. Thus, the monitored values are always expected to exceed the impact from the plant's emissions.

Considering the above information, it is difficult to draw definitive conclusions from the data comparisons. The largest concentration from the monitored data set was about 8 ng/m^3 and this site was near two plants. Similarly, over half of the monitors never detected annual arsenic concentrations above the MDL of 3 ng/m^3, so the highest possible impact at the typical monitoring site must be below 3 ng/m^3. The analysis indicates that the predicted concentrations from the HEM arsenic air dispersion analyses were not radical underestimates of actual plant emission impacts.

6.7.2 Chromium, Nickel, Manganese, and HCl

Chromium and nickel ambient data were also available. The results in analyzing these data led to conclusions similar to those drawn from the arsenic analysis. Much of the data were below detectable levels and did not provide much insight into the relative concentration impacts from utility emissions.

Based on the HEM modeling, manganese and HCl were the two HAPs that appear to be of highest potential concern for noncancer effects due to inhalation exposure. However, in the assessment of noncancer health effects due to inhalation exposure to HAP emissions from utilities, the highest HEM-modeled concentrations of manganese and HCl from the highest-risk plants were estimated to be 10 times lower than the RfC. All other HEM-modeled concentrations for HCl and manganese were even lower. Therefore, regardless of background exposure levels, the emissions of HCl and Mn from utilities are not likely to contribute significantly to an RfC problem. For this reason, the EPA did not conduct an analysis of ambient air background exposures for these two HAPs for this report.

6.8 CHROMIUM SPECIATION UNCERTAINTY AND IMPACT ON RISK ESTIMATES

Available health effects data indicate there are significant differences in the toxicity of the trivalent chromium (Cr III) versus hexavalent chromium (Cr VI). Chromium VI is classified as a human carcinogen (WOE = A) based on human and animal studies that show an increase in lung cancer. Available data are not sufficient to determine the carcinogenicity of Cr III (WOE = D). Cr III appears to be much less toxic than Cr VI.[13,14] However, there are uncertainties in the health effects of Cr III. For more information on chromium toxicity see Appendix E.

Data on speciation of chromium were available from 11 test sites. The limited emissions speciation data (see Appendix H) indicate that somewhere between 0.4 percent and 34 percent of the emitted chromium is chromium VI. The average chromium VI from the coal-fired utilities was 11 percent; the average from oil-fired utilities was 18 percent.

To assess the potential impact of the range of chromium speciation on the risk results, the utilities were modeled using the HEM assuming different speciation percentage assumptions. Tables 6-18 and 6-19 present the results of the assessment.

6.9 ISSUES WITH ARSENIC CANCER UNIT RISK ESTIMATE AND IMPACT ON INHALATION RISK ESTIMATES

Arsenic is considered a human carcinogen (WOE = A). The EPA reviewed the dose-response data in 1986 and established an IURE of 4.29×10^{-3} per $\mu g/m^3$. This IURE is the EPA-verified value

Table 6-18. Chromium Speciation Analysis for Coal-fired Utilities: Inhalation Risk Estimates due to Chromium Emissions Based on Various Assumptions of Percent Cr VI.

% Chromium VI assumption[a]	Lifetime MEI risk	Lifetime MIR	Population w/ > 10^{-6} lifetime cancer risk	Cancer incidence (cases/year)
Assume 100% Cr VI	2×10^{-5}	1×10^{-5}	69,000	0.2
Assume 23% Cr VI	4×10^{-6}	2×10^{-6}	2,300	0.04
Assume 11% Cr VI	2×10^{-6}	1×10^{-6}	110	0.02
Assume 0.4% Cr VI	7×10^{-8}	4×10^{-8}	0.0	0.0007

[a] Based on speciation data from emissions tests for four coal-fired test sites, the average percent Cr VI was 11 percent, the maximum was 23 percent, and the minimum was 0.4 percent. The remaining chromium emissions are assumed to be Cr III. It is assumed that the cancer risk is due only to Cr VI emissions. Because carcinogenicity data for chromium III are very limited and uncertain, it was assumed that Cr III does not pose cancer risk. It is not known whether the Cr III emissions contribute to the cancer risk.

Table 6-19. Chromium Speciation Analysis for Oil-fired Utilities: Inhalation Risk Estimates due to Chromium Based on Various Assumptions of Percent Chromium VI

% Chromium VI assumption[a]	Lifetime MEI risk	Lifetime MIR	Population w/ > 10^{-6} lifetime cancer risk	Cancer incidence (cases/year)
100% Cr VI	3×10^{-5}	3×10^{-5}	40,000	0.1
34% Cr VI	1×10^{-5}	1×10^{-5}	2,300	0.04
18% Cr VI	5×10^{-6}	5×10^{-6}	2,300	0.02
5% Cr VI	1.5×10^{-6}	1.5×10^{-6}	45	0.005

[a] Based on limited speciation data from emissions tests for seven oil-fired test sites, the average percent Cr VI was 18 percent, the maximum was 34 percent, and the minimum was 5 percent Cr VI, it was assumed that chromium III does not pose a cancer risk. It is assumed that the remainder of the chromium emissions are Cr III. It is assumed that the cancer risk is due only to Cr VI emissions.

currently available on IRIS. A more in-depth discussion of the cancer health effects data is provided in Appendix E.

The EPRI submitted a paper on arsenic carcinogenicity to the EPA IRIS office. This paper suggested that the IURE should be approximately three times lower than the current EPA-verified value as a result of reviewing new data. The EPRI asked the EPA to review the new data and consider revising the arsenic unit risk estimate based on the most current data and analyses.

The EPA has initiated the review process. However, to conduct a thorough review and analysis of the data and to calculate a new risk estimate is time consuming. A full review and IRIS update could not be completed in time for this report. However, the EPA has done a cursory review of the paper submitted by EPRI along with other relevant data. Based on this initial review by EPA scientists, it appears that the EPRI-proposed IURE is within the range of plausible estimates of cancer potency.[15] The Canadians have also reviewed the available data recently and established an IURE of 6×10^{-3}. The Canadian IURE also appears to be within the plausible range of potency for arsenic.[15] Table 6-20 compares EPRI, EPA-verified, and Canadian inhalation risk estimates.

Since a full review of the unit risk could not be completed in time for this report, and to help characterize the potential range of risk due to arsenic exposure, an assessment was conducted that presents the estimated risks due to inhalation exposure using three different IUREs (Table 6-20). It should be noted that this presentation does not present the full range of uncertainty, but rather presents the impact on the results due to the three different estimates of the unit risk.

6.10 NICKEL SPECIATION UNCERTAINTY AND IMPACT ON RISK ESTIMATES

There are significant uncertainties associated with nickel speciation. Nickel exists in four different valence states and can be combined with many other elements to form different nickel compounds. Numerous nickel compounds are known to exist.[16]

Total nickel was measured at nearly all sites, but only two sites (both oil-fired utilities) provided data on speciated nickel. The species measured were soluble nickel (water-soluble salts such as nickel sulfate and nickel chloride), sulfidic nickel (such as nickel subsulfide, nickel monosulfide, and nickel sulfide), metallic nickel (including alloys), and oxidic nickel (including nickel oxide, complex oxides, and silicates). The average values of the two test sites were: 58 percent soluble nickel, 3 percent sulfidic nickel, and 39 percent nickel oxides (see Appendix H).

The available health effects data vary significantly from species to species. Human epidemiologic data indicate that at least some forms of nickel are carcinogenic to humans by inhalation exposure.[17,18] Nickel refinery dust and nickel subsulfide are classified as human carcinogens (WOE = A). The IUREs for nickel refinery dust and nickel subsulfide are

Table 6-20. Arsenic Inhalation Risk Estimates: Comparison of Results Using the EPRI, EPA-verified, and Canadian IURE

	Risk estimates using EPRI IURE[a] (1.4 x 10^{-3} per $\mu g/m^3$)			Risk estimates using EPA IURE[2] (4.3 x 10^{-3} per $\mu g/m^3$)			Risk estimate w/ Canadian IURE (6 x 10^{-3} per $\mu g/m^3$)		
	MEI risk	# > 10^{-6}	Incidence	MIR	# > 10^{-6}	Incidence	MIR	# > 10^{-6}	Incidence
Arsenic from Oil-fired Utilities	4 x 10^{-6}	2,200	0.014	1 x 10^{-5}	2,400	0.042	2 x 10^{-5}	3,000	0.05
Arsenic from Coal-fired Utilities	6 x 10^{-7}	0.0	0.015	3 x 10^{-6}	850	0.045	4 x 10^{-6}	850	0.06

[a] The EPRI IURE for arsenic (1.4 x 10-3 per $\mu g/m^3$) is three times lower than the EPA-verified IURE for arsenic (4.3 x 10^{-3} per $\mu g/m^3$). And, the Canadian value is approximately 35 percent greater than the EPA estimate.[15]

2.4 x 10^{-4} and 4.8 x 10^{-4}, respectively. Nickel carbonyl is classified as a probable human carcinogen (WOE = B2), but no IURE has been established. These are the only species currently classified by the EPA as carcinogens. The IARC considers nickel monoxide, nickel hydroxide, and metallic nickel as having sufficient evidence in experimental animals for carcinogenicity.[18] The IARC considers nickel compounds to be carcinogenic to humans and metallic nickel to be possibly carcinogenic. The State of California concludes that the class of nickel compounds is potentially carcinogenic by inhalation.[19] The American Conference of Governmental Industrial Hygienists (ACGIH) has stated that all nickel compounds should be considered carcinogenic.[20] However, there are still significant uncertainties regarding the carcinogenicity of many of the nickel compounds. Available data are insufficient to confirm the carcinogenicity of many nickel compounds.

Cancer IUREs are only available for nickel subsulfide and nickel refinery dust. The cancer potency of the other nickel compounds that may be carcinogenic is not known. Results of animal studies suggest that nickel subsulfide is the most carcinogenic form.[17,18] Based on the limited speciation data, no more than 10 percent of the nickel compounds are likely to be nickel subsulfide. Therefore, the nickel risk estimates presented in previous sections (where it is assumed the mix of nickel compounds emitted from utilities is as carcinogenic as nickel subsulfide) are considered conservative, upper-bound risk estimates.

To assess the potential impact of the speciation uncertainty, the EPA conducted an assessment for cancer risks utilizing different assumptions for speciation and cancer potency. The assessment (summarized in Table 6-21) provides a range of the potential cancer risks due to nickel emissions.

In addition to the cancer effects, nickel also causes noncancer health effects, such as allergenicity and respiratory effects. Currently, no RfC is available for nickel compounds. However, there are various health benchmarks in the literature that are useful for screening purposes to give some idea whether or not the exposure estimates are likely to cause noncancer health effects. The EPA conducted such an assessment (see Table 6-22).

6.11 POTENTIAL INCREASED DIOXIN EMISSIONS FROM UTILITIES WITH ELECTROSTATIC PRECIPITATORS

Emissions data for dioxins and dibenzofurans were available from only nine test sites. None of these sites tested have hot-side ESPs installed for controlling emissions. The EPA discovered that dioxin emissions from municipal waste combustors (MWCs) with hot-side ESPs could be 5 to 15 times greater than emissions from a similar source without a hot-side ESP.[21] Since this phenomenon was observed at MWCs, the EPA assumes that it is possible that the same situation may possibly occur at utilities. Currently, the DOE is planning to conduct an emission test at a facility with a hot-side ESP; however, at this time, no data are available for dioxins from hot-side ESP units. Therefore, as a scoping effort for this report, the dioxins were modeled a second time with the assumption that dioxin emissions are 10 times greater from all utilities that have hot-side ESPs (145 units). The results of this scoping effort showed an increase of roughly double the national total dioxin emissions from utilities, from 1.5×10^{-4} ton/yr to 3.5×10^{-4} ton/yr. The cancer MIR increased by a factor of 3.5, from 5×10^{-8} to 1.8×10^{-7}.

This was based on a hypothesis. Utilities are different than MWCs. There are differences in fuel and operations. Therefore, it is not known whether utility units with hot-side ESPs are likely to emit more dioxins. More data and analyses are needed before any conclusions can be made regarding dioxins from utilities with hot-side ESPs.

6.12 DISCUSSION OF UNCERTAINTY AND ASSUMPTIONS FOR DOSE-RESPONSE ASSESSMENT FOR CARCINOGENS

Information related to dose-response assessment for the HAPs is summarized here to identify the assumptions, methods, data

Table 6-21. Nickel from Oil-Fired Utilities: Inhalation Cancer Risk Estimates Based on Various Assumptions of Speciation and Cancer Potency

Nickel Speciation[a]	**Cancer potency (IURE)**[b]	**MIR**	**# People > 10^{-6} risk**	**Annual Incidence**
100% Ni Subsulfide	4.8×10^{-4}	9.6×10^{-5}	1,600,000	0.4
20% Ni Subsulfide	9.6×10^{-5}	2×10^{-5}	9,900	0.08
10% Ni subsulfide	4.8×10^{-5}	9.6×10^{-6}	2,300	0.04
1% Ni Subsulfide	4.8×10^{-6}	9.6×10^{-7}	0.0	0.004

IURE = Inhalation unit risk estimates.
MIR = Maximum individual risk

[a] The limited nickel speciation data indicate that nickel is a combination of nickel oxide, soluble nickel, sulfidic nickel, and insoluble nickel. The limited speciation data indicate that less than 10 percent of the nickel is nickel subsulfide.

[b] The Inhalation Unit Risk Estimate (IURE) of 4.8×10^{-4} is the IURE for nickel subsulfide found on IRIS. For each of these cases, it is assumed that either 100 percent, 20 percent, 10 percent, or 1 percent of the nickel is nickel subsulfide, and that only this fraction is contributing to the cancer risk. The cancer risk due to the other nickel compounds is not known.

Table 6-22. Comparison of Nickel Exposure to Various Noncancer Health Benchmarks

Various health benchmarks for nickel compounds	**CARB REL**[a] **= 0.24 μg/m³**	**EPRI**[b] **value = 2.4 μg/m³**
# People exposed[c] above the benchmark	0.0	0.0
# People exposed above 1/10th the benchmark	2,300	0.0
Maximum HQ[d]	0.82	0.082

LEGEND:

REL	= Reference exposure level
CARB	= *California Air Resources Board*
EPRI	= Electric Power Research Institute
HQ	= Hazard quotient

[a] This value was obtained from the CARB Hot Spots Program.[3] CARB calculated this number by dividing the Threshold Limit Value (TLV) of 0.1 mg/m³ by 420. The TLV is a level set by the American Conference of Government Industrial Hygienists (ACGIH) as a guideline to protect workers. The 420 accounts for extrapolating from a 40-hour work week to a 168-hour week (4.2x), extrapolating from healthy workers to sensitive subpopulations (10x), and another factor of 10x because adverse health effects are often seen at the TLV.

[b] The EPRI benchmark[22] was calculated by dividing the TLV by 42. The 42 accounts for extrapolating from a 40-hour work week to a 168-hour week, and a 10x is applied to account for sensitive subpopulations.

[c] The exposed population is estimated from the results of the Inhalation Human Exposure Modeling.

[d] The HQ is calculated by dividing the modeled concentration by the health benchmark. It is the ratio of the estimated highest exposed concentration to the benchmark concentration. A value of 1 or higher indicates that the exposure is above the health benchmark.

used, and uncertainty associated with the dose-response measures. This information is useful to place the quantitative risk estimates into context with respect to their associated uncertainty and conservatism.

6.12.1 Default Options

The EPA uses default options when dealing with competing plausible assumptions and uncertainty in estimating cancer unit risks. The use of these default options is intended to lead to unit risk estimates that, although plausible, are believed to be more likely to overestimate than to underestimate the risks. The use of these defaults has led EPA scientists to conclude that the resulting unit risk estimates are upper limits. That is, the actual risks are unlikely to be greater than these estimates, and may be lower; they could also be zero. Below are several of the major default options used in cancer dose-response assessment identified by NRC.[23] However, it must be noted that the preliminary HAPs of interest in this study for cancer risks (i.e., arsenic, chromium VI, and nickel subsulfide) have IUREs and WOE that are based on human epidemiology studies; therefore, many of the assumptions listed below are not relevant for much of this study.

- Laboratory animals are a surrogate for humans in assessing cancer risks; positive cancer-bioassay results in laboratory animals are taken as evidence of a chemical's cancer-causing potential in humans.

- Humans are as sensitive as the most sensitive animal species, strain, or sex evaluated in a bioassay with appropriate study-design characteristics.

- Agents that are positive in long-term animal experiments and also show evidence of promotion or cocarcinogenic activity should be considered as complete carcinogens.

- Benign tumors are surrogates for malignant tumors, so benign and malignant tumors are added in evaluating whether a chemical is carcinogenic and in assessing its potency.

- Chemicals act like radiation at low exposures (doses) in inducing cancer; i.e., intake of even one molecule of a chemical has an associated probability for cancer induction that can be calculated, so the appropriate model for relating exposure-response relationships is the linearized multistage model.

- Important biological parameters, including the rate of metabolism of chemicals, in humans and laboratory animals are related to body surface area. When extrapolating metabolic data from laboratory animals to humans, one may use the relationship of surface area in the test species to that in humans in modifying the laboratory animal data.

- A given unit of intake of a chemical has the same effect, regardless of the time of its intake; chemical intake is integrated over time, irrespective of intake rate and duration.

- Unless there are data to the contrary, individual chemicals act independently of other chemicals in inducing cancer when multiple chemicals are taken into the body; when assessing the risks associated with exposures to mixtures of chemicals, one treats the risks additively.

6.12.2 Models, Methods, and Data

In a dose-response assessment, the likelihood of developing cancer is determined quantitatively for any given level of exposure to a carcinogen.[24] The two basic reasons for conducting a cancer dose-response assessment are (1) to extrapolate from high to low doses, and (2) to extrapolate from animal to human responses. Both epidemiologic and toxicologic studies are conducted at doses higher than those normally encountered in the environment. Therefore, in order to determine response at lower doses, an extrapolation from high to low dose must be performed. Many models are available for dose-response estimation and high- to low-dose extrapolation. The dose-response assessment must also extrapolate from animals to humans if only animal data are available. This interspecies extrapolation is carried out by applying a scaling factor to the experimental data[24] or through the use of physiologically based pharmacokinetic (PBPK) data.

6.12.2.1 Mathematical Dose-Response Extrapolation Models.

No single dose-response model is appropriate in all situations. A dose-response model is usually selected on an agent-specific basis. However, two categories of dose-response models are generally used in carcinogen risk assessment—mechanistic models and tolerance-distribution models.

Mechanistic models describe some mechanism by which carcinogenesis is believed to occur. All of the mechanistic models assume that a tumor originates from a single cell that has been altered by either the agent or one of its metabolites.[24] Examples of mechanistic models are the one-hit, multi-hit, and multistage models.

The one-hit model assumes that a single hit at a critical site can result in malignant transformations. This model is conservative (i.e., reduces the chance of underestimating risk) because it does not account for cellular or deoxyribonucleic acid (DNA) repair mechanisms. The multi-hit mechanistic model, an adaptation of the one-hit model, assumes more than one chemical exposure or biological event is required to elicit a carcinogenic response. The linearized multistage model is the most frequently used of the low-dose extrapolation models. It corresponds to the most commonly accepted theory of carcinogenesis (the multistage process) and is the model most frequently used by EPA in conducting dose-response assessments. This model assumes that a cell progresses through a number of distinct stages before becoming malignant. Like the one-hit model, the multistage model is approximately linear in the low-dose region.

The second type of dose-response model, the tolerance distribution model, is an empirical model that assumes for each individual in a population there is a tolerance level below which that person will not respond to the exposure.[24] These models assume a variability among individual tolerance levels that can be described in terms of a probability distribution. This concept of individual tolerance levels differs from the "threshold" concept used in most noncancer risk assessment, which posits a general level of exposure that is "safe" for most of the population. Tolerance distribution models are actually based on the "nonthreshold" concept of carcinogenesis because they refer to an infinite number of individual tolerance levels or thresholds distributed along a curve. The low-dose extrapolation techniques based on the tolerance distribution theory include the probit (log-probit), logit (log-logistic), and the Weibull model.

If animal data are used in the dose-response assessment, scaling factors are commonly used to calculate a human equivalent dose. These scaling factors are applied to animal data to account for differences between humans and animals regarding body size, lifespan, route, metabolism, and duration of exposure.[1]

Standardized dosage scales such as mg/kg body weight/day, ppm, in the diet or water, and mg/m^2 body surface area/day are commonly used to allow for comparison of data across species.[1] The EPA considers extrapolation on the basis of surface area most appropriate because particular pharmacologic effects commonly correlate to surface area. Because the body surface area is proportional to the animal's weight to the two-thirds power, and because weight is more easily determined than surface area, equivalent dose can be calculated as follows:

$$d_a/bw_a^{(2/3)} = d_h/bw_h^{(2/3)}$$

where

d_a = experimental animal dose (mg)
d_h = equivalent human dose (mg)
bw_a = weight of experimental animals (kg)
bw_h = weight of average human (kg).

6.12.2.2 Discussion of the Derivation of IUREs. An IURE represents an upper limit increased cancer risk estimate from a lifetime (70-year) exposure to a concentration of 1 $\mu g/m^3$ in the ambient air. This IURE is typically derived from the slope factor, which is a plausible upper-bound estimate of the availability of a response per unit intake or exposure concentration of a chemical over a lifetime.[1] When the slope factor is generated from the linearized multistage model, it is denoted as q_1*. Slope factors are usually expressed in terms of $(mg/kg\text{-}day)^{-1}$ when derived from oral data and $(mg/m^3)^{-1}$ when derived from inhalation data. The following equation is used to convert a slope factor to an IURE for air contaminants:

$$\text{IURE} = \text{Slope Factor} \times 1/70\ \text{kg} \times 20\ \text{m}^3/\text{d} \times 10^{-3}.$$

To calculate the IURE, it is assumed that a 70-kg individual with a breathing rate of 20 m^3/d is exposed to the carcinogen over a 70-year lifespan. The factor of 10^{-3} in the IURE equation is required to convert from milligrams to micrograms. The IURE is based on the assumption of low-dose linearity. If a nonlinear low-dose-response extrapolation model were used, the unit risk would differ at different dose levels, and the dose-response assessment output could be expressed as a dose corresponding to a given level of risk, analogous to the risk-specific dose, rather than as a single IURE.

If the IURE is derived from animal data, it usually represents the upper 95th percent confidence limit of the slope factor as suggested by the variation within the animal data. Using the upper 95th percent confidence limit reduces the probability of underestimating the unit risk.

For four priority HAPs (arsenic, chromium, radionuclide, and nickel), human epidemiologic data are available and were used to derive a maximum likelihood estimate (MLE) of the IURE. The MLE is defined as a statistical best estimate of the value of a parameter from a given data set.[25] Therefore, the difference between the upper-bound estimate and the MLE is that the upper bound is a conservative measure of risk while the MLE is a statistically best estimate.

6.12.3 Discussion of Uncertainty in IUREs

Uncertainty is associated with the IURE because many assumptions have been made in the process of deriving it. Uncertainty arises from several areas in a dose-response assessment including intra- and interspecies variability, high- to low-dose extrapolation, route-to-route extrapolation, and the development of equivalent doses. One type of potential uncertainty is often called the "healthy worker effect." This results because the IUREs for some HAPs (e.g., arsenic, chromium VI) are based on studies of workers exposed during their working careers. The sensitivity of the workers to developing cancer may not be the same as the sensitivity of the general population. Therefore, there is uncertainty in the representativeness of the worker population for calculating an IURE for the general population. There may also be uncertainties because of truncation of observation periods in most epidemiology studies. In addition, there are uncertainties in the estimates of individual dose or exposure in the epidemiology studies.

When using animal studies to estimate dose-response, the assumption that administered dose is proportional to delivered dose is typically used when estimating human equivalent doses. However, physiological and pharmacokinetic differences between experimental animal species and humans may result in differences in delivered target organ dose. Not accounting for these may introduce uncertainty in the estimation of human equivalent dose.

Low-dose extrapolation models can result in estimates of risks that differ by several orders of magnitude. Therefore, selection of model is critical. Some uncertainties may result in high biases, others may result in low biases.

The IURE is based on the assumption that exposure to a particular agent occurs over a 70-yr lifetime under constant conditions and assumes that risk is independent of dose rate. Actually, the exposed population is not exposed either continuously or at a constant level. It is unknown how the detoxification and repair mechanisms may act at higher or lower dose rates or with intermittent exposures, thereby introducing uncertainty in the risk estimate. Variable exposure concentrations introduce uncertainty. If detoxification and repair mechanisms are more efficient with intermittent exposures (allowing for recuperation or repair), the IURE would overestimate risks when compared to the total dose received. By contrast, if these mechanisms were less efficient at an intermittently higher dose rate, the IURE may underestimate risk when compared to total dose.

Risks from multiple carcinogens are typically estimated assuming dose additivity. However, uncertainties are associated with this approach. The risk summation technique assumes exposures are in the low-dose range where responses are linear; however, at higher risk levels, nonlinearity may need to be considered. The additivity approach also assumes that each chemical acts independently (i.e., that there are no synergistic or antagonistic chemical interactions and that all chemicals produce the same effect). If these assumptions are incorrect, over- or underestimation of the actual multiple-substance risk could occur.[26] Several other limitations to this approach must be acknowledged. Because the IURE is typically an upper 95th percentile estimate of potency and upper 95th percentiles of probability distributions are not strictly additive, the total cancer risk estimate might become artificially more conservative as risks from a number of different carcinogens are summed. However, the human-derived potency estimates, which are the most important for this analysis (e.g., radionuclides, arsenic, chromium VI, and nickel subsulfide), are not based on the upper 95th percentile. These IUREs are based on a maximum likelihood estimate. Therefore the potential for artificially conservative estimates resulting from summing risks of individual HAPs may not be an issue for this risk assessment.

Uncertainty in the breathing rate relates to the level of activity. The breathing rates in epidemiological studies on which the cancer slope factors (CSFs) are based are typically higher than the standard 20 m^3/d for the general population. Uncertainty in the deposition fractions varies between individuals due to variation in breathing rates, particle sizes, and the sizes of lung passages. Retention half-times typically are distributed lognormally though there is little information on how they differ between the (epidemiological) study population and the general population. Life-time averaged retention half-times should be slightly lower in the general population due to the inclusion of young ages for which the retention half-times are usually lower than adult values.

6.12.4 Variability in Cancer Dose-Response Assessment

Human beings vary substantially in their inherent susceptibility to carcinogenesis. Person-to-person differences in behavior, genetic makeup, and life history can influence susceptibilities. Such interindividual differences can be inherited or acquired. Acquired differences that can significantly affect an individual's susceptibility to carcinogenesis include the presence of concurrent viral or other infectious diseases, nutritional factors such as alcohol and fiber intake, and temporal factors such as stress and aging. Evidence regarding the individual mediators of susceptibility supports the plausibility of a continuous distribution of susceptibility in the human population.

Some researchers have attempted to determine the range of susceptibility due to the general variability in physiological parameters that may affect target organ dose. Their results indicate that the difference in susceptibility between the most sensitive 1 percent of the population and the least sensitive 1 percent might be as small as a factor of 36 (if the logarithmic standard deviation was 0.9) or as large as a factor of 50,000 (if the logarithmic standard deviation were 2.7).[23]

Certain groups of individuals within the population are inherently more sensitive to carcinogen exposure than others. Factors that influence susceptibility include age, race, sex, and genetic predisposition. An example of a sensitive subpopulation is children. This subpopulation can be more sensitive to certain chemicals and more susceptible to cancer for a variety of reasons, including:

- Children have faster breathing rates than adults and, thus, inhale larger quantities of a pollutant, relative to their body weights.
- Organs in children are still growing and developing and are, therefore, more prone to disruption by an environmental agent.
- Children spend substantially more time outdoors than adults and may be exposed to higher concentrations.
- Young organisms appear to be inherently suspectible to many carcinogens. Young experimental animals have been shown repeatedly to acquire more tumors in a shorter time with a smaller dose than adult animals.

In most circumstances, as with this study, there are not enough data available to perform separate quantitative dose-response assessments for these sensitive subpopulations. Obviously, children are not included in the work force at plants where much of the epidemiology data are collected.

As stated above, the IURE is based on the assumption that exposure to a particular agent occurs over a 70-year lifetime under the same conditions to which the study group was exposed. For animals, it is essentially steady and constant exposure over a lifetime; for humans, it is varying exposure over their working career at a particular plant. In effect, this assumes that risk is independent of dose rate. Recent research suggests that cellular repair mechanisms exist that can reverse the damage caused by a carcinogen, and it is likely that these mechanisms operate most effectively after low doses or in the absence of repeated doses. Therefore, variability in exposure would also influence or create a variability in how effective the IURE predicts risk.

6.13 PRELIMINARY QUANTITATIVE UNCERTAINTY AND VARIABILITY ANALYSIS FOR INHALATION EXPOSURE AND RISK ASSESSMENT

6.13.1 Introduction

Risk assessment is a complex process, and uncertainty will be introduced at every step in the analysis. Even using the most accurate data with the most sophisticated models, uncertainty is inherent in the process. There are a number of uncertainties associated with the exposure assessment of emissions from utilities. These include parameter estimation (test results), model choice, and the use of simplifying assumptions.

Uncertainty in emissions and exposure estimates can result from uncertainty (i.e., doubt or ignorance of the true value) or from variability (i.e., known range of values over time, space, or within a population). A quantitative uncertainty analysis was conducted for the direct inhalation exposure part of this risk assessment. The evaluation of uncertainty in the estimation of emissions, dispersion and exposure is summarized here. See Appendix G for details on the uncertainty analysis. This uncertainty evaluation does not include consideration of the impacts (and associated uncertainties) due to long-range transport and multipathway exposures. The focus of this particular analysis is the uncertainties and variability of the inhalation exposure within 50 km of the plants.

The need for formal uncertainty analysis as a part of any risk assessment and its aid in conveying results of the risk assessment are widely accepted, having been proposed in both the EPA Risk Characterization Guidance and the NRC Committee Report: *Science and Judgement in Risk Assessment*.[23] Furthermore, any procedure that relies on a combination of point values (some conservative and some not conservative) yields a point estimate of exposure and risk that falls at an unknown percentile of the full distributions of exposure and risk.

The risk estimates presented in previous sections were derived by utilizing various input data and assumptions. The results were presented as point estimates of risks. The following uncertainty analysis was conducted to determine the degree of conservatism.

The uncertainty analysis focused on the three HAPs (nickel, arsenic, and chromium) that accounted for over 95 percent of cancer incidence. An analysis of uncertainty on these three HAPs accounts for much of the uncertainty in the overall risk estimates.

6.13.2 Approach to Quantitative Uncertainty Analysis

Uncertainty has been classified into four types (parameter uncertainty, model uncertainty, decision-rule uncertainty, and variability). The first two, parameter uncertainty and model uncertainty, are generally recognized by risk assessors as major sources of uncertainty. Parameter uncertainty occurs through measurement errors, random errors, or systematic errors when variables cannot be measured precisely either because of equipment limitations or because the quantity being measured varies spatially or temporally. Model uncertainty can result from surrogate variables, excluded variables, abnormal conditions, and/or incorrect model form. Decision-rule uncertainty arises out of the need to balance different social concerns when determining an acceptable level of risk, which can affect the choice of model, data, or assumptions. Variability is often used interchangeably with the term "uncertainty," but this is not strictly correct. Variability is the unchanging and underlying distribution of a parameter based on physical, chemical, and/or biological processes (e.g., body weight within a population). Even if variability is known (therefore, not in itself uncertain), it still contributes to overall uncertainty of the risk assessment.

This uncertainty analysis focused on parameter uncertainty within the models and data available. Table 6-23 briefly summarizes information regarding the parameters used in the risk estimation process. Model uncertainties are not addressed in the quantitative uncertainty analysis, but are described qualitatively. Variability has been evaluated separately for exposure-response, but is included in the overall estimate of uncertainty related to emissions and exposure. The goal of this uncertainty analysis is to estimate the range of possible risk estimates considering the parameter uncertainty and variability.

Table 6-23. Summary of Basic Parameters Used in Risk Assessment for Electric Utilites

Parameter	Default option/assumptions (and departure from default and reason why)	*Conservatism*	Data (*e.g., source, quality*)	Uncertainty/variability (*quantitative and qualitative*)	Distribution	Judgement/strategy
EMISSIONS CHARACTERIZATION						
Fuel consumption	Coal: •1990 (or 1989, geo mean 1980-8) UDI/EEI data self-reported by the utilities to DOE. •Adjustments made for heating value of different coals •Based on total tonnage Oil: assumed to be residual oil, quantity consumed in gallons is converted to mass based on an assumption of uniform density.	Low, average value used	•UDI Database: self-reported, with no QC or validation •Average heating values used for coal type (lign = 6800 BTU/lb, bit. = 12688 BTU/lb, subbit. = 9967 BTU/lb)	•Accuracy of self-reported values •Fuel consumption over time due to demand, sulfur content, etc...	Normal	Based on engineering judgement.
Coal State of Origin	•Assume all from the state where majority of fuel consumed based on total tonnage	Unknown	•UDI database identifying majority use	•Coal from several states may be used at one plant, mix of states coal actually used •Relative composition of state coals due to availability, cost, sulfur content etc . may change over time	NA	Relative contribution between states held constant
Trace element concentration	•Arithmetic average of coal type for state of origin which is used most at the facility. •Oil: average HAP concentration in test data of residual fuel oil No. 6 (about 80% of all oil burned).	Low, average value used	•USGS core/channel sampling (extraction) of economically feasible coal seams (n= 3331)	COAL: •Conc. measured in extracted coal, not in coal shipments, reductions in trace element conc. may occur during processing •Coal seams measured may not actually being used for shipment •Coal from other states used at plant may contribute significantly •Variability within a coal seam, between coal seams within a state. OIL:•Density will vary among No. 6 fuel oils which means that the volume and mass consumed will vary. Concentration of HAPs within oil will vary.	Log-normal	prevents the possibility of negative concentration with no upper limit
Coal cleaning *factor* (CCF)	Process of preparing coal for shipment may reduce some mineral matter. Since about 77% of eastern and midwestern bituminous shipments are cleaned a CCF was applied to all bit. coal	Low, average value used	Testing of coal shipments from Wyoming, Colorado and Illinois	•Coal cleaning data may not apply to other types of coal •Effectiveness of coal cleaning may vary according to variability in the sulfur and ash content within a coal seam and the variability in processing	Normal	engineering process
EMF: Boiler and APCD	Geometric mean of test data, measured in gas stream, ash stream was ignored	Low, average value used	Coal: 19 facilities tested of varying configurations, combinations of boiler type and APCD. *Oil. testing at 2 facilities.* Each test point was at least a triplicate sample	•were units tested representative of units in operation •Unit performance likely to vary over time due to fuel and operating parameters.	Beta or Triangular (if n=1)	Beta: constrained within 0 and 1 and distribution defined by data.. Tri.. no distribution can *be estimated from single* point, value used as the apex bounded by 0 and 1.

Table 6-23 (continued)

Parameter	Default option/assumptions (and departure from default and reason why)	Conservatism	Data (e.g., source, quality)	Uncertainty/variability (quantitative and qualitative)	Distribution	Judgement/strategy
DISPERSION AND EXPOSURE MODELING						
Dispersion	Gaussian plume	Unknown	Limited data on other models		Not analyzed	Beyond scope of project. Model is EPA default.
Roughness (rural v. urban)	Population density (within 3 km of plant) is assumed to be an indicator or proxy for setting of the plant, and that urban and rural are representative of surface roughness. Binary choice of urban v. rural	Unknown, may not account for values beyond model defaults	Census data on population within 3 km of facility to indicate urban.	Roughness not binary and not always attributable to population density (e.g., buildings) but other land features as well •Urban and rural model default settings may not represent the entire range of surface roughness leading to possible extremes not addressed.	Each plant run in both modes	Population may not be an indicator of actual surface roughness. the urban and rural defaults approximate the range of uncertainty.
Terrain	Assumed to be flat terrain. Flat terrain used in Gaussian plume dispersion.	Low, as shown by complex terrain analysis	Extensive data on terrain surrounding each facility. However, analysis is very exhaustive and cannot be carried out for all facilities.	Terrain effects can be significant leading to minimal dispersion and high exposure •Terrain is not a binary parameter and the degree of terrain differences will vary between plants.	Analyzed separately (see Section 3.2)	Subset of plants analyzed were representative.
Meteorology	The meteorological data from the nearest STAR location are used to represent the meteorology near the plant. Stability classes are assumed to be represented in the STAR data and implicitly addressed in the HEM model.	Unknown, assumed to represent site met conditions	STAR data are typically five-year averages at 350 airports.	Meteorology at plant may be significantly different than the nearest plant Different meteorology may not affect the maximum concentration but may significantly affect the number of persons exposed and at risk. Short-term meteorological conditions (e g., inversion) may affect short-term exposure levels. •Meteorological conditions will vary at a site over time.	Three closest met locations used	The actual site met conditions would be approximated by at least one of three closest stations.
Effective Stack Height	Effective stack height is calculated using the stack height, exit velocity and exit gas temperature as reported in the UDI database.	Medium, actual data with conservative model	Data from UDI database are as reported by the facility. Little or no data on variability in exit gas temperature or exit velocity	The effective stack height may vary significantly from the calculated value due to variation in exit gas temperature and velocity. which would greatly impact the resulting exposure concentration. •Stack height would not vary as it is a physical parameter.	High (1.1) Med (1) Low (0.9)	Based on subjective judgement.
Location of Exposed Population	Population assigned to the centroid of the census block or, if within 0.5 km, to receptor grid location for which concentration is estimated. Assumed to represent a persons average exposure (they may also spend time in areas of higher or lower concentrations)	Low/Medium assignment at 0.5 km accounts for variability	1990 Census Block data.	Location of the centroid is estimated, is it accurately defined. Uncertainty as to where people are actually located with respect to the centroid. •The location of individuals will vary with respect to the centroid, some being in higher concentrations some in lower concentrations.	Analyzed separately	Changing grid or centroid assignment distances showed little influence. resolution of 0.5 km minimizes spatial effects.

Table 6-23 (continued)

Parameter	Default option/assumptions (and departure from default and reason why)	Conservatism	Data (e.g., source, quality)	Uncertainty/variability (quantitative and qualitative)	Distribution	Judgement/strategy
EXPOSURE-RESPONSE ASSESSMENT						
Exposure Duration (Population mobility)	Assumes persons spend 70-year lifetime at the location to which they are assigned. The concentration at the centroid represents their average to which they are exposed.	High (MEI/MIR) tends to maximize exposure	Very little data on mobility which are applicable to the range of populations affected. Most have been focused on small subsets (e.g., residency in apartments).	Individuals mobility will affect exposure. Uncertainty about defining a representative mobility pattern or distribution which also accounts for movement to alternatively polluted areas. ●Population mobility varies dramatically within a population and for an individual over time. ●Alternatively can consider time-activity patterns (e.g., indoor/outdoor, movement within area) and residence time (average = 9 years, 90th = 30 years).	Not analyzed	Given the proportion of the population who are exposed to emissions from utility emissions, it is likely that people who move will still be exposed (though at lower levels).
Exposure Frequency	(Indoor/outdoor concentration) Assumes exposure at 100% of outdoor concentration.	High, maximizes exposure,	*Measurements of indoor/outdoor* concentration ratio, time-activity patterns, and exposure. No data specific to locations of electric utility plants.	*The relationship between indoor and* outdoor concentration is complex because infiltration is affected by climate, building type, ventilation etc.. Infiltration will vary over time due to climate variability (e.g., open windows).	*Lognormal,* variability Normal, uncertainty	*Based on limited data* and accepted EPA defaults.
Breathing Rate	Assumes that workers breathing rate is equal to national average.	Low to Med, workers may have higher BR value	Measurements of minute volumes for different population subgroups.	Breathing rates differ greatly by age and activity.	Lognormal, variability Normal, uncertainty	Variability measured, prevents negative values. Uncertainty from standard error of mean.
Lung Deposition	No adjustment between worker and general populations.	Low, average value used	Measurement of lung deposition fractions	Lung deposition can vary by age and activity level.	Lognormal, variability Normal, uncertainty	Variability measured, prevents negative values. Uncertainty from standard error of mean.
Retention Half-life	No adjustment between worker and general populations.	Low, average value used	Measurement of half-lives, but not for the specific HAPs evaluated.	Retention half-lives will vary by age, activity level, and particle size.	Lognormal, variability Normal, uncertainty	Variability measured, prevents negative values. Uncertainty from standard error of mean.
Slope factors	Used EPA-verified slope factors, best estimate linearized function.	Unknown, but believed to be high	Human epidemiological data.	Slope factors will differ dramatically based on model choice, statistical uncertainty in data	Lognormal, uncertainty	Variability treated qualitatively. Uncertainty from the SEM from existing data. Model uncertainty: qualitatively.

NOTE: Quantitative values for all parameters and their distributions are presented in the body of the text.

Key:
High = Most likely to overestimate than underestimate. Can represent an upper bound estimate.
Medium = May either under- or over-estimate. With the use of conservative models usually more likely to over- than under-estimate risk.
Low = Usually an unbiased estimator using the average value Equally likely to over- or under-estimate risk.
UDI/EEI: Utility Data Institute, Edison Electric Institute
EMF: Emissions modification Factor
APCD: Air pollution control device
Beta Distribution: constrained between two distinct values (e g , 0 and 1), defined by the mean and standard error of mean (SEM) of the original data. This distribution maintains the mean and standard error of mean (SEM). The use of a truncated distribution (either normal or log-normal) can lead to a drift in the mean and/or SEM from the original data.

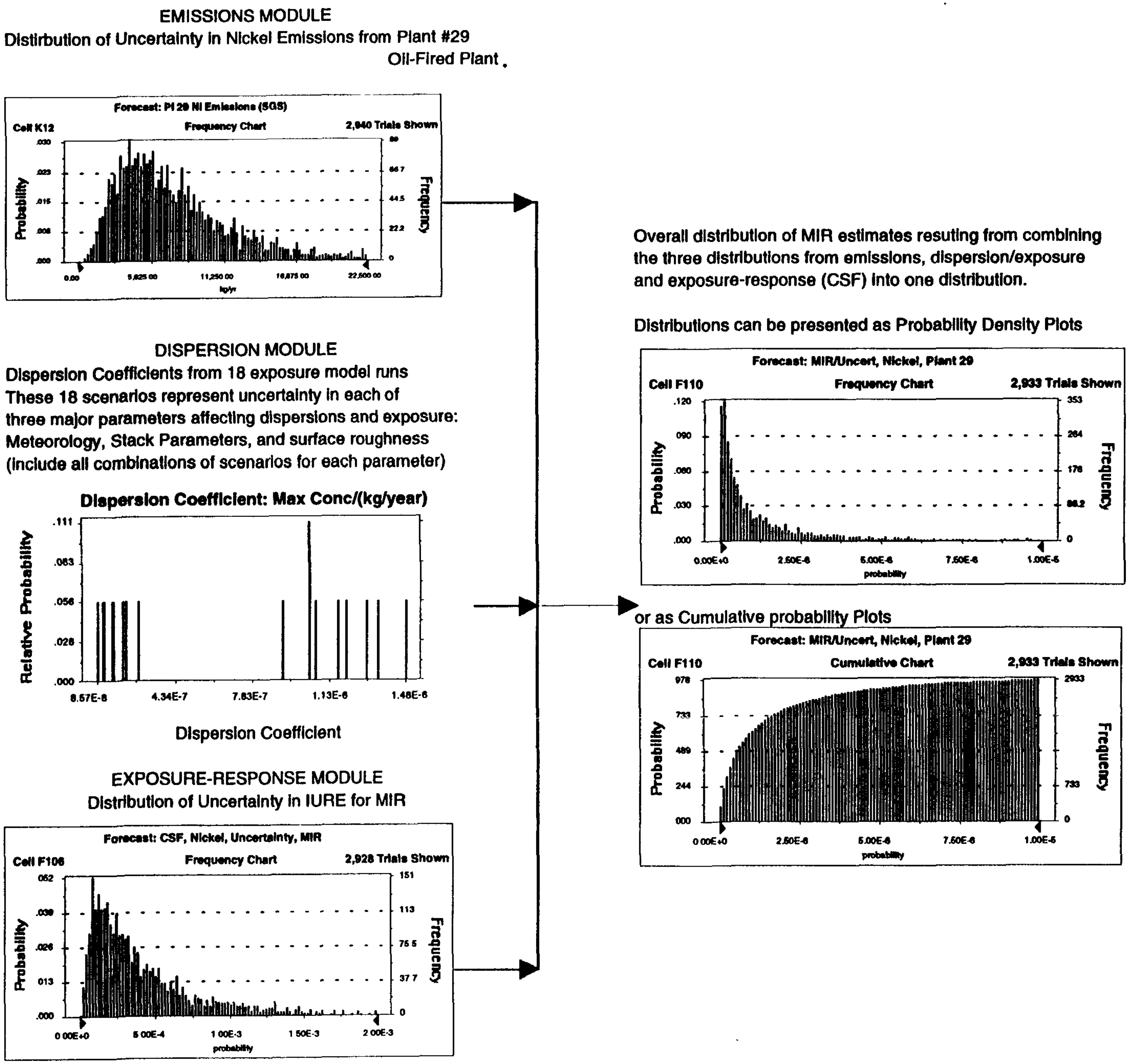
It should also be noted that there are other sources of uncertainty, some of which may be significant, which could not be evaluated quantitatively. These uncertainties are qualitatively discussed.

The approach used in this analysis was to identify the uncertainty with each of the parameters used in the risk estimation process. First, the uncertainty associated with each of these variables was described using an appropriate statistic (e.g., mean and standard error of means) or as a probability density function (the relative probability for discrete parameter values). The standard error of the mean (SE) for each parameter was the estimate of uncertainty and variability used rather than the standard deviation for each parameter. Since the available dose-response data are based on lifetime exposure, and the cancer risk assessment is concerned with long-term average exposures, the SE is a more appropriate statistic. However, it should be noted that using the SE from a sample may be an overconfident estimate (i.e., too narrow a range) of uncertainty (see Appendix G for explanation).

In general, numerical methods (i.e., Monte Carlo simulation) were then used to develop a composite uncertainty distribution by combining the individual distributions. In Monte Carlo simulations, the risk and/or model equations are repeatedly solved using randomly sampled values from the specified distributions to calculate a distribution of risk values. These risk distributions were derived for estimates of MIR and population risks. Because variability was not specifically differentiated in the analysis of emissions and dispersion modeling, uncertainty and variability were simulated together in a one-dimensional Monte Carlo simulation.

The uncertainty analysis was conducted on the three major components of the risk assessment process: emissions characterization, dispersion and exposure modeling, and exposure-response assessment. Each of these is summarized briefly below. Figure 6-5 provides an example of how the uncertainty from each of these components is combined into an overall distribution. A detailed uncertainty analysis could not be conducted on all of the utility plants. Therefore, a total of four plants (two oil-fired and two coal-fired plants) were selected which contribute most to risk, the highest estimated incidence and the highest maximum individual risk. Each of these plants was analyzed for arsenic, nickel, and chromium. The highest incidence oil-fired plant (Plant No. 29) accounted for about 7 percent of the annual cancer incidence and, therefore, was chosen for illustration purposes.

Figure 6-5. DEPICTION OF COMBINING COMPONENT UNCERTAINTY DISTRIBUTIONS (i.e., EMISSIONS, DISPERSION, and EXPOSURE-RESPONSE) INTO AN OVERALL DISTRIBUTION ON UNCERTAINTY (e.g., MIR)



EMISSIONS MODULE
Distirbution of Uncertainty in Nickel Emissions from Plant #29
Oil-Fired Plant.

DISPERSION MODULE
Dispersion Coefficients from 18 exposure model runs
These 18 scenarios represent uncertainty in each of
three major parameters affecting dispersions and exposure:
Meteorology, Stack Parameters, and surface roughness
(include all combinations of scenarios for each parameter)

EXPOSURE-RESPONSE MODULE
Distribution of Uncertainty in IURE for MIR

Overall distribution of MIR estimates resuting from combining the three distributions from emissions, dispersion/exposure and exposure-response (CSF) into one distribution.

Distributions can be presented as Probability Density Plots

or as Cumulative probability Plots

6.13.2.1 Emissions Characterization Uncertainty. An emissions factor program was developed by EPA to estimate plant-specific emissions rates based on fuel type, fuel origin, plant configurations, and emissions testing results. The emission factor program (including principles and rationale) and the data used are described in chapter 3 and appendix D. This program is based on a mass-balance concept, reducing concentrations in the fuel due to the impact of the boiler and control devices.

The parameters used in the emissions characterization were: fuel consumption (coal: ton/yr, oil: barrel/yr), HAP (trace element) concentration in fuel, coal cleaning factor (if needed), emissions modification factors for the boiler (EMF_b, boiler-specific factor to account for amount of HAP entering boiler to that exiting boiler), and the air pollution control device (APCD), if present (EMF_a, APCD-specific factor to account for amount of HAP entering the APCD to that exiting APCD).

6.13.2.2 Plant-Specific Emission Rates. Monte Carlo simulation was used to develop a distribution of possible plant-specific emissions rates. Simulations were carried out randomly sampling values for fuel consumption, HAP concentration, and EMFs. For illustration purposes, Table 6-24 and Figure 6-6 present the summary statistics and graphical representation, respectively, of the emissions predicted for Plant No. 29. This distribution gives some indication of the degree of uncertainty and the possible range of emissions estimates that may be experienced. The emissions estimates used in the baseline risk assessment were generally central tendency estimates (i.e., geometric means). In general, the 95 percent confidence range for long-term average emissions estimates are within a factor of 2 to 3 of the emissions estimates used in the risk assessment. For example, the 95th percentile of the overall range of predicted emissions ranged from about 0.9 times the original emissions estimate from plant No. 343 for nickel to about 2.5 times the original estimate for arsenic. As shown in Table 6-24, the original estimate of emissions from the baseline risk assessment ranged from the 22nd to the 95th percentile of the range of emissions predicted under the uncertainty analysis.

A preliminary evaluation of the EFP was conducted. Comparisons were made of test data from 19 utility boiler stacks (17 coal-fired, 2 oil-fired) against predicted emissions for the same plants using the EFP. For each facility, the emission estimate from the EFP was divided by the reported value from the corresponding test report. A value of 1 meant that the EFP exactly predicted the test results, values lower than 1 indicated

Table 6-24. Summary of Results for Monte Carlo Simulation of HAP Emissions (kg/year) from Oil-Fired Plant No. 29

		Arsenic emissions			Chromium emissions			Nickel emissions		
		FCEM	SGS	Combined	FCEM	SGS	Combined	FCEM	SGS	Combined
Mean		44	216	130	69	20	45	6423	8612	7527
Initial Point Estimate		------------	133	------------	------------	69	------------	------------	8170	------------
(Percentile)		(93)	(22)	(58)	(64)	(95)	(80)	(76)	(59)	(68)
Percentiles:	0.0%	0.0	13	0.0	4	0.2	0.2	225	507	225
(approx.)	2.5%	0.4	66	--	11	0.5	0.6	934	2001	1125
	5.0%	0.8	78	--	15	0.9	1.5	1227	2389	1462
	10%	1.6	98	2.0	20	1.6	2.1	1645	3015	2125
	25%	3.9	142	6.0	33	3.8	8.1	2731	4583	3487
	50%	9.1	197	98	54	8.7	26	4759	7010	5962
	75%	31	269	206	85	20	60	8009	10847	9562
	90%	94	358	303	135	43	106	12736	16034	14625
	95.0%	177	417	378	174	71	149	17441	20173	19060
	97.5%	322	474	455	214	117	208	22936	24544	24750
	100.0%	2177	894	2177	1087	818	1087	55170	82450	82450

FCEM = (Field Chemical Emissions Monitoring [EPRI]) Original oil concentration data, distribution defined by probability plotting technique.

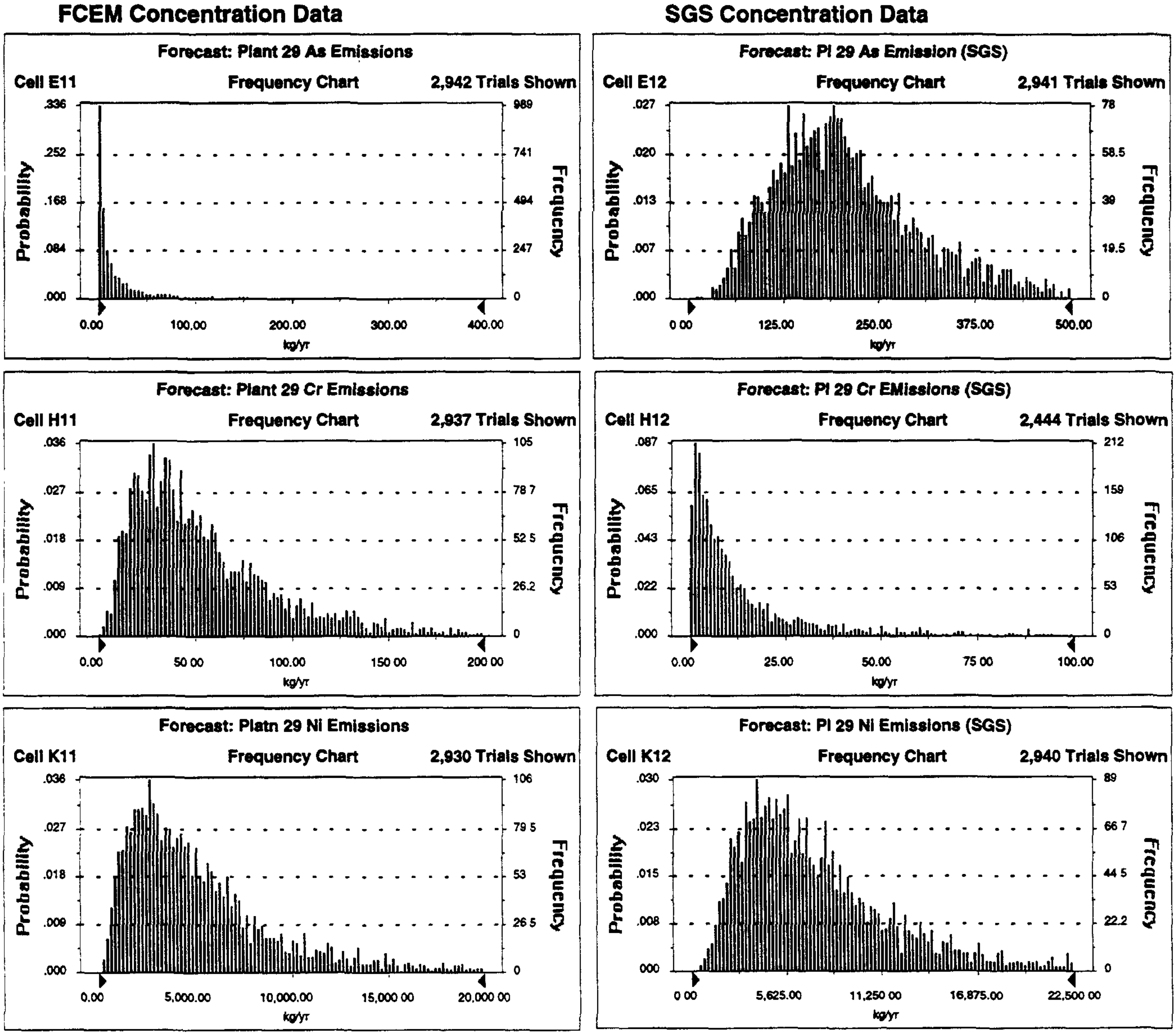
SGS = Subsequent data, trace metal analysis from samples collected for radionuclide analysis.

Combined = Combined forecasts assuming equal probability of the FCEM and SGS data sets.

Initial Point Estimate = The estimate of emissions used in the baseline exposure assessment. This value was based on the average concentration in the FCEM data.

(Percentile) = The percentile of the predicted distribution corresponding to the initial point estimate.

Figure 6-6. Summary of Results of Monte Carlo Simulation of HAP Emissions from Oil-Fired Plant#29



a. Note that the abscissa scales are not the same from the FCEM to SGS data displays.

b. FCEM = Field Chemical Emissions Monitoring from EPRI program.

c. SGS = Subsequent data

the EFP underpredicted emissions, while values higher than 1 indicated the EFP overestimated emissions. In general, the results suggested that the EFP performs reasonably well for predicting emissions on a national basis. The average of the ratios across all stacks and constituents was 1.0, while averages for arsenic, chromium, and nickel are 1.4, 0.7, and 0.9, respectively.

However, while the model did well in predicting overall or average emissions across a range of utility boilers, large differences between predicted and reported values were found for a few individual boilers and constituents. The largest difference was for an individual boiler for which estimated emissions were about 5,000 times lower than reported test results. However, it was determined that this facility was a low-risk plant in the overall analysis; therefore, increasing emissions of this plant by 5,000 would not change the overall results. The EFP tended to underestimate rather than overestimate emissions about 60 percent of the time within this sample of boilers. A preliminary evaluation of facilities with large differences between projected and actual emissions found that these facilities were likely to burn multiple fuel types. Since most of the higher risk plants do not fall into this category, the differences here are not expected to impact significantly on the overall risk estimates. See Appendix G for further discussion.

6.13.2.3 Dispersion and Exposure. Air dispersion modeling is complex and nonlinear, cannot be carried out with the use of spreadsheets, and requires significant time to conduct the modeling and process the data for each run. To better estimate percentile values above 90 percent, a stochastic (Monte Carlo) approach requires large numbers (thousands) of repetitive runs (3,000 was used for the emissions estimates) needed to generate a distribution. Given the time and resources required for single runs, the Monte Carlo approach was not feasible and an alternative approach was needed to evaluate the uncertainty in dispersion and exposure modeling.

The degree of dispersion and resulting exposure is affected by three major parameters: plant stack parameters (e.g., stack height, stack gas temperature, and exit velocity), meteorologic conditions, and surface roughness (urban vs. rural). The uncertainty analysis, therefore, focused on these three parameters. The three factors being evaluated are nonlinear with respect to each other and require a separate HEM run for each parameter value. Therefore, a test matrix approach was used to evaluate uncertainty in the exposure modeling component of the exposure assessment. A limited number of options were developed to represent the expected range of uncertainty for each of these three categories of parameters as follows:

Surface roughness: urban or rural mode

Stack parameters: represented as high (1.1 x UDI values), medium (UDI values), and low (0.9 x UDI values) estimates for stack gas temperature and flue gas exit velocity

Meteorology: three closest meteorology locations in the STAR database.

As a result, for each plant, a total of 18 different HEM runs were made covering each combination of dispersion parameters. For the purposes of this uncertainty analysis, it was assumed that there is insufficient information to determine the relative correctness of each combination and, therefore, each was considered equally likely to represent the possible range of values. The coefficients for estimating maximum concentration and total exposure (per 1 kg/yr emission) resulting from each of these 18 HEM runs were summarized for each plant.

6.13.2.4 Exposure-Response Assessment. The variability of the quantitative relationship between exposure and the excess probability of cancer for different humans and the uncertainty in the mean (taken here also to be the "best estimate" or "maximum likelihood estimate") quantitative relationship between exposure and the excess probability of cancer were both addressed. As with the uncertainty analysis for emissions and dispersion, efforts were limited to arsenic, chromium, and nickel. Specific parameters, for which uncertainty about the mean value (or best estimate for a given parameter within the exposed population) was addressed, include exposure frequency, exposure duration, breathing rate, deposition fractions, and retention half-times. Uncertainty related to the IURE focused on data and the use of epidemiologic data (typically from workers) extrapolated to the general population.

The software program Crystal Ball® (Decisioneering, Inc., Denver, CO) conducted stochastic (Monte Carlo) simulations of the risk estimates, incorporating the uncertainty for each parameter. A probability distribution that best represents the variable, its average value, and a measure of uncertainty about the average value was developed for each parameter. The simulation consists of conducting repeated calculations (thousands) of risk using values for each parameter sampled from the distribution of values for that parameter.

The study of variability focused on how parameter values would be expected to vary among individuals within the general population and how that would affect the estimation of risk and incidence. The parameters for which some measure of variability among individuals within the general population was addressed include exposure duration, exposure frequency, breathing rate, deposition rate, and retention times in the lung. No specific measures of variability were available for how the IURE for these three HAPs may differ among individuals. However, limited data indicate that the IURE differs between smokers and nonsmokers and was incorporated in the analysis.

6.13.3 Discussion of Results of the Quantitative Uncertainty Analysis

The risk estimation process used in the baseline assessment utilized a combination of parameters, each with varying degrees of conservatism (the degree of overestimation, or underestimation). In general, the estimates of maximum individual risk and annual cancer incidence derived in the baseline risk assessment were conservative, generally around the 95th percentile on the distribution. The 95th percentile is roughly 10 times the median and about 5 times the mean. The distribution of estimates of MIR for Plant No. 29 are presented in Table 6-25. The sensitivity analysis indicated that the dispersion coefficient (surface roughness) was the most significant parameter for estimating uncertainty MIR and incidence, followed by the EMFs. The deposition fraction, retention time, and exposure frequency also contributed significantly in the variability of these estimates.

The EPA risk assessments are generally conservative (more likely to overestimate than underestimate risks). Often there is a concern that the use of several conservative assumptions results in risk estimates that are unrealistic and beyond the range of possible risks (i.e., overly conservative). The results of the uncertainty analysis indicate that the baseline inhalation risk estimates are reasonably conservative (generally around the 90th or 95th percentile). The uncertainty analysis supports the baseline risk estimates as reasonable high-end estimates of risks because they are within the range of predicted risks.

However, it should be noted that this analysis has focused only on parameter uncertainty. Also, not all parameters were included. For example, residence time and activity patterns were not assessed quantitatively in the uncertainty analysis. As a

Table 6-25. Distribution of MIR: Plant No. 29: Comparison of FCEM and SGS Concentration Data

MIR, Plant No. 29

Uncertainty

	Arsenic		Chromium		Nickel	
	FCEM	SGS	FCEM	SGS	FCEM	SGS
Mean	1E-07	6E-07	1E-07	4E-08	2E-06	3E-06
Initial Point Estimate	6E-07		2E-07		4E-06	
(percentile)	(96)	(71)	(87)	(98)	(90)	(85)
Percentiles:						
0.0%	2E-12	1E-09	2E-10	1E-11	2E-09	6E-09
2.5%	3E-09	8E-09	2E-09	7E-10	2E-08	4E-08
5.0%	6E-09	2E-08	3E-09	1E-09	5E-08	6E-08
10%	1E-08	3E-08	5E-09	3E-09	9E-08	1E-07
25%	3E-08	7E-08	1E-08	7E-09	2E-07	3E-07
50%	6E-08	2E-07	4E-08	1E-08	6E-07	9E-07
75%	8E-08	7E-07	1E-07	2E-08	2E-06	3E-06
90%	2E-07	1E-06	3E-07	8E-08	4E-06	6E-06
95.0%	5E-07	2E-06	5E-07	1E-07	7E-06	1E-05
97.5%	1E-06	4E-06	7E-07	2E-07	1E-05	2E-05
100.0%	3E-05	2E-05	6E-06	6E-06	9E-05	7E-05
Ratio						
95th : mean	0.8	4.1	2.3	0.7	1.7	2.5
95th : median	8.7	10.6	12.8	9.6	11.1	10.4
95th : baseline	3.5	3.8	4.2	3.9	4.0	3.8

Variability

	Arsenic		Chromium		Nickel	
	FCEM	SGS	FCEM	SGS	FCEM	SGS
Mean	1E-07	6E-07	1E-07	3E-08	2E-06	2E-06
Initial Point Estimate	6E-07		2E-07		4E-06	
(percentile)	(95)	(68)	(90)	(97)	(90)	(90)
Percentiles:						
0.0%	5E-12	4E-09	4E-10	1E-11	5E-09	2E-08
2.5%	1E-09	1E-08	2E-09	3E-10	3E-08	6E-08
5.0%	3E-09	2E-08	3E-09	7E-10	6E-08	1E-07
10%	6E-09	4E-08	6E-09	1E-09	1E-07	2E-07
25%	1E-08	1E-07	1E-08	3E-09	3E-07	4E-07
50%	3E-08	3E-07	4E-08	7E-09	7E-07	1E-06
75%	6E-08	7E-07	1E-07	3E-08	2E-06	3E-06
90%	2E-07	2E-06	3E-07	7E-08	4E-06	6E-06
95.0%	5E-07	2E-06	4E-07	1E-07	6E-06	9E-06
97.5%	1E-06	3E-06	6E-07	2E-07	9E-06	1E-05
100.0%	1E-05	2E-05	3E-06	2E-06	4E-05	5E-05

FCEM = (Field Chemical Emissions Monitoring [EPRI]) Original oil concentration data, distribution defined by probability

SGS = Subsequent data, trace metal analysis from samples collected for radionuclide analysis.

Combined = Combined forecasts assuming equal probability of the FCEM and SGS data sets.

Initial Point Estimate = The estimate of emissions used in the baseline exposure assessment.
This value was based on the average concentration in the FCEM data.

(Percentile) = The percentile of the predicted distribution corresponding to the initial point estimate.

result, the uncertainty presented here may underestimate the overall uncertainty.

6.14 QUALITATIVE DISCUSSION OF ADDITIONAL UNCERTAINTIES

There are several areas of uncertainty that were not covered in the quantitative analysis. Several of these were discussed in previous sections of this report. Further discussion of two areas of uncertainty is provided below.

6.14.1 Uncertainty Using IUREs

As discussed in section 6.12, there are uncertainties associated with the IUREs. Many of these uncertainties were not included in the quantitative uncertainty analysis because adequate data were not available.

6.14.2 Residence Time and Activity Patterns

In the baseline assessment for the MEI risks, it was assumed that people are exposed to the modeled concentration at their residence for 70 years. The EPA realizes that the average person does not live in the same house for 70 years. However, adjusting for exposure due to changes in residence is no easy task, especially for utilities since plants are located nationwide and roughly 80 percent of the U.S. population live within 50 km of at least one plant.

This uncertainty was not quantified for several reasons. First, a person who moves out of one residence may move into another residence still in the high-concentration area (e.g., person moves next door). Second, a person may move away from an area for a period of time, then move back to the same location. Third, since there is typically more than one person located in the high exposure area, if all except one move away (e.g., one person in the census block stays for 70 years), then the assumption of 70-year residence time holds for the MEI. And, fourth, a person may move from the area of exposure of one utility into an exposure area of another. This person's exposure may change, but may not become zero. Therefore, 70-year exposure is considered a conservative, but reasonable, assumption for the MEI. However, it is still quite uncertain how much residence time and activity patterns would affect the risk estimates.

6.15 DISCUSSION OF FEDERAL INTERAGENCY REVIEW COMMENTS

Previous drafts of chapters 1 through 10, along with the appendices, were reviewed by numerous non-EPA scientists representing industry, environmental groups, academia, and other Federal agencies during the summer of 1995. In February, April, and September 1996, all sections of the draft report underwent additional review by EPA, State and local agencies, and other Federal agencies. The EPA has revised the report, as appropriate, based on the reviewers' comments. The EPA revised the report to incorporate the majority of the comments received. However, there were several comments that could not be fully addressed because of limitations in data, methods, and resources. This section presents comments received by other Federal agencies that could not be substantially addressed in this interim report.

6.15.1 Comment

The Department of Energy (DOE) commented that EPA's assumption that 100 percent of nickel emissions are nickel subsulfide "...is not justified..." and "...results in an overly conservative risk assessment..." rather than in a worst-case scenario based on the available data.[27]

6.15.2 Response

Faced with the uncertainties associated with nickel emissions and health effects, the EPA originally chose the conservative assumption that all nickel is as carcinogenic as nickel subsulfide. Several reviewers including State and local representatives and external scientific peer reviewers supported this approach. However, several industry commenters agreed with DOE's position. To address these issues regarding the nickel risk assessment, the EPA has evaluated various alternative assumptions and presented ranges of potential risks using these various assumptions. Also, the EPA has presented in this chapter a thorough discussion and presentation of the uncertainties associated with the nickel risk assessment.

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7.0 MERCURY ASSESSMENT

7.1 OVERVIEW

Mercury is a highly persistent, naturally occurring metal in the environment. Mercury is typically found in the environment in the elemental state (Hg^0). When it bonds to other chemical elements, it is commonly found as a cation. The mercuric ion may bind to a number of inorganic anions; these are generally referred to as species of divalent mercury (Hg[II]). The mercuric ion may also form one or two bonds with a methyl group forming either monomethyl- or dimethylmercury.

The tendency of this metal to bioaccumulate in aquatic food webs has been well documented.[1-3] Mercury is toxic to humans from both the inhalation[4] and oral exposure routes.[4-6] Mercury is also toxic to other mammals[7-10] and to birds.[11-18] Questions remain regarding both the quantity of mercury and the duration of the exposure required to elicit responses in humans and animals, but it is widely accepted that exposures to mercury produce neurotoxicity. Mercury contamination is a potential concern in the U.S. as indicated by numerous fish advisories[19] and mercury-related water quality standards issued by State Agencies.

7.1.1 The Mercury Cycle

Environmental mercury passes through various environmental compartments and may change physical form and chemical species during this process; these movements are conceptualized as a cycle. The mercury cycle has been studied and described in several recent reports[3,20-22] and its understanding continues to undergo refinement.

The movement and distribution of mercury in the environment can be confidently described only in general terms. There has been increasing consensus on many, but not all, of the detailed behaviors of mercury in the environment.[23] The mercury cycle in Figure 7-1 attempts to illustrate mercury release by both natural and anthropogenic sources into environmental media. The figure illustrates the various transport and transformation processes that are expected to occur. The cycle is global in nature and includes a number of intercompartmental transfers (e.g., water to air to water) that result in a series of loops within the cycle.

7.1.2 Sources of Mercury

Contemporary anthropogenic emissions of mercury are only one component of the global mercury cycle. Releases from human activities today are adding to the mercury reservoirs that

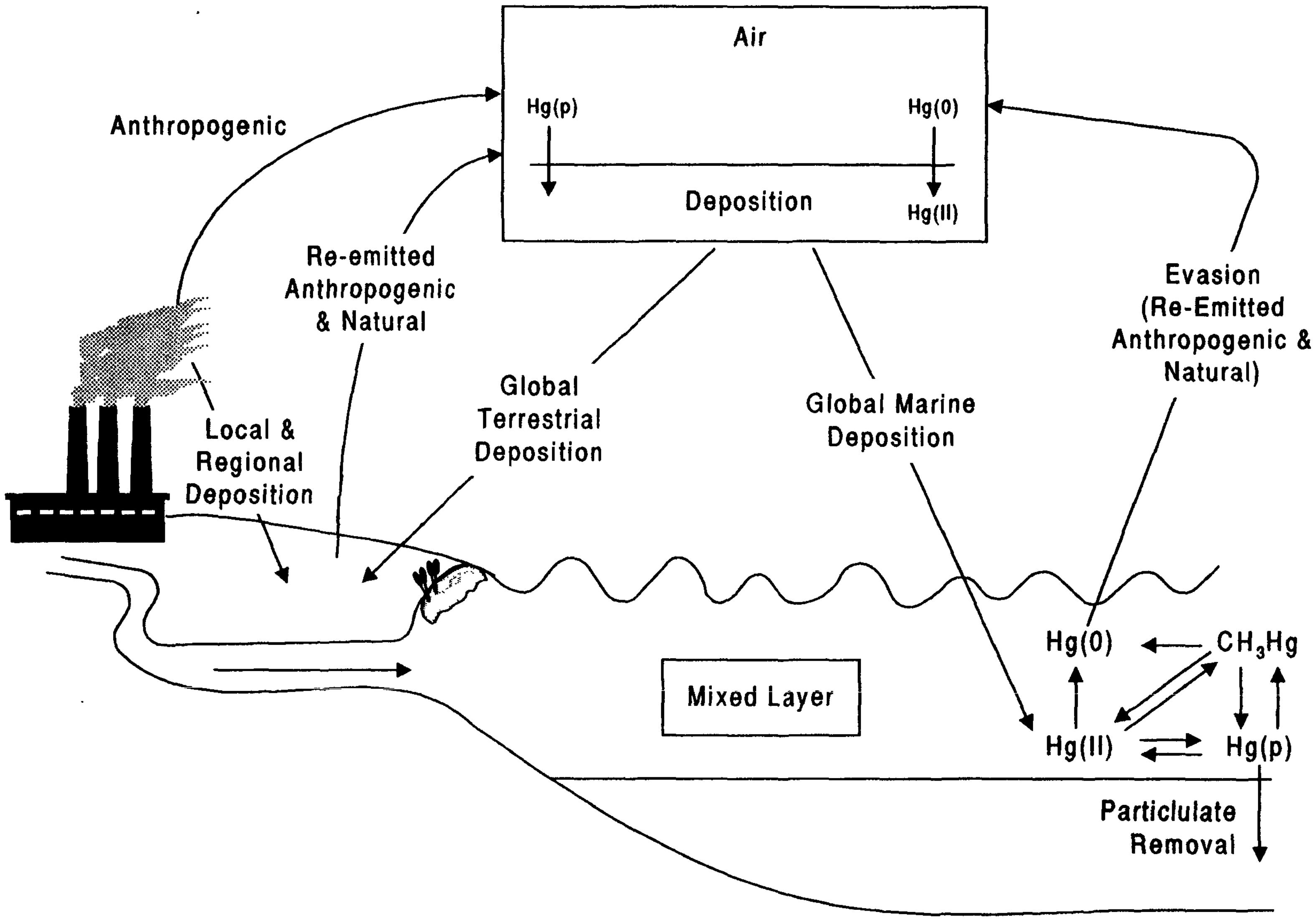


Figure 7-1. The Mercury Cycle

already exist in land, water, and air, both naturally and as a result of previous human activities. Given the present understanding of the global mercury cycle, the flux of mercury from the atmosphere to land or water at any one location is comprised of contributions from the following:

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

Local sources could also include other routes of pollutant release such as direct water discharges in addition to air emissions. Past uses of mercury, such as fungicide application to crops, are also a component of the present mercury burden in the environment.

Different techniques have been used to estimate the mercury concentrations in environmental media that occurred prior to the onset of the industrial revolution. Not surprisingly, there are a broad range of estimates and a great deal of uncertainty associated with each. When these estimates are considered together, they indicate that between 40 and 75 percent of the current atmospheric mercury concentrations are the result of anthropogenic releases.[24-27] This overall range appears to be in agreement with the several-fold increase noted in inferred mercury deposition rates.[28-30] The percentage of current total atmospheric mercury that is of anthropogenic origin may be higher near mercury emissions sources.

A better understanding of the relative contribution of mercury from anthropogenic sources is limited by substantial remaining uncertainties regarding the level of natural emissions as well as the amount and original source of mercury that is reemitted to the atmosphere from existing reservoirs.

Because of the continuous cycling of this element in the environment, it is impossible to separate current mercury concentrations in the environment by origin (i.e., anthropogenic or natural). For example, stack releases of Hg^0 may be oxidized and deposit as Hg(II) far from the source; the deposited mercury may be reduced and re-emitted as Hg^0 only to be deposited again continents away.

Anthropogenic mercury releases are believed to be dominated on the national scale (and global) by industrial processes and by combustion sources that release mercury into the atmosphere. Stack emissions may include both vapor and particulate forms with both divalent and Hg^0 species in the various fractions. The analytic methods needed to speciate mercury in exit gases and emission plumes are being refined, and, at present, there is no scientific consensus in this field of study.

Chemical reactions occurring in the emission plume are also possible. Methods are still being developed to examine the chemistry of emitted mercury close to the emissions source. The speciation of mercury emissions is thought to depend on the fuel used (e.g., coal or oil), flue gas cleaning, and operating temperature. The exit stream may range from almost entirely Hg(II) to nearly all Hg^0. The Hg^0 fraction is thought to be emitted primarily in the gas phase although exit streams containing soot can bind up some fraction of Hg^0. The divalent fraction is thought to be split between gaseous and particle-bound phases.[31] Much of this Hg(II) is thought to be $HgCl_2$.[32]

An emission factor-based approach was used to develop the nationwide emission estimates for utility boilers. The emission estimates for utilities are based on ratios of mass mercury emissions to measures of source activities and nationwide boiler activity levels. The mercury emissions data are estimates; uncertainties occur in the measurement techniques, emission factors, estimates of pollutant control efficiency and nationwide source class activity levels. Details of the emission factor approach are described in chapter 3. Oil- and gas-fired electric utilities are estimated to emit 0.25 ton/yr, and coal-fired electric utilities are estimated to emit 51 ton/yr. This represents approximately 21 percent of the total annual anthropogenic mercury emissions in the U.S. estimated to be about 250 tons for the years 1990 through 1992. If one assumes that the estimate that between 40 and 75 percent of the total current atmospheric emissions are the result of anthropogenic activity is correct, one can calculate that mercury emissions from utilities represent roughly 8 to 15 percent of total emissions of mercury in the U.S. (i.e., anthropogenic plus other sources [e.g., natural emissions]).

Recent estimates of global anthropogenic mercury emissions are about 4,400 tons.[24] Point sources such as fuel combustion, waste incineration, industrial processes (e.g., chlor-alkali plants); and metal ore roasting, refining and processing are the largest point source categories on a world-wide basis.[24] Given this estimate U.S. anthropogenic mercury emissions could account for 6 percent of the global total, and electric utilities in the U.S. would account for about 1 percent of the global anthropogenic mercury emissions, using 1990 emission estimates.

7.1.3 Deposition of Mercury

Most of the mercury species in the atmosphere are in the vapor phase, and measurement data indicate that the vapor phase consists primarily of Hg^0.[3] Particulate-bound mercury species have also been measured in the atmosphere.[33] Concentrations of vapor phase Hg(II) have been predicted[34] near some anthropogenic emission sources. Although methods are being developed,[35] at this time there is no generally accepted method for determining the vapor fraction in the divalent form.

Atmospheric Hg(II) species, in either the vapor or particulate phase, are thought to be subject to more rapid removal from the atmosphere than Hg^0.[34-37] This is the result of the reactivity and water solubility of the gaseous Hg(II) and the gravitational forces that act on particles. Vapor phase Hg(II) species are expected to be scavenged readily by precipitation.[34] These mercury species have lower Henry's law constants than Hg^0 and are assumed to partition strongly to the aqueous phase. Dry deposition (i.e., deposition in the absence of precipitation) of vapor phase Hg(II) is thought to be significant because of its reactivity with surface material. Because of the anticipated rapid depletion, these vapor-phase species are anticipated to be detected in the atmosphere only in the proximity of emission sources. The dry deposition of particulate-bound Hg(II), i.e., Hg(II)(P), is predicted to be dependent on atmospheric conditions and particle size. Particulate mercury is also assumed to be subject to wet deposition due to scavenging by precipitation.

In contrast, Hg^0 vapor is not thought to be susceptible to any major process resulting in direct deposition to the earth's surface. Elemental mercury has a strong tendency to remain airborne[34], and, on non-assimilating surfaces, Hg^0 deposition appears negligible.[38] Elemental mercury can be formed in soil and water[39] through the reduction of Hg(II) species by various mechanisms; Hg^0 typically comprises a minor fraction of the total mercury concentration in these systems. Once formed, the Hg^0 is expected to volatilize to the atmosphere.[24]

There appears to be a potential for deposition of Hg^0 via plant-leaf uptake. Lindberg et al.[36] indicated that forest canopies could accumulate Hg^0 vapor via gas exchange at the leaf surface followed by mercury assimilation in the leaf interior during the daylight hours. This process causes a downward flux of Hg^0 from the atmosphere, resulting in a deposition velocity. Recent evidence[39] indicates that this process does occur but only at air concentrations of Hg^0 well above background for a typical forest area. At more common mercury levels, the forest appears to act as a source of Hg^0 to the atmosphere, with the measured mercury flux in the upward direction. Lindberg et al.[38] noted this may be explained by the volatilization of Hg^0 from the canopy/soil system, most likely the soil. Hanson et al.[40] stated that "dry foliar surfaces in terrestrial forest landscapes may not be a net sink for atmospheric Hg^0, but rather as a dynamic exchange surface that can function as a source or sink dependent on current mercury vapor concentrations, leaf temperatures, surface condition (wet versus dry) and level of atmospheric oxidants." Mosbaek et al.[41] also showed that most of the mercury in leafy plants is due to air-leaf transfer, but that, for a given period of time, the amount of Hg^0 released from the plant-soil system greatly exceeds the amount collected from the air by the plants. It is also likely that many plants accumulate airborne mercury to certain concentrations, after which net deposition of Hg^0 does not occur.

The average residence time of Hg^0 in the atmosphere is approximately 1 year.[3] Atmospheric Hg^0 vapor is thought to deposit to the earth through a series of atmospheric reactions that occur in the aqueous phase (cloud droplets). These reactions may both oxidize Hg^0 to Hg(II) and reduce the Hg(II) to Hg^0. The most important reactions are the oxidation of Hg^0 by ozone, the reduction of Hg(II) by sulfite (SO_3^{-2}) ions, and the production of Hg(II)(P) by complexation with soot:

$Hg^0(g) \rightarrow Hg^0(aq)$ Reaction 1

$Hg^0(aq) + O_3(aq) \rightarrow Hg(II)(aq)$ Reaction 2

$Hg(II)(aq)$ + soot/possible evaporation -> $Hg(II)(p)$ Reaction 3

$Hg(II)(aq) + SO_3^{-2}(aq) \rightarrow Hg^0(aq)$ Reaction 4

(g) = gas phase molecule
(aq) = aqueous phase molecule
(p) = particulate phase molecule
Hg(II) = divalent mercury = Hg^{2+}

The divalent species that result from the oxidation of Hg^0 by ozone can be reduced back to Hg^0 by sulfite; however, the oxidation of Hg^0 by ozone is a much faster reaction than the reduction by sulfite. Thus, a steady-state concentration of aqueous Hg(II) is built up in the atmosphere and can be expressed as a function of the concentrations of $Hg^0(g)$, $O_3(g)$, H^+ (representing acids) and $SO_2(g)$.[42] Note that H^+ and $SO_2(g)$, although not apparent in the listed atmospheric reactions, control the formation of sulfite.

The aqueous-phase Hg(II) is expected to be susceptible to atmospheric removal via wet deposition (precipitation). The third reaction, however, may transform most of the aqueous-phase Hg(II) into the particulate form. Sulfur ions in soot are expected to bind aqueous-phase Hg(II), forming mercury-bearing particles, which may be removed from the atmosphere through wet and dry processes. Concentrations of atmospheric aqueous-phase Hg(II) are expected to limit the reaction with soot. The mercury deposition that results from this proposed chemical pathway may occur far from the mercury emission sources due to the slow reaction rates of the reactions.[25]

7.1.4 Mercury in Soil

Mercury species in soil are subject to a wide array of chemical and biological reactions. Soil conditions (e.g., pH, temperature, and humic content) are typically favorable for the formation of inorganic Hg(II) compounds such as $HgCl_2$, $Hg(OH)_2$, and complexes of these molecules and organic anions.[43] Although inorganic Hg(II) compounds are quite soluble, they form complexes with soil organic matter (mainly fulvic and humic acids) and mineral colloids. This complexing behavior limits the mobility of mercury in soil. Much of the mercury in soil is bound to bulk organic matter and is susceptible to elution in runoff only by being attached to suspended soil particles or humus. Some Hg(II), however, will be absorbed onto dissolvable organic ligands and other forms of dissolved organic carbon (DOC) and may then partition to runoff in the dissolved phase. Currently, the atmospheric input of mercury to soil is thought to exceed greatly the amount leached from soil, and the amount of mercury partitioning to runoff is considered to be a small fraction of the amount of mercury stored in soil. The results of analyses presented in Appendix L, which detail the calibration of soil-water partition coefficients in the watershed model, are consistent with these observations. The affinity of mercury species for soil results in soil acting as a large reservoir for mercury emissions.[22,44]

Elemental mercury can be formed in soil by reduction of Hg(II) compounds/complexes mediated by humic substances.[45] This Hg^0 is expected to vaporize to the atmosphere. Methylmercury may also be formed in soil by various microbiological and abiotic processes acting on inorganic mercury. Typically 1 to 3 percent of the total mercury in surface soil is methylmercury.[46] Some garden soils with high organic content and some contaminated sediments may contain a higher percentage of methylmercury.[47-49]

7.1.5 Plant and Animal Uptake of Mercury

The inorganic Hg(II) and methylmercury complexes in soil are available theoretically for plant uptake and subsequent transfers through the terrestrial food web. Although scientific uncertainty and inter-plant variability are noted, it appears that under typical environmental conditions, plant uptake from soils (especially to aboveground parts of plants) is quite low.[41,43,50] For example, Mosbaek et al.[41] determined that atmospheric mercury uptake accounts for 90 to 95 percent of the total mercury content of the leafy parts of lettuce and rye grass and 40 to 70 percent of the concentration in radish roots. Concentrations of mercury in leafy vegetables generally exceed those of legumes and fruits,[47,51] where it is not clear whether the mercury content results from air and/or soil uptake.

Most plant uptake studies do not explicitly measure both the surrounding soil and air concentrations, and most uptake studies do not examine the species of mercury present in soil. It is unclear if some of the measured variability is attributable to the species of mercury present in the soil or atmosphere, soil properties, or species-specific differences between plants.

Further study of the mechanisms of mercury uptake by plants is needed. Identification of the mercury species present in plants is also needed and may indicate the ability of mercury to be transferred through the food web. Studies indicate that some of the Hg^0 absorbed from the air is readily converted to Hg(II) species in plant tissues.[47] At least some plants appear to methylate some of the accumulated mercury.[52]

Although not well understood, plant uptake of environmental mercury is typically low. Divalent mercury, which appears to be the primary form, does not appear to be as bioavailable to animals as methylmercury. Livestock typically accumulate very low concentrations of mercury in their tissues from foraging or silage/grain consumption.

7.1.6 Mercury in the Freshwater Ecosystem

There are a number of pathways by which mercury can enter the freshwater environment: wet or dry deposition of atmospheric mercury directly into the waterbody; deposition of atmospheric mercury to the watershed followed by transport to waterbodies in runoff (bound to suspended soil/humus or attached to dissolved organic carbon); and leaching of soil mercury species into the waterbody through ground water flow in the upper soil layers. Once in a waterbody, mercury can remain in the water column, be lost from the lake through drainage water, revolatilize into the atmosphere, settle into the sediment or be taken up by aquatic biota. The movements of mercury through any specific waterbody may be unique. The same complexation and transformation processes that occur to mercury species in soil can occur in the aquatic medium in addition to processes specific to the aqueous environment. Only mercury in the water column, the sediment, and other aquatic biota appears to be available to aquatic organisms for uptake.

Much of the mercury in waterbodies binds to matter suspended in the water column and bottom sediment (See Appendix L). The primary form of dissolved mercury in the water column is Hg(II)(see Appendix J). Elemental mercury is produced in freshwater by humic acid reduction of divalent species or demethylation of methylmercury. Some of the Hg^0 will remain in the water column as a dissolved gas, but most will volatilize. Typically, less than 10 percent of the mercury dissolved in the water column exists as a methylmercury complex (See Appendix J). Methylation appears to be a key step in the entrance of mercury into the food chain.[53]

Most of the methylmercury in the water body is the result of methylation of inorganic Hg(II). Biotransformation of inorganic mercury species to methylated organic species in waterbodies can occur in the bottom sediment[54] and the water column.[55] Abiotic processes (e.g., humic and fulvic acids in solution) can also methylate the mercuric ion.[56] There appears to be a large degree of variability among waterbodies concerning the processes that methylate mercury. Bacterial methylation rates appear to increase under aerobic conditions, high temperatures,[57] and low pH.[54,55] Increased quantities of the mercuric species, the proper biologic community, and adequate suspended soil load and sedimentation rate are also important factors; however, these are not the only factors affecting methylation.[57] Anthropogenic acidification of lakes appears to increase methylation rates as well.[54] Not all mercury compounds entering an aquatic ecosystem are methylated, and demethylation reactions[55] as well as volatilization of dimethylmercury decrease the amount of methylmercury available in the aquatic environment.

Most of the methylmercury in the aquatic ecosystem is present in biota, particularly fish.[22,58] Methylmercury accumulates in fish through the aquatic food web; nearly 100 percent of the mercury found in fish muscle tissue is methylated.[58] Methylmercury appears to be primarily passed to planktivorous and piscivorous fish via their diets. Larger, longer-lived fish species at the upper end of the food web typically have the highest concentrations of methylmercury in a given waterbody. A relationship exists between methylmercury content in fish and lake pH, with higher methylmercury content in fish tissue typically found in more acidic lakes.[54,59] (See Appendix J for more details.)

7.2 MERCURY CONCENTRATIONS IN BIOTA

This section provides data on measured levels of mercury in various biota as reported in the scientific literature or as provided by other Federal agencies. It is not feasible, at this time, to determine the sources or source categories responsible for these environmental levels. There are many categories of natural sources (e.g., volcanoes, forest fires) and anthropogenic sources (e.g., municipal waste combustors, medical waste incinerators, lead smelters, chlor-alkali plants, electric utilities) of mercury emissions in the U.S. and worldwide. The utilities are one category of sources of mercury emissions.

The data are being presented to provide general information on estimated levels of mercury in the environment. This information is useful for understanding where mercury is found in the environment and the relative magnitudes of mercury in each of the media or biota. In addition, these data aid in the understanding of environmental fate of mercury, including the bioaccumulation potential of mercury in various species.

In the U.S., the major nonoccupational routes of exposure are expected to be to Hg^0 released from dental amalgams and to mercury in foods. Because of bioaccumulation in the aquatic and marine food webs, the mercury concentrations in fish muscle tissues show the greatest potential for exposure through foods. Mercury concentrations in fish vary greatly, often showing little correlation with proximity to mercury emission sources. Fish in lakes seemingly far removed from anthropogenic sources have been found to have mercury levels of potential concern. In a given waterbody, fish methylmercury concentrations are generally thought to increase with trophic level, as well as with the age and size of the fish. Other lake characteristics such as pH have been found to correlate with methylmercury concentrations in fish.

Tables 7-1 and 7-2 summarize some of the measured mercury concentrations in freshwater sport fish and saltwater fish, respectively. The values in these tables are the average measured concentrations for the fish samples in these studies. They may or may not accurately represent the concentrations in these fish species obtained from other waterbodies or obtained at other times. Also, fish mercury concentrations can vary substantially depending on the size, age, and type of fish within

Table 7-1. Measured Mercury Concentrations in Freshwater Sport Fish (Total Mercury, μg/g wet wt.)

Study	Pike/pickerel	Walleye	Bass	Bottom feeders	Panfish	Trout	Reference
Nationwide Survey (338 locations)	0.52	0.31	0.34 - 0.46	0.09 - 0.27	0.22	0.14	Bahnick et al.[60]
NJDEPE, 1992	0.09-2.82	0.05-8.94	max 1.47				NJDEPE[57]
12 Adirondack Lakes					2-year-old: 0.23 4-year-old: 0.36 *6-year-old: 0.41* 8-year-old: 0.46 10+-year-old: 1.65		*Simonin et al.*[61]
16 New York Lakes					Yellow Perch: 0.01 - 0.64 Pumpkin Seed: 0.01 - 0.19		Mills et al.[62]
42 New Jersey Lakes and Rivers, averages of 9 pH categories	0.15-1.45		0.15-1.16	0.07-0.72	0.10-0.32	0.05-0.64	NJDEPE[57]
Mean concentrations in 65 northern MN lakes	0.14-1.52	0.13-1.75					Sorensen et al.[53]
Ashtabula River, OH; means			LMouth: 0.15 Smouth: 0.12	Carp: 0.05	Bluegill:0.07-0.17		EPA[63]
Saginaw River, MI; means		Fillets: .12		Carp: 0.07			EPA[64]
Northern Michigan lakes	0.10-1.64		0.11-1.00				Gloss et al.[65]
Large fish above dams in 3 Michigan rivers	0.11 - 0.28	< 0.05 to 0.72		< 0.05 to 0.23	< 0.05 to 0.73	0.20 - 0.45	Giesy et al.[66]
38 Wisconsin lakes		0.16-1.74					Lathrop et al.[67]
34 Northern Wisconsin Lakes		0.19 - 0.999					Gerstenberger et al.[68]

Table 7-1. (continued)

Study	Pike/pickerel	Walleye	Bass	Bottom feeders	Panfish	Trout	Reference
53 Florida Lakes			0.04-1.90				Lange et al.[69]
Florida Surface Waters			Lakes: 0.07 - 0.85 Streams: 0.22 - 2.37				Florida DER[70]
9 Canadian Shield Lakes						0.24-3.44	MacCrimmon et al.[71]
Ontario Lakes	0.07-1.28	0.09-3.24					Wren et al.[72]

Note: The U.S. EPA has not determined the source(s) of emissions or effluent responsible for the fish mercury concentrations reported in Tables 7-1 and 7-2. There are many categories of natural and anthropogenic sources of mercury emissions in the U.S. and worldwide. The fossil-fuel-fired utilities in the U.S., which are the sources being studied in this report, are just one category of sources of mercury emissions. The U.S. EPA cannot determine if, or how much, the utilities contribute to the mercury levels reported in Tables 7-1 and 7-2.

Table 7-2. Mercury Concentrations Measured in Some Marine Fish and Shellfish

Fish	Mercury Concentration (µg/g, wet weight)[a]	Comments
Tuna	0.136 - 0.264	Based on NMFS[a] data [75], the mean concentrations measured in 3 types of tuna are as follows: albacore tuna (0.264 µg/g), skipjack tuna (0.136 µg/g) and yellowfin tuna (0.218 µg/g). The U.S. FDA measured the methylmercury concentration in 220 samples of canned tuna in 1991; the average amount of methylmercury measured in these samples was 0.17 µg/g and the measured range was <0.1 - 0.75 µg/g.[76] Richardson, et al.[77] reported an average mercury concentration in Canadian tuna as 0.195 ug/g (wet weight); the range of measured concentrations in tuna was *reported as < 0.01-0.97 ug/g (wet weight).*
Shrimp	0.024 - 0.074	Based on NMFS data[75], the mean concentrations measured in seven types of shrimp are as follows: royal red shrimp (0.074 µg/g), white shrimp (0.054 µg/g), brown shrimp (0.048 µg/g), ocean shrimp (0.053 µg/g), pink shrimp (0.031 µg/g), pink northern shrimp (0.024 µg/g) and Alaska (sidestripe) shrimp (0.042 µg/g).
Pollack	0.04 - 0.15	The Pesticide and Chemical Contaminant Data Base for U.S. FDA (1991/1992) reports the methylmercury concentration in pollack in commerce as 0.04 µg/g. The value 0.15 µg/g came from NMFS.[75]
Salmon	0.019 - 0.063	Based on NMFS data[75], the mean concentrations measured in five types of Salmon are as follows: pink (0.019 µg/g), chum (0.030 µg/g), coho (0.038 µg/g), sockeye (0.027 µg/g), and chinook (0.063 µg/g).
Cod	0.114 - 0.127	Based on NMFS data[75], the mercury concentrations Atlantic Cod is 0.114 µg/g and for Pacific Cod is 0.127 µg/g.
Clam	0.027 - 0.034	Based on NMFS data[75], the mean concentrations measured in four types of clam are as follows: hard (or quahog) clam (0.034 µg/g), Pacific littleneck clam (0 µg/g), soft clam (0.027 µg/g), and geoduck clam (0.032 µg/g).
Flatfish (Flounder)	0.066 - 0.151	Based on NMFS data[75], the mean concentrations measured in nine types of flounder are as follows: Gulf (0.147 µg/g), summer (0.127 µg/g), southern (0.078 µg/g), four-spot (0.090 µg/g), windowpane (0.151 µg/g), arrowtooth (0.020 µg/g), witch (0.083 µg/g), yellowtail (0.067 µg/g), and winter (0.066 µg/g).
Crab	0.07 - 0.183	Based on NMFS data[75], the mean concentrations measured in five types of crab are as follows: blue crab (0.140 µg/g), dungeness crab (0.183 µg/g), king crab (0.070 µg/g), tanner crab (*C. opilio*) (0.088 µg/g), and tanner crab (*C. bairdi*) (0.102 µg/g).

Table 7-2. Continued

Fish	Mercury Concentration (μg/g, wet weight)[a]	Comments
Lobster	0.108 - 0.378	Based on NMFS data[75], the mean concentration in 3 types of lobster are as follows: Spiny (Atlantic) lobster (0.108 μg/g), Spiny (Pacific) lobster (0.210 μg/g), and Northern (Atlantic) lobster (0.378 μg/g).
Fish Sticks	0.02 - 0.15[b]	In an FDA memorandum[78] dated September 24, 1996, "The correct level for fish sticks is 0.02 ppm... These unpublished findings reflect work under the FDA Total Diet Study for the years 1982 to 1990."
Scallop	0.004 - 0.101	Based on NMFS data[75], the mean concentrations measured in four types of scallop are as follows: sea (smooth) scallop (0.101 μg/g), Atlantic Bay scallop (0.038 μg/g), calico scallop (0.026 μg/g), and pink scallop (0.004 μg/g).

Note: The U.S. EPA has not determined the source(s) of emissions or effluent responsible for the fish mercury concentrations reported in Tables 7-1 and 7-2. There are many categories of natural and anthropogenic sources of mercury emissions in the U.S. and worldwide. The fossil-fuel-fired utilities in the U.S., which are the sources being studied in this report, are just one category of sources of mercury emissions. The U.S. EPA cannot determine if, or how much, the utilities contribute to the mercury levels reported in Tables 7-1 and 7-2. There are several other species of saltwater commercial fish (e.g., shark, marlin, swordfish, grouper, croaker) that have been found to contain mercury at concentrations higher than the 10 species included in Table 7-2.

[a] Unless otherwise specified the source of the mercury concentration data is the National Marine Fisheries Service (NMFS).[75]

[b] In the USDA description of Food Codes used in coding the USDA Coding File[79], Fish Sticks are described as containing Atlantic Pollack or Atlantic Cod. The concentration of mercury in these fishes as reported in NMFS[75] are 0.15 ppm for raw Atlantic Pollack and 0.12 for raw Atlantic Cod. The recipe file lists five types of preparation for fish sticks: 1) Cooked, 2) Baked/Broiled, 3) Breaded/Battered Baked, 4) Floured/Breaded, Fried and, 5) Battered, Fried. After adjusting for moisture loss associated with cooking and for the addition of non-fish ingredients (e.g., Batter), the concentration in the fish sticks are expected to be roughly the same concentrations measured in raw Cod and Pollack.

a species or among different species. It is also important to note that the mercury concentrations in fish are in all likelihood the result of both natural and anthropogenic activities.

The modeling assessment did not predict levels in saltwater fish. However, the EPA believes that providing data on measured levels in saltwater fish (Table 7-2) is useful since it is one of the biota that contain mercury. The EPA has not determined if, or how much, mercury emissions from U.S. utilities contribute to these saltwater fish levels. Such an analysis was not possible for this report because of the limitations in the available data, methods, and models and because of the immense complexity of such an assessment.

The little recent data available on mercury in meat products (e.g., beef, chicken, pork) show concentrations to be very low (near the detection limits) for concentrations of both Hg(II) and methylmercury. Meat consumption does not appear to be a major route of exposure to mercury for humans, especially in comparison to mercury concentrations in fish tissues.

Mercury concentrations in green plants are also typically low. Of the plants that do have detectable mercury concentrations, the concentrations tend to be highest in leafy vegetables. Plants grown in mercury-contaminated areas appear to accumulate more mercury than plants in areas where background concentrations are prevalent. However, there are no other noticeable trends in plant concentrations, with mercury levels varying widely among plants and studies. A comparison of measured mercury concentrations in fish with concentrations in plants shows that fish have higher concentrations.

7.3 MEASUREMENT DATA NEAR UTILITIES

The measured mercury concentrations in environmental media around utilities are briefly summarized in this section. These data are not derived from a comprehensive study of mercury around utilities. Despite the need for this effort, such a study does not appear to exist. The quality of the following studies has not been assessed by the U.S. EPA. The data do not appear to be directly comparable among themselves because of differences in analytic techniques and collection methods used. Some of these studies are from older literature and may not reflect current mercury emissions from the sources described.

Anderson and Smith[80] measured mercury levels in environmental media and biota around a 200-MW coal-fired utility in Illinois. The facility had two 152-m-high smokestacks and was equipped with an electrostatic precipitator. Commercial operations at the facility had been ongoing for 6 years when sampling was conducted (from 1973 through 1974). Levels of mercury detected in atmospheric particulate samples collected 4.8 and 9.6 km downwind of the facility were not statistically significantly elevated when compared with samples collected 4.8 km upwind of the site. Mercury levels detected in samples from the upper 2 cm of downwind agricultural soils (sample mean 0.022 ppm mercury) were statistically significantly elevated when compared with upwind samples (0.015 ppm mercury). Core sediment sampling from a nearby lake bed showed statistically significant elevations in sediment mercury concentrations after plant operations began (sample mean 0.049 ppm mercury) when compared with sediment deposits prior to operation (0.037 ppm mercury). No increases were observed in mercury levels in fish from the nearby lake when compared with fish from remote lakes.

Crockett and Kinnison[81] sampled the arid soils around a 2,150-MW coal-fired utility in New Mexico in 1974. The four-stack (two stacks 76 m high and two 91 m high) facility had been operational since 1963 with an estimated mercury release rate of 850 kg/yr. The rainfall in the area averaged 15 to 20 cm/yr. Although a mercury distribution pattern was noted, soil mercury levels near the facility did not differ significantly from background. Given the high amounts of mercury released by the facility and the insignificant amounts detected, the authors speculated that much of the mercury emitted was transported over a larger area, rather than deposited locally.

There are two recent reports of atmospheric mercury measurements in the vicinity of multiple anthropogenic emissions sources. Both studies are of short duration but show elevated mercury concentrations in the local atmosphere or locally collected rain.

Dvonch et al.[33] conducted a 20-day mercury sampling study during August and September of 1993 at four sites in Broward County, Florida. This county contains the city of Ft. Lauderdale as well as an oil-fired utility and a municipal waste combustion (MWC) facility. One of the sample collection sites (site 4) was located 300 m southwest of the MWC facility. Daily measurements of atmospheric particulate and vapor-phase mercury were collected at three of the four sites; daily atmospheric concentrations were not collected at the site near the MWC (site 4) and daily precipitation samples were collected at all sites. The average vapor and particulate phase atmospheric mercury concentrations were higher at the inland sites than at the site near the

Atlantic Ocean, which was considered a background site by the authors. Diurnal variations were also noted; elevated concentrations were measured at night. For example, at site 2, an inland site, the average nighttime vapor-phase concentration was 4.5 ng/m^3; see Table 7-3 for comparison data. This was attributed to little vertical mixing and lower mixing heights that occur in this area at night. Particulate mercury made up less than 5 percent of the total atmospheric mercury. Mercury concentrations in precipitation samples at the four sites were variable; the highest mean concentrations were measured at the inland sites. Given the high levels of precipitation in this area of the U.S. and the short collection period, it is not appropriate to extend this analysis beyond the time frame measured. These mercury concentrations, shown in Table 7-3, are, nonetheless, elevated when compared with the background site.

Keeler et al.[82] and Lamborg et al.[83] reported results of a 10-day atmospheric mercury measurement collection at two sites (labeled as sites A and B) in Detroit, Michigan. There is a large MWC 9 km from site A and a sludge combustor 5 km from site B, and there is a coal-fired utility in the city as well. The vapor-phase mercury concentration encountered at site B during the first days of the experiment exceeded the capacity of the measurement device. Subsequent analyses indicated that the concentrations of mercury encountered were significantly higher than other reported U.S. observations. Table 7-4 presents the concentrations measured at each site.

7.4 MODELING THE FATE OF MERCURY EMISSIONS FROM UTILITIES

It was decided that existing measured mercury data alone were insufficient for an adequate nationwide assessment of the impact of mercury releases from utility boilers; the U.S. EPA therefore chose to examine the fate of mercury released from the stacks of utility boilers using a series of fate and transport models and hypothetical environmental constructs. The mercury concentrations that result from utility emissions were estimated for some environmental media and biota. Significant limitations and uncertainties are associated with the quantitative predictions of the modeling analysis. The results of the analysis provide a qualitative assessment of the environmental fate and transport as well as plausible estimates of environmental mercury concentrations that result from utility emissions. In this analysis, there were three main types of modeling efforts: (1) modeling of mercury atmospheric transport on a regional (i.e., long-range transport) basis (beyond 50 km of source); (2) modeling of mercury atmospheric transport on a local scale (within 50 km of source); and (3) estimating environmental

Table 7-3. Mercury Concentrations in the Atmosphere and Mercury Measured in Rainwater Collected in Broward County, Florida[33]

Site Description	Avg. vapor-phase Hg conc., ng/m^3	Avg. particulate Hg conc. pg/m^3	Avg. tot. Hg conc. in rain, ng/L (range)	Avg. reactive Hg conc. in rain, ng/L (range)
Background near Atlantic Ocean (site 1)	1.8	34	35 (15-56)	1.0 (0.5-1.4)
Inland (site 2)	3.3	51	40 (15-73)	1.9 (0.8-3.3)
Inland (site 3)	2.8	49	46 (14-130)	2.0 (1.0-3.2)
Inland (site 4), 300 m from MWC	-	-	57 (43-81)	2.5 (1.7-3.7)

- = not measured

Table 7-4. Mercury Concentrations Measured at Two Sites in the Atmosphere over Detroit, Michigan[82,83]

Site	Mean vapor-phase mercury concentrations in ng/m^3 (max. meas. value)	Mean particulate-phase mercury concentrations in pg/m^3, (max. meas. value)
Detroit, MI, site A	>40.8 (>74)	341 (1,086)
Detroit, MI, site B	3.7 (8.5)	297 (1,230)

concentrations using the indirect exposure model (IEM) along with the results of the atmospheric modeling. (For a complete description of the models utilized see Appendix M). The models used for these aspects of the study are shown in Table 7-5.

Factors important in modeling the fate, transport, and environmental concentrations are listed in Table 7-6. This table briefly describes the possible effects of these factors on the fate and transport of mercury and the estimates of mercury concentration in various media and the means by which these were addressed.

7.4.1 Long-range Transport Analysis

The long-range transport analysis modeled site-specific, utility mercury emissions data to generate mean, annual atmospheric mercury concentrations and deposition estimates across the continental U.S. (see Figures 7-2, 7-3, 7-4, and 7-5). The RELMAP atmospheric model was used to model cumulative mercury deposition from coal- and oil-fired utility boilers in the U.S. Assumptions were made concerning the form and species of mercury emitted from each source class.

Table 7-5. Models Used to Predict Mercury Air Concentrations, Deposition Fluxes, and Environmental Concentrations

Model	Description
RELMAP	Predicts average annual atmospheric mercury concentration and wet and dry deposition flux for each 40-km^2 grid in the United States due to all anthropogenic sources of mercury in the United States
COMPDEP II	Predicts average concentration and deposition fluxes within 50 km of emission source
IEM2	Predicts environmental concentrations based on air concentrations and deposition rates to watershed and waterbody

Table 7-6. Factors Potentially Important in Multipathway Modeling of Mercury – How They are Addressed in This Assessment

Factor	Importance and possible effect on mercury exposure	Means of addressing in this study
Mercury emission rates from stack	Increased emissions will result in a greater chance of adverse impacts on environment.	Emissions of model plants based on emissions inventory.
Mercury species emitted from stack	More soluble species will tend to deposit closer to the source.	Two species considered to be emitted from source: elemental and divalent mercury
Form of mercury emitted from stack	Transport properties can be highly dependent on form.	Both vapor and particle-bound fractions considered.
Deposition differences between vapor and particulate-bound mercury	Vapor-phase forms may deposit significantly faster than particulate-bound forms.	Deposition (wet and dry) of vapor-phase forms calculated separately from particulate-bound deposition.
Transformations of mercury after emission from source	Relatively nontoxic forms emitted from source may be transformed into more toxic compounds.	Equilibrium fractions estimated in all environmental media for three mercury species: elemental mercury, divalent species, and methylmercury.
Facility locations	Effects of meteorology and terrain may be significant.	Both a humid and less humid site considered. Effect of terrain on results addressed separately.
Location relative to local mercury source	Receptors located downwind are more likely to have higher exposures. Influence of distance depends on source type.	Three distances in downwind direction considered.
Contribution from nonlocal sources of mercury	Important to keep predicted impacts of local sources in perspective.	Results of local mercury source are combined with estimate of impact from nonlocal sources from RELMAP.

Table 7-6. Continued

Factor	Importance and possible effect on mercury exposure	Means of addressing in this study
Uncertainty	Reduces confidence in ability to estimate exposure accurately.	Probabilistic capabilities possible for any combination of sources and scenarios. In the current study, limited uncertainty analyses conducted for major aspects of atmospheric transport modeling.

RELMAP = Regional Lagrangian Model of Air Pollution.

7.4.1.1 RELMAP Model Uncertainty. In general, it is believed that the RELMAP will tend: (1) to overestimate Hg values in urban areas; and (2) to underestimate Hg values in rural areas and in the urban center of larger cities. There were limited data available from several years to check the model and confirm those beliefs. The RELMAP analysis was somewhat generic because the evaluation was based on the meteorology of 1989, thought to be a reasonably typical year, and used 1990 Hg emission estimates. The discussion below summarizes the limited comparison between model predictions and measured data. Please note that the model predictions described in this section include other major sources and a natural background concentration. Overall, the RELMAP seems to predict Hg values within a factor of 2 of measured values and is relatively unbiased in its predictions.

For the ambient vapor-phase mercury concentrations, the data collected at three remote sites agreed very closely with the RELMAP predictions. With limited data in a more urbanized setting in Florida, the RELMAP underpredicted average concentrations at two sites by about a factor of 2 and was in agreement with the data at a third site. For particulate phase Hg, the RELMAP underpredicted concentrations by a factor of 2 at Detroit (18-day sample) and was in approximate agreement in Broward County in Florida (2 weeks of data). At three other sites (Ann Arbor and Pellston, MI, and Underhill, VT), the modeled concentrations were in approximate agreement with a tendency for slight overestimates of the measured values.

For wet deposition, there were 10 sites in Minnesota, upper Michigan, and northeastern North Dakota that recorded averages from about 4 to 10 ug/m^2 (multiple years of data). The RELMAP concentrations for these 10 sites ranged from 2 to 11 ug/m^2, thus showing reasonable agreement with a slight tendency to overestimate. At two lake sites in Wisconsin, the RELMAP

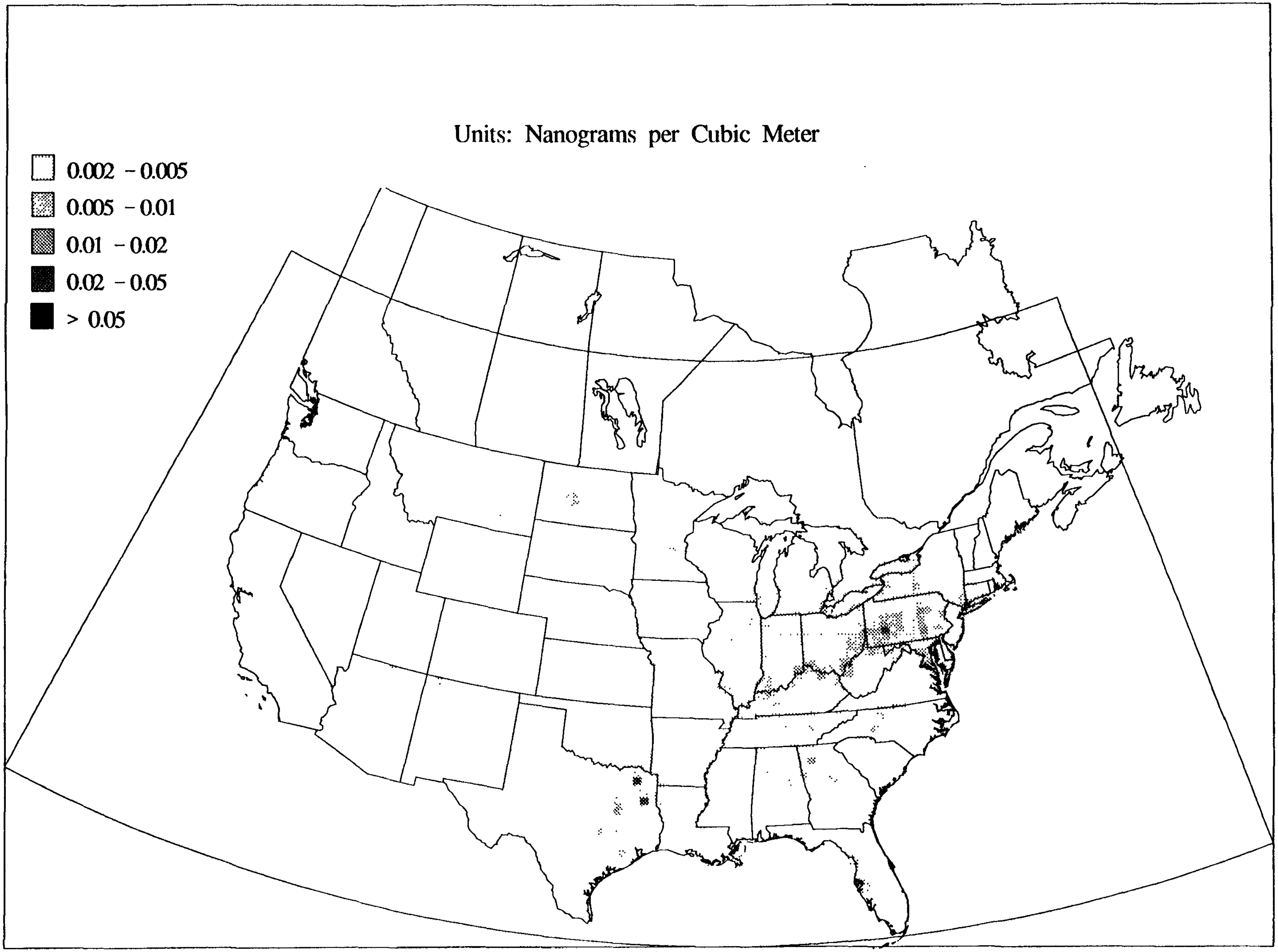


Figure 7-2. Average ionic Mercury (Hg^{+2}) Concentration from Electric Utilities Only (Base)

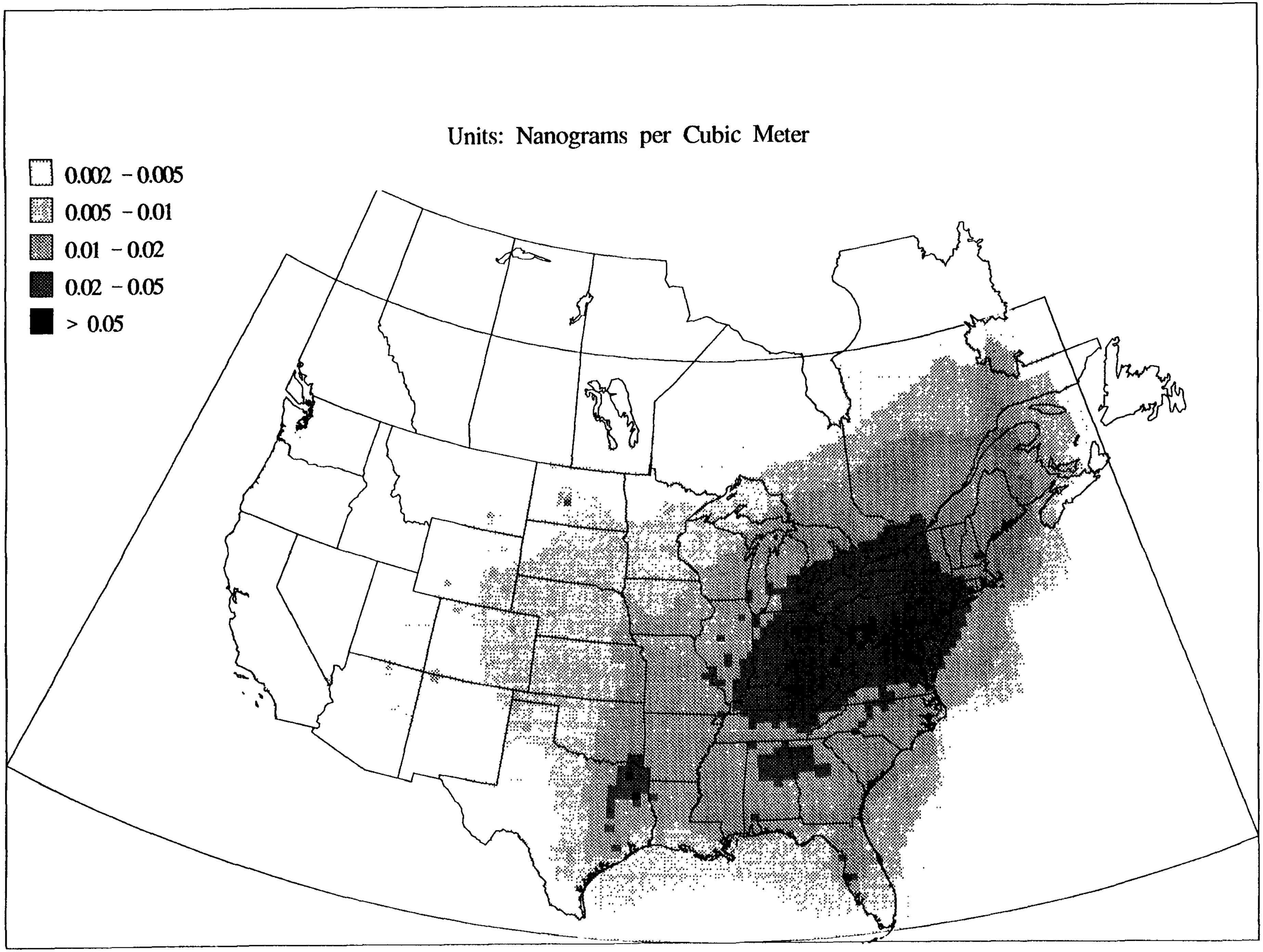


Figure 7-3. Average Elemental mercury (Hg^0) Concentration from electric Utilities Only (Base)

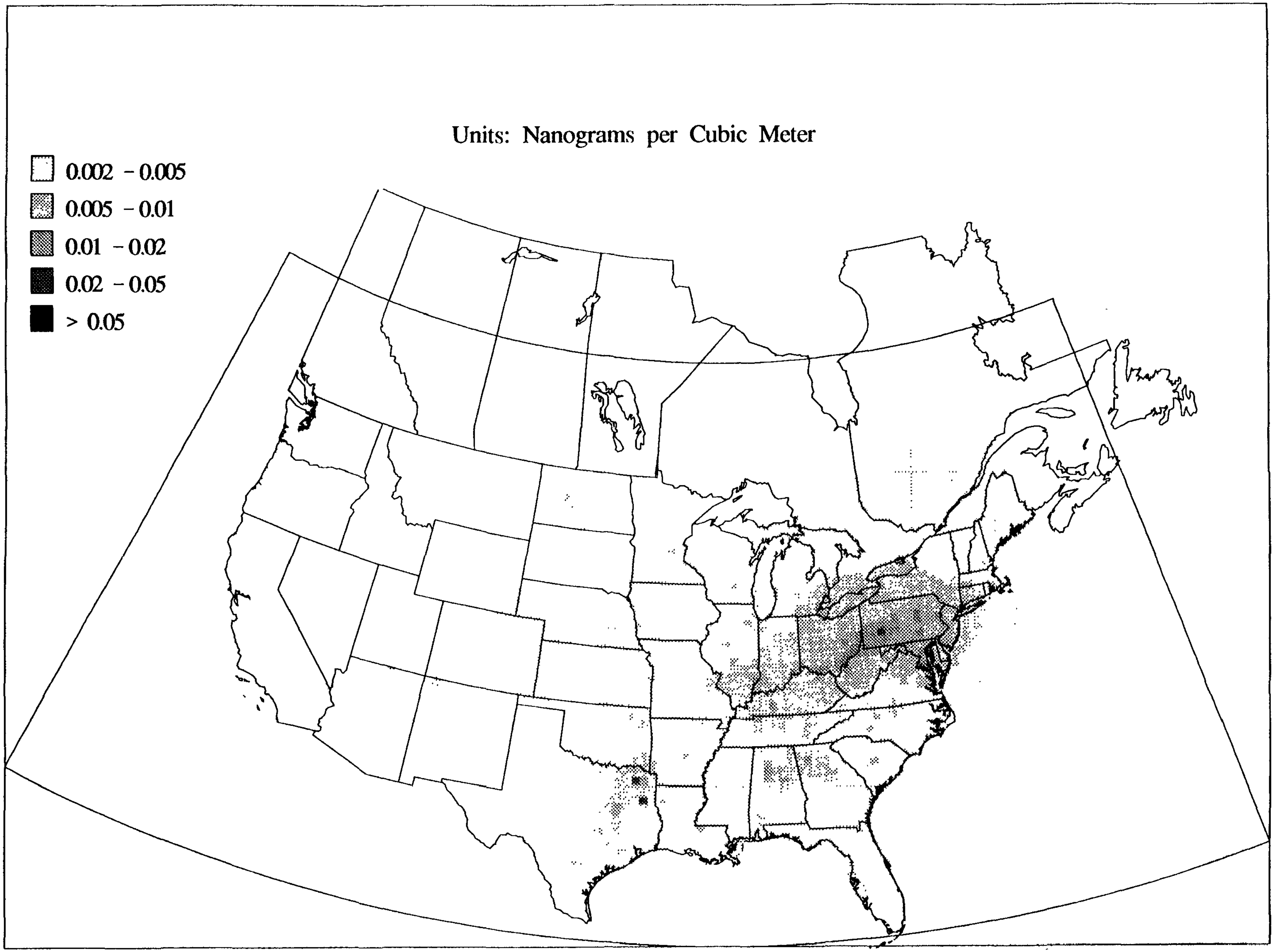


Figure 7-4. Average Particulate Mercury concentration from Electric Utilities Only (Base)

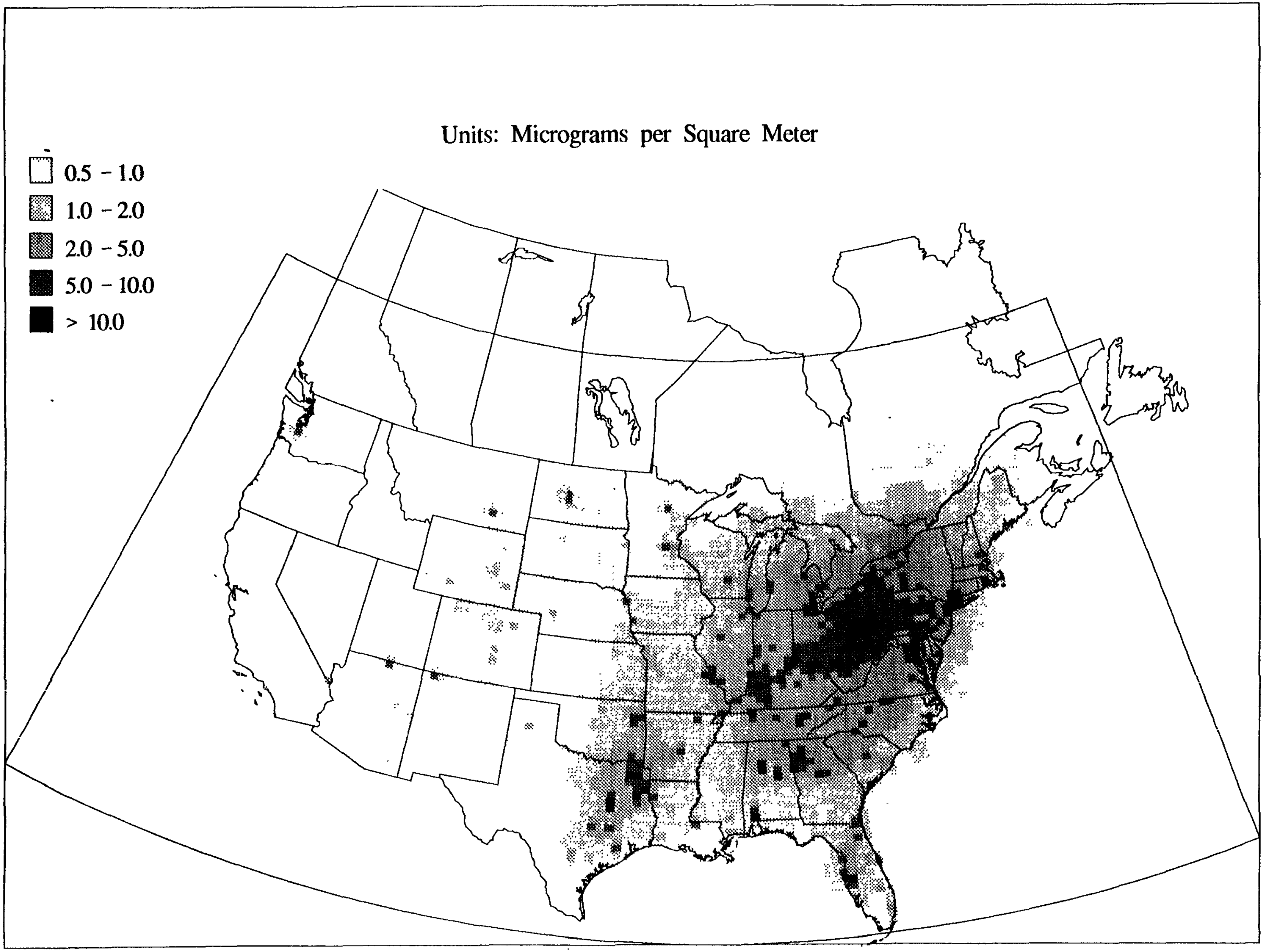


Figure 7-5. Total mercury Wet and Dry Deposition from Electric Utilities Only (Base)

estimates were very close to recorded values. In the early 1990s, there were four sites in somewhat less remote areas that recorded data that were overpredicted by the RELMAP, but the overprediction was within a factor of 2.

Urban wet deposition comparisons cannot be made due to the paucity of long-term data. Limited data (20 days) from four sites in the Fort Lauderdale area indicated that the RELMAP was approximately on target, with a tendency to underestimate.

7.4.1.2 Priliminary Observations from Long-range Transport Modeling Analysis. Based on this modeling analysis, the response from internal and external peer reviewers of the atmospheric mercury modeling strategy, and a review of recent scientific literature on atmospheric mercury, the Agency has arrived at several observations. Tables 7-7 and 7-8 provide specific data on local and regional impacts at an eastern and western site.

The Agency has a high level of confidence in the following observations:

1. Spatial Pattern of Emissions of Mercury from Utilities

The pattern of mercury emissions from utility boilers is fairly uniform over the eastern half of the continental U.S. with the highest concentration of emissions in the Ohio River Valley and in other major coal mining areas.

The Agency's confidence in the following observations is moderate:

2. Chemical and Physical Form of Emissions of Mercury from Utilities

Engineering estimates of the chemical and physical forms of the emissions of mercury from utilities in the continental U.S. have been developed. A number of measurement studies have been conducted with emission samples taken directly from the interior of exhaust stacks. These samples of stack exhaust that may have temperatures of more than 300° C (570° F) probably do not accurately describe the chemical and physical forms of the emissions as they manifest themselves in the regional-scale atmosphere. The Agency has considered many possible effects of cooling and dilution with ambient air of exhaust plume constituents immediately after emission from the stack in the development of the following estimates of mercury emissions to the regional-scale atmosphere by utilities:

Table 7-7. Combination of Local and Regional Impacts: Contribution of Regional Sources to Key Output at Eastern Site[a]

Eastern site	Air (ng/m^3)		Deposition (μg/m^2/yr)		Water concentration (ng/L)		Trophic level 4 fish concentration (μg/g)	
Facility	Predicted value	% RELMAP	Predicted value	% RELMAP	Predicted value	% RELMAP	Predicted value	% RELMAP
2.5 km								
large coal-fired utility boiler	2.5×10^{-2}	94.89%	1.1×10^{1}	22.45%	6.9×10^{-1}	17.01%	1.6×10^{-1}	17.01%
medium coal-fired utility boiler	2.6×10^{-2}	91.40%	9.2×10^{0}	26.72%	5.7×10^{-1}	20.63%	1.3×10^{-1}	20.63%
small coal-fired utility boiler	2.7×10^{-2}	90.37%	3.6×10^{0}	68.35%	2.0×10^{-1}	59.19%	4.7×10^{-2}	59.19%
medium oil-fired utility boiler	2.4×10^{-2}	99.39%	2.7×10^{0}	92.61%	1.3×10^{-1}	89.26%	3.1×10^{-2}	89.26%
10 km								
large coal-fired utility boiler	2.6×10^{-2}	93.78%	4.5×10^{0}	54.92%	2.2×10^{-1}	52.42%	5.3×10^{-2}	52.42%
medium coal-fired utility boiler	2.7×10^{-2}	89.94%	3.8×10^{0}	65.16%	1.9×10^{-1}	61.46%	4.5×10^{-2}	61.46%
small coal-fired utility boiler	2.6×10^{-2}	92.29%	2.7×10^{0}	91.25%	1.3×10^{-1}	90.33%	3.1×10^{-2}	90.33%
medium oil-fired utility boiler	2.4×10^{-2}	99.29%	2.5×10^{0}	98.45%	1.2×10^{-1}	98.25%	2.8×10^{-2}	98.25%
25 km								
large coal-fired utility boiler	2.5×10^{-2}	94.83%	2.9×10^{0}	84.59%	1.4×10^{-1}	84.66%	3.3×10^{-2}	84.66%
medium coal-fired utility boiler	2.6×10^{-2}	91.67%	2.8×10^{0}	88.37%	1.3×10^{-1}	88.17%	3.2×10^{-2}	88.17%
small coal-fired utility boiler	2.5×10^{-2}	96.13%	2.5×10^{0}	97.54%	1.2×10^{-1}	97.49%	2.9×10^{-2}	97.49%
medium oil-fired utility boiler	2.4×10^{-2}	99.53%	2.5×10^{0}	99.60%	1.2×10^{-1}	99.60%	2.8×10^{-2}	99.60%

Table 7-8. Combination of Local and Regional Sources: Contribution of Regional Sources to Key Output at Western Site

Western Site	Air (ng/m3)		Deposition (ug/m^2/yr)		Water concentration (ng/l)		Trophic level 4 fish concentration (ug/g)	
Facility	Predicted value	% RELMAP	Predicted value	% RELMAP	Predicted value	% RELMAP	Predicted value	% RELMAP
2.5 km								
large coal-fired utility boiler	7.0 x 10^{-3}	71.15%	3.4 x 10^{0}	9.13%	2.4 x 10^{-1}	6.74%	6.2 x 10^{-2}	6.74%
medium coal-fired utility boiler	7.6 x 10^{-3}	66.05%	2.6 x 10^{0}	12.19%	1.7 x 10^{-1}	9.29%	4.5 x 10^{-2}	9.29%
small coal-fired utility boiler	7.3 x 10^{-3}	68.19%	7.4 x 10^{-1}	42.26%	4.8 x 10^{-2}	33.65%	1.2 x 10^{-2}	33.65%
medium oil-fired utility boiler	5.1 x 10^{-3}	97.12%	3.8 x 10^{-1}	83.20%	2.1 x 10^{-2}	77.87%	5.3 x 10^{-3}	77.87%
10 km								
large coal-fired utility boiler	6.8 x 10^{-3}	73.38%	1.4 x 10^{0}	22.58%	8.3 x 10^{-2}	19.48%	2.1 x 10^{-2}	19.48%
medium coal-fired utility boiler	7.3 x 10^{-3}	68.54%	9.2 x 10^{-1}	33.99%	5.5 x 10^{-2}	29.03%	1.4 x 10^{-2}	29.03%
small coal-fired utility boiler	6.7 x 10^{-3}	74.82%	4.4 x 10^{-1}	70.89%	2.4 x 10^{-2}	66.85%	6.2 x 10^{-3}	66.85%
medium oil-fired utility boiler	5.1 x 10^{-3}	97.19%	3.3 x 10^{-1}	94.41%	1.7 x 10^{-2}	93.28%	4.5 x 10^{-3}	93.28%
25 km								
large coal-fired utility boiler	6.5 x 10^{-3}	77.40%	6.7 x 10^{-1}	47.14%	3.7 x 10^{-2}	43.75%	9.5 x 10^{-3}	43.75%
medium coal-fired utility boiler	7.0 x 10^{-3}	71.42%	5.3 x 10^{-1}	59.17%	2.9 x 10^{-2}	56.39%	7.4 x 10^{-3}	56.39%
small coal-fired utility boiler	6.0 x 10^{-3}	84.01%	3.6 x 10^{-1}	86.17%	1.9 x 10^{-2}	85.03%	4.9 x 10^{-3}	85.03%
medium oil-fired utility boiler	5.1 x 10^{-3}	97.94%	3.2 x 10^{-1}	97.77%	1.6 x 10^{-2}	97.54%	4.3 x 10^{-3}	0.9754

- About 24 metric ton/yr (50 percent of the total) are emitted in the form of Hg^0

- About 15 metric ton/yr (30 percent of the total) are emitted in the form of Hg(II)

- About 10 metric ton/yr (20 percent of the total) are emitted in the form of Hg(P).

3. Range of Total Mercury Deposition Magnitudes

Based on modeling and emission inventory analysis, the Agency finds that the 10th and 90th percentile range of total mercury deposition from utilities for the continental U.S. spans more than 2 orders of magnitude.

4. Factors Contributing to Fate of Mercury Emissions from Utilities

There are three principal factors contributing to the modeled deposition patterns:

- Emission source locations

- Amount of divalent and particulate mercury emitted or formed in the atmosphere

- Climate and meteorology.

Whether these factors are strongly controlling the actual deposition patterns of utility emissions of mercury cannot be tested due to the absence of source-specific deposition observations.

The Agency's confidence in the following statements is low:

5. Fate of Different Forms of Mercury Emissions from Utilities

Modeling estimates of the transport and deposition of utility air emissions of Hg^0, Hg(II), and Hg(P) in the continental U.S. have revealed the following atmospheric fates.

- Of the total amount of Hg^0 that is emitted, about 1 percent (0.3 metric ton/yr) may be dissolved in cloud and rainwater, most of which is chemically transformed into Hg(II) by dissolved tropospheric ozone and/or adsorbed to particulate soot in aqueous suspension. This mercury is subsequently deposited in rainfall and snowfall to the surface. The vast majority of Hg^0 does not readily deposit to the surface and is transported outside the United States or vertically diffused to the free atmosphere to become part of the global cycle.

- Of the total amount of Hg(II) that is emitted, about 70 percent (10.2 metric ton/yr) deposits to the surface through wet or dry processes within the continental U.S. The remaining 30 percent is transported outside the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle.

- Of the total amount of Hg(P) that is emitted, about 39 percent (3.8 metric ton/yr) deposits to the surface through wet or dry processes within the continental U.S. The remaining 61 percent is transported outside the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle.

The Agency's confidence in the following observation is low due to compounding scientific uncertainties regarding the chemical and physical form of mercury emissions from utilities and the lack of data on atmospheric transformations of mercury between these different forms.

6. Area Most Impacted by Deposition of Mercury from Utilities

Based on modeling analysis of the wet and dry deposition of utility air emissions of all forms of mercury within the continental U.S., the Agency finds that the following geographic areas have the highest annual rate of total deposition of mercury in all forms (above the 90th percentile level):

- Southeastern Great Lakes and Ohio River Valley

- Most of central and western Pennsylvania

- The urban corridor from Washington, DC, to New York City.

- In the vicinity of many of the larger cities in the eastern U.S. and in numerous isolated locations where relatively large coal-fired utilities are located.

The Agency's confidence in this observation is low since there are no observational data with which to evaluate model performance for utilities alone.

7. Area Least Impacted by Atmospheric Deposition of Mercury from Utilities

Based on modeling analysis of the wet and dry deposition of utility emissions of all forms of mercury within the continental U.S., the Agency predicts that the following geographic areas have the lowest annual rate of total deposition of mercury in all forms (below the 10th percentile level):

- Most of the Pacific Coast and Great Basin regions
- Parts of the northern Rocky Mountain region.

The Agency's confidence in this prediction is low since there are no observational data with which to evaluate model performance for utilities alone.

7.4.2 Local Analysis

The local impact analysis estimated the impacts of mercury emissions within 50 km of individual coal- or oil-fired utility plants. Natural and recycled sources of mercury were not factored into the analysis. Model plants, which were developed to represent actual utilities and their emissions, were located at both a hypothetical western and eastern U.S. site (for descriptions of the sites, see Appendix K). Mercury concentrations in environmental media and biota were estimated through fate and transport modeling of the stack emissions. The contribution of regional mercury transported from utilities was also included in the assessment of the impacts of the single source. The models used in the local-scale analysis include a modified version of the COMPDEP air dispersion model, the *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*,[84] which is composed of a series of fate and transport models, and a 1993 addendum to the 1990 methodology document.[85] The COMPDEP is an air dispersion model, based primarily on the COMPLEX I model (the "COMP" part of the acronym), which has been modified and updated to use wet and dry deposition algorithms (the "DEP" part of the acronym). The COMPLEX I is designed for screening analysis of air pollution transport in complex terrain, i.e., for locations where terrain features have elevations above the top of the stack of a facility. Together these models were used to estimate mercury concentrations in environmental media.

The three forms of mercury modeled were Hg^0 vapor, Hg(II) vapor, and particulate mercury Hg(P). The vapor/particle (V/P) ratio was assumed to be equal to the V/P ratio as it would exist in stack emissions. Average ambient U.S. particle size profile data were used.[86] Approximately 93 percent of the particles were assumed to be accumulation particles, approximately 0.3-μm-diameter particles, and approximately 7 percent were assumed to be coarse particles with a diameter of 5.7 μm. Both wet and dry deposition of mercury emissions was considered in the analysis. The assumed deposition parameters for emitted forms of mercury addressed in this study are compared in Table 7-9.

7.4.2.1 Multipathway Modeling. An updated methodology, called Indirect Exposure Modeling 2 (IEM2), uses atmospheric chemical loadings to perform mass balances on a watershed soil element and a surface water element. The mass balances are performed for total mercury, which is assumed to speciate into three components: Hg^0, Hg(II), and methylmercury. The fraction of mercury in each of these components is specified for the soil and the surface water elements. Loadings and chemical properties are given for the individual mercury components, and the overall mercury transport and loss rates are calculated by the methodology. An overview of the IEM2 watershed modules is shown in Figure 7-6.(See Appendix M for more details.)

The IEM2 first performs a terrestrial mass balance to obtain mercury concentrations in watershed soils. Soil concentrations are used in addition to vapor concentrations and deposition rates to calculate concentrations in various food plants. These are used, in turn, to calculate concentrations in terrestrial agricultural animals (i.e., beef, pork, poultry). The IEM2 next performs an aquatic mass balance driven by direct atmospheric deposition as well as the runoff and erosion loads from watershed soils. Methylmercury concentrations in freshwater fish are estimated from total dissolved water concentrations using bioaccumulation factors (BAFs). The modeling did not estimate concentrations in marine fish.

For the analysis, the IEM2 methodology was expanded to handle multiple chemical components in a steady-state relationship. The fraction of each chemical component in the soil and water column was specified by the user. The methodology predicts the total chemical concentration in watershed soils and the waterbody based on loading and dissipation rates specified for each of the components. The model tracks the buildup of watershed soil concentrations over the years given a steady depositional load and long-term average hydrologic behavior.

To predict mercury levels in freshwater fish, a bioaccumulation factor (BAF) approach was used instead of the bioconcentration factor (BCF) approach described in the EPA's *Methodology for Assessing Health Risks Associated with Indirect*

Table 7-9. Comparison of Assumed Deposition Parameters for Emitted Forms of Mercury[a]

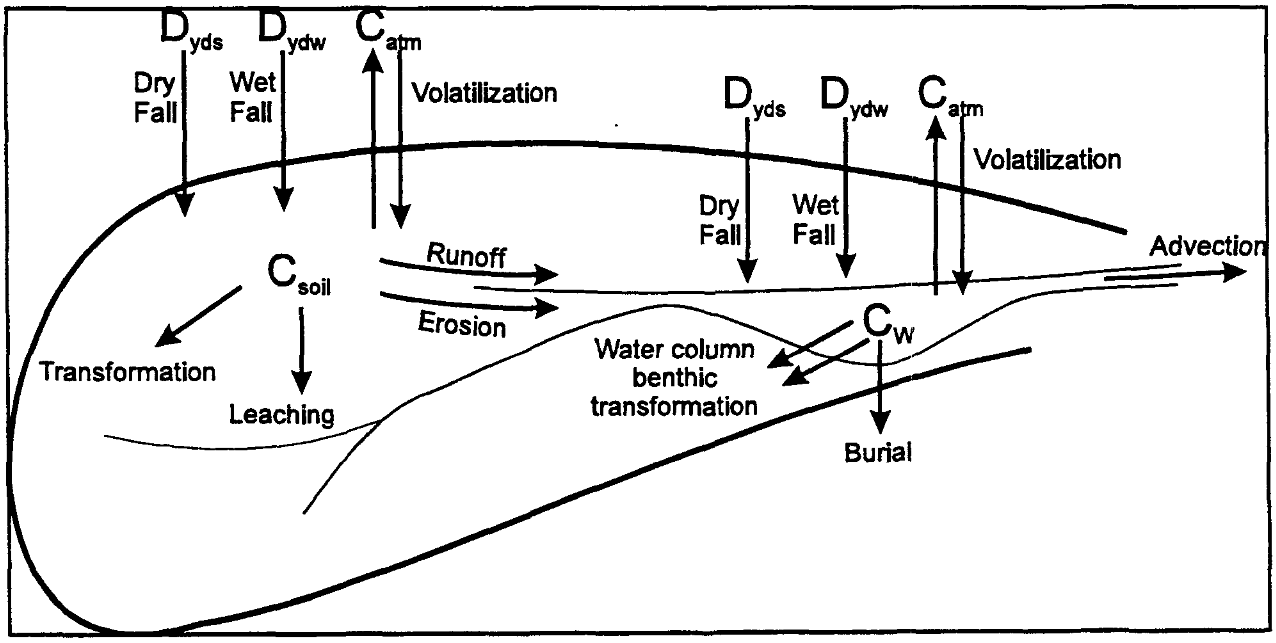
Parameter	Elemental mercury vapor	Divalent mercury vapor	Divalent mercury particulate
Dry deposition velocity (cm/s)	0	0.3 - 1.0	0.2 - 1.4[c]
Washout ratio (unitless)	1.6×10^4	1.6×10^6	NA
Wet deposition scavenging coefficient (/s)[b]	3×10^{-6} to 1×10^{-5}	3×10^{-4} to 1×10^{-3}	2.2×10^{-4} (light precipitation) to 1.5×10^{-3} (heavy precipitation)

[a] See Appendices J through M for the basis for these parameters and other assumptions.

[b] For elemental and divalent vapor, this is calculated by $L = WP/L$, where W is the washout ratio, P is the representative precipitation intensity for the hour, and L is the predicted mixing height for the hour. Due to the dependence on mixing height, the upper end values of the ranges shown routinely occur even for light precipitation.

[c] Based on particle density of 1.8 g/cm^3, particle diameter of 2 μm, surface roughness of 0.3 m, and ambient air temperature of 295 K.

Figure 7-6. Overview of the IEM2 watershed modules



Definitions for Figure 7-6

C_{soil}	=	chemical concentration in upper soil	mg/L
C_w	=	chemical concentration in water body	mg/L
C_{atm}	=	vapor phase chemical concentration in air	μg/m^3
D_{yds}	=	average dry deposition to watershed	mg/yr
D_{yws}	=	average wet deposition to watershed	mg/yr

Exposure to Combustor Emissions.[84] A BAF measures the total uptake rate from water, food, and sediments and is generally derived from field studies. The BAF selected was based on a modification of the concept described in the 1993 U.S. EPA Great Lakes Water Quality Initiative,[87] which previously developed a BAF for mercury of 130,440 based on total measured mercury (all species) in water. The BAFs used for this analysis were 66,200 and 335,000 for trophic level three and trophic level four, respectively. (See Appendices J through M for background information and further discussion of the models, BAFs, and other assumptions, input data, and parameters used in RELMAP, COMPDEP, and IEM modeling.)

7.4.2.2 Model Plants. Model plants representing four types of facilities were developed to represent a range of mercury emissions sources. The facilities selected were these: a large coal-fired utility, a medium coal-fired utility, a small coal-fired utility, and a medium oil-fired utility boiler.

Parameters for each model plant were selected after evaluation of the characteristics of a given source category and current knowledge of mercury emissions from that source category. Important variables for the mercury assessment include mercury emission rates, mercury speciation, and mercury transport/deposition rates. Important model plant parameters include stack height, stack diameter, stack volumetric flow rate, stack gas temperature, plant capacity factor (relative average operating hours per year), stack mercury concentration, and mercury speciation. Values for these parameters are given in Table 7-10. Emission estimates are thought to represent typical levels of mercury emitted from existing utility sources.

Two generic sites were considered: a humid site east of 90 degrees west longitude, and an arid site west of 90 degrees west longitude. The primary differences between the two settings as parameterized are the assumed erosion characteristics for the watershed and the amount of dilution flow from the waterbody. The eastern site has generally steeper terrain in the watershed than the other site (see Appendix K for details about the specific site parameters used in the Assessment).

7.4.2.3 Hypothetical Settings for Estimating Mercury Concentrations in Various Media and Biota. Three types of model plant settings were utilized: rural (agricultural), lacustrine (near lakes), and urban. These three settings were selected because of the variety they encompass and because each was expected to provide "high-end" mercury concentrations in

Table 7-10. Process Parameters for Model Plants

Model plant	Plant size (MW)	Capacity (% of yr)	Stack height (ft)	Stack diamet. (ft)	Hg emission rate (kg/yr)	Spec. % (Hg°/Hg(II)/ Hg part)	Exit veloc. (m/s)	Exit temp. (°F)
Large Coal-fired	975	65%	732	27	230	50/30/20	31.1	273
Medium Coal-fired	375	65%	465	18	90	50/30/20	26.7	275
Small Coal-fired	100	65%	266	12	10	50/30/20	6.6	
Medium Oil-fired	285	65%	290	14	2	50/30/20	20.7	322

environmental media of potential concern (e.g., elevated mercury concentrations are expected in the waters of the lacustrine setting).

Table 7-11 shows the predicted methylmercury concentrations in soil, water, and air for the eastern and western sites. These media concentrations were used to calculate predicted methylmercury concentrations in biota for the two sites, and these values are also shown in Table 7-11.

7.4.2.4 Discussion of COMPDEP and IEM2 Model Limitations, Uncertainty, Validation, and Supporting Empirical Data. For the COMPDEP model results predicted in this interim report, there appear to be no measured data collected around utilities to adequately assess whether the predicted air concentrations and deposition rates can be corroborated. Section 7.3 of this report briefly describes four studies conducted in the vicinity of utilities: Anderson and Smith,[80] Crockett and Kinnison,[81] Dvonch et al.,[33] and Keeler et al.,[82] and Lamborg et al.[83] (The last two studies describe different data collected from the same area over the same period of time.)

Mercury concentrations generated prior to 1980 are generally suspect because of contamination and changes in measurement methods. As a result, comparison of predicted values with the measured data from Anderson and Smith[80] and Crockett and Kinnison,[81] would not appear to be scientifically valid.

The data of Dvonch et al.[33] and Keeler et al.[82] and Lamborg et al.[83] are collected in areas where there are several anthropogenic sources including a utility. The predicted air

Table 7-11. Summary Mercury Concentrations in Media and Biota

Summary Concentrations for Eastern Site					Concentrations in Biota (μg/g)[a,b]													
Plant	Mercury Air Concentration (ng/m3)	Mercury Total Deposition Rate (ug/m²/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg.	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Trophic level 3 Fish	Trophic level 4 Fish
2.5 km with RELMAP 50th Percentile																		
Large Coal-fired Utility Boiler	2.5e-02	1.1e+01	2.1e+01	6.9e-01	6.0e-04	4.4%	6.4e-04	16.7%	3.3e-04	8.3%	3.8e-04	8.9%	2.1e-05	2.1%	2.1e-07	10.1%	3.2e-02	1.6e-01
Medium Coal-fired Utility Boiler	2.6e-02	9.2e+00	1.8e+01	5.7e-01	6.0e-04	4.5%	6.3e-04	17.4%	3.0e-04	9.3%	3.5e-04	9.9%	1.8e-05	2.1%	1.9e-07	11.1%	2.7e-02	1.3e-01
Small Coal-fired Utility Boiler	2.7e-02	3.6e+00	7.0e+00	2.0e-01	5.6e-04	4.9%	5.7e-04	19.6%	1.9e-04	13.8%	2.3e-04	14.4%	7.2e-06	2.2%	1.4e-07	15.1%	9.2e-03	4.7e-02
Medium Oil-fired Utility Boiler	2.4e-02	2.7e+00	5.2e+00	1.3e-01	5.0e-04	4.9%	5.1e-04	19.9%	1.6e-04	14.8%	2.0e-04	15.3%	5.4e-06	2.3%	1.2e-07	15.8%	6.1e-03	3.1e-02
10 km with RELMAP 50th Percentile																		
Large Coal-fired Utility Boiler	2.6e-02	4.5e+00	8.7e+00	2.2e-01	5.4e-04	4.8%	5.6e-04	19.2%	2.0e-04	12.6%	2.4e-04	13.3%	8.9e-06	2.2%	1.4e-07	14.1%	1.0e-02	5.3e-02
Medium Coal-fired Utility Boiler	2.7e-02	3.8e+00	7.4e+00	1.9e-01	5.6e-04	4.8%	5.7e-04	19.5%	1.9e-04	13.6%	2.3e-04	14.2%	7.6e-06	2.2%	1.4e-07	15.0%	8.9e-03	4.5e-02
Small Coal-fired Utility Boiler	2.6e-02	2.7e+00	5.3e+00	1.3e-01	5.4e-04	4.9%	5.5e-04	20.0%	1.7e-04	15.0%	2.1e-04	15.6%	5.5e-06	2.3%	1.2e-07	16.0%	6.1e-03	3.1e-02
Medium Oil-fired Utility Boiler	2.4e-02	2.5e+00	4.9e+00	1.2e-01	5.0e-04	4.9%	5.1e-04	20.0%	1.6e-04	15.0%	1.9e-04	15.6%	5.1e-06	2.3%	1.1e-07	16.0%	5.6e-03	2.8e-02
25 km and RELMAP 50th Percentile																		
Large Coal-fired Utility Boiler	2.5e-02	2.9e+00	5.7e+00	1.4e-01	5.2e-04	4.9%	5.4e-04	19.9%	1.7e-04	14.6%	2.1e-04	15.1%	5.9e-06	2.3%	1.2e-07	15.7%	6.5e-03	3.3e-02
Medium Coal-fired Utility Boiler	2.6e-02	2.8e+00	5.4e+00	1.3e-01	5 4e-04	4.9%	5.5e-04	20.0%	1.7e-04	14.9%	2.1e-04	15.5%	5.6e-06	2.3%	1.3e-07	16.0%	6.2e-03	3.1e-02
Small Coal-fired Utility Boiler	2.5e-02	2.5e+00	4.9e+00	1.2e-01	5.1e-04	4.9%	5.2e-04	20.0%	1.6e-04	15.1%	2.0e-04	15.7%	5.1e-06	2.3%	1.2e-07	16.1%	5.6e-03	2.8e-02
Medium Oil-fired Utility Boiler	2.4e-02	2.5e+00	4.8e+00	1.2e-01	5.0e-04	4.9%	5.1e-04	20.0%	1.6e-04	15.1%	1.9e-04	15.6%	5.0e-06	2.3%	1.1e-07	16.1%	5.5e-03	2.8e-02

Summary Concentrations for Western Site					Concentrations in Biota (μg/g)[a,b]													
Plant	Mercury Air Concentration (ng/m3)	Mercury Total Deposition Rate (ug/m²/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg.	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Tier 3 Fish	Tier 4 Fish
2.5 km with RELMAP 50th Percentile																		
Large Coal-fired Utility Boiler	7.0e-03	3.4e+00	6.9e+00	2.4e-01	1.7e-04	4 3%	1.8e-04	16.2%	1.0e-04	7.7%	1.2e-04	8.3%	6.9e-06	2.1%	6.2e-08	9.5%	1.2e-02	6.1e-02
Medium Coal-fired Utility Boiler	7.6e-03	2.6e+00	5.1e+00	1.7e-01	1.7e-04	4.5%	1.8e-04	17.5%	8.6e-05	9.3%	1.0e-04	10.0%	5.2e-06	2.1%	5.5e-08	11.1%	8.8e-03	4.5e-02
Small Coal-fired Utility Boiler	7.3e-03	7.4e-01	1.5e+00	4.8e-02	1.5e-04	4.9%	1.5e-04	20.0%	4.8e-05	15.0%	5.8e-05	15.6%	1.5e-06	2.3%	3.5e-08	16.1%	2.4e-03	1.2e-02
Medium Oil-fired Utility Boiler	5.1e-03	3.8e-01	7.5e-01	2.1e-02	1.0e-04	5.0%	1.1e-04	20.4%	3.1e-05	16.4%	3.8e-05	16.8%	7.9e-07	2.4%	2.3e-08	17.0%	1.1e-03	5.3e-03
10 km with RELMAP 50th Percentile																		
Large Coal-fired Utility Boiler	6.8e-03	1.4e+00	2 8e+00	8.2e-02	1.5e-04	4.7%	1.5e-04	18.8%	5.9e-05	11.8%	7.0e-05	12.5%	2.8e-06	2.1%	4.0e-08	13.4%	4.2e-03	2.1e-02
Medium Coal-fired Utility Boiler	7.3e-03	9.2e-01	1.8e+00	5.5e-02	1.5e-04	4.9%	1.6e-04	19.7%	5.1e-05	14.1%	6.2e-05	14.7%	1.9e-06	2.2%	3.7e-08	15.3%	2.8e-03	1.4e-02
Small Coal-fired Utility Boiler	6.7e-03	4.4e-01	8.9e-01	2.4e-02	1.4e-04	5.0%	1.4e-04	20.5%	3.9e-05	16.7%	4.8e-05	17.1%	9.4e-07	2.4%	2.9e-08	17.3%	1.2e-03	6.2e-03
Medium Oil-fired Utility Boiler	5.1e-03	3.3e-01	6.6e-01	1.7e-02	1.0e-04	5.0%	1.1e-04	20.6%	3.0e-05	16.8%	3.7e-05	17.2%	7.0e-07	2.4%	2.2e-08	17.3%	8.8e-04	4.4e-03
25 km and RELMAP 50th Percentile																		
Large Coal-fired Utility Boiler	6.5e-03	6.7e-01	1.3e+00	3.7e-02	1.3e-04	4.9%	1.4e-04	20.0%	4.2e-05	15.0%	5.2e-05	15.5%	1.4e-06	2.3%	3.1e-08	16.0%	1.9e-03	9.5e-03
Medium Coal-fired Utility Boiler	7.0e-03	5.3e-01	1.1e+00	2.8e-02	1.4e-04	5.0%	1.4e-04	20.4%	4.2e-05	16.2%	5.2e-05	16.7%	1.1e-06	2.4%	3.1e-08	16.9%	1.5e-03	7.3e-03
Small Coal-fired Utility Boiler	6.0e-03	3.6e-01	7.3e-01	1.9e-02	1.2e-04	5.0%	1.2e-04	20.6%	3.4e-05	17.0%	4.2e-05	17.4%	7.7e-07	2.4%	2.6e-08	17.5%	9.6e-04	4.9e-03
Medium Oil-fired Utility Boiler	5.1e-03	3.2e-01	6.4e-01	1.6e-02	1.0e-04	5.0%	1.0e-04	20.6%	2.9e-05	16.9%	3.6e-05	17.3%	6.8e-07	2.4%	2.2e-08	17.4%	8.4e-04	4.2e-03

[a] Plant and animal product concentrations are ug/g dry weight; fish concentrations are ug/g fresh weight.

[b] Trophic level 3 fish are planktivorous fish such as gizzard shad and trophic level 4 fish are piscivorous fish such as pike.

concentrations around the utility are approximately 0.2 ng/m^3. (This predicted value includes the contribution from the 50th percentile RELMAP value.) Measured average air concentrations over South Florida and near anthropogenic sources are 2.8 and 3.3 ng/m^3 and average concentrations from sites in Detroit are 3.7 and 40 ng/m^3. The measured data were collected over relatively short periods of time and the measured mercury concentrations probably reflect the input from multiple local as well as regional sources. As a consequence, comparison to predicted data are inappropriate.

It must be stressed that COMPDEP predicts annual average results and that no systematic collection of measured mercury data around these sources was identified. The measurement data tend to be collected for short periods of time, and it is difficult in an urban environment to apportion the sources of mercury. It must also be emphasized that: (1) the COMPDEP is a Gausian Plume model, these models have been used for years; (2) the modifications made to the COMPDEP model pertaining to the chemistry of atmospheric mercury were based on "state of the art" assumptions about mercury (see Petersen et al.[34]); and (3) these models have not been validated, but then no other mercury fate and transport models have (at least to the EPA's knowledge) been validated at this time. The EPA is currently conducting research to refine some of the areas of greatest uncertainty. Model validation will take a much longer period of time and will require significant resources.

Hourly and daily predictions of COMPDEP were not used in this report. The EPA's confidence in these predictions is low. Confidence in the longer-term average predictions is greater.

It should also be noted that both the dry deposition of Hg(II) vapor and the chemistry of mercury emitted in a plume are poorly understood. The lack of measured data make it impossible to compare the predicted dry deposition estimates of sources expected to emit Hg(II). The COMPDEP results agree with those of the RELMAP model in that both models predict that the bulk of the emitted mercury will be transported beyond 50 km from the emission source.

Given the uncertainty in the COMPDEP results and the lack of measurement data collected around the local emissions sources, a comparison of the IEM2 models with measured data collected around sources was impossible.

The EPA was unable to locate systematically collected measured mercury data in soils, water, or biota around utility boilers. The lack of measured data precluded a comparison of predicted and measured values and, as a result, a validation of these models is not possible at this time.

7.5 PRELIMINARY OBSERVATIONS FROM LOCAL ANALYSIS

The Agency's confidence in the following observation is high:

1. There is a lack of reported data near the utilities considered in this report. The lack of such measured data precludes a comparison of the model results with measured data around these sources. These data include measured mercury deposition rates and concentrations in the atmosphere, soils, and biota (including fish).

The Agency's confidence in the following observations is moderate:

2. The modeling analysis of utility emissions in flat terrain indicates that relatively low mercury concentrations will result in media and biota in the local modeling domain (i.e., within 50 km of the facility). Depending on the facility, this is predicted due to a combination of the following factors: (1) the predicted plumes are high due to a high effective stack height, and (2) the assumed mercury emission rates are low. The high effective stack heights are a result of high stacks and large stack exit gas velocities associated with this source class. This high effective stack height results in the dispersion of mercury emissions more to the regional scale than to the local scale.

3. Based on the analysis, the larger impact from utility mercury emissions is the result of bioaccumulation of deposited mercury through the aquatic food chain. The magnitude of the increase from any individual utility as configured in this analysis (in particular, in flat terrain) is predicted to be low. There is a great deal of uncertainty and potential variability due to differences in mercury bioaccumulation in various bodies of water.

4. Higher total mercury air concentrations and deposition rates may result in elevated terrain. The magnitude of these impacts compared to flat terrain is uncertain.

5. From the analysis of deposition and on a comparative basis, a utility located in a humid climate has a higher annual rate of deposition than a utility located in an arid

climate. The critical variables are the solubility (estimated washout ratios) of Hg(II) and the annual amount of precipitation.

6. The lack of measurable impacts of the utility model plants is a direct result of the relatively tall stacks of utility boilers. The tall stacks result in impacts beyond the scale of the local-scale analysis (50 km).

7. Based on this analysis, when the results of the local impact analysis were combined with the 50th percentile estimates from the long-range transport analysis for all utilities across the U.S., the regional contribution was generally higher than the local contribution for air concentration and deposition, although there are exceptions.

7.6 GENERAL FINDINGS FOR MERCURY FROM UTILITIES

There are many sources of mercury emissions that occur around the world; these include both natural and anthropogenic sources. Fish mercury levels are probably due to mercury emissions from all of these sources over time. The utilities are one category of anthropogenic sources of mercury, and they are estimated to emit approximately 51 ton/yr of mercury emissions in the U.S., which is approximately 21 percent of the total anthropogenic emissions of mercury in the U.S.

Although the amount of mercury being emitted from any single source may be small, there are a number of reasons why these emissions are of potential concern. The first is that mercury is a persistent element; it is not degraded or removed, but continually accumulates. Consequently, over time, there is potential for levels of biologically available mercury to build up. Second, current scientific evidence demonstrates that most of the mercury emitted from a source, especially sources like utilities that have tall stacks, does not deposit within 50 km of the source but is deposited far away. As a result, even though the concentration of mercury around a single source may not be elevated, there are sufficient data to conclude that the cumulative impact of many small sources leads to accumulation of mercury in the biosphere and bioaccumulation in fish.

The multipathway modeling assessment, in conjunction with available scientific knowledge, supports a plausible link between mercury emissions from utilities and mercury concentrations in air, soil, water, and sediments. The mercury modeling assessment, in conjunction with available scientific knowledge, also supports a plausible link between mercury emissions from utilities and methylmercury concentrations in freshwater fish.

Other studies conducted by various researchers support the EPA's observation that there is a plausible link between utility mercury emissions and the methylmercury found in freshwater fish. For example, in the EPRI's report entitled *Electric Utility Trace Substances Synthesis Report*,[88] EPRI estimated the impacts of utility mercury emissions from two utility plants on methylmercury concentrations in freshwater fish.

The results of the EPRI modeling analysis indicate that, for two utility plant sites within 50 km of four lakes, the methylmercury concentrations in freshwater fish were slightly elevated. The EPRI report notes that these estimated mercury concentrations in fish are well below the multimedia hazard index and that the uncertainty analysis indicated that the estimates were highly conservative. However, the EPRI modeling analysis does support a credible link between mercury emissions from electric utility plants and some incremental increase in methylmercury concentrations in fish located in local waterbodies.

The EPA's multipathway modeling assessment of mercury emissions from four utility model plants suggests that mercury emissions from utilities may contribute, at least to a limited degree, to mercury concentrations in freshwater fish. Also, the modeling assessment suggests that emissions of mercury from all utilities may contribute to the overall mercury loadings to the environment. However, the EPA has not yet determined, at this time, whether the mercury contribution from utilities is a concern for public health.

7.7 DISCUSSION OF FEDERAL INTERAGENCY REVIEW COMMENTS

Previous drafts of chapters 1 through 10, along with the appendices, were reviewed by numerous non-EPA scientists representing industry, environmental groups, academia, and other Federal agencies during the summer of 1995. In February, April, and September 1996, all sections of the draft report underwent additional review by EPA, State and local agencies, and other Federal agencies. The EPA has revised the report, as appropriate, based on the reviewers' comments. The EPA revised the report to incorporate the majority of the comments received. However, there were several comments that could not be fully addressed because of limitations in data, methods, and resources. This section presents comments received by other Federal agencies that could not be substantially addressed in this interim report.

7.7.1 Comments

The Office of Management and Budget (OMB) commented that the April 1996 draft report included "...mercury-related material subject to continued interagency debate..." (e.g., discussion of the health effects of mercury, the level of "safe" exposure to mercury, the potential food safety issues associated with fish consumption). Discussion of this material should be deferred pending completion of further review of the mercury report.[89]

The National Marine Fisheries Service (NMFS) commented that the EPA should defer addressing mercury risks from utilities until completion of the Mercury Report.[90]

The Food and Drug Administration (FDA) commented that the EPA may want to consider issuing the Report without a discussion of human health effects of methylmercury. The FDA also suggested "...that the discussion of the RfD be rewritten..." and the uncertainties acknowledged.[91]

The Council of Economic Advisors (CEA) and the Office of Science and Technology Policy (OSTP) similarly commented that the current report should not refer to the current RfD as a "reasonable estimate" but should state that the current standard is under review.[92]

The NMFS encouraged the EPA "...not to release any documents that characterize [methylmercury] risks..." from exposure through fish consumption until release of the Mercury Report.[93]

The FDA commented that the EPA should "...acknowledge that typical consumers eating fish in moderation from a variety of sources and a variety of species are not believed to be at increased risk from methylmercury." [94]

The OSTP commented that:

> "...quantitative relationships between utility mercury emissions and fish methylmercury levels are unknown. This information is critical for estimating human mercury exposures and risks from utility emissions. Because mercury emitted from utilities can be transported long distances and total deposited anthropogenic emissions in the US are only 1/3 of the total US anthropogenic emissions, it is evident that the majority of US anthropogenic mercury emissions are transported outside of the US. Thus, it is not clear to what extent reductions in US emissions of anthropogenic mercury from utilities will affect methylmercury concentrations in fish. The purpose of this report is to

> determine if utility HAP emissions present a health hazard to humans. The report notes that the fate and transport model predicts that utilities may contribute about 18 percent of the total anthropogenic mercury deposited in the US. Would a 30 percent increase in mercury emissions, which is projected for the year 2010 from coal-fired utilities, lead to greater than a 5 percent increase in fish levels of methylmercury, and would a 50 percent reduction in mercury emissions produce more than a 10 percent reduction in total anthropogenic mercury emissions? More importantly, would either of these changes produce a significant adverse or beneficial health effect? Perhaps health-related issues of anthropogenic mercury emissions need to be examined on a more global scale." [95]

The Centers for Disease Control (CDC) commented "...that mercury is the [HAP] of greatest concern among the pollutants listed." Also, the CDC agreed with one of EPA's draft conclusions (presented in the April 1996 draft report) ..."that mercury emissions should be minimized as part of an overall strategy for reducing power-plant generated pollutants such as sulfur dioxide and particulate matter." [96]

The Department of Energy (DOE) and CEA suggested, as an option, that the report should be revised to remove all discussions of mercury, indicating that this material is being reviewed by the SAB and will be included in the final Mercury Study.[97]

7.7.2 Response

As discussed in the preface, the EPA has removed the assessments of human exposure, health effects, and risk discussions for mercury from this interim report. The EPA plans to consider the above comments and make appropriate changes, as appropriate, and to the extent feasible, before issuing a final report.

The EPA has decided not to include conclusions or policy statements regarding the public health impacts due to mercury emissions from utilities in the interim report because the Agency is awaiting new data from human epidemiology studies that will help address the potential public health impacts. Therefore, several statements that were presented in the April 1996 draft report have been removed from this interim report. The EPA plans to include a more complete assessment of exposures and risks and plans to include conclusions and policy statements, to the extent feasible and appropriate, in the final report.

7.7.3 Comment

The OMB commented that the April 1996 draft report "...does not place in proper context the contribution of anthropogenic U.S. air emissions to the total loading of mercury in the environment." The discussion "...should be expanded to provide a more comprehensive description of the sources of mercury contamination...", including other air deposition sources (e.g., global and naturally occurring sources) and non-air sources (e.g., industrial and municipal water discharges; the release of mercury from sediments).[98]

7.7.4 Response

The EPA has revised the discussion of mercury emission sources to include and emphasize that utility emissions, and anthropogenic emissions as a whole, represent a fraction of total mercury in the environment. However, global emissions were not quantified because of the significant uncertainties associated with estimates of global emissions.

7.7.5 Comment

The FDA commented that "[s]ince methylmercury from marine fish is not considered in the exposure modeling assessment, there is no point in referring to or including Table 7-2 (Measured Mercury Concentrations in Saltwater Commercial Fish) in this discussion. Further, inclusion of information on shark, marlin, etc., is also irrelevant, especially since some of the information on mercury concentrations is inaccurate." [99] The OSTP commented that Table 7-2 is "intended for the mercury study report, should not be in this report." The NMFS submitted the comment: "We recommend that references to marine fish (text in Section 7.2, page 7-10 and Table 7-2) be deleted. Their inclusion is not pertinent to the discussion of mercury concentrations in biota potentially affected by utility emissions as constrained by the modeling techniques used in the hazardous air pollutant study. The last sentence of the footnote to the table is totally irrelevant in the absence of sample sizes, fish lengths (ages), and mercury ranges." The NMFS continued in the comment letter that "If Table 7-2 is retained..." then the EPA should correct the inaccuracies in the table.

7.7.6 Response

The EPA added caveats and discussions of ranges and uncertainty and discussed the relevance of this information. In addition, the EPA revised the table substantially by presenting ranges and explanations of data and by correcting the inaccuracies. The data are being presented to provide general information on estimated levels of mercury in the environment. This information is useful for understanding where mercury is found in the environment and the relative magnitudes of mercury in each of the media or biota. In addition, these data aid in the understanding of environmental fate and transport of mercury, including the bioaccumulation potential of mercury in various species.

The modeling assessment did not predict levels in saltwater fish. However, the EPA believes that providing data on measured levels in saltwater fish (Table 7-2) is useful since it is one of the biota that contain mercury. The EPA has not determined if, or how much, mercury emissions from U.S. utilities contribute to these saltwater fish levels. Such an analysis was not possible for this report because of the limitations in the available data, methods, and models and because of the immense complexity of such an assessment.

7.7.7 Comments

The DOE commented that "[t]he model used in this Report for regional deposition estimation of all pollutants (RELMAP) as well as the model used for estimation of local mercury deposition were both significantly criticized during review of the Mercury Report and are scheduled for review by the EPA's Science Advisory Board." In addition, the DOE adds that "[a]ll sections now using [the RELMAP model] should contain prominent caveats..." because the model is "...believed to significantly overestimate deposition and hence risk of all types..." [100]

The OMB commented that the report "...should clearly state the strengths, limitations, and uncertainties associated with modeling the predicted levels of exposure and indicate the effect of these factors on the overall assessment. The Report should also indicate the extent to which the predicted exposure levels have been validated by empirical data." [101]

The OSTP similarly commented that a "...lack of monitoring data that reflect or validate predictions of mercury exposures based on the fate and transport model..." is a major shortcoming of the report.[102]

7.7.8 Response

The EPA has added discussions of uncertainty and appropriate caveats to various sections (e.g., section 7.10 and others) of the report regarding the exposure models. The models are considered useful for estimating fate and transport of pollutants. The EPA believes that the results of the modeling assessment are reasonable. However, the EPA does recognize there are limitations and uncertainties. The text has been revised in various locations to reflect these limitations and uncertainties.

7.7.9 Comment

With regard to the multipathway modeling assessment for mercury, the FDA commented that the EPA's confidence in the local impact analysis must be considered low, rather than high, based on the stated uncertainties.[103]

7.7.10 Response

Because of scientific uncertainties and variability and a lack of measured mercury data around utility boilers, the EPA interpreted the results of the modeling in a qualitative rather than quantitative manner. The EPA's confidence in the general trends of the quantitative data is greater than the confidence in the actual values predicted. This is reflected in the qualitative nature of the conclusions. As explained in section 7.5, the EPA's confidence in the observations of the local analysis are generally moderate rather than high.

7.7.11 Comments

The FDA commented that "...the existence of fish consumption advisories in 35 States based on conservative methodology does not itself evidence a national mercury fish contamination problem." Rather, it could relate "...more to risk assessment methodologies than to mercury 'contamination' of fish..." and merely be a reflection of the uncertainty surrounding determining a "safe" exposure level and risk assessment methodologies than to actual contamination of fish. The FDA stated that "[t]he discussion of fish advisories and mercury levels in fish should be deleted because it will mislead the reader on the extent of mercury contamination of the environment." [104]

7.7.12 Response

The EPA revised the text in section 7.1 to say that "...there is a potential concern for mercury contamination in the U.S. as indicated by numerous fish advisories..." Also, much of the discussion of fish advisories has been removed from this interim report.

7.7.13 Comment

The DOE commented that the "EPA uses a series of fate and transport models and hypothetical environmental constructs to assess the emission of mercury from utility boilers. EPA extends a level of moderate confidence to a conclusion of its modeling effort that (1) 50 percent of utility mercury is emitted as Hg^0; (2) 30 percent is emitted as Hg(II) (i.e., divalent or ionic form); and (3) 20 percent is emitted as a particulate. This is a very critical conclusion in that, with a high percentage of Hg^0, the model predicts relatively low local (within 50 km) deposition and high long-range deposition due to the fact that Hg^0 readily becomes part of the global mercury cycle."[105] The DOE noted that "...recent bench-, pilot-scale, and field characterization studies...have indicated that coal-fired flue gas can contain high percentages of ionic mercury...studies also show that far less than 20 percent of the mercury is in the form of a particulate (typically less than 5 percent)." It was indicated that the EPA needs to reconcile the discrepancy noted between the various mercury speciation values used in the modeling and those presented by the DOE.

The DOE also commented that "EPA expresses moderate confidence in the model's prediction that local (within 50 km of the source) deposition of, and exposure to, mercury will be low...based on the assumption that most of the utility flue gas mercury is elemental, which remains airborne and is transported greater distances than Hg(II). EPA takes this one step further in concluding that utility mercury impacts are more regional than local in nature." It was noted that the EPA should revisit these conclusions in light of the mercury speciation data provided by the DOE.

7.7.14 Response

The EPA plans to consider these comments and, if feasible and appropriate, conduct additional analyses of the data in the future.

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8.0 QUALITATIVE MULTIPATHWAY ASSESSMENT FOR ARSENIC, DIOXINS, LEAD, AND CADMIUM

8.1 BACKGROUND

A multipathway exposure analysis, an assessment of multiple routes of exposure of humans and/or biota to pollutants, is the appropriate approach for a complete human health risk assessment. Though it would have been preferable to perform a quantitative multipathway exposure assessment of utility emissions of all six HAPs of interest (radionuclides, mercury, arsenic, dioxins, cadmium, and lead), at the time of this study not enough data had been collected or were available to do so. Quantitative multipathway assessments were performed for only one class of HAPs (radionuclides). A general assessment of the fate and transport of mercury is presented in chapter 7. For the other four HAPs - arsenic, dioxins, lead, and cadmium, only qualitative assessments of the potential concerns to human health from non-inhalation exposure were performed.

The completion of quantitative assessments of inhalation exposure risks for all HAPs and of multipathway exposures for only one class of HAPs (radionuclides) is not a reflection of the significance of noninhalation exposure and risks from utility HAP emissions. Rather, it is a reflection of the complexity of assessing multipathway exposure to a HAP. Due to the intensive data requirements of a quantitative multipathway exposure assessment of a HAP, and due to the limited chemical-specific data available (e.g., chemical-specific air to plant biotransfer factor, chemical-specific plant uptake rates) for input into the exposure model, quantitative analyses were not completed for arsenic, dioxins, lead, or cadmium. Though it is important to address the noninhalation exposure pathways for select HAPs, there are complex issues associated with the analysis of all exposure pathways.

The EPA recognizes that, for some of the utility HAPs (e.g., mercury, dioxins), noninhalation exposure is, potentially, a more significant route of exposure than inhalation exposure. The mercury assessment suggests there is a need for further analysis of noninhalation exposures.

Efforts are underway to collect chemical-specific data needed for quantitative multipathway exposure assessment, and further analyses may be undertaken for additional HAPs in the future, in recognition of the need to understand the contribution of air pollutants to risk from noninhalation exposure pathways. For this report, a qualitative discussion of the potential concerns of noninhalation exposure to arsenic, dioxins, lead, and cadmium is presented in the following sections.

8.2 ARSENIC COMPOUNDS

Arsenic is highly persistent in the environment, has a tendency to bioaccumulate, and is toxic from both the inhalation and oral exposure routes. Arsenic is also a known human carcinogen.

For 1990, the estimated arsenic emissions from all coal-, oil-, and gas-fired units were 54 ton/yr, 11 ton/yr, and 0.16 ton/yr, respectively. To put these estimates into perspective, if the total amount of arsenic emitted from these units in 1990 were added to those reported in the toxic release inventory (TRI) database for 1 year (data from 1988) (i.e., 270,000 pounds[1]), the TRI estimate would be increased by 50 percent.[a]

Because arsenic is a naturally occurring compound, it has been found in low levels in all media, including air, soil, water, sediment, fish and shellfish, and other food products.[1] Arsenic is released from anthropogenic activities (e.g., metal smelting, chemical production and use, coal combustion, and waste disposal), and these emissions of arsenic can contribute significantly to environmental contamination.[1]

Once released from stack sources, such as utilities, arsenic can be deposited to various environmental media and undergo complex transformations. In the atmosphere, arsenic exists as PM in the form of trivalent and pentavalent states.[1] Studies indicate that the trivalent state is more toxic to the biological system than the pentavalent species.[2] A typical residence time of particulate arsenic is 9 days. The primary removal mechanism from the atmosphere is wet or dry deposition to soil, water, and plants.[1] Once deposited onto soil, arsenic tends to sorb to soils.[1] However, precipitation may leach soluble forms of the compound into surface waters and ground water. The predominant species of arsenic in soils varies, with pentavalent arsenic

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[a] The emissions reported in the TRI database represent atmospheric releases of arsenic from all facilities included in the Standard Industrial Classification (SIC) 20-39 that manufactured, processed, or otherwise used arsenic above the established threshold. As a consequence, the TRI emissions estimate does not include emissions from the electric utility industry (i.e., SIC 49). Furthermore, the ATSDR *Toxicological Profile for Arsenic*[1] points out that the TRI data do not include emissions data from coal combustion facilities and pesticide spraying operations, two major sources of arsenic.

dominating in aerobic soils, trivalent arsenic dominating in slightly reduced soils, and arsine, methylated arsenic, and elemental arsenic dominating in extremely reduced conditions.[1] Terrestrial plants accumulate arsenic by root uptake from soil or by absorption of airborne arsenic deposition onto leaves.[1] In the aquatic environment, arsenic will partition to sediments or will remain dissolved in the water. In water, arsenic can undergo complex transformations, but the predominant form of arsenic is usually pentavalent arsenic, arsenate.[1] The pH of the water and the redox potential of the aquatic system greatly affect the chemical forms of the inorganic arsenic species.[2]

The highest levels of arsenic are found in seafoods, meats, and grains. U.S. dietary levels of arsenic in these food products typically range from 0.02 ppm in grains and cereals to 0.14 ppm in meat, fish, and poultry.[1] Fish and shellfish are typically associated with the highest levels of arsenic concentration, although, arsenic accumulates in fish and shellfish in a form thought to be relatively nontoxic. Background levels of arsenic reported for milk are 0.5 to 70 $\mu g/kg$[3]. With the exception of significantly contaminated areas, exposures occurring through the intake of air, soil, and water are usually lower than those occurring through food products. Background levels of arsenic for these media have been reported as follows: soil is 5,000 ppb, water 2 ppb, and air 0.02 to 0.10 $\mu g/m^3$. Furthermore, surveys of drinking water in the U.S. indicate that greater than 99 percent of the drinking water supplies have concentrations below the EPA maximum contaminant level (MCL) of 50 ppb.[1] The ATSDR[1] states that, for the general population, the highest levels of exposure to arsenic occur through their diet, with an average intake of approximately 50 $\mu g/d$. Based on an EPA reference dose of 3×10^{-4} mg/kg/d, this intake value corresponds to a hazard quotient of slightly less than 1.[b]

A study that examines the transfer of metals to bovine milk indicates human exposure to arsenic through consumption of milk may be of concern. Because the contribution to human exposure through the food chain has not been thoroughly examined, a study[3] was undertaken to estimate the steady-state bovine milk biotransfer factors (i.e., the rate at which the compounds are transferred to milk) for six metals: arsenic, cadmium, chromium, lead, mercury, and nickel. Results from this study indicate

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[b] This risk estimate was calculated based on the following exposure assumptions: exposure duration of 30 years, exposure frequency of 350 d/y,; body weight of 70 kg, and an average life expectancy of 70 years.

that, of the metals studied, lead and arsenic transfer to milk to the greatest extent.[3] The BCF estimated for inorganic arsenic is 3.0×10^{-5} d/L. To put this value into perspective, Stevens points out the estimated biotransfer factor of tetra chlorodibenzo-p-dioxin (TCDD) is 2.6×10^{-2} d/L and that the particular food chain pathway may be less important for these metals than it is for TCDD.

A study conducted on children in a community located less than 1 mile from a copper smelter indicated that exposure to arsenic through the ingestion of soil may be of concern. Specifically, Polissar et al. conducted a study that attempts to establish a link between direct soil ingestion and arsenic exposure in a community located near a copper smelter (i.e., median distance from the smelter is 0.5 miles). The contamination in the community was predominantly in the form of arsenic trioxide. Analyses of samples obtained from 17 children, ages 6 or under, revealed elevated levels of urinary arsenic (i.e., a relatively high median urinary arsenic concentration of 43.6 ppb). The children of the community appeared to be receiving their arsenic dose from hand-to-mouth activities (i.e., soil ingestion). Calculations showed that inhalation could have contributed only a small fraction of the arsenic, and dermal absorption could be ruled out on the basis of poor dermal absorption of the inorganic arsenic in particulate or bulk soil. Thus, the remaining exposure route is ingestion. Also, arsenic concentrations in homegrown fruits and vegetables in the study area were low, and the arsenic concentrations in drinking water were far below the EPA standard of 50 ppb.

Furthermore, a study conducted by Chen et al. on ground water samples obtained from southwest Taiwan indicated that arsenic consumption through drinking water may be of concern if arsenic is present in the trivalent state.[c] As part of this study, ground water samples obtained from the Blackfoot Disease (BFD) area of Southwest Taiwan and well water samples from a city where no BFD has ever been reported were collected and analyzed. Analyses of these samples revealed that the total dissolved arsenic levels detected in the BFD area samples were approximately 1,000 times higher than in the samples obtained from the control site. In the samples obtained from the BFD area, the ratio of the trivalent species to the less toxic pentavalent species was approximately 3 to 1.[2]

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[c] This study states that Blackfoot Disease (BFD) is a peripheral vascular disease found in a limited area of Taiwan. The cause of this disease is still unknown, but it is generally attributed to the high concentrations of trivalent arsenic found in deep well waters.

Volume 1 of the EPRI's document *Electric Utility Trace Substances Synthesis Report* (Synthesis Report) presents results of a multimedia risk assessment of utility emissions. The assessment was performed by EPRI, using their "Total Risk of Utility Emissions" model, for HAPs emitted by four selected utilities. For arsenic, results of the assessment indicated that the dominant pathway or exposure to arsenic emissions can be ingestion (perhaps as much as 90 percent of total exposure). Further discussion of the assessment is provided in Volumes 1 and 2 of the EPRI Report.[4]

Because utility emission sources have the potential to contribute significantly to the total amount of arsenic emitted annually in the U.S. and because arsenic compounds tend to accumulate in the environment, the risks posed by noninhalation exposures to arsenic compounds in locally and regionally impacted areas may be of concern. Based on the studies discussed above and the physical and chemical properties of arsenic, it can be hypothesized that the primary exposure routes of concern for adults are those that are related to the ingestion of food products and that, for children, the primary route of exposure is ingestion of soil. Exposure through these routes is most likely to occur through the consumption of vegetation or soil contaminated by atmospheric deposition or the consumption of animals or fish contaminated through the ingestion of contaminated media or organisms.

Since inhalation risks are estimated to be above 1×10^{-6} for arsenic emissions from four utilities, since utilities emit approximately 59 ton/yr of arsenic nationwide, and since ingestion of arsenic can pose a cancer risk, the Agency believes that further evaluation of multipathway exposure is necessary to fully understand the risks posed by arsenic emissions from utilities.

8.3 DIOXIN AND DIOXIN-LIKE COMPOUNDS

Polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans, which will be referred to collectively as dioxins, are ubiquitous in the environment.[5] The dioxin and furan compounds chlorinated in the 2,3,7,8 positions on the molecule are highly toxic, environmentally persistent, and have a tendency to bioaccumulate. Exposure to dioxins is a potential concern for both cancer and noncancer effects, even at extremely low levels. The EPA has concluded that 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) and related compounds are probable human carcinogens.[5] In addition, EPA has concluded that there is adequate evidence to support the inference that humans are likely to respond with a broad spectrum of noncancer effects from exposure to dioxins, if exposures are high enough.[5]

It has been hypothesized that the primary mechanism by which dioxins enter the terrestrial food chain is through atmospheric deposition.[5] Therefore, a qualitative assessment of the potential risk to human health attributable to noninhalation exposure to dioxins emitted from utilities is presented here. In general, this assessment focuses on chlorinated dibenzodioxins (CDDs) and chlorinated dibenzofurans (CDFs) as a group and not on individual congeners. A multipathway exposure analysis was not performed for dioxin emissions from utilities. Therefore, data primarily from the external review draft dioxin reassessment report are used here for a qualitative assessment of potential concerns from multimedia exposures to dioxin. The primary sources used for this assessment are the external review draft dioxin reassessment documents (*Health Assessment Document for 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and Related Compounds*[5] and *Estimating Exposure to Dioxin-Like Compounds.*[6,7]) The Final dioxin report is expected to be released by the summer of 1997.

The occurrence of dioxin and dioxin-like compounds in the environment appears to be primarily the result of human activities.[5,6,7] The national estimated loading of these compounds from identified sources into the environment is approximately 12,000 g toxicity equivalents (TEQ)/yr. The draft dioxin reassessment report estimated an average of 9,200 g TEQ/yr from known combustion sources. Based on this study's estimate of 150 g TEQ/yr, dioxin emissions from utilities represent about 1.3 percent of total dioxin emissions.

When this loading of 12,000 g TEQ/yr is compared to annual loadings of other HAPs, it appears to be relatively low. However, dioxin emissions are of concern because these compounds are extremely toxic to humans and wildlife, are persistent in the environment and tend to bioaccumulate.[d]

The CDDs and CDFs have been found throughout the world in all media, including air, soil, water, sediment, and in biota, including fish and shellfish, and other plants and animals used as food products. The ubiquitous nature of these compounds can

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[d] The draft *Health Assessment Document for 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD)*[5] *and Related Compounds* (Volume III of III) states that these compounds are extremely potent in producing a variety of effects in experimental animals based on traditional toxicology studies at levels hundreds or thousands of times lower than most chemicals of environmental interest.

be attributed to their stability under most environmental conditions and also to the great number of sources located throughout the U.S. Once emitted into the atmosphere, the primary removal mechanisms are degradation and deposition to environmental media, such as soil, water, and vegetation. In general, the compounds have very low water solubility and vapor pressures and high octanol/water partition coefficients (K_{ow}) and organic carbon partition coefficients (K_{oc}). These measures of chemical properties indicate that dioxins tend to strongly adsorb to soils and, as a result, resist leaching or volatilization. The fate of dioxins adsorbed to particles includes burial in place, resuspension into the air, and erosion of soils into waterbodies. In the aquatic environment, CDDs and CDFs remain adsorbed to PM, and those compounds that enter surface waters and dissolve will tend to partition to suspended solids or dissolved organic matter. The primary removal mechanism from the water column is through sedimentation and, ultimately, burial of sediments.

Once CDDs and CDFs are deposited and make their way into various media and into biota, they are available for human uptake through ingestion. Consumption of dioxin-contaminated food is considered the primary route of exposure in the general population. Table 8-1 presents background media concentration data and TEQ daily intake rates associated with each medium. The EPA estimates that the highest daily TEQ intakes for humans occur through the consumption of animal fats, particularly fat of beef and veal, dairy products, milk, pork, and chicken. The EPA estimates that the lowest daily TEQ intakes occur through water ingestion, soil ingestion, and inhalation. Based on the data in Table 8-1, exposure occurring through ingestion of beef and veal is estimated to be approximately 16 times higher than exposure through inhalation.[e]

Volume III of the draft Dioxin Reassessment report[5] states that, with regard to average intake, humans are currently exposed to background levels of dioxin-like compounds, including dioxin-like polychlorinated biphenyls (PCBs), on the order of 3 to 6 pg TEQ/kg body weight/d. This value is more than 500-fold higher than the EPA's 1985 risk-specific dose of 0.006 pg TEQ/kg body weight/d associated with an upper-bound risk of 1 in a million (1×10^{-6}) and is several hundredfold higher than the revised risk-specific dose estimates presented in the draft Dioxin Reassessment report.[5]

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[e] It should be noted that the dioxin reassessment documents state that only a small number of samples were available for analysis, particularly for food, and this fact should be considered when evaluating the data.

Table 8-1. Estimated TEQ Background Dioxin Exposures in the United States[8]

	North America				
Media	Conc. TEQ	Contact rate	Daily intake[a] (mg/day)	Daily intake (pg/day)	Percent of total
Soil Ingestion	8.0 ppt	100 mg/day	8.0 x 10-10	0.8	0.7
Fish Ingestion	1.2 ppt	6.5 g/day	7.8 x 10-9	7.8	6.6
Inhalation	0.095 pg/m3	23 m3/day	2.2 x 10-9	2.2	1.8
Water Ingestion	0.0056 ppq	1.4 L/day	7.8 x 10-12	0.008	0.01
Milk Ingestion	0.07 ppt	251 g/day	1.8 x 10-8	17.6	14.8
Dairy Ingestion	0.36 ppt	67 g/day	2.4 x 10-8	24.1	20.3
Egg Ingestion	0.14 ppt	29 g/day	4.1 x 10-9	4.1	3.4
Beef and Veal Ingestion	0.48 ppt	77 g/day	3.7 x 10-8	37.0	31.2
Pork Ingestion	0.26 ppt	47 g/day	1.2 x 10-8	12.2	10.3
Chicken Ingestion	0.19 ppt	68 g/day	1.3 x 10-8	12.9	10.9
Total			1.08 x 10-7	119	100

[a] Daily Intake = contact rate x conc. teq x unit conversion (soil unit conv. = 10^{-12}, all other media unit conv. = 10^{-9}).

As part of the draft dioxin reassessment, methodologies for conducting site-specific indirect exposure modeling are presented. These methodologies are intended for use in evaluating incremental exposures (i.e., not background exposures) associated with specific sources of dioxin-like compounds. In the dioxin reassessment documents, example scenarios were developed and modeling was conducted for six exposure scenarios to demonstrate these methodologies. Of the six scenarios, scenarios 4 and 5 focused on indirect exposure resulting from stack emissions from stationary combustion sources. Of the emission sources modeled, scenarios 4 and 5 best represent exposures that could possibly be roughly similar to exposures resulting from the utility emissions. However, as stated in the draft dioxin reassessment documents, in evaluating the results from this modeling effort, it should be noted that the demonstration scenarios were developed only to illustrate the site-specific methodologies and that the exposure estimates generated for each scenario are not generalizable to other sites.

The results from this indirect exposure modeling effort presented in the draft dioxin reassessment report indicate that exposure levels due to the consumption of fish obtained from an impacted stream dominated the results generated for scenario 4. Under scenario 5, the highest levels of exposure occurred through the ingestion of fish and homegrown beef and milk. In summary, results for the two scenarios indicate that consumption of fish and consumption of beef and milk can be significant routes of exposure of humans to dioxin and dioxin-like compounds.

The transfer of dioxins from air to plants plays a major role in the exposure of terrestrial animals to dioxins. A finding in the draft dioxin reassessment report is that the principal cause for terrestrial food chain contamination is the transfer of dioxin-like compounds from the air to vegetation that animals consume. It has been found that dioxins in the vapor phase can transfer readily to plants and that this is the primary pathway by which plants take up dioxins.[6] This is of significance since the uptake of plants by foraging animals, including cows, is relevant to the concentrations of dioxins to which humans are exposed through the ingestion of beef and other animal products.

In addition to exposure pathways discussed above, exposure through the consumption of breast milk appears to be of potential concern for nursing infants. A study of 42 nursing mothers revealed that an average concentration of 16 ppt of TEQ was found in the lipid portion of their breast milk. A similar study conducted in Germany revealed an average level of 29 ppt of TEQ in the lipid portion of the breast milk. Based on the estimated adult intake of dioxin discussed above, an exposure duration of 1 year (i.e., the infant nurses for 1 year), an average weight gain of 10 kg during the exposure period, and a milk concentration of 20 ppt of TEQ, it is estimated that the average daily dose to the infant over this period is approximately 60 pg of TEQ/kg-d, 20 to 60 times higher than the estimated range for background exposure to adults (1 to 3 pg of TEQ/kg-d).

Because dioxins tend to accumulate in the environment and because they are extremely toxic to humans and wildlife, even small amounts of these compounds emitted from specific sources like utilities may be of concern. Based on background media concentration data and other assessments conducted by the Agency, including the one discussed above, it can be hypothesized that the primary human exposure routes of concern are those that are related to the ingestion of food products (e.g., meat and dairy products). Animal exposure through these routes is most likely to occur through the consumption of animals or fish contaminated through the ingestion of contaminated media or organisms or through the consumption of vegetation contaminated by atmospheric deposition. Since these exposure pathways are likely to be the most significant, they should be considered when assessing the risk from exposure to dioxins. Though some emissions, health effects, and exposure data are available, at this time it is unclear whether utility emissions of dioxins pose a significant risk. Further evaluation may be needed to more comprehensively evaluate the risks posed by emissions of dioxins from utilities.

8.4 LEAD COMPOUNDS

Lead compounds are persistent in the environment and have the tendency to bioaccumulate (see Table 5-8). Lead is known to be toxic by both oral and inhalation route. For these reasons, lead emissions from utilities are a potential concern from noninhalation exposure.

For 1990, the estimated lead emissions from all coal-, oil-, and gas-fired units were 72 ton/yr, 10 ton/yr, and 0.44 ton/yr, respectively. To put these emissions estimates into perspective, it was estimated that a total of 7.2×10^3 metric tons of lead were emitted into the atmosphere from anthropogenic point and nonpoint sources during 1989.[9] When this estimate is compared to the 1990 lead emissions estimate for utilities, it appears that utilities are responsible for emitting approximately 1 percent of the total amount of lead emitted annually. In 1989, the major contributors of atmospheric lead included industrial processes (2.3×10^3 metric tons), solid waste management (2.3×10^3 metric tons), transportation (2.2×10^3 metric tons), and fuel combustion (0.5×10^3 metric tons).[f]

Lead is a naturally occurring metal that can be found in small amounts in the earth's crust. However, the primary source of lead found in the environment is anthropogenic activities. Once emitted to the atmosphere from stack sources, such as utilities, lead can be deposited onto environmental media such as soil, water, and vegetation. In the atmosphere, lead exists primarily as PM in the form of lead sulfate ($PbSO_4$) and lead carbonate ($PbCO_3$). However, it is not clear how the chemical composition of lead changes during dispersion. Because a typical residence time of particulate lead is 10 days, long-range transport up to thousands of miles can occur. For example, lead has been found in sediment cores of lakes in Canada that were not located near any point sources of lead, indicating that long-range atmospheric transport may have been occurring. The primary removal mechanism from the atmosphere is wet or dry deposition onto soil, water, and plants. Atmospheric deposition is the primary source of lead found in soils.[10] Evidence supports that lead enters soil as lead sulfate or quickly converts to lead sulfate at the soil surface. Once deposited onto soil, lead tends to sorb strongly to soils and becomes extremely persistent. Because lead is strongly sorbed to soils, leaching of significant amounts of lead to ground water or surface water is not likely to occur. With the exception of highly acidic environmental conditions, leaching of lead into ground water and surface water occurs very slowly. The major contributors to surface water lead are atmospheric deposition and urban runoff. Typically in the aquatic environment, lead is associated with suspended solids and sediments. The concentration of dissolved lead in water is low because lead tends to form compounds that have low water solubilities and will precipitate out of the water column. The ratio of suspended lead to dissolved lead is found to vary from 4:1 in rural streams to 27:1 in urban streams. Many terrestrial plants accumulate lead by root uptake from soil or by absorption of airborne lead deposited onto plants. However, the bioavailability of lead to plants from soil is limited due to the strong absorption of lead to soil.[9]

The highest background levels of lead are found in soils (<10 to 30 μg/g) and in sediments (i.e., the average concentration of lead in river sediments is 20,000 μg/g).[10] In 1988, the average ambient air concentration for 139 sites

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[f] Industrial processes include nonferrous smelters, battery plants, and chemical plants.

monitored by the National Air Surveillance Network (NASN) was determined to be 0.085 $\mu g/m^3$.[9] This value is well below the NAAQS of 1.5 $\mu g/m^3$.[10]

Concentrations of lead found in foods are given in Table 8-2. These concentrations range from a low of about 0.002 $\mu g/g$ of food to a high of more than 0.8 $\mu g/g$ (found in milk). Background levels of lead in milk can range from 23 to 79 $\mu g/kg$.[3] The ATSDR[9] states that, for the general population, the highest levels of exposure to lead are most likely to occur through the ingestion of contaminated food and drinking water and by the inhalation of lead particles in ambient air. Furthermore, fruits, vegetables, and grains may contain levels of lead in excess of background levels as a result of deposition of lead onto plant surfaces and by plant uptake from soil. As seen in Table 8-2, the average adult dietary intake of lead for the years 1980-82 was estimated to be 56.5 μg/day.[9,11] However, recent data (1992) indicate that average dietary intakes have reduced significantly over the past decade to approximately 2 to 4 μg/day.[12] In general, human exposure to lead is most likely to occur through the ingestion of contaminated food and drinking water and by inhalation of lead particulates emitted from an emission source or reentrainment of lead-contaminated soil. Lead emissions from utilities do not contribute substantially to the total amount of lead released annually for anthropogenic activities (i.e., approximately 1 percent). However, lead emissions from utilities were not modeled for noninhalation exposures. Therefore, it is unclear whether the impact of the lead emissions from utilities is significant.

8.5 CADMIUM COMPOUNDS

As shown in Table 5-8, cadmium is persistent in the environment and has the potential to bioaccumulate. Health effects data indicate that cadmium is relatively toxic by both oral and inhalation routes. Cadmium is a probable (B1) human carcinogen by the inhalation route and is relatively potent (IURE = 1.8×10^{-3} per $\mu g/m^3$). However, there are insufficient data to assess the carcinogenicity from oral exposure. Regarding noncancer effects, cadmium exposure has been linked to kidney effects, primarily proteinuria. The RfD for cadmium is 5×10^{-4} mg/kg/d, and is associated with a high confidence level since it is based on human data. For these reasons, cadmium emissions from anthropogenic sources have the potential to be a concern for noninhalation exposure.

For 1990, the estimated cadmium emissions from all coal-, oil-, and gas-fired units were 1.9 ton/yr, 3.5 ton/yr, and

Table 8-2. Concentration of Lead in Various Food Products[9,10]

Food group	Concentration (μg/g)
Dairy Products	0.003-0.83
Meat, Fish, and Poultry	0.002-0.159
Grain and Cereal Products	0.002-0.136
Vegetables	0.005-0.649
Fruit and Fruit Juices	0.005-0.223
Oils, Fats, and Shortenings	0.002-0.028
Sugar and Adjuncts	0.006-0.073
Beverages	0.002-0.041

0.086 ton/yr, respectively. To put these estimates into perspective, it was estimated that a total of 700 ton/yr of cadmium were emitted into the atmosphere from anthropogenic point and nonpoint sources in the early 1980s.[13] In the early 1980s, the major contributors of atmospheric cadmium included fossil fuel combustion, smelting operations, manufacturing plants, and incinerators. The total amount of atmospheric releases of cadmium reported in the 1988 TRI database was approximately 60 tons.[13] [g]

Cadmium is a naturally occurring metal found in small amounts throughout the earth's crust. However, the primary source of cadmium found in the environment is anthropogenic activities. Once released from stack sources, such as utilities, cadmium can be deposited to various environmental media. Cadmium emitted from combustion processes typically exists in the atmosphere as small PM. Cadmium oxide is the predominant form of cadmium in the atmosphere. However, the toxicology of cadmium appears not to be dependent on the chemical's form. Because a typical residence time of particulate cadmium is between 1 and 10 days, long-range transport, up to thousands of kilometers, can occur. The primary removal mechanism from the atmosphere is wet or dry deposition onto soil, water, and plants. Atmospheric

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[g] The emissions reported in the TRI database represent atmospheric releases of arsenic from all facilities included in SIC 20 to 39 that manufactured, processed, or otherwise used cadmium above the established threshold. As a consequence, the TRI emissions estimate does not include emissions from the utility industry (i.e., SIC 49). Furthermore, the ATSDR *Toxicological Profile For Cadmium*[13] points out that the TRI data do not include emissions data from the combustion of fossil fuel and incineration of municipal or industrial wastes, two major sources of cadmium.

deposition can contribute significantly to the concentration of cadmium in soil in areas surrounding emission sources, such as incinerators and areas of heavy vehicular traffic. Once deposited onto soil, cadmium can be leached into water, especially under acidic conditions. In the aquatic environment, cadmium exists primarily as a soluble hydrated ion and, as a result, is more mobile than other heavy metals, such as lead. However, under certain environmental conditions, cadmium concentrations have been found to be at least 1 order of magnitude higher in sediment than in the water column. Accumulation of cadmium in terrestrial and aquatic plants can occur by root uptake from soil. In addition, terrestrial plants can be contaminated by absorption of airborne cadmium deposited onto plants. Accumulation of cadmium in feed crops has the potential to result in high levels of cadmium in the liver and kidneys of animals that eat the contaminated feed. Data indicate cadmium bioaccumulates in all levels of the food chain.[13] Table 8-3 presents concentrations of cadmium in various foods.

The highest background levels of cadmium are found in soils.[13] The mean concentration of cadmium in uncontaminated soil is 0.25 ppm. Topsoil concentrations can be as much as two times higher than subsurface concentrations due to atmospheric exposition and contamination (e.g., landfarming of municipal sludge). Average ambient air concentrations can range from 1×10^{-6} mg/m^3 in remote areas to 4×10^{-5} in urban areas. Concentrations of cadmium in surface water, ground water, and drinking water are typically less than 1 μg/L. In a study conducted in 27 U.S. cities, 12 food groups were tested and cadmium was detected in nearly all samples. As seen in Table 8-3, the highest levels of cadmium were found in leafy vegetables and potatoes and the lowest levels were found in beverages. Liver and kidney meats and shellfish were associated with higher concentrations than other types of meats. Cadmium can accumulate in freshwater and marine animals up to thousands of times higher than the levels of cadmium found in the water. In the U.S., the adult intake of cadmium attributable to diet is estimated to be approximately 30 μg/d. Assuming a gastrointestinal absorption of 5 to 10 percent, the amount of cadmium absorbed from diet is approximately 1 to 3 μg/d. In addition, human exposure can occur at the same level through cigarette smoking. Cadmium has been found to accumulate in tobacco plants. The average concentration of cadmium in a cigarette is between 1 and 2 μg/cigarette. A cadmium exposure

Table 8-3. Concentration of Cadmium in Various Food Products[13]

Food group	Average concentration (ppm)	Range of concentrations (ppm)
Potatoes	0.0421	0.016 to 0.142
Leafy Vegetables	0.0328	0.016 to 0.061
Grain and Cereal Products	0.0237	0.002 to 0.033
Root Vegetables	0.0159	trace-0.028
Garden Vegetables	0.0171	trace-0.093
Oils and Fats	0.0108	trace-0.033
Sugars and Adjuncts	0.0109	trace-0.053
Meat, Fish, and Poultry	0.0057	trace-0.014
Legume Vegetables	0.0044	trace-0.016
Dairy Products	0.0035	trace-0.016
Fruits	0.0021	trace-0.012
Beverages	0.0013	trace
All groups		trace-0.142

and absorption level of 1 to 3 μg/d can result from smoking one pack of cigarettes per day. Based on these data, the ATSDR[13] states that, for the general nonsmoking population, the highest levels of exposure to cadmium are most likely to occur through consumption of food. However, smoking can result in double the exposure level. Individuals living near emission sources may be exposed to above-average exposure levels through multiple pathways, such as ingestion of contaminated drinking water or garden vegetables, inhalation of airborne dust, and incidental ingestion of contaminated soil.

A study that examines the transfer of metals to bovine milk indicates that human exposure to cadmium through the consumption of milk may not be of concern. Because the contribution to human exposure through the food chain has not been thoroughly examined, a study[3] was undertaken to estimate the steady-state bovine milk biotransfer factors (i.e., the rate at which the compounds are transferred to milk) for six metals: arsenic, cadmium, chromium, lead, mercury, and nickel. Results from this study indicated that, of the metals studied, lead and arsenic transferred to milk to the greatest extent and cadmium to a lesser extent.[3] The bioconcentration factor estimated for cadmium is 1.3×10^{-6} L/kg. To put this value into perspective, Stevens points out the estimated biotransfer factor of TCDD is 2.6×10^{-2} L/kg and that this particular food chain pathway may be less important for these metals than it is for TCDD.

In general, human exposure to cadmium is most likely to occur through the consumption of food products for nonsmokers and through the consumption of food products and cigarette smoking for smokers. Because cadmium emissions from utilities do not contribute significantly to the total amount of cadmium released annually for anthropogenic activities (i.e., approximately 1 percent) and because there are numerous units dispersed throughout the country, it is unclear whether the impact of the cadmium emissions on the background concentration of cadmium in the various media is significant. Exposure resulting from utility emissions of cadmium in excess of background levels cannot readily be determined.

8.6 OVERALL SUMMARY

Further quantitative analysis of noninhalation exposure to HAPs that are persistent in the environment and that have the potential to bioaccumulate may be needed. Due, in part, to low emissions of these HAPs from utilities relative to other anthropogenic sources, a quantitative assessment of noninhalation exposure to dioxins, lead, and/or cadmium has not been given as high a priority as arsenic for further multipathway assessment. However, dioxins, lead, and cadmium are persistent in the environment, have a tendency to bioaccumulate, and are toxic by the inhalation and ingestion routes of exposure. Therefore, further assessment of the multipathway exposures and risks may be needed to more comprehensively evaluate the impacts of emissions of dioxins, lead, and cadmium from utilities.

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9.0 MULTIPATHWAY EXPOSURE AND RISK ASSESSMENT FOR RADIONUCLIDES

9.1 SUMMARY OF RADIONUCLIDE ANALYSIS

The EPA assessed the exposure and risks due to radionuclide emissions from coal-, oil-, and gas-fired utilities. The details of this assessment are contained in an EPA report entitled: *Estimates of Health Risks Associated with Radionuclide Emissions from Fossil-Fueled Steam Electric Generation Plants.*[1] This section summarizes that report.

Shortly after the discovery of radioactivity at the turn of the century, investigators became aware that nearly all natural materials contained trace quantities of radioactivity. Natural radioactivity is derived from two sources. A small percentage of natural radioactivity is derived from the interaction of cosmic radiation with specific elements (e.g., carbon-14, tritium). The majority of naturally occurring radionuclides are classified as primordial radioisotopes or their radioactive decay products. Primordial radionuclides are believed to have been formed along with all other terrestrial elements except hydrogen, by nuclear fusion reactions, neutron absorption, and beta decay in a former star, which exploded as a super-nova.[2]

The behavior of primordial radionuclides and their decay products in the environment is complex. Pathways leading to significant human exposures include external radiation from the emission of gamma rays from the ground and building materials. Internal exposure may result from the transfer of radioactivity through root uptake by plants that serve as food to domestic animals or are directly ingested by humans. Internal exposure may also result from the inhalation of airborne radioactivity.

The three major fossil fuels—coal, oil, and natural gas—contain varying quantities of the naturally occurring radionuclides of the uranium-238 and thorium-232 series and potassium-40. When these fuels are burned to produce steam in the production of electricity, radionuclides are entrained in the combustion gases and may be emitted into the environment. As early as 1954, Anderson, Mayneord, and Turner[3] suggested that man's activities, in particular the burning of coal, might significantly perturb the natural radiation environment by transferring additional radioactivity into the air, where it is more readily available for human intake by inhalation.

Radionuclides are among the HAPs included in section 112(b). Over the years, the EPA has reviewed available information and provided estimates regarding the radionuclide content of fossil fuels, environmental emissions, human exposure, and health risks. This information has been reported by the EPA in several reports, including the Background Information Document supporting the decision not to regulate radionuclide emissions from coal-fired boilers issued in 1989.[4] This report updates previously published data and estimates with more recently available information regarding the radionuclide contents of fossil fuels, associated emissions by utilities, and potential health effects to exposed population groups.

9.1.1 Natural Radionuclide Content in Fossil Fuels-Coal

The decay series of uranium and thorium constitute the major radionuclides contained in coal. Uranium-238 has 13 major radioactive decay products and thorium-232 has 9. For coal, it is generally assumed that primary members within each of the two decay series are in secular equilibrium. Secular equilibrium denotes that the radioactivity concentrations among primary decay chain members are constant. A national database of nearly 7,000 coal samples was analyzed with regard to uranium and thorium content of the major ranks of coal used by utilities. Concentrations spanned a wide range of values that were lognormally distributed. Table 9-1 summarizes the data by providing the geometric mean concentration values expressed in units of parts per million and identifies the relative percent utility consumption of coal types.

Concentration values expressed in parts per million are readily converted to radioactivity concentrations by means of the specific activity values for uranium-238 and thorium-232. For U-238, 1 ppm is equal to 0.33 pCi/g of coal; for Th-232, 1 ppm is equal to 0.11 pCi/g of coal. For example, with an average content of 1.24 ppm uranium and 2.18 ppm thorium in bituminous coal, there is a corresponding activity of 0.41 pCi/g for each member of the U-238 series and 0.24 pCi/g for each member of the Th-232 series.

The radionuclide content of coal is not unique when compared to other natural materials. In fact, it is generally assumed that the average radioactivity of the earth's crust (i.e., soil and rocks) is about twice that of coal.

9.1.2 Natural Radionuclide Content in Fossil Fuels-Natural Gas

Radioactivity in natural gas is almost exclusively radon-222, which migrates from proximal geologic formations into gas reservoirs. In 1989, the American Gas Association identified 262,482 production wells that yielded more than 18 trillion cubic feet (Tcf) of natural gas. An additional 1.53 Tcf of gas were imported primarily from Canada. About 2.77 Tcf of gas were consumed by utilities to produce electricity.

Table 9-1. Utilization and Radionuclide Content by Coal Rank

Coal rank	Percent utilization	Average uranium (ppm)	Average thorium (ppm)
Bituminous	69.0	1.24	2.18
Subbituminous	24.7	1.07	2.28
Lignite	6.3	1.41	2.38

The radon content of natural gas at the wellhead has been measured in thousands of wells over several decades. However, these measurements are of limited use for estimating radon concentrations at the point of consumption for several reasons. Radon concentrations vary by geographic location and over time. Also, radon content is markedly reduced when natural gas is processed to remove commercially valuable heavier hydrocarbons (ethane, propane, butane). Further reductions in radon concentrations reflect the natural decay that occurs during the gathering, processing, and distribution/storage of gas prior to consumption. The main radon isotope, Rn-222, has a half-life of about 4 days; the other isotopes have half-lives of less than 1 minute.

A more meaningful approach is to assess the radon content in gas distribution lines. Analyses of gas in the distribution lines eliminates well-to-well variations and accounts for radon reduction from processing and natural decay. Radon measurements of natural gas in distribution lines near the point of consumption suggest an average value of 20 pCi/L. In this report, therefore, radon emissions from gas-fired boilers are based on a radon concentration of 20 pCi/L of processed gas.

9.1.3 Natural Radionuclide Content in Fossil Fuels-Oil

Residual fuel oil denotes a general classification of fuel obtained as liquid still bottoms from the distillation of crude oil. Nonradiometric analyses show that crude oil and various petroleum products may contain as many as 60 different metals in measurable quantities. Uranium and thorium are among the trace metals commonly found in crude oil and petroleum products. The presence of these two radioactive trace metals also implies the presence of their radioactive decay products.

A comprehensive literature search, however, revealed that data specific to the radionuclide content of residual fuel oil are not only sparse but are considerably more difficult to interpret than those for coal or gas. Contributing to the difficulty in data interpretation is the absence of secular equilibrium among primary members of the U-238 and Th-232 decay chains. Due to the paucity of data, the EPA concluded that there was a need for additional data and conducted its own study.

The EPA enlisted the help of the Utility Air Regulatory Group (UARG) and the EPRI to solicit the voluntary participation of individual utilities in providing samples of residual oils for radioanalysis. The selection of a utility was based on the utility's geographic location, along with its generator nameplate capacity, capacity factor, and/or annual fuel-oil consumption. Selection, therefore, favored larger facilities with the highest capacity factors/fuel consumption and accounted for radionuclide variability based on origin of crude oil.

In total, 12 utilities provided 42 samples of residual fuel oil for analysis. Participating utilities represented major regions of the United States where fuel oil serves as a primary fuel source. Quantitatively, the 12 utilities had an annual consumption of about 2 billion gallons, which was estimated to be about 24 percent of the fuel oil consumed by all U.S. oil-fired units.

Radionuclide analysis, data interpretation, and data verification involved the efforts of a major commercial analytical laboratory, the EPA's National Air and Radiation Environmental Laboratory (NAREL), and the National Institute of Standards and Technology (NIST).

Table 9-2 provides estimates of the average radionuclide values of the 42 residual fuel oil samples evaluated in the EPA study. Values are well within the range of the limited study data reported by others and support the conclusion that the radionuclide content of residual fuel oil is low relative to coal.

9.1.4 Radionuclide Emissions from Fossil-Fueled Plants

Radionuclide emissions from utilities are affected by the radionuclide content in fossil fuel, by plant design features, and by operating parameters. Important design features involve the size of the plant, type of furnace used, and the emission control systems designed to remove pollutants from the flue gas. The most significant operational factors, which dictate the rate of fuel consumption, involve the percentage of time a plant is operating, the power level, and the efficiency by which a plant converts thermal energy to electric energy.

In this study, estimates of radionuclide emissions and associated human health risks are based on fossil-fired boiler

Table 9-2. Estimates of Average Radionuclide Concentrations in 42 Residual Fuel Oil Samples

U-238 Series	Concentration (pCi/g)
U-238	0.0018
Th-234	0.0018
Pa-234	0.0018
U-234	0.0034
Th-230	0.0068
Ra-226	0.0043
Rn-222	0.0043
Po-218	0.0043
Pb-214	0.0043
Bi-214	0.0043
Po-214	0.0043
Pb-210	0.44
Bi-210	0.44
Po-210	0.44
Th-232 Series	**Concentration (pCi/g)**
Th-232	0.0030
Ra-228	0.068
Ac-228	0.068
Th-228	0.068
Ra-224	0.068
Rn-220	0.068
Po-216	0.068
Pb-212	0.068
Po-212	0.068

units with generating capacities of 25 MWe or more. The 25-MWe selection criterion reflects the low probability of significant emissions for small plants regardless of unit-specific operating parameters. Of the Nation's 2,298 boiler units (Table 9-3), 1,748 units have a generating capacity of 25 MWe or more.

From data reported to the EEI that include annual fuel consumption and particulate removal efficiencies, emissions were estimated for each of the 1,748 boiler units and aggregated by plant affiliation. (The 1,748 fossil-fired boiler units represent a total of 684 utility plants.) These unit- and plant-specific emission data are contained in a separate addendum to the report. Table 9-3 provides average annual emissions per operating boiler unit, as well as per billion kilowatt-hour of electricity generated. For coal-fired units, the average annual emissions for particulates range from a fraction of a millicurie (mCi) to several millicuries among primary radionuclides.

Although the average radionuclide content of residual fuel oil is 2 to 3 orders of magnitude lower than that of coal, Table 9-3 reveals that average emission rates are nearly comparable.

Table 9-3. Average Annual Radionuclide Emissions per Operating Boiler Unit and per Billion Kilowatt-Hour Electricity Generated

Radionuclide	Emission Rates					
	Per Operating Unit (mCi/y)			Per Billion kWe-h Generated (mCi/y)		
	Coal	Gas	Oil	Coal	Gas	Oil
U-238	2.3 x 10^{0}	1.3 x 10^{2}	1.1 x 10^{-1}	1.5 x 10^{0}	2.6 x 10^{-2}	1.8 x 10^{-1}
Th-234	1.2 x 10^{0}	1.3 x 10^{-2}	1.1 x 10^{-1}	7.7 x 10^{-1}	2.6 x 10^{-2}	1.8 x 10^{-1}
Pa-234m	1.2 x 10^{0}	1.3 x 10^{-2}	1.1 x 10^{-1}	7.7 x 10^{-1}	2.6 x 10^{-2}	1.8 x 10^{-1}
Pa-234	1.2 x 10^{0}	1.3 x 10^{-2}	1.1 x 10^{-1}	7.7 x 10^{-1}	2.6 x 10^{-2}	1.8 x 10^{-1}
U-234	2.3 x 10^{0}	2.5 x 10^{-2}	2.1 x 10^{-1}	1.5 x 10^{0}	4.9 x 10^{-2}	3.4 x 10^{-1}
Th-230	1.2 x 10^{0}	4.9 x 10^{-2}	4.1 x 10^{-1}	7.7 x 10^{-1}	9.5 x 10^{-2}	6.7 x 10^{-1}
Ra-226	1.7 x 10^{0}	2.9 x 10^{-2}	2.6 x 10^{-1}	1.2 x 10^{0}	5.7 x 10^{-2}	4.3 x 10^{-1}
Rn-222	3.0 x 10^{2}	2.5 x 10^{3}	3.8 x 10^{2}	2.0 x 10^{2}	4.9 x 10^{3}	6.2 x 10^{2}
Po-218	5.6 x 10^{0}	3.1 x 10^{-2}	2.7 x 10^{-1}	3.8 x 10^{0}	6.0 x 10^{-2}	4.4 x 10^{-1}
Pb-214	5.6 x 10^{0}	3.1 x 10^{-2}	2.7 x 10^{-1}	3.8 x 10^{0}	6.0 x 10^{-2}	4.4 x 10^{-1}
Bi-214	1.2 x 10^{0}	3.1 x 10^{-2}	2.7 x 10^{-1}	7.7 x 10^{-1}	6.0 x 10^{-2}	4.4 x 10^{-1}
Po-214	5.6 x 10^{0}	3.1 x 10^{-2}	2.7 x 10^{-1}	3.8 x 10^{0}	6.0 x 10^{-2}	4.4 x 10^{-1}
Pb-210	5.6 x 10^{0}	3.1 x 10^{0}	2.7 x 10^{1}	3.8 x 10^{0}	6.0 x 10^{0}	4.4 x 10^{1}
Bi-210	1.2 x 10^{0}	3.1 x 10^{0}	2.7 x 10^{1}	7.7 x 10^{-1}	6.0 x 10^{0}	4.4 x 10^{1}
Po-210	5.6 x 10^{0}	3.1 x 10^{0}	2.7 x 10^{1}	3.8 x 10^{0}	6.0 x 10^{0}	4.4 x 10^{1}
Th-232	7.1 x 10^{-1}	2.1 x 10^{-2}	1.8 x 10^{1}	4.7 x 10^{-1}	4.1 x 10^{-2}	3.0 x 10^{-1}
Ra-228	1.0 x 10^{0}	4.7 x 10^{-1}	4.1 x 10^{0}	7.1 x 10^{-1}	9.1 x 10^{-1}	6.7 x 10^{0}
Ac-228	7.1 x 10^{-1}	4.7 x 10^{-1}	4.1 x 10^{0}	4.7 x 10^{-1}	9.1 x 10^{-1}	6.7 x 10^{0}
Th-228	7.1 x 10^{-1}	4.7 x 10^{-1}	4.1 x 10^{0}	4.7 x 10^{-1}	9.1 x 10^{-1}	6.7 x 10^{0}
Ra-224	1.0 x 10^{0}	4.7 x 10^{-1}	4.1 x 10^{0}	7.1 x 10^{-1}	9.1 x 10^{-1}	6.7 x 10^{0}
Rn-220	1.6 x 10^{2}	5.7 x 10^{-1}	8.4 x 10^{0}	1.1 x 10^{2}	1.1 x 10^{0}	1.4 x 10^{1}
Po-216	3.5 x 10^{0}	4.7 x 10^{-1}	4.1 x 10^{0}	2.4 x 10^{0}	9.1 x 10^{-1}	6.7 x 10^{0}
Pb-212	3.5 x 10^{0}	4.7 x 10^{-1}	4.1 x 10^{0}	2.4 x 10^{0}	9.1 x 10^{-1}	6.7 x 10^{0}
Bi-212	7.1 x 10^{-1}	4.7 x 10^{-1}	4.1 x 10^{0}	4.7 x 10^{-1}	9.1 x 10^{-1}	6.7 x 10^{0}
Tl-208	2.1 x 10^{-1}	1.4 x 10^{-1}	1.2 x 10^{0}	1.4 x 10^{-1}	2.7 x 10^{-1}	1.9 x 10^{0}
K-40	7.8 x 10^{0}	6.2 x 10^{-3}	5.2 x 10^{-3}	5.3 x 10^{0}	1.2 x 10^{-2}	8.5 x 10^{-3}

This is explained by the fact that, unlike coal-fired units, the majority of oil-fired units lack particulate emission control systems that remove radionuclides from the flue gas with efficiencies of 95 percent or more. Due to the fact that coal-fired units on average have a higher capacity factor, the degree of comparability between coal-fired and oil-fired units is further enhanced when emissions are defined per unit of billion kilowatt-hours.

Particulate emissions for units designated as gas-fired are generally small when compared to either coal- or gas-fired units. Moreover, radionuclide emissions other than radon from units designated as gas-fired principally result from the combustion of a secondary fuel.

9.1.5 Summary of CAP-93 Model

For a given facility, atmospheric releases may be modeled for as many as six independent sources. Plume rise can be calculated assuming either a momentum- or buoyancy-driven plume that reflects facility-specific plant parameters. Plume dispersion is based on a modified Gaussian plume equation and accounts for plume depletion that includes precipitation scavenging and dry deposition. Based on availability, primary model parameters for plume dispersion and depletion are based on site-specific meteorological data. (A library of meteorological data that include wind data files, annual precipitation, ambient temperatures, and lid-height for all major cities is provided by the code.)

From plume dispersion and plume depletion calculations, the program computes radionuclide concentrations in air and rates of deposition and buildup on ground surfaces and in soil. Estimates of the radionuclide concentrations in produce, leafy vegetables, milk, and meat are made by coupling the output of the atmospheric transport models with the terrestrial food-chain models defined in the U.S. Nuclear Regulatory Commission's Regulatory Guide 1.109. The quantities of foodstuff produced locally are based on the average agricultural productivity data of the State in which the assessment area is located.

For dose and risk estimates, the population distribution at each of the 684 assessed sites was developed by means of the GENPOP computer code and 1990 Census Bureau data. Dose estimates reflect the exposure from external (air immersion and ground surface) and internal (inhalation and ingestion) sources. For low-LET external radiation, CAP-93 employs the nominal risk coefficient of 3.9×10^{-4} fatal cancers per rem.

For internal exposures, dose and risk estimates are defined by ICRP tissue/organ weighting factors that account for route of entry, clearance class, and transfer factors within body compartments. In summary, dose and cancer risks can be tabulated for individual exposure pathways, radionuclides, and tissues/organs. All risk estimates pertain to the risk of fatal cancer and assume that exposure occurs over the lifetime of individuals within the assessed population.

EPA's methodology for estimating risks from Rn-222 emissions is based on an extrapolation of epidemiologic findings of underground miners (NAS 1988, NAS 1991) exposed to radon. CAP-93 calculates working levels (WL), not concentrations of specific radon daughter products. A WL is defined as any combination of short-lived radon decay products in 1 liter of air that will result in the emission of 1.3×10^5 MeV of alpha-particle energy. Risk is not derived from dose but from time-integrated exposure expressed in working level months (WLM). Under typical residential exposure conditions, it is assumed that 1 WLM corresponds to 170 hours of exposure at 200 pCi/L of radon gas. CAP-93 employs a risk coefficient of 3.6×10^{-4} fatal lung cancers per WLM.[a]

CAP-93 assesses risk for a circular grid that is defined by sixteen sectors and up to 20 radial distances around a specified facility. For this study radial distances of 400, 1,500, 3,500, 7,500, 10,000, 15,000, 25,000, 35,000, 45,000, and 50,000 meters were used. Risk to the population is determined by summing individual risks by distance and section for the 0 to 50-km grid around each assessed facility. Risk to the maximally exposed individual(s) corresponds to that location (i.e., distance and sector of highest exposure) where individuals are believed to reside.

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[a] Recently, the Agency revised its estimates of radiogenic cancer risks to reflect the current epidemiological data and scientific consensus on extrapolations from the available data to chronic low dose exposures.[5] The revised estimates yield a nominal value of 5.1×10^{-4} fatal cancer per rad for uniform whole body exposure to low-LET radiation and 2.2×10^{-4} fatal lung cancers per WLM for exposure to radon-222 and its decay products. The radon risks reported in this study can be adjusted to the new radon risk coefficient simply by applying a correction factor of about 0.6. No simple adjustment can be made to the non-radon risks to reflect the Agency's current values. However, since the ground surface pathway dominates the risk for maximally exposed individuals, an upward adjustment of approximately 30 percent would bound their risks.

The population risk frequency distribution identifies the number of people at various levels of risk. The risk categories are divided into powers of 10, in which the individual lifetime cancer risk ranges from one chance in ten to less than one chance in a million. Risk data for each of the 684 assessed plants are provided in the previously identified Addendum. Only a summary of these data is provided below.

9.1.6 Estimates of Population Health Risks

Radionuclide emissions from utilities may result in public exposure from multiple pathways that include (1) external radiation from activity suspended in air or deposited on the ground and (2) internal exposure from the inhalation of airborne contaminants or ingestion of contaminated food products. Although the potential health risks are essentially independent of whether a dose was internal or external, the assessment of internal exposures is considerably more complex. For ingested or inhaled radionuclides, dose assessment requires biokinetic information that describes the distribution and retention of individual nuclides, the type of radiation emitted, and the amount of energy absorbed by individual target tissues/organs.

Estimates of population doses from chronic atmospheric releases require the use of a computer code that accounts for atmospheric dispersion, radionuclide concentrations in environmental media, and radionuclide intakes by inhalation and ingestion. In support of National Emission Standards for Hazardous Air Pollutants, the EPA, with support from Oak Ridge National Laboratory, developed the CAP-88 computer model. The CAP-88 (Clean Air Act Assessment Package-1988) computer model is a composite of computer programs, databases, and associated utility programs.

The CAP-88 programs are considered among the best available verified models for population dose and risk assessment for radionuclide air emissions. For a given facility, atmospheric releases and dose assessment may be modeled for up to six independent sources that take into account plant- and site-specific model parameters.

Since it was first introduced, CAP-88 has been revised periodically to reflect changes in database information and improved risk methodologies. For this study, the most recent version of the code, designated as CAP-93, was used. The CAP-93 contains a correction to the procedure used to calculate wet deposition of radionuclides from the plume.

For low doses of radiation, potential health effects may not appear for years or even decades following exposure. Such delayed effects are termed "stochastic" and are thought to result from highly selective molecular changes in individual cell(s). Although these highly selective changes occur rarely, when they do, the altered cell may develop into cancer. Among the stochastic effects that have been associated with radiation exposure, medical scientists consider cancer induction the primary health effect of concern.

A key characteristic of a stochastic effect is that the **severity** of the effect is **not** dose-dependent. However, the **probability** that a stochastic event (i.e., cancer) may occur is dictated by the radiation dose. The stochastic nature of low-dose radiation is not unique but is universal to all carcinogenic agents that act by primary genetic mechanisms.

The current method of estimating radiation risks relies on select human studies in which cancer rates were observed at a higher incidence among exposed individuals than would normally occur spontaneously. The most intensely studied people are the Japanese atomic bomb survivors of Hiroshima and Nagasaki. Data through 1985 show that, among the 76,000 individuals studied, 5,935 survivors have died of cancer from all causes. It is estimated that about 340 of these cancers (i.e., 80 leukemias and 260 nonleukemias) were the result of radiation exposure.

The data also define a dose response in which increasing doses yielded an increased percentage of excess cancers, especially for leukemia. However, some numerical estimates embody substantial statistical uncertainties as to the number of cancer deaths induced by radiation. Thus, for doses less than 50,000 mrem (50 rem), the small number of excess cancers above normal expected levels may reflect random fluctuations that are not linked to radiation exposure. When doses exceed 50,000 mrem (50 rem), the number of excess cancers is sufficient to support a causal link to human cancers.

For low-dose exposures, a causal link and a quantitative relationship between radiation dose and cancer has not been established. Yet, scientists conservatively assume that any dose of radiation, no matter how small, may pose a risk to human health. Estimates of health risks from low-level radiation are, therefore, derived by extrapolating risks from high doses to lower doses using a linear nonthreshold dose-response model contained in the CAP-88 and CAP-93 computer codes.

The CAP-93 assesses risk for a circular grid that is defined by 16 sectors and a radial distance of 50 km around a facility. Risk to the population is determined by summing individual risks by distance and sector for the 0- to 50-km grid around each assessed facility. Risk to the maximally exposed individual(s) corresponds to that location (i.e., distance and sector) of highest exposure where individuals are believed to reside.

The population risk frequency distribution identifies the number of people at various levels of risk. The risk categories are divided into powers of 10, in which the individual lifetime cancer risk ranges from one chance in ten to less than one chance in a million. Risk data for each of the 684 assessed plants are provided in a separate addendum. These data are summarized in section 9.2.

9.2 RADIONUCLIDE UNCERTAINTY ANALYSIS[6]

Uncertainties in the estimates of risk presented for radionuclides emitted from fossil-fuel-fired steam electric-generating units were assessed using both qualitative judgments and quantitative techniques. Since, as in almost all assessments of environmental health risk, the risk estimates were based on modeling rather than direct measurements of exposure and risk, the results were subject to uncertainties in modeling, completeness, and parameter values.

Modeling uncertainties pertain to the formulation of the mathematical models used to predict risk and the degree to which they accurately represent reality. Completeness uncertainties pertain to whether or not all significant radionuclides and pathways of exposure are addressed. And parameter uncertainties pertain to the specific values assigned to the parameters that are input to the calculational models.

Census-tract, air dispersion, environmental transport, metabolic, and dose-response models were used to predict the location of individuals around the plants, the dispersion of the pollutants in the environment, their concentrations in soil and air at receptor locations, their accumulation and removal from soil, their uptake and transfer from soil to foodstuffs, and their intakes, translocations, accumulations, and removal from the various organs and tissues of the body, and the resulting risks to the individuals in the exposed population.

The modeling uncertainty associated with the use of the GENPOP census-tract model used to locate the individuals within 50 km of each plant was limited to its ability to properly place individuals living in proximity to the plant. The potential magnitude of this uncertainty was partially assessed in a quantitative manner using field-verification techniques to identify actual locations of nearby individuals for the plants with the highest estimated maximum individual risks. The results of these plant-specific assessments indicated that, on a plant-by-plant basis, the reported MIR might be high by an order of magnitude. However, they also demonstrated that the reported MIR of about 10^{-5} is correct when viewed as an upper bound for the entire population of electric utility steam-generating units.

Modeling uncertainties associated with the air dispersion, environmental transport, metabolic, and dose-response models were considered via model input parameters. Significant model inputs were included in a rigorous Monte Carlo analysis of parameter uncertainties associated with two plants with the highest estimated MIRs. For the dispersion and environmental transport models that were used, which are widely recognized as appropriate for the physical processes that govern dispersion and environmental concentration, assessment of parameter uncertainty only was clearly appropriate. For the metabolic and dose-response models, the parameter uncertainty relied on lumped parameters. This approach reflected the limited data available on the uptake and retention of radionuclides within the various organs and tissues of the body and the necessity of extrapolating the dose-response relationship from data reflecting much higher (orders of magnitude) exposures.

Uncertainties in completeness are limited to source terms and exposure pathways. Because the source terms for utilities are well characterized, there is very little likelihood that significant unaccounted for radionuclide releases are occurring at these facilities. With respect to pathways of exposure, the analysis assumed that four pathways of exposure (ingestion of milk, meat, and vegetables; inhalation; immersion in contaminated air; and exposure to contaminated ground) were present at all sites. The ground water pathway was not included because the deposited material is on the ground surface in a physical and chemical form that minimizes its potential to leach to ground water.

Exposure to multiple sources is one potentially significant exposure pathway that was not accounted for by the air dispersion and environmental transport modeling. To evaluate the potential significance of this pathway, explicit hand calculations were performed for the plants with the highest estimated MIRs to estimate the impacts from all plants within 50 km. The results of these explicit calculations showed that omission of multiple plants from the estimates resulted in less than a 5 percent error in the MIRs. A related completeness uncertainty was the impact on individuals residing beyond the 50-km assessment area around each plant. A semiquantitative assessment of this uncertainty, which affects the estimate of deaths per year in the exposed populations, indicated that the risk was not understated by more than a factor of 3. Given these results, completeness uncertainties are not judged to be a significant contributor to the overall uncertainty in the analysis.

The largest uncertainties were associated with the parameter values that were used in the assessment. As noted above, a rigorous assessment of parameter uncertainties was conducted for the two plants with the highest estimated MIRs. Nominal values used in the assessment were assigned a distribution and range based on available data and expert judgment. Based on this analysis, it was determined that the 90 percent confidence interval for the reported MIR values of approximately 10^{-5} ranges from about 10^{-6} to 10^{-4}.

9.2.1 Summary Findings

Table 9-4 gives the distribution of fatal cancer risks to the combined populations residing within the 50-km (35-mile) radii of the 684 utility plants. The aggregate of assessed populations living within a 50-km radius of a plant is estimated to be 196.1 million, which represents approximately 75 percent of the U.S. population. The individual lifetime risk of fatal cancer to more than 99.9 percent of the assessed population (i.e., 196.1 million) is less than one chance in a million. The data further suggest that, under current operating conditions, there are no instances in which the release of radioactivity is likely to result in a lifetime fatal cancer risk to any one person that is equal to or greater than 1 chance in 10,000. It is estimated that about 1,027 individuals residing within a 50-km distance of a plant may receive radiation exposures for which the lifetime risk is between 1 in 10,000 and 1 in 100,000 (i.e., 1×10^{-4} to 1×10^{-5}).

It must also be pointed out that the distribution of individual risks within each risk range is heavily skewed toward the lower value. This is evidenced by the fact that the average individual lifetime risk is a small fraction of the midpoint value within each of the risk ranges. Correspondingly, the probability of a single fatal cancer occurrence within the highest risk group of 1,027 individuals is less than 2 chances in 10,000 per year. For the entire assessed population of 196,100,000 within 50 km of these plants, the estimated cancer

Table 9-4. Frequency Distribution of Lifetime Fatal Cancer Risks for All Plants

Lifetime cancer risk range	Number of people	Average individual lifetime risk	Deaths per year in this risk range	Death per year in this risk range or higher
1×10^{0} to 1×10^{-1}	0	0	0	0
1×10^{-1} to 1×10^{-2}	0	0	0	0
1×10^{-2} to 1×10^{-3}	0	0	0	0
1×10^{-3} to 1×10^{-4}	0	0	0	0
1×10^{-4} to 1×10^{-5}	1,027	1.3×10^{-5}	1.92×10^{-4}	1.92×10^{-4}
1×10^{-5} to 1×10^{-6}	95,745	2.2×10^{-6}	3.06×10^{-3}	3.26×10^{-3}
Less Than 1×10^{-6}	196,000,000	1.2×10^{-7}	3.32×10^{-1}	3.36×10^{-1}

risk attributable to radionuclide emissions from electric utility SGUs (includes coal-, oil-, and gas-fired utilities)is less than 1 cancer death per year (i.e., 3.36×10^{-1} deaths/year is the risk equivalent of about 1 in 3 chances that a single cancer death will occur in a year). Exposures and risks to individuals residing beyond 50 km are not explicitly evaluated. However, using the assumption that radionuclides dispersion and exposure beyond 50 km would be similar to arsenic, which was modeled with the RELMAP (see chapter 6), the EPA estimates that the overall cancer incidence may be seven times greater. That is, considering both local and long-range transport, the cancer incidence could be roughly 2 cases per year (i.e., 0.3 x 7). Most (approximately 99 percent) of the cancer incidence is due to inhalation exposure. The EPA estimates that coal-fired utilities are contributing about 25 percent of the cancer incidence and oil-fired utilities the other 75 percent.

Based on radionuclide emissions and plant-specific/site-specific data, CAP-93 also calculates the MIR for each of the 684 plants. Table 9-5 characterizes those plants with the highest estimated MIR values expressed in **lifetime** fatal cancer risk. There were a total of 17 plants for which the lifetime risk of fatal cancer to the MIR is estimated to exceed 1×10^{-5} due to multipathway exposures to radionuclide emissions from utilities. The highest MIR value of 3×10^{-5} corresponds to a five-unit coal-fired facility that generated 3,340 MW of electricity in 1990. Of the 17 plants with the highest MIR values, 11 are exclusively designated as coal boilers. Only two facilities are identified as exclusively oil-fired plants.

Table 9-5. Plants with the Highest Estimated Maximum Individual Risk (MIR)*

Plant Name	MIR*	Coal-Fired		Gas-Fired		Oil-Fired	
		Units	MWe	Units	MWe	Units	MWe
Plant #222	3×10^{-5}	5	3,340				
Plant #247	3×10^{-5}	4	900				
Plant #60	2×10^{-5}	4	3,160				
Plant #301	2×10^{-5}	2	750	3	262		
Plant #251	2×10^{-5}	4	1,540				
Plant #406	2×10^{-5}	4	2,777				
Plant #256	2×10^{-5}	3	1,728				
Plant #17	2×10^{-5}					2	1,112
Plant #133	2×10^{-5}	2	1,135			2	66
Plant #318	1×10^{-5}	6	1,100				
Plant #672	1×10^{-5}	8	1,965				
Plant #668	1×10^{-5}	7	2,304				
Plant #82	1×10^{-5}					2	804
Plant #207	1×10^{-5}					2	558
Plant #253	1×10^{-5}	3	2,052				
Plant #489	1×10^{-5}	4	1,872				
Plant #651	1×10^{-5}					6	372

* MIR is the maximum individual risk expressed as lifetime fatal cancer risk.

The remaining four plants are represented by a combination of boilers, where coal is at least one of the designated primary fuels.

The MEI risk due to inhalation exposure to radionuclides from the highest risk oil-fired plant is estimated to be 1×10^{-5}. The MEI risk due to inhalation exposure to radionuclides from the second highest risk oil-fired utility is estimated to be 3×10^{-6}. The other 135 oil-fired utilities and all coal-fired utilities are estimated to pose cancer risks less than 1×10^{-6} due to inhalation exposure to radionuclides.

Because of limitations in the GENPOP computer code used for identifying locations of individuals, the MIRs shown for each plant should be viewed with caution; errors of a few hundred meters in the location of individuals can result in an over- or underestimate of risk by factors of 2 or more. The UARG reestimated the risks for the 17 plants with the highest MIRs using refined population grids. Their results show lower MIRs for the majority of these plants, but their highest MIR of 1×10^{-5} is consistent with the EPA's estimates. Thus, the EPA believes the GENPOP methodology is sufficiently accurate to establish the magnitude of MIRs for all utilities.

9.3 REFERENCES

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3. W. Anderson, Mayneord W.V., and Turner R.C. "The Radon Content of the Atmosphere," Nature. 174:424. 1954.

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5. U.S. Environmental Protection Agency. Estimating Radiogenic Cancer Risks. EPA-402-R-93-076. Office of Radiation and Indoor Air. Washington, D.C. 1994.

6. National Council on Radiation Protection and Measurements. "A Guide for Uncertainty Analysis in Dose and Risk Assessment Related to Environmental Contamination." NCRP Commentary #14, May 10, 1996. pp. 54.

10.0 ALTERNATIVE CONTROL STRATEGIES FOR HAZARDOUS AIR POLLUTANT EMISSIONS REDUCTIONS

This chapter presents methods of reducing HAP emissions through precombustion controls, combustion controls, postcombustion controls, and alternative controls. Also, strategies for maximizing total HAP control or minimizing total HAP emissions are reviewed.

The HAPs of concern include the trace elements identified in Chapter 5 as a potential health risk. These consist of arsenic, cadmium, chromium, lead, manganese, mercury, and nickel; dioxins and furans (due to the toxicity of the organic chemical); and HCl and HF (due to the estimated emissions of the compounds).

10.1 PRECOMBUSTION CONTROLS

To reduce SO_2 emissions and thereby comply with the Phase II requirements of the Acid Rain Provisions of the Act, some utilities will switch to fuels that contain lower amounts of sulfur. The effects of fuel switching on HAP emissions is briefly reviewed in this section. Emissions of trace elements from coal-fired units may be controlled through precombustion control techniques such as coal cleaning and coal gasification. The effectiveness of these control techniques is also reviewed.

10.1.1 Fuel Switching

Utilities that switch fuel may change from higher to lower sulfur-containing coal (less than 1.5 weight percent sulfur) or elect to burn a different type of fuel (e.g., switching from oil to natural gas combustion). A concern with fuel switching is whether or not it will increase HAP emissions, due to potentially increased concentrations of trace elements in the fuel.

The qualitative effects of switching the type of fuel may be noted through comparisons of the averages of trace element concentrations in utility fuels. Table 10-1 lists the arithmetic average, as well as the standard deviation of the average, for trace element concentrations in coal, residual oil, and natural gas. As indicated by the magnitude of the standard deviations listed in Table 10-1, trace element concentrations vary considerably in coal and residual oil. Some of the standard deviations are large enough that comparable concentrations of trace elements may occur in some coal and residual oil samples. For discussion purposes, any overlap in trace element concentrations was ignored, and generalizations on the effects of fuel switching were made from comparisons of average trace element concentrations in the three fuels.

Table 10-1. Comparison of Average Concentrations of Trace Elements in Utility Fuels[a,b]

	Coal[c]	Residual	Natural gas[d]
Sulfur	(lb/MMBtu)	(lb/MMBtu)	(lb/MMBtu)
average[e]	1.24	0.31	0.00006
SD (mean)[f]	0.19	0.07	0.00006
No. averages	26	13	2
	Coal[c]	**Residual**	**Natural gas[e]**
Trace elements:	(lb/trillion)	(lb/trillion)	(lb/trillion)
Arsenic			
average	660	17	0.19
SD (mean)	120	11	0.06
No. averages	26	6	2
Cadmium			
average	60	5.4	
SD (mean)	30	3.9	--
No. averages	26	3	
Chloride			
average	27,000	7,400	
SD (mean)	6,600	3,300	--
No. averages	20	11	
Chromium			
average	600	17	
SD (mean)	98	3	--
No. averages	26	11	
Fluoride			
average	5,300	600	
SD (mean)	720	200	--
No. averages	26	3	
Lead			
average	800	73	
SD (mean)	190	43	--
No. averages	26	5	
Mercury [1,2]			
average	7.7	0.6	0.001
SD (mean)	0.6	0.3	1
No. averages	152	4	
Nickel			
average	700	1,300	
SD (mean)	69	200	-
No. averages	26	13	

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[a] The coal data listed in Table 10-1 were not weighted for coal production by State of coal origin.

[b] There were only two sets of data for concentrations of trace elements in natural gas in Table 10-1.

[c] With the exception of the mercury data, coal values were determined from modified U.S. Geological Survey (USGS) data, by State of coal origin, and coal shipment data for coals that originated from three States. Modified USGS data are USGS data that were modified to account for the effects of bituminous coal cleaning. Mercury data were reported by EPRI for samples of coal shipments.[2] None of the data were weighted for coal production.

[d] Natural gas values were determined from the preliminary EPRI test reports for Sites 120 and 121. The listed values are detected concentrations.

[e] Averages of averaged data sets.

[f] This is the standard deviation of the number of averages directly below.

10.1.1.1 Switching to Natural Gas Combustion. As shown in Table 10-1, natural gas has the lowest average concentrations, on a lb/trillion Btu basis, of sulfur, arsenic, and mercury when compared with the corresponding values for residual oil and coal. The averages listed for coal and residual oil exceed those listed for natural gas by factors that range from approximately 100 (for the concentration of arsenic in residual oil) to as much as approximately 21,000 (for the concentration of sulfur in coal). Thus, of the three utility fuels, natural gas contains the least amounts of the trace elements, and switching from coal or residual oil to natural gas combustion would ultimately reduce emissions of trace elements. Some total and seasonal conversion of coal- and oil-fired units to natural gas firing is expected to affect compliance with the various ozone and NO_x control provisions of Titles I and IV of the Act. However, a complete conversion of all utility boilers to natural gas is not practical. Even though the natural gas transmission network is expanding, delivery of natural gas to each utility unit cannot yet be accomplished. In addition, there is concern over the long-term availability of natural gas (particularly with respect to other fuels) given the projected usage (and increase in usage) in the residential, commercial, and industrial sectors and the estimates of proved and supplemental reserves of natural gas. Estimates of "proved reserves" of natural gas have decreased each year (but one) for the past 10 years.

10.1.1.2 Switching from Coal to Residual Oil Combustion. As shown in Table 10-1, with the exception of the average concentrations of nickel, the average concentrations of trace elements listed for coal exceed those listed for residual oil by factors that range from approximately 4 (for the concentrations of sulfur and chloride) to as much as approximately 40 (for the concentration of arsenic). However, the average concentration of nickel in coal is approximately half the corresponding value for residual oil. Thus, switching from coal to residual oil combustion could result in increased emissions of nickel and decreased emissions of the other trace elements.

10.1.1.3 Switching from Higher to Lower Sulfur Coals. The effects of coal switching will be reviewed first for mercury and then for the other trace elements.

Figure 10-1 shows the relationship between the concentrations of mercury and sulfur in 153 samples of coal shipments.[2] As shown in Figure 10-1, there is no relationship between the sulfur and mercury content in the sampled coal shipments; mercury concentrations below approximately 15 lb/trillion Btu are present in coal with both higher sulfur

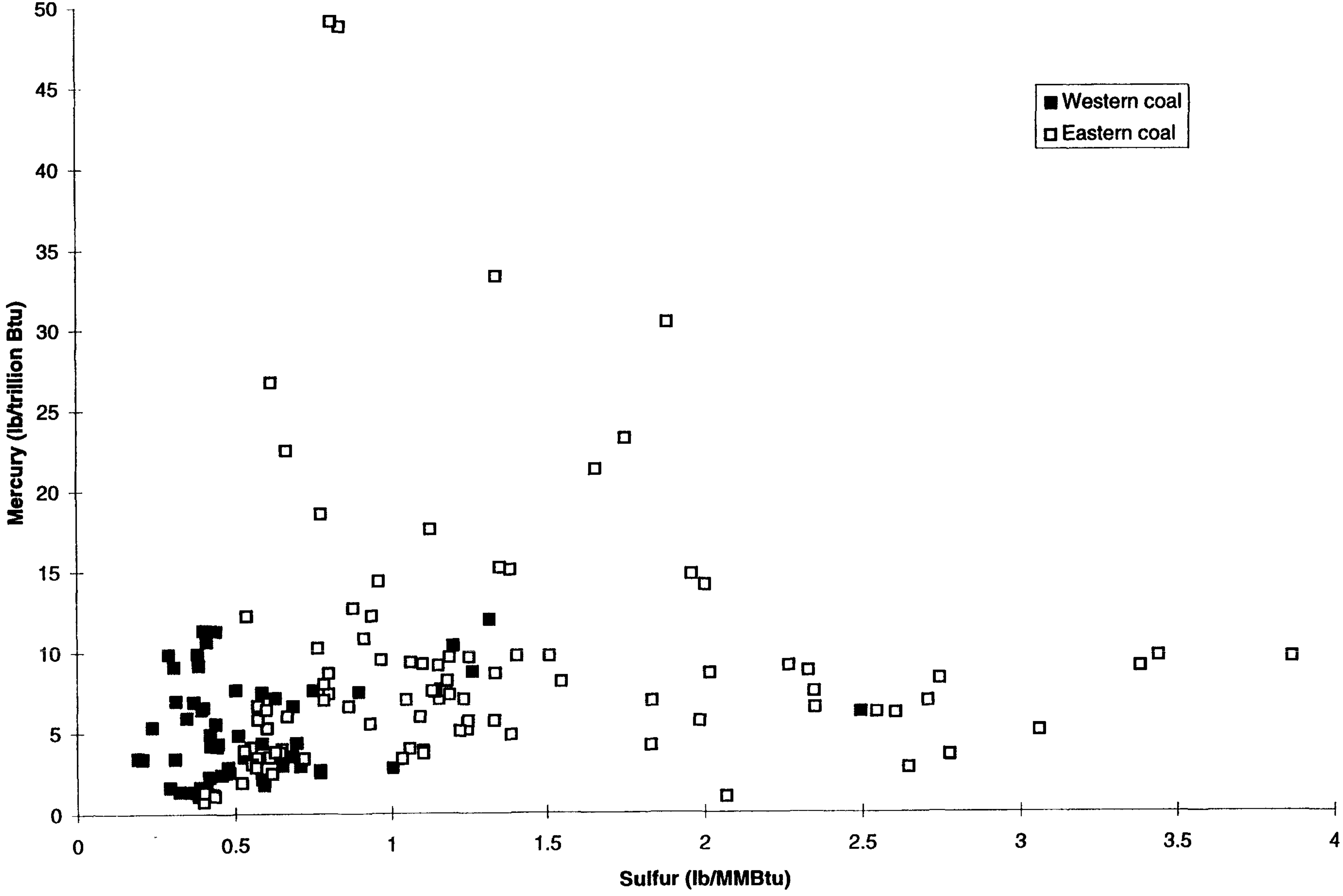

Figure 10-1. Relation between the concentrations of mercury and sulfur in 153 samples of coal shipments.[2]

concentrations (above 2.5 lb/MMBtu) and lower sulfur concentrations (below 1.5 lb/MMBtu).

A conceivable control strategy would involve blending higher mercury-containing coals with lower mercury-containing coals to reduce mercury emissions. Such a practice would be comparable to blending high and low sulfur-containing coals in order to meet SO_2 emission limits. However, coal blending for mercury control is not a proven control strategy. Changes in the electrical resistivity and amount of flyash resulting from coal blending could reduce PM capture efficiencies by ESPs and subsequently lead to increased emissions of PM and HAP metals. However, these effects on ESP performance can be addressed by gas conditioning and/or modifications to the ESP. Blending for mercury control could also increase levels of other HAPs or sulfur. Another uncertainty with coal blending for mercury control is the possibility of blending coals that contain different species of mercury, and changes in the amount of vapor- and PM-phase species of mercury would affect mercury control with PM control devices. Another factor is that the blending of two different coals might change the higher heating value of the resulting mixture.

The qualitative effects of switching to lower-sulfur-containing coals on other metallic HAPs are examined in Figure 10-2(a-g) through plots of the average concentrations of each HAP, excluding mercury, with sulfur content in coal. As shown in Figure 10-2(a-g), the average concentration of trace elements in coal shipments, as approximated by the modified U.S. Geological Survey (USGS) data (modified for the effect of coal cleaning on bituminous coals), generally show no clear trends with sulfur content (i.e., decreasing the sulfur content of coal does not generally lead to reduced concentrations of trace elements in coal).

Based upon average concentrations of trace metals in coal from the modified USGS data, fuel switching to lower-sulfur-containing coals will not generally result in consistently reduced emissions of the trace elements. Trace elements associated with the PM (e.g., arsenic, cadmium, chromium, lead, and nickel) could be removed from coal-fired flue gas with a PM control device.

It should be stressed that the effects of coal switching were drawn from comparisons of average concentrations of trace elements in modified USGS coal data. The concentrations of trace elements in actual coal shipments may vary from the USGS averages.

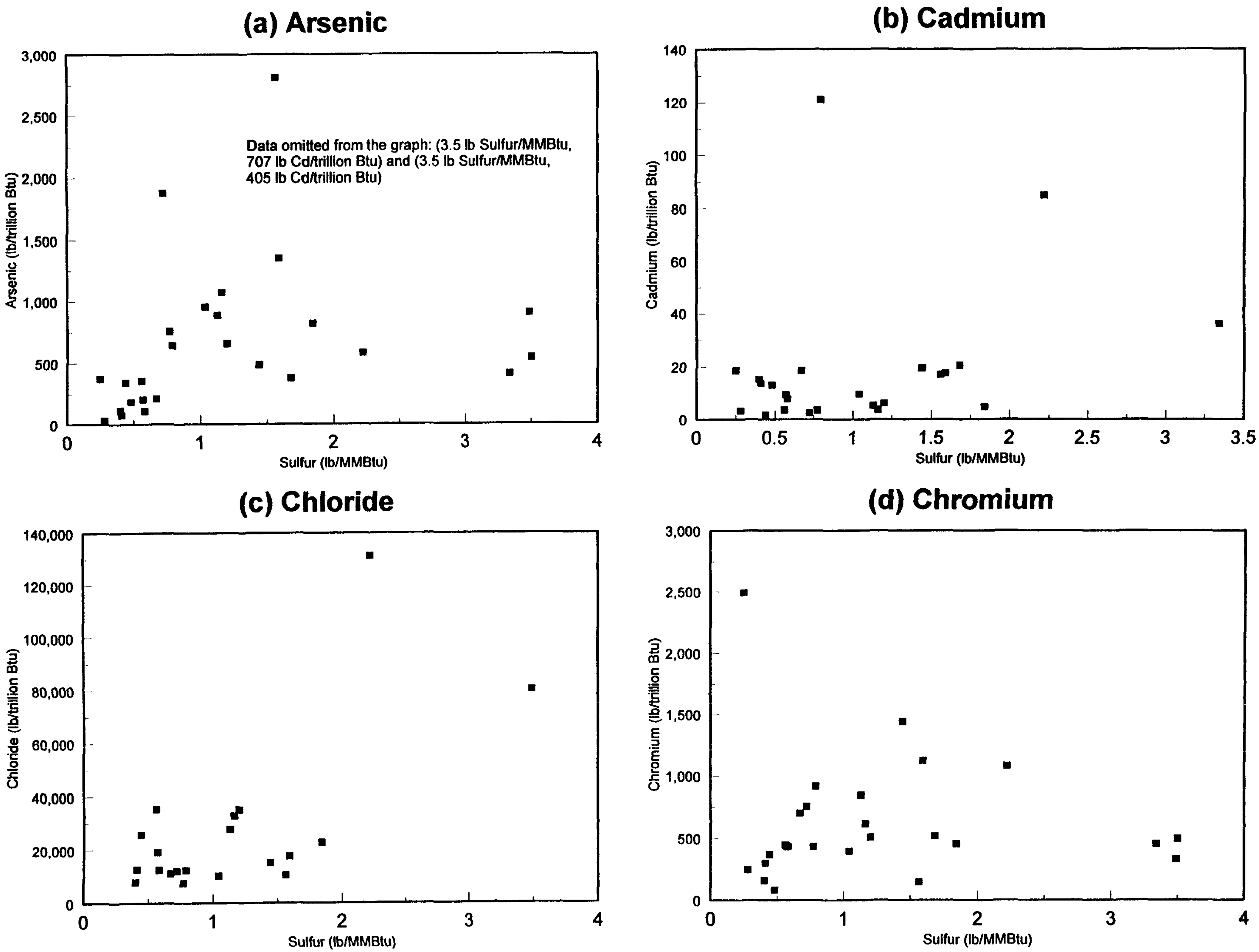

Figure 10-2 (a-g). Relation between concentration of selected trace elements and sulfur in modified USGS data.

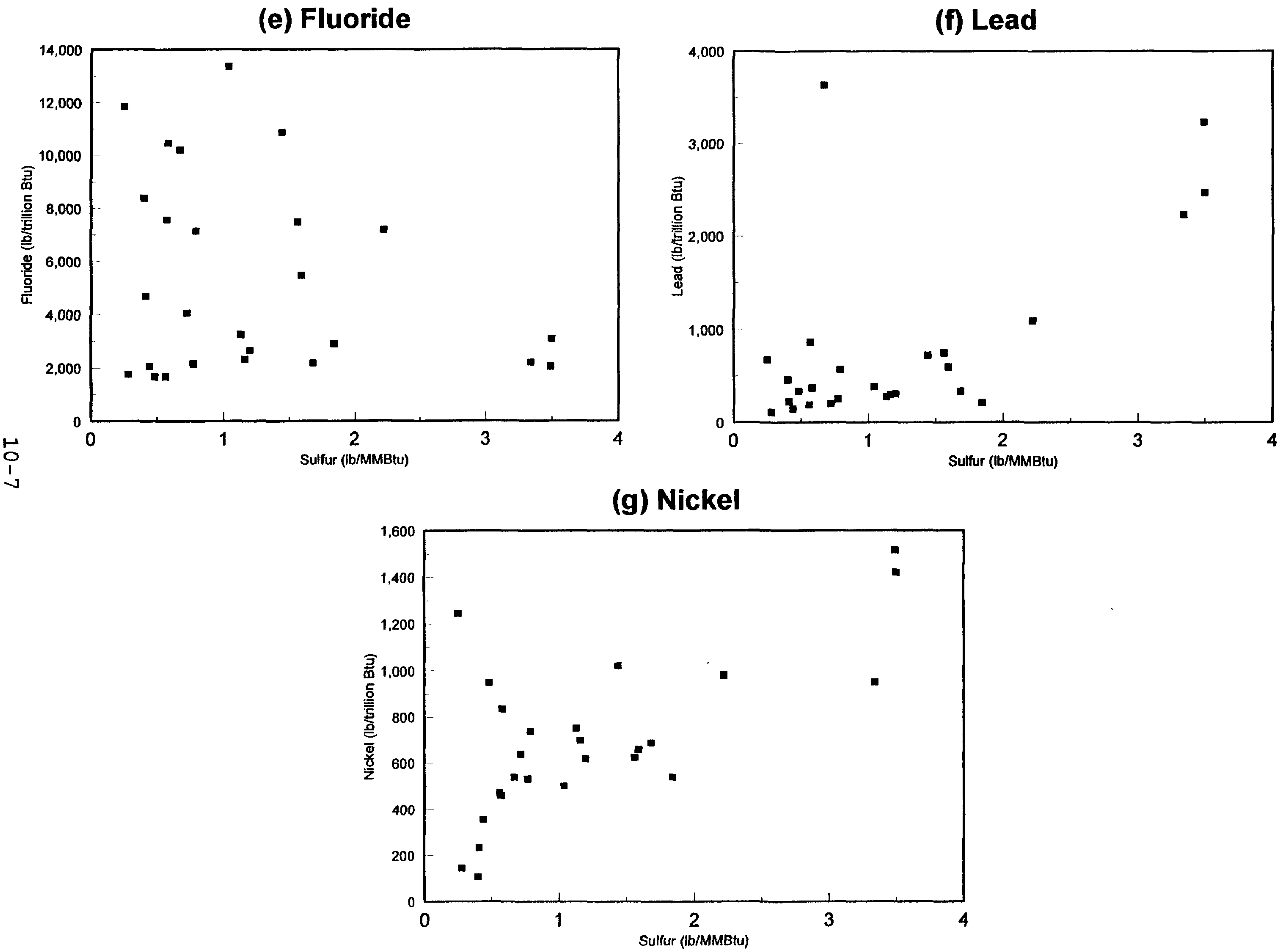

Figure 10-2 (a-g). (continued).

10.1.2 Coal Cleaning

Approximately 77 percent of eastern and midwestern[3] bituminous coal shipments are cleaned to meet customer specifications on heat, ash, and sulfur content.[g] Subbituminous and lignite coals are not routinely cleaned.[4] Conventional coal cleaning removes mineral matter and, in the process, may also remove some of the trace elements contained in the mineral matter. The mineral matter is removed from the coal by either crushing and screening or by coal washing (through the difference in specific gravities of the constituents).[5] In the process of removing the mineral matter, coal cleaning generates solid refuse that contains trace elements; the solid refuse must be handled carefully and disposed of properly. Any coal cleaning liquid wastes will also contain trace elements, but the liquid wastes may be properly clarified and then recycled.

Table 10-2 lists the limited amount of available data on trace element reductions achieved through conventional coal cleaning. In Table 10-2, some of the trace element reductions are negative. Negative percentages occur when part of the coal is removed but the element is not contained in the extracted portion of the coal, so that the same weight of the element that was contained in the uncleaned coal is contained within a relatively smaller weight of the cleaned coal. Because the weight of the trace element does not change, negative removal percentages are considered to indicate that no trace element reduction occurred or that the trace element reduction was effectively 0 percent.

As shown in Table 10-2, for the limited amount of available data, trace element removal percentages may vary for coals obtained from the same seam. The variability occurs because trace element concentrations, in the mineral portion of coal, vary from coal to coal. For the data listed in Table 10-2, the variation in trace element reductions may also be due to the use of various cleaning methods, the accuracy of the analytical techniques used to measure concentrations of trace elements in cleaned and uncleaned coals, and sample collection methods. With regard to analytical techniques, CONSOL, Inc., noted the following difficulties in analyzing mercury concentrations in coal: the volatility of mercury compounds, low mercury concentrations in coal, large variability of approximately 50 percent in the interlaboratory reproducibility of mercury concentrations, lack of certified mercury-in-coal standards, and lack of standard sample preparation and analysis methods.[6,7]

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[g] Research has only recently begun to look at coal cleaning specifically for trace metal removal.

Table 10-2. Trace Element Reductions Achieved Through Conventional Coal Cleaning[a]

Seam	State	Reference	% Removal arsenic	% Removal cadmium	% Removal chromium	% Removal fluoride	% Removal lead	% Removal mercury	% Removal nickel
C. App. A		a	39	14	75		57	-11	50
C. App. B		a	22	33	67		65	8.3	39
IL	IL	e	54	59	21		38	55	24
IL #6	IL	a	47	76	72		35	43	40
IL #6	IL	c	3	32	23	27	37	-8.3	21
IL 2,3,5	IL	d	30		44	41	68	17	58
IL 2,3,5	IL	d	48		46	60	59	42	37
Ky #11	Ky	d	52		60	83	88		68
Ky #11	Ky	d	13		69	92	84	20	73
Ky #9 & 14	Ky	c	40	53	29	42	49	13	33
Lower Kittanning	Pa	c	66	43	44	64	59	23	35
Pittsburgh	Pa	b	74		71	-459	67	20	33
Pittsburgh	Pa	c	53	57	56	67	65	15	51
Pittsburgh	Pa	c	27	20	24	30	34	7.7	31
Pittsburgh A	Pa	a	65	40	42		37	27	51
Pittsburgh B	Pa	a	67	58	66		69	36	62
Pittsburgh C	Pa	a	67	63	63		63	7.1	53
Pittsburgh D	Pa	a	77	55	64		63	-20	69
Pittsburgh E	Pa	a	50	50	71		72	20	69
Pratt	Al	d	29	0	48	19	58	3.4	46
Pratt	Al	d	7.1	0	54	54	24	29	21
Pratt/Utley	Al	d	29	0	41	65	53	21	54
Sewickley	Pa	c	36	52	46	57	94	0	52
Upper Freeport	Pa	b	38		11	47	53	-200	26
Upper Freeport	Pa	d	73	40	50	80	8.3	64	15
Upper Freeport	Pa	d	74	42	45	75	61	60	30
Utley	Al	d	23	0	17	6.5	28	21	24
All seams:									
min			3.4	0	11	-459	8.3	-200	15
max			77	76	75	92	94	64	73
average			45	38	49	50	55	21	43

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[a] Negative percentages are listed as entries. However, averages were determined with negative percentages treated as zero removal.

Table 10-2. (continued)

References

Data for references a - d were taken from the report:

Akers, David, Robert Dospoy, and Clifford Raleigh, The Effect of Coal Cleaning on Trace Elements, Draft Report, Development of Algorithms, December 16, 1993, prepared for EPRI by CQ Inc..

Data for reference e were taken from the report:

Demir, Ilham, Richard D. Harvey, Rodney R. Ruch, Heinz H. Damberger, Chusak Chaven, John D. Steele, Wayne T. Frankie, Ken K. Ho, "Characterization of Available Coals from Illinois Mines," draft report, December 28, 1993, Illinois State Geological Survey file number to be assigned.

Specific references that were mentioned in the report by Akers, Dospoy, and Raleigh:

a DeVito, M., L. Rosendale, and V. Conrad, "Comparison of Trace Element Contents of Raw and Clean Commercial Coals," presented at the DOE Workshop on Trace Elements in Coal-Fired Power Systems, Scottsdale, AZ, April 1993.

c Ford, C. and A. Price, "Evaluation of the Effects of Coal Cleaning on Fugitive Elements: Final Report, Phase III," DOE/EV/04427-62, July 1982.

The average trace element reductions, listed in Table 10-2 for a limited amount of available data, were determined with the negative percentages treated as 0 percent removal and the averages not weighted by coal production since the EPA does not believe sufficient data exist at this time to follow a production-weighted approach. The average values for the limited amount of available data indicate that, in general, lead concentrations were reduced the most (approximately 55 percent) while mercury concentrations were, on average, reduced the least (approximately 21 percent). It should be stressed that better and worse trace element reductions may be found for specific coals. For instance, mercury removals of at least 50 percent should occur during conventional cleaning of Upper Freeport coal, based upon the modes of occurrence of mercury and available conventional coal cleaning data.[8] In Table 10-2, mercury removals were reported for three samples of Upper Freeport coal; two indicated approximately 62 percent removal while the third indicated effectively no mercury removal. More research is needed to establish trace element removal efficiencies, achieved through conventional coal cleaning, on a statistically representative sampling of commercially viable coal seams.

Although there is variability in trace element reductions, the data suggest that coal cleaning techniques may be useful in reducing trace element concentrations in selected coals. More studies are needed on diverse samples of coal to establish the effectiveness of coal cleaning in reducing trace element concentrations and to determine the causes of variability in cleaning effectiveness.

10.1.3 Coal Gasification

Coal gasification converts coal to a form that emits lower quantities of pollutants at the utility boiler than if the coal were not converted. Although there are some disadvantages to this process (the cost of gasification and the addition of another combustion source), the total quantity of air pollutants emitted from the combination of gasification and combustion is expected to be lower than burning coal in a conventional system.[9,10]

The gasification process typically described for near-term generation projects uses integrated gasification combined cycle technology (IGCC) and conventional cold-gas cleanup. In this process, gas from coal is used to generate electricity from both a steam turbine and a gas turbine. Steps in the process, shown in Figure 10-3, include coal preparation, coal oxidation and gasification, gas cooling, and gas cleanup. A large part of the pollutant mass is transferred to the slag or ash produced during gasification, and more of the impurities are transferred to water streams used in the gas cleanup. Slag or ash from the gasification step may be treated for recovery of salable products, and the stream from gas cleanup may be treated for recovery of sulfur. Heat transferred from the cooling step is used to produce steam for the steam turbine generator, while fuel gas made from the coal is burned to produce more electricity from the gas turbine generator. The IGCC technology can produce up to 25 percent more electricity from a given amount of coal than is currently obtained from conventional boilers.[9]

Statements by the DOE[9] suggest that IGCC technology is almost certain to be one of the lowest-cost fossil fuel options for generating electricity in the 21st century. When used to refurbish an existing plant, the technology is less expensive than building a conventional coal-fired plant. Other claims include higher thermal efficiency (to about 40 or 45 percent from about 35 percent), higher plant output (by 50 to 150 percent), and lower SO_2 and NO_x emissions as described above. The IGCC process has been demonstrated in a limited number of commercial-scale projects. In addition, IGCC is being utilized in several DOE CCT projects.

There are limited data available on the impact of IGCC on HAP emissions. Experiments with a different gasifier, an air-blown, fixed-bed gasifier coupled to a turbine simulator, produced trace metal concentrations as shown in Table 10-3.[10] The hot gas from the gasifier was treated in a moving bed with zinc titanate sorbent, followed by a cyclone. Emissions from the hot-gas cleanup were vented to a flare.

Hot-gas cleaning may fail to condense volatile trace species (e.g., mercury) and may produce a high concentration of trace elements in the vapor phase. This could lead to higher trace element emissions compared to the concentrations normally encountered in conventional coal-fired power plants.[11]

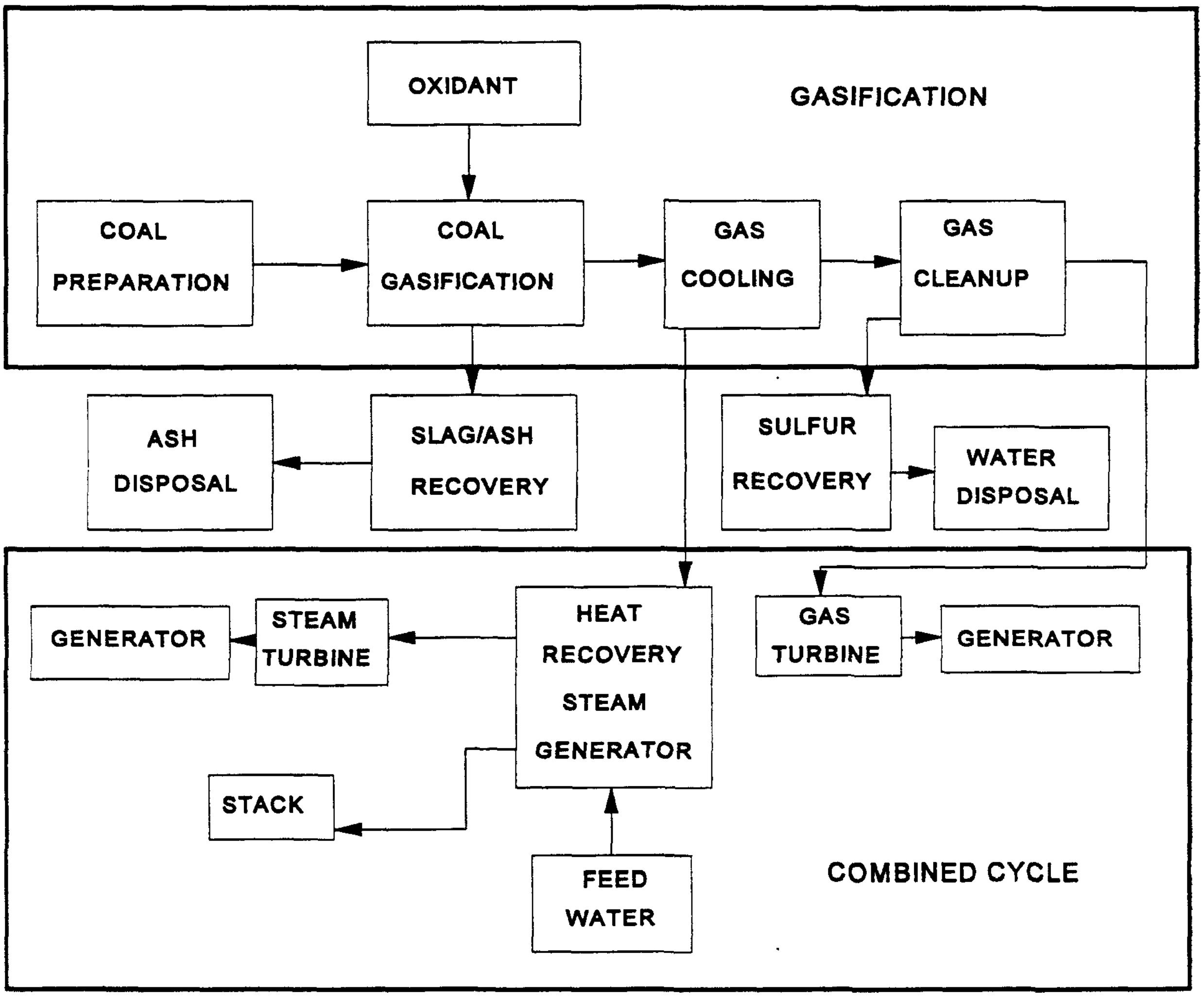

Figure 10-3. Coal gasification combined cycle technology

Table 10-3. Emissions from an Air-Blown, Fixed-Bed Gasifier

Trace metal	Emissions to flare, μg/Nm³	Emissions from turbine simulator, μg/Nm³	Total air, μg/Nm³
Arsenic	639	8	647
Cadmium	16	0.19	16.2
Chromium	155	20	175
Mercury	20	2	22
Nickel	1,530	26	1,556
Selenium	68	0.56	68.6

10.2 COMBUSTION CONTROL

Combustion control deals with the effect of furnace type (firing method and bottom type) and furnace modifications (such as the addition of low-NO_x burners) on HAP formation. Since most of the recent emission testing on utility units dealt with the generation and control of trace metals and not organic HAPs, trace metals are used to analyze the effect of combustion control. The trace metals for coal-fired units examined in this section are arsenic, beryllium, cadmium, chromium, lead, manganese, and mercury; those for oil-fired units are arsenic, lead, nickel, and mercury.

While the majority of recently collected HAP data has focused on metals, some small-scale studies have been conducted to evaluate changes in combustion conditions on organic HAPs. In one test, coal was burned at normal and elevated excess air levels and with air staging to simulate combustion modification NO_x controls.[12] A large number of organic HAPs were sampled in each case. Some increases in HAP emissions were noted for the air-staging conditions, but the conclusion was that this increase would not result in emissions at significant levels, even for a large utility boiler. During a second small-scale test, combustion conditions were varied between very high excess air and substoichiometric conditions.[13] This study concluded that low-NO_x firing conditions did not necessarily exacerbate emissions of organic HAPs. In both studies, the organic emissions were found to be one or more orders of magnitude less than emissions of any of the metallic HAPs, even under the worst combustion conditions tested. Although these results are from small-scale units and are relatively limited in their scope, they provide additional information supporting the position that, in general, organic HAP emissions are not likely to increase significantly due to the installation of low-NO_x combustion equipment.

The effect of NO_x control on metallic HAP generation was examined by developing an average emission output in lb/trillion Btu from one oil- and several coal-fired units. These units were tested before and after the installation of NO_x control or after the addition of greater NO_x control to an existing NO_x-controlled unit. As shown in Table 10-4, there appears to be a trend toward reductions in HAP emissions through the addition of NO_x control. However, this trend is neither uniform (see arsenic, beryllium,

Table 10-4. Comparison of Electric Utility Emissions Before and After Application of NO_x Control or Application of Greater NO_x Control on a Unit With Lesser NO_x Control (lb/trillion Btu)[14-17]

Trace metal	EPRI Site 110 (with low NO_x burners)	EPRI Site 110 (with low NO_x burners and offset air)	EPRI Site 110 percentage change	EPRI Site 114 (without NO_x control)	EPRI Site 114 (with overfire air and reburn burners)	EPRI Site 114 percentage change	EPRI Site 13 (without NO_x control)	EPRI Site 13 (with burners out of service)	EPRI Site 13 percentage change	EPRI Site 16 (with overfire air)	EPRI Site 16 (with overfire air and low NO_x burners)	EPRI Site 16 percentage change
Arsenic	141.73	64.15	-121%	137.22	151.4	9%	7.1	3.7	-92%	1789.01	1805.89	1%
Beryllium	43.87	45.26	3%	39.86	36.21	-10%	ND	ND	ND	108.33	130.43	17%
Cadmium	448.83	120.9	-271%	17.38	20.41	15%	10.13	13.84	27%	10.93	21.07	48%
Chromium	745.12	743.02	-0.28%	241.77	164.57	-47%	2.95	8.95	67%	934.26	845.76	-10%
Lead	878.75	503.23	-75%	1365.7	1007.17	-36%	8.02	4.6	-74%	437.31	351.15	-25%
Manganese	1021.84	999.7	-2%	392.1	375.22	-4%	4.7	7.84	40%	745.42	890.91	16%
Mercury	3.58	0.02	-17981%[a]	6.47	4.48	-44%	0.23	0.17	-36%	7.06	10.74	34%
Nickel	538.55	511.13	-5%	1073	1437.58	25%	1827.15	1355	-35%	664.17	655.97	-1%
Average Overall Percentage Change	—	—	-67%	—	—	-11%	—	—	-15%	—	—	10%
	Lesser NO_x control	Greater NO_x control		Lesser NO_x control	Greater NO_x control		Lesser NO_x control	Greater NO_x control		Lesser NO_x control	Greater NO_x control	

ND = This trace metal was below the detection limit in both the coal feed and the boiler exit emissions on both tests.

[a] This percentage difference seems too large and is not used in the average overall percentage change calculation.

and cadmium percentage change between Sites 110 and 114) nor universal (see Site 16 vs. all other sites). The differences in the percentage change could be due to the variability of trace metal concentrations in the oil or in different sources of bituminous coal; differences in the carbon, chloride, or ash content of the coal burned; differences in the age or operating condition of the units; or a combination of all of these or other factors.

The effect of the bottom type (furnace type) on HAP generation was examined by analyzing an average emission output of coal-fired units (see Table 10-5) with either a wet bottom furnace or a dry bottom furnace, both burning pulverized coal. Emissions were further segregated by coal type. Since there was only one test done on a conventional, dry bottom, lignite-fired unit, no lignite-fired units were analyzed. Site 111, which burned a mixture of bituminous and subbituminous coal, was grouped with the subbituminous coal-fired units for this analysis. To support the emission results, the EMFs for these furnaces and the concentrations of the trace metals found in feed coal are also included in Table 10-5. In this way, the effects of furnace type and low-NO_x operation and of trace metal concentration in the coal can be observed. Furnace type with and without low-NO_x operation may affect the partitioning of ash between bottom ash and fly ash. Trace metal concentration in the coal affects the trace metal concentration in the flue gas in either vapor or solid form. Oil-fired units could not be separated into wet or dry bottom configurations, as all oil-fired units use dry bottom furnaces. Therefore, the impact on HAP emissions of bottom type is not addressed for oil-fired boilers.

When units firing bituminous and subbituminous coal were analyzed, their averages showed that arsenic, beryllium, lead, and mercury seemed to be emitted in higher amounts by wet bottom units while cadmium, chromium, and manganese seemed to be emitted in higher amounts by dry bottom units (see Figure 10-4a). When units firing only bituminous coal were analyzed, the same effect was observed (see Figure 10-5a). When units firing only subbituminous coal were analyzed, their averages showed that emissions of almost all HAPs stated above were emitted in higher amounts from dry bottom units than from wet bottom units (see Figure 10-6a). Mercury was the exception, being emitted in similar amounts by both bottom types. The percent removal by bottom type and the concentrations of trace metals in the feed

Table 10-5. Comparison of wet bottom vs. dry bottom electric utility unit metallic HAP emissions, trace metal removal, and trace metal concentrations in feed coal [14,15,17,18-32]

Compound		EPRI Site 114 (without NOx control)	EPRI Site 114 (with overfire air and reburn burners)	DOE Niles test (with SNOX SCR)	DOE Niles	Northern States Power Riverside 8	EPRI Site 102 (same as NSP A.S. King)			
Bottom Type		WET	WET	WET	WET	WET	WET			
Furnace Type		Cyclone	Cyclone	Cyclone	Cyclone	Cyclone	Cyclone			
	Fuel type	BIT	BIT	BIT	BIT	SUB	SUB	Total bituminous & subbituminous	Bituminous	Subbituminous
	Emissions in lb/trillion Btus							Average emissions	Average emissions	Average emissions
Arsenic		137.2	151.4	2349.3	1583.3	32.4	51.4	718	1055	42
Beryllium		39.9	36.2	55.6	41.3	5.0	3.1	30	43	4
Cadmium		17.4	20.4	4.8	2.6	2.1	1.9	8	11	2
Chromium		241.8	164.6	434.0	366.1	54.7	146.0	235	302	100
Lead		1365.7	1007.2	637.5	593.0	56.3	152.0	635	901	104
Manganese		392.1	375.2	420.4	322.5	66.4	599.0	363	378	333
Mercury		6.5	4.5	21.3	19.7	3.0	6.4	10	13	5
	Emission modification factors							Average percent removal by boiler (1-average EMF)	Average percent removal by boiler (1-average EMF)	Average percent removal by boiler (1-average EMF)
Arsenic		0.15	0.25	0.85	0.58	0.51	0.48	53%	54%	50%
Beryllium		0.15	0.15	0.30	0.26	0.08	0.04	84%	79%	94%
Cadmium		0.01	0.01	0.20	0.11	0.16	0.02	92%	92%	91%
Chromium		0.30	0.23	0.35	0.28	0.22	0.25	73%	71%	77%
Lead		0.50	0.84	0.60	0.56	0.38	0.61	42%	38%	50%
Manganese		0.20	0.18	0.19	0.15	0.13	0.33	80%	82%	77%
Mercury		0.73	0.54	1.00	1.00	1.00	1.00	12%	18%	0%
	Trace metal concentrations in coal (micrograms/gram)							Average trace metal concentrations in coal (lb/trillion Btu)	Average trace metal concentrations in coal (lb/trillion Btu)	Average trace metal concentrations in coal (lb/trillion Btu)
Arsenic		609.94	889.55	2773.70	2735.93	63.25	106.95	1196.6	1752.3	85.1
Beryllium		246.23	269.83	184.91	158.68	60.45	74.62	165.8	214.9	67.5
Cadmium		3539.16	2891.03	24.47	24.62	13.95	82.91	1096.0	1619.8	48.4
Chromium		707.83	815.42	1250.88	1313.24	252.99	580.37	820.1	1021.8	416.7
Lead		1204.82	2742.77	1060.53	1067.01	148.82	248.73	1078.8	1518.8	198.8
Manganese		2033.13	1964.42	2175.45	2079.30	509.70	1824.03	1764.3	2063.1	1166.9
Mercury		8.28	8.90	20.94	17.24	2.79	4.97	10.5	13.8	3.9

BIT = bituminous
SUB = subbituminous
SNOX = wet sulfuric acid-selective catalytic reduction process

Table 10-5. (continued)

Compound		EPRI Site 110 (w/LNB and offset air)	EPRI Site 110 (w/LNB)	DOE Yates	EPRI Site 15	DOE Cardinal	EPRI Site 12	EPRI Site 14	EPRI Site 115	EPRI Site 16 OFA	EPRI Site 16 OFA/LNB	***Compound***
Bottom Type		DRY	DRY	DRY	DRY	DRY	DRY	DRY	DRY	DRY	DRY	***Bottom Type***
Furnace Type		Tangential	Tangential	Tangential	Tangential	Front	Opposed	Opposed	Vertical	Opposed	Opposed	***Furnace Type***
	Fuel type	BIT	BIT	BIT	BIT	BIT	BIT	BIT	BIT	BIT	BIT	
	Emissions in lb/trillion Btus											
Arsenic		64.2	141.7	393.1	598.9	1043.2	498.4	230.4	23.1	1789.0	1805.9	Arsenic
Beryllium		45.3	43.9	90.1	45.8	113.0	58.6	30.1	9.0	108.3	130.4	Beryllium
Cadmium		120.9	448.8	23.8	7.4	32.4	33.8	5.3	2.3	10.9	21.1	Cadmium
Chromium		743.0	745.1	2798.8	1164.3	976.0	1252.7	735.6	50.2	934.3	845.8	Chromium
Lead		503.2	878.8	688.6	811.3	623.0	381.2	115.2	64.0	437.3	351.2	Lead
Manganese		999.7	1021.8	2055.9	1752.6	1350.8	1723.6	1152.2	195.2	745.4	890.9	Manganese
Mercury		0.0	3.6	12.4	2.8	1.7	7.6	0.8	1.3	7.1	10.7	Mercury
	Emission modification factors											
Arsenic		0.89	0.39	1.00	0.60	0.91	1.00	0.50	0.61	1.00	1.00	Arsenic
Beryllium		0.93	0.43	1.00	0.54	0.96	1.00	0.92	0.52	1.00	0.82	Beryllium
Cadmium		1.00	0.70	1.00	0.01	1.00	0.14	0.02	0.58	1.00	0.11	Cadmium
Chromium		1.00	1.00	1.00	0.58	0.61	1.00	0.67	0.57	0.58	0.69	Chromium
Lead		1.00	0.36	1.00	1.00	1.00	1.00	0.79	0.38	1.00	0.66	Lead
Manganese		0.71	0.76	1.00	0.81	0.27	1.00	0.93	0.58	0.60	0.88	Manganese
Mercury		1.00	0.75	1.00	1.00	0.41	0.74	0.74	0.78	0.64	1.00	Mercury
	Trace metal concentrations in coal (micrograms/gram)											
Arsenic		163.45	159.93	183.73	N/A	1146.58	451.47	458.05	38.20	1240.88	1666.67	Arsenic
Beryllium		106.24	47.14	86.61	84.62	117.32	48.79	32.72	17.51	102.19	159.42	Beryllium
Cadmium		171.62	138.89	23.62	615.38	11.11	247.58	218.12	3.98	8.03	188.41	Cadmium
Chromium		735.52	462.96	1952.74	2000.00	1599.89	1237.89	1090.59	87.54	1605.84	1231.88	Chromium
Lead		1389.31	420.88	629.92	307.69	602.62	174.76	145.41	167.13	372.26	528.99	Lead
Manganese		1307.59	1430.98	1842.51	2153.85	5092.97	1310.71	1236.00	334.26	1240.88	1014.49	Manganese
Mercury		6.54	5.39	6.04	10.77	4.30	10.19	17.45	1.67	10.95	10.14	Mercury

BIT = bituminous
SUB = subbituminous
SNOX = wet sulfuric acid-selective catalytic reduction process

Table 10-5. (continued)

	EPRI Site 11	DOE Springerville	DOE Boswell	NSP Sherburne 3	NSP Riverside 6,7			
	DRY	DRY	DRY	DRY	DRY			
	Tangential	Tangential	Front	Opposed	Front			
Fuel type	SUB	SUB	SUB	SUB	SUB	Total bituminous & subbituminous	Bituminous	Subbituminous
Emissions in lb/trillion Btus						Average emissions	Average emissions	Average emissions
	124.1	81.0	34.4	64.4	47.1	463	659	70
	9.9	93.4	11.7	21.8	9.4	55	67	29
	58.7	172.4	11.0	7.0	10.3	64	71	52
	248.2	607.0	446.7	253.6	326.7	809	1025	376
	143.1	91.0	176.5	525.4	236.1	402	485	234
	1406.5	5627.6	6278.5	4978.2	480.0	2044	1189	3754
	2.7	5.4	4.7	6.0	4.3	5	5	5
Emission modification factors						Average percent removal by boiler (1-average EMF)	Average percent removal by boiler (1-average EMF)	Average percent removal by boiler (1-average EMF)
	0.92	0.29	0.23	0.79	0.99	26%	21%	36%
	0.79	0.87	0.60	0.58	0.40	24%	19%	35%
	0.35	1.00	1.00	0.99	0.25	39%	44%	28%
	0.72	0.68	1.00	0.49	1.00	23%	23%	22%
	1.00	0.19	0.42	0.49	0.19	30%	18%	54%
	0.98	0.72	0.57	0.00	0.77	24%	24%	24%
	0.29	1.00	0.87	1.00	1.00	19%	19%	17%
Trace metal concentrations in coal (micrograms/gram)						Average trace metal concentrations in coal (lb/trillion Btu)	Average trace metal concentrations in coal (lb/trillion Btu)	Average trace metal concentrations in coal (lb/trillion Btu)
	134.45	279.14	147.23	81.94	47.66	489.7	740.8	138.1
	12.61	107.81	19.49	37.46	23.27	65.4	81.1	40.1
	168.07	48.13	6.50	7.02	40.67	122.0	164.5	54.1
	344.54	886.19	341.02	515.04	326.70	1016.9	1350.8	482.7
	117.65	481.28	421.13	1065.20	1220.31	479.6	366.1	661.1
	1428.57	7780.67	11042.55		626.60	2925.3	1778.2	5219.6
	9.24	3.56	5.41	2.34	4.27	7.4	8.9	5.0

BIT = bituminous
SUB = subbituminous
SNOX = wet sulfuric acid-selective catalytic reduction process

Figure 10-4a. Average emissions

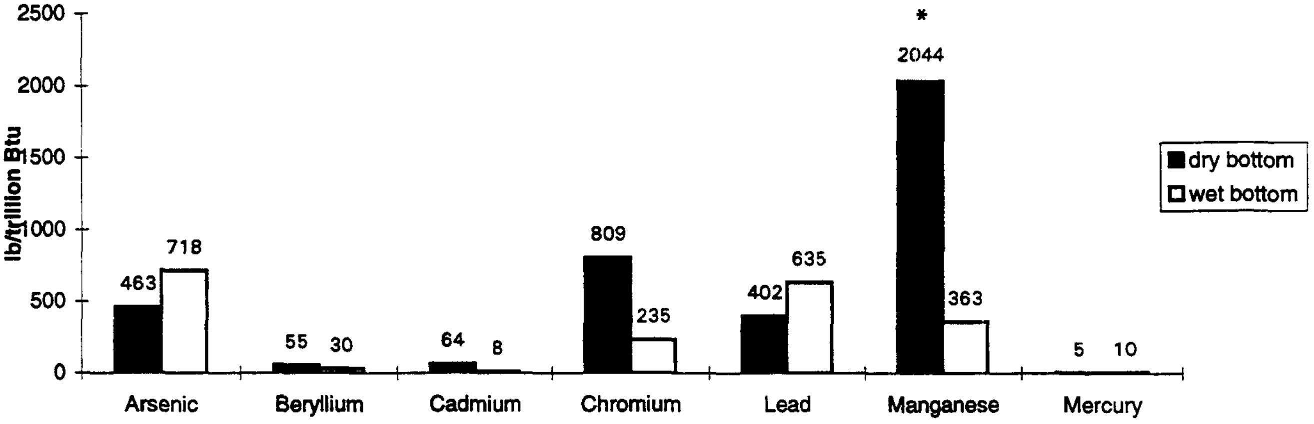

Figure 10-4b. Average trace metal removal by boiler

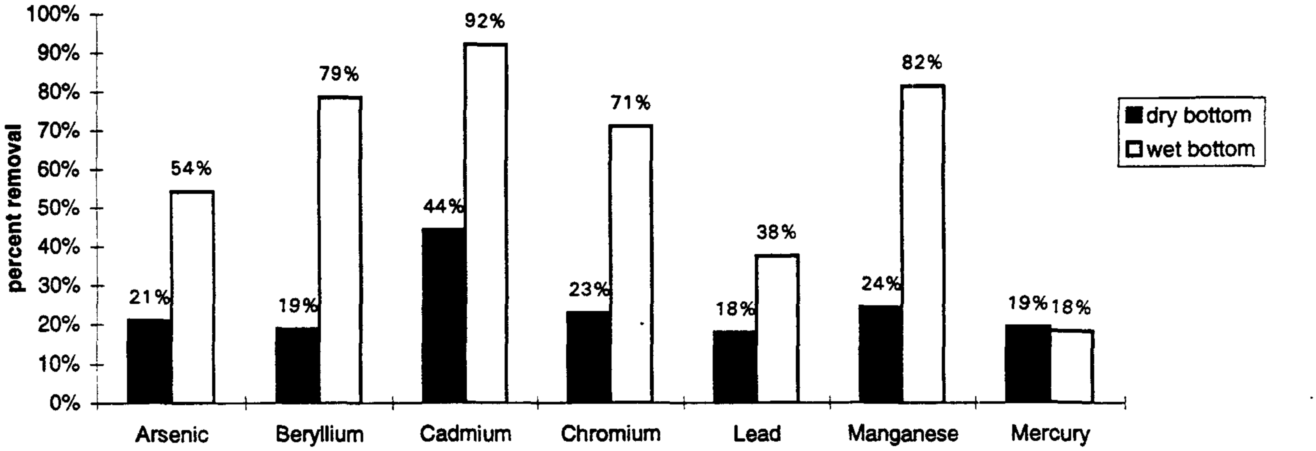

Figure 10-4c. Average trace metal concentration in feed coal

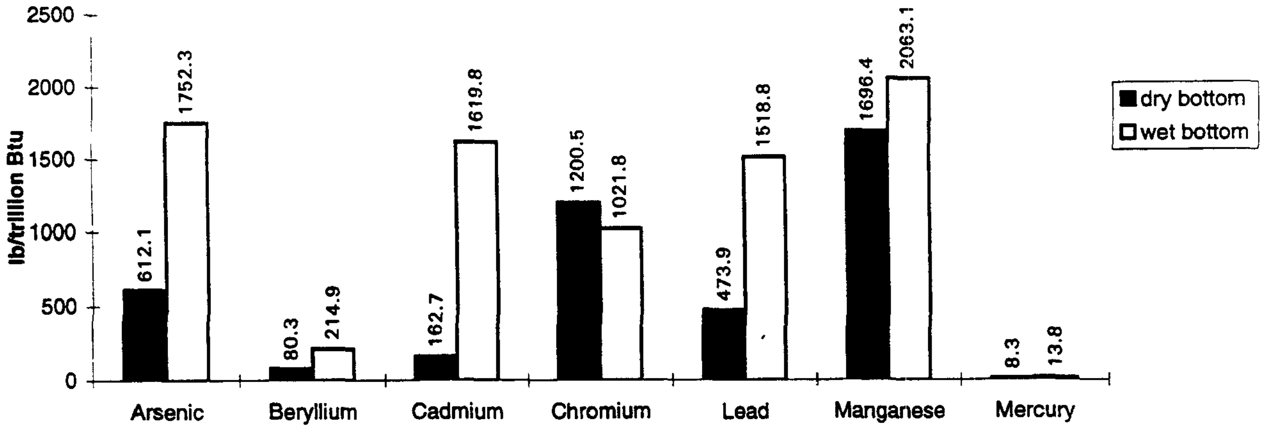

Figure 10-4 (a-c). Average coal-fired emissions, average trace metal removal, and average trace *element concentration in feed coal vs. bottom type* (bituminous and subbituminous coal-fired)

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* Denotes a boiler emission average higher than the concentration found in the feed coal. This is caused by figures (a) and (b) having been computed from data with EMFs limited to a maximum of 1.0.

Figure 10-5a. Average emissions

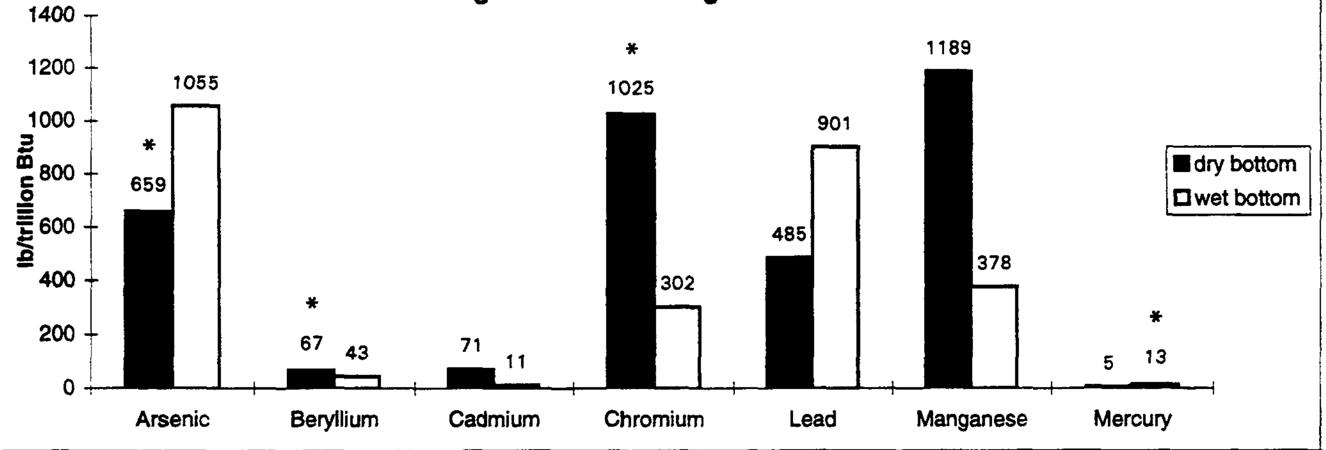

Figure 10-5b. Average trace metal removal by boiler

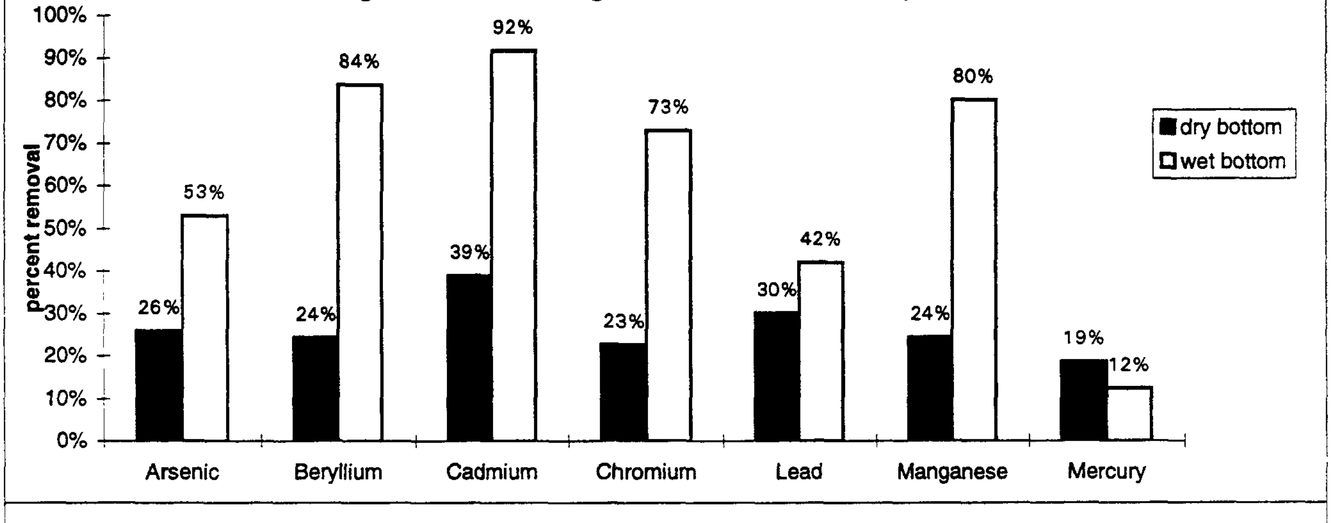

Figure 10-5c. Average trace metal concentration in feed coal

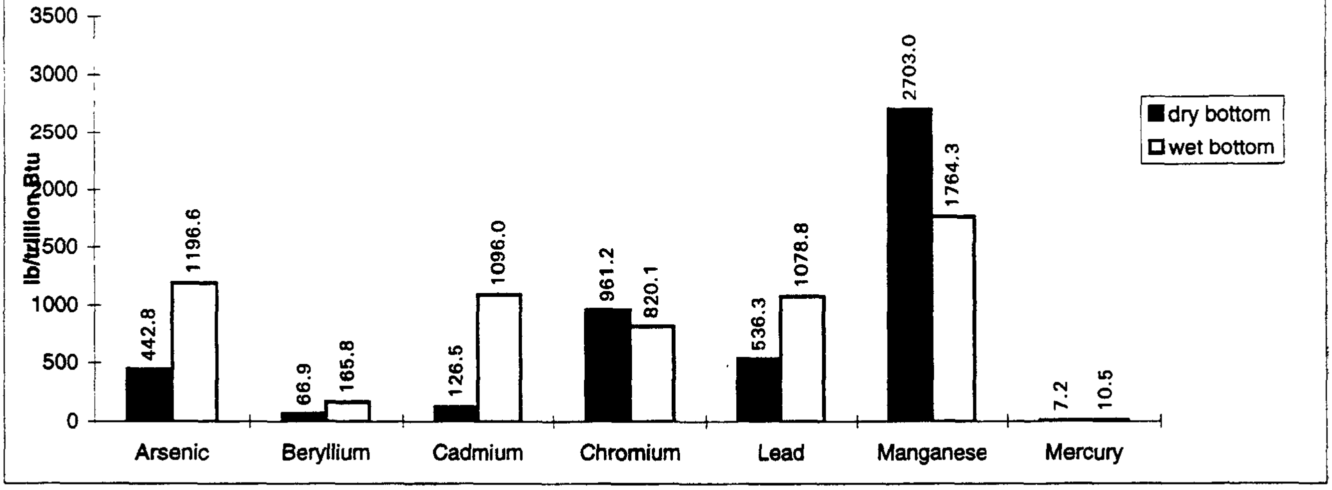

Figure 10-5 (a-c). Average coal-fired emissions, average trace metal removal, and average trace element concentration in feed coal vs. bottom type (bituminous coal-fired only)

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* Denotes a boiler emission average higher than the concentration found in the feed coal. This is caused by figures (a) and (b) having been computed from data with EMFs limited to a maximum of 1.0.

Figure 10-6a. Average emissions

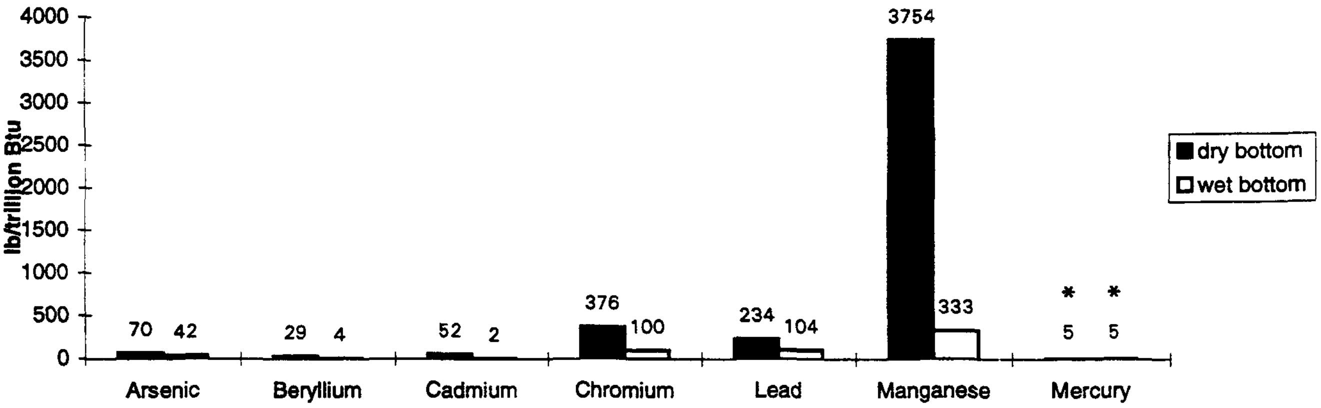

Figure 10-6b. Average trace metal removal by boiler

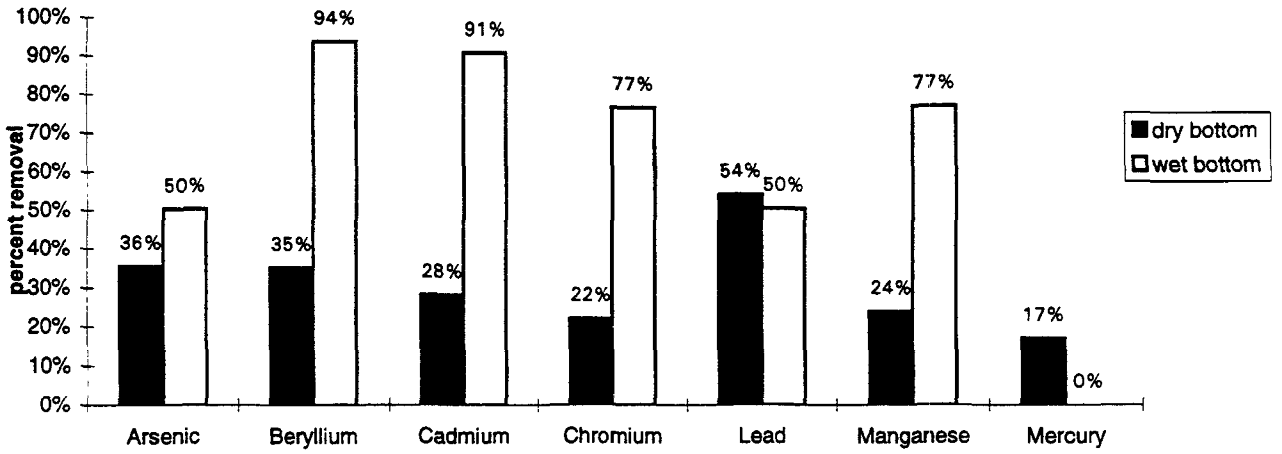

Figure 10-6c. Average trace metal concentration in feed coal

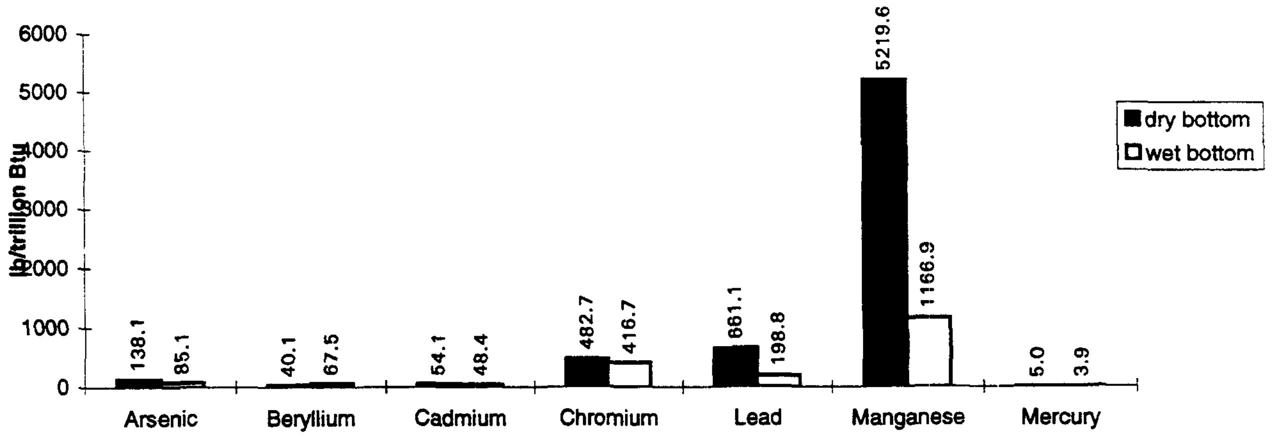

Figure 10-6 (a-c). Average coal-fired emissions, average trace metal removal, and average trace element concentration in feed coal vs. bottom type (subbituminous coal-fired only)

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* Denotes a boiler emission average higher than the concentration found in the feed coal. This is caused by figures (a) and (b) having been computed from data with EMFs limited to a maximum of 1.0.

coal show a logical relationship between the trace metal concentration observed in the feed coal and the amount emitted from the boiler. This relationship appears to be a consequence of the ash and trace metal partitioning in the boiler.

Based on this analysis, wet bottom furnaces seem to have better trace metal removal than dry bottom furnaces. Trace metal removal in a furnace is likely due to the partioning mentioned above. A possible explanation for the effect would be that bottom ash in a wet bottom furnace is kept in a molten state, and, thus, the trace metals in the bottom ash are less likely to reentrain into the fly ash. It needs to be emphasized that these analyses are based on limited data and may not hold true for all units and coals. More data and analyses are needed.

It needs to be noted that the averages in Figures 10-4(a and b), 10-5(a and b), and 10-6(a and b) were computed from data with EMFs limited to a maximum of 1.0, meaning that no more HAP could exit a device than entered it. All of the data used in these figures had at least one instance in which an EMF of 1.0 was used. This did not occur in Figures 10-4(c), 10-5(c), or 10-6(c) because these data were taken directly from the coal feed without modification. The result of this is several sets of data averages where more HAP is emitted than was present in the feed coal. These data averages were composed of a large number of EMFs of 1.0.

10.3 POSTCOMBUSTION CONTROL

The following sections assess how different APCDs affect removal of selected HAPs from fossil-fuel-fired electric utility flue gas.

10.3.1 Particulate Phase Controls

Figures 10-7 through 10-14 and Tables 10-6 through 10-9 show the relationship between the HAP metal removal and PM collection efficiency of different particulate controls (namely ESPs and FFs). The HAP removal effectiveness is shown in the tables in this section as percent removal. Percent removal is equivalent to 1 minus the EMF (see chapter 3, section 3.4). A 90 percent removal indicates that 90 percent of that HAP has been collected by a PM control device. The HAP metals that exist primarily in

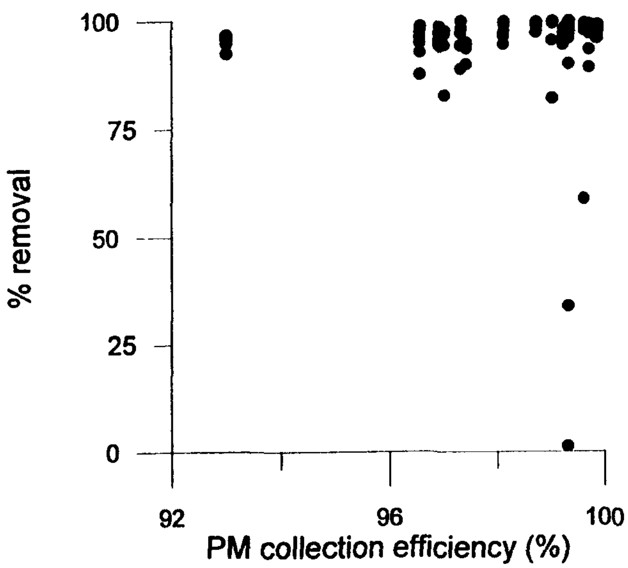

Figure 10-7. Removal of metallic HAPs by electrostatic precipitators (cold-side, coal) (includes, arsenic, beryllium, cadmium, chromium, lead, and manganese).

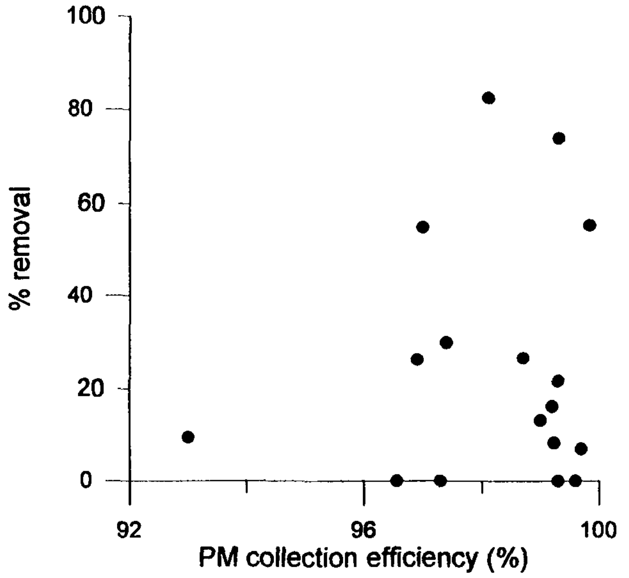

Figure 10-8 Removal of mercury by electrostatic precipitators (cold-side, coal).

	Mean	Median	SDev	Min	Max	Count
Arsenic	97	99	2	94	> 99	17
Beryllium	98	98	2.6	90	> 99	17
Cadmium	85	94	24	1.3	98	17
Chromium	97	97	2.3	93	> 99	17
Lead	93	97	15	34	> 99	17
Manganese	98	98	1.5	95	> 99	17
Mercury	25	16	26	0	82	17

15, 17, 18, 20-22, 24-26, 33-37

Table 10-6. Descriptive statistics for removal efficiencies shown in Figures 10-7 and 10-8

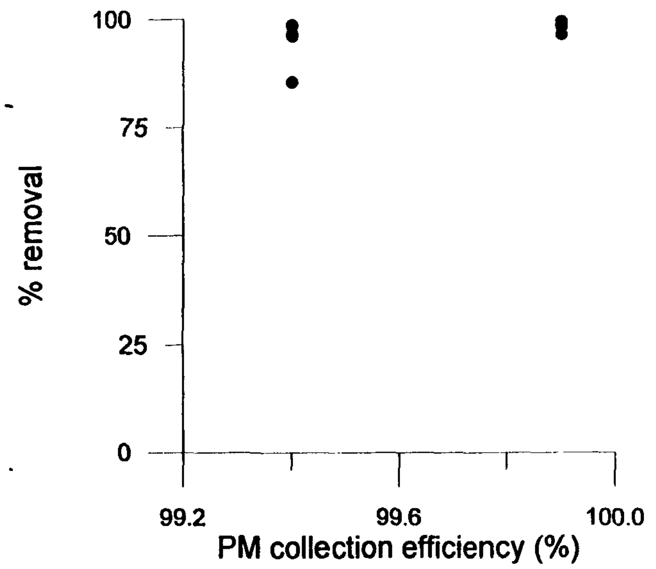

Figure 10-9. Removal of metallic HAPs by electrostatic precipitators (hot-side, coal) (includes, arsenic, beryllium, cadmium, chromium, lead, and manganese)

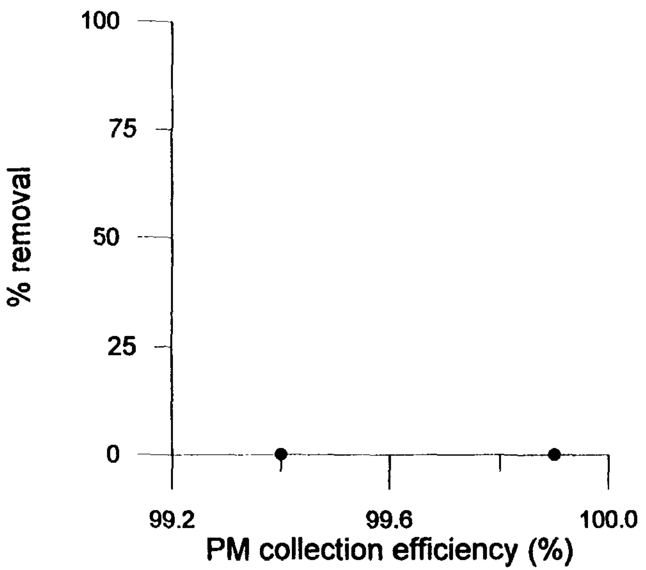

Figure 10-10. Removal of mercury by electrostatic precipitators (hot-side, coal).

	Mean	Median	SDev	Min	Max	Count
Arsenic	92	92	10	85	99	2
Beryllium	99	99	0.1	99	99	2
Cadmium	99	99	0.7	99	> 99	2
Chromium	97	97	1.6	96	98	2
Lead	97	97	1.2	97	98	2
Manganese	97	97	1.5	96	99	2
Mercury	0	0	0	0	0	2

Table 10-7. Descriptive statistics for removal efficiencies shown in Figures 10-9 and 10-10[14]

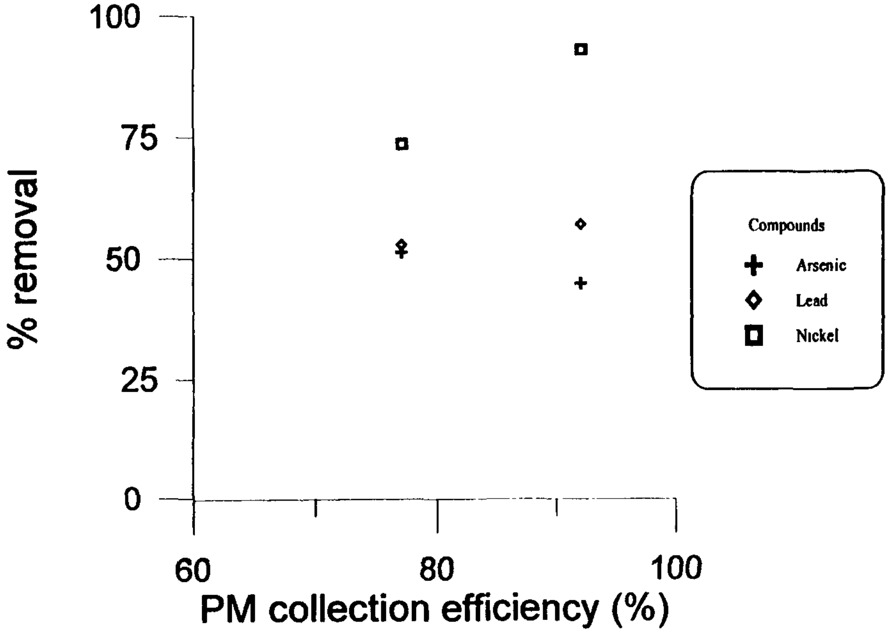

Figure 10-11 Removal of metallic HAPs by an electrostatic precipitator (oil) (includes, arsenic, lead, and nickel).

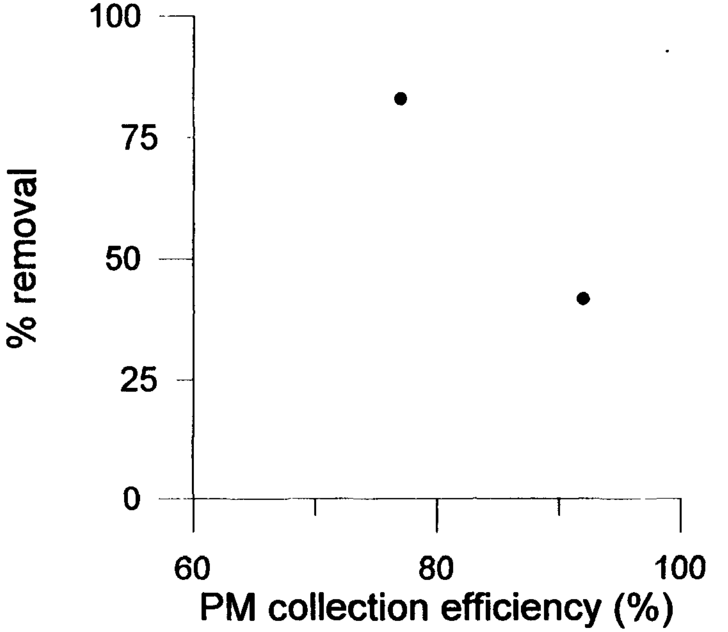

Figure 10-12. Removal of mercury by an electrostatic precipitator (oil).

	Mean	Median	SDev	Min	Max	Count
Arsenic	48	48	4.6	45	51	2
Lead	55	55	2.9	53	57	2
Mercury	62	62	29	42	83	2
Nickel	83	83	14	73	93	2

Table 10-8. Discriptive statistics for removal efficiencies shown in Figures 10-11 and 10-12[38, 39]

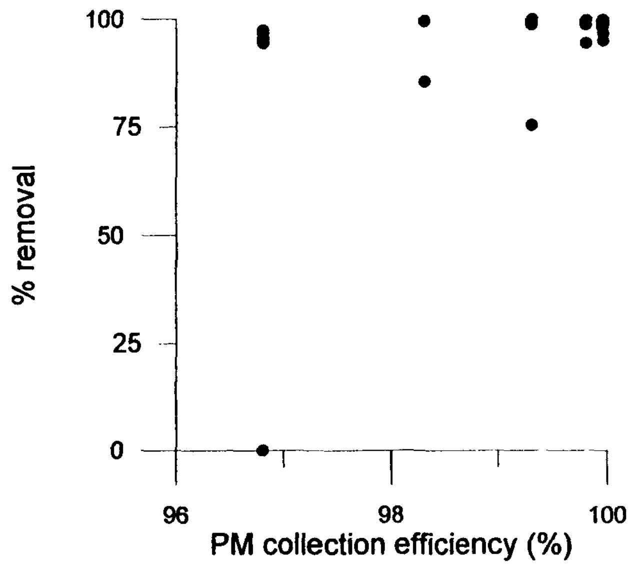

Figure 10-13. Removal of metalic HAPs by a fabric filter (coal) (includes, arsenic, beryllium, cadmium, chromium, lead, and manganese).

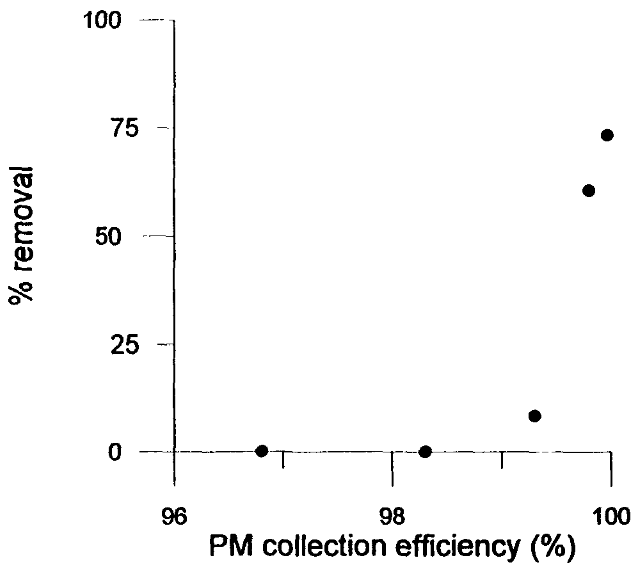

Figure 10-14. Removal of mercury by a fabric filter (coal).

	Mean	Median	SDev	Min	Max	Count
Arsenic	99	99	1.3	97	> 99	6
Beryllium	99	> 99	2.4	94	> 99	5
Cadmium	72	95	48	0	99	4
Chromium	94	99	10	75	> 99	5
Lead	99	99	1.2	97	> 99	6
Manganese	96	99	5.6	85	> 99	6
Mercury	28	8.4	36	0	73	5

16, 19, 28-30, 40

Table 10-9. Descriptive statistics for removal efficiencies shown in Figures 10-13 and 10-14 particulate form are readily controlled by PM control devices. These HAPs include arsenic, beryllium, cadmium, chromium, lead, and manganese. Table 10-10 shows the percentage of data, for all listed HAPs other than mercury, with a control device HAP removal efficiency greater than 90 percent. For example, 90 percent of the particulate from metallic HAPs data points for cold-side ESPs fall into the 90 percent or better removal category. For the two oil-fired sites for which ESP removal data were available, the control of particulate metallic HAPs was not clear. It should be noted that the concentrations of metallic HAPs in oil, with the exception of nickel, which is not discussed here, are significantly lower than those in coal, and the fuel-ash characteristics are also quite different. These factors could explain the spread of PM HAP removal of 51 to 93 percent.

Mercury, however, is not well controlled by PM APCDs. This situation would be expected because mercury is emitted as a mixture of solid and gaseous forms.

Dioxin removal in utility boiler PM control equipment has been measured at one coal-fired boiler and one oil-fired boiler. In both cases, measurements contained many values of uncertain accuracy. However, removal efficiencies could be estimated for one dioxin and three furans at the coal-fired boiler. This unit was a 615-MWe boiler firing Pennsylvania bituminous coal that had an ESP with an overall PM efficiency of 99 percent. The ESP's apparent efficiency for the following compounds was:

• 1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin	7 percent
• 2,3,7,8-tetrachlorodibenzofuran	38 percent
• 2,3,4,6,7,8-hexachlorodibenzofuran	35 percent
• 1,2,3,4,6,7,8-heptachlorodibenzofuran	29 percent.

The oil-fired boiler was an 850-MWe unit firing residual oil and had an ESP with an overall PM collection efficiency of 92 percent. All of the measurements for this site indicated negative efficiencies for dioxins and furans in the ESP.

Under certain conditions, dioxins may be formed in particulate control equipment. Studies of MWC systems have shown that dioxins and furans can be formed in particulate-laden flue

Table 10-10. Particulate Metallic HAP Removal Percentage from ESPs and FFs (Excluding Mercury)

Particulate Control Device (Coal)	Number of data points	Percentage of data with a HAP removal efficiency greater than 90 percent
ESP (cold-side)	102	90
ESP (hot-side)	12	92
Fabric Filter	32	91

gases in a temperature range of 480° to 1,020° F (250° to 550° C).[41-43] Units that are equipped with hot-side ESPs (ESPs upstream of an FGD unit) are of particular concern with regard to this formation mechanism since their operating temperatures typically fall within this range. Little information is available regarding dioxin formation in utility particulate removal equipment, however, and additional information is needed to adequately quantify the potential for dioxin formation in utility pollution control systems. Thus, at utility plants, any strategy for dioxin control must consider adequately treating large volumes of gas in order to remove relatively small concentrations of dioxin.

10.3.2 Vapor Phase Controls

Figures 10-15 through 10-18 and Tables 10-11 and 10-12 show the relationship between HAP metal removal and the inlet temperature for SO_2 control devices. The correlation between FGD scrubber inlet temperature and HAP metal removal is difficult to determine. This difficulty is compounded by having a maximum of five data sites at which two of the five test sites employ flue gas bypass in their design. A bypass means that part of the flue gas is diverted around the FGD or SDA/FF while the majority of the flue gas is treated by these control devices. Bypass is used to minimize the size and the lime/limestone cost of the FGD unit while still meeting SO_2 emission limits. Another factor is that FGDs usually follow an ESP or an FF; thus, the concentrations of metallic HAPs that reach the FGDs are generally less than 10 percent of the amounts intercepted by primary PM control devices.

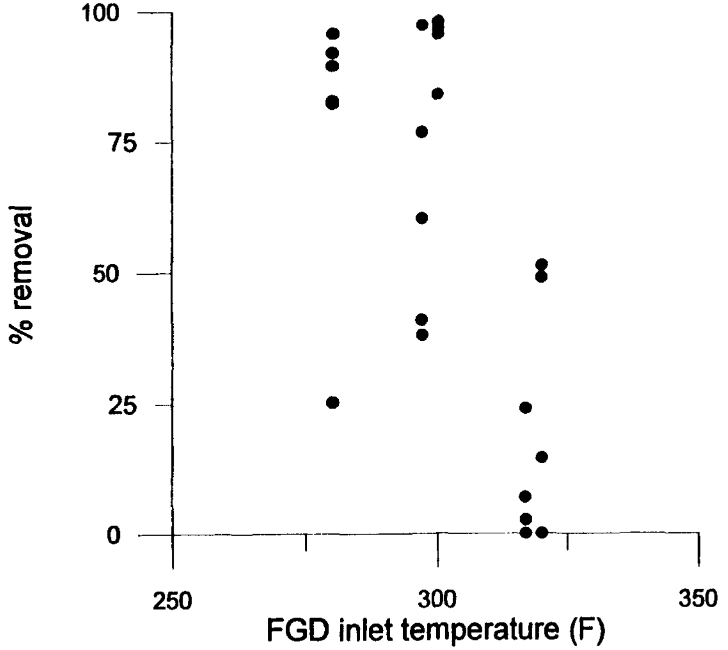

Figure 10-15. Removal of metallic HAPs by an FGD (coal) (includes, arsenic, beryllium, cadmium, chromium, lead, and manganese).

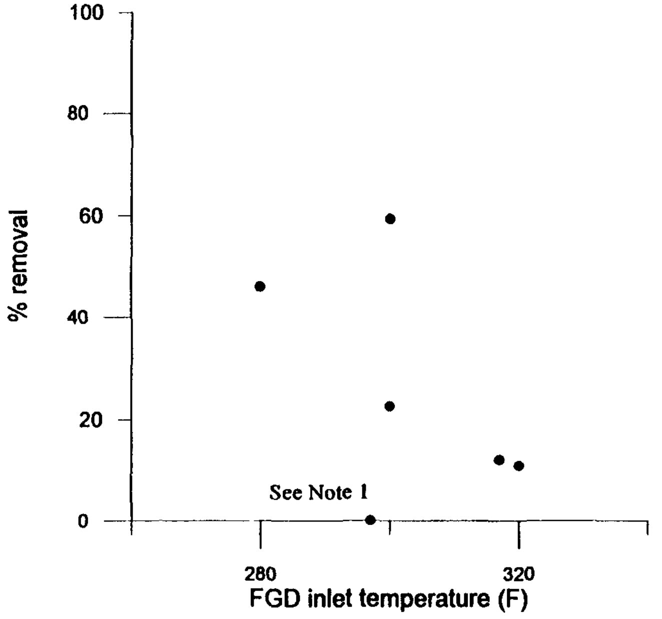

Figure 10-16. Removal of mercury by an FGD (coal).

Note 1 - This unit (EPRI Site 12) was retested for mercury, but was tested as a combined ESP/FGD system. Since there was no way of determining which component was responsible for the mercury removal, the ESP was given the full credit for removal This explains the "zero" data point in Figure 10-16 (The ESP for this site was given an "82" percent removal)

	Mean	Median	SDev	Min	Max	Count
Arsenic	68	77	30	24	96	5
Beryllium	63	89	53	2.6	98	3
Cadmium	40	32	30	15	84	4
Chromium	58	60	37	0	97	5
Lead	48	41	47	0	98	5
Manganese	45	41	52	0	97	4
Mercury	25	17	23	0	59	6

Table 10-11. Descriptive statistics for removal efficiencies shown in Figures 10-15 and 10-16[22, 24, 26, 37, 44, 45]

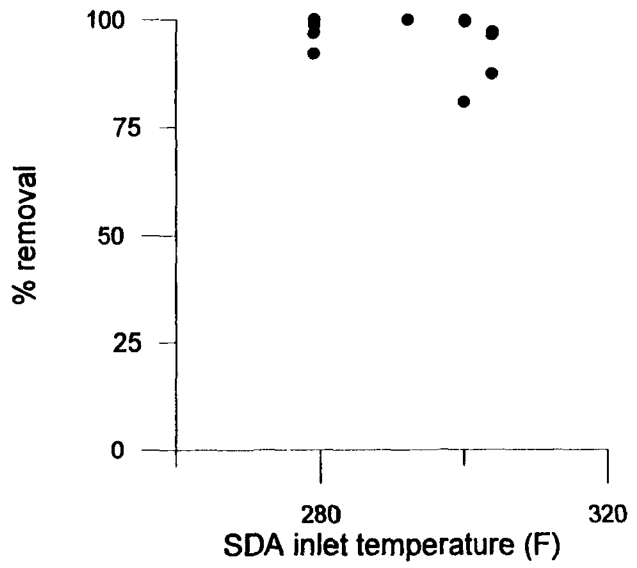

Figure 10-17. Removal of metallic HAPs by a spray dryer adsorber/fabric filter (coal) (includes, arsenic, beryllium, cadmium, chromium, lead, and manganese).

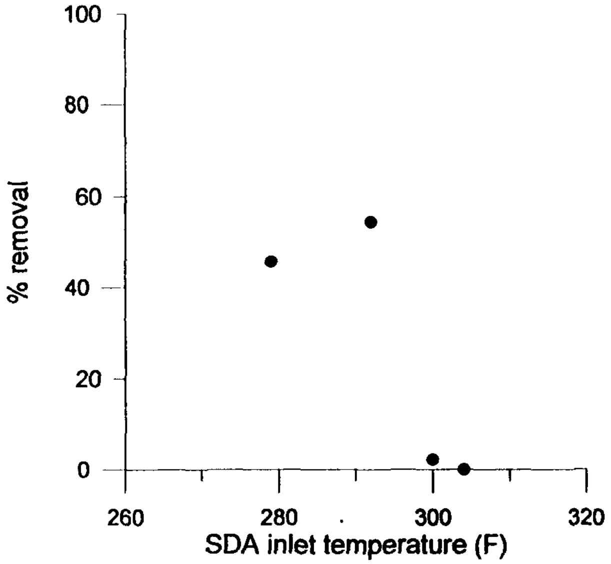

Figure 10-18. Removal of mercury by a spray dryer adsorber/fabric filter (coal).

	Mean	Median	SDev	Min	Max	Count
Arsenic	99	> 99	1.8	96	> 99	4
Beryllium	> 99	> 99	0.3	99	> 99	3
Cadmium	90	90	8.2	81	> 99	4
Chromium	98	97	1.5	97	> 99	3
Lead	99	> 99	0.7	99	> 99	3
Manganese	> 99	> 99	0.1	> 99	> 99	3
Mercury	26	24	29	0	55	4

Table 10-12. Descriptive statistics for removal efficiencies shown in Figures 10-17 and 10-18[23, 31, 32, 44]

The HAP metal removal by SDA/FF-equipped units seems to follow the results found in FF PM-controlled units, i.e., metallic HAPs are controlled to approximately 90 percent or better. This situation follows because an SDA/FF contains an FF.

10.3.3 Acid Gas Control

There was a limited amount of data (using EPA Method 26a) available on the removal efficiencies for HCl and HF from air pollution control devices. Since utilities operate with varying amounts of bypass, removal percentages for FGDs and SDA/FFs were estimated for operations with, respectively, 17 percent and 14 percent bypass. The test report data indicated that: ESPs removed less than 6 percent of the acid gases; FFs removed approximately 44 percent of the HCl and essentially none of the HF; an FGD with 17 percent bypass was estimated to remove approximately 79 percent of the HCl and approximately 28 percent of the HF; and an SDA/FF with 14 percent bypass was estimated to remove approximately 82 percent of the acid gases.[45] Despite the inconsistencies in removal efficiencies achieved for HCl and HF with FFs and FGDs, the data indicate that the SO_2 control devices remove more of the acid gases than do PM controls.

10.3.4 Carbon Adsorption

A possible way of further reducing the amount of vapor phase HAPs emitted from utilities is through the use of carbon adsorption. Activated carbon (AC) 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 AC that contains multiple internal pores and has a very high specific surface area. With this internal pore structure, the AC can adsorb a broad range of contaminants.

Activated carbon can be introduced through the use of a fixed-carbon filter bed, a moving bed, a fluidized carbon filter bed, or through direct AC injection into the flue gas stream.

Other than for mercury (see section 10.6.1.2), no utility data were found for AC removal of HAPs. However, for other industries (e.g., MWCs and medical waste incinerators [MWIs]), dioxin removal from the flue gas stream has also been achieved by AC injection.

10.4 ALTERNATIVE CONTROLS

An alternative to pollution control systems applied directly to boiler systems is to use alternative, nontraditional control methods such as demand side management (DSM) and energy conservation. Demand side management addresses the issue of reducing potential HAP emissions by reducing the amount of electricity needed. This objective can be accomplished by several different methods.

Through the use of progressively higher efficiency electrical devices (e.g., more efficient home appliances, lighting systems, and industrial machinery), the overall tonnage of fossil fuel burned would be reduced.[47] Also, campaigns to reduce the use of fossil-fuel-generated electricity have a positive effect on reducing HAPs by slowing down the necessity of building additional plants. Research into more efficient electrical generators and transmission equipment could produce units that deliver the same amount of energy for less fuel.[48] Future potential electrical transmission equipment could include the development and use of superconductive power lines, which substantially reduce the amount of electricity that needs to be generated to move the high voltage electricity through power lines because of the negligible power loss (due to lower resistance).[49] A final method would be to switch to a source of renewable energy (e.g., wind, solar, biomass-firing), but to date these methods have been limited to certain geographic locations only. (However, such campaigns may also lead to slower introduction of new, more efficient fossil fuel-burning technologies.)

Another potential fuel option is the use of liquid or gaseous fuels derived from biomass. Currently, fuels such as ethanol and methanol, derived from corn, grains, and other crops, are being used to produce "gasohol" (a blend of up to 10 percent ethanol in gasoline). Future use of these fuels by utilities, as well as hydrogen fuels derived from biomass, could reduce (but perhaps not eliminate) HAP emissions. However, DOE has estimated that development of technologies to produce sufficient quantities of biomass-derived fuels may not be commercially viable until 2030.[50]

10.5 POLLUTANT TRADEOFFS

10.5.1 HAP Increase/Decrease

The various strategies for limiting HAP emissions, discussed in sections 10.1 through 10.4, have different effects in controlling air emissions of all HAPs. Table 10-13 presents the qualitative effects of the different control strategies on air emissions.

Table 10-13 provides a comparison of HAP removal effectiveness of different existing and alternative control strategies. As shown in Table 10-13, the effect on HAP emissions of: (1) switching from a higher- to a lower-sulfur coal, (2) NO_x controls, and (3) boiler types cannot be predicted from the available data. Techniques that would reduce emissions of the HAPs of concern include: (1) switching from coal or oil to natural gas; (2) coal gasification; and (3) alternative controls, such as energy conservation or DSM. The remaining strategies control certain HAPs. Existing PM control devices, such as ESPs and FFs, generally do not remove the vapor-phase HAPs (i.e., organics, elemental mercury, HCl, and HF). (However, these controls do provide some reduction of ionic mercury emissions.) Emissions of the vapor-phase HAP, dioxin, are not controlled by ESPs but are controlled to some extent with FFs because dioxins adsorb onto the filter cake. As noted in section 10.3.1, there is the potential for dioxins to be produced in hot-side ESPs due to the temperature and catalytic effects of the fly ash. With the exception of elemental mercury emissions, the existing SO_2 control devices, namely FGDs and SDA/FFs, tend to reduce emissions of the HAPs and provide some reduction of emissions of ionic mercury.

Emissions of HAPs could be reduced through energy conservation or DSM. Finally, pilot-scale studies suggest that ionic and elemental mercury emissions could be controlled, respectively, with unimpregnated- and impregnated-carbon adsorption, but full-scale testing is needed to establish the effectiveness of these techniques.

10.5.2 Water/Solid Waste Considerations

Coal cleaning can produce a variety of waste problems.[51] The process creates a liquid waste containing fine coal particles and inorganic elements and compounds dissolved from the ash in

Table 10-13. Qualitative Effects of Different Control Strategies on Air Emissions of HAPs

Type of alternative control strategy	Effect on particulate HAPs		Effect on mercury		Effect on acid gases (HCl, HF)	Effect on dioxin
	Primarily organically bound	Primarily inorganically bound	Primarily oxidized mercury	Primarily elemental mercury		
PRECOMBUSTION CONTROL						
Conventional coal cleaning	No removal	Decrease in emission	Decrease in emission (highly coal-specific)		Decrease in emission if chloride or fluoride is reduced	
Coal gasification	Decrease in emission		Cannot predict the effect		Decrease in emission	
Fuel switching						
From a higher to lower sulfur coal	*Cannot predict the effect*		Cannot predict the effect		Cannot predict the effect	
To western and certain eastern coals	Cannot predict the effect		Cannot predict the effect		Cannot predict the effect	
Coal or oil to gas	Decrease in emission		Decrease in emission		Decrease in emission	
COMBUSTION CONTROLS						
NO$_x$ controls	Cannot be determined without further testing					
Boiler type	Cannot be determined without further testing					
POSTCOMBUSTION CONTROLS						
Particulate phase controls						
ESP	Decrease in emission		Some decrease in emission	No effect	No effect[a]	
FF	Decrease in emission		*Some decrease in emission*	No effect	No effect	Decrease in emission because of filter cake adsorption
Vapor phase controls						
FGD	Decrease in emission		Decrease in emission	No effect	Decrease in emission	
SDA/FF	Decrease in emission		Some decrease in emission	No effect	Decrease in emission	
Carbon adsorption	Decrease in emission. (Elemental and ionic mercury were removed, respectively, by impregnated and unimpregnated carbon adsorption, in pilot-scale studies.)					
NON-TECHNOLOGY-BASED CONTROL OPTIONS						
Demand management	Decrease in emission					

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[a] There is the potential for dioxins to be produced in hot-side ESPs due to the temperature and catalytic effects of the fly ash.

the coal. The large volumes of water used in the cleaning process and the large amount of suspended solids generated dictate that process water be clarified and recycled. The usual means of clarification is retention in large sedimentation ponds. Contamination of surface water or ground water can occur from coal wastepiles or storage piles if water is allowed to infiltrate them. Contaminants such as iron and manganese and heavy metals such as cadmium and silver may leach from the wastes. Because processing wastes are higher in ash (material) content than in the cleaned coal, leachate from waste poses a greater threat.[51]

Precombustion controls such as fuel switching could have an effect on reducing this waste. If coal with lower amounts of ash and sulfur was mined, there would be less need to clean the coal and therefore fewer tailings would be created at the coal washing site. Switching from coal to natural gas would reduce the need to mine and wash the coal.

In either case, the amount of bottom ash and captured fly ash is quite large. Because the metals are not destroyed in the combustion process, the ash will have a higher concentration of metals than the coal, and water contamination may occur if water runoff from the ash is not controlled. The amount of metals in ash disposal pits is not likely to increase significantly if particulate controls are already in place. If new controls are installed, larger amounts of ash will require disposal, leading to an increase in the potential for water contamination from ash leachate.

Coal gasification would not necessarily reduce the waste potential of coal, but the processing of coal into a gasified form would tend to localize the waste and transfer it from a gaseous stream to a solid stream that could be more easily disposed of after being properly treated.[10] For example, gasifier residue may contain contaminants in a nearly vitreous matrix that is not easily leached. In conventional coal combustion, these contaminants would largely appear in the flue gas stream.

Combustion controls, such as different boiler/furnace types or the adding of NO_x controls, would have the effect of changing combustion conditions in the coal-fired furnace and, thus, changing the ratio of bottom ash to fly ash. Increasing the amount of bottom ash in the furnace would consequently reduce the amount of fly ash exiting the furnace. However, if the amount of fly ash exiting the furnace was increased, there would be greater fly ash loading on the PM controls. If the PM controls were not designed to accept this additional load, excess fly ash could cause stack opacity problems and higher HAP emissions.

The wet FGD processes use a liquid absorbent to absorb SO_2 gases. This absorbent is most likely an alkaline slurry composed of lime or limestone slurried with water that can be used in a regenerable process or in a nonregenerable process. Both processes produce a calcium sludge effluent that must be disposed of properly. The sludge can be stored in a settling pond or dewatered to take up less space in a landfill.[52] In addition to removing sulfur oxides, regenerable processes generate a usable product from the sludge, such as gypsum, that can be used in the manufacture of wallboard.

In a dry FGD system, the flue gas is contacted with an alkaline material to produce a dry waste product for disposal. The alkaline material can be added either in the fuel prior to combustion, through dry injection into the flue gas, or as an alkaline slurry.[53] For example, SDAs inject a lime/limestone alkaline slurry into the flue gas steam. The reagent droplets absorb SO_2 while simultaneously being dried. In all these methods, the resulting dried reagent and fly ash are then captured by an FF or ESP and, thus, do not represent an additional liquid or solid waste problem except that the sorbent may contaminate salable fly ash.

The use of all forms of carbon adsorption have the potential to add additional PM loading to existing PM controls. Also, the spent AC either has to be disposed of as an additional solid waste or regenerated and reused. Studies on other industries indicate that the adsorbed HAP (e.g., mercury, dioxin) does not have a tendency to leach out of the AC.

Section 3001 (b)(3)(c) of the Resource Conservation and Recovery Act (RCRA) required that the EPA determine, based on the results of a study required by section 8002(n) of RCRA, whether RCRA subtitle C regulation of fossil fuel combustion wastes is warranted. On August 9, 1993, the Agency determined that regulation of four large-volume waste streams resulting from fossil fuel combustion (specifically, coal combustion in utility steam-generating units) was inappropriate. These waste steams are fly ash, bottom ash, boiler slag, and flue gas emission control waste. However, any change in the characteristics of these wastes resulting from actions taken to specifically control HAPs could necessitate a review of that decision by the Agency.[54]

All elements of the alternative controls, from conservation to technology improvements, can reduce the amount of waste produced by the utility industry.

10.6 AVAILABLE CONTROL TECHNOLOGY AND STRATEGIES FOR MERCURY CONTROL

Typical mercury removal efficiencies for conventional emission controls are discussed in section 10.3. Conventional controls are generally inconsistent in their effectiveness and range from 0 to more than 80 percent removal.

There has been some evidence that a higher carbon content in fly ash may lead to lower levels of stack mercury emissions, with the hypothesis being that the mercury is adsorbed by the carbon in the flue gases. This hypothesis comes from data on MWCs, particularly from units burning refuse-derived fuel, which tend to have greater carbon carryover and a larger drop in mercury concentrations across particulate control devices than do mass burn systems, which tend to have lower carbon levels in the fly ash. Because combustion modification NO_x controls can lead to increased carbon in the fly ash of coal combustion units, some level of mercury capture may take place in units with combustion control for NO_x through a similar mechanism. This has not been verified for the conditions occurring in utility units, however.

Strategies for reducing mercury emissions from electric power generation include demand reduction to decrease overall fossil fuel use, use of other forms of generation (such as nuclear power), switching to fuels having less mercury, and adding controls that remove mercury more effectively than conventional controls. The mercury content in utility flue gas ranges from 0.2 μg/dry standard cubic meter (μg/dscm) to 21 μg/dscm at 7 percent oxygen and standard conditions of 20° C and 1 atmosphere, while utility flue gas flow rates may range from 400,000 to 4,000,000 dscm/h.[55] Thus, at utility plants, any strategy for mercury control must consider adequately treating large volumes of gas in order to remove relatively small concentrations of mercury as well as addressing any resulting impacts on power plant equipment operations (such as particulate control devices) and on waste disposal issues.

The rest of this section describes the impact of process variables on mercury removal in existing systems and then describes a potential new control, mercury removal by adsorption onto AC by various methods. Other potential technologies are also briefly described.

10.6.1 Impact of Fuels and Temperature on Mercury Emissions

Fuels, their sources and precombustion treatment, and the way they are burned can have a significant impact on the quantity of mercury emissions in the flue gas from a boiler.

10.6.1.1 Fuels and Mercury Speciation. Mercury is contained in the coal and oil fuels burned in utility boilers. During combustion, mercury readily volatilizes from the fuel and is found predominantly in the vapor phase in the flue gas[56] in one of three forms: (1) elemental, (2) ionic, or (3) organic. It is important to understand mercury speciation because it will indicate potential ways to reduce mercury emissions. For example, the ionic mercury form (i.e., Hg^{++}) is water soluble and less volatile than elemental mercury (i.e., Hg^0). Thus, reducing the temperature of the flue gas or wet scrubbing of the flue gas may result in increased ionic mercury removal. The speciation information is also needed to understand what is emitted from utility stacks, how it is affected by atmospheric chemistry, and the subsequent deposition of mercury.

The distribution of ionic mercury, most likely mercuric chloride ($HgCl_2$), in coal-fired utility flue gas ranged from 12 to 99 percent of the total mercury content and averaged 79 percent, while 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 suggests that mercury is predominantly in the elemental form when the fuel is oil. 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;[57] however, validated methods are needed to establish the speciation of mercury before any relationship between mercury speciation and control device performance can be inferred.

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), but more data are needed to verify this association.[58-60] 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).[61,62] The association between fuel chloride content and the concentration of ionic mercury in the flue gas may also apply to fuel oil, but this association has not been examined.

The speciation of mercury is important in planning control strategies but is still in the early stages of investigation. Studies of a pilot-scale wet FGD system treating coal-fired flue gas indicate that more than 90 percent of the ionic mercury was removed while hardly any of the elemental mercury was removed.[60] 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.[59]

10.6.1.1.1 Coal cleaning. As mentioned in section 10.2.2, conventional coal cleaning may be effective for reducing mercury concentrations only in specific coals and, at this time, cannot be considered a mercury control technique for all coals.[56] Advanced coal cleaning techniques are being investigated for improved trace metal removal potential.

10.6.1.1.2 Coal switching. As mentioned in section 10.1.1.3, switching to certain higher sulfur coals (above 2.5 lb/MMBtu) and selected lower sulfur coals (below 1.5 lb/MMBtu), which contain less than 15 lb mercury/trillion Btu, could reduce mercury emissions from utility units. However, coal blending for mercury control is not a proven control strategy. Changes in the electrical resistivity and amount of fly ash resulting from coal blending could reduce PM capture efficiencies by ESPs and subsequently lead to increased emissions of PM and HAP metals. Blending for mercury control could also increase levels of other HAPs or sulfur. Another uncertainty with coal blending for mercury control is the possibility of blending coals that would result in different species of mercury, and changes in the amount of vapor- and particulate-phase species of mercury would affect mercury control with PM control devices.

10.6.1.2 Temperature. Utility flue gas typically has a temperature range of 121° to 177° C (250° to 350° F) after leaving an air preheater, which is a heat exchanger commonly used to heat incoming combustion air.[55] Mercury is found predominantly in the vapor phase in utility flue gas.[56] If the vapor-phase mercury was condensed onto PM, the PM could be removed with existing PM control devices. Theoretically, cooler temperatures will give relatively more mercury condensation onto PM[56] and, subsequently, increased mercury removal with existing PM control devices. There is limited, preliminary evidence for the temperature dependence of mercury removal in a pilot-scale FF study. The pilot study suggests that mercury removal efficiencies apparently increase as the temperature of the flue gas decreases. Specifically, as the flue gas temperature decreased from 107° C (225° F) to 99° C (210° F) to 96° C (205° F), the mercury removal percentages for a pilot-scale FF correspondingly increased from 27 percent to 33 percent to 51 percent.[63] However, mercury control is not entirely temperature dependent – FGD systems have exhibited no mercury removal (see Figure 10-17) with outlet temperatures below 107° C (225° F).

10.6.2 Developing Technologies: Activated Carbon

Research has been performed on AC injection at a pilot-scale SDA/FF plant[59] at a pilot-scale pulse-jet FF (by EPRI)[63] and at a pilot combustor and FF (by the University of North Dakota Energy and Environmental Research Center [UNDEERC] and EPRI).[61]

10.6.2.1 Factors Affecting Mercury Removal Efficiency. Preliminary data from two studies suggest that factors besides the optimum amount of AC that is injected may affect mercury removal. These factors are temperature, the form of the vapor-phase mercury, and the type of activated carbon[61] injected into the flue gas.

10.6.2.1.1 Temperature. Mercury removal efficiencies and the required amount of AC injection were apparently temperature dependent within a range of 88° to 121° C (190° to 250° F) in a pilot-scale study conducted on reducing mercury levels in utility flue gas through AC injection upstream of an FF.[63] At lower temperatures, 88° to 96° C (190° to 205° F), mercury concentrations were reduced by 97.7 percent with an AC injection rate of approximately 155 μg carbon/μg of inlet mercury. At higher temperatures, 110° to 121° C (230° to 250° F) mercury concentrations were reduced by only 75 to 87 percent with an AC injection rate of approximately 3,500 μg carbon/μg of inlet mercury.

These studies suggest that more mercury is removed and less carbon needed at lower flue gas temperatures. However, it may not be possible to lower the flue gas temperature sufficiently at a given utility plant because utility plants typically operate with a stack gas temperature between 121° and 177° C (250° and 351° F) upstream of any PM control device to avoid acid condensation and, consequently, equipment corrosion. The stack gas temperature may be lowered below 96° C (205° F) and still avoid acid condensation, provided low-sulfur coals (less than about 1 weight percent sulfur) are burned.[64] If the utility burns low-sulfur coal and uses an ESP for PM control, the flue gas will probably require conditioning to reduce the high resistivity of the fly ash since high resistivity makes the fly ash hard to collect in an ESP.

10.6.2.1.2 Speciation of Mercury. The effectiveness of AC injection in recovering different forms of mercury is still in the early stages of investigation. In testing at a pilot-scale SDA/ESP system in Denmark,[59] the flue gas contained from 66.6 percent to 83.4 percent ionic mercury with an average of 75.2 percent ionic mercury, and elemental mercury constituted the remainder of the total mercury concentration. Without AC 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 (i.e., the system removed 72.5 percent of the total mercury). During testing with AC 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 made up the remainder of the total mercury concentration. 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.[59] In testing at another facility that had a full- and pilot-scale SDA/FF system, the flue gas contained 85 to 90 percent elemental mercury. Without AC injection, the full- and pilot-scale SDA/FF systems removed 10 to 20 percent of the total mercury from the coal-fired flue gas,[59] and the low removal of total mercury may be attributed to essentially complete removal of the ionic mercury and little 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 AC increased the removal of total mercury to approximately 90 percent.[59] Thus, the studies at this SDA/FF system suggest that sulfur- and iodide-impregnated ACs are needed for total mercury removals of 90 percent when elemental mercury is the predominant mercury species. Furthermore, the studies suggest that total mercury removal efficiencies are dependent upon mercury speciation.

Finally, recent tests conducted at the pilot-scale combustor and FF at UNDEERC also suggest that mercury removal is dependent upon mercury speciation and the type of carbon used.[61]

Since mercury speciation affects total mercury removal from utility flue gas with AC injection and because the speciation of mercury is not understood at this time, more data are needed to establish the factors that affect, as well as to characterize, mercury speciation in utility flue gas.

10.6.2.2 Comparison of Characteristics of MWC/utility Flue Gas. The level of mercury control achieved in utility flue gas may depend upon flue gas characteristics such as temperature, chloride content (in the fuel), mercury content (in the flue gas), and the volume of flue gas. As shown in Table 10-14, these properties distinctly differ from those in MWC flue gas. In particular, mercury concentrations in MWC flue gas streams may be up to several orders of magnitude greater than those seen in utility flue gas streams.

Due to the differences between the flue gas characteristics at MWCs and utility units, the application of AC injection to utility flue gas has not been directly scaled from the application at MWCs. At utility plants, the small concentrations of mercury are contained in a large volume of flue gas, and large amounts of AC may be needed to provide adequate contact between the carbon particles and mercury. Pilot-scale studies of AC injection on utility flue gas have been conducted, but full-scale testing is needed to determine the feasibility of using AC at utility plants.

10.6.2.3 Results of Pilot-scale Carbon Injection. Figure 10-19 shows mercury removal efficiency ranges for existing control devices with AC injection at different temperatures based on limited available information.[59,63] Figure 10-19 shows that the tested SDA/FF systems with AC injection had a median mercury

Table 10-14. Comparison of Typical Uncontrolled Flue Gas Parameters at Utilities and MWCs[a]

Uncontrolled flue gas parameters	Coal-fired utility boiler[65]	Oil-fired utility boiler[64,65,66]	MWC[b] [63,67]
Temperature (°C)	121 - 177	121 - 177	177 - 299
Mercury content (μg/dscm)	1 - 25	0.2 - 2	400 - 1,400
Chloride content (μg/dscm)	1,000 - 140,000	1,000 - 3,000	200,000 - 400,000
Flow rate (dscm/min)	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.

removal efficiency of 60 percent with a range from 50 percent to 99 percent removal. (The 99 percent removal may have been obtained with an impregnated carbon, but the test report did not specify the type of carbon. The use of impregnated carbon is discussed in section 10.6.1.2.4.) Figure 10-19 also shows that the 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.

Figure 10-19 suggests that FFs with AC injection ahead of the FF have a median mercury removal efficiency that varies with temperature and AC injection rate.[63] It should be noted that the higher temperature test data, at flue gas temperatures between 107° and 121° C (225° and 250° F), had prior PM control while the lower temperature test data, at flue gas temperatures between 88° and 107° C (190° and 225° F), were obtained with and without prior PM control. The effect of the presence and absence of the prior PM control on mercury removal efficiency was not apparent in the lower temperature test data. With a low AC injection rate (<1,000 wt C/wt inlet Hg) and an average flue gas temperature between 107° and 121° C (225° and 250° F), the median mercury removal efficiency was 29 percent with a range from 14 to 47 percent removal. With a low AC injection rate (same as above) and an average flue gas temperature between 88° C (190° F) and 107° C (225° F), the median mercury removal efficiency was 97 percent with a range from 76 to 99 percent removal. A high AC injection rate (>1,000 wt C/wt inlet Hg) and an average flue gas temperature between 107° and 121° C (225° and 250° F) produced a

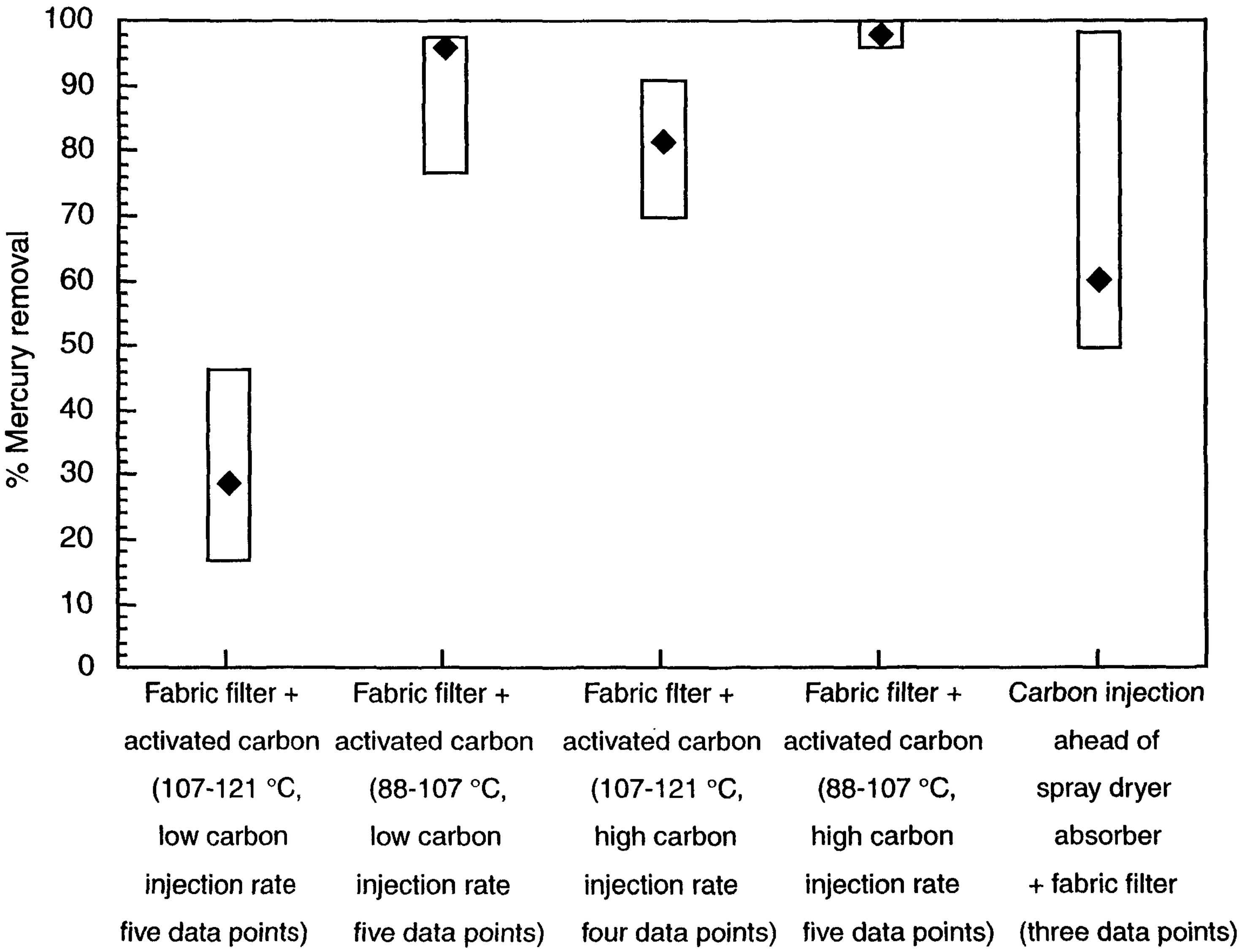

Figure 10-19. Comparison of mercury removal efficiencies with activated carbon injection.

median mercury removal efficiency of 81.5 percent with a range from 69 to 91 percent removal. A high AC injection rate (same as above) and an average flue gas temperature between 88° and 107° C (190 and 225°F) produced a median mercury removal efficiency of 98 percent with a range from 95 to 99 percent removal.

Any strategy for add-on mercury control at utility plants must consider adequately treating large volumes of gas in order to remove relatively small concentrations of mercury, as well as any resulting impacts on power plant equipment operations (such as particulate control devices) and on waste disposal issues. Later testing done on FF-equipped units using AC injection showed much more variability and generally lower collection efficiencies than those shown in Figure 10-19.[68] Thus, while AC injection shows promise as a mercury control technology, the limited results to date are inconsistent and more data and research are needed.

10.6.2.4 Emerging Technologies for Controlling Mercury Emissions from Utilities. Research continues on developing potential technologies for mercury emission reduction from utility plants. Although AC injection has been shown to be a promising technology, research with impregnated ACs, sodium sulfide (Na_2S) injection, and an AC circulating fluidized bed suggest that greater mercury removal is possible.

In sulfur-impregnated AC injection, the carbon-bound sulfur reacts with mercury to form mercuric sulfide (HgS) on the carbon and the carbon is removed by a PM 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 AC.[59]

With iodide-impregnated AC injection, the carbon-bound iodide reacts with mercury to form mercuric iodide (HgI_2) on the carbon and the carbon is removed by a PM control device. In a pilot-scale study, iodide-impregnated carbon increased mercury removal to nearly 99 percent, an increase of nearly 45 percent over results achieved with an equal amount of nonimpregnated AC.[59] While all testing to date has shown that iodide-impregnated AC injection has a substantial effect on the mercury removal capability of AC, further testing has shown that, under certain conditions (certain coal types at temperatures of 177° C [350° F] and higher), a portion of the captured mercury (postulated to be mercuric iodide) may be revolatilized as oxidized mercury.[69]

Chloride-impregnated AC injection has been tested only on MWCs in Europe. The chloride reacts with mercury to form $HgCl_2$ on the carbon and the carbon is removed by a PM control device. European MWC experiments have shown that impregnating AC with chloride salts increases the adsorptive capacity of the AC 300-fold.[67] Although the amount is small, chloride-impregnated AC injection would introduce additional chlorine (a HAP) into the flue gas stream.

Another potential method of improving mercury collection efficiency is to combine calcium hydroxide (lime, $Ca(OH)_2$) with AC. This reagent, consisting of approximately 95 to 97 percent lime and 3 to 5 percent AC, is known under the product name Sorbalit.[66] Sorbalit has only been tested on European MWCs and MWIs.

Sulfur-, iodide-, chloride salt-, and $Ca(OH)_2$-impregnated ACs show promise for increasing the mercury removal efficiency, but further testing is needed. The cost of these modified carbons can be as much as 20 times higher than that of unmodified AC.[70]

Mercury reduction has been achieved at MWCs through injection of Na_2S solution into the flue gas prior to the acid gas control device. The resulting solid, HgS, can be collected by an FF.[71] There are several potential limitations to Na_2S injection. These include reaction of Na_2S with calcium 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.[72] 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 utility units.

Another potential process for the reduction of mercury emissions is the use of AC in a CFB.[56] In a CFB, the AC 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 AC is withdrawn from the process and replaced by fresh material.[67]

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

As noted earlier, another potential technique, advanced coal cleaning, may reduce the concentrations of mercury contained in the mineral and organic phases of coal. The reliability and feasibility of advanced coal cleaning techniques in reducing the concentration of mercury in coal are unknown at this time, but are being investigated.

Additional advanced processes for controlling mercury emissions include the condensing heat exchanger, gold sorbent technology, other sorbent injection processes, advanced fine particulate control technologies, and enhancement of wet scrubbers.[73] However, data on these processes were not available for this report; data will be incorporated into the final Report to Congress as they become available.

10.7 DISCUSSION OF FEDERAL INTERAGENCY REVIEW COMMENTS

Previous drafts of chapters 1 through 10, along with the appendices, were reviewed by numerous non-EPA scientists representing industry, environmental groups, academia, and other Federal agencies during the summer of 1995. In February, April, and September 1996, all sections of the draft report underwent additional review by EPA, State and local agencies, and other Federal agencies. The EPA has revised the report, as appropriate, based on the reviewers' comments. The EPA revised the report to incorporate the majority of the comments received. However, there were several comments that could not be fully addressed because of limitations in data, methods, and resources. This section presents comments received by other Federal agencies that could not be substantially addressed in this interim report.

10.7.1 Comment

The DOE commented that the report "...understates the possibilities for air toxics control through fuel switching and coal cleaning." The DOE "...has found that coal cleaning holds significant promise for many forms of air quality control. Efforts in 2010 should be significantly more successful in emission reductions than indicated in the Report." [74]

10.7.2 Response

There is no way to tell which plants will use advanced coal cleaning technologies or which coals are going to be cleaned in the future. The current report considers HAP reductions through fuel switching. In addition, it was assumed that 100 percent of bituminous coal is currently washed using conventional coal cleaning techniques (even though only 90 percent was washed in 1993). At the time the report was being developed, there were no data available to characterize advanced coal cleaning. For subbituminous and lignite coals, to the Agency's knowledge there is currently no coal cleaning being performed and no projections for cleaning these coals in the future. Therefore, no coal washing factors were added to HAP concentrations for lignite and subbituminous coals. Data on advanced coal cleaning will be incorporated into the final Report to Congress as they become available.

10.7.3 Comment

The Office of Science and Technology Policy (OSTP) commented that "[t]he report does not adequately address the impact of demand-side management strategies though a number of utilities are pursuing this aggressively rather than building out generation capacity. The report implies that investments in alternative energy technologies may occur at the expense of investments in more efficient coal-burning technologies (a theoretical possibility but no data are presented). The report maintains that biomass will not become a viable option until the year 2030. With realistic investments in Research & Development (R&D) in both crop production and generation techniques, biomass can be competitive (without subsidies) within 5 years in niche markets (with high energy prices) and within 10 years in larger markets. [DOE and IPCC studies]." Also, OSTP adds that "...control strategies dominate pollution avoidance strategies, both in discussion and analysis." [75]

10.7.4 Response

Demand-side management's potential for reduction of HAPs is mentioned in this chapter, based on the limited data available. The EPA will review the data as they become available and provide additional discussion, as appropriate, of other alternative control strategies in the final report.

10.7.5 Comment

The DOE and the Office of Management and Budget (OMB) commented that the discussion of control options should include cost estimates.[76,77]

10.7.6 Response

As noted in Chapter 1, the EPA has not included any cost estimates in this report. However, as a part of the mercury study, the EPA did evaluate general costs of mercury control.[78]

This analysis was based on a model-plant approach and assumed that all plants within a source category would achieve the same emission reduction and incur the same costs as the model plants. The cost estimates derived, therefore, were only for relative comparisons among source categories. The cost of mercury control incurred by any specific facility may be underestimated because of the variability inherent in the assumptions that were made (e.g., mercury reduction efficiency of the various control techniques; flue gas stream mercury content; site-specific factors such as down-time and labor costs). Costs for monitoring and record keeping were not included. The costs represent retrofit application of controls; installation of controls at new facilities can be significantly less expensive than retrofitting. These costs also do not include the benefit of co-control of other pollutants in addition to mercury. In addition, the cost estimates represent analysis of relatively new applications of mercury control technologies to utility units. New or emerging control technologies will undoubtedly have lower costs. (Cost analyses were not performed for such emerging technologies as coal gasification or for other HAPs.)

Based on this model plant analysis, the EPA estimated the national cost of mercury control (using activated carbon) to be approximately $3 billion. The EPRI has estimated the cost to the industry for mercury control to range from $1 to $10 billion.[79]

Should the EPA embark on a regulatory program, detailed cost analyses of the various control options and strategies considered would be undertaken."

10.7.7 Comment

The OMB commented that it was "...concerned about the experimental nature of the technologies for controlling mercury emissions and the uncertainties associated with costs and effectiveness of those technologies." [80]

10.7.8 Response

The EPA shares this concern and would like to see the technology tested on full-scale units to establish their actual emissions reduction capacity and costs across a range of operating conditions.

10.7.9 Comment

The OMB commented that "...the discussion of the effectiveness of potential control strategies for HAPs should include a full discussion of the strengths, limitations, and uncertainties of the information provided for each option." [81]

10.7.10 Response

The EPA will look in more depth at these strengths, limitations, and uncertainties when more data on these alternative controls become available. This material will be presented in the final Report to Congress.

10.7.11 Comment

The DOE commented that the "EPA characterizes the mercury removal effectiveness of the...FGD as poor. Based on...1992-1994 utility emissions characterization studies...this statement is generally valid. However, recent pilot-scale testing has shown relatively high removal of divalent mercury (Hg++) across a wet scrubber...similar results were found in tests recently performed at a utility boiler...due in part to developments in the methods used to measure (speciate) mercury. It is also more directly the result of a concerted effort to better understand the role of conventional flue gas cleanup technology in controlling mercury emissions. Continued research may show that existing FGD technology can have a significant impact on mercury emissions, particularly Hg++. This will become more relevant if additional FGD i[s] installed in compliance with Phase II, and will have an impact in terms of lowering EPA's mercury emission projections through the year 2010."

10.7.12 Response

The EPA was not able to obtain the necessary data, or conduct appropriate analyses and review of the data, from the recent pilot-scale studies referred to by the DOE before publication of this interim report. However, the EPA plans to consider these data and, if appropriate, conduct additional analyses of the data from the pilot-scale studies before completing the final utility study Report to Congress.

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11.0 PRELIMINARY OBSERVATIONS

The following observations are based on this study and the currently available scientific data.

11.1 INDUSTRY GROWTH AND HAP EMISSIONS

1. Utility units emit a significant number of the 189 HAPs included on the section 112(b) list, with coal-fired units emitting the largest number of HAPs. Coal-, oil-, and gas-fired utilities emit a mix of HAPs, including organics (e.g., polycyclic aromatic hydrocarbons, dioxins) and heavy metals (e.g., arsenic, lead, cadmium, chromium, nickel, mercury). Utilities are responsible for approximately 21 percent (between 18 and 23 percent) of the U.S. anthropogenic emissions of mercury.

2. Estimated growth in the number of utility units during the period 1990-2010 is predicted to result in an overall increase in HAP emissions. Over this period, utility coal consumption is estimated to increase by approximately 32 percent, oil consumption is estimated to decrease by approximately 50 percent; and natural gas consumption is expected to increase by about 59 percent. Because coal combustion is the primary source of HAP emissions from utility units, its relative increased usage accounts for the increase in HAP emissions, even though much of the growth will be at units subject to emission regulations. This results from the fact that the emission controls used to comply with these regulations are not 100 percent efficient in removing HAPs. (The year 2010 was selected as the end year for analysis because it is projected that all measures to effect compliance with other provisions of the Act will be completed by then.)

3. Anticipated future actions taken by the utility industry to comply with other provisions of the Act through installation of add-on controls are not expected to significantly impact HAP emissions. Similarly, actions taken to enhance existing controls or to utilize emerging technologies may also impact HAP emissions. Unknown actions that, for example, take the form of pollution prevention or fuel switching may significantly impact HAP emissions. Planned installations of SO_2 scrubbers are running below estimates made at the time of enactment of the 1990 Amendments to the Act, with compliance with Title IV being effected by switching from relatively higher to

relatively lower sulfur coals, allowance trading, switching from coal to oil or natural gas, etc. rather than by installation of add-on controls. In addition, these SO_2 scrubbers do not exhibit significant control over the majority of HAPs, including mercury. Upgrades and modifications planned for PM control devices appear to be related to accounting for changes in coal ash properties when switching from relatively higher to relatively lower sulfur coals, rather than to any improvement in overall PM (and associated trace metal) control efficiency. The limited data available do not suggest that the various NO_x controls being installed to comply with Titles I and IV have any impact on HAP emissions. However, the impact on HAP emissions of advanced processes (e.g., the condensing heat exchanger, gold sorbent technology, other sorbent injection processes, advanced fine particulate control technologies, enhanced wet scrubbers) cannot be estimated at this time but depend on the extent to which they are utilized by the industry. Switching from one coal to another does not impact overall HAP emissions (i.e., there are no "low-HAP" coals as there are low-sulfur coals). The extent of the impact of pollution prevention (e.g., repowering for efficiency, DSM) on HAP emissions cannot be estimated but depends on the extent to which these practices are utilized by the industry. They would at least serve to reduce the rate of increase in HAP emissions. Switching from coal or oil to natural gas will effectively reduce HAP emissions, but the extent of this practice over the period 1990-2010 cannot be estimated.

4. <u>The emission test program provided valid and representative results for the purpose of this study</u>. The utility industry is composed of a wide variety of units employing a mix of fuel types, burner types, boiler types, and control device configurations. The HAP emission test data were obtained by a variety of organizations using common test and analytical protocols. The EPA had a hand in establishing these protocols. The test sites covered the range of types of facilities and configurations to the extent that the EPA finds that the units tested are representative of the industry.

11.2 INHALATION RISK ASSESSMENT

5. The incremental increased cancer risk due to inhalation exposure to HAP emissions (based on 1990 emission levels and calculated in the baseline assessment) from the "highest risk" coal-fired unit for the MEI for the year 1990 is estimated to be approximately 4 in 1,000,000 (i.e., 4 x 10^{-6}). This does not include risks due to either multipathway exposure or background exposures. Arsenic and chromium VI are the HAPs that contribute most to this cancer risk. The MEI inhalation risk for arsenic emissions is estimated to be 3 x 10^{-6}, and the MEI inhalation risk for chromium VI is estimated to be 2 x 10^{-6}. The cancer risks from other HAPs are estimated to be lower than those from arsenic and chromium VI.

6. For the year 2010, the MEI cancer risks from coal-fired utilities are not expected to be significantly different. However, due to unknown actions and uncertainties in the projections, the EPA has low confidence in this observation.

7. The incremental increased cancer risk due to inhalation exposure to HAP emissions from the "highest-risk" oil-fired utility for the MEI for the year 1990 is estimated to be between 3 in 100,000 (i.e., 3 x 10^{-5}) and 1 in 10,000 (1 x 10^{-4}). Nickel, arsenic, chromium, and radionuclides are the HAPs contributing most to this estimated increased risk.

8. For the year 2010, the MEI cancer risks from oil-fired utilities are predicted to be approximately 30 to 50 percent lower than the 1990 estimates.

9. Based on the quantitative parameter uncertainty analysis conducted for the inhalation risk assessment, the EPA estimates that the baseline MEI risk estimates due to inhalation exposure presented above are reasonable, high-end estimates. The quantitative variability and uncertainty of many of the input parameters such as emission estimates, stack height, breathing rates, and exposure duration were considered in the uncertainty analysis. This resulted in an estimated range and distribution of potential risks due to inhalation exposure only. The median ("central tendency estimate") MEI inhalation risk estimate is roughly 5 to 10 times lower than the MEI baseline risk

estimates presented above. That is, these MEI risk estimates are more likely to be overestimating rather than underestimating the true risks due to inhalation exposure for the MEI. However, there are limitations to the uncertainty analysis (e.g., it did not consider multipathway exposures) and limitations in available data and the range of uncertainty is, most likely, larger than estimated by this study.

10. The risk estimates contain additional uncertainties that are not represented in the quantitative uncertainty analysis. The impact of using different high-to-low dose extrapolation models was not quantitatively assessed in this study. In addition, there are other factors, such as variation in population sensitivity (e.g., sensitive subpopulations) and other uncertainties and variabilities, that were not quantitatively assessed.

11. Based on the baseline assessment, 2 of the 426 coal-fired utilities are estimated to pose an MIR of cancer greater than 1×10^{-6} due to inhalation exposure to HAP emissions. The exposure model predicts that approximately 2,477 people are exposed to air concentrations from the two coal plants that result in an estimated inhalation cancer risk of greater than 1×10^{-6}. The other 424 coal-fired utilities are estimated to present inhalation cancer risks less than 1×10^{-6}.

12. The risk estimates from the "highest-risk" oil plant (MEI cancer risk of 1×10^{-4} assuming all nickel is as carcinogenic as nickel subsulfide) are estimated to be 3 to 4 times higher than the second "highest-risk" oil plant. The MEI risk from the second "highest-risk" oil-fired utility is estimated to be 3×10^{-5} (assuming all nickel is as carcinogenic as nickel subsulfide). The third "highest-risk" plant is estimated to present a risk of 1×10^{-5}. All remaining plants are estimated to be below 10^{-5}.

13. Using the assumption that the mix of nickel compounds are as carcinogenic as nickel subsulfide, 20 oil-fired utilities present an estimated MEI risk of greater than 10^{-6}. If the mix of nickel compounds is assumed to be 20 percent as carcinogenic as nickel subsulfide, then approximately four oil-fired utilities present an MEI risk greater than 1×10^{-6}.

14. The population cancer incidence due to inhalation exposure to HAP emissions from all coal-fired utilities, considering only local (within 50 km of each plant) impacts, is estimated to be 0.1 case per year.

15. The population cancer incidence due to inhalation exposure to HAP emissions from all oil-fired utilities, considering only local impacts, is estimated to be up to 0.5 case per year.

16. The test-case evaluation of long-range transport of particle-bound arsenic indicates that the population inhalation exposures could increase the estimated population cancer incidence by a factor of 7 (0.6 case per year for all fuels) in comparison to the local dispersion within 50 km of the source (less than 0.1 case per year. Thus, the cancer incidence may be dominated by exposures on the regional, rather than on the local, scale. Arsenic, as well as other carcinogenic heavy metals of concern (e.g., chromium and nickel), are mostly sorbed to fine particulate in utility stack emissions, will experience long residence times in the atmosphere, and are mostly subject to long-range transport over large geographical areas.

17. With regard to noncancer effects, the highest HQ for any HAP, considering both short- and long-term exposures, is approximately 0.1 for HCl from coal-fired utilities. This estimate represents the exposure due to utilities only and does not consider background exposures or long-range transport.

11.3 MERCURY

18. Mercury emitted by utility units has the potential to be transported long distances. The modeling analysis indicates that the deposition of mercury emitted from such units is dominated on the regional rather than the local scale. Predictions of the COMPDEP and RELMAP models indicate that most of the mercury emitted by utilities is transported further than 50 km from the emission source.

19. Based on the modeling conducted for this study, along with evidence from related studies, the EPA finds that there is a plausible link between mercury emissions from anthropogenic combustion and industrial sources,

including utility boilers, and mercury concentrations in air, soil, water, and sediments.

20. Based on the modeling conducted for this study, along with evidence from related studies, the EPA finds that there is a plausible link between methylmercury concentrations in freshwater fish and mercury emissions from utility units.

11.4 QUALITATIVE ASSESSMENT OF DIOXINS, ARSENIC, CADMIUM, AND LEAD

21. Dioxins, arsenic, cadmium, and lead are highly persistent in the environment, have the potential for bioaccumulation, and exhibit toxicity. These characteristics make it likely that noninhalation exposure pathways are a significant route of exposure to these HAPs. Therefore, further evaluation of multipathway exposure may be needed to more comprehensively assess the risks.

22. Exposure to arsenic through noninhalation exposure routes may pose additional risk, beyond the risk posed by inhalation exposure to arsenic; however, no quantitative assessment of noninhalation exposure to arsenic emissions from utility units has been performed at this time. Available data indicate that consumption of fruits, grains, and vegetables; soil ingestion; and ingestion of milk may be significant routes of exposure to arsenic. These noninhalation pathways are of potential concern due to the emissions of arsenic by utility units, because arsenic persists in the environment, bioaccumulates, and can cause cancer through the oral route of exposure.

11.5 RADIONUCLIDE ANALYSIS

23. The highest MIR risk to any individual, within a 50-km radius, resulting from multipathway exposure to radionuclide emissions from utility units is estimated to be 3×10^{-5}, and 17 of the 684 plants have an MIR greater than 10^{-5}. Based on the limited analyses of multiplant exposure, neither of these values would be changed if the analysis had systematically addressed multiplant exposure.

24. For the vast majority of the 196+ million persons living within 50 km of any utility unit, the lifetime fatal cancer risk is less than 1×10^{-6}.

25. Based on the multipathway exposure modeling conducted with the CAP-93 model, which estimates exposure within 50 km of each utility unit, the estimated deaths/year resulting from multipathway exposure to radionuclide emissions from utility units is approximately 0.3 per year. Had the exposures and risks due to overlapping plumes been explicitly addressed, this value would not increase by more than a factor of 3.

26. The quantitative uncertainty analysis performed for the radionuclide analysis indicates that these risk estimates are central values of the true probability distribution.

11.6 ALTERNATIVE CONTROL STRATEGIES

27. There are a number of alternative control strategies that are effective in controlling some of the HAPs emitted from utility units. These strategies are summarized below.

    A. Conversion of coal- and oil-fired units to natural gas firing effectively eliminates emission of HAPs.

    B. Conversion of coal-fired units to oil combustion will effect decreases in emissions of some HAPs but could lead to increased emissions of others (e.g., nickel).

    C. Because of the wide variability in the trace metal contents of coals, switching from one coal to another will not generally result in consistently reduced overall HAP emissions.

    D. Current methods of coal cleaning are able to remove portions of the trace metals contained within the coal. These emission reductions range from approximately 20 percent for mercury to approximately 50 percent for lead. Further research is needed in methods of effecting greater trace metal removals during coal cleaning.

    E. Newer forms of fuel combustion (e.g., coal gasification) show promise as being cleaner sources of electricity but available data are limited and further research is needed.

F. The impact of combustion controls on HAP emissions is inconclusive given the current level of knowledge. While available data appear to indicate that installation of low-NO_x burners results in a trend toward lower HAP emissions, the trend is neither universal nor uniform.

G. Particulate matter control devices (i.e., FFs, ESPs) generally effect good control (i.e., greater than 90 percent removal) of the trace metallic HAPs, with the exception of mercury. Organic HAPs are not well controlled by PM control devices. Fabric filters appear to moderately control HCl (i.e., 50 percent removal) but not HF, and ESPs do not generally effect good acid gas HAP control.

H. Wet acid gas control devices (e.g., FGDs) by themselves do not uniformly effect good control of the trace metallic HAPs, including mercury. Based on limited data, SDA/FF combinations, however, appear to be as effective as do FFs alone. Flue gas desulfurization units (as operated on utility units) generally effect good control (i.e., greater than 80 percent) of HCl, but control of HF is not uniform.

I. Add-on technologies for the control of mercury have not been demonstrated on utility units in the U.S. Pilot-scale work on activated carbon injection indicates that mercury removal is possible but that such removal is inconsistent and variable and that further research is needed.

J. Pollution prevention methods (e.g., DSM, energy conservation, repowering for energy efficiency) have the potential to result in reduced HAP emissions. The extent that these methods will be utilized by the industry is not known and, thus, the extent of any emission reductions cannot be forecast at this time.

Each of these alternative control strategies may have significant cost, economic, technical, and research implications before they can be widely utilized in the utility industry.

11.7 AREAS FOR FURTHER RESEARCH AND ANALYSES

There are numerous uncertainties and data gaps described throughout this report. This section identifies several of the important areas in which further research or scientific and technical work are needed.

11.7.1 Emissions Data for Dioxins

Dioxins emissions data were available from only eight of the tested utility plants. Therefore, there are greater uncertainties with the dioxin emissions than for many of the other HAPs. All types of utility units were not tested (e.g., there are no data available from coal-fired units with hot-side ESPs).

11.7.2 Speciation of Nickel and Chromium

There are significant uncertainties regarding the forms of nickel being emitted from oil-fired utilities and the health effects associated with those different forms. Therefore, further research and evaluation of the emissions to determine what forms are being emitted and of the health effects associated with those different forms would be of value for this study. Further evaluation of chromium speciation is also needed.

11.7.3 Multipathway Risk Assessment

As mentioned previously, further work is needed to study the risks due to multipathway exposure to HAPs that are persistent and bioaccumulate. Mercury, arsenic, dioxins, cadmium, and lead are five HAPs identified as priority for further multipathway assessment.

11.7.4 Long-Range Transport Exposures

Uncertainties remain regarding long-range transport of HAPs. Further modeling and evaluation is needed to assess the impacts of long-range transport of HAPs from utilities.

11.7.5 Mercury Issues

There are numerous areas regarding mercury that need further research and evaluation, which are discussed in the draft mercury study being reviewed by the Science Advisory Board (SAB). A few areas relevant to the content of this interim report are the following: (1) what percent of mercury emissions is elemental versus divalent mercury; (2) how much mercury is removed during coal cleaning; and (3) what control technologies or pollution prevention options are available that could potentially reduce mercury emissions, and what are the feasibilities and economic impacts of such options.

11.7.6 Projections to the Year 2010

There are significant uncertainties and unknowns in the emissions and risk projections made to the year 2010. Further research and evaluation in this area is needed.

11.7.7 Ecological Risks

Other than the ecological assessment for mercury, the effects of HAPs on wildlife, endangered species, and terrestrial and aquatic ecosystems were not evaluated. Although not mandated by section 112(n)(1)(A), further evaluation of ecological risks due to HAP emissions is needed to fully evaluate the impacts of utility HAP emissions.

11.7.8 Criteria Pollutant and Acid Rain Programs

Further evaluation is needed to assess the impacts of the Acid Rain and Criteria Pollutant programs on HAP emissions.

11.7.9 Short-Term Emissions

A limited assessment of short-term exposures was completed. However, further evaluation of short-term releases, especially high-end, peak releases, is needed to fully assess the potential impacts to public health due to emissions of HAPs (particularly HCl and HF) from utilities.

11.8 FEDERAL INTERAGENCY REVIEW COMMENTS

11.8.1 Comments

The Food and Drug Administration (FDA) commented that it was concerned that "...this report reaches conclusions about the possible public health impact of mercury emissions that are inconsistent with actions involving the Mercury Study report." The FDA also commented that "...it seems inappropriate to conclude that mercury emissions do in fact pose a potential concern for public health." [1]

The Office of Science Technology and Policy (OSTP) recommended that "... the utility report...[should] be delayed until completion of the SAB and stakeholder review [of the Mercury Report] and revision." [2]

The Department of Energy (DOE) commented that "...it is premature to conclude that it is necessary 'to pursue an evaluation of a range of options to minimize mercury emissions from various sources, including utilities'..." The DOE also commented that the language of the conclusions should be revised to say that "...it is not appropriate or necessary to regulate [HAPs] from [utilities] under section 112 at this time." The EPA should indicate that "...it will revisit this determination if information to the contrary becomes available." [3]

The Centers for Disease Control (CDC) commented "...that mercury emissions should be minimized as part of an overall strategy for reducing power-plant generated pollutants such as sulfur dioxide and particulate matter." [4]

The Council of Economic Advisors (CEA) commented that the April 1996 draft report "...is not clear on the health related benefits of reducing mercury emissions from utilities...the relationship between reductions in utility generated mercury emissions and deposition of mercury in the [U.S.] from anthropogenic sources is not clear." [5]

11.8.2 Response

Conclusions regarding the impacts to public health due to mercury emissions have not been included in this interim report. Also, policy decisions and regulatory decisions are not included in this interim report. The EPA plans to include conclusions regarding the impacts to public health, policy statements, and regulatory determination in the final report, as appropriate. Also, the EPA has not conducted a cost/benefit analysis for this interim report.

11.9 REFERENCES

1. Letter and enclosures from Lake, L.R., FDA, to Maxwell, William H., EPA/CG. April 30, 1996.

2. Memorandum and attachment from Gibbons, Jack, OSTP, to Maxwell, William H., EPA/CG. May 6, 1996.

3. Letter from Chupka, Marc W., DOE, to Maxwell, William H., EPA/CG. May 3, 1996.

4. Letter from Mannino, David M., CDC, to Maxwell, William H., EPA/CG. May 2, 1996.

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15 SUPPLEMENTARY NOTES

16 ABSTRACT

This report has been prepared pursuant to section 112(n)(1)(A) of the Clean Air Act, and provides the Congress and the public with information regarding the emissions, fate, and transport of utility HAPs. The primary components of this report are: (1) a description of the industry; (2) an analysis of emissions data; (3) an assessment of hazards and risks due to inhalation exposures to numerous HAPs (excluding mercury); (4) an assessment of risks due to multipathway (inhalation plus non-inhalation) exposure to radionuclides; and (5) a discussion of alternative control strategies. The assessment for mercury includes a description of emissions, deposition estimates, control technologies, and a dispersion and fate modeling assessment which includes predicted levels in various media based on modeling from four representative utility plants using hypothetical scenarios. The EPA plans to publish a final report at a later date which will include (1) a more complete assessment of the exposures, hazards, and risks; (2) conclusions, as appropriate and feasible, regarding the significance of the risks and impacts to public health; and (3) a determination as to whether regulation of utility HAPs is appropriate and necessary.

17 KEY WORDS AND DOCUMENT ANALYSIS

a DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c COSATI Field/Group
Air Pollution Atmospheric Dispersion Modeling Electric Utility Steam Generating Units Hazardous Air Pollutants/Air Toxics	Air Pollution Control	

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