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Air

# Study of Hazardous Air Pollutant Emissions from Electric Utility Steam Generating Units -- Final Report to Congress

# Volume 1.

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# Glossary

| | |
|---|---|
| AALG | ambient air level goal |
| AC | activated carbon |
| AECDP | Advanced Emissions Control Development Program (Babcock & Wilcox) |
| AFBC | atmospheric fluidized-bed combustor |
| APCD | air pollution control device |
| ACGIH | American Conference of Government Industrial Hygienists |
| ARD | Acid Rain Division (EPA) |
| ATSDR | Agency for Toxic Substances and Disease Registry |
| BAF | bioaccumulation factor |
| BAP | benzo[a]pyrene |
| BBF | biased burner firing |
| BCF | bioconcentration factor |
| BFB | bubbling fluidized bed |
| BOO | burners out of service |
| BSAF | biota sediment accumulation factor |
| CAA | Clean Air Act |
| CAP | Clean Air Act Assessment Package |
| CAPCOA | California Air Pollution Control Officers Association |
| CARB | California Air Resource Board |
| CCT | clean coal technology |
| CEDF | Clean Environment Development Facility (Babcock & Wilcox) |
| CFB | circulating fluidized bed |
| CLD | certainly lethal dose |
| CNS | central nervous system |
| CSF | cancer slope factor |
| DNA | deoxyribonucleic acid |
| DOC | dissolved organic carbon |
| DOE | Department of Energy |
| DSM | demand side management |
| E | (ratio of) exposure |
| ECTC | Environmental Control Test Center (EPRI) |

| | |
|---|---|
| EEI | Edison Electric Institute |
| EFP | emission factor program |
| EIA | Energy Information Administration (DOE) |
| E-LIDS™ | Enhanced Limestone Injection Dry Scrubbing |
| EMF | emission modification factor |
| ENAMAP | Eastern North American Model of Air Pollution |
| EPA | Environmental Protection Agency |
| EPRI | Electric Power Research Institute |
| ESP | electrostatic precipitator |
| EURMAP | European Regional Model of Air Pollution |
| FBC | fluidized-bed combustor |
| FCEM | field chemical emissions monitoring |
| FETC | Federal Energy Technology Center |
| FF | fabric filter |
| FGD | flue gas desulfurization |
| FGR | fluidized gas recirculation |
| FTIR | Fourier transform infrared |
| GEIA | Global Emissions Inventory Activity |
| GI | gastrointestinal |
| GIS | Geographic Information System |
| HAP | hazardous air pollutant |
| HEC | human equivalent concentration |
| HEM | Human Exposure Model |
| HI | hazard index |
| HQ | hazard quotient |
| IARC | International Agency for Research on Cancer |
| ICRP | International Commission on Radiological Protection |
| IDLH | immediately dangerous to life and health |
| IEM | indirect exposure methodology |
| IGCC | integrated gasification combined cycle |
| IPP | independent power producer |
| IRIS | Integrated Risk Information System |
| IRP | Inerts Ranking Program |
| ISC3 | Industrial Source Complex Version 3 |
| ISCLT2 | Industrial Source Complex Long Term Version 2 |

| | |
|---|---|
| ISCLT3 | Industrial Source Complex Long Term Version 3 |
| ISCST3 | Industrial Source Complex Short Term Version 3 |
| IURE | inhalation unit risk estimate |
| LADD | lifetime averaged daily dose |
| $LC_{50}/LD_{50}$ | lethal concentration/dose that kills 50% of test animals |
| LCUB | large coal-fired utility boiler |
| LET | linear energy transfer |
| L/G | liquid-to-gas |
| LOAEL | lowest-observed-adverse-effect level |
| LRT | long-range transport |
| MACT | maximum achievable control technology |
| MCL | maximum contaminant level |
| MCLG | maximum contaminant level goal |
| MCUB | medium coal-fired utility boiler |
| MDL | minimum detectable level |
| MEI | maximally exposed individual |
| MFP | monofluorophosphate |
| MIR | maximum individual risk |
| MLE | maximum likelihood estimate |
| MRL | minimal risk level |
| MWC | municipal waste combustor |
| MWe | megawatts electric |
| MWI | medical waste incinerator |
| NAAQS | national ambient air quality standards |
| NAPAP | National Acid Precipitation Assessment Program |
| NAREL | National Air and Radiation Environmental Laboratory |
| NAS | National Academy of Science |
| NASN | National Air Surveillance Network |
| NGM | nested grid model |
| NIEHS | National Institute of Environmental Health Sciences |
| NIOSH | National Institute for Occupational Safety and Health |
| NIST | National Institute of Standards and Technology |
| NMHC | nonmethane hydrocarbon |
| NOAA | National Oceanographic and Atmospheric Administration |
| NOAEL | no-observed-adverse-effect level |

| | |
|---|---|
| NOEC | no-observed-effect concentration |
| NRC | National Research Council |
| NSPC | Northern States Power Company |
| NSPS | new source performance standards |
| NTP | National Toxicology Program |
| NWS | National Weather Service |
| OAQPS | Office of Air Quality Planning and Standards (EPA) |
| OAR | Office of Air and Radiation (EPA) |
| OFA | overfire air |
| ORD | Office of Research and Development (EPA) |
| ORIA | Office of Radiation and Indoor Air (EPA) |
| OSF | off-stoichiometric firing |
| OSHA | Occupational Safety and Health Administration |
| OTAG | Ozone Transport Assessment Group |
| OURE | oral unit risk estimate |
| PAB | Pollutant Assessment Branch (EPA) |
| PAH | polycyclic aromatic hydrocarbon |
| PBPK | physiologically based pharmacokinetic |
| PCB | polycyclic biphenyl |
| PCDD | polychlorinated dibenzo-p-dioxin |
| PCDF | polychlorinated dibenzofuran |
| PEL | permissible exposure limit |
| PFBC | pressurized fluidized-bed combustor |
| PM | particulate matter |
| PSCCo | Public Services Company of Colorado |
| PURPA | Public Utility Regulatory Policies Act |
| RAC | Risk Assessment Council (EPA) |
| RBC | risk based concentration |
| RCRA | Resource Conservation and Recovery Act |
| REL | reference exposure level |
| RELMAP | Regional Lagrangian Model of Air Pollution |
| RfC | reference concentration |
| RfD | reference dose |
| SAB | Science Advisory Board |
| SAMSON | Solar and Meteorological Surface Observation Network |

| | |
|---|---|
| SBS | small boiler simulator |
| SCR | selective catalytic reduction |
| SCRAM-BBS | Support Center for Regulatory Air Models Bulletin Board System |
| SCUB | small coal-fired utility boiler |
| SDA | spray dryer absorber |
| SE | standard error |
| SGU | steam generating unit |
| SIP | State Implementation Plan |
| SNCR | selective noncatalytic reduction |
| SPC | Science Policy Council (EPA) |
| STAR | STability ARray |
| TCDD | tetrachlorodibenzo-p-dioxin |
| TE | trace element |
| TEF | toxicity (or toxic) equivalency factor |
| TEQ | toxicity equivalent |
| TLV | threshold limit value |
| TOC | total organic compounds |
| tpy | tons per year |
| TRI | Toxics Release Inventory |
| TSP | total suspended particulate |
| TTN | Technology Transfer Network |
| UARG | Utility Air Regulatory Group |
| UDI | Utility Data Institute |
| UNDEERC | University of North Dakota Energy and Environmental Research Center |
| URE | unit risk estimate |
| USGS | U.S. Geological Survey |
| USPHS | U.S. Public Health Service |
| VOC | volatile organic compound |
| WHO | World Health Organization |
| WL | working level |
| WLM | working level month |
| WOE | weight of evidence |

# EXECUTIVE SUMMARY

## ES.1 LEGISLATIVE MANDATE

In section 112(n)(1)(A) of the Clean Air Act, as amended (the Act), Congress directs the United States Environmental Protection Agency (EPA) to:

> "... 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 ... [hazardous air pollutants] ... after imposition of the requirements of this Act."

Section 112(a)(8) of the Act 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 MWe output to any utility power distribution system for sale is also considered an electric utility steam-generating unit (i.e., utility unit).

Section 112(n)(1)(A) also requires that:

- The EPA develop and describe alternative control strategies for hazardous air pollutants (HAPs) that may warrant regulation under section 112; and

- The EPA proceed with rulemaking activities under section 112 to control HAP emissions from utilities if EPA finds such regulation is appropriate and necessary after considering the results of the study.

## ES.2 REGULATORY DETERMINATION

This report does not contain a determination as to whether or not regulations to control HAP emissions from utility units are appropriate and necessary. The Agency has deferred the regulatory determination until a later date.

## ES.3 OVERVIEW APPROACH TO COMPLETING THE STUDY

The study included numerous separate and interrelated analyses. First, HAP emissions test data were gathered from 52 utility units (i.e., boilers), including a range of coal-, oil-, and natural gas-fired utility units. Second, the emissions test data along with facility specific information (e.g., boiler type, control device, fuel usage) were used to estimate HAP emissions from all 684 utility plants in the United States (U.S.). Third, a screening level hazard/risk assessment was completed to prioritize the HAPs for further analyses. Fourth, various priority HAPs were analyzed for inhalation and

multipathway exposures and risks and other potential impacts. In addition, potential control strategies were analyzed for the priority HAPs. The overall summary of the study is presented in Figure ES-1.

This report presents the findings of the study. 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 67 HAPs; (4) assessments of risks due to multipathway (inhalation plus non-inhalation) exposures to four HAPs (radionuclides, mercury, arsenic, and dioxins); and (5) a discussion of alternative control strategies.

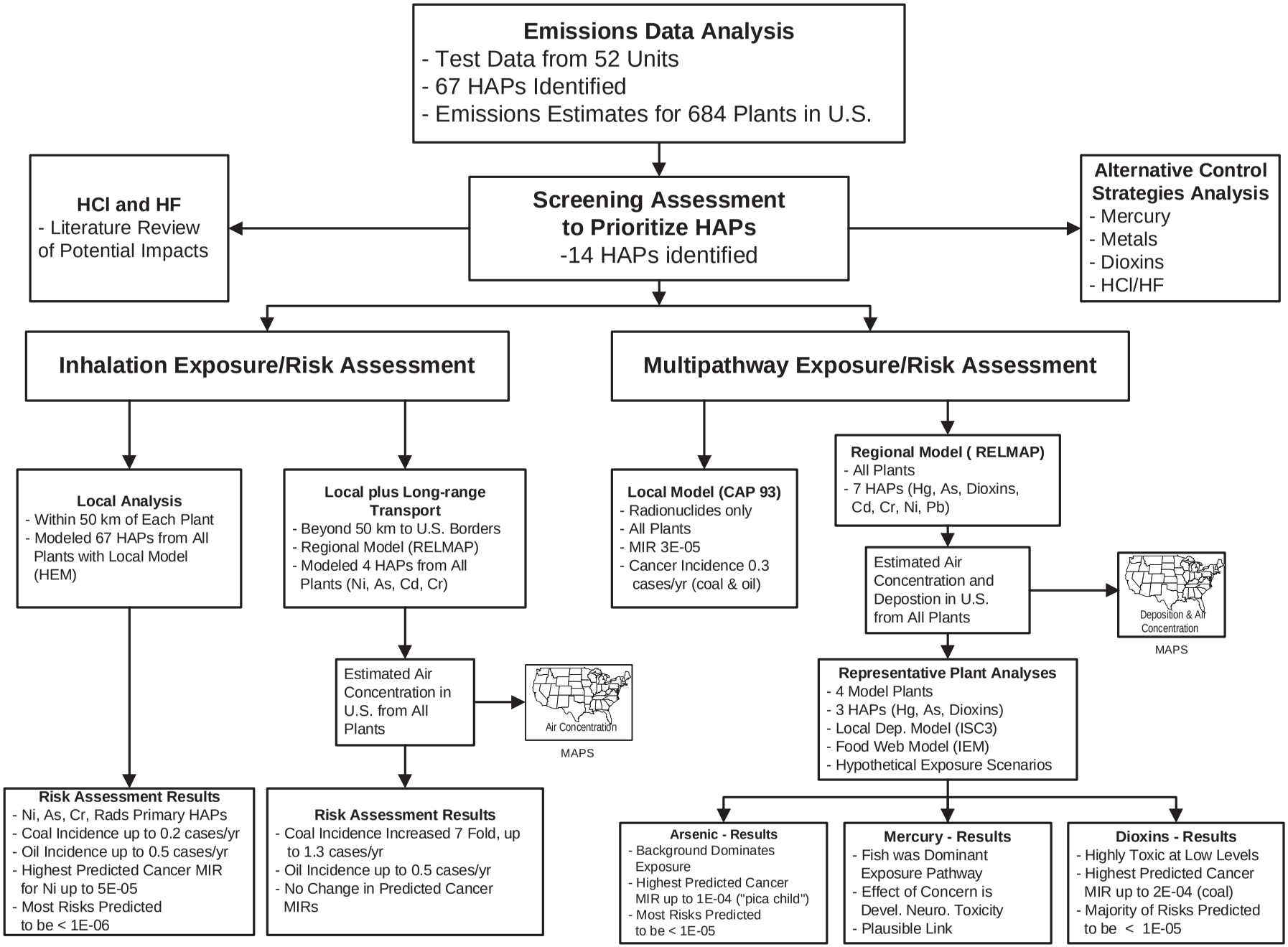
The study was based primarily on two scenarios: (1) 1990 base year emissions; and (2) 2010 emissions. In addition, emissions for 1994 were estimated using the most recent data. The 1990 scenario was chosen since that was the year the Amendments to the Act were passed and was the latest year for which utility operational data were available at the time the study was initiated. 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 acid rain program is in place. The 2010 scenario also included estimated changes in HAP emissions resulting from projected trends in fuel choices and projected increases in electric power demands. However, the effects of other on-going or potential activities that were not factored into the 2010 projections (e.g., industry restructuring, new ozone and particulate matter [PM] standards, global climate change programs) may result in the 2010 projections being either underestimated or overestimated.

## ES.4 EMISSIONS DATA ANALYSIS

A total of 684 utility plants (i.e., utilities) were identified as meeting the criteria for the study in 1990 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 and several plants burn more than one type of fuel (e.g., contain both coal- and oil-fired units). In 1990, there were 426 plants that burned coal as one of their fuels, 137 plants that burned oil, and 267 plants that burned natural gas.

Emission estimates for the years 1990, 1994, 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.

# Figure ES-1. Overall Structure of Utility Air Toxics Study Analyses



These test data provided the basis for estimating average annual emissions for each of the 684 plants. A total of 67 of the 188 HAPs listed in section 112 of the Act were identified in the emissions testing program as potentially being emitted by utilities. Tables ES-1 and ES-2 present estimated emissions for, respectively, a subset of priority HAPs for 1990, 1994, and 2010, and for a set of characteristic boilers for 1994.

Although the EPA used average annual emissions estimates in assessing long-term exposures to individual HAPs on a national basis, emissions test data were not available for each utility in the U.S. Therefore, estimates for individual plants are particularly uncertain. Based on an uncertainty analysis, the average annual emissions estimates are expected to be roughly within a factor of plus or minus three of actual annual emissions. However, even this uncertainty analysis had limitations. For example, the uncertainty analysis did not include data on potential upsets or unusual operating conditions; therefore, the range of uncertainty could be greater.

## ES.5 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. For many of the 67 HAPs, inhalation exposure is believed to be the dominant exposure pathway. However, for HAPs that are persistent and/or bioaccumulate, and are toxic by ingestion (or are radioactive), the non-inhalation exposure pathways could be more important. Based on a screening and prioritization assessment, which is described below, the EPA identified four high priority HAPs (radionuclides, mercury, arsenic, dioxins) to assess for non-inhalation exposures. In addition, cadmium and lead were identified as next highest priority. Multipathway assessments are presented for radionuclides, mercury, arsenic, and dioxins. The other two HAPs (lead and cadmium) were examined qualitatively for their potential for multipathway hazards.

## ES.6 SCREENING ASSESSMENT

As outlined in Figure ES-1, EPA initially 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).

Table ES-1. Nationwide Utility Emissions for Thirteen Priority HAPs[a]

| HAP | Nationwide HAP emission estimates (tons per year)[b] | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Coal | | | Oil | | | Natural gas | | |
| | 1990 | 1994 | 2010 | 1990 | 1994 | 2010 | 1990 | 1994 | 2010 |
| Arsenic | 61 | 56 | 71 | 5 | 4 | 3 | 0.15 | 0.18 | 0.25 |
| Beryllium | 7.1 | 7.9 | 8.2 | 0.46 | 0.4 | 0.23 | NM[c] | NM | NM |
| Cadmium | 3.3 | 3.2 | 3.8 | 1.7 | 1.1 | 0.9 | - | - | - |
| Chromium | 73 | 62 | 87 | 4.7 | 3.9 | 2.4 | - | - | - |
| Lead | 75 | 62 | 87 | 11 | 8.9 | 5.4 | 0.43 | 0.47 | 0.68 |
| Manganese | 164 | 168 | 219 | 9.3 | 7.3 | 4.7 | - | - | - |
| Mercury | 46 | 51 | 60 | 0.25 | 0.2 | 0.13 | 0.0015 | 0.0017 | 0.024 |
| Nickel | 58 | 52 | 69 | 390 | 320 | 200 | 2.2 | 2.4 | 3.5 |
| Hydrogen chloride | 143,000 | 134,000 | 155,000 | 2,900 | 2,100 | 1,500 | NM | NM | NM |
| Hydrogen fluoride | 20,000 | 23,000 | 26,000 | 140 | 280 | 73 | NM | NM | NM |
| Acrolein | 25 | 27 | 34 | NM | NM | NM | NM | NM | NM |
| Dioxins[d] | 0.000097 | 0.00012 | 0.00020 | $1 \times 10^{-5}$ | $9 \times 10^{-6}$ | $3 \times 10^{-6}$ | NM | NM | NM |
| Formaldehyde | 35 | 29 | 45 | 19 | 9.3 | 9.5 | 36 | 39 | 57 |

[a] Radionuclides are the one priority HAP not included on this table because radionuclide emissions are measured in different units (i.e., curies per year) and, therefore, would not provide a relevant comparison to the other HAPs shown. Radionuclide emissions are presented in chapter 9.

[b] 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 (see section ES.4 for discussion).

[c] NM = Not measured.

[d] 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).

Table ES-2. Estimated Emissions for Nine Priority HAPs from Characteristic Utility Units (1994; tons per year)[a]

| Fuel: | Coal | Oil | Natural gas |
|---|---|---|---|
| **Unit size (MWe):** | **325** | **160** | **240** |
| Arsenic | 0.0050 | 0.0062 | 0.0003 |
| Cadmium | 0.0023 | 0.0014 | NC[b] |
| Chromium | 0.11 | 0.0062 | NC |
| Lead | 0.021 | 0.014 | NC |
| Mercury | 0.05 | 0.0012 | NC |
| Hydrogen chloride | 190 | 9.4 | NC |
| Hydrogen fluoride | 14 | NC | NC |
| Dioxins[c] | 0.00000013 | 0.000000023 | NC |
| Nickel | NC | 1.7 | 0.004 |

[a] There are uncertainties in these numbers. Based on an uncertainty analysis, 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 d of Table ES-1.

If the MEI risk was above a minimum measure (e.g., exposure greater than one-tenth the inhalation reference concentration [RfC][a] 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 five criteria: (1) persistence; (2) tendency to bioaccumulate; (3) toxicity; (4) emissions quantity; and (5) radioactivity.

Based on this screening assessment, a total of 14 HAPs were identified as priority. Twelve HAPs (arsenic, beryllium, cadmium, chromium, manganese, nickel, hydrogen chloride [HCl], hydrogen fluoride [HF], acrolein, dioxins, formaldehyde, and radionuclides) were identified as priority pollutants for further study based on potential for inhalation exposures and risks. Four of these 12 HAPs (arsenic, cadmium, dioxins, and radionuclides) plus 2 additional HAPs (mercury and lead) were considered priority for multipathway exposure); of these 6 HAPs, 4 (arsenic, mercury, dioxins, and radionuclides) were identified as the highest priority to assess for

[a] 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.

multipathway exposures and risks. Overall, a total of 14 of the 67 HAPs were considered priority. The other 53 HAPs were not evaluated beyond the screening assessment.

## ES.7 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 (i.e., local analysis). For 13 of the 14 priority HAPs, the HEM was used; for radionuclides, the Clean Air Act Assessment Package-1993 (CAP-93) model was used. The HEM exposure modeling conducted for the inhalation risk assessment was very similar to the modeling conducted for the screening assessment. The same default options and same input data were used. However, there is one important difference. For the inhalation risk assessment, a distinction was made between urban and rural locations. If a 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). Dispersion of the pollutant plume in an urban area is expected to exhibit greater turbulence because of heat transfer and obstacles (i.e., large buildings). If a 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 (i.e., higher) estimates of human exposures, which is appropriate for a screening assessment. However, using the urban and rural distinction is believed to reflect more realistic conditions.

The cancer risks for all gas-fired plants were well below one chance in one million (i.e., $< 1 \times 10^{-6}$) 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 estimate (i.e., conservative, but within the bounds of reasonable estimates) of the risks due to inhalation exposure within 50 km of plants. Within the limits of current scientific information, this approach is, therefore, most likely to overestimate health risks for these pollutants. The uncertainty analysis suggests that the most likely estimated inhalation MIRs (i.e., central tendency MIRs) may be roughly 2 to 10 times lower than the high-end MIRs presented below. The average individual risks due to inhalation exposure to utility HAP emissions for the total exposed U.S. population (roughly 200,000,000 people) are predicted to be roughly 100 to 1000 times lower than the high-end inhalation MIRs.

ES.7.1 Inhalation Cancer Risks for Coal-Fired Utilities Based on Local Analysis (1990)

The vast 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 $1 \times 10^{-6}$ due to inhalation exposure to utility HAP emissions. Only two of the 426 plants are estimated to potentially pose inhalation risks greater than $1 \times 10^{-6}$ (see Figure ES-2).

The increased lifetime cancer MIR due to inhalation exposure to coal-fired utility HAP emissions, based on the local analysis, is estimated to be no greater than $3 \times 10^{-6}$. Arsenic and chromium are the HAPs contributing most to the inhalation risks (see Table ES-3). All other HAPs, including radionuclides, were estimated to present inhalation risks less than $1 \times 10^{-6}$ for coal-fired units.

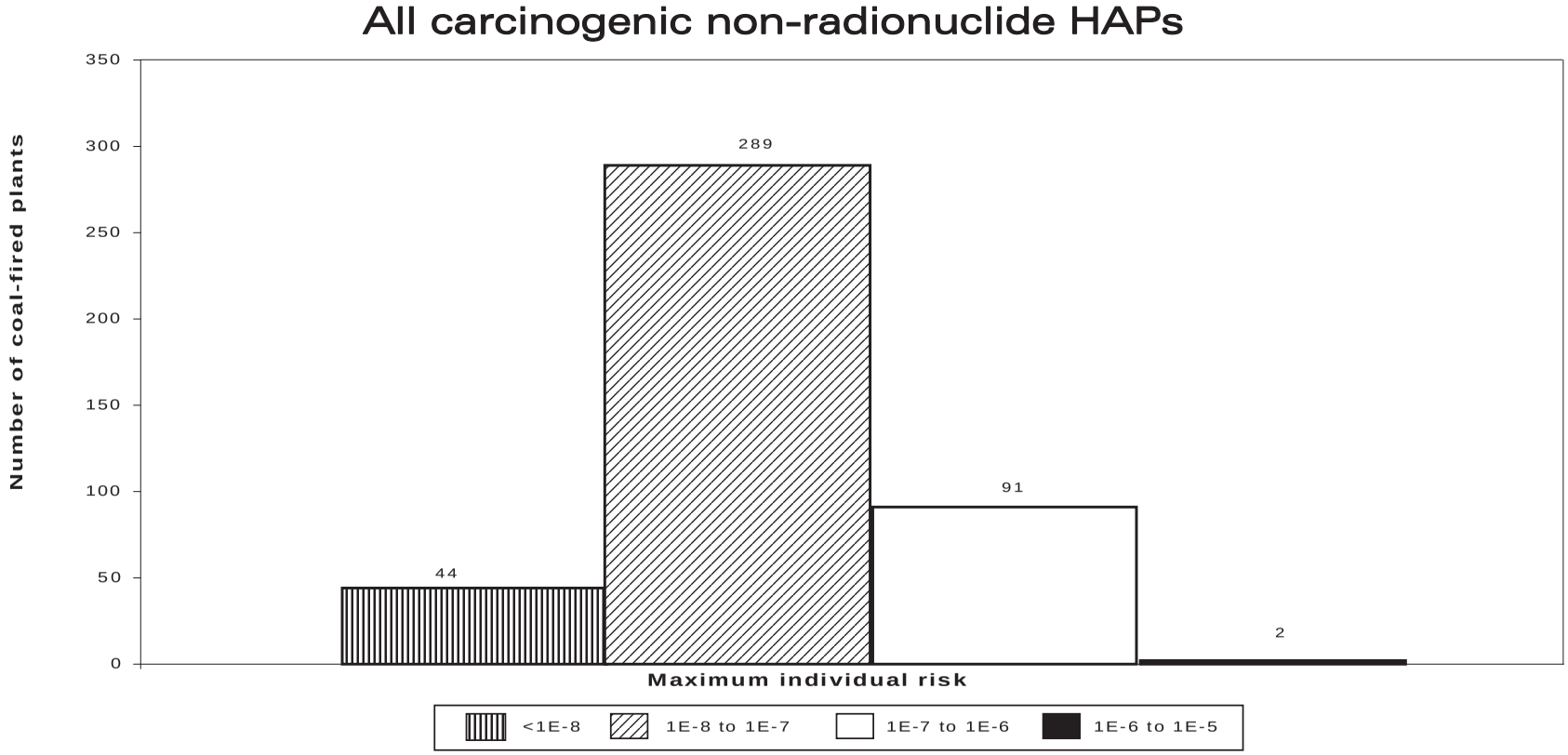
The cancer incidence in the U.S. due to inhalation exposure to HAPs (including radionuclides) from all 426 coal-fired plants **based on the local analysis** is estimated to be no greater than approximately 0.2 cancer case per year (cases/yr), or 1 case every 5 years. However, as described in later sections, the consideration of long-range dispersion of HAPs (beyond 50 km) results in increased estimates for cancer incidence.

ES.7.2 Inhalation Cancer Risks for Oil-Fired Utilities Based on Local Analysis (1990)

The majority of the oil-fired plants (125 of the 137 plants) are estimated to pose inhalation cancer MIRs less than $1 \times 10^{-6}$. However, up to 11 of the 137 oil-fired plants are estimated to potentially present inhalation MIRs above $1 \times 10^{-6}$ (see Figure ES-3). Nickel, arsenic, radionuclides, and chromium are the primary contributors to these cancer risks.

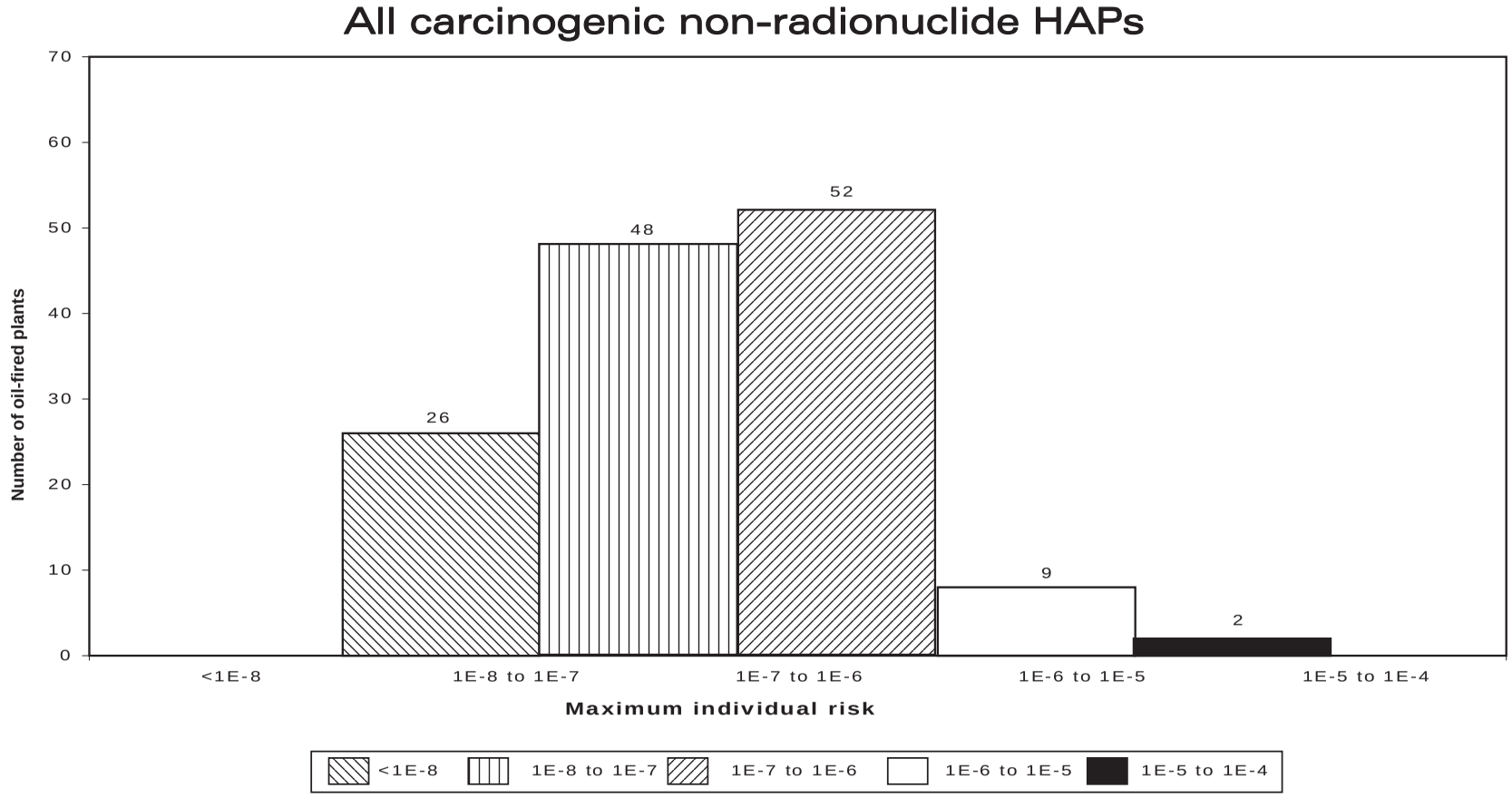
For oil-fired utilities, the highest contribution to the MIRs is from nickel. However, there are substantial uncertainties with the nickel risk estimates. Nickel is emitted in several different forms (e.g., nickel oxides, soluble nickel, sulfidic nickel) and the health effects of these different forms vary, and for some forms are unknown or uncertain. Nickel subsulfide (which is one of the possible forms of sulfidic nickel) is a known human carcinogen and appears to be the most carcinogenic form based on available data. Based on limited data, 3 to 26 percent of the nickel emissions are believed to be sulfidic nickel. It is not known how much of the sulfidic nickel emissions are nickel subsulfide. Several other nickel species (e.g., nickel oxides) are also potentially carcinogenic but the potencies are not known.

# Figure ES-2. Number of Coal-Fired Utilities Posing Various Levels of Maximum Individual Risks (By Levels of MIR)




Note: The high-end maximum individual risks (MIRs) are presented as exponents in this figure. For example, an increased cancer risk of one chance in one million (i.e., 1 x $10^{-6}$) is shown as 1E-6 in this Figure. The figure shows that 91 plants are estimated to pose an MIR between 1 x $10^{7}$ and 1 x $10^{6}$ .

# Figure ES-3. Number of Oil-Fired Utilities Posing Various Levels of Maximum Individual Risks (By Levels of MIR)




Note: The MIRs are presented as exponents in this figure. For example, an increased cancer risk of 1 x $10^{-6}$ is shown as 1E-6 here. The figure shows there are 9 oil-fired plants with estimated MIRs between 1 x $10^{-6}$ and 1 x $10^{-5}$.

Table ES-3. Summary of High-End Inhalation Cancer Risk Estimates from Local Analysis for Coal-Fired Utilities for the Year 1990

| HAP | Highest Cancer MIR[a] | Population with lifetime risk > 1 x $10^{-6}$ | Number plants with MIR > 1 x $10^{-6}$ |
|---|---|---|---|
| Arsenic | 2 x $10^{-6}$ | 850 | 2 |
| Chromium | 1 x $10^{-6}$ | 110 | 1 |
| Total[b] (Aggregate of HAPs) | 3 x $10^{-6}$ | 850 | 2 |

[a] Estimated lifetime maximum individual risk (MIR) due to inhalation exposure for the "highest risk" coal-fired plant. Based on an uncertainty analysis, these estimates are considered reasonable high-end estimates (see section ES.7.4 for discussion).

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

To evaluate the range of potential risks due to nickel emissions, the EPA estimated risks using various assumptions for nickel cancer potency (presented in chapter 6). For example, assuming the nickel mix is 50 percent as carcinogenic as nickel subsulfide, the highest inhalation cancer MIR due to the aggregate of HAP emissions from the highest risk oil-fired utility plant is estimated to be 6 x $10^{-5}$. Assuming the nickel mix is 10 percent as carcinogenic as nickel subsulfide, the highest inhalation cancer MIR due to the aggregate of HAP emissions from the highest risk oil-fired utility plant is approximately 3 x $10^{-5}$. The values in Table ES-4 and Figure ES-3 are based on the conservative assumption that the nickel mix is 50 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.

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 no greater than 0.5 cancer case/yr.

### ES.7.3 Inhalation Cancer Risks Based on Long-Range Transport

In addition to the above analyses, the EPA conducted long-range transport analyses to assess emissions dispersion and exposures on a national scale for 1990. 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.

Table ES-4. Summary of High-end Inhalation Cancer Risk Estimates Based on Local Analysis for Oil-Fired Utilities for the Year 1990

| HAP | Highest MIR[a] | Population with lifetime risk > 1 x $10^{-6}$ | Number plants with MIR > 1 x $10^{-6}$ |
|---|---|---|---|
| Nickel[b] | 5 x $10^{-5}$ | 110,000 | 11 |
| Arsenic | 1 x $10^{-5}$ | 2,400 | 2 |
| Radionuclides | 1 x $10^{-5}$ | 2,400 | 2 |
| Chromium | 5 x $10^{-6}$ | 2,300 | 1 |
| Cadmium | 2 x $10^{-6}$ | 45 | 1 |
| Total[c] (aggregate) | 6 x $10^{-5}$ | 110,000 | 11 |

[a] Estimated lifetime maximum individual risk (MIR) due to inhalation exposure for the "highest risk" oil-fired plant. Based on an uncertainty analysis, these estimates are considered reasonable high-end estimates (see section ES.7.4 for discussion).

[b] The estimates for nickel and total HAPs are based on the assumption that the mix of nickel compounds is 50 percent as carcinogenic as nickel subsulfide.

[c] Estimated risk due to inhalation of the aggregate of HAPs assuming additivity of risk for 14 individual carcinogenic HAPs.

The RELMAP modeling was conducted for all coal- and oil-fired utilities, but was limited to mercury, cadmium, chromium, arsenic, nickel, lead, and dioxins. Only inhalation exposures to the carcinogenic HAPs are discussed in this section. Deposition and multipathway concerns are discussed elsewhere in this report. The long-range transport modeling indicates that the local HEM analysis alone does not account for a substantial percentage of the population exposures due to coal-fired utility emissions. A comparison of the HEM results to the RELMAP results indicates a significant portion of emissions disperse further than 50 km, as would be expected for these HAPs, which are mostly fine particulate substances emitted from elevated stacks.

The RELMAP results for arsenic, cadmium, chromium, and nickel (which are emitted mainly as PM) were used to estimate the potential long-range transport inhalation exposures for other carcinogenic HAPs. Using this methodology, the highest cancer incidence due to inhalation exposure to HAPs from coal-fired utilities considering both local and long-range transport is estimated to be up to 1.3 cases/yr, which is about 7 times greater than the incidence estimated in the local analysis alone. The cancer incidence for oil-fired utilities did not change (see Table ES-5).

Table ES-5. Summary of High-End 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 individual risk (MIR) | Annual increased cancer Incidence | Maximum individual risk (MIR) | Annual increased cancer incidence |
| Radionuclides | $1 \times 10^{-5}$ | 0.2 | $2 \times 10^{-8}$ | 0.1 |
| Nickel[a] | $5 \times 10^{-5}$ | 0.2 | $7 \times 10^{-7}$ | 0.005 |
| Chromium | $5 \times 10^{-6}$ | 0.02 | $1 \times 10^{-6}$ | 0.02 |
| Arsenic | $1 \times 10^{-5}$ | 0.04 | $2 \times 10^{-6}$ | 0.05 |
| Cadmium | $2 \times 10^{-6}$ | 0.005 | $2 \times 10^{-7}$ | 0.0006 |
| All Others[b] | $8 \times 10^{-7}$ | 0.005 | $8 \times 10^{-7}$ | 0.004 |
| Total[c] | $6 \times 10^{-5}$ | 0.5 | $3 \times 10^{-6}$ | 0.2 |
| LOCAL PLUS LONG-RANGE IMPACTS (dispersion from utility emission points to borders of continental U.S.) | | | | |
| | OIL-FIRED PLANTS | | COAL-FIRED PLANTS | |
| Pollutant | Maximum individual risk (MIR) | Annual increased cancer incidence | Maximum individual risk (MIR) | Annual increased cancer incidence |
| Radionuclides | $1 \times 10^{-5}$ | 0.2 | Not estimated | 0.7 |
| Nickel[a] | $5 \times 10^{-5}$ | 0.2 | $1 \times 10^{-8}$ | 0.038 |
| Chromium | $5 \times 10^{-6}$ | 0.02 | $2 \times 10^{-6}$ | 0.15 |
| Arsenic | $1 \times 10^{-5}$ | 0.05 | $3 \times 10^{-6}$ | 0.37 |
| Cadmium | $2 \times 10^{-6}$ | 0.006 | $3 \times 10^{-7}$ | 0.005 |
| All Others[b] | $8 \times 10^{-7}$ | 0.006 | $1 \times 10^{-6}$ | 0.028 |
| Total[c] | $6 \times 10^{-5}$ | 0.5 | $4 \times 10^{-6}$ | 1.3 |

[a] Assumes that the nickel mixture is 50 percent 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] Aggregate risk (risk due to inhalation exposure to all carcinogenic HAPs, assuming additivity of risks).
[d] There are uncertainties associated with these risk estimates. See sections ES.7.4 for discussion.

A comparison between the HEM local dispersion results and the long-range transport modeling results indicates that long-range transport is much less important for the MIR than it is for cancer incidence. For example, the MIR from the local analyses for coal-fired utilities (i.e., inhalation risk of $3 \times 10^{-6}$) is predicted to

increase by roughly 10 to 20 percent to about 4 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 inhalation MIR because of the remote location of the two highest risk oil-fired plants.

ES.7.4 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 uncertainty analysis indicates that the inhalation cancer MIRs and incidence estimates presented above 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., predicted to be roughly the 90th to 95th percentile). The uncertainty analysis suggests that the most likely estimated inhalation MIRs (i.e., central tendency MIRs) may be roughly 2 to 10 times lower than the high-end MIRs presented above. The average individual risks due to inhalation exposure to utility HAP emissions for the total exposed U.S. population (roughly 200,000,000 people) are predicted to be roughly 100 to 1,000 times lower than the high-end inhalation MIRs.

ES.7.5 Summary of the Inhalation Cancer Risks

For the majority of utility plants (approximately 671 of the 684 plants), the estimated inhalation cancer risks due to HAP emissions are less than 1 x $10^{-6}$. However, several plants (2 coal plants and up to 11 oil plants) are estimated to potentially pose inhalation cancer risks above 1 x $10^{-6}$. One oil plant is estimated to pose a high-end inhalation cancer MIR of up to 6 x $10^{-5}$. Based on the assessment, no greater than 1.8 cancer cases/yr are estimated to occur in the U.S. due to inhalation exposure to HAP emissions from all coal- and oil-fired utilities. Further research and evaluation may be needed to more comprehensively assess the inhalation cancer risks, especially to reduce the uncertainties associated with the nickel risk estimates.

ES.7.6 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, 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.

Using a short-term air dispersion model that considers all reasonable meteorological conditions, 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.8 MERCURY MULTIPATHWAY ASSESSMENT

### ES.8.1 Background Discussion for Mercury

Mercury cycles in the environment as a result of natural and human (anthropogenic) activities. The amount of mercury mobilized and released into the biosphere has increased since the beginning of the industrial age. Most of the mercury in the atmosphere is elemental mercury vapor, which circulates in the atmosphere for up to a year, and hence can be widely dispersed and transported thousands of miles from likely sources of emission. After it deposits, mercury commonly is emitted back to the atmosphere either as a gas or associated with particles, to be re-deposited elsewhere. As it cycles between the atmosphere, land, and water, mercury undergoes a series of complex chemical and physical transformations, many of which are not completely understood.

Mercury is a persistent element and bioaccumulates in the food web. Mercury accumulates most efficiently in the aquatic food web. Predatory organisms at the top of the food web generally have higher mercury concentrations. Nearly all of the mercury that accumulates in fish tissue is methylmercury. Inorganic mercury, which is less efficiently absorbed and more readily eliminated from the body than methylmercury, does not tend to bioaccumulate.

Fish consumption dominates the pathway for human and wildlife exposure to methylmercury. The EPA's 1997 Mercury Study Report to Congress supports a plausible link between anthropogenic releases of mercury from industrial and combustion sources in the U.S. and methylmercury in fish. However, these fish methylmercury concentrations also result from existing background concentrations of mercury (which may consist of mercury from natural sources, as well as mercury which has been re-emitted from the oceans or soils) and deposition from the global reservoir (which includes mercury emitted by other countries). Given the current scientific understanding of the environmental fate and transport of this element, it is not possible to quantify how much of the methylmercury in fish consumed by the U.S. population is contributed by U.S. emissions relative to other sources of mercury (such as natural sources and re-emissions from the global pool). As a result, it cannot be assumed that a change in total mercury emissions will be linearly related to any resulting change in methylmercury in fish, nor over what time period these changes would occur. This is an area of ongoing study.

ES.8.2 Methylmercury Health Effects

Epidemics of mercury poisoning following high-dose exposures to methylmercury in Japan and Iraq demonstrated that neurotoxicity is the health effect of greatest concern when methylmercury exposure occurs to the developing fetus. Dietary methylmercury is almost completely absorbed into the blood and distributed to all tissues including the brain; it also readily passes through the placenta to the fetus and fetal brain. The reference dose (RfD) is an amount of methylmercury, which when ingested daily over a lifetime is anticipated to be without adverse health effects to humans, including sensitive subpopulations. At the RfD or below, exposures are expected to be safe. The risk following exposures above the RfD is uncertain, but risk increases as exposures to methylmercury increase.

Extrapolating from the high-dose exposures that occurred in the Iraq incident, the U.S. EPA derived a RfD for methylmercury of 0.1 microgram per kilogram body weight per day (µg/kg bw/day). While the U.S. EPA was advised by scientific reviewers to employ this RfD for this analysis, new data are emerging. Currently ongoing are two large epidemiology studies in the Seychelle Islands and in the Faroe Islands that were designed to evaluate childhood development and neurotoxicity in relation to fetal exposures to methylmercury in fish-consuming populations. Because of various limitations and uncertainties in all of the available data, the U.S. EPA and other Federal agencies intend to participate in an interagency review of the human data on methylmercury, including the most recent studies from the Seychelle Islands and the Faroe Islands. The purposes of this review are to refine the estimates of the level of exposure to mercury associated with subtle neurological endpoints and to further consensus between all of the Federal agencies. After this process, the U.S. EPA will determine if a change in the RfD for methylmercury is warranted. (Note: see the 1997 EPA *Mercury Study Report to Congress* for further discussion and assessment of mercury health effects and public health impacts).

ES.8.3 Mercury Multipathway Exposure Assessment

Mercury was considered highest priority for multipathway exposure analysis. 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 modeling, (2) local scale modeling, and (3) modeling of environmental concentrations. The RELMAP was used to predict long-range dispersion and deposition across the U.S. For the local analysis, a model designed to predict deposition of HAPs within 50 km, the Industrial Source Complex Version 3 (ISC3) air dispersion model, was used. Next, the EPA's Indirect Exposure Model Version 2M (IEM-2M) was used to estimate mercury environmental concentrations and human exposures. Hypothetical exposure scenarios were evaluated for four model plants (a large coal-fired, a medium coal-fired, a small coal-fired, and a medium oil-fired utility boiler). The analysis included three types of plant locations: (1) rural (agricultural), (2) near lakes

(lacustrine), and (3) urban. Three human fish consumption scenarios were considered.

The modeling provided information on whether local and/or long-range transport of mercury is significant in a variety of scenarios. The models indicate that most of the mercury from utilities is transported further than 50 km from the source. The fate and transport models provided an assessment of potential inhalation and ingestion exposures.

ES.8.4 Summary of Mercury Assessment Results for Utilities

Recent estimates of annual total global mercury emissions from all sources (natural and anthropogenic) are about 5,000 to 5,500 tons per year (tpy). Of this total, about 1,000 tpy are estimated to be natural emissions and about 2,000 tpy are estimated to be contributions through the natural global cycle of re-emissions of mercury associated with past anthropogenic activity. Current anthropogenic emissions account for the remaining 2,000 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.

For the year 1994, coal-fired utilities were estimated to emit approximately 51 tpy of mercury in the U.S., which is estimated to be 33 percent of the 158 tpy of airborne anthropogenic emissions of mercury in the U.S. If one assumes that current anthropogenic activity represents between 40 and 75 percent of the total airborne emissions (anthropogenic plus other emissions [e.g., natural emissions]), one can calculate that U.S. utilities emit roughly 13 to 26 percent of the total (natural plus anthropogenic) airborne emissions of mercury in the U.S.

Given the global estimates of 5,000 to 5,500 tpy (which are highly uncertain), U.S. anthropogenic mercury emissions are estimated to account for roughly 3 percent of the global total, and U.S. utilities are estimated to account for roughly 1 percent of total global emissions.

A computer simulation of long-range transport of mercury emissions from all U.S. sources conducted for the EPA's 1997 Mercury Study Report to Congress suggests that about one-third (~ 52 tons) of the 158 tpy of U.S. anthropogenic emissions are deposited, through wet and dry deposition, within the lower 48 States. The remaining two-thirds (~ 107 tons) is transported outside of U.S. borders where it diffuses into the global reservoir. In addition, the computer simulation suggests that another 35 tons of mercury from the global reservoir is deposited for a total deposition of roughly 87 tpy in the U.S. Although this type of modeling is uncertain, the simulation suggests that about three times as much mercury is being added to the global reservoir from U.S. sources as is being deposited from it. What is not uncertain is that additional emissions to air will

contribute to levels in the global reservoir and deposition to water bodies.

Long-range transport modeling conducted as part of this Utility Study predicts that approximately 30 percent (15 tpy) of the 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. Long-range transport modeling also predicts that the highest deposition occurs 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 receptor monitoring data, the RELMAP model seems to be accurate within a factor of plus or minus 2. That is, the RELMAP model seems to over- and underestimate mercury values within a factor of two and appears to be relatively unbiased in its predictions.

The modeling assessment in conjunction with available scientific knowledge, supports a plausible link between anthropogenic mercury emissions and mercury found in freshwater fish. As noted above, there are many sources of mercury emissions worldwide, both natural and anthropogenic. The coal-fired utilities are one category of the mercury sources.

Mercury is considered the highest priority for multipathway analyses because it is an environmentally persistent, toxic element. Mercury is deposited to soil and terrestrial vegetation but at levels that do not result in human exposures likely to be detrimental to health through terrestrial exposure pathways. However, in its methylated form mercury bioaccumulates in the food web (especially the aquatic food web). Modeling results suggest that most of the mercury emitted to the atmosphere is deposited more than 50 km away from the source, especially sources that have tall stacks. As stated above, the modeling assessment from the Mercury Study in conjunction with available scientific knowledge, supports a plausible link between anthropogenic mercury emissions and mercury found in freshwater fish. Additional emissions to air will contribute to levels in the global reservoir and deposition to water bodies. As a result, mercury emissions from utility units may add to the existing environmental burden.

At this time, the available information, on balance, indicates that utility mercury emissions are of sufficient potential concern for public health to merit further research and monitoring. The EPA recognizes that there are substantial uncertainties that make it difficult to quantify the magnitude of the risks due to utility mercury emissions, and that further research and/or evaluation would be needed to reduce these uncertainties. Remaining questions include the following: (1) what is the quantitative relationship between a change in U.S. mercury emissions and the resulting change in

methylmercury levels in fish; (2) what are the actual consumption patterns and estimated methylmercury exposures of the subpopulations of concern; (3) what are the actual mercury levels in a statistically valid and representative sample of the U.S. population and susceptible subpopulations; (4) what exposure levels are likely to result in adverse health effects; (5) what affects the formation of methylmercury in waterbodies and its bioaccumulation in fish; (6) how much mercury is emitted from natural sources and past anthropogenic sources; and (7) how much mercury is removed during coal cleaning and other ongoing practices for pollution control. New data that could reduce some of the uncertainties are likely to become available in the next several years, and EPA plans to review and consider these data, as appropriate, in future decisions.

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, total mercury removal by existing PM control devices on coal-fired utilities varies considerably, ranging from 0 to 82 percent removal (with a median efficiency of 15 percent removal) for cold-side electrostatic precipitators (ESPs), and from 0 to 73 percent removal (with a median efficiency of 8 percent removal) for fabric filters. Also, hot-side ESPs exhibited no mercury control. Existing flue gas desulfurization (FGD) units exhibit limited mercury control, ranging from 0 to 62 percent removal, with a median removal of 23 percent. The mercury control efficiency of FGD units is a function of several factors including temperature, plant configuration, and type of coal. 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.9 SCREENING LEVEL MULTIPATHWAY ASSESSMENT FOR ARSENIC

Arsenic is a naturally occurring element found normally, in various concentrations, in soil. In addition, arsenic can also be naturally present in other media (e.g., various food sources and water). Arsenic levels have been measured in a variety of foods. Even though shellfish and other marine foods contain the greatest concentrations of total arsenic, much of the arsenic present in fish and shellfish exists in the less toxic organic form. Other food products, such as meats, rice, and cereals, contain higher percentages, and often higher total amounts, of inorganic arsenic, which is the form of primary toxicological concern.

Arsenic is also naturally present in trace amounts in coal and oil. When coal or oil are burned, some of this naturally occuring arsenic is released to the atmosphere. The quantity of arsenic released from any utility plant is dependent on many factors including the concentration of arsenic in the fuel, control device efficiency, and other factors.

Utilities emit about 62 tpy of arsenic nationwide, about 3 to 4 percent of the total anthropogenic arsenic emissions in the U.S. Because of its chemical and physical characteristics, arsenic emitted to the atmosphere may be transported to other environmental media (soil or water), thus allowing non-inhalation exposures to occur.

ES.9.1 Exposure Modeling

It was not possible to model every utility plant for arsenic multipathway exposures. Therefore, a screening level model plant approach was used. Four model plants (i.e., a large coal-fired, a medium coal-fired, a small coal-fired, and a medium oil-fired utility boiler) were designed to characterize typical utility plants. In taking the model plant approach, it was realized that there would be a great deal of uncertainty surrounding the predicted fate and transport of arsenic as well as the exposures. However, the assessment was useful for estimating potential risks due to utility arsenic emissions. Three models were used to predict environmental arsenic concentrations and exposure: the RELMAP, the ISC3, and the Indirect Exposure Model Version 2 (IEM-2). These models were used to predict the fate and transport of arsenic emissions and to estimate human exposures to arsenic through multiple exposure routes, including food consumption, water ingestion, and inhalation. Three basic exposure scenarios were considered: a subsistence farmer (adult and child), a subsistence fisher (adult and child), and a pica child (i.e., a child that ingests significant quantities of soil). These scenarios were considered because they represent possible high-end scenarios for exposure to arsenic.

ES.9.2 Health Effects of Arsenic

Inhalation exposure to inorganic arsenic has been strongly associated with lung cancer in humans. Human exposure to inorganic arsenic, via ingestion, has been associated with an increased risk of several types of cancer, including skin, bladder, liver, and lung cancers. Oral exposure to inorganic arsenic has also been associated with noncancer effects, including effects to the central nervous system, cardiovascular system, liver, kidney, and blood.

ES.9.3 Approach for Estimating Screening Level Arsenic Risks

Increased cancer risks were estimated for each hypothetical scenario, for the four model plants, each of which was placed in two different hypothetical locations (i.e., an eastern humid site and a dry western site). For each of the exposure scenarios, except for the pica child, it is assumed that the hypothetical person is exposed for 30 years. For the pica child, it is assumed that exposure occurs for

7 years. Risks were estimated by multiplying the estimated intakes of arsenic by the EPA's cancer potency factor for arsenic.

ES.9.4 Screening Level Arsenic Risk Assessment Results

The results of the screening level multipathway arsenic exposure assessment provide an indication of the potential hazards and risks that may occur due to emissions from a utility plant. However, the results are not applicable to any particular plant. There are uncertainties and limitations to the analysis.

Exposures to inorganic arsenic due to background levels and due to emissions from the model utility boilers were predicted to be mainly through the ingestion of grains. Exposure to inorganic arsenic through the ingestion of fish was not predicted to be a major pathway of exposure because there is considerable evidence that little of the total arsenic in fish tissue is inorganic arsenic. Soil ingestion is the major route of exposure to inorganic arsenic for the pica child.

ES.9.4.1 Arsenic Cancer Risks. The cancer risks due to multipathway exposures to inorganic arsenic, as estimated in the model plant analysis using hypothetical scenarios, due to utility emissions alone (no background) were estimated to range from $4 \times 10^{-7}$ to $1 \times 10^{-4}$. The highest estimated risk ($1 \times 10^{-4}$) was for a pica child assumed to be living at the point of maximum deposition. The arsenic emissions from the large coal-fired model utility boiler at the eastern humid site were estimated to pose this highest risk for the pica child. When the risk from background exposure ($2 \times 10^{-4}$) is added to the maximum risk from utility exposure, the risk for the pica child is estimated to be up to $3 \times 10^{-4}$. The "pica child" is considered a high-end, conservative scenario.

Background exposures were estimated to dominate the exposures and risks in all scenarios. When considering only the arsenic emissions from the model utility units (not including background), in all scenarios it was the large coal-fired unit that was estimated to pose the greatest multipathway risks and the medium coal-fired unit was estimated to pose the next highest risks. The small coal-fired unit and the oil-fired unit were estimated to present lower risks.

ES.9.4.2 Uncertainty Discussion. There are uncertainties associated with the cancer risk estimates from arsenic. The analysis was based on model plants and hypothetical constructs; therefore, the results are not applicable for any specific utility plant. Further analyses are needed to better characterize the risks posed by arsenic emissions from utilities. A few uncertainties are discussed here.

Exposure to arsenic through the ingestion of tap or well water was not included in this assessment. The exposure modeling assessment was based on a model plant analysis, hypothetical scenarios, and incorporated data with varying degrees of uncertainty. Also, there are uncertainties associated with the health effects data for arsenic.

For example, the animal ingestion studies have not clearly shown an association between arsenic ingestion exposure and cancer.

## ES.10 DIOXIN SCREENING LEVEL MULTIPATHWAY ASSESSMENT

The highest MEI inhalation cancer risk due to dioxin emissions from any utility plant based on the HEM analysis (described in section ES.7) was estimated to be $1 \times 10^{-7}$. The EPA estimates that coal-fired utilities emit 0.2 pounds per year (lb/yr) of dioxin (toxic equivalents, TEQ) and that oil-fired utilities emit 0.01 lb/yr. These estimates combined are roughly 1 percent of the nationwide anthropogenic dioxin emissions. However, dioxin emissions data were only available for twelve utility plants and 42 percent of the measurements were below the minimum detection limit. Moreover, dioxins are not part of the naturally occuring fossil fuel. They are formed in highly complicated reactions which may occur with unknown frequency during combustion. Therefore, the emissions data for dioxins from utilities, which are the basis of exposure modeling, are considered more uncertain than the emissions data for many of the other HAPs.

For the screening level multipathway analysis, the transport, deposition, multipathway exposures, and human cancer risks were assessed for utility emissions of polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs), collectively referred to as dioxins. Atmospheric deposition of dioxin emissions can be important because dioxins tend to persist in the environment and bioaccumulate in the food web. Environmental persistence and bioaccumulation, coupled with carcinogenic effects at very low levels, make multipathway exposure an important consideration for dioxins.

### ES.10.1 Methods

The basic approach for estimating screening level multipathway exposures to dioxins was similar to the methods described above for mercury and arsenic. However, there were some differences. The EPA's ISCST3 model was used to predict deposition and air concentrations of dioxins within 50 km of each of four model plants. Model plants were selected to represent both large and small coal- and oil-fired utilities. A modified version of the IEM spreadsheet model was used to estimate environmental concentrations, exposures to the environmental concentrations for 16 hypothetical human scenarios, and the resulting cancer risks. Pathways assessed include inhalation, dermal contact with soil, and ingestion of water, soil, fish, plants, and animals.

### ES.10.2 Results

Since the analysis was based on model plants, using hypothetical scenarios, the results are not applicable to any specific plant and contain substantial uncertainties about the risks due to dioxin emissions. Total modeled screening level lifetime cancer risks related to multipathway exposure to dioxins for the four-model plant analysis ranged from $1 \times 10^{-10}$ to $2 \times 10^{-4}$. The results of this

analysis indicate that the exposures and risks due to fish consumption are the highest of all pathways considered. The highest modeled result of $2 \times 10^{-4}$ lifetime cancer risk was obtained for the subsistence fisher exposure scenario. In all modeled scenarios, the non-inhalation exposures were at least one order of magnitude larger than the inhalation exposures, thus demonstrating the potential significance of including multipathway exposure analysis in the risk assessments for pollutants that are environmentally persistent and tend to bioaccumulate. Also, unlike the results for arsenic, modeled exposures to dioxins for each pathway exceed the background exposure estimates for dioxins.

ES.10.3 Uncertainty Discussion

Several sensitivity analyses were completed for the screening level multipathway assessment of utility dioxin risks to assess the reasonableness of the results. The assumptions with the greatest impact on the predicted risk to the subsistence fisher were those made about the biota-sediment accumulation factor. This sensitivity analysis suggests that the modeling results are reasonable for a screening level analysis.

## ES.11 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 $1 \times 10^{-5}$. The highest estimated multipathway radiation exposure for the MEI due to radionuclide emissions from utilities was predicted to be 1.5 millirems (mRems) per year, which is estimated to pose an increased cancer risk of $3 \times 10^{-5}$. Seventeen plants (13 coal- and 4 oil-fired plants) were estimated to pose multipathway risks between $1 \times 10^{-5}$ and $3 \times 10^{-5}$. 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. 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.

The risks due to exposure to radionuclides from utilities are substantially lower than the risks due to natural background radiation. The average exposure to natural background radiation (excluding radon) for the U.S. population has been estimated to be

roughly about 100 mRems per year, which is about 67 times higher than the highest exposure due to utility radionuclide emissions.

## ES.12 QUALITATIVE MULTIPATHWAY EXPOSURE ASSESSMENT

The EPA recognizes that non-inhalation exposure pathways could be important for additional HAPs that are persistent and tend to bioaccumulate. A few additional HAPs that were not modeled for multipathway exposures are discussed below.

### ES.12.1 Cadmium and Lead

Cadmium emissions from the vast majority of plants (683 of the 684 plants) are estimated to pose inhalation risks less than $10^{-6}$, and the highest modeled air concentration of lead was 200 times below the national ambient air quality standard (NAAQS). Cadmium and lead are persistent, may bioaccumulate, and are toxic by ingestion. However, since the emission quantities and inhalation risks are relatively low, the EPA does not plan to conduct future evaluations of multipathway exposures of cadmium and lead from utilities.

### ES.12.2 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, there are considerable uncertainties about the noncancer toxicity of nickel and chromium at relatively low ingestion doses (below the toxic threshold). Also, it is uncertain whether they pose a carcinogenic risk by ingestion. Hence, EPA does not plan to assess multipathway exposures for nickel and chromium for utilities.

## ES.13 POTENTIAL IMPACTS OF HYDROGEN CHLORIDE AND FLUORIDE

No exceedances of the health benchmarks (e.g., RfCs) for HCl or HF were identified in the inhalation exposure assessment. However, emissions of HCL and HF may contribute to acid deposition and, to a lesser extent to PM fine and visibility problems. To the extent that these emissions may contribute to such problems, they could be addressed through other Titles of the Act.

## ES.14 ALTERNATIVE CONTROL AND PREVENTION STRATEGIES

There are numerous potential alternative control strategies for reducing 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 approaches that prevent pollution by improving efficiency in supply (e.g., promoting energy efficiency in combustion) or demand (e.g., demand side management [DSM], pollution prevention, energy conservation). The degree of feasibility, cost, 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.15 OTHER ISSUES AND FINDINGS

### ES.15.1 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 report, HAP emissions from coal-fired utilities are predicted to increase by 10 to 30 percent by the year 2010. Predicted changes that were included in the 2010 emissions projections include the installation of scrubbers for a small number of facilities, the closing of a few facilities, and an increase in fuel consumption of other facilities. However, based on EPA's exposure modeling analysis for the year 2010, 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 the acid rain program 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, acid rain precursors, or global warming compounds (e.g., use of fuel switching or add-on controls to reduce $SO_x$, $NO_x$, and/or carbon dioxide emissions), these actions could result in reductions in HAP emissions. For example, analyses performed to assess compliance with the revised NAAQS for ozone and PM indicate that mercury emissions in 2010 may be reduced by approximately 16 percent (11 tpy) over those projected in this report. Other potential (but unknown) actions (e.g., repowering, restructuring) may have a significant impact on HAP emissions; however, these unknowns were not included in the 2010 projection.

### ES.15.2 Peer Review

Draft versions of Chapters 1 through 9 and 13 of this report and draft technical support documents were reviewed by many non-EPA scientists representing industry, environmental groups, academia, and other parties. Chapters 10, 11, and 12 are new chapters produced in response to major comments from the reviewers. 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. Additional review occurred during 1997. 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. In addition, there were some comments that EPA did not agree with. Also, the new chapters (10 to 12) have only undergone a limited review. 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 fee) on request by calling the above number.

ES.15.3 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 $1 \times 10^{-6}$ 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.

The EPRI's risk estimates are generally similar to, but in several cases lower than, those of EPA. Differences between the studies include: (1) EPA's use of a higher unit risk factor for arsenic; (2) EPA's assumption that nickel was 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) EPRI's 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 (the carcinogenic form), 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.15.4 Potential Environmental Impacts Not Included in Study

There are other potential environmental issues associated with utilities not assessed in this report. These include: (1) 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; (2)an assessment of ecological impacts of HAPs; (3) the impacts of carbon dioxide emissions and climate; and (4) the impacts resulting from restructuring, 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 Section 112(n) of the Act.

ES.15.5 Link to Particulate Matter

Arsenic, cadmium, chromium, lead, nickel, radionuclides, and several other HAPs 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 (about two-thirds of oil-fired units are uncontrolled for PM). Impacts for PM were not addressed in this study, but are being studied under Title I of the Act. If additional controls of PM emissions are utilized, this could result in reductions in HAP emissions.

ES.16 OVERALL TECHNICAL SUMMARY AND CONCLUSIONS

Based on available information and current analyses, the EPA believes that mercury from coal-fired utilities is the HAP of greatest potential concern and merits additional research and monitoring. There are uncertainties regarding the extent of risks due to mercury exposures including those from utility emissions. Further research and evaluation are needed to gain a better understanding of the risks and impacts of utility mercury emissions. In addition, further research and evaluation of potential control technologies and strategies for mercury are needed.

For a few other HAPs, there also are still some remaining potential concerns and uncertainties that may need further study. First, the screening multipathway assessments for dioxins and arsenic suggest that these two HAPs are of potential concern (primarily from coal-fired plants); however, further evaluations and review are needed to better characterize the impacts of dioxins and arsenic emissions from utilities. Second, nickel emissions from oil-fired utilities are of potential concern, but significant uncertainties still exist with regards to the nickel forms emitted from utilities and the health effects of those various forms. The impacts due to HAP emissions from gas-fired utilities are negligible based on the results of this study; therefore, the EPA feels that there is no need for further evaluation of the risks of HAP emissions from natural gas-fired utilities.

## ES.17 AREAS FOR FURTHER RESEARCH AND ANALYSIS

There are many uncertainties and data gaps described throughout this report. This section summarizes several important areas in which further research or scientific work may be needed.

### ES.17.1 Emissions Data for Dioxins

Emissions data for dioxin compounds were available from less than 12 utility plants. Many of the measurements were near the detection limits. Therefore, there are greater uncertainties with the dioxin emissions than for the other HAPs. Research may be needed to gain a better understanding of the dioxin emissions from utilities and the dioxin formation, if any, in various utility boiler types (e.g., units with cold-side or hot-side ESPs).

### ES.17.2 Speciation of Nickel

There are significant uncertainties regarding the forms of nickel emitted from oil-fired utilities and their associated health effects. Research would be useful to determine the emissions quantities of various nickel forms and the health effects of various nickel forms.

### ES.1.7.3 Multipathway Risk Assessment

Further work may be needed to better characterize the risks due to multipathway exposure to certain HAPs (e.g., arsenic and dioxins).

### ES.17.4 Local, Regional, and Long-range Transport Exposures

Further modeling and evaluation may be needed to better characterize the impacts of local, regional, and long-range transport of HAPs from utilities.

### ES.17.5 Mercury

There are numerous areas regarding mercury that may need further research, study, or evaluation. A few potential areas for further study include the following:

(1) additional data on mercury content of various types of coal;

(2) improved methods for measuring mercury levels in water;

(3) the impact of reducing mercury emissions from coal-fired facilities on the bioaccumulation of mercury in fish;

(4) statistically valid and reliable estimates of methylmercury exposure levels in the U.S. population and susceptible subpopulations, as measured in human hair;

(5) the occupational, dietary and behavioral factors that affect mercury exposures for people who are determined to be exposed above a threshold of concern;

(6) the human health and environmental benefits that would be expected by reducing mercury emissions from U.S. utilities;

(7) control technologies or pollution prevention options that are available, or will be available, that could potentially reduce mercury emissions and what are the costs of those options;

(8) how do other regulations, programs and activities (e.g., acid rain program, electricity restructuring, NAAQSs, and climate change) affect mercury emissions; and

(9) additional data on mercury emissions (e.g., how much is emitted from various types of units, how much is divalent vs elemental mercury, and how do factors such as control device, fuel type, and plant configuration affect emissions and speciation).

Several additional uncertainties and potential areas for further research on mercury are discussed in other sections of this report.

ES.17.6 Projections to the Year 2010

There are significant uncertainties and unknowns in the emissions and risk projections made to the year 2010 (e.g., impact of electricity restructuring; impact of State efforts to regulate such restructuring; impact of any climate change abatement initiatives). Research and evaluation in these areas may be needed.

ES.17.7 Ecological Risks

The effects of HAPs on wildlife, endangered species, and terrestrial and aquatic ecosystems were not evaluated in this study. Although not mandated by section 112(n)(1)(A), further evaluation of ecological risks due to HAP emissions would be needed to fully evaluate the impacts of utility HAP emissions.

ES.17.8 Criteria Pollutant and Acid Rain Programs

Further evaluation is needed to assess the impacts of the Acid Rain and Criteria Pollutant programs (e.g., impact of revisions to the PM-fine and ozone NAAQS; impact of Ozone Transport Assessment Group [OTAG] activities) on HAP emissions, especially for mercury.

# 1.0 INTRODUCTION

This chapter presents an introduction to the study of hazardous air pollutant (HAP) emissions from electric utility steam-generating units (i.e., 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 (CAA or 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 directs the U.S. Environmental Protection Agency (EPA) to:

> "... 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."

Section 112(a)(8) of the Act 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 MWe output to any utility power distribution system for sale is also considered an electric utility steam-generating unit.

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; and
- The EPA proceed with rulemaking activities under section 112 to control HAP emissions from utilities if it determines from the study that such regulation is appropriate and necessary.

Section 112(n)(1)(A) does not include a requirement to analyze the cost(s) of alternative control strategies in the study. Therefore, no cost analyses (e.g., control costs, economic, cost-benefit) have been performed as a part of this study. These analyses would be conducted as part of the rulemaking process should EPA determine that regulations are appropriate and necessary.

The EPA began work in 1991 to develop and collect the information and data needed to prepare this study of HAP emissions from electric utilities. At that time, only a small amount of reliable data on HAP

emissions from utilities were available. In October 1996, the Agency published a three-volume report, *Study of Hazardous Air Pollutant Emissions from Electric Utility Steam Generating Units--Interim Final Report*. (EPA-453/R-96-013). This final report incorporates additional analyses and includes more recent data on emissions, control technologies, and health effects.

This report discusses the possible impact of pollution controls required by other Federal regulations or sections of the Act, estimates which HAPs are present in utility unit emissions, and estimates exposures and risk to humans from the emission of these HAPs.

## 1.2 CAA PROVISIONS AND STUDIES RELATED TO THIS STUDY

The CAA contains several provisions relating to electric utilities that will impact the industry well into the future. Environmental regulations implementing many of these requirements are now in effect; others have been established since the date of the last report; and others are under development.

This section summarizes the major provisions of the Act affecting electric utilities and their relevance to this study. These include nonattainment provisions, acid deposition control programs, and new source performance standards (NSPS) discussed in sections 1.2.1 through 1.2.3. The development of regulations for HAP under section 112 of the Act and other related studies required by section 112 are discussed in sections 1.2.4 and 1.2.5, respectively.

### 1.2.1 Nonattainment Provisions

Title I of the Act includes requirements for attaining and maintaining the national ambient air quality standards (NAAQS). Section 108 of the Act directs EPA to identify certain pollutants which may reasonably be anticipated to endanger public health and welfare. Section 109 directs the Administrator to establish primary and secondary NAAQS for the identified pollutants. Under section 110 of the Act and related provisions, States are primarily responsible for ensuring attainment and maintenance of the ambient standards. The EPA has established NAAQS for six criteria pollutants: ozone ($O_3$), carbon monoxide (CO), particulate matter (PM), lead, sulfur dioxide ($SO_2$), and nitrogen oxides ($NO_x$) under Title 40, Part 50 of the Code of Federal Regulations (40 CFR Part 50). Electric utilities are significant emitters of $SO_2$ and $NO_x$; $NO_x$ emissions from electric utilities account for about one-third of nationwide emissions.[1] Electric utilities also emit other criteria pollutants such as PM as well as air toxics.

The EPA issued revised NAAQS for $O_3$ and PM on July 18, 1997 (*Federal Register*, volume 62, page 38856 [62 FR 38856]). The new rules strengthened the primary standard for $O_3$, added standards for PM less than 2.5 microns in size ($PM_{2.5}$) to supplement the $PM_{10}$ primary standard, and revised secondary standards.[2] As part of this

rulemaking, the EPA also proposed rules requiring States to develop programs to reduce regional haze.

To achieve the new standards, EPA has developed an integrated strategy that will require reductions in $NO_x$ and $SO_2$ as well as volatile organic compounds (VOC) and PM. Nitrogen oxide is a precursor to the formation of ground-level $O_3$, and $SO_2$ is a precursor to the formation of PM in the atmosphere. Electric utilities will be affected as States reduce emissions to meet the new standards. One EPA study predicts that nationwide $NO_x$ reductions ranging from 25 to 90 percent, depending on the particular State or non-attainment area, will be needed to attain the revised $O_3$ ambient standard.[3]

The Regulatory Impact Analysis for the revised standards assumes that much of the needed emission reductions would be achieved through the Acid Deposition Program for $SO_2$ and $NO_x$ discussed in section 1.2.2 and through the revised NSPS discussed in section 1.2.3. According to the analysis, $SO_2$ and $NO_x$ emissions from utilities will be reduced by approximately 40 and 50 percent, respectively, by the year 2010.[4] These analyses also estimate a 16 percent reduction (approximately 11 tons per year) in utility mercury emissions (in 2010) as a result of compliance with the revised NAAQS (primarily related to the impact of the $SO_2$ strategy to meet the PM NAAQS).[5]

In a related action, EPA proposed rules requiring 22 States and the District of Columbia to submit State Implementation Plans (SIPs) that address the regional transport of ground-level $O_3$. The proposed rule would decrease ozone transport in the eastern half of the United States by reducing $NO_x$ emissions. Under the proposed rule, States may reduce emissions from sources they choose, although utility and large nonutility point sources are expected to be affected. Implementation of the proposed rule would reduce total emissions of $NO_x$ by 35 percent based on analyses by EPA and the Ozone Transport Assessment Group (OTAG). The EPA estimates that this action will bring areas into attainment with the revised $O_3$ standard without additional local controls. Many of these States are expected to reduce $NO_x$ emissions by participating in the cap-and-trade program discussed in section 1.2.2.[6]

### 1.2.2 Acid Deposition Control

Title IV of the Act sets as its primary goal the reduction of annual $SO_2$ emissions by 10 million tons below 1980 levels. To achieve these reductions, the law requires a two-phase tightening of the restrictions placed on fossil-fuel power plants (i.e., utilities). Phase I of EPA's $SO_2$ Program (40 CFR Parts 72 through 75) began in 1995 and affects 263 units at 110 mostly coal-burning electric utility plants in 21 States. An additional 182 units joined Phase I as substitution or compensating units, bringing the total of Phase I units to 445. Emissions data indicate that 1995 $SO_2$ emissions at these units nationwide were reduced by almost 40 percent below their required level of 8.7 million tons.[7] The second phase begins in the year 2000 and covers an additional 1,600 boilers. The EPA believes the 10 million ton goal will be met before the year 2010. To reduce

$SO_2$ emissions, an affected source may: (1) install flue gas scrubbers, (2) switch to a fuel that contains less sulfur, 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.

Under the $SO_2$ program, affected units are allocated allowances based on their historic fuel consumption and a specific emissions rate. Each allowance permits a unit to emit one ton of $SO_2$ per year. For each ton discharged, one allowance is retired. Allowances may be bought, sold, or banked and are tracked through a computerized system. However, no source can emit at a level violating Federal or State limits set under Title I of the Act. Sources also must obtain a permit and meet continuous emission monitoring requirements for $SO_2$, $NO_x$, and carbon dioxide ($CO_2$), as well as volumetric flow and opacity monitoring requirements.

Section 407 of the Act establishes the $NO_x$ Emission Reduction Program with the goal of reducing emissions by 2 million tons from 1980 levels. Like the $SO_2$ emission reduction program, the $NO_x$ program is implemented in two phases beginning in 1996 and 2000. Under Phase I rules (40 CFR Part 76), approximately 277 dry-bottom wall-fired boilers and tangentially-fired boilers (Group I) must meet applicable annual average emission rates of 0.45 pound per million British thermal units (lb/MMBtu) and 0.50 lb/MMBtu, respectively, by January 1, 1996. Utilities can meet the limits by installing low-$NO_x$ burner technology or other combustion control technology or by averaging emissions among several units. An affected unit also may obtain an alternative emission limit under specified conditions. Implementation of Phase I will decrease annual $NO_x$ emissions by over 400,000 tons per year (tpy) between 1996 and 1999 (60 FR 18751, April 13, 1995).[8,9]

The EPA issued final rules implementing Phase II of the program in late 1996 (61 FR 67112, December 19, 1996). In these rules, EPA determined that more effective low $NO_x$ burner technology is available to establish more stringent standards for Phase II, Group I boilers than those established for Phase I. Emission limits for Group II boilers (wet bottom, cyclones, cell burners, and vertically-fired boilers) were also established based on $NO_x$ control technologies comparable in cost to low $NO_x$ burners (selective catalytic reduction [SCR]). Selective catalytic reduction is a commercially available flue gas treatment technology that injects ammonia into the flue gas in the presence of a catalyst. The catalyst promotes reactions that convert $NO_x$ to nitrogen and water. By the year 2000, the Phase II rule (affecting 775 units) will achieve an additional reduction of 1.17 million tons of $NO_x$ per year. Phase I and Phase II together are estimated to decrease nationwide annual $NO_x$ emissions by 2.06 million tpy beginning in the year 2000.[10]

The final Phase II rule includes an option allowing a State or group of States to petition EPA to accept an emissions cap-and-trade program as a substitute for compliance with the Group 2 limits and

additional reductions required for Group 1 boilers. The petition may be granted if the Administrator finds that alternative compliance through the cap-and-trade program will achieve lower total $NO_x$ emissions from Group 1 and Group 2 boilers than if the new limits were applicable. The Phase I limits established in 1995 would apply to Group 1 boilers in a cap-and-trade program. This provision is expected to affect boilers located in the OTAG region which contains about 87 percent of the units covered by the Phase II rule.[10]

In related developments, the EPA is currently developing a model cap-and-trade program to facilitate $NO_x$ emission reductions from large stationary sources choosing to participate. The Agency intends to propose the rule in early 1998 and finalize the action in conjunction with the ozone transport rulemaking in September 1998.[11]

### 1.2.3 New Source Performance Standards

Section 111 of the Act requires the development of NSPS for newly constructed or modified affected facilities. Section 403 of the Act, as amended, revised the definition of the term "standard of performance" to mean:

> "...a standard for emissions of air pollutants which reflects the degree of emission reduction achievable through the application of the best system of emission reduction which (taking into the cost of achieving such reduction and any nonair quality health and environmental impact and energy requirements) the Administrator determines has been adequately demonstrated."

New source performance standards currently provide the major regulatory authority for the control of air emissions from utilities. Fossil-fuel-fired steam generating units greater than 73 MW heat input that were constructed or modified after August 17, 1971, are subject to requirements of 40 CFR Part 60, Subpart D; units constructed or modified after September 18, 1987, are subject to 40 CFR Part 60, Subpart Da. These rules define "fossil fuel" 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. Subparts D and Da include limits for emissions of $SO_2$, $NO_x$, and PM based primarily on the use of scrubbers or low sulfur coal, combustion modification techniques (overfire air, low excess air, and reduced heat release rate), and PM control devices. Provisions also are included for the use of continuous opacity monitoring systems and continuous emission monitoring systems for $SO_2$ and $NO_x$ and oxygen ($O_2$) or $CO_2$.

Section 407 of the Act requires EPA to revise the NSPS for $NO_x$ emissions from utility and nonutility units to reflect improvements in emission reduction methods. The EPA proposed revisions to the $NO_x$ limit for utility units (i.e., boilers) in 40 CFR Part 60, Subpart Da (regardless of fuel type) based on coal-firing and the performance of SCR control technology, in combination with combustion controls (62 FR

36947; July 9, 1997). Thus, units can meet the proposed standards by using clean fuels such as natural gas or by installing more effective control systems. The proposed rule also revised the emission limit to incorporate an output-based format that will encourage unit operating efficiency and pollution prevention. The EPA estimates that about 43,600 tons of $NO_x$ per year would be emitted from 17 new utility boilers expected to be constructed over the next 5 years. The proposed revised standards would reduce these emissions by about 25,800 tpy.[12]

The NSPS program 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, the use of $SO_2$ scrubbers (currently on about 14 percent of the units) will also control some vapor-phase HAPs, such as hydrogen chloride (HCl) and hydrogen fluoride (HF), in addition to providing some control of mercury.

### 1.2.4 Hazardous Air Pollutants

Section 112(d) of the Act requires that EPA promulgate regulations for the control of HAPs listed in section 112(b) of the Act from both new and existing major sources. A "major" source means a source that:

> "... emits or has the potential to emit, considering controls, 10 tons per year or more of any HAP or 25 tons per year or more of any combination of HAPs."

Regulations developed under section 112(d) must reflect the maximum degree of reduction in emissions of HAP that is achievable, taking into consideration the cost of achieving the emissions reduction, and any non-air quality health and environmental reduction and energy requirements. This level of control is commonly known as the maximum achievable control technology (MACT). For new sources, MACT standards cannot be less stringent than the emission control that is achieved in practice by the best-controlled similar source. The MACT standards for existing sources cannot be less stringent than the average emission limitation achieved by the best-performing 12 percent of existing sources for categories and subcategories with 30 or more sources, or the best-performing 5 sources for categories or subcategories with fewer than 30 sources. Section 112(d) also provides that the Administrator may distinguish among classes, types, and sizes of sources within a source category when establishing standards. Regulations for the control of HAP emissions from utilities will be developed under this authority if such regulations are determined to be necessary and appropriate.

### 1.2.5 Other Studies

The 1990 amendments to section 112 of the Act also mandate 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) the Great Waters study, and (5) the Presidential Risk Commission.

1.2.5.1 Mercury Study. Section 112(n)(1)(B) requires the EPA to complete a study of mercury emissions from utilities, municipal waste combustion units, and other sources, including area sources, by November 15, 1994. The study is 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. The EPA just recently (December 19, 1997) published the *Final Mercury Study Report to Congress*. The 3-year delay was necessary to allow sufficient time for data gathering, analyses, writing, and extensive peer review. The Mercury Study is closely related to this Utility Study because utilities are the largest anthropogenic source of mercury emissions. Utilities (primarily coal-fired utilities) are estimated to emit approximately 33 percent of the airborne anthropogenic mercury in the United States. Several analyses and conclusions contained in the Mercury Study are applicable to utilities, and are discussed in Chapter 7 of this report.

1.2.5.2 NIEHS Health Effects of Mercury Study. Under section 112(n)(1)(c), the NIEHS is required to perform a study identifying the threshold level of mercury exposure that would not adversely affect human health. A report on the NIEHS study was published in 1993.[13]

1.2.5.3 NAS Risk Assessment Methodologies Study. In January 1995, the NAS finalized a report[14] on the risk assessment methodologies used by the EPA. The results of the NAS study were consulted to help develop the methodologies for the risk assessment portions of this study.

1.2.5.4 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.[15] 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.

Following the first Report to Congress, the EPA published the "Final Water Quality Guidance for the Great Lakes System" required by section 118(c)(2) of the Clean Water Act (60 FR 15366, March 23, 1995). This guidance document established minimum water quality criteria, methodologies, policies, and procedures for the Great Lakes System. States and Tribes in the Great Lakes Basin were required to incorporate these provisions into their water quality standards and National Permit Discharge Elimination System permit programs by March 1997. In the guidance, EPA recognized that non-point sources of mercury, particularly by air deposition, are the most significant remaining contributors of mercury to the Great Lakes System. The EPA followed the guidance with the "Final Water Quality Guidance for the Great Lakes System Draft Mercury Permitting Strategy," released for public comment in June 1997 (62 FR 31025). The final permitting strategy will be finalized in the near future.

The second report to Congress on the atmospheric deposition of pollutants to the Great Waters was completed in June 1997. The report confirmed, and provided additional support for, the findings of the first Report to Congress that persistent and bioaccumulative toxic pollutants and excessive nitrogen can adversely affect the environmental condition of the Great Waters. Electric utilities and mobile sources are identified, in modeling studies and emission data, as major contributors of $NO_x$ to the Chesapeake Bay and its watershed.[16]

1.2.5.5 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 has issued the report in two volumes. Volume 1 entitled, "Framework for Environmental Health Risk Assessment," was issued in February 1997. Volume 2 entitled, "Risk Assessment and Risk Management in Regulatory Decision-Making," was issued in April 1997.

## 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. Portions of this report, and the data and methodologies utilized, were 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 through 11 address multipathway exposures to mercury, lead, cadmium, radionuclides, arsenic, and dioxins. Chapter 12 discusses the potential impacts of HCl and HF. Alternative control strategies for HAP emissions reductions are given in Chapter 13. Chapter 14 presents the conclusions of the study. Additional supporting material is provided in the appendices.

[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.

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16. U.S. Environmental Protection Agency. *Deposition of Air Pollutants to the Great Waters: Second Report to Congress*. EPA-453/R-97-011. Office of Air Quality Planning and Standards, Research Triangle Park, NC. June 30, 1997, pp. i-vi, 1-14.

# 2.0 CHARACTERIZATION OF THE INDUSTRY

This chapter presents a characterization of the fossil-fuel-fired 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 13. 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 1994.[a] 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 is defined (section 112(a)(8) of the Act) as any fossil-fuel-fired combustion unit of more than 25 megawatts electric (MWe) 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 MWe output to any utility power distribution system for sale.

Fossil fuel-fired electric utility steam-generating units are fueled primarily by coal, oil, or natural gas. Figure 2-1 shows the 1994 distribution of fossil fuels burned by the electric utility industry by unit (i.e., individual boiler) and by total megawatts.[1] Coal-fired boilers account for the largest portion of the industry by number of units (1,026 units, 61 percent), representing 68 percent of the industry's total megawatts. Gas-fired boilers make up 30 percent of the industry's units (493 units) and account for 23 percent of the total megawatts.[b] Oil-fired boilers account for 9 percent of the units

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[a] 1994 was chosen as the analysis year for this final report because that is the last year for which complete, plant-specific data were available from the Utility Data Institute (UDI), the same data source as was used for the 1990 analyses and upon which the 2010 projections are based. The UDI database is based on the same data as that used by the Energy Information Administration (which may have more recent information) but, because of varying uses and means of data verification, the data sets may not agree completely.

[b] It should be noted that the 1994 data for the number of individual combined-cycle turbine systems were not available. However, the number of plants (34, all gas-fired) and their total number of megawatts (10,047.87) was available, and these totals were included in Figure 2-1. This would have the effect of biasing the gas-fired boiler unit numbers lower than normal, but would have no effect on the gas-fired boiler MW numbers.

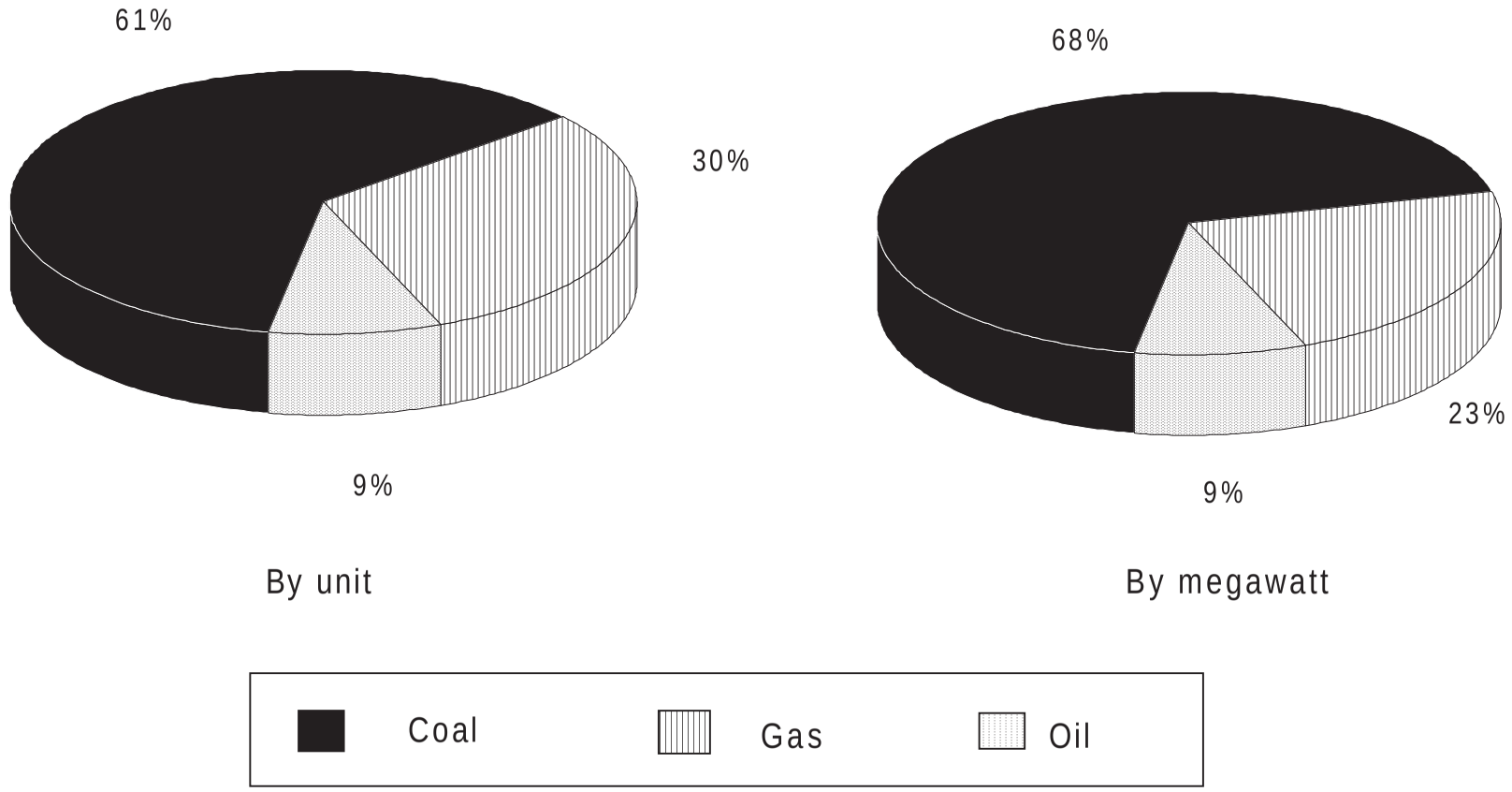



**Figure 2-1. Fossil fuel use in the utility industry in 1994.[1]**

(149 units) and represent 9 percent of the megawatts. Fossil-fuel-fired electric utility steam generating units accounted for 81 percent of the total industry in 1994.[1] Other fuels utilized include biomass at 0.4 percent and "other" (including nuclear, geothermal, hydroelectric, etc.) at 17.9 percent. Units less than 25 MWe (the defining limit set by the Act) comprised 0.7 percent of the industry.

## 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.

### 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 whereas 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. In 1994, there were 1,668 conventional utility steam-generating units in the United States, with 1,026 burning coal of some type. The total output was 464.8 gigawatts electrical (GWe).[1]

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 United States 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 244 separate companies, investor-owned utilities provided 74.3 percent of kilowatt hour (kWh) generation of electric power to the Nation in 1994. Publicly owned utility companies, which consist of approximately 2,020 separate companies, represent 10.8 percent of the Nation's electric power supply. The 10 Federal power agencies generate 9.2 percent of the Nation's electric power supply. Rural electric cooperatives, numbering approximately 931 separate companies, provides 5.7 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]

One of the fastest growing areas of the electric utility industry has been nonutility generators. 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.

Ownership of nonutility generators can be further divided into ownership by:

- Units that cogenerate steam and electricity (qualifying facilities[6]);[c]

- Small power producers (<80 MWe) that generate electricity primarily from a renewable source; and

- Other nonutility generators (e.g., independent power producers [IPPs], units that cogenerate steam and electricity [nonqualifying facilities], and other commercial and industrial units).

In the last few years, the electric utility industry has undergone a large restructuring brought on by the impending deregulation of the United States electric utility industry. Under this deregulation, consumers will be able to buy their electric power from any supplier willing to provide power in their area, thus breaking up the virtual monopolies that certain power companies and agencies have had since electric power was first provided.

Many older performing units are being sold or shut down for the sake of efficiency so that electric utility companies have an inventory of newer, more efficient units with modern pollution controls. Also, some companies are buying units in States where they traditionally have never supplied power in the past. In the next decade, these reorganizations will substantially change the makeup of electric power ownership in the utility industry.

## 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 (FBCs).

2.3.1.1 Stoker-Fired Boilers. 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.[7] The

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[c] 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."

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.[8] Because of their design, stokers are used only for smaller furnaces firing coal.

2.3.1.2 Cyclone-Fired Boilers. 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 furn ace. Because of this slagging system, cyclone-firing furnaces are almost exclusively coal-fired; however, some units can fire oil.[9]

2.3.1.3 Tangential-Fired Boilers. 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.[10] Tangential-fired boilers can fire coal, gas, or oil.

2.3.1.4 Wall-Fired Boilers. 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 walls that face each other). Circular register burners and cell burners are types of burner configurations found in single-wall or opposed-wall-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.[11] 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.[12] "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.[13]

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, whereas the BFB combustors have relatively low velocities and coarse bed-particle size.[14,15]

Most FBCs are of the atmospheric fluidized-bed combustor (AFBC) type, which, as the name suggests, operate at atmospheric pressure. A newer 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.[16]

2.3.1.6 Distribution of Furnace Types. Figure 2-2 shows the 1994 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 (48.8 percent), which represents 48.3 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 41.2 percent of the units (43.3 percent of the total megawatts), and cyclone firing is used in 8.5 percent of the units (8 percent of the total megawatts). Stoker-fired boilers and FBCs account for about 1.5 percent of designs among the coal-fired units (0.4 percent of the total coal-fired megawatts). Wall-fired designs represent the largest portion of gas- and oil-fired units by number of units (66 percent), which represents 62.1 percent of the total megawatts. The second most common design is the tangential-fired unit. Tangential-fired units represent 28.5 percent (31 percent of the total megawatts) of the gas- and oil-fired units, and combined-cycle gas turbine units account for about 5.3 percent (6.9 percent of the total megawatts) of designs for gas- and oil-fired units.[d] There is one known cyclone-fired unit fueled by oil. This unit represents 0.2 percent (0.1 percent of the total megawatts) of the 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. By comparison, essentially all elemental HAPs leaving the furnace enter with 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 13 provides a discussion, from

[d] It should be noted that the 1994 data for the number of individual combined-cycle turbine systems were not available. However, the number of plants (34, all gas-fired) and their total number of megawatts (10,047.87) was available, and these totals were included in Figure 2-2. This would have the effect of biasing the gas-fired boiler unit numbers lower than normal, but would have no effect on the gas-fired boiler MW numbers.

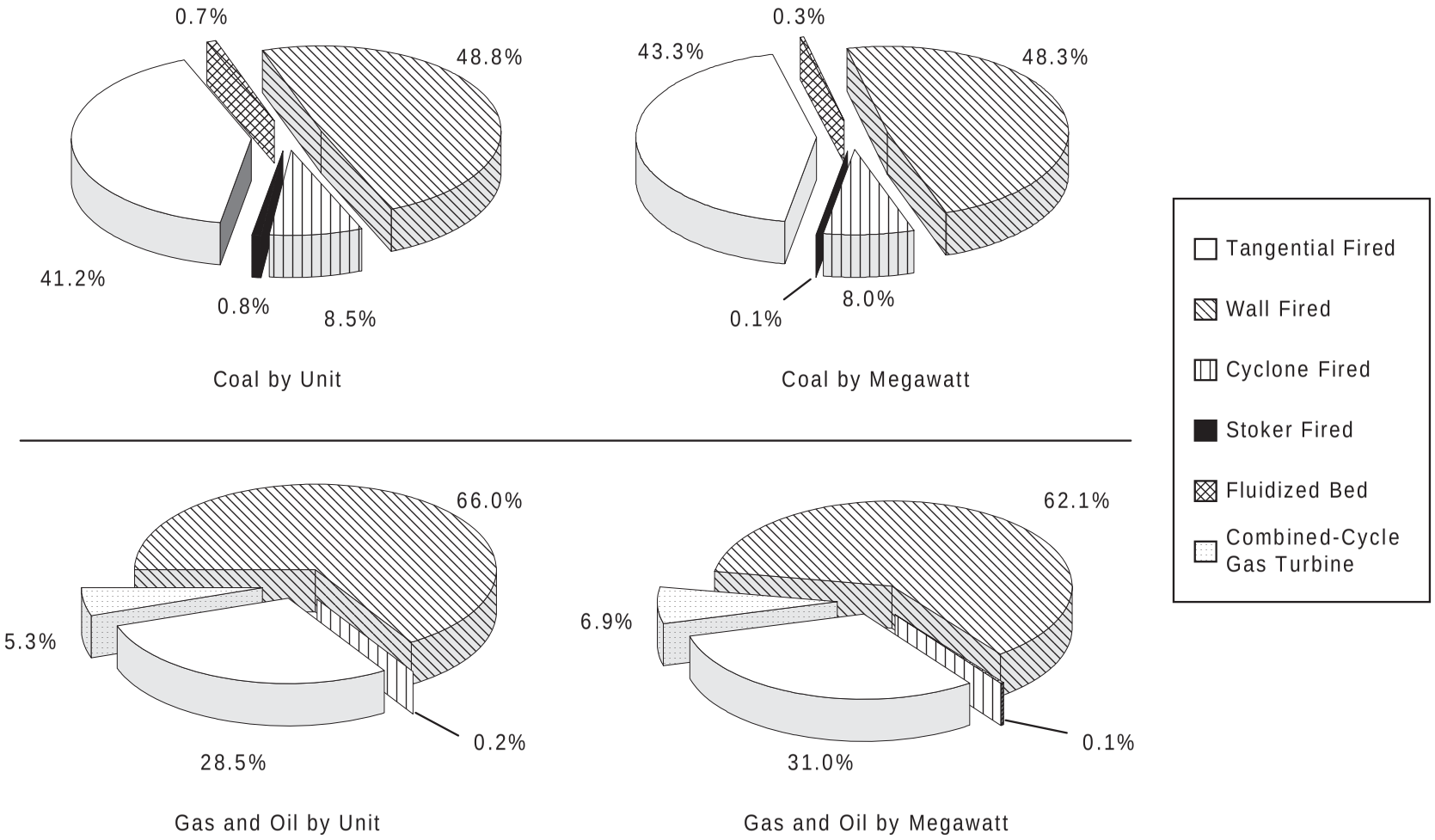



**Figure 2-2. Unit types in the utility industry by fuel type in 1994.[1]**

limited data, suggesting that, for example, organic HAP emissions are increased as furnace conditions are changed. Similarly for elemental HAPs, chapter 13 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 Nationwide emission factor program described later in this report. Appendix D describes the construction of the program 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 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.[17]

### 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.[18-20]

### 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. An example would be a combustion gas turbine's 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 1994 distribution of PM control by fuel in the utility industry by unit and by total MW.[1]

In 1994, 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 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 the largest portion of the oil-fired units (56 percent) and accounted for 48 percent of the oil-fired industry's total MWs. Electrostatic precipitators were used on 22 percent of the oil-fired units or at 27.4 percent of the MWe capacity of the oil-fired industry. Mechanical controls (cyclones) were used on 21 percent of the oil-fired units (24.5 percent of the total MWs). There is one known oil-fired unit controlled by a fabric filter. This unit represents 1 percent (0.1 percent of the total megawatts) of the oil-fired units. 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-micrometers (µm) to 50-µm range[21] (on a mass basis), HAPs tend to concentrate preferentially on particles smaller than about 7 µm, and especially on those around 0.3 µm.[22] 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. Much of the efficiency data by particle size originates 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 ($PM_{10}$).

### 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.[23] 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 (and was assumed to have no control effect on HAPs in the calculations).[24]

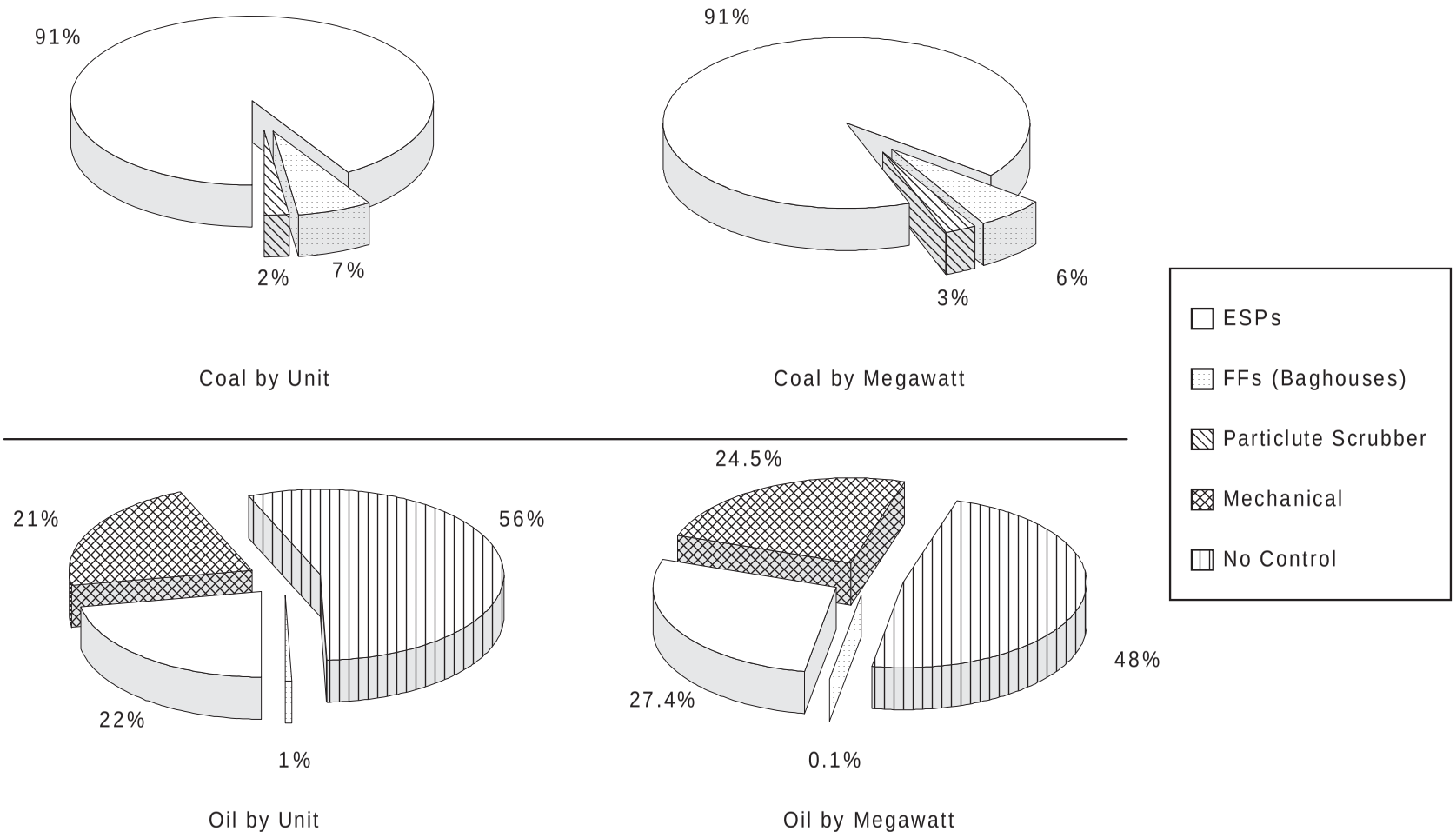



**Figure 2-3. Particulate control in the utility industry by fuel type in 1994.[1]**

### 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 this value, 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 (not to be confused with total PM) are typically collected with efficiencies from 95 to 99.9 percent.[25] Particles near the 0.3 µm size are in a charging transition region that reduces collection efficiency.[26] These particles have been shown to have lower collection efficiency (about 80 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.[27] 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.[28] 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 those of 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 $\mu$m, efficiency may be reduced to less than 50 percent.[29] 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.[30]

### 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 $\mu$m to about 10 $\mu$m (the range of the measuring equipment).[31] Because of its high collection efficiency for small particles, the baghouse should be particularly effective for removing particles that have been enriched with HAPs.[32,33] 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 13-6 and 13-9, 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.

Table 2-1. Comparison of Particulate Matter Collection Systems[34]

| Collector | Typical mass efficiency, % | Efficiency at 0.3 $\mu$m, % | 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.

## 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 (or flue gas desulfurization [FGD]) 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 1994, all oil- and gas-fired units burn compliance fuel, whereas 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 six coal-fired units, representing a total capacity of 815.6 MWe. These units are FBCs and control $SO_2$ in the combustion zone by using limestone as a sorbent.[1]

Figure 2-4 shows $SO_2$ control devices used in coal-fired utilities in 1994 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 FGD was used at approximately 14 percent of the units (approximately 21 percent of the coal-fired total electric capacity), whereas 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 pounds per million British thermal units heat input (lb/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 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 $SO_2$ emission limits. 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.[35] Approximately 77 percent of the eastern and midwestern bituminous coal shipments are subjected to some physical cleaning process.[36] Subbituminous and lignite coals are not routinely cleaned.[37,38] 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.[39] Bituminous coals from the eastern United States, 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.[40]

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.[35,41] 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,

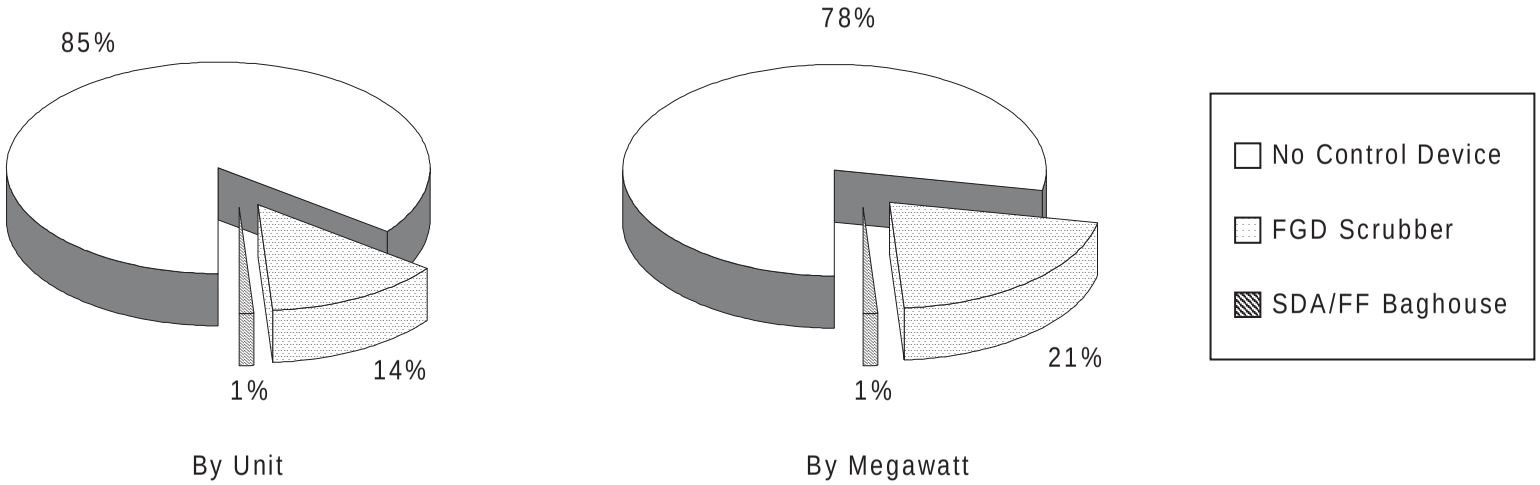



**Figure 2-4. $SO_2$ control in the utility industry in 1994 (coal-fired boilers only).[1]**

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 13.1.2.

2.5.2 Postcombustion Control: Flue Gas Scrubbing for $SO_2$ Control

According to the 1995 compilation of the Edison Electric Institute's (EEI) Power Statistics database (examining 1994 data), scrubbers were installed on 152 boiler units (out of about 1,026 coal-fired units in the United States) with a total rated capacity of 70,458 MWe.[1] Table 2-2 lists the different types of scrubbing installations used in United States utility power plants. As shown in Table 2-2, wet limestone/lime slurry scrubbing represents the 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 specifically constructed for the purpose.[42]

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.[43]

The basic wet limestone scrubbing process is simple and well established. Limestone sorbent is cheap and generally locally available in the United States. The $SO_2$ removal efficiencies of existing wet limestone scrubbers range from 31 to 97 percent, with an average of 78 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.

Table 2-2. Distribution of $SO_2$ Control Technologies in 1994[1]

| Scrubber type | No. of boiler units | Installed FGD capacity (MWe) | Total percent of installed FGD capacity (%) |
|---|---|---|---|
| Wet limestone | 70 | 35,101 | 49.8 |
| Wet lime | 44 | 21,172 | 30.0 |
| Dry lime/SDA | 15 | 5,615 | 8.0 |
| Sodium carbonate | 9 | 3,181 | 4.5 |
| Dual-alkali | 6 | 2,267 | 3.2 |
| Wellman-Lord | 4 | 1,779 | 2.5 |
| Mag-Ox | 3 | 895 | 1.3 |
| Dry aqueous carbonate | 1 | 450 | 0.7 |
| Total | | | 100.0 |

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

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 1994, wet limestone scrubbers were used at 70 units, or at 35,101 MWe of the total scrubbing capacity.[1]

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.[44]

Wet lime scrubbing is a proven technology; considerable operating experience has been gained in 44 utility units.[1] These units represented 21,172 MWe of the total scrubbing capacity in 1994. The $SO_2$ removal efficiencies of existing wet lime scrubbers range from 30 to 95 percent. Recent advances include the use of additives to improve performance, reduce scaling problems, and produce a salable gypsum byproduct. Lime scrubbing processes require 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.[45] 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 85 percent.[1] Spray dryers were used at 15 units and constituted 5,615 MWe of scrubbing capacity in 1994.

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 1994) using wet sodium carbonate scrubbing in the United States, 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 75 to 90 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 United States, 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.[46] 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 (1994) on six boiler units in the United States with a combined capacity of 2,267 MWe. The $SO_2$ removal efficiencies at these plants range from 77 to 93 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 $SO_2$.[47] The treated flue gas is demisted and reheated before it is exhausted through a stack. The Wellman-Lord process has been installed on four United States boiler units with a combined capacity of 1,779 MWe (1994), with $SO_2$ removal efficiencies ranging from 65 to 74 percent.[1]

2.5.2.7 Magnesium Oxide. Similar to Wellman-Lord, the magnesium oxide (MAG-OX) 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.[48]

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 United States with a combined capacity of 895 MWe (1994). The $SO_2$ removal efficiencies at these plants range from 85 to 94 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 (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.[49] Only one unit, of 450 MWe capacity, uses the dry aqueous carbonate system for FGD (1994); 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.[50-53]

Figure 2-5 shows $NO_x$ control approaches used in 1994 based on the number of units and total MW capacity.[1] Around 67 percent of coal-fired plants, representing about 50.4 percent of the coal-fired MW capacity, had no $NO_x$ control, whereas around 33 percent of the units, representing about 49.6 percent of the coal-fired MW capacity, used some kind of $NO_x$ control. Approximately 72 percent of the gas- and oil-fired units, with about 61 percent of the MW capacity, did not use $NO_x$ control, whereas approximately 28 percent of the units, representing about 39 percent of the gas- and oil-fired MW capacity, used some kind of $NO_x$ control. The gas- and oil-fired portion of Figure 2-5 does not contain data from combined-cycle turbine systems.

The chemical species nitrogen dioxide ($NO_2$) and nitric 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 $NO_2$.[54] 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.[55] 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).[56] Fuel switching, then, may yield reduced $NO_x$ emissions.

The formation of $NO_x$ in coal-fired units depends on factors such as the type of boiler, type of burner, and facility operation.[57] Any of these factors that increase temperature or residence time at high temperature will promote $NO_x$ formation.[58] In general, cyclone and other wet-bottom boilers have relatively higher $NO_x$ emissions, with an approximate range of 1 to 2 lb/MMBtu, than do dry-bottom boilers, which have an approximate range of 0.4 to 1.5 lb/MMBtu.[59] 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/MMBtu, 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.[60] 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.[61]



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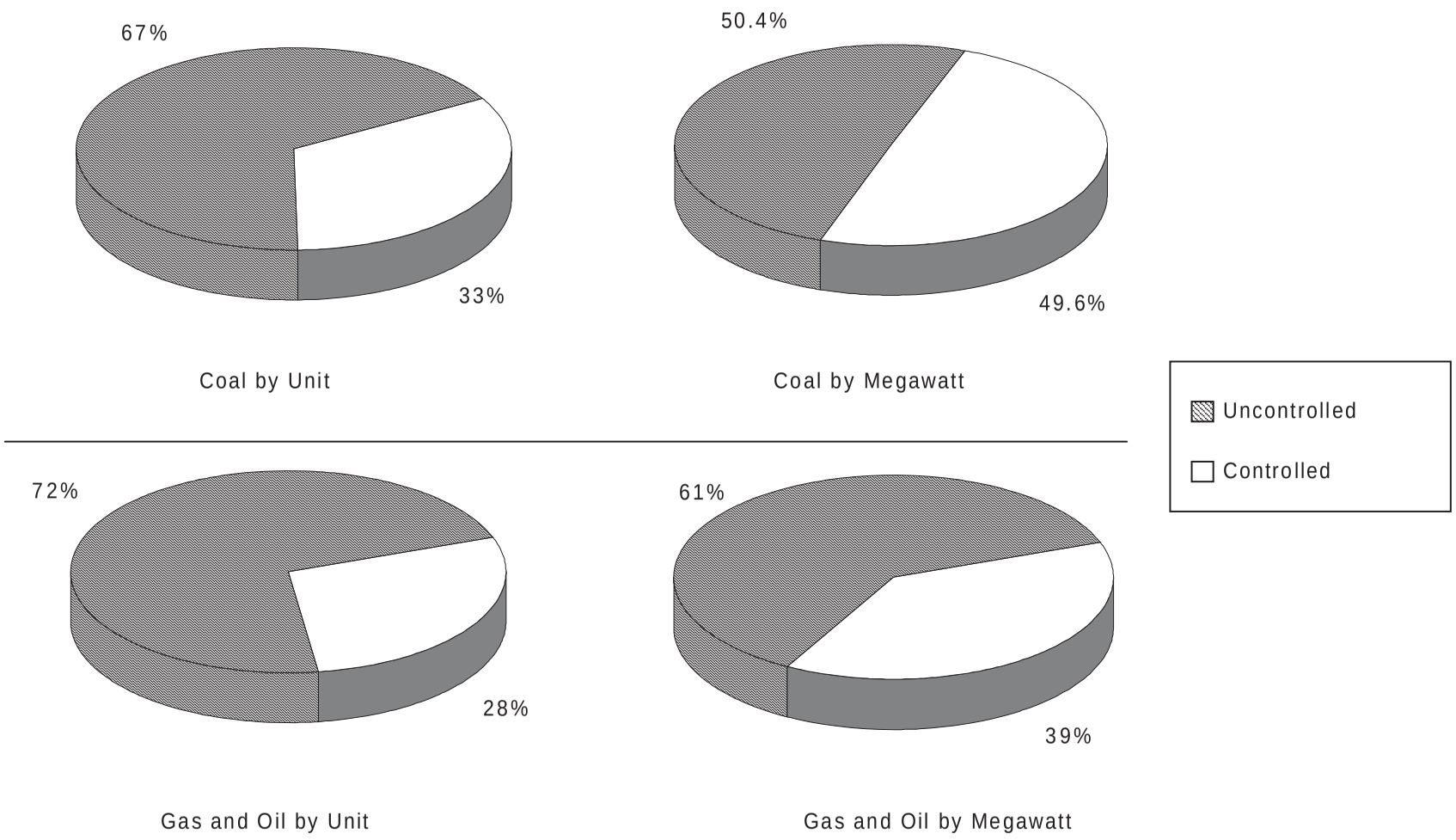



**Figure 2-5. Nitrogen oxide control in the utility industry by fuel type in 1994 .[1]**

The reduction of $NO_x$ emissions is important for controlling acid rain and ozone formation.[62] 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.[63] 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.[64] 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).[65] 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.[66]

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.[67] Overfire air may be used as a single $NO_x$ control technique, with $NO_x$ reductions of 15 to 50 percent.[68,69] When OFA is combined with low $NO_x$ burners, reductions of up to 60 percent may result.[70] The actual $NO_x$ reduction achieved with a given control technique may vary from site to site.[71]

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.[72] 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.[73,74] Reburn with other fuels, primarily coal, is currently under development, as are improvements in the process.[75]

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.[76] By using FGR to return part of the flue gas to the primary combustion zone, 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.

Temperatures may also be reduced in the primary combustion zone by increasing the spacing between burners for greater heat transfer to heat-absorbing surfaces.[77] 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 (CO) and soot emissions.[78]

### 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 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]).[79] Postcombustion control had, up to 1994, 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.[80] Since 1994, SCR has been installed on seven utility boiler units, five of which are cogeneration units.[81]

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.[82] Selective catalytic reduction units provide up to 70 to 90 percent $NO_x$ reduction[83] 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).[84]

Selective noncatalytic reduction is currently achieved commercially in one of two ways: THERMAL DeNO$_x$®, an Exxon process, or NO$_x$OUT®, an EPRI process. THERMAL DeNO$_x$® reduces $NO_x$ to nitrogen through injection of ammonia into the air-rich flue gas. NO$_x$OUT® achieves $NO_x$ reduction by injecting urea into the oxygen-rich and/or high-temperature convection part of the boiler.[85]

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 1995 EEI power statistics database (1994 data).[1] As shown in Table 2-3, most of the utility industry has no $NO_x$ control; 64 percent of the dry-bottom coal-fired boiler units, 87 percent of the wet-bottom coal-fired boiler units, 76 percent of the oil-fired boiler units, 70 percent of the gas-fired boiler units, and 100 percent of the combined-cycle turbine units had no $NO_x$ control in 1994 (see Note b in Table 2-3). 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 33 percent of the dry-bottom coal-fired units, 11 percent of the wet-bottom coal-fired units, 24 percent of the oil-fired units, and 30 percent of the gas-fired boiler units. Table 2-3 also shows that approximately 3 percent of the dry-bottom coal-fired units and 2 percent of the wet-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 Act 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 Title IV 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, repowering, 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.

Since the Interim Final Report to Congress, the EPA has obtained and analyzed current information and future projections on energy production (by fuel) in the electric utility industry.[86] It appears that the fuel usage projections listed below are being met and are proceeding toward what the EPA (using the Acid Rain Division's [ARD] model projections) predicted in the Interim Final Report to Congress. Therefore this section and the 2010 projected emissions were not changed in this Final Report.



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Table 2-3. Distribution of $NO_x$ Control by Fuel Burned, by Unit, in 1994 [1]

| | Percent $NO_x$ control[a] | | |
|---|---|---|---|
| **Fuel** | **None** | **Staged combustion** | **Boiler design** |
| Coal, boiler bottom | | | |
| Dry | 64 (46) | 33 (49) | 3 (5) |
| Wet | 87 (85) | 11 (12) | 2 (1) |
| Oil | 76 (66) | 24 (34) | -- |
| Gas | 70 (60) | 30 (40) | -- |
| Combined-cycle turbine | 100 (100)[b] | -- | -- |

[a] Values listed in parentheses are the percent distribution by MWe for each type of fuel.

[b] 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).[87] However, this technique is not used for utility boilers because it has a high efficiency penalty (about 10 percent).[87] Steam or water injection was present in the 1990 utility data but seems to missing from the 1994 utility data set. In the 1990 data, approximately 36 percent of the combined-cycle turbine units used steam or water injection for $NO_x$ control, whereas only approximately 2 percent of the boilers reported using this technique. The EPA believes that this technique is still being used but the companies that were surveyed neglected to include this in the responses.

### 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.[88] 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 ARD of the EPA's Office of Air and Radiation (OAR) by ICF Resources Incorporated. This single projection is used by the Office of Air Quality Planning and Standards (OAQPS) in this report to maintain consistency with the ARD.

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).[89] On this basis, the predominant fossil fuel both in 1990 and projected for 2010 is coal, at approximately 81 percent of the total industry fossil-fuel usage (22 quadrillion Btu/yr [Quads] in 2010). Oil and gas consumption in 1990 were, respectively, 6 and 13 percent of the total industry fossil-fuel usage on a Btu/yr basis. For 2010, oil consumption was projected to decrease to 2 percent (0.6 Quads), and gas consumption was projected to increase to 17 percent (4.5 Quads) on a Btu/yr basis for the total industry fossil-fuel usage.[90] Based on the ARD model projections, coal consumption in 2010 is expected to be the same percentage of the total utility fossil-fuel usage as in 1990 (81 percent).

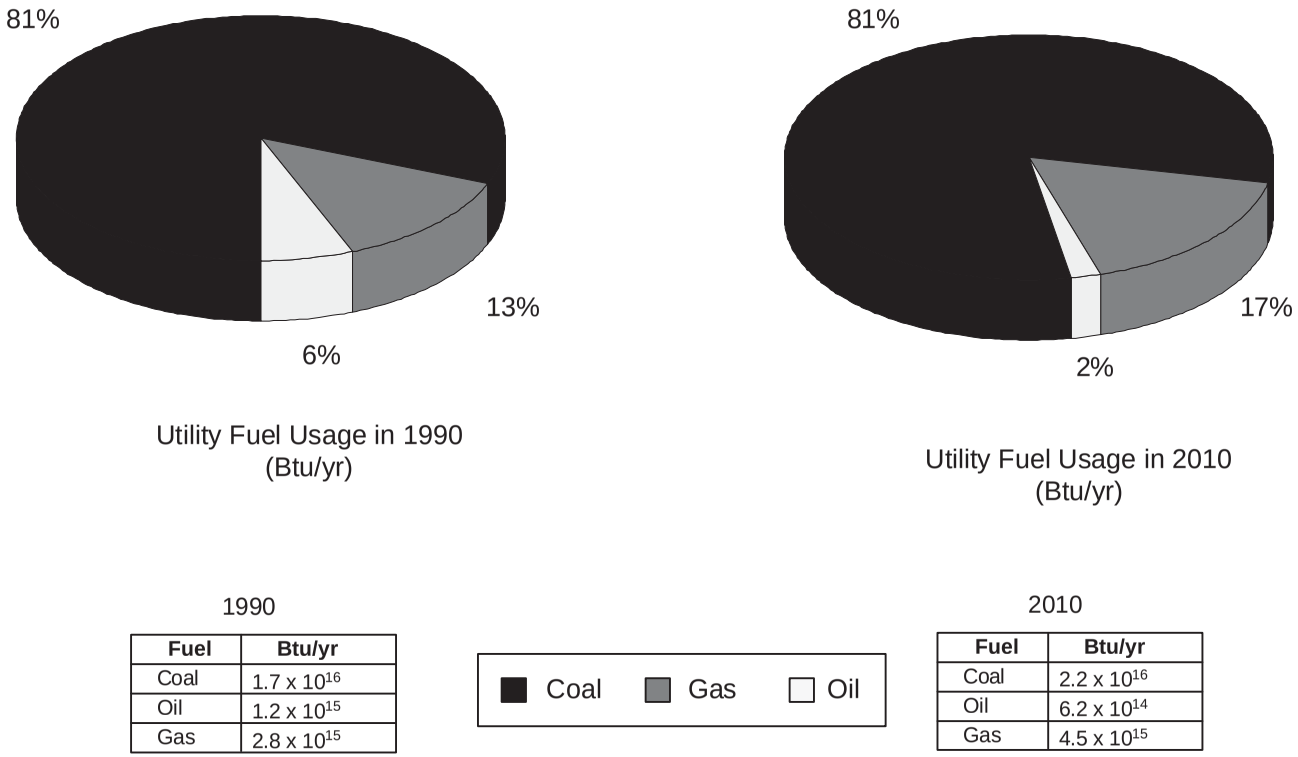



**Figure 2-6. Fuel use in the utility industry by fuel type in 1990 and projections for the year 2010.** [89,90]

Recent projections by the United States Department of Energy's (DOE) Energy Information Administration (EIA) were reviewed to determine the current validity of this projected fuel use scenario (see Table 2-4).[86] Based on the 1998 Annual Energy Outlook, coal use in the electric utility industry in 2010 is projected to be 21.34 Quads (73 percent of total utility fossil-fuel usage), oil use 0.35 Quads (1 percent), and natural gas use 7.38 Quads (25 percent). The EIA projections include consideration of issues related to decreased electricity production from nuclear power, lower coal prices, lower capital costs for coal-fired generating technologies, higher electricity demand, and industry restructuring. Although the share of coal generation declines in the 1998 estimate relative to earlier projections (primarily due to restructuring considerations), the projection for coal-fired fuel use in 2010 is higher than the 1997 projection and is essentially the same as that of EPA's ARD. Thus, the Agency has not changed its projections related to emissions in 2010.

Figure 2-7 shows the projected growth of each utility fuel between 1990 and 2010.[90] Between 1990 and 2010, fuel consumption is projected to change as follows: coal will increase by 29 percent, oil will decrease by 48 percent, and natural gas will increase by 61 percent. Based on the recent EIA projection noted above, coal consumption will increase by 26 percent, oil consumption will decrease by 71 percent, and natural gas consumption will increase by 164 percent.

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) 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 can estimate neither the size nor the location of the new units, the increased consumption has been allocated to existing units (in 2010) for the analyses. This allocation is believed reasonable because (1) many new units would be built on the site of existing utility facilities (thus, "co-locating" the emissions) and (2) the analyses are based on total fuel used (rather than on capacity factor, etc.).

The decrease in oil consumption could result in (or result from) units being retired or in a decrease in capacity factor, or a mix. The decreased consumption has been allocated among those oil-fired units EPA believes will be operative 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.

Table 2-4. Fuel Use in the Electric Utility Industry by Fuel Type, Quadrillion Btu/yr

| Fuel | EPA, projection | | EIA, actual | | | | | | | EIA, projection | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1990[a] | 2010[a] | 1990[b] | 1991[b] | 1992[b] | 1993[b] | 1994[b] | 1995[b] | 1996[b] | 2010[c] | | 2010[d] |
| | | | | | | | | | | Reference case | Forecast ranges | |
| Coal | 17 | 22 | 16.19 | 16.03 | 16.21 | 16.79 | 16.90 | 16.99 | 17.93 | 19.91 | 19.31 - 21.09 | 21.34 |
| Oil | 1.2 | 0.62 | 1.25 | 1.18 | 0.95 | 1.05 | 0.97 | 0.66 | 0.73 | 0.57 | 0.56 - 0.6 | 0.35 |
| Gas | 2.8 | 4.5 | 2.88 | 2.86 | 2.83 | 2.74 | 3.05 | 3.28 | 2.80 | 7.09 | 6.37 - 7.89 | 7.38 |

[a] Utility Data Institute.[91]
[b] U.S. Department of Energy, Energy Information Agency.[92]
[c] Annual Energy Outlook. 1997.[93]
[d] Annual Energy Outlook. 1998.[94]

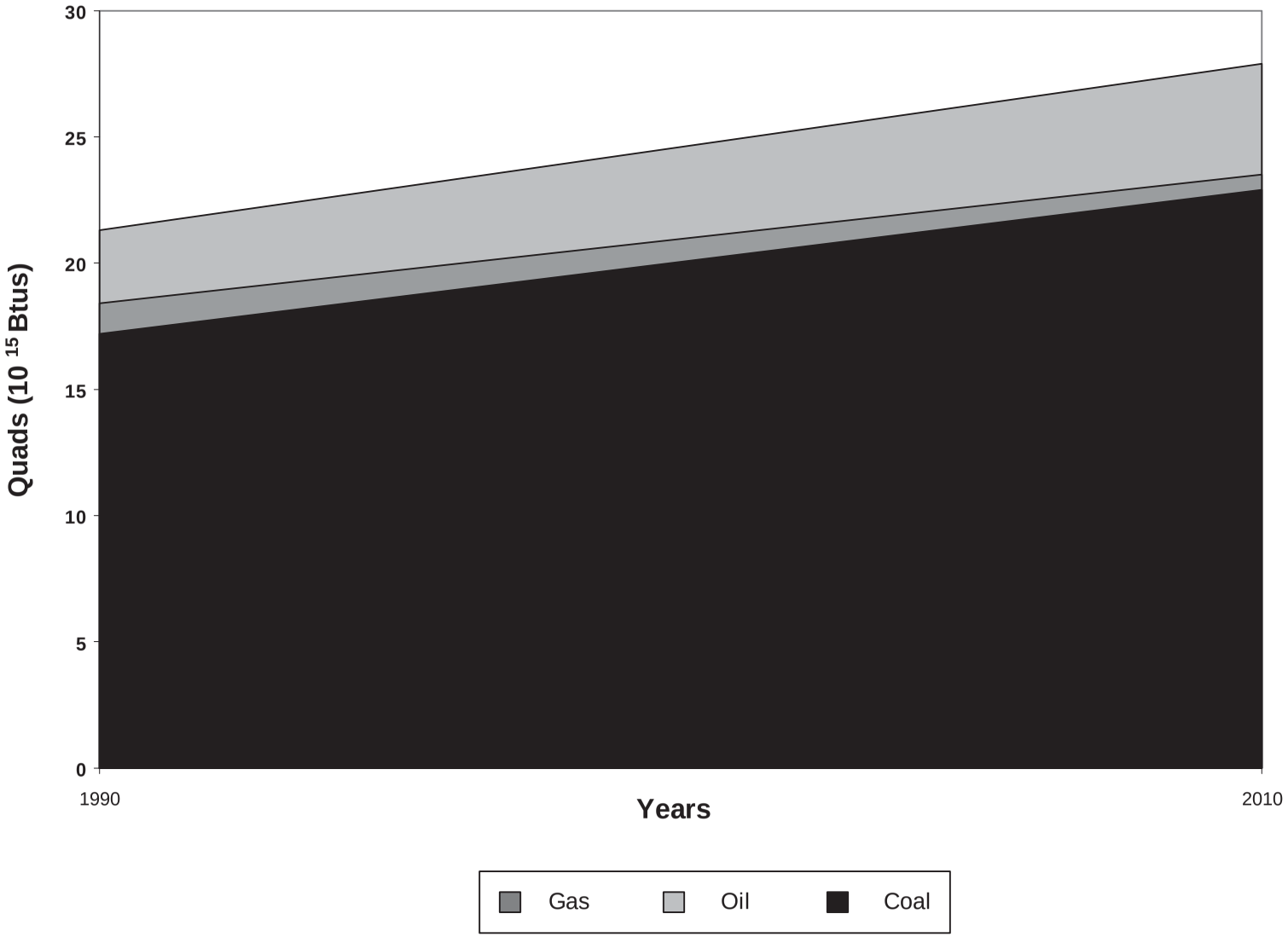



**Figure 2-7. Projected use of fuels by 2010 for utility industry.** [89]

The Agency projects that 135 units will be retired during the period from 1990 to 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 Program establish a cap on the national, annual $SO_2$ emissions. To achieve compliance with the requirements, utilities may use 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, but not natural gas-fired) replacement, (4) use of a Phase II compensating unit, or (5) purchase of emission allowances.

The Phase I requirements affected 261 generating units (435 with substitution or compensating generating units).[95] The 174 substitution or compensating generating units are not included in the following discussion. Examining the method used by the 261 Phase I generating units to comply with the provisions, the following was found:[96]

- 53 percent (136 units accounting for 59 percent of the 1995 $SO_2$ emission reductions) switched to a coal that contains less sulfur
- 27 percent (83 units accounting for 9 percent of the 1995 $SO_2$ emission reductions) purchased additional emission allowances
- 16 percent (27 units accounting for 28 percent of the 1995 $SO_2$ emission reductions) installed flue gas scrubbers
- 2 percent (7 units accounting for 2 percent of the 1995 $SO_2$ emission reductions) were retired
- 2 percent (8 units accounting for 2 percent of the 1995 $SO_2$ emission reductions) either repowered using new boiler technology, or switched to natural gas or low sulfur oil.

Each of the 27 units known to be installing scrubber units was modeled with the scrubber unit in place for the 2010 scenario. The EPA modeled the remaining 234 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 Program, an additional approximately 1,600 generating units will be covered by the year 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 by using alternate methods.[97] This assumption is based on several factors including: (1) the increased availability of low-sulfur coal at favorable prices; (2) the introduction of processes that reduce $SO_2$ 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.[98]

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 versus 2010 analysis. No change in a unit's burner configuration (i.e., "old" versus new low-$NO_x$) was included in the 2010 scenario. The impact of low-$NO_x$ burner installation is discussed in chapter 13.

Under the Acid Rain Program, the rules for $NO_x$ control (40 CFR Part 7b) require tangential-fired and dry-bottom wall-fired boilers subject to Phase I $SO_2$ reduction requirements to meet annual average $NO_x$ emission limits of 0.45 lb/MMBtu and 0.50 lb/MMBtu, respectively, by January 1, 1996. Utilities can meet the Title I and IV requirements by installing low-$NO_x$ burner technology or by averaging emissions among several units.

Since the Interim Report to Congress, additional rules (e.g., revised NAAQS, revised NSPS, and Acid Rain Program for $NO_x$ and $SO_x$) have been promulgated that could pressure the electric utility industry to consider options beyond those considered earlier. These include additional fuel switching or the adoption of SCR or SNCR rather than the addition of low-$NO_x$ burners because of tightening $NO_x$ emission standards. Since limited data were available assessing the HAP removal potential of SCR and SNCR, they were not addressed in the 2010 program output. Also, additional FGD units may be installed to comply with the revised PM NAAQS (which impact on sulfate rather than traditional PM). However, as no area has been determined to be in non-compliance yet, no units have been planned and such installations were not addressed in the 2010 analyses. To the extent that additional FGD units are installed (for NSPS or NAAQS compliance) the overall effect could be lower HAP emissions in 2010.

Under Phase II of the Acid Rain Program, the EPA established $NO_x$ emission limits for all other boilers, including wet-bottom wall-fired boilers and cyclones, by January 1, 1997; affected units must be in compliance by January 1, 2000.[99] EPA also revised the emission limits for dry-bottom wall-fired boilers and 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.[97] 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.[100] 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.[101] It is not known how transient these excess emissions will be.

### 2.7.3 Compliance Strategy Impacts of Other Activities

Other activities, not directly related to CAA mandates, will also impact on electric utility industry control strategies and emissions. These include developments related to electricity industry restructuring, such the Federal Energy Regulatory Commission's (FERC) Open Access Rule (Order 888) finalized in April 1996, changes in the energy production from other sources (e.g., accelerated retirement of nuclear plants, increases in the use of biomass), overall national demand for electricity, relative differences in fuel costs, and any future "global warming" abatement initiatives. Responses to these activities (i.e., specific control strategies) were not included in the analyses for this report. As can be seen from the EIA projections presented earlier, it is expected that, over the long-term, natural gas will increase its share of the fossil-fuel generation. However, it should be noted (see Table 2-4) that between 1995 and 1996 (the first year of industry restructuring), coal consumption for electricity generation increased by approximately one quad while natural gas consumption decreased by approximately 0.5 quad. Preliminary data for 1997 indicate that coal use continues to increase while natural gas use has leveled off.[102] It is not known what factors are involved in this trend (e.g., nuclear outages, relative fuel prices, seasonal weather conditions) but some parties believe that restructuring is playing a role.[103] In addition, it is not known how this short-term trend may ultimately factor into long-term projections.

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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 data in the literature 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 beginning in 1990 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 beginning in late 1993.

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 performed 36 emissions tests at 34 locations 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 eight test reports from five of its coal-fired plants for testing performed from 1990 to 1992.

The DOE, through what is now its Federal Energy Technology Center (FETC), 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 several of the DOE emission tests.

The EPA also completed the initial development of the Fourier transform infrared (FTIR) spectrometry 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 58 tests were conducted at 52 sites using FTIR and conventional sampling and analysis methods from the EPRI, the DOE, the NSPC, and the EPA. Although 58 test reports were received by EPA in time for inclusion in this study, 4 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. For draft versions of this report, emissions were estimated for 1990 based on the 48 usable tests then available. Estimates for 1994 are based on the 54 usable tests. Emission estimates for 1994 use the same modeling procedure as for 1990, but with minor revisions as noted hereafter.

Data reliability and the precision and accuracy of the analytical techniques for each test were addressed by the individual test contractors in their test reports. Where the contractor had major concerns about the quality of the data or found gross departures from expected precision or accuracy of a particular test analysis, the EPA refrained from using the data in its computations.

## 3.2 POLLUTANTS STUDIED

As many as possible of the 189 HAPs listed in section 112(b) were included in this study. Table A-1 (Appendix A) lists the organic HAPs that were detected at least once in the utility test data (excluding FTIR-detected data), the estimated nationwide HAP emissions in 1990 and 1994, and the 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 United States (plus any additional unit types tested that were below this cutoff). The matrix was then used only 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. It should be noted that the totals in Table B-1 were taken from the 1991 EEI Power Statistics Database and do not correlate with the 1994 industry statistics given in chapter 2.

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, whereas 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.

### 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 the UDI/EEI Power Statistics database (1991 and 1994 editions) and an extract from Production Costs, U.S. Gas Turbine and Combined-Cycle Power Plants (for 1994 estimates). These databases were 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 (EMF) 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

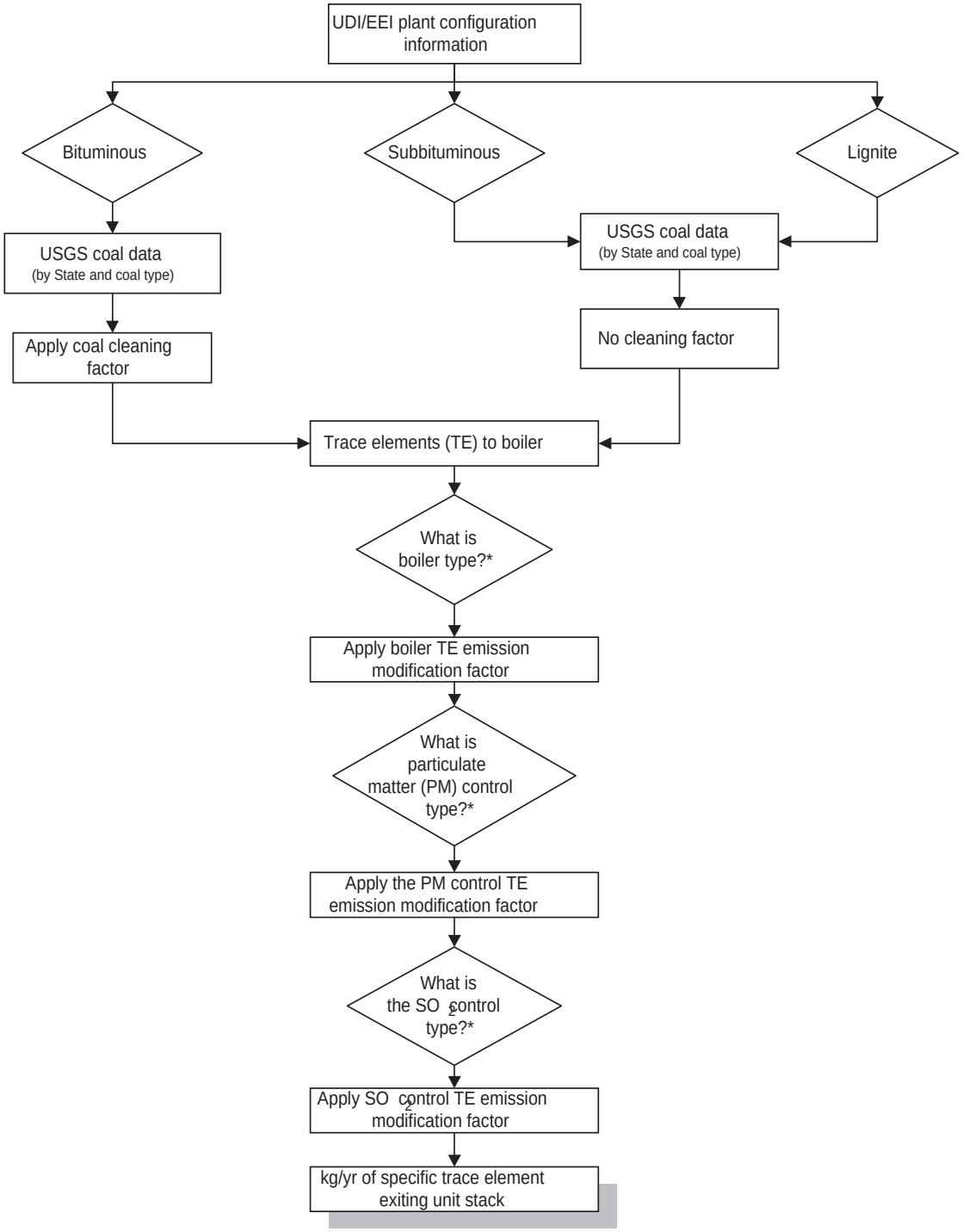



**Figure 3-1. Trace elements in coal.**

*Taken from UDI/EEI data.

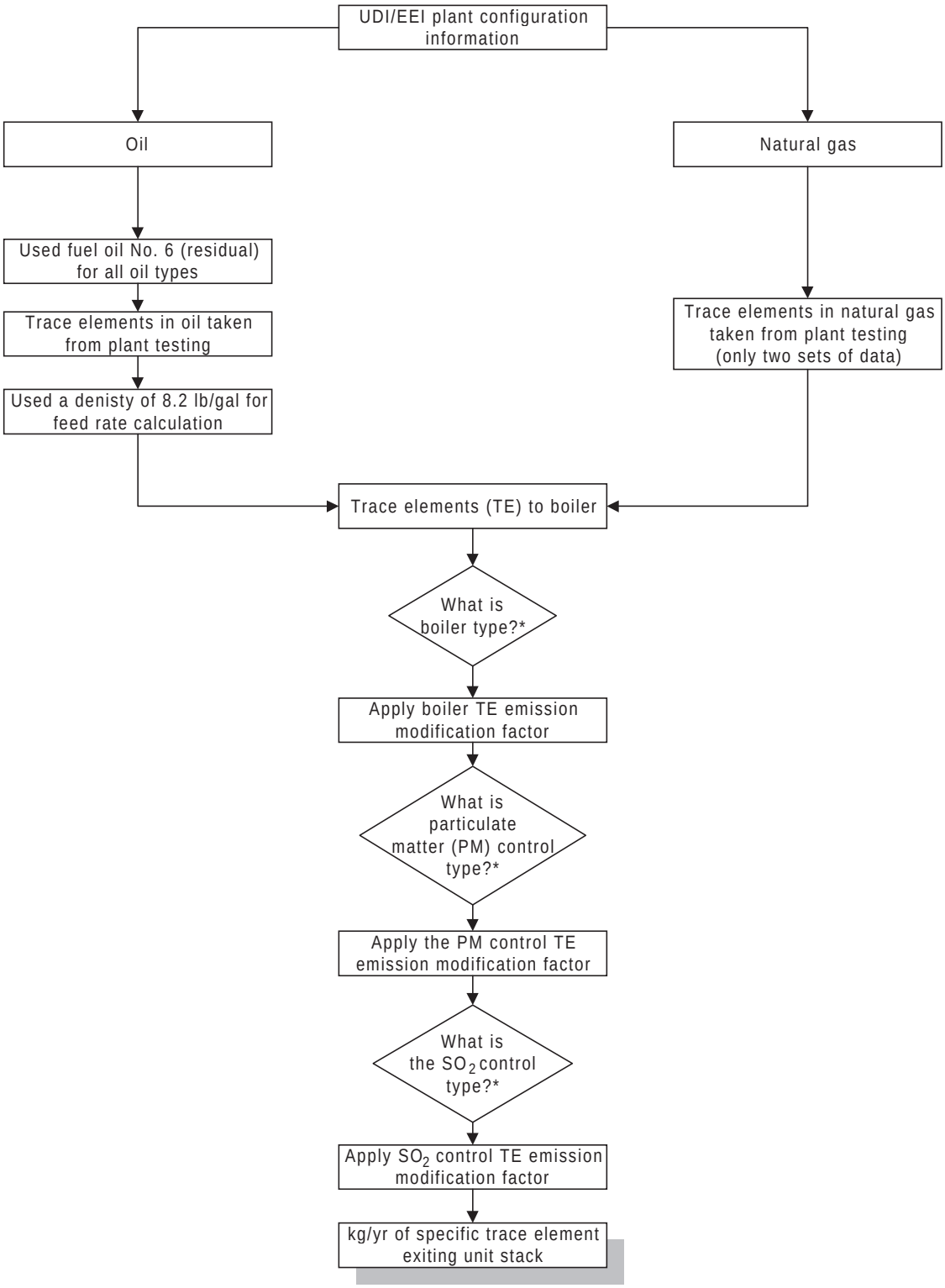
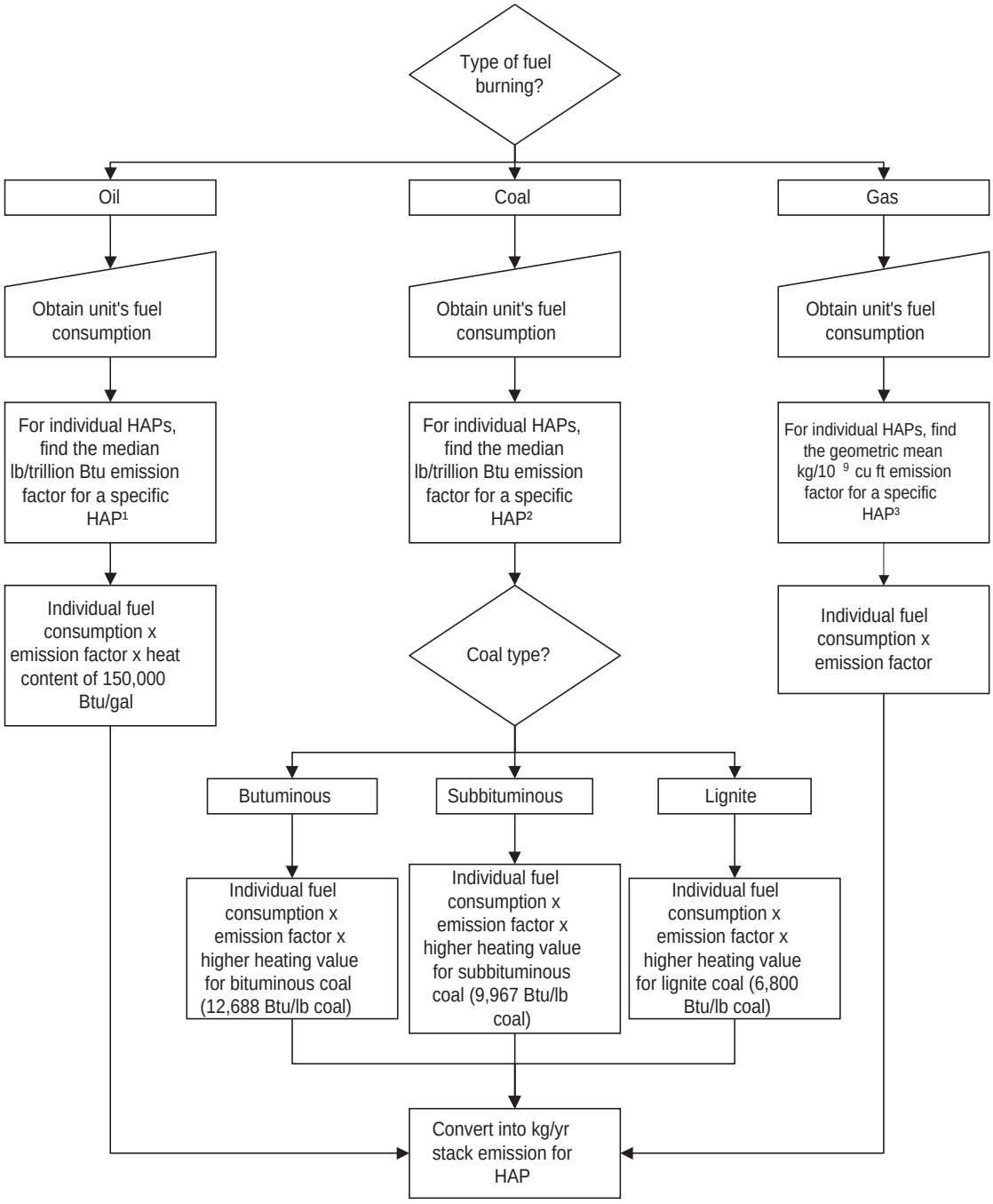



**Figure 3-2. Trace elements in oil and natural gas.**

*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.**

State of origin in the United States Geological Survey (USGS) database, which contains analyses of 3,331 core and channel samples of coal. The samples come from either the top 50 economically feasible coal seams in the United States during 1990, or from seams associated with 1991 coal receipts for electric utility plants.

### 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) or the similar file for 1994 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. Other units listed on indefinite standby (i.e., no fuel burned) were excluded from 1994 emission estimates.

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 of >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.

The 1990 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.[5] 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. Similar problems in the 1994 UDI/EEI database were solved by using 1990 data where possible and by similar methods to those stated above when not possible.

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.[6] 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 [7] to meet customer specifications on heat, ash, and sulfur content, a coal cleaning factor was applied to most bituminous coals in the EFP.[8] Two exceptions were bituminous coals from Illinois and Colorado, for which analyses were on an as-shipped basis representative of the coal to be fired.

For a unit that burned bituminous coal, the feed rate in kilograms per year (kg/yr) 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 Appendix D for listings of trace elements in coal, coal cleaning factors, and equations (Nos. 1 and 2 in Table D-2) used in the EFP.

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 test reports on gas-fired units 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.[9] For example, for each lb/hr of chloride in the feed coal at one of the test sites, 0.63 lb/hr of HCl was found in the gas stream leaving the boiler. Similarly for HF, the boiler emissions were 0.64 lb/hr for each lb/hr of fluoride in the coal. 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 0.63 or 0.64, respectively. The resulting numbers allowed direct conversion into boiler emissions that could be further modified for systems with PM control or $SO_2$ control. For the 1990 emission estimates, before obtaining further test reports, the factors were 0.61 for HCl and 0.56 for HF.

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 3-1, for coals originating from these States.[10]

### 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 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.[11] These EMFs are averaged by taking the geometric mean of similar devices (e.g., all oil-fired tangential boilers, all cold-side ESPs). Geometric means are 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 are then applied to the kg/yr feed rates entering the boiler, the effect of which either reduces or leaves unchanged the emissions that pass through them. Those EMFs calculated as being greater than 1.0 (i.e., more material exiting a device than entering it) were set to equal 1.0.

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. The EMFs developed for 1990 emission estimates were revised to include additional test report data for 1994 emission estimates. 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 generally 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 or 1994. After accounting for coal cleaning (bituminous coal only), the emission concentration of an inorganic HAP was converted to 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 kg/yr 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.

Table 3-1. Assigned Chloride and HCl Concentrations in Coal, by State of Coal Origin [10]

| State | Conversion of assigned ppmw chloride to assigned HCl ppmw | Assigned ppmw HCl in coal |
|---|---|---|
| Alaska | 54 x 0.63 = | 34.0 |
| Illinois | 1,136 x 0.63 = | 715.7 |
| Indiana | 1,033 x 0.63 = | 650.8 |
| Iowa | 1,498 x 0.63 = | 943.7 |
| Missouri | 1,701 x 0.63 = | 1,071.6 |
| Utah | 220 x 0.63 = | 138.6 |
| Washington | 104 x 0.63 = | 65.5 |

Appendix C contains all of the EMFs used to develop the 1990 and 1994 unit emission estimates for inorganic HAPs.

3.4.7 Acid Gas HAPs

The method used with HCl or HF emissions allowed direct conversion from coal chlorine or fluorine content into boiler emissions, as described earlier, 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 test reports in which contractors conducted tests individually for HCl, chlorine, HF, and fluorine before and after each control device. These tests were in contrast to the remaining tests for which HCl and HF values were estimated or omitted rather than measured.

The next steps after obtaining amounts of HCl or HF leaving the boiler were to construct EMFs for the PM control device, and then for the $SO_2$ control device. Using chlorine as an example, the measured amount of HCl entering the PM control device (in kg/yr with suitable conversion factors) was compared with the measured amount of HCl leaving the PM control device. Using these two quantities, an EMF was formed as described in section 3.4.6.

In the final step, EMFs were formed for HCl and HF through the $SO_2$ control device based on the measured mass of HCl or HF entering that device (leaving the PM control device) and the mass measured at the exit of the $SO_2$ control device. However, a modification was required to account for flue-gas bypass around the $SO_2$ control device. A portion of the flue gas is bypassed to maintain $SO_2$ removal at the minimum permitted amount. This action is used as a means of reducing energy required to reheat the flue gas for effective plume rise from the stack. In developing the HCl and HF EMFs for wet FGDs and dry scrubbers, the effect of flue gas bypass was treated by analyzing

utility test data from the four plants (of eight tested) that used bypasses, reviewing municipal waste incinerator results that showed a typical HCl or HF removal efficiency of 95 percent, and having discussions with industry representatives. Based on the 95 percent removal efficiency coupled with the measured values for quantity of flue gas bypassed, an industry average effective value for flue gas bypass in 1994 was estimated. The value was assumed to be 15 percent (17 percent for 1990 data) for wet FGDs and 14 percent (for 1990 and 1994 data) for dry scrubber systems. These assumptions were used only in the development of HCl and HF EMFs.[12]

### 3.4.8 Organic HAPs

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.

Organic stack emissions from coal-fired boilers were first determined on an emission factor basis (lb/trillion Btu) to account for different coal heating values, then converted to a rate basis (kg/yr of individual HAP). This procedure was necessary because different coal ranks had 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 3-2).[13]

If stack emission or APCD exit emission data were reported as nondetected, and, if at least one-third of the data samples at the inlet of the APCD were detected concentrations with values comparable to the nondetected outlet values, EPA used the inlet data directly as a measure of outlet concentration at the stack. If the outlet nondetected values were significantly different from the inlet detected values, the data were not used. 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.

### 3.4.9 Model Estimates for the Year 2010

Emission estimates for 2010 were derived from the same basic 1990 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.

In summary, the input files for the 2010 analyses were modified to account for the expected increases in nationwide coal and natural gas, and expected decrease in oil, use in the utility industry resulting from the industry growth described in section 2.7.1. Units

Table 3-2. Average Higher Heating Values of Coal[14]

| 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.

known to be retired during the period 1990-2010 were removed from the input files. Announced options (described in section 2.7.2) taken for compliance with the Acid Rain Program (e.g., coal switching, FGD installation) were accounted for in the input files. Thus, emission changes resulting from these activities were included in the 2010 analyses.

However, as noted in section 2.7.3, control strategies taken in response to such activities as industry restructuring and global warming abatement initiatives were not included in the 2010 analyses. To the extent that such strategies include increased use of FGD units or natural gas, the projected 2010 HAP emissions could be over-estimated in this report. For example, analyses performed to assess compliance with the revised national ambient air quality standards (NAAQS) for ozone and PM indicate that mercury emissions in 2010 may be reduced by approximately 16 percent (11 tons per year) over those projected in this report, primarily due to increased FGD use and fuel switching.[15] To the extent that such strategies include the increased use of coal, as may be happening at least in the short term under industry restructuring, the projected 2010 HAP emissions could be underestimated in this report.

## 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, 1994, and 2010 nationwide emissions of this subset of HAPs from utility units (see also Table A-1, Appendix A). With expected changes in input data and new test data for EMFs, the opportunity was taken to improve the model for more effective use in estimating 1994 emissions. Depending on reported and projected fuel switching and fuel use, units brought on line or taken off line, the revisions to EMFs between 1990 and 1994, and improvements in the model's handling of fuel combustion, estimated HAP emissions across the 1990 to 2010 span may increase uniformly, decrease uniformly, peak in 1994, or show a minimum in 1994. These varied responses among the HAPs are thought to occur primarily because of industry changes rather than changes to the model.

## 3.6 COMPARISON OF EFP ESTIMATES WITH TEST DATA

Comparisons were made between test data from 19 utility boiler stacks and 1994 predicted emissions for the same plants using the EFP.[16] 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. It should be stressed that the EFP is designed to provide a reasonable estimate of nationwide emissions based on summing a large number of imperfect boiler estimates that are expected to cluster about a reasonable estimate of the true value. The EFP program also provides reasonable estimates for the boiler-specific emissions used for the exposure modeling analyses (see chapter 6). However, there are uncertainties and, based on an uncertainty analysis, it is estimated that emission values may be over- or underestimated by as much as roughly a factor of three for any specific boiler.

Table 3-3. Selected Nationwide HAP Emissions

| | Selected nationwide HAP emissions (estimated) in tons/year for 1990 | Selected nationwide HAP emissions (estimated) in tons/year for 1994 | Selected nationwide HAP emissions (estimated) in tons/year for 2010 |
|---|---|---|---|
| ***Coal-fired electric utility plants*** | | | |
| **Arsenic** | 60.93 | 55.81 | 70.61 |
| **Beryllium** | 7.13 | 7.93 | 8.20 |
| **Cadmium** | 3.33 | 3.15 | 3.82 |
| **Chromium** | 73.27 | 61.60 | 87.43 |
| **Lead** | 75.47 | 61.77 | 86.89 |
| **Manganese** | 163.97 | 167.72 | 219.02 |
| **Mercury** | 45.80 | 51.34 | 59.74 |
| **Hydrogen chloride** | 143,000 | 134,000 | 155,000 |
| **Hydrogen fluoride** | 19,500 | 23,100 | 25,700 |
| **2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)** | 0.000097 | 0.000121 | 0.000108 |
| **n-nitrosodimethylamine** | 5.84 | 6.09 | 7.73 |
| | | | |
| ***Oil-fired electric utility plants*** | | | |
| **Arsenic** | 5.02 | 3.51 | 2.54 |
| **Beryllium** | 0.46 | 0.40 | 0.23 |
| **Cadmium** | 1.71 | 1.09 | 0.86 |
| **Chromium** | 4.74 | 3.91 | 2.40 |
| **Lead** | 10.58 | 8.92 | 5.35 |
| **Manganese** | 9.28 | 7.30 | 4.70 |
| **Mercury** | 0.25 | 0.19 | 0.13 |
| **Nickel** | 392.8 | 322.0 | 198.2 |
| **Hydrogen chloride** | 2,860 | 2,100 | 1,450 |
| **2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)** | 0.000016 | 0.000009 | 0.000003 |
| | | | |
| ***Natural-gas-fired electric utility plants*** | | | |
| **Arsenic** | 0.15 | 0.18 | 0.25 |
| **Nickel** | 2.19 | 2.42 | 3.49 |
| **Formaldehyde** | 35.62 | 39.23 | 56.58 |

For three elements and 19 boilers, averages for estimates of three individual elements (arsenic, chromium, and nickel) were different from the test values by +60, -32, and -3 percent, respectively. The highest individual difference between predicted and reported values was represented by a factor of 2,600.

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. Plants 1 through 17 fire coal, whereas plants 18 and 19 fire oil. At least one of the plants fires a combination of coal and petroleum coke.

Possible reasons were examined for large differences between projected and actual emissions. In the 1990 EFP, only one fuel (and if coal, from only one State) was assumed to be burned. Although the 1994 version of the EFP was designed to accommodate multiple types of fuel, the one-State-of-origin restriction for coal was still used. However, at least one of the plants burned combinations of coal and petroleum coke, but the EFP recognized only coal from one State. The petroleum coke used by the plant had nickel concentrations that may have been more than 100 times higher than that found in the Montana coal assigned to that plant by the EFP, and concentrations of nickel in the plant's 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 2,600 as mentioned above. The EFP was not sufficiently detailed to recognize mixtures of coal and petroleum coke (i.e., the EFP did not include the nickel contributed by the petroleum coke). Plants burning such mixtures will likely have their nickel emissions underestimated by the EFP.

## 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 United States. Once the specific plant was chosen, its 1994 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).

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 | 0.14 | 0.0035 |
| 2 | 0.24 | 0.10 | 0.30 |
| 3 | 1.84 | 0.48 | 0.13 |
| 4 | 1.66 | 1.30 | 1.97 |
| 5 | 9.83 | 1.63 | 1.08 |
| 6 | 0.92 | 0.26 | 0.36 |
| 7 | 2.00 | 0.51 | 0.81 |
| 8 | 0.18 | 1.17 | 10.00 |
| 9 | 0.0010 | 0.0036 | 0.0004 |
| 10 | 0.65 | 0.27 | 0.20 |
| 11 | 0.20 | 0.19 | 0.16 |
| 12 | 0.09 | 0.0041 | 0.0730 |
| 13 | 0.04 | 2.40 | 1.37 |
| 14 | 4.13 | 1.15 | 0.17 |
| 15 | 0.01 | 0.04 | 0.14 |
| 16 | 6.40 | 0.72 | 0.93 |
| 17 | 2.01 | 1.23 | 0.65 |
| 18 | 0.05 | -- | 0.02 |
| 19 | 0.03 | -- | 0.03 |
| | | | |
| Average: EFP/test | 1.60 | 0.68 | 0.97 |

EFP = emission factor program

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

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 1994 fuel consumption values, and the emission testing (on which the EFP is based) was performed under essentially steady-state conditions (with little or no variation from the baseline operating condition). Therefore, the characteristic emissions from testing 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. That is why the fuel heat content was used as an indicator of emissions rather than the plant capacity or utilization.

Table 3-5. Emissions from a Characteristic Coal-Fired Electric Utility Plant (1994)

| | 1994 |
|---|---|
| **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 1994** |
| **Arsenic** | 0.050 |
| **Beryllium** | 0.0081 |
| **Cadmium** | 0.0023 |
| **Chromium** | 0.110 |
| **Lead** | 0.021 |
| **Manganese** | 0.092 |
| **Mercury** | 0.045 |
| **Hydrogen chloride** | 191.8 |
| **Hydrogen fluoride** | 14.31 |
| **2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)** | 0.000000103 |
| **n-nitrosodimethylamine** | 0.0052 |

Table 3-6. Emissions from a Characteristic Oil-Fired Electric Utility Plant (1994)

| | 1994 |
|---|---|
| **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 1994** |
| **Arsenic** | 0.0062 |
| **Beryllium** | 0.0002 |
| **Cadmium** | 0.0140 |
| **Chromium** | 0.0062 |
| **Lead** | 0.014 |
| **Manganese** | 0.019 |
| **Mercury** | 0.0012 |
| **Nickel** | 1.69 |
| **Hydrogen chloride** | 9.4 |
| **2,3,7,8-tetrachlorodibenzo-p-dioxin (TEQ)** | 2.27E-08 |

Table 3-7. Emissions from a Characteristic Natural Gas-Fired Electric Utility Plant (1994)

| | 1994 |
|---|---|
| **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 1994** |
| **Arsenic** | 0.0003 |
| **Nickel** | 0.0041 |
| **Formaldehyde** | 0.067 |

## 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.

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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 National Research Council 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 used in 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 (NAS) 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, and presentations of central tendency and worst-case portions of the range of risk; 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, 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 multipathway analyses of radionuclides, mercury, arsenic, and dioxins; a long-range transport modeling analysis for mercury, arsenic, chromium, nickel, cadmium, and lead; 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 a qualitative discussion of lead and cadmium. Chapter 9 presents the assessment for radionuclides. Chapter 10 and 11 present screening level multipathway assessments for arsenic and dioxins, respectively; and Chapter 12 discusses potential impacts of hydrogen chloride and hydrogen fluoride emissions.

## 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 EPA's Integrated Risk Information System (IRIS). IRIS is an online database maintained by the EPA, which contains 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 used. If other data sources were used, they are indicated by footnotes in tables, end notes, 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 chemical 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

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 |

> 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 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 mean 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 noncancer hazards. The RfD is expressed in units of 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 and other health effects information for HAPs identified in the emissions data.

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 and non-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 for air pollutants typically 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. The emissions estimates are believed to be

reasonable estimates of the emissions from the utility plants. However, there are uncertainties in the emissions estimates. The emissions estimates, which were calculated using the geometric mean of the test data, are believed to be central tendency estimates. That is, the true emissions could be higher or lower than predicted with the emissions model. Based on an uncertainty analysis, the EPA predicts that the emissions estimates for any individual plant are likely to be within a factor of plus or minus three of actual emissions (see chapters 3 and 6 for further discussion of emissions data and emissions estimates).

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 ARray (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 United States 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; 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 ($\mu g/m^3$) 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 Cancer 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 ($\mu g/m^3$) 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 $\mu g/m^3$) of a pollutant in the air. The true cancer risk due to exposure to any particular HAP is unknown and unlikely to be higher than that predicted with the IURE and could be lower, possibly as low as zero.

If the highest modeled ambient air concentration occurs in an area (i.e., 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 based on the assumption 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 $3 \times 10^{-6}$, and the MIR for arsenic from coal-fired utilities was $2 \times 10^{-6}$.

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 were 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.[7] 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, cancer risks and noncancer risks are estimated differently. 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 (e.g., 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.[7,16]

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. If the HI exceeds unity, the HI should be reevaluated using HQ values summed only for noncarcinogens with similar target organs based on EPA risk assessment guidelines[7] and assuming 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[17] that contains information about 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 and assumptions 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.

- The closest meteorological station to each utility plant is assumed to represent the weather patterns at the utility plant site.

- 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 the assumption that 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 impacts of using some of these default values (e.g., emissions estimates, indoor vs. outdoor concentrations, breathing rates, closest meteorological station, flat terrain and others) are evaluated in sections 6.12 and 6.13 and Appendix G of the EPA's 1996 Interim Utility Air Toxics Report. Other default parameters are described and discussed in Appendix F and in various sections of chapter 6 of this report.

## 4.6 REFERENCES


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# 5.0 SCREENING 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 plant for 66 of the 67 HAPs identified in the emissions database. The emissions estimates used for all the HEM modeling presented in this study (chapters 5 and 6) were estimates of annual average emissions for all plants (see chapter 3 for details on how emissions estimates were calculated). The HEM modeling predicted annual average ambient air concentrations in all the census blocks located within 50 km of each facility (see chapters 4, 6, and Appendix F for detailed discussions of the HEM modeling for this study,and the input data, assumptions and default parameters). 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., $1 \times 10^{-7}$), 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 priorities 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 about 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 | IURE[b] ($\mu g/m^3$) | MEI cancer risk[c] | Primary type of cancer assoc. w/ inhalation[d] | MEI cancer risk >$10^{-7}$ |
|---|---|---|---|---|---|---|
| Arsenic compounds | 0.0014 | A | 0.0043 | > $10^{-6}$ | Lung | Yes |
| Chromium (11 percent VI)[e] | 0.0023 | A | 0.0016[e] | > $10^{-6}$ | Lung | Yes |
| Beryllium | 0.00025 | B2 | 0.0024 | > $10^{-7}$ | Lung | Yes |
| Cadmium | 0.00009 | B2 | 0.0018 | > $10^{-7}$ | Lung | Yes |
| Nickel compounds[f] | 0.0027 | A[f] | 0.00048 | > $10^{-7}$ | Lung & nasal | Yes |
| Dioxins[g] | 2 x $10^{-9}$ | B2 | 30.0[g] | 7 x $10^{-8}$ | Tongue, lung, nasal, liver | No |
| PAHs[h] | 0.00002 | B2 | 0.0021[h] | 4 x $10^{-8}$ | Lung (BAP) | No |
| Naphthalene | 0.00009 | C | 4 x $10^{-6}$ | 4 x $10^{-10}$ | NA | No |
| Hexaclorobenzene | 9 x $10^{-6}$ | B2 | 0.00046 | 4 x $10^{-9}$ | NA | No |
| Carbon tetrachloride | 0.00038 | B2 | 0.000015 | 6 x $10^{-9}$ | Liver | No |
| Quinoline[i] | 0.000006 | C | 0.0035[j] | 2 x $10^{-8}$ | NA | No |
| Vinylidene chloride | 0.0011 | C | 5 x $10^{-5}$ | 6 x $10^{-8}$ | NA | No |
| Formaldehyde | 0.00047 | B1 | 1 x $10^{-5}$ | 6 x $10^{-7}$ | Nasal, lung | No |
| n-Nitrosodimethylamine[j] | 0.00008 | B1 | 0.014 | 1 x $10^{-6}$ | Liver & other | Yes[j] |
| 1,1,2-Trichloroethane | 0.00054 | C | 2 x $10^{-5}$ | 9 x $10^{-9}$ | NA | No |
| Acetaldehyde | 0.00078 | B2 | 2 x $10^{-6}$ | 2 x $10^{-9}$ | Nose & larynx | No |
| Benzene | 0.00029 | A | 8 x $10^{-6}$ | 2 x $10^{-9}$ | Leukemia | No |
| Benzyl chloride | 6 x $10^{-7}$ | B2 | 5 x $10^{-5}$ | 3 x $10^{-11}$ | NA | No |
| Bis(2-ethylhexyl)phthalate[i] | 0.00047 | B2 | 4 x $10^{-6}$ [i] | 2 x $10^{-9}$ | NA | No |
| Bromoform | 0.00077 | B2 | 1 x $10^{-6}$ | 9 x $10^{-10}$ | NA | No |
| Chloroform | 0.00037 | B2 | 2 x $10^{-5}$ | 9 x $10^{-9}$ | Kidney & liver | No |
| Ethylene dichloride | 0.00036 | B2 | 3 x $10^{-5}$ | 9 x $10^{-9}$ | NA | No |
| Isophorone[i] | 0.003 | C | 3 x $10^{-7}$ [i] | 9 x $10^{-10}$ | NA | No |
| Methyl chloride[i] | 0.0007 | C | 2 x $10^{-6}$ [i] | 1 x $10^{-9}$ | Kidney | No |

(continued)

Table 5-1. (Continued)

| Hazardous air pollutant | Highest MEI conc.[a] (µg/m$^3$) | EPA WOE | IURE[b] (µg/m$^3$) | MEI cancer risk[c] | Primary type of cancer assoc. w/ inhalation[d] | MEI cancer risk >$10^{-7}$ |
|---|---|---|---|---|---|---|
| Methylene chloride | 0.0015 | B2 | $5 \times 10^{-7}$ | $7 \times 10^{-10}$ | Liver & lung | No |
| Trichloroethylene[i] | 0.00036 | B2/C | $2 \times 10^{-6}$ [i] | $6 \times 10^{-10}$ | Lung, liver, & testicular | No |
| Pentachlorophenol[i] | $1 \times 10^{-6}$ | B2 | $3 \times 10^{-5}$ [i] | $3 \times 10^{-11}$ | NA | No |
| Tetrachloroethylene[i] | 0.00036 | B2/C | $6 \times 10^{-7}$ [i] | $2 \times 10^{-10}$ | Liver | No |

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.
MEI = Maximally Exposed Individual.
NA = Not available.
WOE = Weight of Evidence, for carcinogenicity. See section 4.3.1 and Table 4-1. for explanation of WOE.

[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.
[b] IUREs obtained from EPA's Integrated Risk Information System (IRIS),[2] unless indicated otherwise by footnotes.
[c] This is the estimated increased lifetime cancer risk to the highest MEI due to inhalation exposure.
[d] 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.
[e] 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 limited speciation data described in Appendix H.[3] 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 ($1.2 \times 10^{-2}$) by 0.11 (11 percent).
[f] 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.
[g] The emissions were estimated using the toxic equivalency (TEQ) approach described in the draft EPA Dioxin Reassessment Report.[4] Exposure was estimated by modeling the TEQ emissions with HEM. The IURE is for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and was obtained from the draft EPA Dioxin Reassessment Report.
[h] 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).[5] (These are listed in Appendix H).[3] 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) ($2.1 \times 10^{-3}$). 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 $2.1 \times 10^{-4}$ per µg/L to inhalation units. The conversion assumes equal absorption and metabolism and assumes equal risk from the different routes of exposure, which may not be the case.
[i] 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.*[6] 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.
[j] 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. Based on available information, the MEI risk estimate presented here is likely to be a significant overestimate of the true risks posed by n-Nitrosodimethylamine.

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 ($\mu g/m^3$) | Noncancer health effect on which RfC is based[a] | Confidence in RfC[b] | Highest MEI conc.[c] ($\mu g/m^3$) | Max. HQ | Highest HQ[d] > 0.1 |
|---|---|---|---|---|---|---|
| 2-Chloro-acetophenone | 0.03 | Hyperplasia of nasal resp. epith. in rats | low | $3 \times 10^{-5}$ | 0.001 | No |
| Acrolein | 0.02 | Metaplasia and inflammation rat nasal epithel. | med | $4 \times 10^{-4}$ | 0.02 | No |
| Cumene[e] | 9.0[e] | — | NA | 0.00003 | $3 \times 10^{-6}$ | No |
| Ethyl benzene | 1000 | Developmental effects | low | 0.00005 | $5 \times 10^{-8}$ | No |
| Ethyl chloride | 10000 | Delayed fetal ossification | med | 0.0003 | $2 \times 10^{-8}$ | No |
| Hexane | 200 | CNS & nasal epith. lesions humans | med | 0.00009 | $5 \times 10^{-6}$ | 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[f,g] | 1.5[f] | CNS & devel. humans | NA | 0.007 | 0.0057 | No |
| Manganese | 0.05 | CNS, humans | med | 0.02 | 0.4 | Yes |
| Mercury[g] | — | — | -- | 0.001 | -- | No |
| Methyl bromide | 5.0 | Lesions of olfactory epithelium | high | 0.0001 | $2 \times 10^{-5}$ | No |
| Methyl chloroform[e] | 1000[e] | Hepatotoxicity | NA | 0.0004 | $4 \times 10^{-7}$ | No |
| Methyl ethyl ketone | 1000 | Decreased fetal birth weight (mice) | low | 0.0009 | $9 \times 10^{-7}$ | No |
| MTBE | 3000 | Increased liver & kidney weight (rat) | med | 0.0002 | $7 \times 10^{-8}$ | No |
| Styrene | 1000 | CNS in humans | med | 0.00036 | $4 \times 10^{-7}$ | No |
| Toluene | 400 | Neurological effects; degeneration of nasal epithelium | med | 0.0004 | $1 \times 10^{-6}$ | No |

(continued)

Table 5-2. (Continued)

| Hazardous air pollutant | RfC ($\mu g/m^3$) | Noncancer health effect on which RfC is based[a] | Confidence in RfC[b] | Highest MEI conc.[c] ($\mu g/m^3$) | Max. HQ | Highest HQ[d] > 0.1 |
|---|---|---|---|---|---|---|
| 1,3-Dichloropropene | 20 | Hypertrophy/hyperplasia of nasal respiratory epithelium | high | 0.00054 | $3 \times 10^{-5}$ | No |
| Vinyl acetate | 200 | Nasal epithelium lesions | high | 0.00005 | $3 \times 10^{-7}$ | No |

CNS = Central nervous system.
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).
MEI = Maximally Exposed Individual.
NA = Not applicable.
RfC = Reference concentration (inhalation).

[a] This is the critical adverse noncancer health effect that was observed in animal or human studies.[2,7]
[b] This is the overall confidence in the RfC as reported on IRIS.
[c] 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.
[d] 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.
[e] The RfC was obtained from the 1992 EPA Health Effects Summary Tables.[8] It has not been verified by the EPA's RfC/RfD workgroup.
[f] No RfC is available for lead compounds. Therefore, as a substitute, the lead National Ambient Air Quality Standard (1.5 $\mu g/m^3$) was used in this assessment.[9] However, the lead NAAQS is note 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 carcinogen.[2,6]
[g] 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 discussed 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[a] ($\mu g/m^3$) | OSHA PEL/420[a] ($\mu g/m^3$) | ACGIH TLV/420[a] ($\mu g/m^3$) | Highest MEI conc.[b] ($\mu g/m^3$) | Maximum HQ | HQ > 0.1 |
|---|---|---|---|---|---|---|
| Acetophenone | NA | NA | NA | 0.00008 | NA | NA |
| Antimony compounds | 1.2 | 1.2 | 1.2 | 0.0005 | $4 \times 10^{-4}$ | No |
| Carbon disulfide | 7.1 | 29 | 74 | 0.0005 | $7 \times 10^{-5}$ | No |
| Chlorobenzene | NA | 833 | 830 | 0.00037 | $4 \times 10^{-7}$ | No |
| Cobalt compounds | 0.12 | 0.12 | 0.12 | 0.0017 | 0.014 | No |
| o & p-Cresols | 24 | 52 | | 0.0003 | $1 \times 10^{-5}$ | No |
| Cumene | 580 | 580 | 580 | 0.00003 | $3 \times 10^{-6}$ | No |
| Dibutyl phthalate | 11.9 | 11.9 | 11.9 | 0.00033 | $3 \times 10^{-5}$ | No |
| Hydrogen fluoride | 6.0 | 6.0 | 6.0[c] | 0.365 | 0.06 | No |
| Methyl methacrylate | NA | 980 | 980[c] | 0.00013 | $1 \times 10^{-7}$ | No |
| MIBK | 490 | 980 | 490 | 0.00058 | $1 \times 10^{-6}$ | No |
| Phenol | 48 | 45 | 45[c] | $7 \times 10^{-4}$ | $2 \times 10^{-5}$ | No |
| Phthalic anhydride | 14 | 14 | 14 | $6 \times 10^{-4}$ | $4 \times 10^{-5}$ | 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 | $5 \times 10^{-7}$ | No |
| 2,4-Dinitrotoluene | 3.6 | 3.6 | 0.36 | $1 \times 10^{-6}$ | $3 \times 10^{-6}$ | No |
| Methyl iodide | 24 | 67 | 29 | 0.00005 | $2 \times 10^{-6}$ | No |

ACGIH = American Conference of Government Industrial Hygienists, which is a professional society, not a government agency.
HQ = Hazard Quotient. The ratio of exposure concentration/RfC. An HQ < 0.1 indicates that no adverse health effects are expected to occur (see Chapter 4 for discussion of HQ).
MEI = Maximally Exposed Individual.
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

Footnotes for Table 5-3 (continued)

PEL = Permissible Exposure Levels. These are legal limits established by OSHA.
REL = Recommended Exposure Levels. NIOSH develops these 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.

[a] The NIOSH, OSHA, and ACGIH are primarily involved with the safety and health of workers. The RELs, PELs, and TLVs are similar. Breathing concentrations below these levels are expected to be reasonably protective of health workers, exposed for 8 hours per day, 5 days per week (~40 hours). However, there are uncertainties and often the data are less then complete. Also, for some of these values (especially the PELs), measurement techniques and economic factors are sometimes factored in. [10,11,12]
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 a 40 hr/week to a 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. [13] CARB also divides the TLV by 420 to calculate some of their noncancer reference exposure levels (4.2 to account for exposure time adjustment, 10 to account for sensitive individuals, and another 10 because health effects are sometimes observed at the TLV level).

[b] 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."[13]

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.[a] ($\mu g/m^3$) | EPA WOE | IURE[b] per $\mu g/m^3$ | Cancer MEI Risk[c] | Primary type of cancer associated w/ Inhalation[d] | MEI cancer risk >$10^{-7}$ |
|---|---|---|---|---|---|---|
| Arsenic | 0.0032 | A | 0.0043 | > $10^{-6}$ | Lung | Yes |
| Chromium (18 percent Cr VI)[e] | 0.0025 | A | 0.0028 | > $10^{-6}$ | Lung | Yes |
| Beryllium | 0.0003 | B2 | 0.0024 | > $10^{-7}$ | Lung | Yes |
| Cadmium | 0.0009 | B2 | 0.0018 | > $10^{-6}$ | Lung | Yes |
| Nickel compounds[f] | 0.21 | A2 | 0.00048 | > $10^{-6}$ | Lung & nasal | Yes |
| Dioxins[g] | 4 x $10^{-9}$ | B2 | 30.0 | 1 x $10^{-7}$ | Tongue, lung, nasal, liver, thyroid | Yes |
| PAHs[h] | 0.00003 | B2 | 0.0021 | 6 x $10^{-8}$ | Lung (BAP) | No |
| Formaldehyde | 0.007 | B1 | 1.3 x $10^{-5}$ | 9 x $10^{-8}$ | Nasal, lung | No |
| Acetaldehyde | 0.0019 | B2 | 2.2 x $10^{-6}$ | 4 x $10^{-9}$ | Nasal & laryngeal | No |
| Benzene | 0.0003 | A | 8.3 x $10^{-6}$ | 3 x $10^{-9}$ | Leukemia | No |
| Methylene chloride | 0.008 | B2 | 4.7 x $10^{-7}$ | 4 x $10^{-9}$ | Liver & lung | No |
| Naphthalene | 0.00008 | C | 4.2 x $10^{-6}$ | 3 x $10^{-10}$ | -- | No |
| Tetrachloroethylene[i] | 0.00013 | B2/C | 5.8 x $10^{-7}$ | 8 x $10^{-11}$ | Liver | No |

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.
MEI = Maximally Exposed Individual.
WOE = Weight of Evidence, for carcinogenicity. See section 4.3.1 and Table 4-1. for explanation of WOE.

[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.
[b] IUREs obtained from EPA's Integrated Risk Information System (IRIS)[2], unless indicated otherwise by footnotes.
[c] This is the estimated increased lifetime cancer risk to the highest MEI due to inhalation exposure.
[d] 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.
[e] 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.3 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 x $10^{-2}$) by 0.11 (11 percent).
[f] 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.
[g] The emissions were estimated using the toxic equivalency (TEQ) approach described in the draft EPA Dioxin Reassessment Report.[4] Exposure was estimated by modeling the TEQ emissions with HEM. The IURE is for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and was obtained from the draft EPA Dioxin Reassessment Report.

Footnotes for Table 5-4. (Continued)

[h] 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).[5] (These are listed in Appendix H3). 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) ($2.1 \times 10^{-3}$). 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 $2.1 \times 10^{-4}$ per $\mu$g/L to inhalation units. The conversion assumes equal absorption and metabolism and assumes equal risk from the different routes of exposure, which may not be the case.

[i] 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*.[6] 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.

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 ($\mu g/m^3$) | Critical noncancer health effect that RfC is based on[a] | Overall confidence in RfC[b] | Highest MEI[c] conc. ($\mu g/m^3$) | Highest HQ | HQ >0.1[d] |
|---|---|---|---|---|---|---|
| Ethyl benzene | 1000 | Developmental effects | Low | $1 \times 10^{-4}$ | $1 \times 10^{-7}$ | No |
| Hydrogen chloride | 20 | Hyperplasia of nasal mucosa, larynx, and trachea in rats | Low | 1.1 | 0.16 | Yes |
| Lead[e] | 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[f] | 1000[f] | Hepatotoxicity[f] | NA | 0.0018 | $2 \times 10^{-6}$ | No |
| Toluene | 400 | Neurological effects | Medium | 0.002 | $5 \times 10^{-6}$ | No |
| Vinyl acetate | 200 | Nasal lesions | High | 0.0012 | $6 \times 10^{-6}$ | No |

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).
MEI = Maximally Exposed Individual.
NA = Not applicable.
RfC = Reference concentration (inhalation).

[a] This is the critical adverse noncancer health effect that was observed in animal or human studies.[2,7]
[b] This is the overall confidence in the RfC as reported on IRIS.
[c] 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.
[d] 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.
[e] No RfC is available for lead compounds. Therefore, as a substitute, the lead National Ambient Air Quality Standard (1.5 $\mu g/m^3$) was used in this assessment.[9] However, the lead NAAQS is note 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 carcinogen.[2,6]
[f] 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 discussed in section 5.5.

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[a] ($\mu g/m^3$) | OSHA PEL/420[a] ($\mu g/m^3$) | ACGIH TLV/420[a] ($\mu g/m^3$) | Highest MEI conc. ($\mu g/m^3$)[b] | Max HQ | 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 | No |
| Selenium compounds | 0.48 | 0.48 | NA | 0.001 | 0.002 | No |
| m,o,p-Xylenes | 1040 | 1040 | 1040 | 0.0005 | $5 \times 10^{-7}$ | No |

ACGIH = American Conference of Government Industrial Hygienists, which is a professional society, not a government agency.
HQ = Hazard Quotient. the ratio of exposure concentration/RfC. An HQ < 0.1 indicates that no adverse health effects are expected to occur (see Chapter 4 for discussion of HQ).
MEI = Maximally Exposed Individual.
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
PEL = Permissible Exposure Levels. These are legal limits established by OSHA.
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.

[a] The NIOSH, OSHA, and ACGIH are primarily involved with the safety and health of workers. The RELs, PELs, and TLVs are similar. Breathing concentrations below these levels are expected to be reasonably protective of health workers, exposed for 8 hours per day, 5 days per week (~40 hours). However, there are uncertainties and often the data are less then complete. Also, for some of these values (especially the PELs), measurement techniques and economic factors are sometimes factored in. [10,11,12] 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 a 40 hr/week to a 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. [13] CARB also divides the TLV by 420 to calculate some of their noncancer reference exposure levels (4.2 to account for exposure time adjustment, 10 to account for sensitive individuals, and another 10 because health effects are sometimes observed at the TLV level).

[b] 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.

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

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

HEM = Human Exposure Model
HQ = Hazardous 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).
IURE = Inhalation Unit Risk Estimates
MEI = Maximally Exposed Individual.

[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.
[b] IUREs obtained from EPA's Integrated Risk Information System (IRIS),[2] unless indicated otherwise by footnotes.
[c] This is the estimated increased lifetime cancer risk to the highest MEI due to inhalation exposure.
[d] 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.
[e] 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.
[f] These values are not RfCs. They are TLV/420. See Tables 5-3 and 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.[14] 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.[15] Many studies indicate that deposition of atmospherically emitted pollutants can result in indirect avenues of exposure for humans.[16] 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.[15] 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.[17] 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.[17]

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, 8, 10, and 11 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*.[18] 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 five HAPs were chosen 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 $1 \times 10^{-3}$]), as well as the HAPs with EPA-verified OUREs and a WOE classifica-tion of A or B, were selected. Also, due to their extremely 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

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 |

HAP = Hazardous air pollutant
TCDD = Tetrachlorodibenzo-p-dioxins
PAH = Polycyclic aromatic hydrocarbons
POM = Polycyclic organic matter

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 polycyclic 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) nationwide. 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. 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

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[a] (ton/yr) | Oil-fired emissions estimates[a] (ton/yr) |
|---|---|---|---|---|---|
| 2,3,7,8-TCDD (dioxins)(TEQ) | NA | $3 \times 10^{+0}$ [b] | B2 | $9.7 \times 10^{-5}$ | $0.69 \times 10^{-5}$ |
| Lead compounds | No threshold [c] | -- | B2 | $7.5 \times 10^{+1}$ | 11 |
| Mercury compounds | $1 \times 10^{-4}$ | -- | C | $4.6 \times 10^{+1}$ | 0.3 |
| Arsenic compounds[d] | $3 \times 10^{-4}$ | $5.0 \times 10^{-5}$ | A | $6.1 \times 10^{+1}$ | 5 |
| Cadmium compounds | $5 \times 10^{-4}$ | -- | B2 | 3.3 | 2 |
| Hexachlorobenzene | $8 \times 10^{-4}$ | $4.6 \times 10^{-5}$ | B2 | 0.7 | NA |
| Selenium compounds | $5 \times 10^{-3}$ | -- | -- | $1.5 \times 10^{+2}$ | 2 |
| Beryllium compounds | $5 \times 10^{-3}$ | $1.2 \times 10^{-4}$ | B2 | 7.1 | 0.5 |
| Cyanide compounds | $5 \times 10^{-3}$ | -- | -- | $2.2 \times 10^{+2}$ | NA |
| Manganese compounds | $5 \times 10^{-3}$ | -- | -- | $1.6 \times 10^{+2}$ | 10 |
| Pentachlorophenol | NA | -- | B2 | $6.4 \times 10^{-2}$ | NA |
| Cobalt compounds | NA | -- | -- | $2.1 \times 10^{+1}$ | 20 |
| POM (PAH)[e] | NA | $2.1 \times 10^{-4}$ | B2 | 1.9 | < 1 |

HAP = Hazardous air pollutant
NA = Not available
OURE = Oral unit risk estimate
PAH = Polycyclic aromatic hydrocarbons
POM = Polycyclic organic matter
RfD = Reference dose
TEQ = Toxicity equivalent
TCDD = Tetrachlorodibenzo-p-dioxin
WOE = weight of evidence

[a] This is an estimate of total nationwide emissions from the source category.
[b] This is an unverified oral unit risk estimate.[4]
[c] 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.
[d] RfD is for inorganic arsenic. There was not a clear consensus for developing this value. See Appendix E and/or the IRIS database for details.
[e] POM emissions were estimated by summing the emissions estimates for each individual PAH listed in Appendix H of the **Interim** Final Utility HAP Report.[3]

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. This prioritization method for HAPs for multipathway analysis has limitations and uncertainties, was based on limited data, was not quantitative, and was based largely on subjective decisions; therefore, the selection of among the most important to assess for multipathway exposures and are considered a reasonable starting point for further multipathway analyses.

## 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., $1 \times 10^{-7}$); (2) maximum inhalation exposure concentration was 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., $1 \times 10^{-7}$ and 1/10 the RfC) are lower than levels that have been considered historically as levels for regulatory and policy decisions (e.g., $1 \times 10^{-6}$ for cancer and RfC for noncancer). These lower values were chosen for screening purposes so that it would be unlikely that potentially important HAPs would be missed by screen. That is, these conservative levels were chosen to ensure that all potentially important HAPs would be identified by 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 $1 \times 10^{-6}$ 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, emissions estimates, and recommendations from the peer review panel.[2,12,13] 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.



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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 multipathway 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[c] | No | 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 |

HQ = Hazard quotient
MEI = Maximally exposed individual
NA = Not applicable

[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]

[c] The MEI risk estimate for n-Nitrosodimethylamine is highly uncertain (and likely to be a high estimate) because emission estimates were based on one measured value and several measurements below the maximum detection limit. Therefore, n-Nitrosodimethylamine was not selected as priority for further 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 are not likely to present significant risks to public health. Based on available data and the screening analyses, the 53 HAPs that were not chosen for further study are not likely to be a concern for public health. However, due to uncertainties and limitations in the data, it is not possible to fully and conclusively determine that they do not pose a threat to public health. It is possible (although unlikely) that future data, such as revised emissions data or new toxicologic data, could possibly warrant further evaluation of some of these 53 HAPs in the future.

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# 6.0 INHALATION RISK ASSESSMENT

## 6.1 BASELINE ASSESSMENT OF INHALATION EXPOSURES AND RISKS FOR 13 PRIORITY POLLUTANTS

This chapter presents estimates of risks due to inhalation exposure to 13 of the 14 priority HAPs identified in the screening assessment (chapter 5). The assessment of risks presented in this chapter is more refined and complete than the screening level assessment presented in chapter 5. The risk estimates presented in this chapter are believed to be more accurate and more comprehensive than those presented in chapter 5. However, it is important for the reader to understand that the risk estimates presented in this chapter are still generally considered high-end estimates, and there are still substantial uncertainties and data gaps in the risk assessment presented in this chapter. Further assessment would be needed in several areas to gain a better understanding of the actual risks posed by electric utilities.

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 this section, for the 13 priority HAPs, risks have been calculated using the HEM for HAP emissions from all 684 utilities, and using 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 presented in section 6.1 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] The IUREs represent upper bound estimates of the cancer risks posed by these HAPs. The true risks are not known and could be as low as zero. 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). Dispersion of the pollutant plume in an urban area is expected to exhibit greater turbulence because of heat transfer and obstacles (i.e., large buildings). Therefore, using the urban default setting typically results in higher predicted air concentrations as compared to the concentrations predicted using the rural default setting. 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 and risks. 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 in later sections of chapter 6) suggests that the baseline risk estimates are generally conservative, but within the range of reasonable estimates. 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. Central tendency estimates of risks as well as background exposures and risks are discussed, and analyzed to a limited degree, in later sections of this chapter.

### 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 $1 \times 10^{-9}$ and were excluded from the presentation of results, thus reducing the total number of plants to 424. Table 6-1 summarizes the following: the predicted high-end MEI risks; high-end MIRs; the high-end estimate of the number of persons predicted to be exposed above individual cancer risk levels of 1 chance in 1 million (i.e., $1 \times 10^{-6}$); 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

Table 6-1. Summary of High-End Risk Estimates from Chronic Inhalation Exposure by HAP for 424 U.S. Coal-Fired Utilities Based on the Baseline Inhalation Risk Assessment

| Pollutant | Carcinogens | | | | | Noncarcinogen |
|---|---|---|---|---|---|---|
| | Highest MEI cancer risk[a] | Highest cancer MIR[b] | Population with risk >$10^{-6}$ | # Plants with MIR >$10^{-6}$ | Cancer incidence (cases/yr)[c] | Max. HQ |
| Arsenic | 3 x$10^{-6}$ | 2 x $10^{-6}$ | 850 | 2 | 0.05 | NA |
| Beryllium | 3 x$10^{-7}$ | 2 x $10^{-7}$ | 0 | 0 | 0.004 | NA |
| Cadmium | 2 x$10^{-7}$ | 1 x $10^{-7}$ | 0 | 0 | 0.0006 | NA |
| Chromium[d] | 2 x$10^{-6}$ | 1 x $10^{-6}$ | 110 | 1 | 0.02 | NA |
| Dioxin/furans | 5 x$10^{-8}$ | 3 x $10^{-8}$ | 0 | 0 | 0.001 | NA |
| Hydrogen chloride | NA | NA | NA | NA | NA | 0.1 |
| Lead | NA | NA | NA | NA | NA | 0.001 |
| Manganese | NA | NA | NA | NA | NA | 0.05 |
| Mercury | NA | NA | 0 | 0 | NA | -- |
| Nickel[e] | 4 x $10^{-7}$ | 2 x $10^{-7}$ | 0 | 0 | 0.003 | NA |
| Total | 4 x$10^{-6}$ | 3 x $10^{-6}$ | NA | 2 | 0.1 | 0.1 to 0.2 |

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 = Maximally 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] Of all 424 coal-fired plants modeled with the HEM, this is the estimated increased inhalation cancer risk for a theoretical person assumed to be exposed for 70 years to the highest modeled HAP ambient air concentration around any of the 424 plants.
[b] Of all 424 coal-fired plants modeled with the HEM, this is the highest MIR.
[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 6.6 for discussion of long-range transport.
[d] 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.
[e] The nickel emitted is a mixture of various nickel compounds such as soluble nickel. This analysis assumes that the mix of nickel compounds emitted is 50% as carcinogenic as nickel subsulfide.

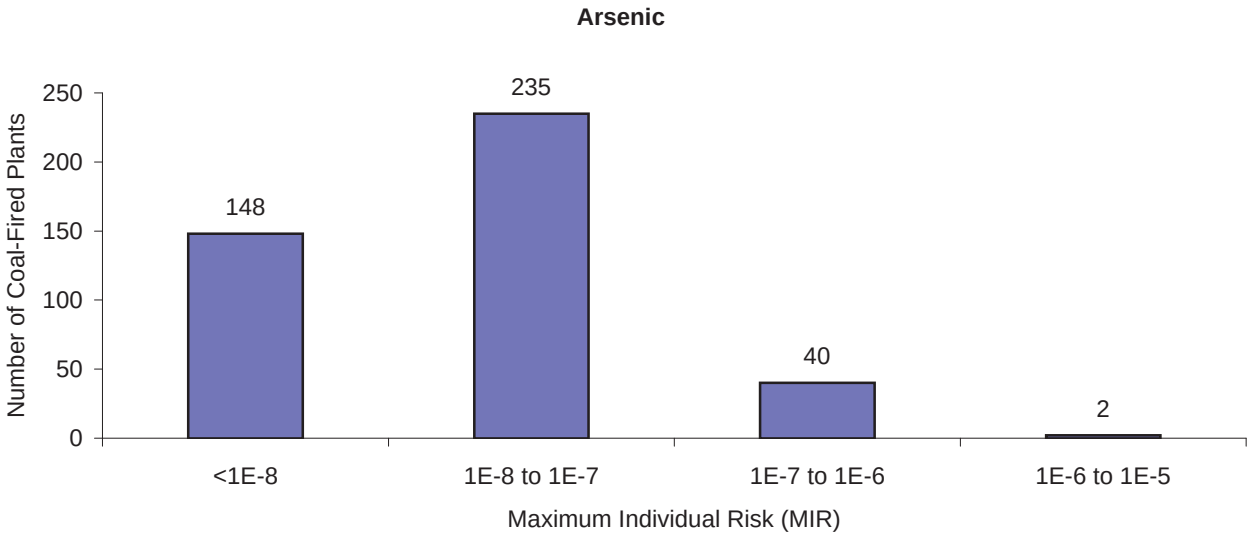
calculated based on the maximum modeled ambient concentration even though a person may or may not reside in the vicinity of the maximum concentration. The MEI risk was highest for arsenic (a Class A, human carcinogen) at 3 x $10^{-6}$ for the "highest-risk" coal-fired plant. The highest estimated MIR at a single plant was 2 x $10^{-6}$ for arsenic. Table 6-1 shows that arsenic emissions from two plants resulted in

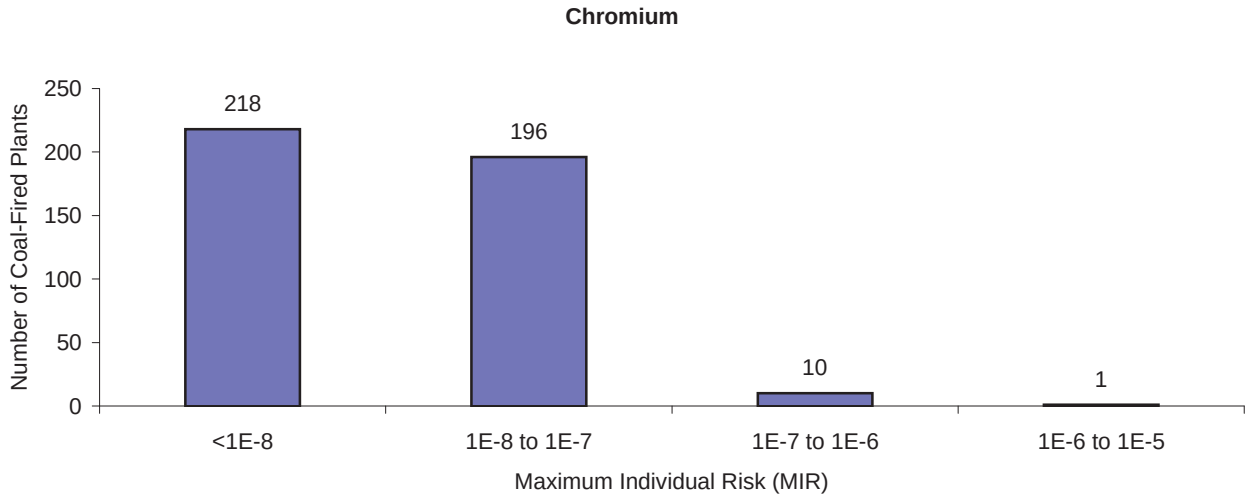
MIRs greater than or equal to $10^{-6}$. The MIRs for the remaining 424 coal-fired plants were lower than $1 \times 10^{-6}$. Figures 6-1a and 6-1b show that most inhalation risks were considerably lower than $1 \times 10^{-6}$. The risk for chromium assumes that 11 percent of total chromium is hexavalent chromium, (a Class A, human carcinogen). The limited emissions speciation data (described in Appendix H of the EPA Interim Final Utility Report, Volume III)[2] found hexavalent chromium between 0.3 and 34 percent of total chromium. The average percentage of 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 $2 \times 10^{-8}$ for arsenic and $2 \times 10^{-9}$ for chromium. The 90th percentile MIR of all 424 plants modeled is $1 \times 10^{-7}$ for arsenic and $4 \times 10^{-8}$ 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 $1 \times 10^{-6}$. The highest total MIR for a single coal-fired plant is $3 \times 10^{-6}$. Of the 424 coal-fired plants modeled, the median total MIR is $5 \times 10^{-8}$, and the 90th percentile is $2 \times 10^{-7}$ (that is, 10 percent of plants pose an MIR greater than $2 \times 10^{-7}$).

6.1.1.2 Population Cancer Risk. The population distribution at various risk levels is shown in Table 6-2 for each of the five major carcinogenic HAPs. As with the MIR, arsenic and chromium are the major contributors. The high-end estimate of number of people exposed to risks of $1 \times 10^{-6}$ or greater from exposure to arsenic is 850 and from exposure to chromium is about 107. That is, based on this assessment, it is unlikely that more than 850 people are exposed to inhalation risks greater than $1 \times 10^{-6}$ due to coal-fired utility nonradionuclide HAP emissions, and most likely fewer people are exposed (possibly as few as zero) to this level of inhalation risk.

The HEM also calculated the annual incidence of cancer expected for each of the HAPs due to inhalation exposure within 50 km. As shown in Table 6.1, the high-end estimate for total cancer incidence from the nonradionuclide carcinogenic HAPs was estimated, using the HEM, to be as high as 0.1 cancer case per year for coal-fired plants due to emissions within 50 km of each plant. However, the estimate for incidence increases by about a factor of seven when considering emissions dispersion beyond 50 km (see section 6.6). Arsenic and chromium are again the major contributors and account for almost 90 percent of the estimated cancer incidences.



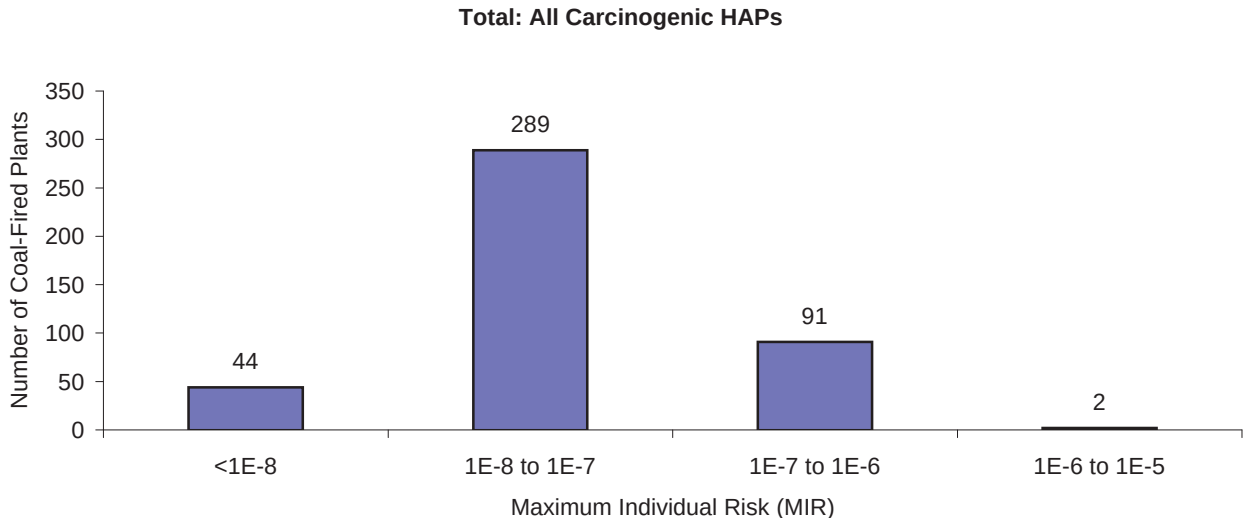




Figure 6-1a. Number of Coal-Fired Utilities Posing Various Levels of Maximum Individual Risk (by levels of MIR)

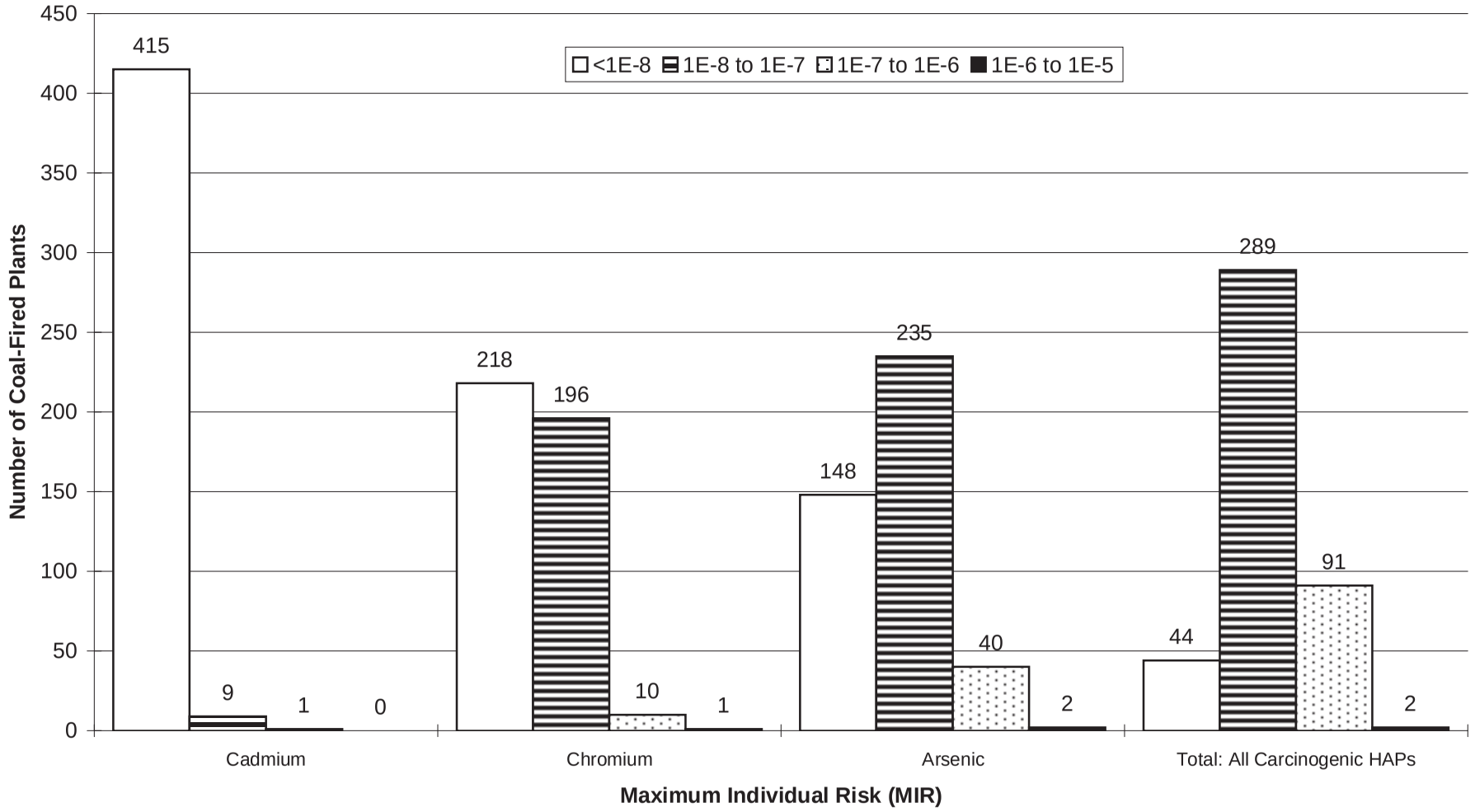



**Figure 6-1b. Number of Coal-Fired Utilities Posing Various Levels of Maximum Individual Risk (by levels of MIR)**

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

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

6.1.1.3 Noncancer Risk. The maximum HQ estimated for noncarcinogenic HAPs emitted from coal-fired power plants was 0.1 for HCl. The next highest was 0.05 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.

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 high-end individual and population risks for each of the HAPs evaluated. Eight plants had risks less than $1 \times 10^{-9}$ 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 $5 \times 10^{-5}$ 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, the mix of nickel compounds emitted by oil-fired utilities was assumed to be 50 percent as carcinogenic as 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 3 to 26 percent of nickel emissions (from oil-fired utilities) are sulfidic nickel.[2,3] It is not known how much of the sulfidic nickel is in form of nickel subsulfide. 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 the High-End Risk Estimates from Inhalation Exposure for Priority HAPs for 137 U.S. Oil-Fired Utilities Based on the Baseline Risk Assessment

| Pollutant | Carcinogens | | | | Noncarcinogen |
|---|---|---|---|---|---|
| | Highest Cancer MIR | Population with risk $>10^{-6}$ | # Plants with MIR $>10^{-6}$ | Cancer incidence[a] (cases/yr) | MAX HQ |
| Arsenic | $1 \times 10^{-5}$ | 2,400 | 2 | 0.04 | NA |
| Beryllium | $7 \times 10^{-7}$ | 0 | 0 | 0.002 | NA |
| Cadmium | $2 \times 10^{-6}$ | 45 | 1 | 0.005 | NA |
| Chromium[b] | $5 \times 10^{-6}$ | 2,300 | 1 | 0.02 | NA |
| Dioxin/furans | $1 \times 10^{-7}$ | 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[c] | $5 \times 10^{-5}$ | 110,000 | 11 | 0.2 | NA |
| Total | $6 \times 10^{-5}$ | NA | 11 | 0.3 | NA |

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.
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.
Total = Total MIR is the sum of the MIR for individual HAPs within a plant. The total HQ (=HI) is the sum of the HQs within a plant.

[a] 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 6.6 for discussion of long-range transport.

[b] 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.

[c] This analysis conservatively assumes that all nickel emitted from utilities is 50 percent as carcinogenic 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, nickel oxides, and sulfidic nickel. Emissions tests indicate that 3 to 26 percent of the nickel emissions are sulfidic nickel. Nickel subsulfide is one of the possible forms of sulfidic nickel. It is not known how much of the sulfidic nickel is in the form of nickel subsulfide. 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.

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 $1 \times 10^{-7}$ for nickel and $1 \times 10^{-8}$ for arsenic. The 90th percentile for MIR among plants is $5 \times 10^{-7}$ for nickel (that is, 90 percent of plants are estimated to pose risks less than $5 \times 10^{-7}$ due to nickel emissions) and $1 \times 10^{-7}$ for arsenic.

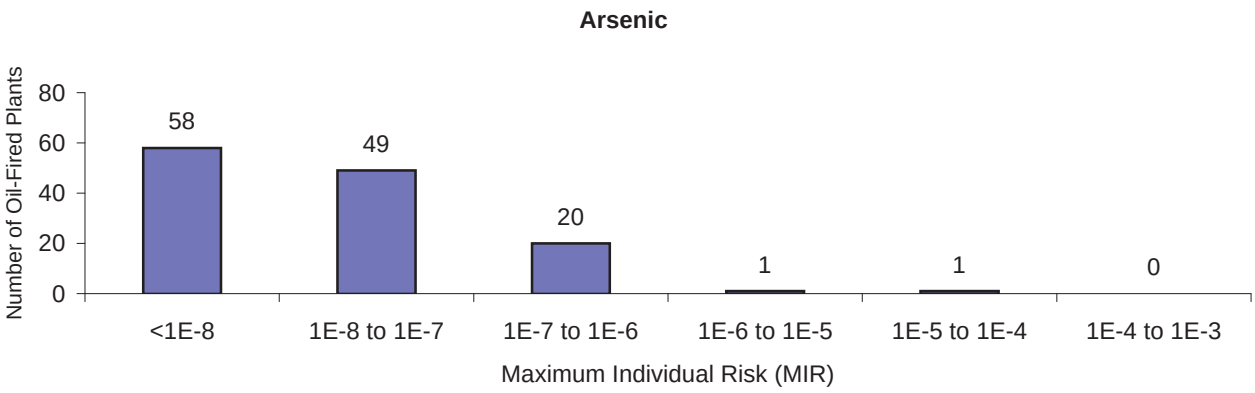
The total MIR was calculated for each facility by summing the MIRs for individual HAPs. The highest total high-end MIR from the sum of high-end risks for each carcinogen is 6 in 100,000 ($6 \times 10^{-5}$) at only one plant. The second and third highest-risk oil-fired plants pose MEI inhalation risks of $2 \times 10^{-5}$ and $1 \times 10^{-5}$, respectively. The total high-end MIR exceeded $1 \times 10^{-6}$ as a result of HAP emissions from 11 oil-fired plants. The median total MIR for all plants is approximately $4 \times 10^{-7}$, and the 90th percentile is approximately $2 \times 10^{-6}$. 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 50 percent as carcinogenic as nickel subsulfide (see section 6.10 for discussion).

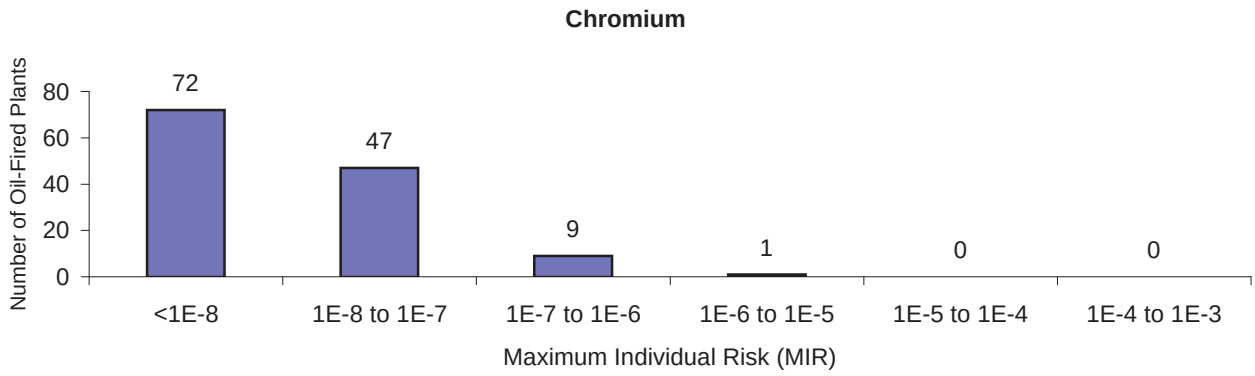
6.1.2.2 Population Cancer Risk. The population distribution at various risk levels is shown in Table 6-4 for each of six carcinogens. As with the MIR, nickel, arsenic, and chromium are the major contributors to the total population exposed to risk levels of 1 in 1,000,000 ($1 \times 10^{-6}$) or more. The high-end estimate for number of people exposed to risks of $1 \times 10^{-6}$ or greater is 110,000 for nickel, and about 2,400 for arsenic and chromium. That is, based on this assessment, it is unlikely that more than 110,000 people are exposed to inhalation risks greater than $1 \times 10^{-6}$ due to oil-fired utility nonradionuclide HAP emissions, and most likely fewer people are exposed to this level of inhalation risk.

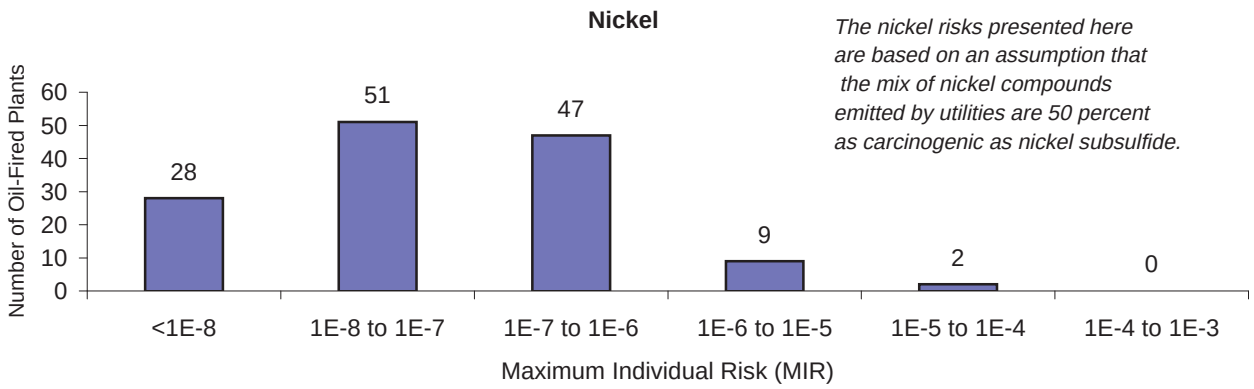
Incidences from each nonradionuclide HAP were summed to estimate total cancer incidence, which was estimated to be as high as 0.3 case per year from these 137 oil-fired plants. Nickel accounts for over 60 percent of the total annual incidence and arsenic contributes roughly about 15 percent.

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 and other uncertainties as discussed in later sections of this report.

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



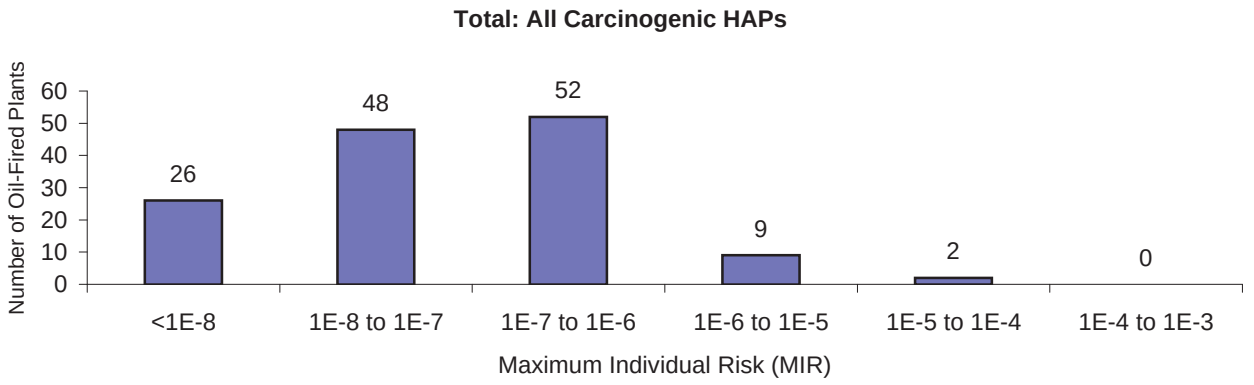




Figure 6-2a. Number of Oil-Fired Utilities Posing Various Levels of Maximum Individual Risk (by levels of MIR)

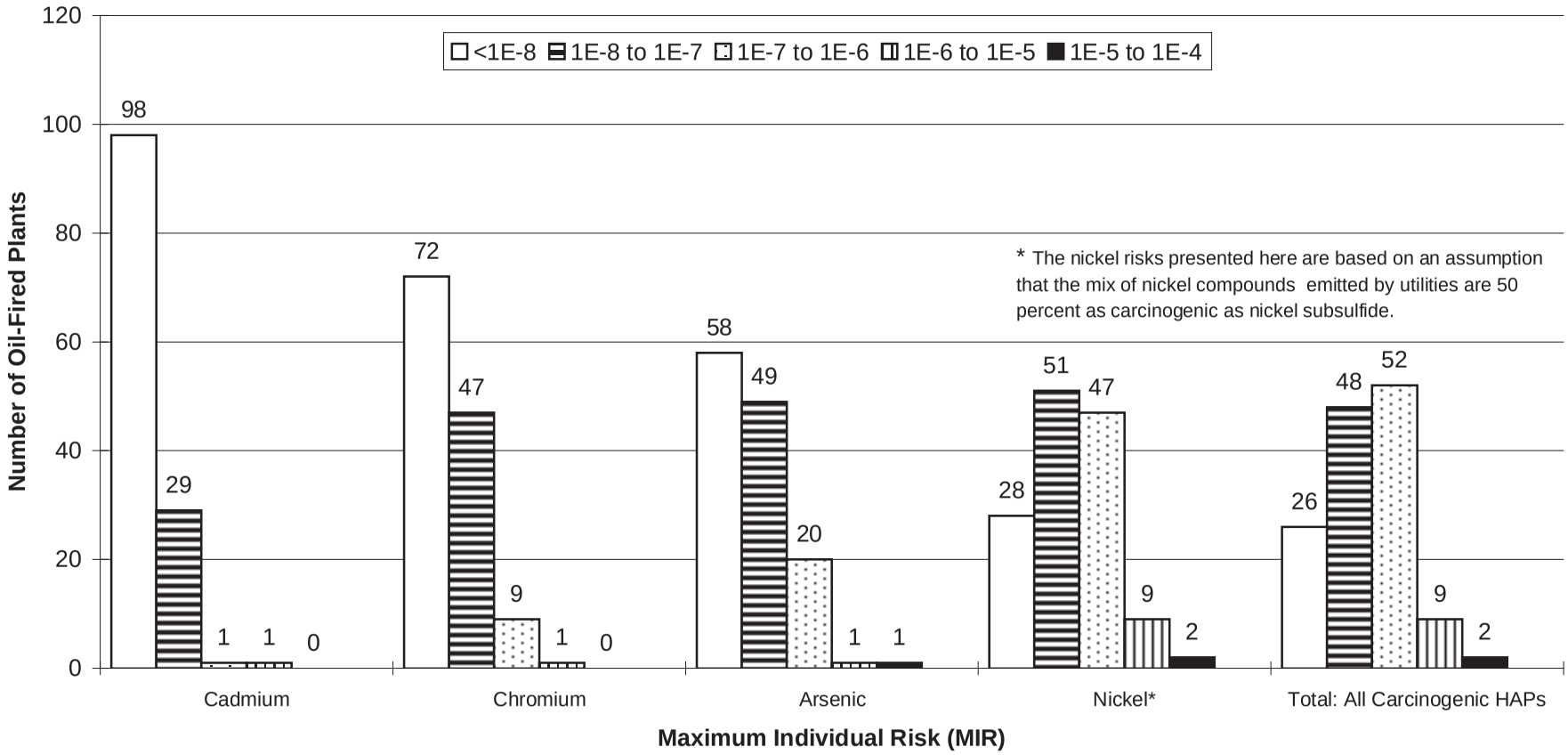



**Figure 6-2b. Number of Oil-Fired Utilities Posing Various Levels of Maximum Individual Risk (by levels of MIR)**

Table 6-4. Summary of High-End Estimates of Population Exposed Through Inhalation at Various Levels of Risk or Greater from Oil-Fired Utilities

| Risk level | Nickel | Arsenic | Chromium | Cadmium | Beryllium | Dioxins/ furans |
|---|---|---|---|---|---|---|
| $5 \times 10^{-5}$ | 45 | 0 | 0 | 0 | 0 | 0 |
| $2.5 \times 10^{-5}$ | 89 | 0 | 0 | 0 | 0 | 0 |
| $1 \times 10^{-5}$ | 2,200 | 45 | 0 | 0 | 0 | 0 |
| $5 \times 10^{-6}$ | 2,300 | 89 | 45 | 0 | 0 | 0 |
| $2.5 \times 10^{-6}$ | 9,900 | 2,280 | 89 | 0 | 0 | 0 |
| $1 \times 10^{-6}$ | 110,000 | 2,370 | 2,280 | 45 | 0 | 0 |
| $5 \times 10^{-7}$ | 1,600,000 | 32,600 | 2,280 | 89 | 45 | 0 |
| $2.5 \times 10^{-7}$ | 7,000,000 | 287,000 | 9,490 | 2,280 | 89 | 0 |

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.

### 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 high-end risks greater than 1 in 10 million ($10^{-7}$) with 23 persons exposed above that level. For noncarcinogens, the maximum HQ was $1 \times 10^{-7}$. The estimated risks due to HAP emissions from gas-fired utilities are low.

## 6.2 DISTINGUISHING BETWEEN URBAN AND RURAL LOCATIONS

The HEM has two distinct modeling options (urban or rural) intended to simulate atmospheric dispersion behavior of gases via different surface roughness.[4] 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(presented in chapter 5), 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

Table 6-5. Summary of High-End Inhalation Risk Estimates for Gas-Fired Utilities

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

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.
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

[a] The nickel emitted is a mixture of various nickel compounds. This analysis assumes that all nickel emitted is 50 percent as carcinogenic as nickel subsulfide.

Quality Models (40 CFR, Appendix W to Part 51),[4] 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.[4] 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 urban 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 are 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 all the inhalation exposure modeling analyses presented in chapter 6 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.

Table 6-6. Comparison of High-End 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 | | |
|---|---|---|---|---|---|---|
| **Pollutant and 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, (Coal) | $6 \times 10^{-6}$ | 0.08 | 21,000 | $3 \times 10^{-6}$ | 0.05 | 850 |
| Cr, Coal (assuming 11% Cr VI) | $3 \times 10^{-6}$ | 0.03 | 890 | $2 \times 10^{-6}$ | 0.02 | 110 |
| Be, Coal | $6 \times 10^{-7}$ | 0.006 | 0.0 | $3 \times 10^{-7}$ | 0.004 | 0.0 |
| Cr, Oil (assuming 18% Cr VI) | $5 \times 10^{-6}$ | 0.02 | 2,300 | $5 \times 10^{-6}$ | 0.02 | 2,300 |
| Be, Oil | $7 \times 10^{-7}$ | 0.002 | 0.0 | $7 \times 10^{-7}$ | 0.002 | 0 |
| Cd, Oil | $1.6 \times 10^{-6}$ | 0.007 | 45 | $1.6 \times 10^{-6}$ | 0.005 | 45 |
| Ni, Oil | $5 \times 10^{-5}$ | 0.3 | 155,000 | $5 \times 10^{-5}$ | 0.2 | 110,000 |
| As, Oil | $1 \times 10^{-5}$ | 0.05 | 4,600 | $1 \times 10^{-5}$ | 0.04 | 2,400 |

MEI = Maximally exposed individual

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

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

| **Pollutant and fuel** | **Urban default MEI HQ** | **Selection of appropriate setting (rural vs. urban) MEI HQ** |
|---|---|---|
| Hcl, from Coal | 2.3/20 = 0.12 | 2.3/20 = 0.12 |
| Mn, from Coal | 0.02/0.05 = 0.4 | 0.002/0.05 = 0.04 |

HQ = Hazard quotient
MEI = Maximally exposed individual.

## 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 Amendments of 1990. 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. Moreover, there are several other potential future actions or programs (e.g., PM and ozone NAAQS implementation, climate change programs, electricity restructuring), which could have an impact on HAP emissions, that were not considered in the projections made for this 2010 analysis because of the uncertainties and unknowns about how these programs will affect HAP emissions. However, even with these limitations, the method used by EPA is considered reasonable given the available data.

The exposures and risks for the year 2010 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 inhalation risks from utility emissions. The analysis of this subset of priority HAPs provides information regarding the anticipated potential public health risks due to inhalation for the year 2010.

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 POTENTIAL RISKS DUE TO SHORT-TERM EXPOSURE

The potential for exceeding short-term reference exposure levels (RELs)[5] 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.[5] The CAPCOA RELs are listed in Table 6-10.

Table 6-8. Estimated High-End 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[a]** | **Cancer incidence (cases/year)** | **Population w/ MIR >$10^{-6}$** | **MEI risk[a]** | **Cancer incidence[b] (cases/year)** | **Population w/ MIR >$10^{-6}$** |
| As from Coal | $3 \times 10^{-6}$ | 0.051 | 590 | $3 \times 10^{-6}$ | 0.045 | 852 |
| Be from Coal | $3 \times 10^{-7}$ | 0.004 | 0.0 | $3 \times 10^{-7}$ | 0.0035 | 0.0 |
| Cd from Coal | $3 \times 10^{-8}$ | 0.0007 | 0.0 | $2 \times 10^{-7}$ | 0.0006 | 0.0 |
| Cr from Coal (11% Cr VI) | $1.4 \times 10^{-6}$ | 0.021 | 399 | $2 \times 10^{-6}$ | 0.02 | 107 |
| Dioxins from Coal | $6 \times 10^{-8}$ | 0.0012 | 0.0 | $5 \times 10^{-8}$ | 0.001 | 0.0 |
| Ni from Coal | $2 \times 10^{-7}$ | 0.003 | 0.0 | $4 \times 10^{-7}$ | 0.003 | 0.0 |
| Cr from Oil (18% Cr VI) | $3 \times 10^{-6}$ | 0.009 | 89 | $5 \times 10^{-6}$ | 0.02 | 2,300 |
| Be from Oil | $4 \times 10^{-7}$ | 0.0008 | 0.0 | $7 \times 10^{-7}$ | 0.0017 | 0.0 |
| Cd from Oil | $8 \times 10^{-7}$ | 0.0026 | 0.0 | $2 \times 10^{-6}$ | 0.0053 | 45 |
| Ni from Oil | $3 \times 10^{-5}$ | 0.1 | 11,000 | $5 \times 10^{-5}$ | 0.2 | 110,000 |
| Dioxins from Oil | $7 \times 10^{-8}$ | 0.0004 | 0.0 | $1 \times 10^{-7}$ | 0.0007 | 0.0 |
| As from Oil | $7 \times 10^{-6}$ | 0.026 | 2,300 | $1 \times 10^{-5}$ | 0.042 | 2,400 |

Note: The EPA used urban vs. rural modeling data distinction in this analysis.

MEI = Maximally exposed individual.
MIR = Maximum individual risk is the highest risk identified at the centroid of a census tract to which a population is assigned.

[a] These MEI risk estimates are for the “highest risk” plant.
[b] This is the estimated cases of cancer predicted to occur in the United States due to emissions of this HAP from all utilities of that fuel type based on the HEM analysis.

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

| **Pollutant** | **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 |

HQ = Hazard quotient.
MEI = Maximally exposed individual.
RfC = Reference concentration.

Table 6-10. Noncancer Reference Exposure Levels (Acute) from CAPCOA[5]

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

CAPCOA = California Air Pollution Control Officers Association.
REL = Reference exposure level.

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 (called TSCREEN) that considers the potential range in meteorological conditions at the utility plant site to estimate the maximum 1 hour concentration of the three compounds in the vicinity of selected coal- and oil-fired utilities. 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 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.

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 |

To illustrate this methodology, a sample utility is presented in Table 6-11. 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 HEM analysis were located in relatively flat terrain. (The effects of terrain are analyzed in Appendix G of the EPA Interim Final Utility Report, Volume II.)[6] 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

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.

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 United States 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.[7] This regional model simulated monthly $SO_2$ and sulfate ($SO_4^{2-}$) concentrations and 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.[7,8] 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 $\mu$m) and coarse (2.5 $\mu$m < diameters < 10.0 $\mu$m) 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.*[9] The next section discusses modifications made to the original sulfur version of RELMAP to enable the simulation of atmospheric particulate metals (arsenic, cadmium, chromium, lead and nickel).

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 | $3 \times 10^{-6}$ | 0.05 | 850 | $3 \times 10^{-6}$ | 0.05 | 850 |
| As, Oil, 1990 | $1 \times 10^{-5}$ | 0.04 | 2,200 | $1 \times 10^{-5}$ | 0.04 | 2,400 |
| As, Coal, 2010 | $3 \times 10^{-6}$ | 0.05 | 590 | $3 \times 10^{-6}$ | 0.05 | 590 |
| As, Oil, 2010 | $7 \times 10^{-6}$ | 0.03 | 2,200 | $7 \times 10^{-6}$ | 0.03 | 2,300 |

HAP = Hazardous air pollutants.
MEI = Maximally exposed individual.

### 6.6.2 RELMAP Modeling Approach for Particulate Metals

6.6.2.1 Introduction. Previous versions of RELMAP have been described by Eder et al.[9] and Clark et al.[10] The goal of the current effort was to model the emission, transport, and fate of airborne cadmium, arsenic, chromium, nickel, and lead from utilities in the continental United States for the year 1989. Modifications to the RELMAP simulation for arsenic, cadmium, chromium, lead, and nickel were based on the assumption that these 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 the five metals, the data accounting task of a source-receptor run for all 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.[9] were preserved for this RELMAP modeling study.

6.6.2.2 Physical Model Structure. Because of the long atmospheric residence time of fine PM, significant long-range transport 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 United States.

Since the descriptive document by Eder et al.[9] was produced, the original three-layer puff structure of RELMAP was replaced by a four-layer structure. The following model layer definitions were used for the RELMAP particulate metal 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 particulate metals 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 particulate metals, as for most other pollutants, emission rates for each source cell were defined from input data and a time step of three 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 particulate metal simulations. 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 are 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.

6.6.2.5 Vertical Exchange of Mass with the Free Atmosphere. To accurately simulate the long atmospheric lifetime of some pollutants, the RELMAP was adapted to allow a treatment of the exchange of mass between the surface-based mixed layer and the free atmosphere above. The RELMAP no longer requires that the pollutant mass remain entirely within the mixed layer. As an intuitive approximation, a pollutant depletion rate of 5 percent per three-hour time step was chosen to represent this diffusive mass exchange. This rate of mass exchange used in the RELMAP was chosen to approximate the average daily exchange of mass obtained from similar Lagrangian model exercises in Europe. When compounded over a 24-hour period, the mass exchange rate of 5 percent every three hours removes 33.6 percent of an inert, non-depositing pollutant. Since each of the modeled particulate metals undergo significant wet and dry deposition, their effective diffusion rate out of the top of the model is less than 33.6 percent per day. The mass lost through this vertical exchange through the top of the model is accounted for and is reported as a model output for mass balance checks.

## 6.6.3 Model Parameterizations

6.6.3.1 Chemical Transformation. The simplest pollutant type to model with RELMAP is the inert type. To model inert pollutants, one

can simply omit chemical transformation calculations and not be concerned with chemical interactions with other chemical species. Arsenic, cadmium, chromium, lead and nickel were treated as inert pollutants.

6.6.3.2 Dry Deposition. All five metals were assumed to be totally in particulate form. Since each of these metals and their compounds make up only a small fraction of the total particulate mass loading of the atmosphere, we modeled each as a minor component of the general population of conglomerate aerosol particles. Lead has been generally associated with fine particle sizes (<1/$\mu$m diameter), but there is evidence that larger particles may play a significant role in its dry deposition in urban areas.[11,12] Therefore, metal particles were modeled in five sizes; 0.1, 0.3, 1.0, 3.0, and 10.0 $\mu$m 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 semi-empirical dry deposition model as described in Sehmel[11] was used to estimate dry deposition velocity for conglomerate particles in the five size classes. 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 that sulfate, nitrate, and organic compounds make up most of the particulate mass for particles less than 10 $\mu$m in diameter, a density value of 2 g/cm$^3$ was used to represent all particulate metals. Although the metals and their 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 metals were calculated using a FORTRAN subroutine developed by the California Air Resource Board (CARB).[12] Table 6-15 shows the windspeed (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.

Some of the parameters used for dispersion and deposition modeling are shown in Tables 6-15 and 6-16. These parameters include windspeed versus stability category and roughness length versus land-use category.

6.6.3.3 Wet Deposition. A review of literature showed a range of measured values for the scavenging ratios for the metal particulates.[13,14] Considering the source of the metal particulates (i.e., utilities) and a review of the literature, a scavenging ratio of 0.2 x 10$^6$ was used for nickel, chromium and lead; 0.3 x 10$^6$ for arsenic; and 0.4 x 10$^6$ for cadmium.

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

| Stability category | Windspeed (m/s) |
|---|---|
| A | 2.0 |
| B | 3.0 |
| C | 4.0 |
| D | 5.0 |
| E | 3.0 |
| F | 2.0 |

CARB = California Air Resource Board.

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

| Land-use category | Roughness length (ms) | |
|---|---|---|
| | Spring-summer | Autumn-winter |
| 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 |

CARB = California Air Resource Board.

### 6.6.4 Exposure and Risk Estimates

6.6.4.1 Air HAP Concentration Estimates. Table 6-17 presents the average and maximum annual air particulate metal concentration for the continental United States as predicted by the RELMAP analysis for four metal particulates (i.e., arsenic, cadmium, chromium, and nickel). The results of the RELMAP modeling for lead are presented in chapter 8. Results are presented as both combined impacts (e.g., emissions from both coal and oil utility combustion combined) and segregated impacts(e.g., emissions from coal and oil utilities separately).

Figures 6-3 through 6-14 graphically present the RELMAP air concentration results for each of the metals and each combustion fuel type. In general, air concentrations of the four metal particulates, as a result of utility oil and coal combustion, are predicted by RELMAP to be maximum in the eastern part of the United States.

Table 6-17. RELMAP Predicted Air Concentrations

| Pollutant | Fuel | Max concentration ($\mu g/m^3$) | Average concentration ($\mu g/m^3$) |
|---|---|---|---|
| Arsenic | Coal | 2.5 E-04 | 8.0 E-06 |
| | Oil | 4.4 E-05 | 5.0 E-07 |
| | Coal & Oil | 2.5 E-04 | 8.5 E-06 |
| Cadmium | Coal | 8.0 E-06 | 3.0 E-07 |
| | Oil | 1.0 E-05 | 1.6 E-07 |
| | Coal & Oil | 1.1 E-05 | 4.6 E-07 |
| Chromium | Coal | 2.2 E-04 | 1.2 E-05 |
| | Oil | 3.0 E-05 | 4.7 E-07 |
| | Coal & Oil | 2.2 E-04 | 1.2 E-05 |
| Nickel | Coal | 1.7 E-04 | 7.8 E-06 |
| | Oil | 2.6 E-03 | 3.9 E-05 |
| | Coal & Oil | 2.6 E-03 | 4.7 E-05 |

RELMAP = Regional Lagrangian Model of Air Pollution.

Air concentrations as predicted with RELMAP for emissions from coal-fired utilities are predicted to be maximum along the western slopes of the Appalachians Mountains and the northern Ohio River Valley. Air concentrations as a result of utility oil combustion are predicted to be maximum along the coastal northeast and the Florida Peninsula.

6.6.4.2 Estimating Exposures and Risks. Once the grid cell concentrations are known (as predicted by RELMAP), public exposure and risks can be calculated. The population database within the HEM model contains the centroid coordinates and number of people living within each census block group. By applying this database to the predicted RELMAP air concentrations both average and total population exposure and risk can be estimated by the following six-step approach:

1. For each census block group determine which RELMAP grid cell the census block group centroid is located within. All the people living within the census block group are assumed to be exposed to the predicted RELMAP HAP concentration for that corresponding grid cell.

$$C_{(n,pol)} = C_{(i,j,pol)}$$

where:

$C_{(n,pol)}$ = the concentration of pollutant "pol" at census block group "n" in $\mu g/m^3$.

$C_{(i,j,pol)}$ = the RELMAP concentration of pollutant "pol" at grid cell (i,j) in $\mu g/m^3$ where the centroid of census block group "n" is locate within grid cell (i,j).

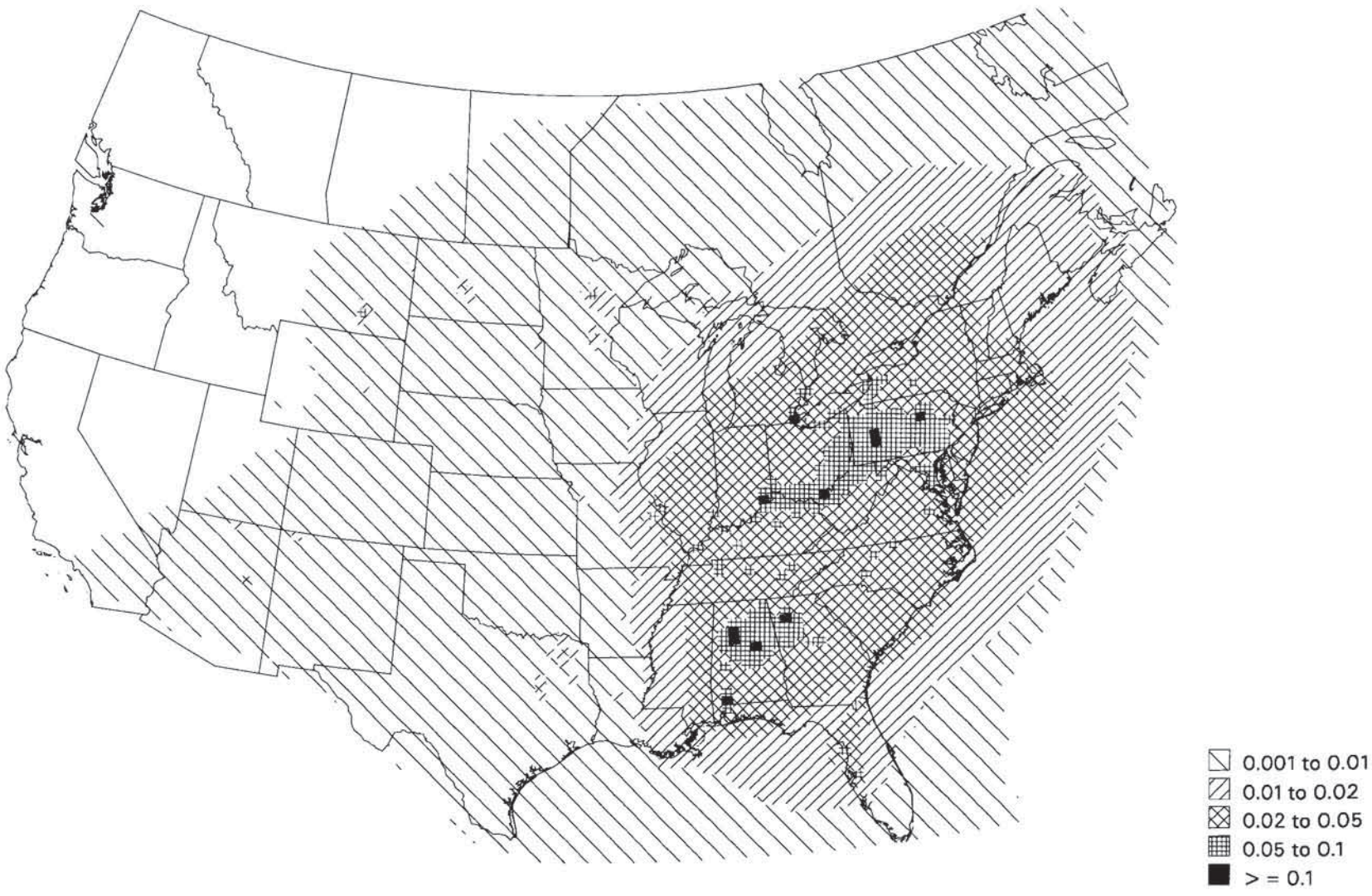



Figure 6-3. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Arsenic from Coal Utilities: Predicted Air Concentration of Arsenic, Units: $ng/m^3$



Exhibit 2024
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Figure 6-4. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Arsenic from Oil Utilities: Predicted Air Concentration of Arsenic, Units: $ng/m^3$



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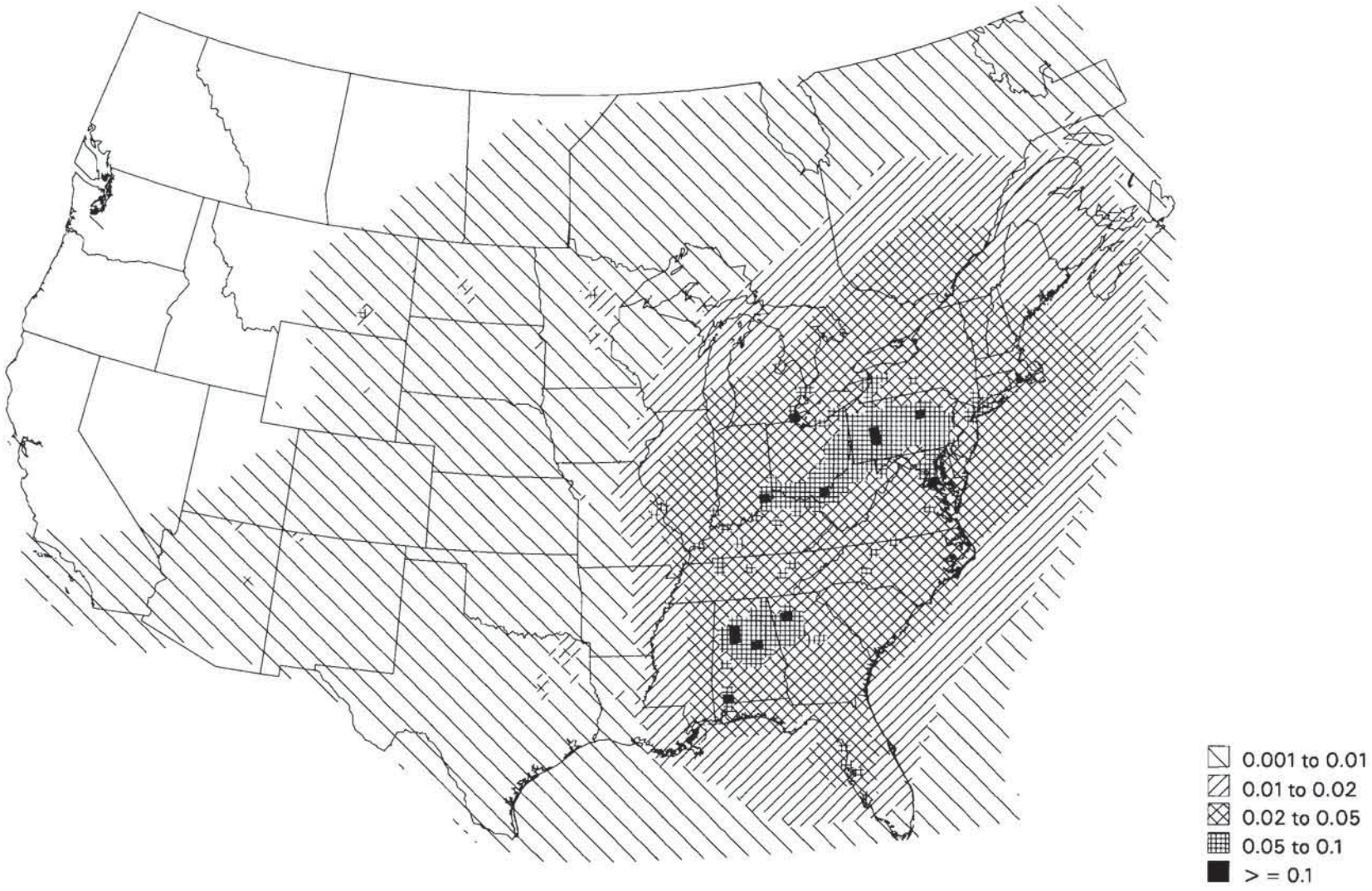



Figure 6-5. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Arsenic from Coil and Oil Utilities: Predicted Air Concentration of Arsenic, Units: $ng/m^3$



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Figure 6-6. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Cadmium from Coal Utilities: Predicted Air Concentration of Cadmium, Units: ng/m$^3$



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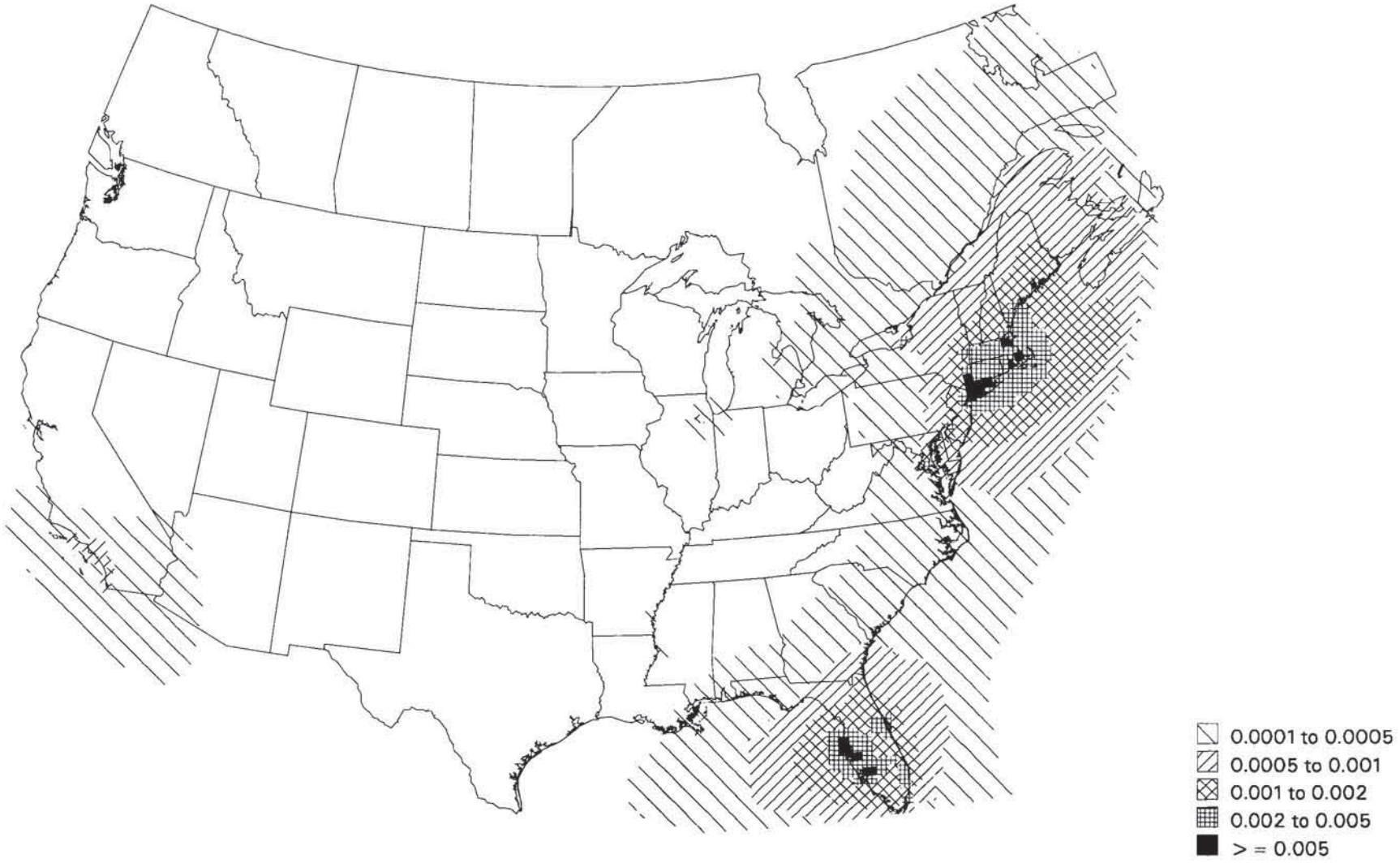



Figure 6-7. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Cadmium from Oil Utilities: Predicted Air Concentration of Cadmium, Units: $ng/m^3$



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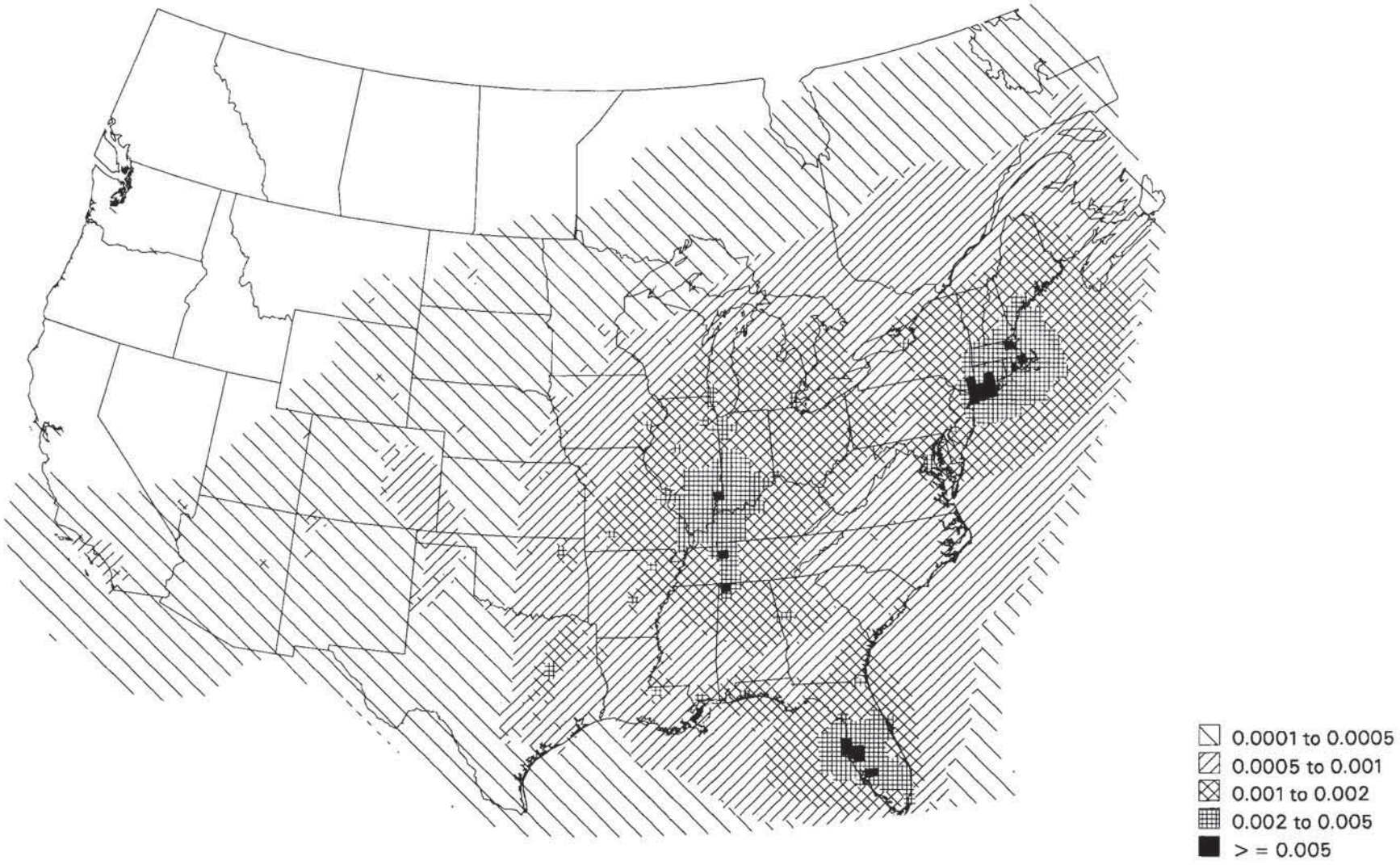



Figure 6-8. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Cadmium from Coal and Oil Utilities: Predicted Air Concentration of Cadmium, Units: ng/m$^3$



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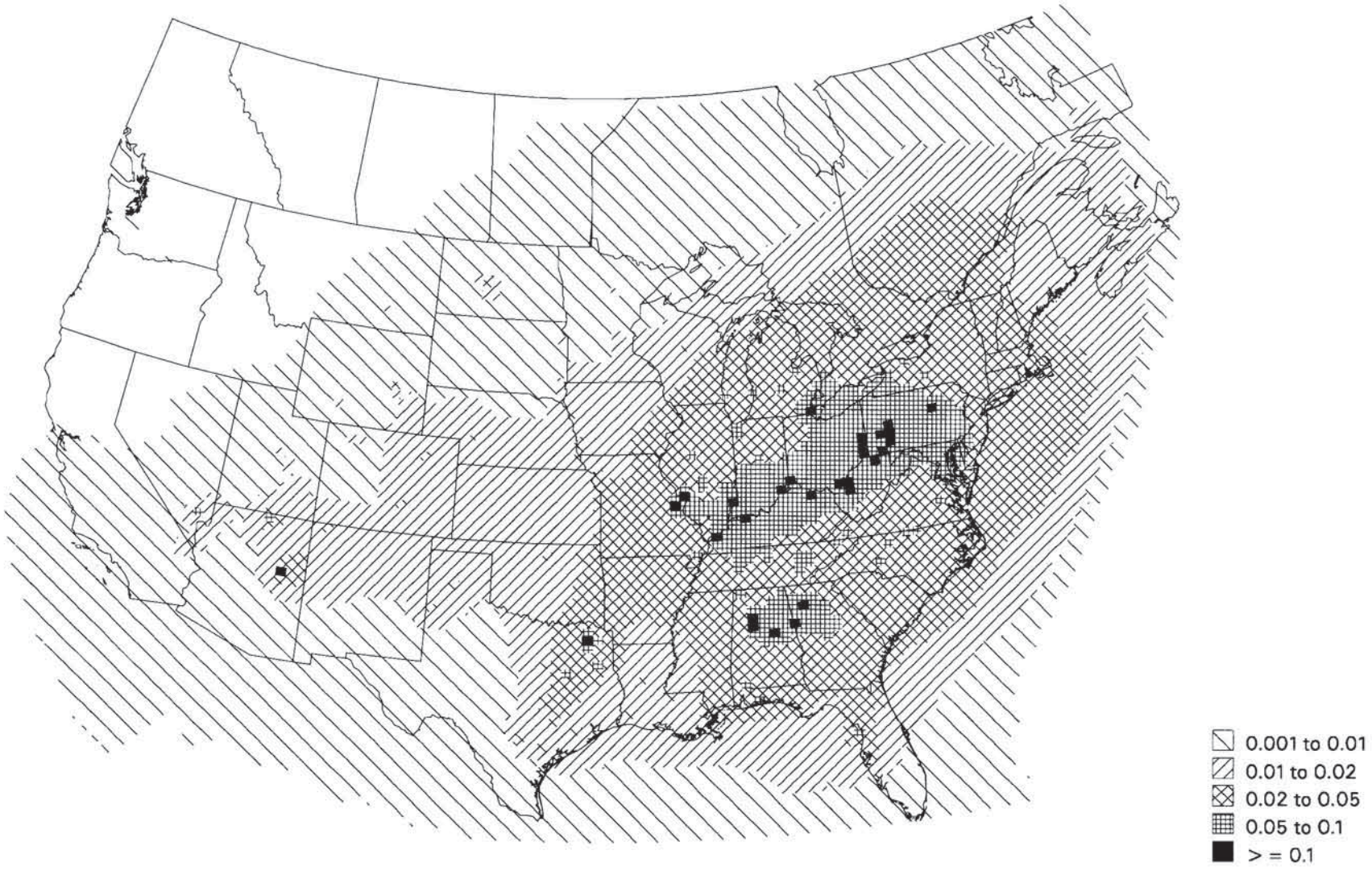



Figure 6-9. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Chromium from Coal Utilities: Predicted Air Concentration of Chromium, Units: $ng/m^3$



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Figure 6-10. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Chromium from Oil Utilities: Predicted Air Concentration of Chromium, Units: $ng/m^3$



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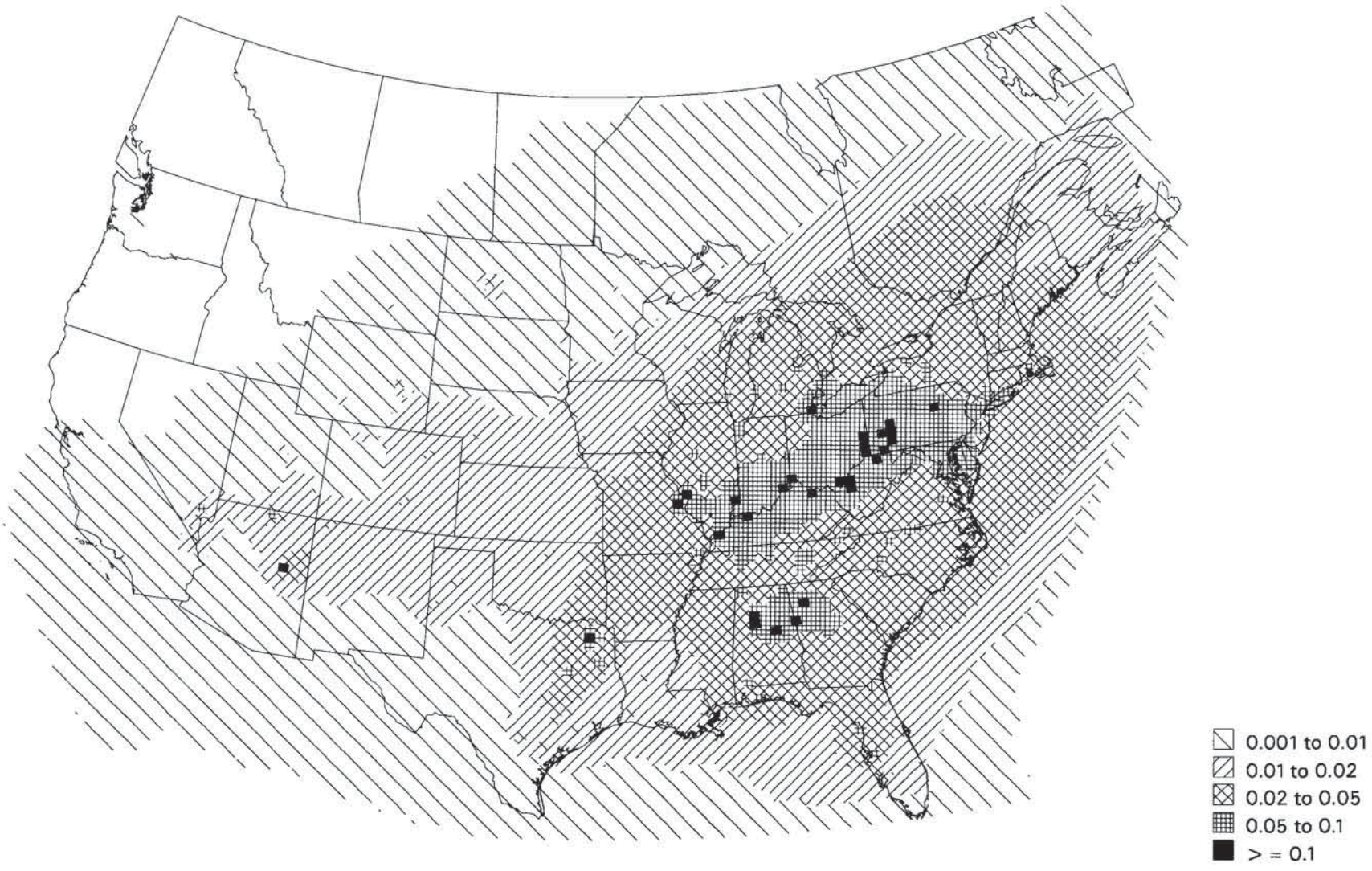



Figure 6-11. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Chromium from Coal and Oil Utilities: Predicted Air Concentration of Chromium, Units: $ng/m^3$



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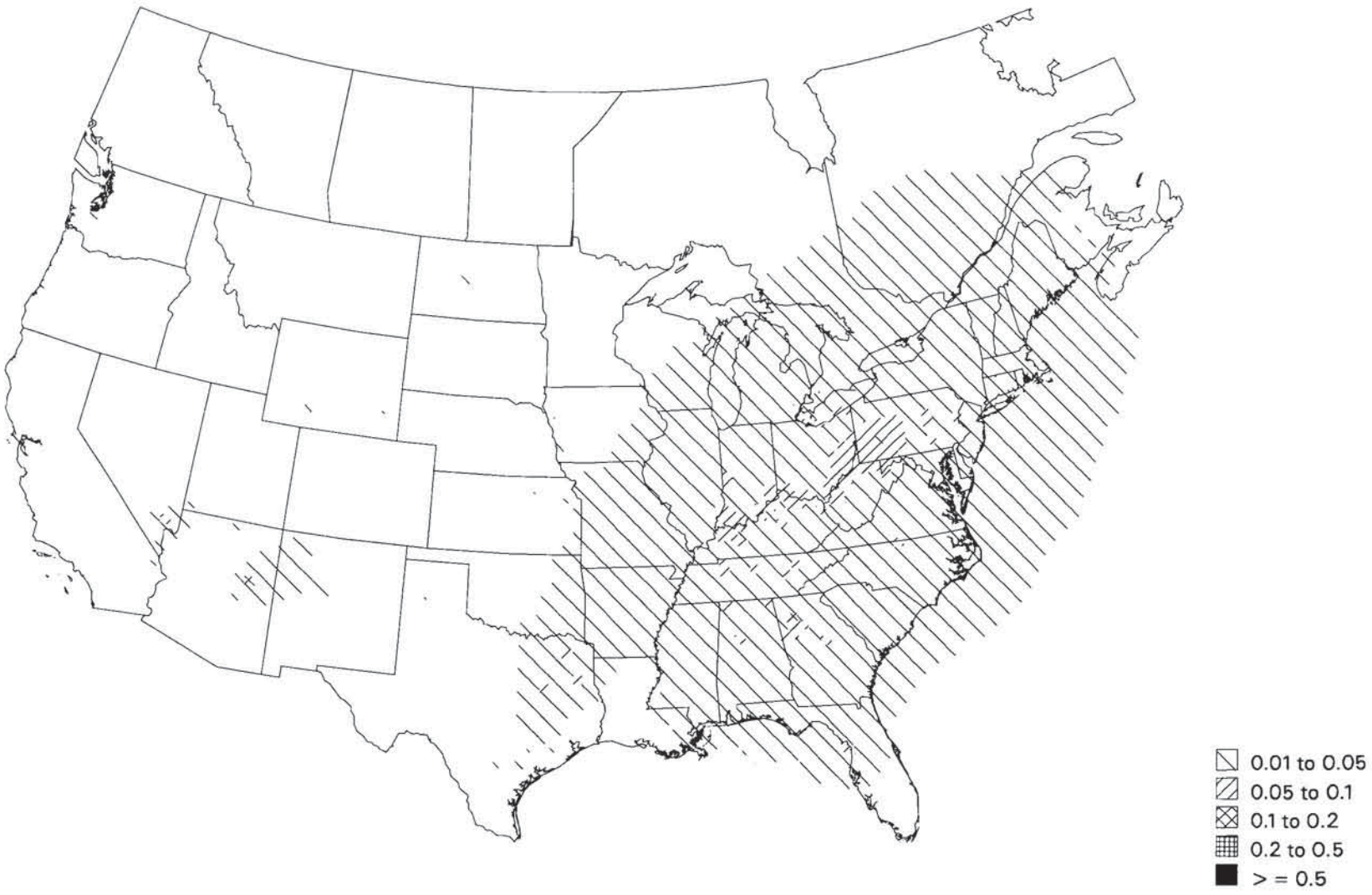



Figure 6-12. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Nickel from Coal Utilities: Predicted Air Concentration of Nickel, Units: $ng/m^3$



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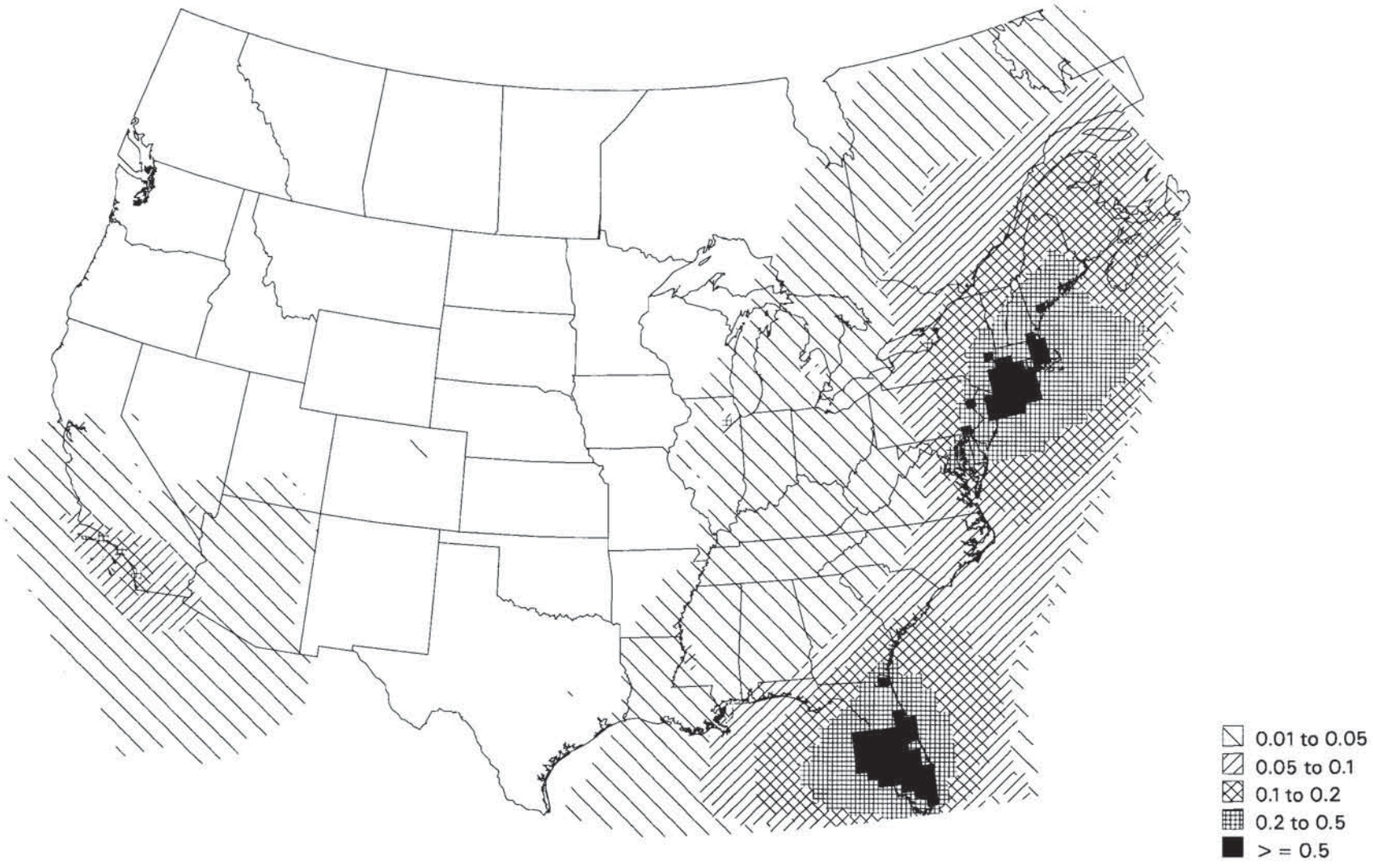



Figure 6-13. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Nickel from Oil Utilities: Predicted Air Concentration of Nickel, Units: $ng/m^3$



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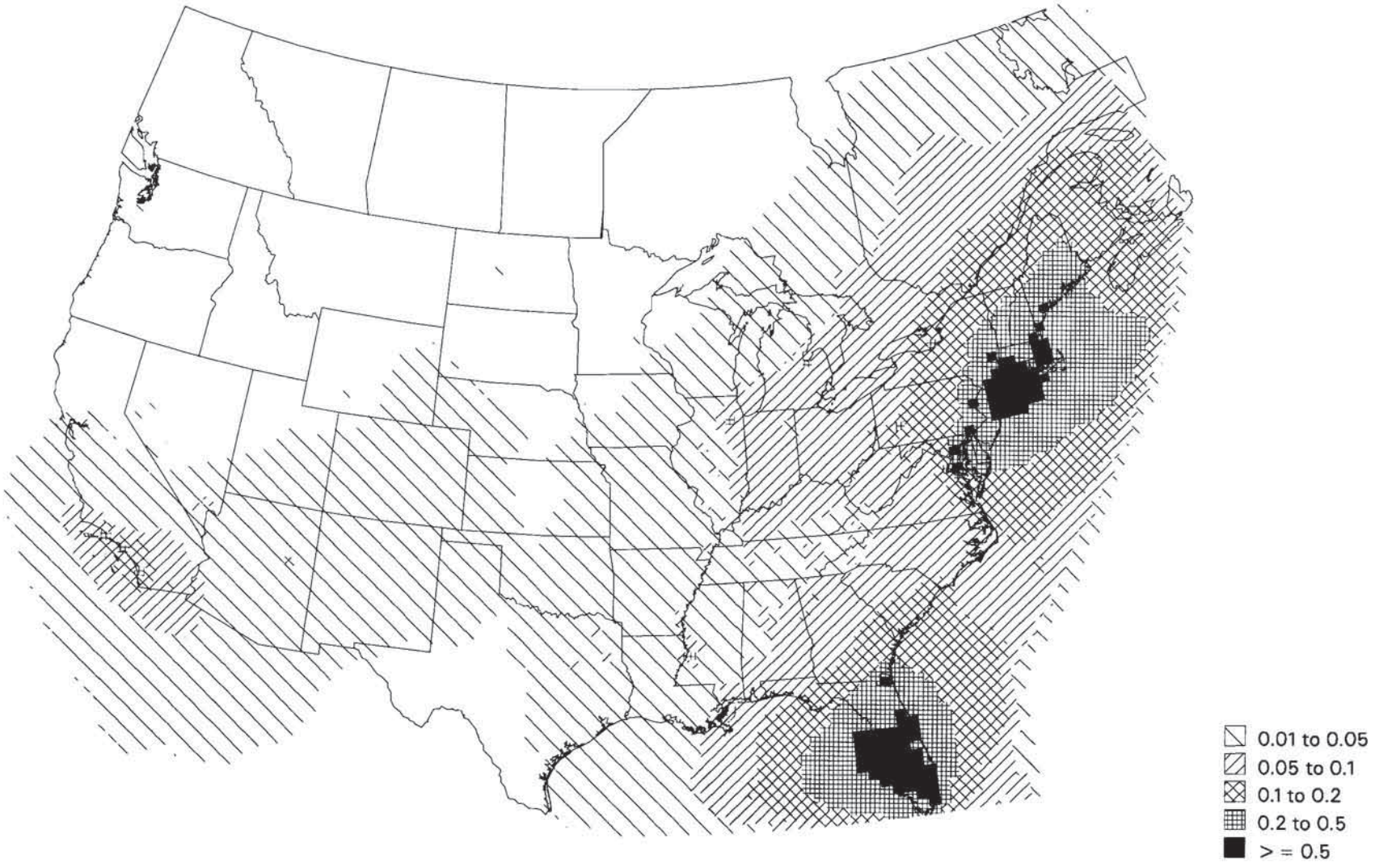



Figure 6-14. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Nickel from Coal and Oil Utilities: Predicted Air Concentration of Nickel, Units: $ng/m^3$



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2. Determine the average individual lifetime cancer risk for people living in the census block group by multiplying the HAP concentration for the census block group (determined in step 1) by the IURE (which is the estimated increase risk of cancer for an individual exposed to the pollutant concentration of 1 $\mu g/m^3$ for 70 years). The IURE is explained in detail in chapter 4.

$$\mathbf{R_{(n,pol)} = C_{(n,pol)} * IURE_{(pol)}}$$

where:

$R_{(n,pol)}$ = the average individual lifetime cancer risk for pollutant "pol" at census block group "n" in $\mu g/m^3$ .

$IURE_{(pol)}$ = the estimated increase risk of cancer for an individual exposed to a concentration of 1 $\mu g/m^3$ of pollutant "pol" for 70 years (1 per $\mu g/m^3$ ).

3. Determine the lifetime population cancer incidence for each census block group by multiplying the average individual risk (from step 2) by the corresponding population of the census block group.

$$\mathbf{LCI_{(n,pol)} = R_{(n,pol)} * Pop_{(n)}}$$

where:

$LCI_{(n,pol)}$ = the Lifetime (70 years) Population Cancer Incidence for census block group "n" for pollutant "pol" in cases/lifetime.

$Pop_{(n)}$ = the 1990 Census total population for census block group "n".

4. The lifetime cancer incidence can be estimated by summing the lifetime population cancer incidence for each census block group (from step 3).

$$\mathbf{LCI_{(total,pol)} = \sum LCI_{(n,pol)}} \quad n=1,466,318$$

where:

$LCI_{(total,pol)}$ = the Lifetime (70 years) Population Cancer Incidence for the continental U.S. (466,318 census block groups in the continental United States) for pollutant "pol" in cases/lifetime.

5. The annual population cancer incidence in the continental U.S. is predicted by dividing the lifetime cancer incidence (from step 4) by 70 years.

$$\mathbf{ACI_{(total,pol)} = LCI_{(total,pol)} / 70\ years}$$

where:

$ACI_{(total,pol)}$ = the Annual Population Cancer Incidence for the continental United States.

6. The average individual exposure over the entire continental U.S. population for a pollutant is estimated by dividing the total population exposure by the total number of people living in the continental United States.

$$\mathbf{AIE_{(total,pol)} = PE_{(total,pol)} / Pop_{(total)}}$$

where:

$AIE_{(total,pol)}$ = the average individual annual exposure for the continental United States for pollutant "pol"

$PE_{(total,pol)}$ = the sum of the concentration of pollutant "pol" for all census block groups ($\sum C_{(n,pol)\ n=1,466,318}$) in $\mu g/m^3$.

$Pop_{(total)}$ = the 1990 Census total population for the continental United States. (≈247,000,000 people)

The results of the exposure and risk estimates are shown in Table 6-18.

To evaluate potential impacts due to long-range transport (LRT), the coal and oil utility emissions were modeled both together and separately. By applying the algorithm described above, a combined (e.g.,oil and coal emissions) value of 0.42 cancer cases/year was estimated for arsenic emissions over the continental United States. The contribution of oil emissions is predicted to be about 11 percent of the total inhalation risks from arsenic emissions.

The LRT population exposure and cancer incidence for the four metals from RELMAP for coal-fired utilities is about seven times greater (i.e., 600 percent greater) than the population risks estimated in the local analysis alone (i.e., within 50 km) using the HEM. However, the population exposure and cancer incidence for oil-fired utilities from the RELMAP study are only slightly (about 10 percent) greater than the exposures and incidence predicted with the HEM analysis (See Table 6-19).

The total (coal and oil)population risk estimate for chromium emissions is predicted to be 0.22 cases/year. The contribution of oil fired chromium emissions is predicted to be about 10 percent of the total risk from chromium emissions. The LRT population risk estimate for chromium from RELMAP is about five times greater than the population risks estimated modeling chromium emissions using the HEM model.

The high-end estimate for total population cancer incidence due to nickel emissions is predicted to be up to 0.2 cases/year. The contribution of coal-fired nickel emissions is predicted to be about 10 percent of the total risk from nickel emissions. The LRT population risk estimate for nickel is about equal to the population risks

Table 6-18. Predicted Exposure and High-End Risk Estimates Based on RELMAP Modeling of Particulate Metal Emissions from All Oil- and Coal-Fired Utilities in the United States.

| Pollutant | Fuel | Maximum Concentration ($\mu$g/m3) | Average Exposure Concentration $\mu g/m^3$) | Unit Risk Estimate (per $\mu g/m^3$) | Maximum Exposed Individual (MEI) Risks | Population Risks (cases/yr) |
|---|---|---|---|---|---|---|
| Arsenic | Coal | 2.5e-04 | 2.4e-05 | 4.3e-03 | 1.1e-06 | 0.4 |
| | Oil | 4.4e-05 | 3.1e-06 | 4.3e-03 | 1.9e-07 | 0.05 |
| | Coal & Oil | 2.6e-04 | 2.8e-05 | 4.3e-03 | 1.1e-06 | 0.4 |
| Cadmium | Coal | 8.0e-06 | 7.6e-07 | 1.8e-03 | 1.4e-08 | 0.005 |
| | Oil | 1.1e-05 | 9.4e-07 | 1.8e-03 | 1.9e-08 | 0.006 |
| | Coal & Oil | 1.1e-05 | 1.7e-06 | 1.8e-03 | 1.9e-08 | 0.01 |
| Chromium | Coal | 2.2e-04 | 3.3e-05 | 1.3e-03 | 2.9e-07 | 0.2 |
| | Oil | 3.0e-05 | 2.7e-06 | 2.2e-03 | 6.6e-08 | 0.02 |
| | Coal & Oil | 2.2e-04 | 3.5e-05 | 1.7e-03 | 3.9e-07 | 0.2 |
| Nickel | Coal | 1.7e-04 | 2.2e-05 | 2.4e-04 | 4.0e-08 | 0.02 |
| | Oil | 2.6e-03 | 2.4e-04 | 2.4e-04 | 5.0e-07 | 0.2 |
| | Coal & Oil | 2.6e-03 | 2.6e-04 | 2.4e-04 | 5.0e-07 | 0.2 |

MEI = Maximally exposed individual.
RELMAP = Regional Lagrangian Model of Air Pollution.

estimated by modeling nickel emissions using the HEM model because most of the nickel exposure is due to oil-fired utilities.

The potential impacts to the MEIs appear to be considerably less than for population exposures for each metal particulate. The maximum RELMAP ambient concentrations (Table 6-17) are each less than 20 percent of the highest HEM concentration for coal- and oil-fired utilities (see Tables 5-1 and 5-4). Also, a comparison of results for MEI risks in Table 6-18 with MEI risks in Tables 6-1 and 6-3 shows the differences in MEI results from the local versus long-range transport analyses.

These metal particulates are also associated with other HAP 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, the factor of 7 can be applied to these other HAPs from coal-fired utilities to roughly estimate the potential impact of long-range transport of HAPs on the overall cancer incidence. Therefore, considering local and LRT, the cancer incidence due to inhalation exposure to HAP emissions is estimated to be as high as about 0.5 cancer cases per year for oil-fired utilities and as high as about 1.3 cases per year for coal-fired utilities. Gas-fired utilities are estimated to present far less population cancer risks than oil-, and

Table 6-19. Summary of the High-End Estimates of the Inhalation Risk Estimates Due to Local and Long-Range Transport

| LOCAL IMPACTS (dispersion within 50 km of each utility plant) [a] | | | | |
|---|---|---|---|---|
| | Oil-fired plants | | Coal-fired plants | |
| Pollutant | Maximally exposed individual (MEI) | Annual increased cancer Incidence | Maximally exposed individual (MEI) | Annual increased cancer incidence |
| Radionuclides[b] | $1 \times 10^{-5}$ | 0.2 | $2 \times 10^{-8}$ | 0.1 |
| Nickel[c] | $5 \times 10^{-5}$ | 0.2 | $4 \times 10^{-7}$ | 0.003 |
| Chromium | $5 \times 10^{-6}$ | 0.02 | $2 \times 10^{-6}$ | 0.02 |
| Arsenic | $1 \times 10^{-5}$ | 0.04 | $3 \times 10^{-6}$ | 0.05 |
| Cadmium | $2 \times 10^{-6}$ | 0.005 | $2 \times 10^{-7}$ | 0.0006 |
| All others[b,d] | $8 \times 10^{-7}$ | 0.005 | $8 \times 10^{-7}$ | 0.004 |
| Total[e] | $6 \times 10^{-5}$ | 0.5 | $4 \times 10^{-6}$ | 0.18 |
| LOCAL PLUS LONG-RANGE IMPACTS (dispersion from utility emission points to borders of continental U.S.) | | | | |
| | Oil-fired plants | | Coal-fired plants | |
| Pollutant | Maximally exposed individual (MEI) | Annual increased cancer incidence | Maximally exposed individual (MEI) | Annual increased cancer incidence[b,f] |
| Radionuclides[b] | $1 \times 10^{-5}$ | 0.2 | Not estimated | 0.7 |
| Nickel[c] | $5 \times 10^{-5}$ | 0.4 | $4 \times 10^{-7}$ | 0.02 |
| Chromium | $5 \times 10^{-6}$ | 0.02 | $3 \times 10^{-6}$ | 0.15 |
| Arsenic | $1 \times 10^{-5}$ | 0.05 | $4 \times 10^{-6}$ | 0.4 |
| Cadmium | $2 \times 10^{-6}$ | 0.006 | $3 \times 10^{-7}$ | 0.005 |
| All others[b,d] | $8 \times 10^{-7}$ | 0.006 | $1 \times 10^{-6}$ | 0.028 |
| Total[e] | $6 \times 10^{-5}$ | 0.5 | $5 \times 10^{-6}$ | 1.3 |

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

[b] Radionuclides and “all others were not modeled with RELMAP. The cancer incidence for these HAPs in the Local Plus Long-Range Impacts section were estimated by assuming the results from the RELMAP modeling for Cr, As, Cd, and Ni could be applied to these other HAPs. Hence, the cancer incidence for radionuclides and “all others” for oil-fired utilities were assumed to be the same as incidence from the local (HEM) analysis, and the incidence for radionuclides and “all others” from coal-fired utilities were assumed to increase by a factor of seven.

[c] Assumes that the nickel mixture is 50 percent as carcinogenic as nickel subsulfide.

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

[e] Aggregate risk (risk due to inhalation exposure to all carcinogenic HAPs, assuming additivity of risks).

[f] These population risk estimates are predicted directly from RELMAP which includes the local and regional impacts.

coal-fired utilities. Therefore, adding these estimates (0.5 + 1.3), it is predicted that up to about 2 cancer cases per year occur due to inhalation exposure to HAP emissions from all utility plants (coal-, oil-, and natural gas-fired) in the continental United States. However, as stated earlier, the IUREs are upper bound estimates of the cancer risks posed by HAPs at low exposure levels, and the true risk is unknown and could be as low as zero. Also, the inhalation high-end risk estimates are conservative. Therefore, the cancer incidence due to inhalation exposure to utility HAP emissions is predicted to be no greater than 2 cases per year in the United States. Most likely fewer than 2 cases/yr occur in the United States due to inhalation exposure to utility HAP emissions.

There are numerous uncertainties in the modeling, the assumptions, the extrapolations, and the resulting cancer incidence estimates. 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, there are considerable uncertainties in the risk estimates and incidence estimates for nickel because of the uncertainties associated with the emissions of different nickel species and the uncertainties in the health effects for each of those forms. Further evaluation of the data, models, and methods could be useful to reduce the uncertainties and to fully evaluate the impacts of long-range transport.

## 6.7 DISCUSSION OF BACKGROUND EXPOSURES

### 6.7.1 Arsenic

Over 250 sites have 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. If a person were exposed to this highest measured concentration of 8 ng/$m^3$ (or 0.008 $\mu$g/$m^3$) for a lifetime and the IURE is used to estimate the cancer risk, this person would have an estimated increased cancer risk of 3 x $10^{-5}$. However, typical background arsenic inhalation exposures, which are likely to be less than 3 ng/$m^3$ would likely pose risks lower than 1 x $10^{-5}$.

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/$m^3$.

The information presented above is useful for gaining a general sense of the potential background air concentrations of arsenic. However, it is difficult to draw conclusions from the comparison of the modeled concentrations and the measurement data. 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.

### 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. However, data presented in a 1994 EPA Report[15] indicate that chromium levels in some urban areas have been measured to be roughly from 0.8 to 16 ng/$m^3$, which would equate to a high-end increased cancer risk of 1 x $10^{-5}$ to 2 x $10^{-4}$ if it is assumed that the chromium is hexavalent and that a person is exposed to those levels for 70 years (i.e., lifetime). In addition, based on data presented in the above 1994 EPA report, nickel levels in a few urban areas are roughly between 0.1 to 20 ng/$m^3$. Assuming the nickel mix is 50 percent as carcinogenic as nickel subsulfide and assuming people are exposed to these levels for a lifetime, this concentration range would correspond to high-end risks of roughly between 3 x $10^{-8}$ to 7 x $10^{-6}$.

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 exceedance. 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 toxicity between the trivalent chromium (Cr III) and the 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.[16,17] 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[2] 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 6-20 and 6-21 present the results of the assessment.

Table 6-20. Chromium Speciation Analysis for Coal-Fired Utilities: Inhalation Risk Estimates due to Chromium Emissions 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) |
|---|---|---|---|---|
| Assume 100% Cr VI | 2 x $10^{-5}$ | 1 x $10^{-5}$ | 69,000 | 0.2 |
| Assume 23% Cr VI | 4 x $10^{-6}$ | 2 x $10^{-6}$ | 2,300 | 0.04 |
| Assume 11% Cr VI | 2 x $10^{-6}$ | 1 x $10^{-6}$ | 110 | 0.02 |
| Assume 0.4% Cr VI | 7 x $10^{-8}$ | 4 x $10^{-8}$ | 0.0 | 0.0007 |

MEI = Maximally exposed individual.
MIR = Maximum individual risk.

[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 6-21. 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 x $10^{-5}$ | 3 x $10^{-5}$ | 40,000 | 0.1 |
| 34% Cr VI | 1 x $10^{-5}$ | 1 x $10^{-5}$ | 2,300 | 0.04 |
| 18% Cr VI | 5 x $10^{-6}$ | 5 x $10^{-6}$ | 2,300 | 0.02 |
| 5% Cr VI | 1.5 x $10^{-6}$ | 1.5 x $10^{-6}$ | 45 | 0.005 |

MEI = Maximally exposed individual.
MIR = Maximum individual risk.

[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.

## 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 $4.29 \times 10^{-3}$ per $\mu g/m^3$.[17] This IURE is the EPA-verified value 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.[18] 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.[19] The Canadians have also reviewed the available data recently and established an IURE of $6 \times 10^{-3}$.[20] The Canadian IURE also appears to be within the plausible range of potency for arsenic.[3] Table 6-22 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 6-22). 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.[21]

At the time emissions data were being analyzed for this report (1992 to 1994), 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,

Table 6-22. High-End Arsenic Inhalation Risk Estimates: Comparison of Results Using the EPRI, EPA-Verified, and Canadian IUREs[a]

| | Risk estimates using EPRI IURE (1.4 x $10^{-3}$ per $\mu g/m^3$) | | | Risk estimates using EPA IURE (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 |

EPA = U.S. Environmental Protection Agency
EPRI = Electric Power Research Institute
IURE = Inhalation unit risk estimate
MEI = Maximally exposed individual
MIR = Maximum individual risk

[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. [17]

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.[2] More recently, EPRI submitted a fax to the EPA summarizing nickel speciation data from 5 test sites.[3] Based on the data presented in the fax from EPRI, 25 to 60 percent of nickel emissions are soluble nickel compounds, 4 to 26 percent are sulfidic nickel compounds, 0 to 4 percent are metallic nickel compounds, and 27 to 70 percent are oxidic nickel compounds.

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.[22,23] Nickel refinery dust and nickel subsulfide are classified as human carcinogens (WOE = A). The IURE for nickel refinery dust is 2.4 x $10^{-4}$. Based on an assumption that nickel subsulfide constitutes 50 percent of the refinery dust, a potency estimate (IURE) of 4.8 x $10^{-4}$ was assigned to nickel subsulfide. 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 International Agency for Research on Cancer (IARC) considers nickel monoxide, nickel hydroxide, and metallic nickel as having sufficient evidence in experimental animals for carcinogenicity.[24] 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.[24] The American Conference of Governmental Industrial Hygienists (ACGIH) has stated that all nickel compounds should be considered carcinogenic

for risk management purposes.[25] 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.[23,24] 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 are 50 percent as carcinogenic as nickel subsulfide) are considered conservative, high-end risk estimates.

To assess the potential impact of the speciation uncertainty, the EPA conducted an assessment for cancer risks using different assumptions for speciation and cancer potency. The assessment (summarized in Table 6-23) provides a range of the potential cancer risks due to nickel emissions.

6.10.1 Alternative Analysis for Estimating Population Risks. Figure 6-15 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 is 0.3 case per year using the assumption that the potency (IURE) of the mixture of nickel compounds emitted from oil-fired utilities is 50 percent the potency of nickel subsulfide, about 0.15 case/yr if the IURE is assumed to be 20 percent as potent as nickel subsulfide, and about 0.1 case per year if the IURE is assumed to be 10 percent nickel subsulfide. Likewise, there would be changes in the number of persons potentially exposed at various risk levels. If the IURE were 20 percent nickel subsulfide, about 9,930 persons would be exposed at an MIR > $1 \times 10^{-6}$. Figure 6-15 does not capture the full potential range of estimated population risks. It is possible that the potency of the mix of nickel compounds emitted from oil-fired utilities is less than 10 percent as carcinogenic as nickel subsulfide. Therefore, the cancer incidence could possibly be lower than that shown in Figure 6-15. The cancer incidence due to nickel emissions could possibly be as low as zero.

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 6-24).

Table 6-23. 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 | $4.8 \times 10^{-4}$ | $9.6 \times 10^{-5}$ | 1,600,000 | 0.4 |
| 20% Ni subsulfide | $9.6 \times 10^{-5}$ | $2 \times 10^{-5}$ | 9,900 | 0.08 |
| 10% Ni subsulfide | $4.8 \times 10^{-5}$ | $9.6 \times 10^{-6}$ | 2,300 | 0.04 |
| 1% Ni subsulfide | $4.8 \times 10^{-6}$ | $9.6 \times 10^{-7}$ | 0.0 | 0.004 |

IURE = Inhalation unit risk estimate
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 $4.8 \times 10^{-4}$ 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.

## 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 have hot-side electrostatic precipitators (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.[26] Since this phenomenon was observed at MWCs, the EPA assumes that it is possible that the same situation may possibly occur at utilities. However, at this time, sufficient information is not available to assess the potential risks due to dioxin emissions from the utility plants with hot-side ESPs. 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.

## 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 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.

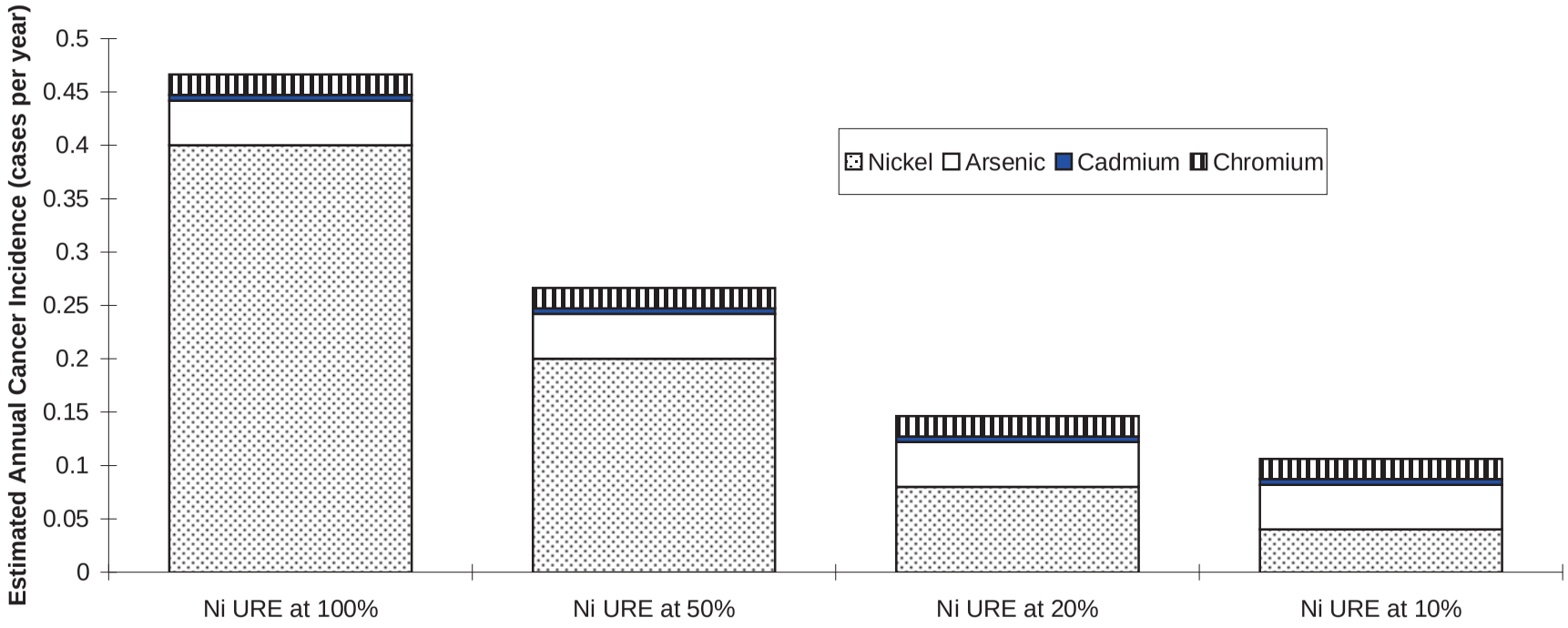



**Figure 6-15. Estimates of Annual Cancer Incidence Due to Inhalation Exposure to HAP Emissions from Oil-fired Electric Utilities Based on the Local Analysis Using Alternative UREs for Nickel (as % of Nickel Subsulfide)**

Table 6-24. Comparison of Nickel Exposure to Various Noncancer Health Benchmarks

| Various health benchmarks for nickel compounds | CARB REL[a] = 0.24 $\mu$g/m$^3$ | EPRI[b] value = 2.4 $\mu$g/m$^3$ |
|---|---|---|
| # People exposed[c] above the benchmark | 0.0 | 0.0 |
| Maximum HQ[d] | 0.82 | 0.082 |

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

[a] This value was obtained from the CARB Hot Spots Program.[5] CARB calculated this number by dividing the Threshold Limit Value (TLV) of 0.1 mg/m$^3$ 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 [27] 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.

### 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.[28] 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.[29] 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[30] 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.[30] 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.[30] 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:

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

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.[30] 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 over-estimate 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.[31] 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).[28]

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.

- Young organisms appear to be inherently susceptible 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 of the EPA Interim Final Utility Report for details on the uncertainty analysis.[6] 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*.[28] 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 associated with these point estimates.

The uncertainty analysis focused on the three HAPs (nickel, arsenic, and chromium) that accounted for over 95 percent of the high-end estimate 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. Other uncertainties which were not amenable to quantitative evaluation are discussed qualitatively in section 6.12. Table 6-25 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. 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 of the EPA Interim Final Utility Report for explanation).[6]

In general, numerical methods (e.g., 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.

## Table 6-25. SUMMARY OF BASIC PARAMETERS USED IN THE INHALATION RISK ASSESSMENT FOR UTILITIES

| 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:<br>●1990 (or 1989, geo mean 1980-8) UDI/EEI data self-reported by the utilities to DOE.<br>●Adjustments made for heating value of different coals<br>●Based on total tonnage<br>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<br>●Average heating values used for coal type (lign = 6800 BTU/lb, bit. = 12688 BTU/lb, subbit. = 9967 BTU/lb) | ●Accuracy of self-reported values<br>●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<br>●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.<br>●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:<br>●Conc. measured in extracted coal, not in coal shipments, reductions in trace element conc. may occur during processing<br>●Coal seams measured may not actually being used for shipment<br>●Coal from other states used at plant may contribute significantly<br>●Variability within a coal seam, between coal seams within a state.<br>OIL:<br>●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<br>●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 facilitestested of varying configurations, combinations of boiler type and APCD.<br>Oil: teesting at 2 facilities.<br>Each test point was at least a triplicate sample | ●Were units tested representative of units in operation<br>●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..<br>Tri.: no distribution can be estimated from single point, value used as the apex bounded by 0 and 1. |

## Table 6-25. 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.<br>●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<br>●Terrain is not a binary parameter and the degree of terrian 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.<br>●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 orexit 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.<br>●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 concentratio nis 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.<br>●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 6-25. 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.<br>●Population mobility varies dramatically within a population and for an individual over time.<br>●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..<br>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 depostion 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 originial 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.

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 6-16 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.

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).

It should be noted that two different trace metal concentrations in oil were used. The original data were from the EPRI's Field Chemistry Emissions Monitoring (FCEM). A subsequent data set was provided by UARG and their contracting lab (SGS Environmental Laboratories). An analysis of these data indicated that the samples were discretely different. It is unknown which "sample population" is most representative of the oil burned. Therefore, the two data sets are treated as two distinct cases and are assumed to be representative of the range of oil being burned by utilities.

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 6-26 and Figure 6-17 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

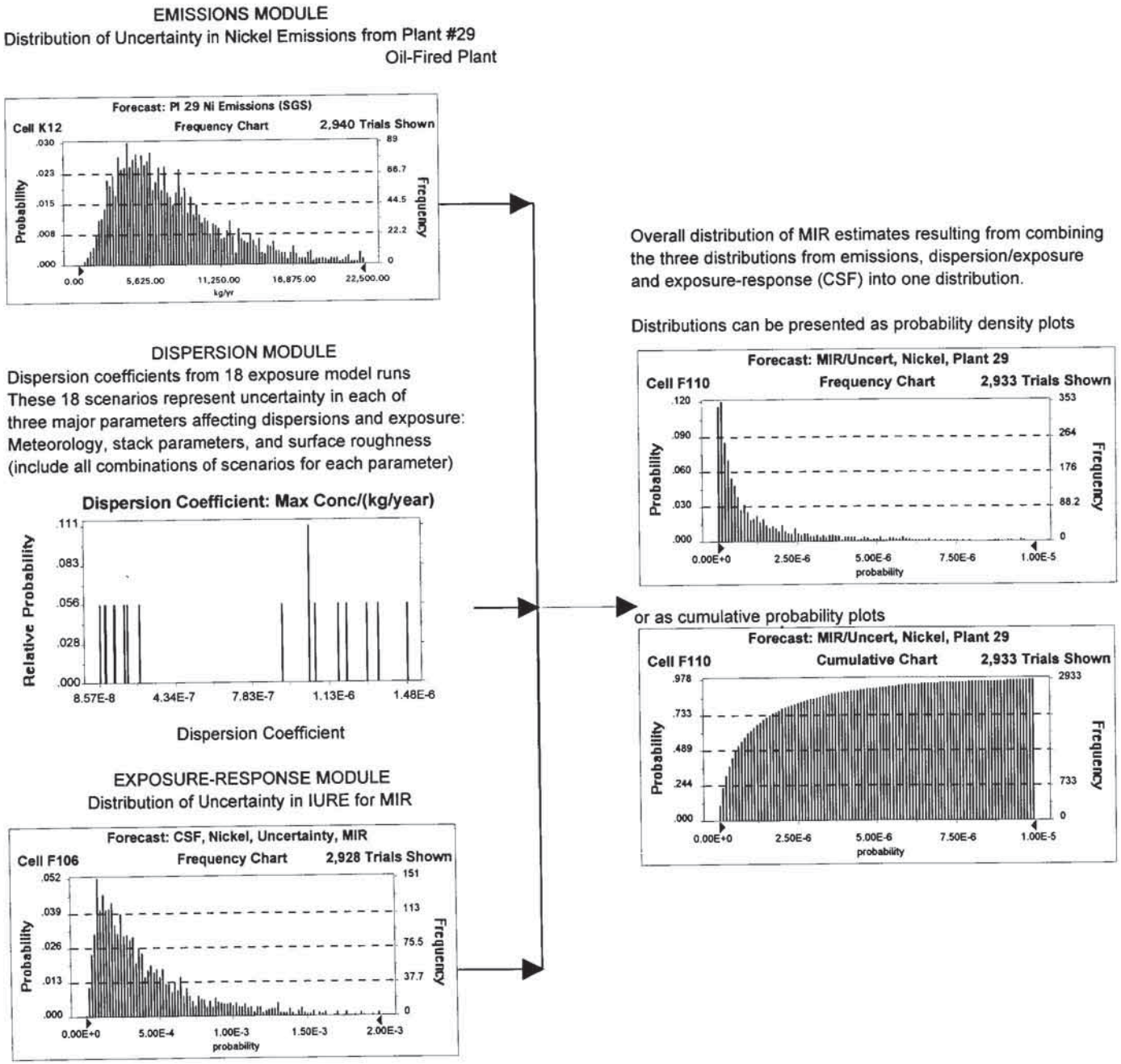



**Figure 6-16. Depiction of Combining Component Uncertainty Distributions (i.e., Emissions, Dispersion, and Exposure-Response) into an Overall Distribution of Uncertainty (e.g., MIR)**



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## Table 6-26. Summary of Results for Monte Carlo Simulation of HAP Emissions (kg/year) from Oil-Fired Plant No. 29

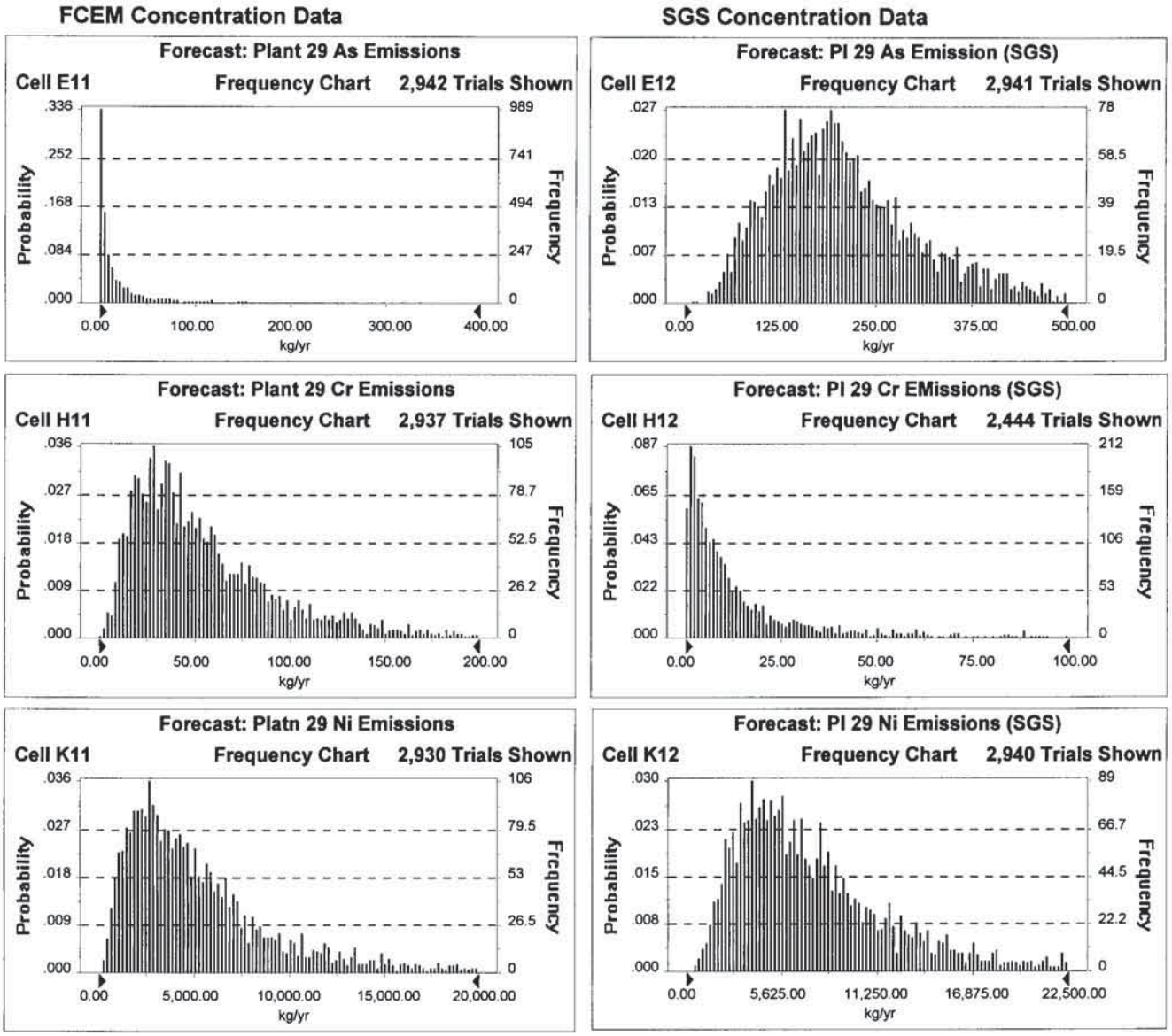
| | | Arsenic emissions | | | Chromium emissions | | | Nickel emissions | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | FCEM | SGS | Combined | FCEM | SGS | Combined | FCEM | SGS | Combined |
| Mean | | 44 | 216 | 130 | 69 | 20 | 45 | 6423 | 8612 | 7527 |
| Initial Point Estimate | | ---------- | 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 |

FCEM = Field Chemical Emissions Monitoring from EPRI program. Original oil concentration data.
SGS = Subsequent data, trace metal analysis conducted by Utility Air Regulatory Group (UARG) HAP committee from samples collected for radionuclide analysis. The concentration was determined by SGS Environmental Laboratories, a contractor to UARG.
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.



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Figure 6-17. Summary of Results of Monte Carlo Simulation of HAP Emissions from Oil-Fired Plant No. 29




FCEM = Field Chemical Emissions Monitoring from EPRI program. Original oil concentration data.

SGS = Subsequent data, trace metal analysis conducted by Utility Air Regulatory Group (UARG) HAP corr from samples collected for radionuclide analysis. The concentration was determined by SGS Environmental Laboratories, a contractor to UARG.

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.

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 6-26, 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 Table 6-26. Summary of Results for Monte Carlo Simulation of HAP Emissions (kg/year) from Oil-fired Plant #29 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 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 1.08, while averages for arsenic, chromium, and nickel were 1.6, 0.68, and 0.97, 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 2,600 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 2,600 would not change the overall results. The EFP tended to underestimate rather than overestimate emissions about 70 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. In addition, variability in fuel composition might also lead to large differences between measured and calculated emissions. 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 of the EPA Interim Final Utility Report for further discussion.[6]

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 this difference 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 6-27. 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 (predicted to be roughly around the 90th or 95th percentile). The uncertainty analysis supports the general conclusion that the baseline risk estimates are likely to be reasonable high-end estimates.

## Table 6-27. 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 |
| Ratio | | | | | | |
| 95th : baseline | 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 : mean | 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 |

FCEM = Field Chemical Emissions Monitoring from EPRI program. Original oil concentration data.

SGS = Subsequent data, trace metal analysis conducted by Utility Air Regulatory Group (UARG) HAP committee from samples collected
The concentration was determined by SGS Environmental Laboratories, a contractor to UARG.

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.

The uncertainty analysis suggests that the most likely inhalation MIRs (i.e., central tendency MIRs) and most likely cancer incidence values (i.e., central tendency cancer incidence estimates) may be roughly 2 to 10 times lower than the high-end MIRs and incidence estimates presented above. In addition, based on results of the HEM modeling and the uncertainty analysis, it is predicted that the average individual risks due to inhalation exposure to utility HAP emissions for the total exposed U.S. population (roughly 200,000,000 people) are roughly 100 to 1,000 times lower than the high-end MIRs.

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 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. This approach assumes that people spend most of their time at home and that the average concentration at their residence represents the average concentration to which they are exposed. Electric utility plants typically have high stacks compared with many other air pollutant point sources. As a result, ground-level concentrations (and concomitant exposures) would tend to vary less with distance than other sources. Therefore, movement by individuals within the grid would have minimal impact on exposures. 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 United States 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.

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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 of freshwater fish is a potential concern in the United States as indicated by numerous fish advisories[19] and mercury-related water quality standards issued by State Agencies. The 1997 EPA *Mercury Study Report to Congress*[20] presents a more complete assessment of the health effects, exposures, risks, ecological effects, sources, and control technologies. This chapter presents an abbreviated assessment of mercury as it is relevant to utilities, which is largely based on information presented in EPA's *Mercury Study Report to Congress*.[20]

Fish consumption dominates the pathway for human and wildlife exposure to methylmercury. The EPA's 1997 *Mercury Study Report to Congress* supports a plausible link between anthropogenic releases of mercury from industrial and combustion sources in the United States and methylmercury in fish. However, these fish methylmercury concentrations also result from existing background concentrations of mercury (which may consist of mercury from natural sources, as well as mercury which has been re-emitted from the oceans or soils) and deposition from the global reservoir (which includes mercury emitted by other countries). Given the current scientific understanding of the environmental fate and transport of this element, it is not possible to quantify how much of the methylmercury in fish consumed by the U.S. population is contributed by U.S. emissions relative to other sources of mercury (such as natural sources and re-emissions from the global pool). As a result, it cannot be assumed that a change in total mercury emissions will be linearly related to any resulting change in methylmercury in fish, nor over what time period these changes would occur. This is an area of ongoing study.

### 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 and its understanding continues to undergo refinement.[3,21-24]

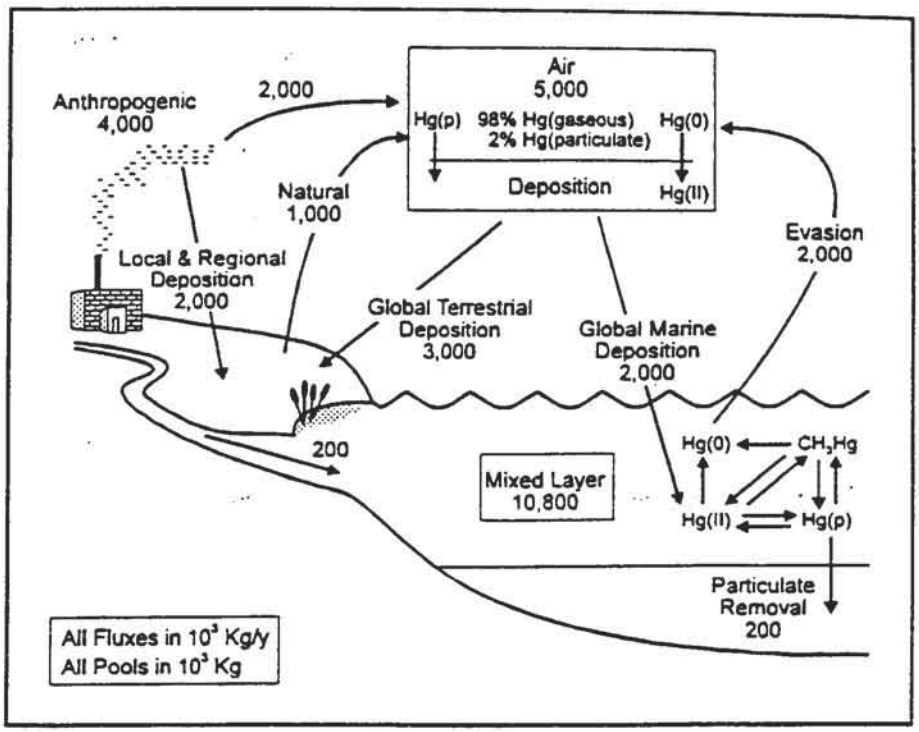
Given the present understanding of the mercury cycle, the flux of mercury from the atmosphere to land or water at any one location is comprised of contributions from: the natural global cycle; the global cycle perturbed by human activities; regional sources; and local sources. Recent advances allow for a general understanding of the global mercury cycle and the impact of anthropogenic sources. It is more difficult to make accurate generalizations of the fluxes on a regional or local scale due to the site-specific nature of emission and deposition processes.

7.1.1.1 The Global Mercury Cycle Past and Present. As a naturally occurring element, mercury is present throughout the environment in both environmental media and biota.[25] In a 1979 report edited by Nriagu, various authors estimated the global distribution of mercury and concluded that by far the largest repository is ocean sediments. Ocean sediments contain an estimated $10^{17}$ g of mercury, mainly as HgS. According to estimates in the report edited by Nriagu, ocean waters contain around $10^{13}$ g, soils and freshwater sediments $10^{13}$ g, the biosphere $10^{11}$ g (mostly in land biota), the atmosphere $10^{8}$ g, and freshwater contains on the order of $10^{7}$ g. This budget excludes "unavailable" mercury in mines and other subterranean repositories. A more recent estimate of the global atmospheric repository by Fitzgerald[3] is 25 Mmol or approximately $5\times10^{9}$ g; this is 50 times the previous estimate of Nriagu.[25]

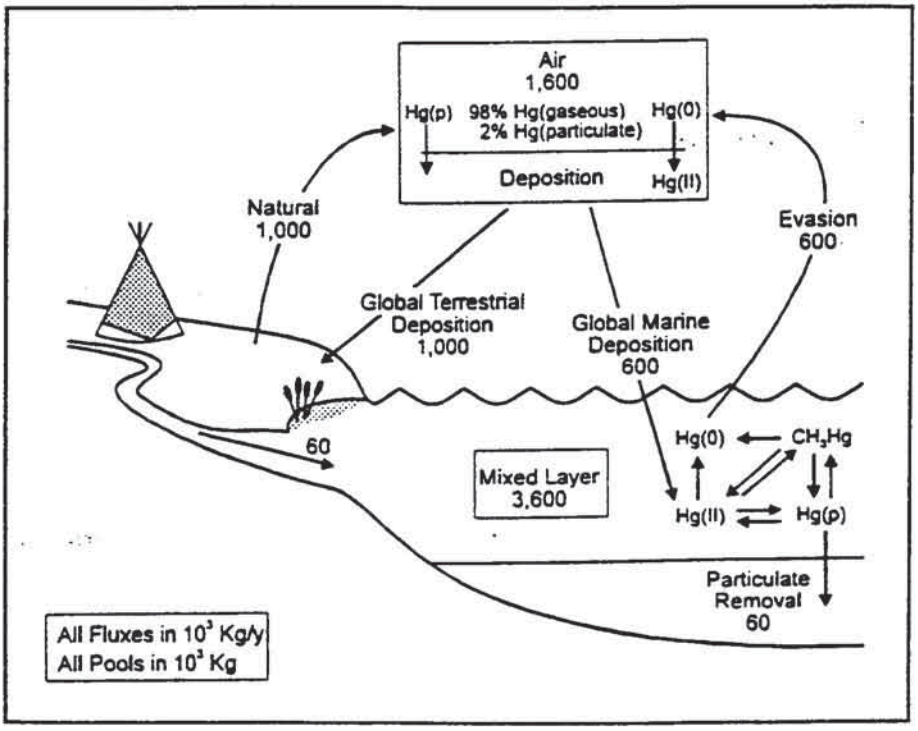
Recent estimates of annual total global mercury emissions from all sources (natural and anthropogenic) are about 5,000 to 5,500 tpy.[26] Of this total, about 1,000 tpy are estimated to be natural emissions and about 2,000 tpy are estimated to be contributions through the natural global cycle of re-emissions of mercury associated with past anthropogenic activity. Current anthropogenic emissions account for the remaining 2,000 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 the global estimates of 5,000 to 5,500 tpy (which are highly uncertain), U. S. anthropogenic mercury emissions are estimated to account for roughly 3 percent of the global total, and U. S. utilities are estimated to account for roughly 1 percent of total global emissions.

A number of different techniques have been used to estimate the pre-industrial mercury concentrations in environmental media before anthropogenic emissions contributed significantly to the global mercury cycle. Figure 7-1 shows estimated current and preindustrial budgets and fluxes. It is difficult to separate current mercury concentrations by origin (i.e., anthropogenic or natural) because of the continuous cycling of the element in the environment. For

**Estimated Current Mercury Budgets and Fluxes**




**Estimated Pre-Industrial Mercury Budgets and Fluxes**




Source: Adapted from Mason et al., 1994.

**Figure 7-1. Comparison of Estimated Current and Pre-Industrial Mercury Budgets and Fluxes**

example, anthropogenic releases of elemental mercury may be oxidized and deposit as divalent mercury far from the source; the deposited mercury may be reduced and re-emitted as elemental mercury only to be deposited again continents away. Not surprisingly, there is a broad range of estimates and a great deal of uncertainty with each. When the estimates are combined, they indicate that between 40 and 75 percent of the current atmospheric mercury concentrations are the result of anthropogenic releases. The Expert Panel on Mercury Atmospheric Processes[27] concluded that pre-industrial atmospheric concentrations constitute approximately one-third of the current atmospheric concentrations. The panel estimated that anthropogenic emissions may currently account for 50-75 percent of the total annual input to the global atmosphere.[27] The estimates of the panel are corroborated by Lindqvist et al.,[28] who estimated that 60 percent of the current atmospheric concentrations are the result of anthropogenic emissions and Porcella,[29] who estimated that this fraction was 50 percent. Horvat et al.[30] assessed the anthropogenic fraction as constituting 40 to 50 percent of the current total. This overall range appears to be in agreement with the several-fold increase noted in inferred deposition rates.[31,32,33] The percentage of current total atmospheric mercury which is of anthropogenic origin may be much 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 re-emitted to the atmosphere from existing reservoirs. Recent estimates indicate that of the approximately 200,000 tons of mercury emitted to the atmosphere since 1890, about 95 percent resides in terrestrial soils, about 3 percent in the ocean surface waters, and 2 percent in the atmosphere.[27] More study is needed before it is possible to accurately differentiate natural fluxes from these reservoirs from re-emissions of mercury originally released from anthropogenic sources. For instance, approximately one-third of total current global mercury emissions are thought to cycle from the oceans to the atmosphere and back again to the oceans, but a major fraction of the emissions from oceans consists of recycled anthropogenic mercury. It is believed that as little as 20 to 30 percent of the current oceanic emissions are from mercury originally mobilized by natural sources.[34] Similarly, a potentially large fraction of terrestrial and vegetative emissions consists of recycled mercury from previously deposited anthropogenic and natural emissions.[27]

Comparisons of contemporary (within the last 15-20 years) measurements and historical records indicate that the total global atmospheric mercury burden has increased since the beginning of the industrialized period by a factor of between two and five. Contamination from some anthropogenic processes that are no longer in use produces continuing significant releases to surface water, groundwater, and the atmosphere. It is estimated that the mercury content of typical lakes and rivers has been increased by a factor of

two to four since the onset of the industrial age.[25] For example, analysis of sediments from Swedish lakes shows mercury concentrations in the upper layers that are two to five times higher than those associated with pre-industrialized times. More recently, researchers in Sweden estimated that mercury concentrations in soil, water and lake sediments have increased by a factor of four to seven in southern Sweden and two to three in northern Sweden in the 20th century.[23] In Minnesota and Wisconsin, an investigation of whole-lake mercury accumulation indicates that the annual deposition of atmospheric mercury has increased by a factor of three to four since pre-industrial times. Similar increases have been noted in other studies of lake and peat cores from this region; results from remote lakes in southeast Alaska also show an increase, though somewhat lower than found in the upper midwest United States.[27]

Although it is accepted that atmospheric mercury burdens have increased substantially since the preindustrial period, it is uncertain whether overall atmospheric mercury levels are currently increasing, decreasing, or remaining stable. Measurements over remote areas of the Atlantic Ocean show increasing levels up until 1990 and a decrease for the period 1990-1994.[35] At some locations in the upper Midwest of the United States, measurements of deposition rates suggest decreased deposition. However, other measurements at remote sites in northern Canada and Alaska show deposition rates that continue to increase.[36,37] Since these sites are subject to global long-range sources and few regional sources, these measurements may indicate a still increasing global atmospheric burden. More research is necessary; a multi-year, world-wide atmospheric mercury measurement program may help to better determine current global trends.[38]

7.1.1.2 Regional and Local Mercury Cycles. According to one estimate, roughly one half of the total anthropogenic mercury emissions eventually enter the global atmospheric cycle;[39] the remainder is removed through local or regional cycles. Mercury emissions from utilities are believed to exist primarily in two forms, divalent or elemental mercury. Divalent mercury, or Hg(II), is a positive ion (missing two electrons) with a electric charge of plus 2 (i.e., $Hg^{++}$, or oxidized mercury). Elemental mercury, or Hg(0), has a neutral charge (i.e., $Hg^{0}$). An estimated 5 to 10 percent of primary Hg(II) emissions are deposited within 100 km of the point of emission and a larger fraction on a regional scale. Hg(0) that is emitted may be removed on a local and regional scale to the extent that it is oxidized to Hg(II). Some Hg(0) may also be taken up directly by foliage; most Hg(0) that is not oxidized will undergo long-range transport due to the insolubility of Hg(0) in water. In general, primary Hg(II) emissions will be deposited on a local and regional scale to the degree that wet deposition processes remove the soluble Hg(II). Dry deposition may also account for some removal of atmospheric Hg(II). Assuming constant emission rates, the quantity of mercury deposited on a regional and local scale can vary depending on source characteristics (especially the species of mercury emitted), meteorological and topographical attributes, and other factors.[27] For

example, deposition rates at some locations have been correlated with wind trajectories and precipitation amounts.[40,41] Although these variations prohibit generalizations of local and regional cycles, such cycles may be established for specific locations. For example, unique mercury cycles have been defined for Siberia on a regional scale[42] and for the area downwind of a German chlor-alkali plant on a local scale.[43] Mercury cycles dependent on local and regional sources have also been established for the Upper Great Lakes region[44,45] and the Nordic countries.[40]

While the overall trend in the global mercury burden since pre-industrial times appears to be increasing, there is some evidence that mercury concentrations in the environment in certain locations have been stable or decreasing over the past few decades. For example, preliminary results for eastern red cedar growing near industrial sources (chlor-alkali, nuclear weapons production) show peak mercury concentrations in wood formed in the 1950s and 1960s, with stable or decreasing concentrations in the past decade.[27] Some results from peat cores and lake sediment cores also suggest that peak mercury deposition in some regions occurred prior to 1970 and may now be decreasing.[31,32,33,37] Data collected over 25 years from many locations in the United Kingdom on liver mercury concentrations in two raptor species and a fish-eating grey heron indicate that peak concentrations occurred prior to 1970. The sharp decline in liver mercury concentrations in the early 1970s suggests that local sources, such as agricultural uses of fungicides, may have led to elevated mercury levels two to three decades ago.[46] Similar trends have been noted for mercury levels in eggs of the common loon collected from New York and New Hampshire.[47] The downward trend in mercury concentrations observed in the environment in some geographic locations over the last few decades generally corresponds to regional mercury use and consumption patterns over the same time frame (consumption patterns are discussed in Volume II of the *Mercury Study Report to Congress*).[20]

### 7.1.2 Atmospheric Processes

Basic processes involved in the atmospheric fate and transport of mercury include: (1) emissions to the atmosphere; (2) transformation and transport in the atmosphere; (3) deposition from the air; and then (4) re-emission to the atmosphere. Each of these processes is briefly described below.

7.1.2.1 Emissions of Mercury. As discussed fully in Volume II of the *Mercury Study Report to Congress*,[20] mercury is emitted to the atmosphere through both naturally occurring and anthropogenic processes. Natural processes include volatilization of mercury in marine and aquatic environments, volatilization from vegetation, degassing of geologic materials (e.g., soils), and volcanic emissions. The natural emissions are thought to be primarily in the elemental mercury form. Conceptually, the current natural emissions can arise from two components: mercury present as part of the pre-industrial equilibrium and mercury mobilized from deep geologic deposits and added to the global cycle by human activity.

Anthropogenic mercury releases are thought to be dominated on the national scale by industrial processes and combustion sources that release mercury into the atmosphere. Available information indicates that stack emissions include both gaseous and particulate forms of mercury. Gaseous mercury, Hg(g), emissions are believed to include both elemental and oxidized chemical forms, while particulate mercury, Hg(p), emissions are thought to be composed primarily of oxidized compounds due to the relatively high vapor pressure of Hg(0). The analytic methods for mercury speciation of exit gasses and emission plumes are being refined, and there is still controversy in this field. Chemical reactions occurring in the emission plume are also possible. Available information suggests that the speciation of mercury emissions depend on the fuel used (e.g., coal, oil), flue gas cleaning and operating temperature, and possibly other factors. The exit stream is thought to range from almost all divalent mercury to nearly all elemental mercury. Most of the mercury emitted at the stack outlet is found in the gas phase although exit streams containing soot can bind up some fraction of the mercury. The divalent fraction is split between gaseous and particle bound phases.[28] Much of this Hg(II) is believed to be mercuric chloride ($HgCl_2$).[48]

An emission factor-based approach was used to develop the nationwide emission estimates for the fossil fuel combustion categories presented in Table 7-1. The emission factors presented are estimates based on ratios of mass mercury emissions to measures of source activities and nationwide source activity levels. The reader should note that the data presented in this table are estimates; uncertainties include the precision of measurement techniques and the calculation of emission factors, estimates of pollutant control efficiency, and nationwide source class activity levels. The estimates may also be based on limited information for a particular source class, thereby increasing the uncertainty in the estimate further. Due to these and other uncertainties, other sources have calculated different total emissions estimates using similar methods.[49]

7.1.2.2 Transformation and Transport of Atmospheric Mercury. Hg(0) has an atmospheric residence time of about one year and will thus be distributed fairly evenly in the troposphere. Oxidized mercury may be deposited relatively quickly if it is precipitated out, leading to a residence time of hours to months. Longer residence times are possible as well; the atmospheric residence time for some Hg(II) associated with fine particles may approach one year.[49]

The transformation of Hg(0)(g) to Hg(II)(aqueous) and Hg(II)(p) in cloud water demonstrates a possible mechanism by which natural and anthropogenic sources of Hg(0) to air can result in mercury deposition to land and water. This deposition can occur far from the source due to the slow rate of Hg(0)(g) uptake in cloud water. It has been suggested that this mechanism is important in a global sense for Hg pollution, while direct wet deposition of anthropogenic Hg(II) is the most important locally.[3,28] Gaseous Hg(II) is expected to deposit at a faster rate after release than particulate Hg(II) assuming that most

Table 7-1. Best Point Estimates of National Mercury Emission Rates by Category

| Sources of mercury[a] | 1994-1995 Mg/yr[b] | 1994-1995 tons/yr[b] | % of Total inventory[b] |
|---|---|---|---|
| Area sources | 3.1 | 3.4 | 2.2 |
| Lamp breakage | 1.4 | 1.5 | 1.0 |
| General laboratory use | 1.0 | 1.1 | 0.7 |
| Dental preparations | 0.6 | 0.7 | 0.4 |
| Landfills | <0.1 | <0.1 | 0.0 |
| Mobile sources | c | c | c |
| Paint use | c | c | c |
| Agricultural burning | c | c | c |
| Point Sources | 140.9 | 155.7 | 97.8 |
| Combustion sources | 125.2 | 137.9 | 86.9 |
| Utility boilers | 46.8 | 51.5 | 32.6 |
| Coal | (46.7)[d] | 51.3 | 32.5 |
| Oil | (0.2) | (0.2) | (0.1) |
| Natural gas | (<0.1) | (<0.1) | (0.0) |
| MWCs[h] | 26.9 | 29.6 | 18.7 |
| Commercial/industrial boilers | 25.8 | 28.4 | 17.9 |
| Coal | (18.8) | (20.7) | (13.1) |
| Oil | (7.0) | (7.7) | (4.9) |
| MWIs[h] | 14.6 | 16.0 | 10.1 |
| Hazardous waste combustors[e] | 6.4 | 7.1 | 4.4 |
| Residential boilers | 3.3 | 3.6 | 2.3 |
| Oil | (2.9) | (3.2) | (2.0) |
| Coal | (0.4) | (0.5) | (0.3) |
| SSIs | 0.9 | 1.0 | 0.6 |
| Wood-fired boilers[f] | 0.2 | 0.2 | 0.1 |
| Crematories | <0.1 | <0.1 | 0.0 |
| Manufacturing sources | 14.4 | 15.8 | 10.0 |
| Chlor-alkali | 6.5 | 7.1 | 4.5 |
| Portland cement[e] | 4.4 | 4.8 | 3.1 |
| Pulp and paper manufacturing | 1.7 | 1.9 | 1.2 |
| Instruments manufacturing | 0.5 | 0.5 | 0.3 |
| Secondary Hg production | 0.4 | 0.4 | 0.3 |
| Electrical apparatus | 0.3 | 0.3 | 0.2 |
| Carbon black | 0.3 | 0.3 | 0.2 |
| Lime manufacturing | 0.1 | 0.1 | 0.1 |
| Primary lead | 0.1 | 0.1 | 0.1 |
| Primary copper | <0.1 | <0.1 | 0.0 |
| Fluorescent lamp recycling | <0.1 | <0.1 | 0.0 |
| Batteries | <0.1 | <0.1 | 0.0 |
| Primary Hg production | c | c | c |
| Mercury compounds | c | c | c |
| Byproduct coke | c | c | c |
| Refineries | c | c | c |
| Miscellaneous sources | 1.3 | 1.4 | 0.9 |
| Geothermal power | 1.3 | 1.4 | 0.9 |
| Turf products | g | g | g |
| Pigments, oil, etc. | g | g | g |
| TOTAL | 144 | 158 | 100 |

[a] MWC = Municipal waste combustor; MWI = medical waste incinerator; SSI = sewage sludge incinerator.
[b] Numbers do not add exactly because of rounding.
[c] Insufficient information to estimate 1994-1995 emissions.
[d] Parentheses denote subtotal within larger point source category.
[e] For the purpose of this inventory, cement kilns that burn hazardous waste for fuel are counted as hazardous waste combustors.
[f] Includes boilers only; does not include residential wood combustion (wood stoves).
[g] Mercury has been phased out of use.
[h] U.S. EPA has finalized emission guidelines for these source categories which will reduce mercury emissions by at least an additional 90 percent over 1995 levels.

of the particulate matter is less than 1 μm in diameter. An atmospheric residence time of ½ - 2 years for elemental mercury compared to as little as hours for some Hg(II) species is expected.[50] This behavior is observed in the modeling results presented in this effort as well. It is possible that dry deposition of Hg(0) can occur from ozone mediated oxidation of elemental mercury taking place on wet surfaces, but this is not expected to be comparable in magnitude to the cloud droplet mediated processes.

This great disparity in atmospheric residence time between Hg(0) and the other mercury species leads to very much larger scales of transport and deposition for Hg(0). Generally, air emissions of Hg(0) from anthropogenic sources, fluxes of Hg(0) from contaminated soils and water bodies, and natural fluxes of Hg(0) all contribute to a global atmospheric mercury reservoir with a holding time of ½ to 2 years. Global atmospheric circulation systems can take Hg(0) emissions from their point of origin and carry them anywhere on the globe before transformation and deposition occur. Emissions of all other forms of mercury are likely to be deposited to the earth's surface before they thoroughly dilute into the global atmosphere. Continental-scale atmospheric modeling, such as that performed for this study using the Regional Lagrangian Model of Air Pollution (RELMAP), can explicitly simulate the atmospheric lifetime of gaseous and particulate mercury species, but not Hg(0). Although Hg(0) is included as a modeled species in the RELMAP analysis, the vast majority of Hg(0) emitted in the simulation transports outside the spatial model domain without depositing, and the same is generally thought to happen in the real atmosphere. Natural Hg(0) emissions and anthropogenic Hg(0) emissions from outside the model domain are simulated in the form of a constant background Hg(0) concentration of 1.6 ng $m^{-3}$, approximating conditions observed in remote oceanic regions.[3] This background Hg(0) concentration is subject to simulated wet deposition by the same process as explicitly modeled anthropogenic sources of Hg(0) within the model domain.

Explicit numerical models of global-scale atmospheric mercury transport and deposition have not yet been developed. As the general understanding of the global nature of atmospheric mercury pollution develops, numerical global-scale atmospheric models will surely follow.

7.1.2.3. Deposition of Mercury. The divalent species emitted, either in the vapor or particulate phase, are thought to be subject to much faster atmospheric removal than elemental mercury.[51,52] Both particulate and gaseous divalent mercury is assumed to dry deposit (this is defined as deposition in the absence of precipitation) at significant rates when and where measurable concentrations of these mercury species exist. The deposition velocity of particulate mercury is dependent on atmospheric conditions and particle size. Particulate mercury is also assumed to be subject to wet deposition due to scavenging by cloud microphysics and precipitation. The gaseous divalent mercury emitted is also expected to be scavenged readily by

precipitation. Divalent mercury species have much lower Henry's law constants than elemental mercury, and thus are assumed to partition strongly to the water phase. Dry deposition of gas phase divalent mercury is thought to be significant due to its reactivity with surface material. Overall, gas phase divalent mercury is more rapidly and effectively removed by both dry and wet deposition than particulate divalent mercury,[51,52,53] a result of the reactivity and water solubility of gaseous divalent mercury.

In contrast, elemental mercury vapor is not thought to be susceptible to any major process of direct deposition to the earth's surface due to its relatively high vapor pressure and low water solubility. On non-assimilating surfaces elemental mercury deposition appears negligible,[51] and though elemental mercury can be formed in soil and water due to the reduction of divalent mercury species by various mechanisms, this elemental mercury is expected to volatilize into the atmosphere.[27] In fact, it has been suggested that *in-situ* production and afflux of elemental mercury could provide a buffering role in aqueous systems, as this would limit the amount of divalent mercury available for methylation.[3] Water does contain an amount of dissolved gaseous elemental mercury,[54] but it is minor in comparison to the dissolved-oxidized and particulate mercury content.

There appears to be a potential for deposition of elemental mercury via plant-leaf uptake. Lindberg et al.[51] indicated that forest canopies could accumulate elemental mercury 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 elemental mercury from the atmosphere, resulting in a deposition velocity. Recent evidence[55] indicates that this does occur but only when air concentrations of elemental mercury are above an equilibrium level for the local forest ecosystem. At lower air concentration levels, the forest appears to act as a source of elemental mercury to the atmosphere, with the measured mercury flux in the upward direction. Lindberg et. al.[56] noted this may be explained by the volatilization of elemental mercury from the canopy/soil system, most likely the soil. Hanson et al.[55] stated that "dry foliar surfaces in terrestrial forest landscapes may not be a net sink for atmospheric elemental mercury, 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." Similarly, Mosbaek et al.[57] convincingly 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 elemental mercury released from the plant-soil system greatly exceeds the amount collected from the air by the plants. It is also likely that many plant/soil systems accumulate airborne elemental mercury when air concentrations are higher than the long-term average for the particular location, and release elemental mercury when air concentrations fall below the local long-term average. On regional and global scales, dry deposition of Hg(0) does not appear to be a

significant pathway for removal of atmospheric mercury, although approximately 95 percent or more of atmospheric mercury is Hg(0).[3]

There is an indirect pathway, however, by which elemental mercury vapor released into the atmosphere may be removed and deposited to the earth's surface. Chemical reactions occur in the aqueous phase (cloud droplets) that both oxidize elemental mercury to divalent mercury and reduce the divalent mercury to elemental mercury. The most important reactions in this aqueous reduction-oxidation balance are thought to be oxidation of elemental mercury with ozone, reduction of divalent mercury by sulfite ($SO_3^{-2}$) ions, or complexation of divalent mercury with soot to form particulate divalent mercury:

$Hg(0)(g) \rightarrow Hg(0)(aq)$
$Hg(0)(aq) + O_3(aq) \rightarrow Hg(II)(aq)$
$Hg(II)(aq) + \text{soot/possible evaporation} \rightarrow Hg(II)(p)$
$Hg(II)(aq) + SO_3^{-2}(aq) \rightarrow Hg(0)(aq)$
(g) = gas phase molecule
(aq) = aqueous phase molecule
(p) = particulate phase molecule

The Hg(II) produced from 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 of Hg(II) by sulfite. Thus, a steady state concentration of Hg(II)(aq) 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 $SO_2(g)$.[28] Note that $H^+$ and $SO_2(g)$, although not apparent in the listed atmospheric reactions, control the formation of sulfite.

The Hg(II)(aq) produced would then be susceptible to atmospheric removal via wet deposition. The third reaction, however, may transform most of the Hg(II)(aq) into the particulate form, due to the much greater amounts of soot than mercury in the atmosphere. The soot concentration will not be limiting compared to the concentration of Hg(II)(aq), and S atoms in the soot matrix will bond readily to the Hg(II)(aq). The resulting Hg(II)(p) can then be removed from the atmosphere by wet deposition (if the particle is still associated with the cloud droplet) or dry deposition (following cloud droplet evaporation). It is possible that dry deposition of Hg(0) can occur from ozone mediated oxidation of elemental mercury taking place on wet surfaces, but this is not expected to be comparable in magnitude to the cloud droplet mediated processes.[58]

Mercury released into the atmosphere from natural and anthropogenic sources deposits mainly as Hg(II), from either direct deposition of emitted Hg(II) or from conversion of emitted elemental Hg(0) to Hg(II) through ozone-mediated reduction. The former process may result in elevated deposition rates around atmospheric emission sources and the latter process results in regional/global transport followed by deposition.

There is still a great deal of uncertainty with respect to the amount of dry deposition of mercury. Once deposited, mercury appears to bind tightly to certain soil components. The deposited Hg(II) may revolatilize through reduction and be released back to the atmosphere as Hg(0). Soil Hg(II) may also be methylated to form methylmercury; these two forms may remain in the soil or be transported through the watershed to a water body via runoff and leaching. Mercury enters the water body through direct deposition on the watershed, and mercury in water bodies has been measured in both the water column and the sediments. Hg(II) in the water body may also be methylated to form methylmercury; both Hg(II) and methylmercury may be reduced to form Hg(0) which is reintroduced to the atmosphere.

7.1.2.4 Re-emissions of Mercury into the Atmosphere. Re-emission of deposited mercury results most significantly from the evasion of elemental mercury from the oceans. In this process, anthropogenically emitted mercury is deposited to the oceans as Hg(II) and then reduced to volatile Hg(0) and re-emitted. According to one estimate, this process accounts for approximately 30 percent (10 Mmol/year) of the total mercury flux to the atmosphere.[39] Overall, 70 to 80 percent of total current mercury emissions may be related to anthropogenic activities.[34] By considering the current global mercury budget and estimates of the preindustrial mercury fluxes, Mason et al.[39] estimate that total emissions have increased by a factor of 4.5 since preindustrial times, which has subsequently increased the atmospheric and oceanic reservoirs by a factor of 3. The difference is attributed to local deposition near anthropogenic sources. Although the estimated residence time of elemental mercury in the atmosphere is about 1 year, the equilibrium between the atmosphere and ocean waters results in a longer time period needed for overall change to take place for reservoir amounts. Therefore, by substantially increasing the size of the oceanic mercury pool, anthropogenic sources have introduced long- term perturbations into the global mercury cycle. Fitzgerald and Mason[34] estimate that if all anthropogenic emissions were stopped, it would take about 15 years for mercury pools in the oceans and the atmosphere to return to pre-industrial conditions. The Science Advisory Board, in its review of the EPA's Mercury Study, concluded that it could take significantly longer. There is scientific agreement, however, that the slow release of mercury from terrestrial sinks to freshwater and coastal waters will persist for a long time, probably decades, which effectively increases the length of time anthropogenic emissions would impact the environment. This is particularly significant given that the surface soils contain most of the pollution-derived mercury of the industrial period. As a result, it is uncertain at this time how long it would take after reductions in anthropogenic emissions for mercury levels in the global environment, including fish levels, to return to true background levels. The slow release of mercury from terrestrial sinks to freshwater and coastal waters will likely persist for much longer, possibly decades, effectively increasing the lifetime of anthropogenic mercury further.[34] This may be particularly significant considering that surface soils currently contain most of the pollution-derived

mercury of the industrial period. Thus, re-emissions of past anthropogenic mercury emissions will contribute to long-term influences on the global biogeochemical cycle for mercury.

### 7.1.3 Terrestrial and Aquatic Fate of Mercury

7.1.3.1 Mercury in Soil. Once deposited, the Hg(II) species are subject to a wide array of chemical and biological reactions. Soil conditions (e.g., pH, temperature and soil humic content) are typically favorable for the formation of inorganic Hg(II) compounds such as $HgCl_2$, $Hg(OH)_2$ and inorganic Hg(II) compounds complexed with organic anions.[59] Although inorganic Hg(II) compounds are quite soluble (and, thus, theoretically mobile) they form complexes with soil organic matter (mainly fulvic and humic acids) and mineral colloids; the former is the dominating process. This is due largely to the affinity of Hg(II) and its inorganic compounds for sulfur-containing functional groups. This complexing behavior greatly 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 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 affinity of mercury species for soil results in soil acting as a large reservoir for anthropogenic mercury emissions.[60,23] For example, note the mercury budget proposed by Meili et al.[60] Even if anthropogenic emissions were to stop entirely, leaching of mercury from soil would not be expected to diminish for many years.[23] Hg(0) can be formed in soil by reduction of Hg(II) compounds/complexes mediated by humic substances.[25] This Hg(0) will vaporize eventually and re-enter the atmosphere. Methylmercury can be formed by various microbial processes acting on Hg(II) substances. Approximately 1-3 percent of the total mercury in surface soil is methylmercury, and as is the case for Hg(II) species, it will be bound largely to organic matter. The other 97-99 percent of total soil mercury can be considered largely Hg(II) complexes, although a small fraction of Hg in typical soil will be Hg(0).[61]

7.1.3.2 Plant and Animal Uptake of Mercury. While there is a great deal of uncertainty surrounding air-to-plant transfer of mercury, some evidence indicates that this pathway may be an important source of mercury to soils via defoliation. Overall, mercury concentrations in plants, even those whose main uptake appears to be from the air, are expected from modeling results to be low. This prediction is corroborated by low reported mercury concentrations in most green plants, although the data set of these values is not complete and there are some exceptions. The bulk of the mercury in plants appears to be inorganic.[50] Livestock typically accumulate little mercury from foraging or silage/grain consumption, and the mercury content of meat is low. Due to these factors, the terrestrial

pathway is not expected to be significant, particularly when compared to the consumption of fish by humans. Since this is not an exposure pathway of concern for mercury, it was not included in the modeling that follows.

7.1.3.3 Mercury in the Freshwater Ecosystem. There are a number of pathways by which mercury can enter the freshwater environment: Hg(II) and methylmercury from atmospheric deposition (wet and dry) can enter water bodies directly; Hg(II) and methylmercury can be transported to water bodies in runoff (bound to suspended soil/humus or attached to dissolved organic carbon); and Hg(II) and methylmercury can leach into the water body from groundwater flow in the upper soil layers. Once in the freshwater system, the same complexation and transformation processes that occur to mercury species in soil will occur in aquatic media along with additional processes due to the aqueous environment. Mercury concentrations are typically reported for particular segments of the water environment; the most common of these are the water column (further partitioned as dissolved or attached to suspended material), the underlying sediment (further divided into surface sediments and deep sediments), and biota (particularly fish).

Most of the mercury in the water column, Hg(II) and methylmercury, will be bound to organic matter, either to dissolved organic carbon[62,28] or to suspended particulate matter. In most cases, studies that refer to the dissolved mercury in water include mercury complexes with DOC. Studies indicate that about 25-60 percent of Hg(II) and methylmercury organic complexes are particle-bound in the water column. The rest is in the dissolved, bound-to-DOC phase.[25,63] Hg(0) is produced in fresh water by humic acid reduction of Hg(II) or demethylation of methylmercury. Some will remain in the dissolved gaseous state, but most will volatilize. As noted previously, Hg(0) constitutes very little of the total mercury in the water column but may provide a significant pathway for the evolution of mercury out of the water body via Hg(II) or methylmercury -> Hg(0) -> volatilization. For many lakes, however, sedimentation of the Hg(II) and methylmercury bound to particulate matter is expected to be the dominant process for removal of mercury from the water column.[24]

Generally, no more than 25 percent of the total mercury in a water column exists as a methylmercury complex; typically, less than 10 percent is observed. This is a result of methylation of Hg(II) which is thought to occur in the bottom sediment and the water column by microbial action and abiotic processes. An equilibrium is soon established between Hg(II) and methylmercury in freshwater systems; in a number of sediment-water systems, it has been found that methylmercury concentrations in waters were independent of water column residence time or time in contact with sediments.[64] Methylmercury in the water column which is lost through demethylation, exported downstream, or taken up by biota is thought to be replaced by additional methylation of Hg(II) compounds to sustain equilibrium.

Once entering a water body, 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. After entry, the movements of mercury through any specific water body may be unique. Only mercury in the water column, the sediment, and other aquatic biota appears to be available to aquatic organisms for uptake.

Methylation appears to be a key step in the entrance of mercury into the food chain.[24] The biotransformation of inorganic mercury species to methylated organic species in water bodies can occur in the sediment and the water column. Abiotic processes (e.g., humic and fulvic acids in solution) also appear to methylate the mercuric ion. All mercury compounds entering an aquatic ecosystem are not methylated, and demethylation reactions as well as volatilization of dimethylmercury decrease the amount of methylmercury available in the aquatic environment. It is clear that there is a large degree of scientific uncertainty and variability among water bodies concerning the processes that methylate mercury.[24]

Methylmercury is very bioavailable and accumulates in fish through the aquatic food web; nearly 100 percent of the mercury found in fish muscle tissue is methylated.[24] 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 water body. Most of the total methylmercury production ends up in biota, particularly fish. Overall, methylmercury production and accumulation in the freshwater ecosystem places this pollutant into a position to be ingested by fish-eating organisms.

Methylmercury appears to be efficiently passed through the aquatic food web to the highest trophic level consumers in the community (e.g., piscivorous fish). At this point it can be contacted by fish-consuming humans through ingestion. Methylmercury appears to pass from the human gastrointestinal tract into the bloodstream more efficiently than the divalent species.

7.1.3.4 Fate of Mercury in Marine Environments. As noted earlier, mercury is an atmophillic element and, as such, its global transport occurs primarily through the atmosphere. Elemental mercury, the principle species found in the atmosphere, has a high vapor pressure and a low solubility in water. As a result of these properties, the half-life of atmospheric mercury is thought to be a year or longer. Elemental mercury appears to be deposited to ocean waters primarily through wet deposition. Oxidizing reactions in the atmosphere may also play a role in the conversion of elemental mercury to more reactive atmospheric species which are subsequently deposited.

Mercury found in ocean waters and sediments comprises a large reservoir of the total mercury on the planet. The conceptualization of oceans as reservoirs of mercury is fitting for they serve both as

sources of mercury to the atmosphere and as environmental mercury sinks.[65,66,67] The forms and species of mercury present in the ocean waters and sediments may be transformed as a result of both biotic and abiotic factors within the ocean. The most significant species of mercury from a human health perspective is monomethylmercury (MHg). MHg shows strong evidence of bioaccumulation and biomagnification in the marine food web, potentially posing risks to consumer species (particularly apex marine predators and piscivores).[24]

## 7.2 MERCURY HEALTH EFFECTS

A brief summary of the health effects of methylmercury is presented here. The 1997 EPA *Mercury Study Report to Congress*[68] contains more information on the health effects of mercury and mercury compounds.

Most of the population of the earth have some exposure to mercury as a result of normal daily activities. The general population may be exposed to mercury through inhalation of ambient air; consumption of contaminated food, water, or soil; and/or dermal exposure to substances containing mercury. In addition, some quantity of mercury is released from dental amalgam.

The health effects literature contains many investigations of populations with potentially high exposure to mercury, including industrial workers, people living near point sources of mercury emissions, people who consume large amounts of fish, and dental professionals. There also are numerous studies of populations exposed to high levels of mercury, such as the Minamata poisoning episode in Japan. Volume IV of the EPA's *Mercury Study Report to Congress*[69] presents measured and predicted mercury exposure for various U.S. populations.

The form of mercury which is emphasized here is methylmercury because methylmercury is the form of primary interest for human exposures for this report. It is acknowledged that humans can be exposed to elemental and inorganic mercury and that certain populations can be exposed to many types of organic mercurials, such as antiseptics and pesticides, which are not discussed here.

### 7.2.1 Toxicokinetics

The toxicokinetics (i.e., absorption, distribution, metabolism, and excretion) of mercury is highly dependent on the form of mercury to which a receptor has been exposed. Below is a brief summary of the toxicokinetics information for methylmercury.

Methylmercury is rapidly and extensively absorbed through the gastrointestinal tract. Absorption information following inhalation exposures is limited. This form of mercury is distributed throughout the body and easily penetrates the blood-brain and placental barriers in humans and animals. Methylmercury transport into tissues appears to be mediated by the formation of a methylmercury-cysteine complex.

This complex is structurally similar to methionine and is transported into cells via a widely distributed neutral amino acid carrier protein. Methylmercury in the body is considered to be relatively stable and is only slowly demethylated to form mercuric mercury in rats. It is hypothesized that methylmercury metabolism may be related to a latent or silent period observed in epidemiological studies observed as a delay in the onset of specific adverse effects. Methylmercury has a relatively long biological half-life in humans; estimates range from 44 to 80 days. Excretion occurs via the feces, breast milk, and urine.

7.2.2 Biological Effects

The primary targets for toxicity of mercury and mercury compounds are the nervous system, kidney, and developing fetus. Other systems that may be affected include the respiratory, cardiovascular, gastrointestinal, hematologic, immune, and reproductive systems. A brief summary of the biological effects of methylmercury is presented here.

Three human studies that examined the relationship between methylmercury and cancer incidence were considered extremely limited because of study design inappropriate for risk assessment or incomplete data reporting. Evidence from animal studies provides limited evidence of carcinogenicity. Male ICR and B6C3F1 mice exposed orally to methylmercuric chloride were observed to have an increased incidence of renal adenomas, adenocarcinomas, and carcinomas. Renal epithelial cell hyperplasia and tumors, however, were observed only in the presence of profound nephrotoxicity suggesting that the tumors may be a consequence of reparative changes to the damaged kidneys. Tumors were observed at a single site, in a single species and sex.

Methylmercury appears to be clastogenic but not a potent mutagen. Studies have also shown evidence that methylmercury may induce mammalian germ cell chromosome aberrations. There are a number of studies in both humans and experimental animals that show methylmercury to be a developmental toxicant. Neurotoxicity in offspring is the most commonly observed effect and the effect seen at lowest exposures.

A significant body of human studies exists for evaluating the potential systemic toxicity of methylmercury. This data base is the result of studying two large scale poisoning episodes in Japan and Iraq as well as several epidemiological studies assessing populations that consume significant quantities of fish. In addition, much research on the toxicity of methylmercury has been conducted in animals including non-human primates.

The critical target for methylmercury toxicity is the nervous system. The developing fetus may be at particular risk from methylmercury exposure. Offspring born of women exposed to high doses of methylmercury during pregnancy have exhibited a variety of developmental neurological abnormalities, including the following:

delayed onset of walking, delayed onset of talking, cerebral palsy, altered muscle tone and deep tendon reflexes, and reduced neurological test scores. Maternal toxicity may or may not have been present during pregnancy for those offspring exhibiting adverse effects. For the general population, the critical effects observed following methylmercury exposure are multiple central nervous system effects including ataxia and paresthesia.

A latent or silent period has been observed in some epidemiological and animal studies indicating a delay in the onset of adverse effects. It is hypothesized this delay may be related to methylmercury metabolism.

7.2.3 Sensitive Subpopulations

A susceptible population is a group that may experience more severe adverse effects at comparable exposure levels or adverse effects at lower exposure levels than the general population. The greater response of these sensitive subpopulations may be a result of a variety of intrinsic or extrinsic factors. For mercury, the most sensitive subpopulations may be developing organisms. Data are also available indicating that other factors may be associated with the identification of sensitive subpopulations including the following: age; gender; dietary insufficiencies of zinc, glutathione, or antioxidants; predisposition for autoimmune glomerulonephritis; and predisposition for acrodynia.

7.2.4 Interactions

There are data demonstrating that a number of substances affect the pharmacokinetics and/or toxicity of mercury compounds. Of most interest is the potential interaction of selenium and mercury. Selenium is known to bioaccumulate in fish, so exposure to methylmercury from fish consumption may be associated with exposure to increased levels of selenium. There are data indicating that selenium co-administered with methylmercury can form selenium-methylmercury complexes. The formation of these complexes may temporarily prevent methylmercury-induced tissue damage but also may delay excretion of the methylmercury. Thus, formation of selenium-methylmercury complexes may not reduce methylmercury toxicity but rather may delay onset of symptoms. More information is needed to understand the possible interaction of selenium with methylmercury. There is also potential for interaction between various forms of mercury and ethanol, thiol compounds, tellurium, potassium dichromate, zinc, atrazine, and vitamins C and E.

7.2.5 Hazard Identification/Dose-Response Assessment

The available toxicological and epidemiological evidence was evaluated, and U.S. EPA risk assessment guidelines and methodologies were applied to hazard identification for various endpoints; namely, carcinogenicity, germ cell mutagenicity, developmental toxicity, and

general systemic toxicity. Data supported quantitative assessments of systemic toxicity. An oral reference dose (RfD[a]) was calculated for methylmercury. U.S. EPA derived the RfD for methylmercury by extrapolating from the high-dose exposures that occurred in the Iraq incident. Data for carcinogenicity of inorganic and methylmercury were judged to be inadequate in humans and limited from animal bioassays. The carcinogenicity data for methylmercury were not sufficient to support a quantitative assessment. Table 7-2 summarizes the hazard identification and dose-response information for organic mercury.

### 7.2.6 Ongoing Research

While much data has been collected on the potential toxicity of mercury and mercury compounds, much is still unknown. Two ongoing epidemiological studies are now providing critical information on the developmental toxicity of methylmercury. One study, being conducted in the Seychelles Islands, is evaluating dose-response relationships in a human population with dietary exposures (fish) at levels believed to be in the range of the threshold for developmental toxicity. The second study, conducted in the Faroe Islands, is assessing mercury exposure in a population that consumes a relatively large quantity of marine fish and marine mammals. Children exposed to methylmercury *in utero* and followed through 6 years of age have been assessed for mercury exposure and neurological developmental. Because of various limitations and uncertainties in all of the available data, the U.S. EPA and other Federal agencies intend to participate in an interagency review of the human data on methylmercury, including the most recent studies from the Seychelle Islands and the Faroe Islands. The purposes of this review are to refine the estimates of the level of exposure to mercury associated with subtle neurological endpoints and to further consensus among all of the Federal agencies. After this process, the U.S. EPA will determine if a change in the RfD for methylmercury is warranted.

### 7.2.7 Research Needs

Specifically, information is needed to reduce the uncertainties associated with the current oral RfD for methylmercury. More work with respect to both dose and duration of exposure would also allow for potentially assessing effects above the RfD. Limited evidence suggests that methylmercury is a possible human carcinogen. Research on mode of action in induction of tumors at high doses will be of particular use in defining the nature of the dose response relationship for carcinogenicity.

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[a] The oral RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subpopulations) that is likely to be without an appreciable risk of deleterious health effects during a lifetime.

Table 7-2. Summary of U.S. EPA Hazard Identification/Dose-Response Assessment for Methylmercury

| Form of mercury | Oral RfD (mg/kg-day) | Inhalation RfC ($mg/m^3$) | Cancer weight-of-evidence rating | Cancer slope factor | Germ cell mutagenicity | Developmental toxicity data base characterization |
|---|---|---|---|---|---|---|
| Organic | 0.0001[a] (methyl-mercury) | n/a | C, possible human carcinogen | n/a | High weight of evidence | Sufficient human and animal data |

[a] Critical effect is neurological toxicity in progeny of exposed women, RfD calculated using a benchmark dose (10%).

There are many uncertainties associated with the health effects data analysis, due to an incomplete understanding of the toxicity of methylmercury. The sources of uncertainty include the following:

- The data serving as the basis for the methylmercury RfD were from a population ingesting contaminated seed grain. The nutritional status of this group may not be similar to that of U.S. populations. The exposure was for a short, albeit critical, period of time. It is likely that there is a range of response among individuals to methylmercury exposure. The selenium status of the exposed Iraqi population is not certain, nor is it established the extent to which selenium has an effect on mercury toxicity.

- There was no NOAEL (no-observable-adverse-effect level) for estimation of a threshold for all developmental endpoints. A benchmark was estimated using a Weibull model on grouped data. Use of an estimate other than the 95 percent lower limit on 10 percent response provides alternate estimates. Other modeling approaches using data which have not been grouped provide similar estimates. Benchmark doses, NOAELs, and LOAELs from other human studies provide support for the benchmark used in the RfD.

- Ingestion levels of methylmercury associated with measured mercury in hair were estimated based on pharmacokinetic parameters derived from evaluation of the extant literature. Use of other plausible values for these parameters results in (relatively small) changes in the exposure estimate.

- While there are data to show that the developing fetus is more susceptible to methylmercury toxicity than adults, there are not sufficient data to support calculation of a separate RfD for children (vs. adults).

To improve the risk assessment for methylmercury, U.S. EPA would need the following:

- Results from ongoing studies in human populations with measurable exposure to methylmercury, and new research on actual consumption patterns and estimated methylmercury exposure of the subpopulations of concern, with validation by analysis of hair samples from a representative sample of members of this subpopulation.

- Reproductive studies and analysis.

- Data on mode of action of methylmercury tumor induction.

- Validated physiologically-based pharmacokinetic models for mercury which include a fetal component.

Based on the extant data and knowledge of developing studies, the following outcome can be expected:

- Human populations exposed to sufficiently high levels of methylmercury either *in utero* or *post partum* will have increased incidence of neurotoxic effects.

## 7.3 MERCURY CONCENTRATIONS IN BIOTA

The *Mercury Study Report to Congress* documents many concentrations in animals and plants.[24] Concentrations in abiotic environmental components consist primarily of inorganic species. While these concentrations may be elevated in specific areas, fish concentrations are generally of highest concern when assessing risks posed by emitted mercury. The concern stems from the consumption of fish by humans and the form of mercury, methylmercury, which fish bioaccumulate. Methylmercury, which is the primary form of mercury found in fish tissue, is a human neurotoxin and is readily absorbed into the human body through the gastrointestinal tract. Fish methylmercury concentrations result from existing environmental concentrations of mercury (which may consist of mercury from anthropogenic and natural sources, as well as mercury which has been re-emitted from the oceans or soils) and deposition from the global reservoir (which includes mercury emitted by other countries).

Given the current scientific understanding of the environmental fate and transport of this element, it is not possible to quantify how much of the methylmercury in fish consumed by the U.S. population is contributed by U.S. emissions relative to other sources of mercury (such as natural sources and re-emissions from the global pool). As a result, it cannot be assumed that a change in total mercury emissions will be linearly related to any resulting change in methylmercury in fish, nor over what time period these changes would occur. This is an area of ongoing study.

## 7.4 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[70] 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[71] 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 and had 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. Measurement data near other types of anthropogenic sources are discussed in the 1997 EPA *Mercury Study Report to Congress*.[24]

## 7.5 MODEL FRAMEWORK

This section describes the models and modeling scenarios used to predict the environmental fate of mercury. Measured mercury concentrations in environmental media were used when available to parameterize these models. Human exposures to mercury were predicted based on modeling results.

### 7.5.1 Models Used

The extant measured mercury data alone were judged insufficient for a national assessment of mercury exposure for humans from utility units. Thus, the decision was made to model the mercury emissions. In this study, there were three major types of modeling efforts: (1) modeling of mercury atmospheric transport on a regional basis; (2) modeling of mercury atmospheric transport on a local scale (within 50 km of source); and (3) modeling of mercury fate in soils and water bodies into biota, as well as the resulting exposures to human fish consumers. The models used are described in Table 7-3.

### 7.5.2 Modeling of Long-Range Fate and Transport of Mercury

The goal of this analysis was to model the emission, transport, and fate of airborne mercury over the continental United States using the meteorologic data for the year of 1989 and the most current utility mercury emissions data. The results of the simulation were intended to be used to answer a number of fundamental questions. Probably the most general question was "How much mercury emitted by utility boilers is deposited back to United States soils and water bodies over a typical year?" It is known that year-to-year variations in accumulated precipitation and wind flow patterns affect the observed quantity of mercury deposited to the surface at any given location. Meteorological data for the year 1989 was used since most of the continental United States experienced near average weather conditions during that year. To estimate the quantity of mercury emitted by utilities that deposits in the United States, and specifically which geographic regions may be more highly impacted, information on chemical and physical forms of the mercury emissions was needed since these characteristics determine the rate and location of the wet and dry deposition processes for mercury.

The RELMAP model was used to predict the average annual atmospheric mercury concentration and the wet and dry deposition flux for each ½ degree longitude by ⅓ degree latitude grid cell (approximately 40 km square) in the continental United States. The emission, transport, and fate of airborne mercury over the continental United States was modeled using meteorological data for the year 1989. The utility emission data used were those presented in the *Mercury Study Report to Congress*.[20] Emission data are shown in Table 7-4.

The RELMAP model was originally developed to estimate concentrations of sulfur and sulfur compounds in the atmosphere and rainwater in the eastern United States. The primary modification of RELMAP was the handling of three species of mercury (elemental, divalent, and particulate) and carbon soot (or total carbon aerosol).[24] A complete description of the RELMAP mercury model is presented in the *Mercury Study Report to Congress*.[24] The results of the RELMAP modeling are shown in Figures 7-2 through 7-4.

Table 7-3. 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 U.S. due to all anthropocentric sources of mercury in the U.S. |
| ISC3 | Predicts annual average atmospheric concentrations and deposition fluxes within 50 km of mercury emission source |
| IEM-2M | Predicts environmental mercury concentrations based on air concentrations and deposition rates to watershed and water body. Predicts human exposure based on these predicted concentrations and human activity patterns. |

RELMAP = Regional Lagrangian Model of Air Pollution
IEM = Indirect exposure methodology
ISC = Industrial Source Complex

Table 7-4. Mercury Emissions Inventory Used in the RELMAP Modeling (Based on the 1994-95 Estimates)

| Mercury emission source type | Emissions (kg/yr) | Assumed speciation percentages | | |
|---|---|---|---|---|
| | | Hg(0) [a] | $Hg^{2+}$ [b] | $Hg_P$ [c] |
| Electric utility boilers (coal, oil and gas) | 46,183 | 50 | 30 | 20 |

[a] Hg(0) represents elemental mercury gas
[b] $Hg^{2+}$ represents divalent mercury gas
[c] $Hg_P$ represents particulate mercury

Figure 7-2. Total Modeled Mercury Deposits from Wet and Dry Deposition from Coal Utilities Based on 1994 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$




Figure 7-3. Total Modeled Mercury Deposits from Wet and Dry Deposition from Oil Utilities Based on 1994 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$

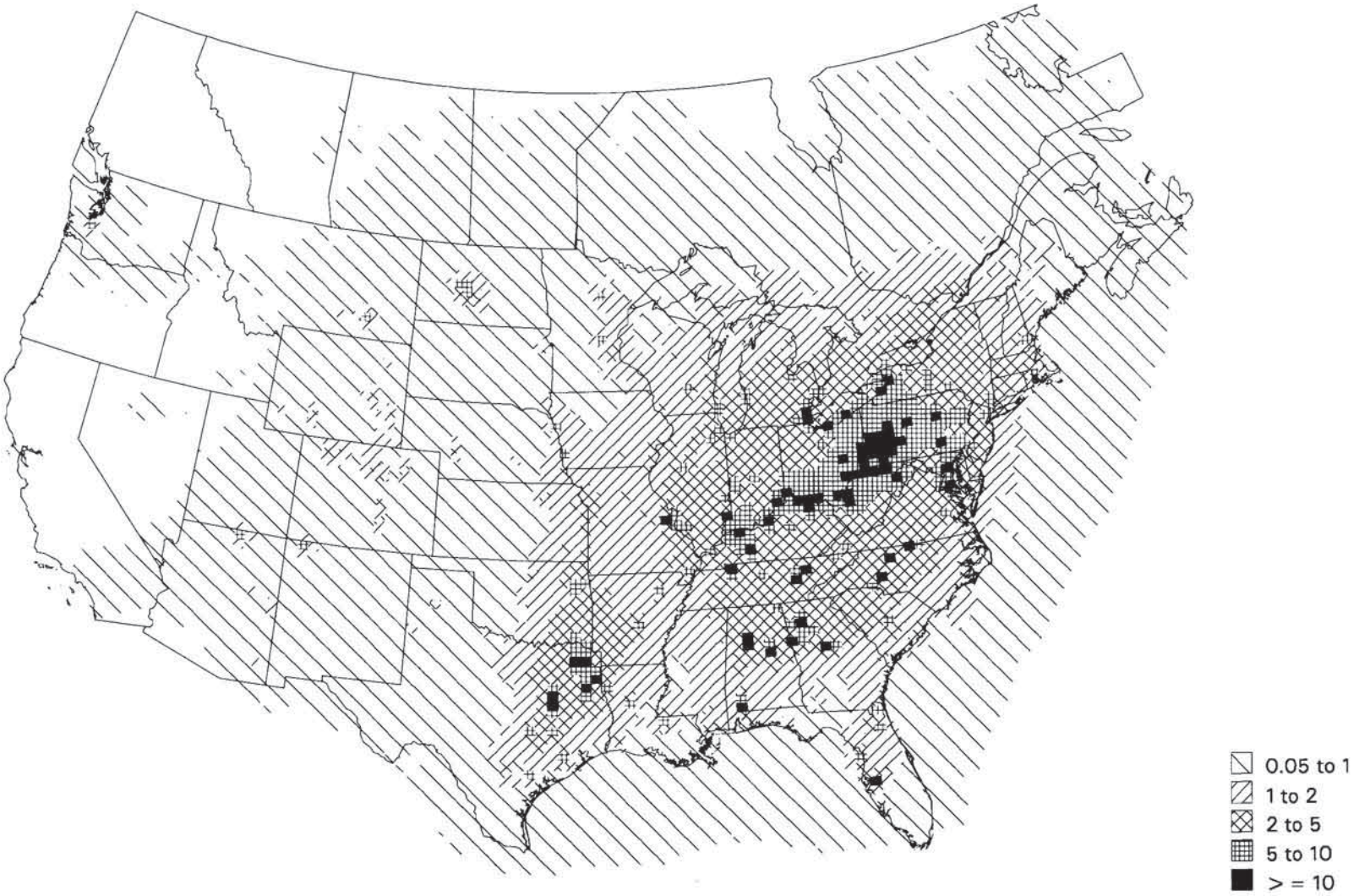



Figure 7-4. Total Modeled Mercury Deposits from Wet and Dry Deposition from Coal and Oil Utilities Based on 1994 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$

A computer simulation of long-range transport of mercury emissions from all United States sources conducted for the EPA's 1997 Mercury Study Report to Congress suggests that about one-third (~ 52 tons) of the 158 tpy of United States anthropogenic emissions are deposited, through wet and dry deposition, within the lower 48 States. The remaining two-thirds (~ 107 tons) is transported outside of United States borders where it diffuses into the global reservoir. In addition, the computer simulation suggests that another 35 tons of mercury from the global reservoir is deposited for a total deposition of roughly 87 tons. Although this type of modeling is uncertain, the simulation suggests that about three times as much mercury is being added to the global reservoir from United States sources as is being deposited from it. What is not uncertain is that additional emissions to air will contribute to levels in the global reservoir, and concomitant deposition to water bodies.

Long-range transport modeling conducted as part of this Utility Study predicts that approximately 30 percent (i.e., 15 tpy) of the utility mercury emissions deposit in the continental United States. The estimated annual deposition rates resulting from utility mercury emissions range from 0.5 to greater than 10 $\mu$g per square meter.

### 7.5.3 Modeling the Local Transport of Mercury in the Atmosphere

The program used to model the transport of the anthropogenic mercury within 50km of an emissions source was the Industrial Source Complex Version 3 (ISC3) gas deposition model obtained from the United States EPA's Support Center for Regulatory Air Models (SCRAM) website (the program is called GDISCDFT). This model has a gas dry deposition model that was applied in this study. The issues related to using this program to model emitted mercury in the local atmosphere are detailed in Volume III of the *Mercury Study Report to Congress*.[24]

The phase and oxidation state of emitted mercury is thought to be of critical importance in determining atmospheric fate. Only Hg(0) and $Hg^{+2}$ were considered in the air dispersion modeling. At the point of stack emission and during atmospheric transport, the contaminant is partitioned between two physical phases: vapor and particle-bound. It was assumed that 25 percent of the divalent emissions from an individual source would attach to particles in the plume; particle sizes were assumed to reflect ambient particle data.

7.5.3.1 Development and Description of Model Plants. Model plants representing four utility boilers were developed to represent a range of mercury emissions from this source. 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 risk assessment included mercury emission rates, mercury speciation, and mercury transport/deposition rates. Important model plant parameters included 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 (see Table 7-5). Emission estimates were assumed to represent typical emission levels emitted from existing sources.[20,24]

7.5.3.2. Hypothetical Locations of Model Plants. There are a variety of geographic aspects that can influence the dispersion of mercury emissions from a utility boiler. These aspects include factors that affect the environmental chemistry of a pollutant and the physics of plume dispersion. Environmental chemistry can include factors such as the amount of wet deposition in a given area. Factors affecting plume dispersion include terrain, wind direction and average wind speed.

Because wet deposition may be an important factor leading to mercury exposures, especially for the more soluble species emitted, the meteorology of a location was used as a selection criterion. Two different types of meteorology were deemed necessary to characterize the environmental fate and transport of mercury: an arid/semi-arid site and a humid site. The criterion specifically utilized was total yearly rainfall. (See Volume III of the *Mercury Study Report to Congress* for details.)[24]

Terrain features refer to the variability of the receptor height with respect to a local source. Two main types of terrain were used in the modeling: simple, and complex. Simple terrain is defined as a study area that is relatively level and well below stack top (rather, the effective stack height). Complex terrain refers to terrain that is not simple, such as source located in a valley or a source located near a hill. This included receptors that are above or below the top of the stack of the source. Complex terrain can affect concentrations, plume trajectory, and deposition. Due to the complicated nature of plume flow in complex terrain, it is probably not possible to predict impacts in complex terrain as accurately as for simple terrain. In view of the wide range of uncertainty inherent in accurately modeling the deposition of the mercury species considered, the impacts posed by complex terrain were not incorporated in the local scale analysis.

Two generic sites are considered: a humid site east of 90 degrees west longitude, and a more arid site west of 90 degrees west longitude (these are described in Volume III of the *Mercury Study Report to Congress*).[24] The primary differences between the two sites as parameterized were the assumed erosion characteristics for the watershed and the amount of dilution flow from the water body. The eastern site had generally steeper terrain in the watershed than the other site. A circular drainage lake with a diameter of 1.78 km and average depth of 5 m, with a 2 cm benthic sediment depth, was modeled at both sites. The watershed area was 37.3 $km^2$.

7.5.4 Modeling Mercury in a Watershed

Atmospheric mercury concentrations and deposition rates estimated from RELMAP and ISC3 drive the calculations of mercury in watershed soils and surface waters. The soil and water concentrations, in turn,

Table 7-5. Process Parameters for Model Plants

| Model plant | Plant size | Capacity (% of year) | Stack height (ft) | Stack diameter (ft) | Hg emission rate (kg/yr) | Speciation percent ($Hg(0)/Hg^{2+}/Hg^{P}$) | Exit velocity (m/sec) | Exit temperature (°F) |
|---|---|---|---|---|---|---|---|---|
| Large coal-fired utility boiler | 975 Megawatts | 65% | 732 | 27 | 230 | 50/30/20 | 31.1 | 273 |
| Medium coal-fired utility boiler | 375 Megawatts | 65% | 465 | 18 | 90 | 50/30/20 | 26.7 | 275 |
| Small coal-fired utility boiler | 100 Megawatts | 65% | 266 | 12 | 10 | 50/30/20 | 6.6 | 295 |
| Medium oil-fired utility boiler | 285 Megawatts | 65% | 290 | 14 | 2 | 50/30/20 | 20.7 | 322 |

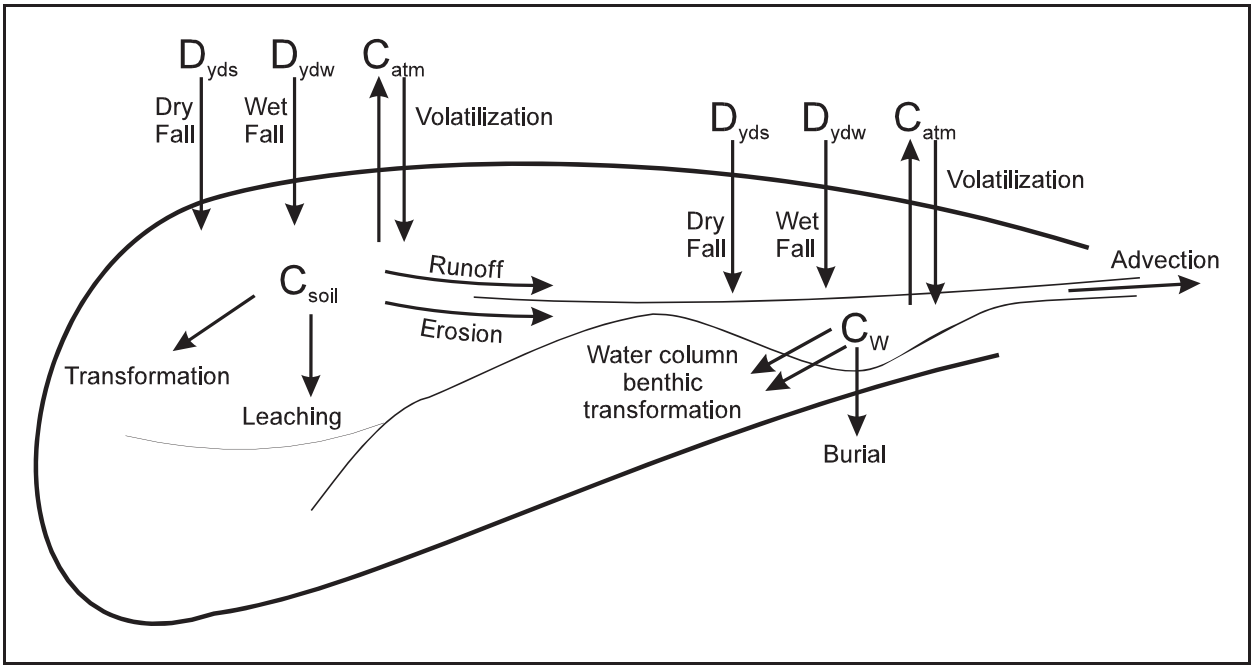
Hg(0) = elemental mercury;
$Hg^{2+}$ = divalent vapor phase mercury;
$Hg^{P}$ = particle-bound mercury

drive calculations of concentrations in the associated biota and fish, which humans are assumed to consume. The watershed model used for this report, Indirect Exposure Methodology Version 2M (IEM-2M), was adapted from the more general IEM-2 methodology[30,31] to handle mercury fate in soils and water bodies. It is described completely in the EPA *Mercury Study Report to Congress*, Volume III.[24]

7.5.4.1 Overview of the Watershed Model. IEM-2M simulates three chemical components: elemental mercury, Hg(0); divalent mercury, HgII; and methylmercury, MHg. In the previous version of IEM-2, these components were assumed to be in a fixed ratio with each other as specified by the fraction elemental ($f_1$) and fraction methyl ($f_3$). This version calculates the fractions in each component based on specified or calculated rate constants.

IEM-2M is composed of two integrated modules that simulate mercury fate using mass balance equations describing watershed soils and a shallow lake, as illustrated in Figures 7-5 and 7-6. The mass balances are performed for each mercury component, with internal transformation rates linking Hg(0), HgII, and MHg. Sources include wetfall and dryfall loadings of each component to watershed soils and to the water body. An additional source is diffusion of atmospheric Hg(0) vapor to watershed soils and the water body. Sinks include leaching of each component from watershed soils, burial of each component from lake sediments, volatilization of Hg(0) and MeHg from the soil and water column, and advection of each component out of the lake.

At the core of IEM-2M are nine differential equations describing the mass balance of each mercury component in the surficial soil layer, in the water column, and in the surficial benthic sediments. The equations are solved for a specified interval of time, and predicted concentrations output at fixed intervals. For each




| | | |
|---|---|---|
| $C_{soil}$ | total mercury concentration in upper soil | ng/g |
| $C_w$ | total mercury concentration in water body | ng/L |
| $C_{atm}$ | vapor phase mercury concentration in air | ng/$m^3$ |
| $D_{yds}$ | average dry deposition to watershed | $\mu$g/$m^2$-yr |
| $D_{yws}$ | average wet deposition to watershed | $\mu$g/$m^2$-yr |

Figure 7-5. Overview of the IEM-2M Watershed Modules

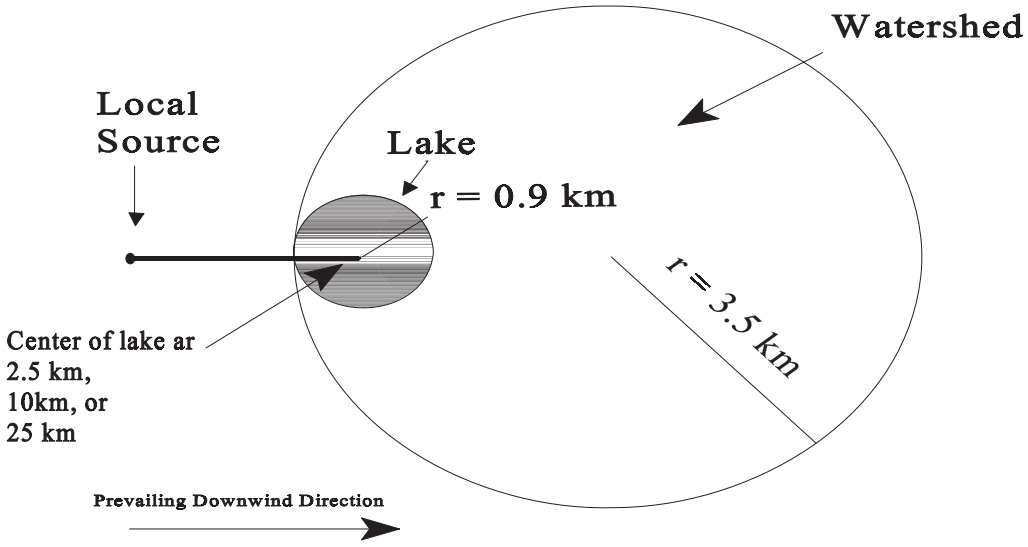



Figure 7-6. Configuration of Hypothetical Water Body and Watershed Relative to Local Source

calculational time step, IEM-2M first performs a terrestrial mass balance to obtain mercury concentrations in watershed soils. Soil concentrations are used along with vapor concentrations and deposition rates to calculate concentrations in various food plants. These are used, in turn, to calculate concentrations in animals. IEM-2M next performs an aquatic mass balance driven by direct atmospheric deposition along with runoff and erosion loads from watershed soils.

The nature of this methodology is basically steady with respect to time and homogeneous with respect to space. While it tracks the buildup of soil and water concentrations over the years given a steady depositional load and long-term average hydrological behavior, it does not respond to unsteady loading or meteorological events. There are limitations on the analysis and interpretations imposed by these simplifications. The model's calculations of average water body concentrations are less reliable for unsteady environments, such as streams, than for more steady environments, such as lakes.[24] The description includes a "benchmarking" exercise with an independent model, the Mercury Cycling Model.

Mhg concentrations in fish are derived from dissolved MHg water concentrations using bioaccumulation factors (BAF).[24] Methylmercury concentrations in fish were derived from predicted water column concentrations of dissolved methylmercury by using BAFs for trophic level 4 fish (Table 7-6). The BAFs selected for these calculations were estimated from existing field data. The BAF (dissolved methylmercury basis) for trophic level 4 fish is $1.6 \times 10^6$. Methylmercury was estimated to constitute 7.8 percent of the total dissolved mercury in the water column, and 65 percent of this was assumed to be freely dissolved. The potential variability around these predicted fish residue values is highlighted in Table 7-6, which shows percentile information for the BAF estimates.

There are several limitations to the modeling analyses. First, there is a lack of adequate mercury measurement data near the anthropogenic atmospheric mercury sources considered in this report. To assess how well the modeled data predict actual mercury concentrations in different environmental media at a variety of geographic locations requires a database against which to make these comparisons. The lack of such measured data preclude a comparison of the modeling results with measured data around these sources. These data include measured mercury deposition rates as well as measured concentrations in the atmosphere, soils, water bodies and biota. Substantial additional monitoring data would facilitate such comparison. Second, the IEM-2M has not been validated with site-specific data. The model was benchmarked against the independently-derived Regional Mercury Cycling Model (R-MCM), which itself has been calibrated to several Wisconsin lakes. When driven by the same atmospheric loading and solids concentrations, IEM-2M predictions of mercury concentrations compare well with those calculated by R-MCM for a set of Wisconsin lakes. Additional limitations are discussed in later sections below.

Table 7-6. Percentiles of the Methylmercury Bioaccumulation Factor

| Parameter | Percentile of distribution | | | | |
|---|---|---|---|---|---|
| | 5th | 25th | 50th | 75th | 95th |
| Trophic 4 BAF | $3.3x10^6$ | $5.0x10^6$ | $6.8x10^6$ | $9.2x10^6$ | $1.4x\ 10^7$ |

BAF = bioaccumulation factor

7.5.5 Exposure Modeling

7.5.5.1 Description of Hypothetical Human Exposure Scenarios. Human exposure to environmental mercury is the result of mercury concentrations at specific human exposure points (e.g., ingested fish). For each location, mercury exposure was estimated only for individuals representing several specific subpopulations that consumed the freshwater fish that inhabited one of the three local lakes. The individuals representing the subpopulations were defined to model both average and high-end exposures.

The fish ingestion pathway was the only source of methylmercury intake assessed. For this assessment, four human fish consumption scenarios were considered for the hypothetical lakes: (1) an adult subsistence fish consumer who was assumed to ingest large amounts of locally-caught fish; (2) a child of a subsistence local fish consumer; (3) a high-end recreational angler; and 4) an average local fish consumer. These consumption scenarios were thought to represent identified fish-consuming subpopulations in the United States.

Fish for human consumption from local water bodies can be derived from many sources including self-caught, gifts, and grocery and restaurant purchases. For the purposes of this study, all fish consumed were assumed to originate from the hypothetical lakes, which were considered to represent several small lakes that might be present in the type of hypothetical locations considered. No commercial distribution of locally caught fish was assumed; exposure to locally-caught fish was modeled for the fish-consuming subpopulations described above.

Fish consumption rates for the three fish-consuming subpopulations were derived from the Columbia River Inter-Tribal Fish Commission report[72] and the draft EPA *Exposure Factors Handbook*.[73] Other estimates of human fish consumption rates are reported in the *Exposure Factors Handbook*.[73] The estimates presented highlight the broad variability in consumption rates. The Columbia River Inter-Tribal Fish Commission report[72] estimated fish consumption rates for members of four tribes inhabiting the Columbia River Basin. The estimated fish consumption rates were based on interviews with 513 adult tribe members who lived on or near the reservation. The

participants had been selected from patient registrations lists provided by the Indian Health Service. Adults interviewed provided information on fish consumption for themselves and for 204 children under five years of age.

Fish consumption rates for tribal members are shown in Tables 7-7 and 7-8. The values used in this study are shown in Table 7-9. The values listed below reflect an annual average, but monthly variations were also reported. For example, the average daily consumption rate during the two highest intake months was 107.8 g/day, and the daily consumption rate during the two lowest consumption months was 30.7 g/day. Fish were consumed by over 90 percent of the surveyed population, with only 9 percent of the respondents reporting no fish consumption. The maximum daily consumption rate for fish reported by one member of this group was 972 g/day. Since most of the population consisted of fish consumers, utilization of per capita estimates was considered appropriate.

The *Exposure Factors Handbook*[73] describes many freshwater fish consumption studies. The mean daily consumption rates derived for recreational freshwater anglers from the compiled studies range from 5-17 g/day; the derived 95th percentile range was 8-25 g/day. The value of 30 g/day clearly exceeds the 95th percentile; this individual is a high-end consumer. The recommended mean intake for subsistence populations was 70 g/day, and the 95th percentile was 170 g/day. The value of 60 g/day which is used for the subsistence adult is lower than the recommended mean. The body weights used for the adult and child were 70 and 17 kg.[74]

## 7.6 RESULTS

Tables 7-10 through 7-13 present the results of the multipathway modeling analysis. The results are based on a model plant analysis and are for hypothetical scenarios. Therefore, the results do not apply to any specific utility plant and contain significant uncertainties.

In all cases, the average air concentrations are predicted to be dominated by the regional contribution of utilities rather than the single local source modeled. This is largely due to the high effective stack heights exhibited by the sources. The largest contribution of 35 percent is for the medium coal-fired utility boiler (MCUB) in the western site. The western site is predicted to have lower concentrations of mercury as a result of regional transport.

At both the eastern and western sites using the 50th percentile RELMAP results, the deposition rates, soil concentrations, and fish concentrations are usually dominated by the local coal-fired utility source within 10 km of the source. The small coal-fired utility boiler (SCUB) at the eastern site is the exception due to the higher deposition rate from regional sources. In the eastern site regional sources dominate the deposition rates, soil concentrations, and fish

Table 7-7. Fish Consumption Rates for Columbia River Tribes[72]

| Subpopulation | Mean daily fish consumption (g/day) |
|---|---|
| Total adult population, aged 18 years and older | 59 |
| Children, aged 5 years and younger | 20 |
| Adult females | 56 |
| Adult males | 63 |

Table 7-8. Daily Fish Consumption Rates Among Adults in the Columbia River Tribes[72]

| Percentile | g/day |
|---|---|
| 50th | 29-32 |
| 90th | 97-130 |
| 95th | 170 |
| 99th | 389 |

Table 7-9. Fish Consumption Rates Used in This Study

| Subpopulation | Fish consumption rate (g/day)[a] |
|---|---|
| Subsistence adult | 60 [a] |
| High-end child | 20 [a] |
| Recreational angler | 8 |
| High-end recreational angler | 30 [a] |

[a] Columbia River Inter-Tribal Commission, 1994.

Table 7-10. Model Results for Eastern Site, RELMAP 50th Percentile (utilities only)

| Facility | Distance | Air concentration (ng/m3) | %RELMAP | %ISC3 | Deposition (µg/m2/yr) | %RELMAP | %ISC3 | Soil concentration (ng/g) | %RELMAP | %ISC3 | Tier 4 fish concentration (ug/g) | %RELMAP | %ISC3 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Large coal-fired utility boiler | 2.5 km | 0.026 | 93% | 7% | 17.9 | 13% | 87% | 31 | 13% | 87% | 0.43 | 10% | 90% |
| | 10 km | 0.026 | 91% | 9% | 5.27 | 44% | 56% | 9.1 | 44% | 56% | 0.11 | 42% | 58% |
| | 25 km | 0.026 | 92% | 8% | 3.4 | 69% | 31% | 5.8 | 69% | 31% | 0.064 | 68% | 32% |
| Medium coal-fired utility boiler | 2.5 km | 0.027 | 89% | 11% | 9.12 | 26% | 74% | 16 | 26% | 74% | 0.21 | 21% | 79% |
| | 10 km | 0.028 | 88% | 12% | 4.17 | 56% | 44% | 7.2 | 56% | 44% | 0.081 | 54% | 46% |
| | 25 km | 0.027 | 90% | 10% | 3.19 | 73% | 27% | 5.5 | 73% | 27% | 0.06 | 73% | 27% |
| Small coal-fired utility boiler | 2.5 km | 0.028 | 87% | 13% | 3.94 | 59% | 41% | 6.8 | 59% | 41% | 0.08 | 55% | 45% |
| | 10 km | 0.027 | 91% | 9% | 2.93 | 80% | 20% | 5 | 80% | 20% | 0.056 | 78% | 22% |
| | 25 km | 0.025 | 96% | 4% | 2.54 | 92% | 8% | 4.4 | 92% | 8% | 0.048 | 92% | 8% |
| Medium oil-fired utility boiler | 2.5 km | 0.024 | 99% | 1% | 2.53 | 93% | 7% | 4.3 | 93% | 7% | 0.048 | 91% | 9% |
| | 10 km | 0.024 | 99% | 1% | 2.41 | 97% | 3% | 4.1 | 97% | 3% | 0.045 | 97% | 3% |
| | 25 km | 0.024 | 99% | 1% | 2.37 | 99% | 1% | 4.1 | 99% | 1% | 0.044 | 99% | 1% |

Table 7-11. Model Results for Western Site, RELMAP 50th Percentile (utilities only)

| Facility | Distance | Air concentration (ng/m3) | %RELMAP | %ISC3 | Deposition (ug/m2/yr) | %RELMAP | %ISC3 | Soil concentration (ng/g) | %RelMap | %ISC3 | Tier 4 fish concentration (ug/g) | %RELMAP | %ISC3 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Large coal-fired utility boiler | 2.5 km | 0.0061 | 87% | 13% | 3.9 | 10% | 90% | 6.4 | 10% | 90% | 0.12 | 8% | 92% |
| | 10 km | 0.0066 | 80% | 20% | 1.51 | 25% | 75% | 2.5 | 25% | 75% | 0.04 | 22% | 78% |
| | 25 km | 0.0073 | 73% | 27% | 1.4 | 27% | 73% | 2.3 | 27% | 73% | 0.035 | 25% | 75% |
| Medium coal-fired utility boiler | 2.5 km | 0.007 | 76% | 24% | 2.41 | 16% | 84% | 3.9 | 16% | 84% | 0.066 | 14% | 86% |
| | 10 km | 0.0081 | 65% | 35% | 1.75 | 22% | 78% | 2.9 | 22% | 78% | 0.047 | 19% | 81% |
| | 25 km | 0.0076 | 69% | 31% | 1.26 | 30% | 70% | 2.1 | 30% | 70% | 0.032 | 28% | 72% |
| Small coal-fired utility boiler | 2.5 km | 0.0077 | 69% | 31% | 1.44 | 26% | 74% | 2.4 | 26% | 74% | 0.04 | 22% | 78% |
| | 10 km | 0.0067 | 79% | 21% | 0.836 | 45% | 55% | 1.4 | 45% | 55% | 0.023 | 39% | 61% |
| | 25 km | 0.006 | 89% | 11% | 0.535 | 71% | 29% | 0.88 | 71% | 29% | 0.013 | 68% | 32% |
| Medium oil-fired utility boiler | 2.5 km | 0.0054 | 97% | 3% | 0.471 | 80% | 20% | 0.77 | 80% | 20% | 0.011 | 79% | 21% |
| | 10 km | 0.0054 | 97% | 3% | 0.439 | 86% | 14% | 0.72 | 86% | 14% | 0.011 | 83% | 17% |
| | 25 km | 0.0054 | 98% | 2% | 0.405 | 93% | 7% | 0.66 | 93% | 7% | 0.0097 | 93% | 7% |

Table 7-12. Predicted Exposure Results for Eastern Site, RELMAP 50th Percentile (utilities only)

| Facility | Distance | MHg Exposure (mg/kg/day) Subsistence Fisher | Child of Subsistence Fisher | Recreational Angler | Average recreational angler | %RELMAP | %ISC3 |
|---|---|---|---|---|---|---|---|
| Large coal-fired utility boiler | 2.5 km | 3.7E-04 | 5.1E-04 | 1.8E-04 | 4.9E-05 | 10% | 90% |
| | 10 km | 9.0E-05 | 1.2E-04 | 4.5E-05 | 1.2E-05 | 42% | 58% |
| | 25 km | 5.5E-05 | 7.6E-05 | 2.8E-05 | 7.4E-06 | 68% | 32% |
| Medium coal-fired utility boiler | 2.5 km | 1.8E-04 | 2.4E-04 | 8.9E-05 | 2.4E-05 | 21% | 79% |
| | 10 km | 7.0E-05 | 9.6E-05 | 3.5E-05 | 9.3E-06 | 54% | 46% |
| | 25 km | 5.2E-05 | 7.1E-05 | 2.6E-05 | 6.9E-06 | 73% | 27% |
| Small coal-fired utility boiler | 2.5 km | 6.9E-05 | 9.4E-05 | 3.4E-05 | 9.2E-06 | 55% | 45% |
| | 10 km | 4.8E-05 | 6.6E-05 | 2.4E-05 | 6.4E-06 | 78% | 22% |
| | 25 km | 4.1E-05 | 5.6E-05 | 2.0E-05 | 5.5E-06 | 92% | 8% |
| Medium oil-fired utility boiler | 2.5 km | 4.1E-05 | 5.7E-05 | 2.1E-05 | 5.5E-06 | 91% | 9% |
| | 10 km | 3.9E-05 | 5.3E-05 | 1.9E-05 | 5.2E-06 | 97% | 3% |
| | 25 km | 3.8E-05 | 5.2E-05 | 1.9E-05 | 5.1E-06 | 99% | 1% |

Table 7-13. Predicted Exposure Results for Western Site, RELMAP 50th Percentile (utilities only)

| Facility | Distance | MHg Exposure (mg/kg/day) Subsistence fisher | Child of subsistence fisher | Recreational angler | Average recreational angler | %RELMAP | %ISC3 |
|---|---|---|---|---|---|---|---|
| Large coal-fired utility boiler | 2.5 km | 1.0E-04 | 1.4E-04 | 5.1E-05 | 1.4E-05 | 8% | 92% |
| | 10 km | 3.5E-05 | 4.8E-05 | 1.7E-05 | 4.6E-06 | 22% | 78% |
| | 25 km | 3.0E-05 | 4.2E-05 | 1.5E-05 | 4.0E-06 | 25% | 75% |
| Medium coal-fired utility boiler | 2.5 km | 5.7E-05 | 7.8E-05 | 2.8E-05 | 7.6E-06 | 14% | 86% |
| | 10 km | 4.0E-05 | 5.5E-05 | 2.0E-05 | 5.4E-06 | 19% | 81% |
| | 25 km | 2.8E-05 | 3.8E-05 | 1.4E-05 | 3.7E-06 | 28% | 72% |
| Small coal-fired utility boiler | 2.5 km | 3.5E-05 | 4.7E-05 | 1.7E-05 | 4.6E-06 | 22% | 78% |
| | 10 km | 2.0E-05 | 2.7E-05 | 9.9E-06 | 2.6E-06 | 39% | 61% |
| | 25 km | 1.1E-05 | 1.6E-05 | 5.7E-06 | 1.5E-06 | 68% | 32% |
| Medium oil-fired utility boiler | 2.5 km | 9.7E-06 | 1.3E-05 | 4.9E-06 | 1.3E-06 | 79% | 21% |
| | 10 km | 9.3E-06 | 1.3E-05 | 4.6E-06 | 1.2E-06 | 83% | 17% |
| | 25 km | 8.3E-06 | 1.1E-05 | 4.2E-06 | 1.1E-06 | 93% | 7% |

concentrations at 25 km from all four sources considered. In the western site at 25 km the local coal combustion source emissions still dominate at 25 km except for the SCUB. The deposition rates at both sites are dominated by the regional sources when compared to the medium oil-fired utility model plant.

The contribution of the local source is identical for the deposition and soil concentrations, but this is not true for the fish concentration. This is because the surface water receives input of mercury from both direct deposition and from erosion/runoff from the watershed. The water body is assumed to lie at the end of the watershed closest to the facility, and so the contribution of the local source to the deposition rate to the water body is generally larger than that for the watershed. This results in a slightly higher contribution of the local source to the predicted fish concentrations.

The multipathway exposure modeling analysis presented in this chapter contains substantial uncertainties and is based on model plants and hypothetical scenarios. Therefore, the results do not apply to any existing utility plant. The analysis and results are useful for gaining a better qualitative understanding of the potential environmental fate of mercury emissions from a model utility plant. However, the quantitative results are uncertain. Further research and analyses are needed to gain a more complete understanding of the mercury exposures due to utility emissions.

Based on the model plant, multipathway exposure modeling analysis of hypothetical scenarios, the daily average methylmercury exposure of the average hypothetical recreational angler (8 g fish consumed per day) is not predicted to exceed the RfD of $1 \times 10^{-4}$ mg/kg/day under any combination of source, site, and distance. The daily average methylmercury exposure of the high-end hypothetical recreational angler (30 g fish consumed per day) is predicted to exceed the RfD in the eastern site at 2.5 km from the large coal-fired utility boiler (LCUB). The daily average methylmercury exposure of the hypothetical subsistence angler (60 g fish consumed per day) is predicted to exceed the RfD in the eastern site at 2.5 km from the LCUB and at 2.5 km from the MCUB. The daily average methylmercury exposure of the hypothetical subsistence angler is predicted to exceed the RfD in the western site at 2.5 km from the LCUB. Fish consumption by children is predicted to exceed the RfD for several hypothetical general cases: at 2.5 km from the LCUB at either site and at 2.5 km from the MCUB at the eastern site. Background exposures were not considered in this analysis. If background exposures due to other anthropogenic and natural sources were considered, this would obviously result in higher predicted exposures. Total exposures and background exposures are discussed and analyzed in the 1997 EPA *Mercury Study Report to Congress*.[24,26,69]

## 7.7 CONCLUSIONS

Long-Range Transport Analysis

Based on modeling analysis of the wet and dry deposition of utility air emissions of all forms of mercury within the continental United States, 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 United States and in numerous isolated locations where relatively large coal-fired utilities are located.

Areas Predicted to be 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 United States, 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 three principal factors that contribute to these modeled deposition patterns are:

- the emission source locations,
- the amount of divalent and particulate mercury emitted or formed in the atmosphere, and
- climate and meteorology.

A facility located in a humid climate is predicted to have a higher annual rate of mercury deposition than a facility located in an arid climate. The critical variables within the model are:

- the estimated washout ratios of elemental and divalent mercury, and
- the annual amount of precipitation.

Precipitation is important because it removes various forms of mercury from the atmosphere and deposits them to the surface of the earth.

The chemical form of emitted mercury is a critical factor in its fate, transport, and toxicity in the environment. The form distributions, or speciation factors, define the estimated fraction of mercury emitted as elemental mercury (Hg0), divalent mercury ($Hg^{2+}$), or mercury associated with particulates (Hgp). A wide variety of alternate speciation scenarios have been investigated to measure the sensitivity of the RELMAP results to this uncertainty.[74,75] The results show that the total simulated wet and dry deposition of mercury to the continental United States is strongly and positively correlated to the fraction of mercury emitted as $Hg^{2+}$ and Hgp for all major source types.

The differences between the results for the eastern and western sites are due primarily to the differences in the frequency and intensity of precipitation. At the eastern site, precipitation occurs about 12 percent of the year, with about 5 percent of this precipitation of moderate intensity (0.11 to 0.30 inches per hour). By comparison, at the western site, precipitation occurs about 3 percent of the year, with about 2 percent of the precipitation of moderate intensity.

Assessment of Watershed Fate

The atmospheric mercury concentrations and deposition rates estimated using the RELMAP and ISC3 were then used as inputs in the watershed model, IEM-2M, to derive calculations of mercury in watershed soils and surface waters. The soil and water concentrations, in turn, drive calculations of concentrations in the associated biota and fish, which humans are assumed to consume.

IEM-2M Model Sensitivity

For a specific atmospheric deposition rate, mercury concentrations in watersheds and water bodies can vary significantly. Several intrinsic and extrinsic watershed and water body characteristics influence the mercury concentrations in soil, water, and fish. These should cause significant variability in mercury concentrations between regions and among individual lakes within a region.

Mercury concentrations in watershed soils are strongly influenced by atmospheric loading and soil loss processes. The influence of plant canopy and roots in mediating both the loading to the soil and the loss from the soil is not well characterized at present, although published studies indicate its potential importance. Reduction of Hg(II) in the upper soil layer appears to control the volatile loss of mercury, and variations in this reaction can cause significant variations in soil mercury levels. The factors controlling mercury reduction are not well characterized at present. Soil erosion from a watershed can vary more than 3 orders of magnitude depending on

rainfall patterns, soil type, topography, and plant cover. High levels of soil erosion should significantly diminish soil mercury concentrations. Runoff and leaching are not expected to affect soil mercury concentrations significantly.

Total mercury concentrations in a water body are strongly influenced by atmospheric loading and, for drainage lakes, by watershed loading. Variations in watershed size and erosion rates can cause significant variability in lake mercury levels. Hydraulic residence time, the water body volume divided by total flow, affects the maximum possible level of total water column mercury for a given loading rate. Parameters controlling mercury loss through volatilization and net settling can also cause significant variations among lakes. Mercury loss through settling is affected by *in-situ* productivity, by the supply of solids from the watershed, and by the solids-water partition coefficient. DOC concentrations can significantly affect partitioning, and thus overall mercury levels. Mercury loss through volatilization is controlled by the reduction rate, which is a function of sunlight and water clarity. Reduction may also be controlled by pH, with lower values inhibiting this reaction and leading to higher total mercury levels.

Fish mercury levels are strongly influenced by the same factors that control total mercury levels. In addition, fish concentrations are sensitive to methylation and demethylation in the water column and sediments. A set of water body characteristics appear to affect these reactions, including DOC, sediment TOC, sunlight, and water clarity. Variations in these properties can cause significant variations in fish concentrations among lakes. Other factors not examined here, such as anoxia and sulfate concentrations, can stimulate methylation and lead to elevated fish concentrations. Fish mercury levels are sensitive to factors that promote methylmercury mobility from the sediments to the water column; these factors include sediment DOC and sediment-pore water partition coefficients.

Limitations of the Local Scale and Watershed Analyses

There are limitations associated with the fate and transport analyses. These have to do to a large degree with the current state-of-the-science concerning mercury fate and transport in the terrestrial and aquatic environments and variability between waterbodies. A few important limitations were discussed previously. Additional important limitations are discussed below.

- There is a lack of information characterizing the movement of mercury from watershed soils to water bodies and the rates at which mercury converts from one chemical species to another. There appears to be a great deal of variability in these factors among watersheds.

- There are not conclusive data on the amount of and rates of mercury methylation in different types of water bodies. In addition, there is a lack of data on the transfer of mercury between environmental compartments and biologic compartments; for example, the link between the amount of mercury in the water body and the levels in fish appears to vary from water body to water body.

Conclusions Regarding Mercury Fate and Transport in the Environment

The uncertainty inherent in the modeled estimates in this study arises from many individual assumptions present within the three models. Because of these uncertainties, EPA interpreted the model results qualitatively rather than quantitatively as follows.

The analyses conducted for this study as well as for the EPA's Mercury Study and available scientific knowledge indicate that human activities today are adding to the mercury reservoirs that already exist in land, water, and air, both naturally and as a result of previous human activities.

The analysis of mercury fate and transport conducted for this study, in conjunction with available scientific knowledge, supports a plausible link between mercury emissions from utility combustion sources and mercury concentrations in air, soil, water, and sediments. The critical variables contributing to this linkage are:

- the species of mercury that are emitted from the sources, with Hg0 mostly contributing to concentrations in ambient air and $Hg^{2+}$ mostly contributing to concentrations in soil, water and sediments;
- the overall amount of mercury emitted from a combustion source;
- the watershed soil loss rates, including reduction and erosion;
- the water body loss rates, including outflow, reduction, and settling; and
- the climate conditions.

In addition, this study also supports a plausible link between mercury emissions from utility combustion sources and methylmercury concentrations in freshwater fish. The critical variables contributing to this linkage are:

- the species of mercury that are emitted, with emitted divalent mercury mostly depositing into local watershed areas and, to a lesser extent the atmospheric conversion of

elemental mercury to divalent species which are deposited over greater distances;

- the overall amount of mercury emitted from a source;
- the watershed soil loss rates, including reduction and erosion;
- the water body loss rates, including outflow, reduction, and settling;
- the extent of mercury methylation in the water body;
- the extent of food web bioaccumulation in the water body; and
- the climate conditions.

From the analysis of deposition and on a comparative basis, the deposition of $Hg^{2+}$ close to an emission source is greater for receptors in elevated terrain (i.e., terrain above the elevation of the stack base) than from receptors located in flat terrain (i.e., terrain below the elevation of the stack base). The critical variables are parameters that influence the plume height, primarily the stack height and stack exit gas velocity.

On a national scale, an apportionment between sources of mercury and mercury in environmental media and biota cannot be described in quantitative terms with the current scientific understanding of the environmental fate and transport of this pollutant.

Human Exposure

The only exposure route considered was the consumption of freshwater fish. Consumption of fish is the dominant pathway of exposure to methylmercury for fish-consuming humans. There is a great deal of variability among individuals in these populations with respect to fish consumption rates. As a result, there is a great deal of variability in exposure to methylmercury in these populations. While EPA interprets these models qualitatively, some freshwater fish-consuming individuals are predicted to exceed the RfD as a result of mercury emissions from the sources considered. Measuring methylmercury concentrations in fish from these waters and more direct measures of exposure (e.g., hair or blood data) to humans consuming fish around these sources should be a research priority.

It is important to note that the utility contribution is only one component of the total amount of methylmercury in fish. Other anthropogenic sources, natural sources and the existing background are expected to influence fish methylmercury levels.

## 7.8 DISCUSSION OF POTENTIAL CONCERNS OF MERCURY EMISSIONS FROM UTILITIES

Mercury is considered the highest priority for multipathway analyses because it is an environmentally persistent, toxic element. Mercury is deposited to soil and terrestrial vegetation through terrestrial exposure pathways, but at levels that do not result in human exposures likely to be detrimental to health. However, in its methylated form mercury bioaccumulates in the food web (especially the aquatic food web). Modeling results suggest that most of the mercury emitted to the atmosphere is deposited more than 50 km away from the source, especially sources that have tall stacks. As stated above, the modeling assessment from the Mercury Study, in conjunction with available scientific knowledge, supports a plausible link between anthropogenic mercury emissions and mercury found in freshwater fish. Additional emissions to air will contribute to levels in the global reservoir and deposition to water bodies. As a result, mercury emissions from utility units may add to the existing environmental burden.

At this time, the available information, on balance, indicates that utility mercury emissions are of sufficient potential concern for public health to merit further research and monitoring. The EPA recognizes that there are substantial uncertainties that make it difficult to quantify the magnitude of the risks due to utility mercury emissions, and that further research and/or evaluation would be needed to reduce these uncertainties. Remaining questions include the following: (1) what is the quantitative relationship between a change in United States mercury emissions and the resulting change in methylmercury levels in fish; (2) what are the actual consumption patterns and estimated methylmercury exposures of the subpopulations of concern; (3) what are the actual mercury levels in a statistically valid and representative sample of the U.S. population and susceptible subpopulations; (4) what exposure levels are likely to result in adverse health effects; (5) what affects the formation of methylmercury in waterbodies and its bioaccumulation in fish; (6) how much mercury is emitted from natural sources and past anthropogenic sources; and (7) how much mercury is removed during coal cleaning and other ongoing practices for pollution control. New data that could reduce some of the uncertainties are likely to become available in the next several years, and EPA plans to review and consider these data, as appropriate, in future decisions.

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# 8.0 QUALITATIVE MULTIPATHWAY ASSESSMENT FOR LEAD AND CADMIUM

## 8.1 BACKGROUND

A multipathway exposure analysis, which is 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 priority HAPs (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. However, multipathway assessments were performed for radionuclides, mercury, arsenic, and dioxins. For the other two HAPs, lead and cadmium, only qualitative assessments of the potential concerns to human health from noninhalation exposure were performed.

The completion of quantitative assessments of inhalation exposures for all HAPs and of multipathway exposures for only radionuclides, mercury, arsenic, and dioxins does not reflect 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 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 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 a potentially more significant route of exposure than inhalation exposure. The mercury assessment suggests that there is a need for further analysis of noninhalation exposures.

Efforts are underway to collect the 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 about noninhalation exposure to lead and cadmium is presented in the following sections.

## 8.2 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 the oral and inhalation routes. For these reasons,

noninhalation exposure to lead emissions from utilities are a potential concern.

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 $7.2 \times 10^3$ metric tons per year of lead were emitted into the atmosphere from anthropogenic point and nonpoint sources during 1989.[1] 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 ($2.3 \times 10^3$ metric tons/year), solid waste management ($2.3 \times 10^3$ metric tons/year), transportation ($2.2 \times 10^3$ metric tons/year), and fuel combustion ($0.5 \times 10^3$ metric tons/year).[a]

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.[2] There is evidence 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 groundwater or surface water is not likely to occur. With the exception of highly acidic environmental conditions, leaching of lead into groundwater 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 with low water solubilities that 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.

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[a] Industrial processes include nonferrous smelters, battery plants, and chemical plants.

However, the bioavailability of lead to plants from soil is limited due to the strong absorption of lead to soil.

The highest background levels of lead are found in soils (<10 to 30 $\mu$g/g) and in sediments (the average concentration of lead in river sediments is 20,000 $\mu$g/g).[2] In 1988, the average ambient air concentration for 139 sites monitored by the National Air Surveillance Network (NASN) was determined to be 0.085 $\mu$g/m$^3$.[1] This value is well below the NAAQS of 1.5 $\mu$g/m$^3$.[2]

Concentrations of lead found in foods are given in Table 8-1. 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 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 8-1, the average adult dietary intake of lead for the years 1980-82 was estimated to be 56.5 $\mu$g/day.[1] However, recent data (1992) indicate that average dietary intakes have decreased significantly over the past decade to approximately 2-4 $\mu$g/day.[4] Presumably, this is at least partially due to the phasing out of lead in gasoline over the past two decades. 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-based paints, dust, and chips are also potential avenues of significant exposures for subpopulations (e.g., children) that may ingest these items.

Lead emissions from utilities do not contribute substantially to the total amount of lead released annually from 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. Air concentrations and deposition rates for lead emissions were estimated using the RELMAP model. The methods and model are described in detail in chapter 6 (section 6.6). Figures 8-1 through 8-6 show the results of the lead RELMAP modeling.

## 8.3 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 x $10^{-3}$ 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

Table 8-1. Concentration of Lead in Various Food Products[1,2]

| 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 |

is 5 x $10^{-4}$ 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, 1.7 ton/yr, and 0.054 ton/yr, respectively. To put these estimates into perspective, a recent study by the EPA estimates that about 233 tpy were emitted in the United States by anthropogenic sources in 1990.[5] Therefore, utilities are estimated to emit slightly more than 1 percent of anthropogenic cadmium emissions in the United States.

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 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; as a result, it 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

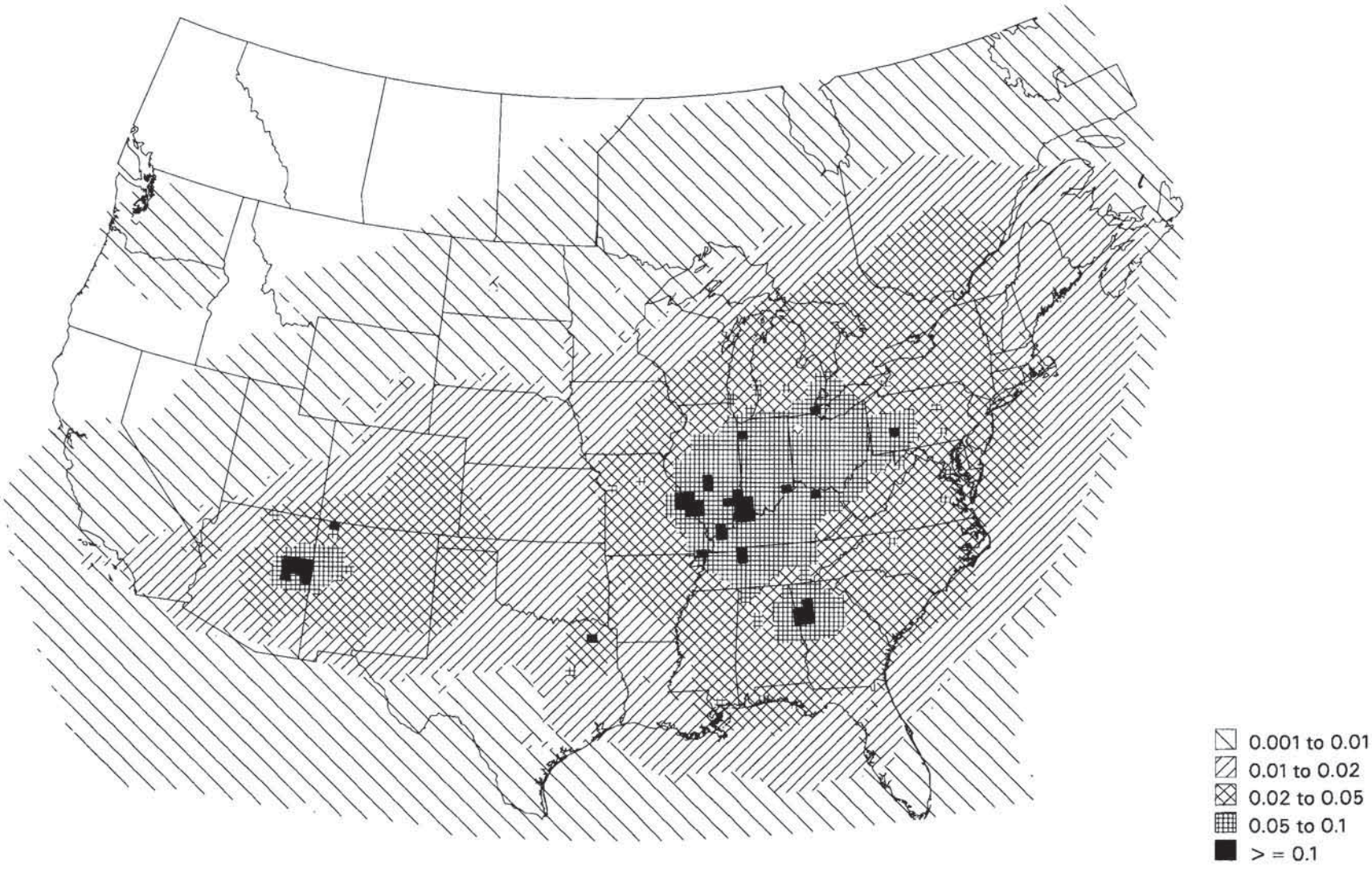



Figure 8-1. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Lead from Coal Utilities: Predicted Air Concentration of Lead, Units: ng/m$^3$



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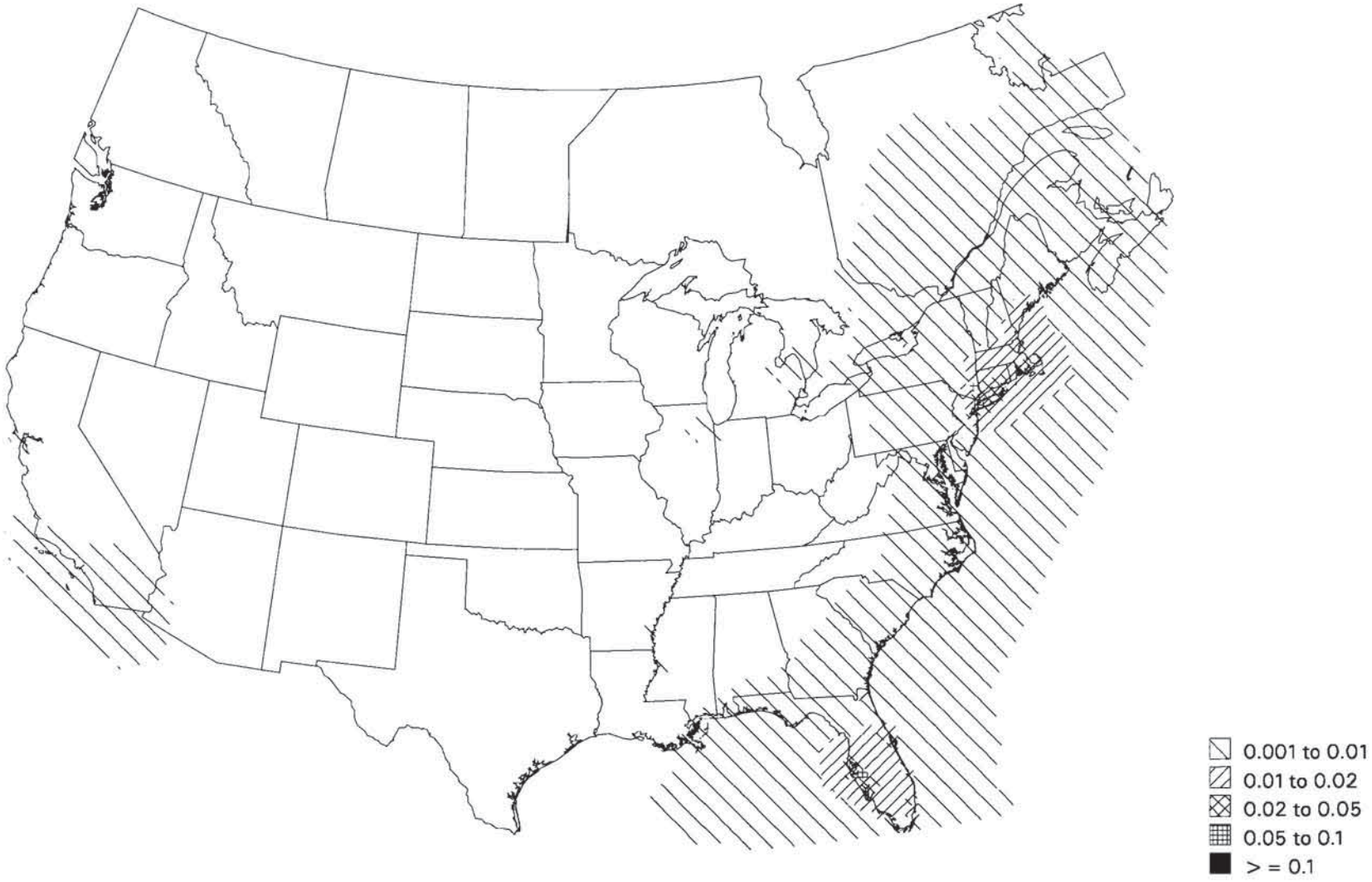



Figure 8-2. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Lead from Oil Utilities: Predicted Air Concentration of Lead, Units: $ng/m^3$



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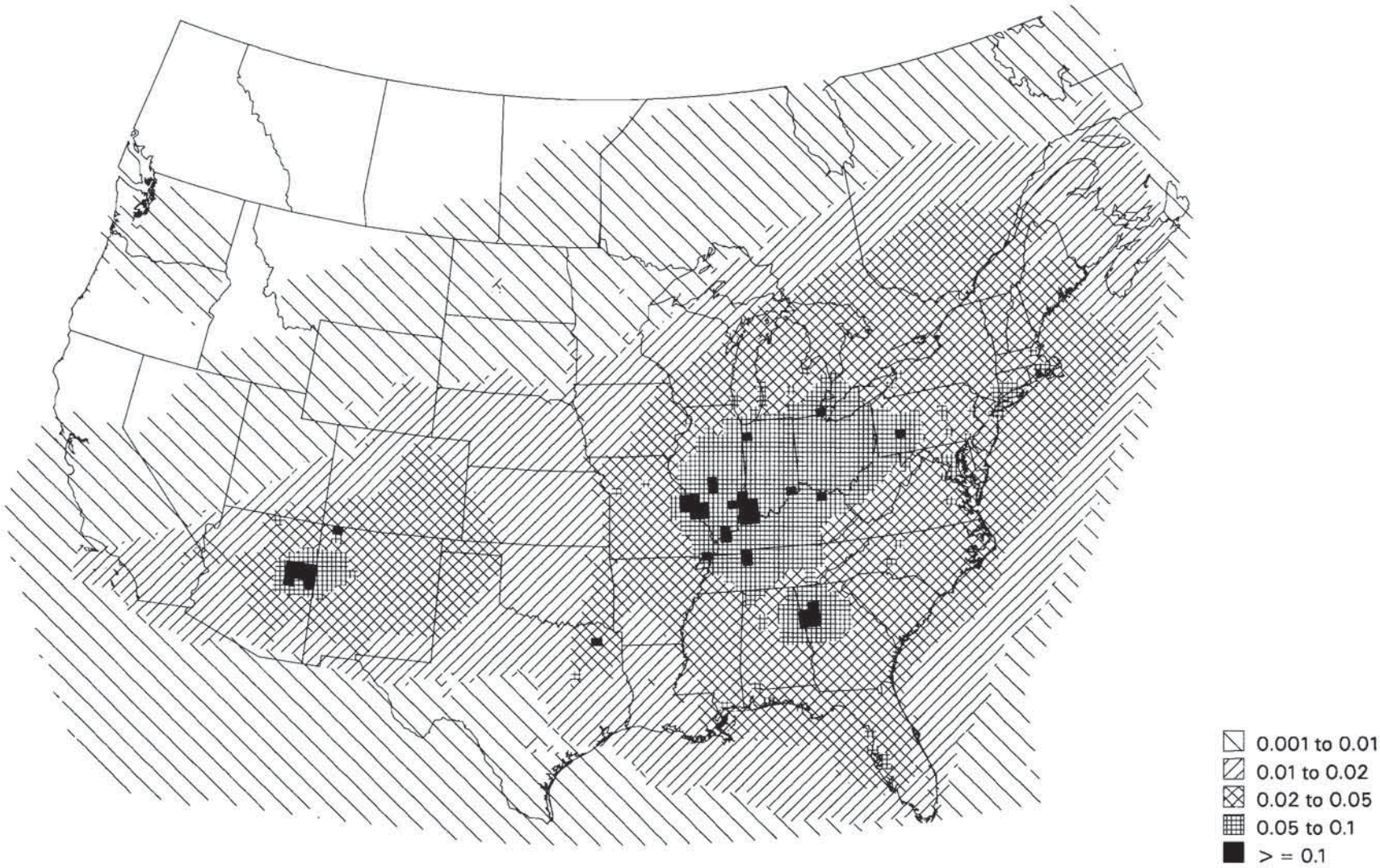



Figure 8-3. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Lead from Coal and Oil Utilities: Predicted Air Concentration of Lead, Units: $ng/m^3$



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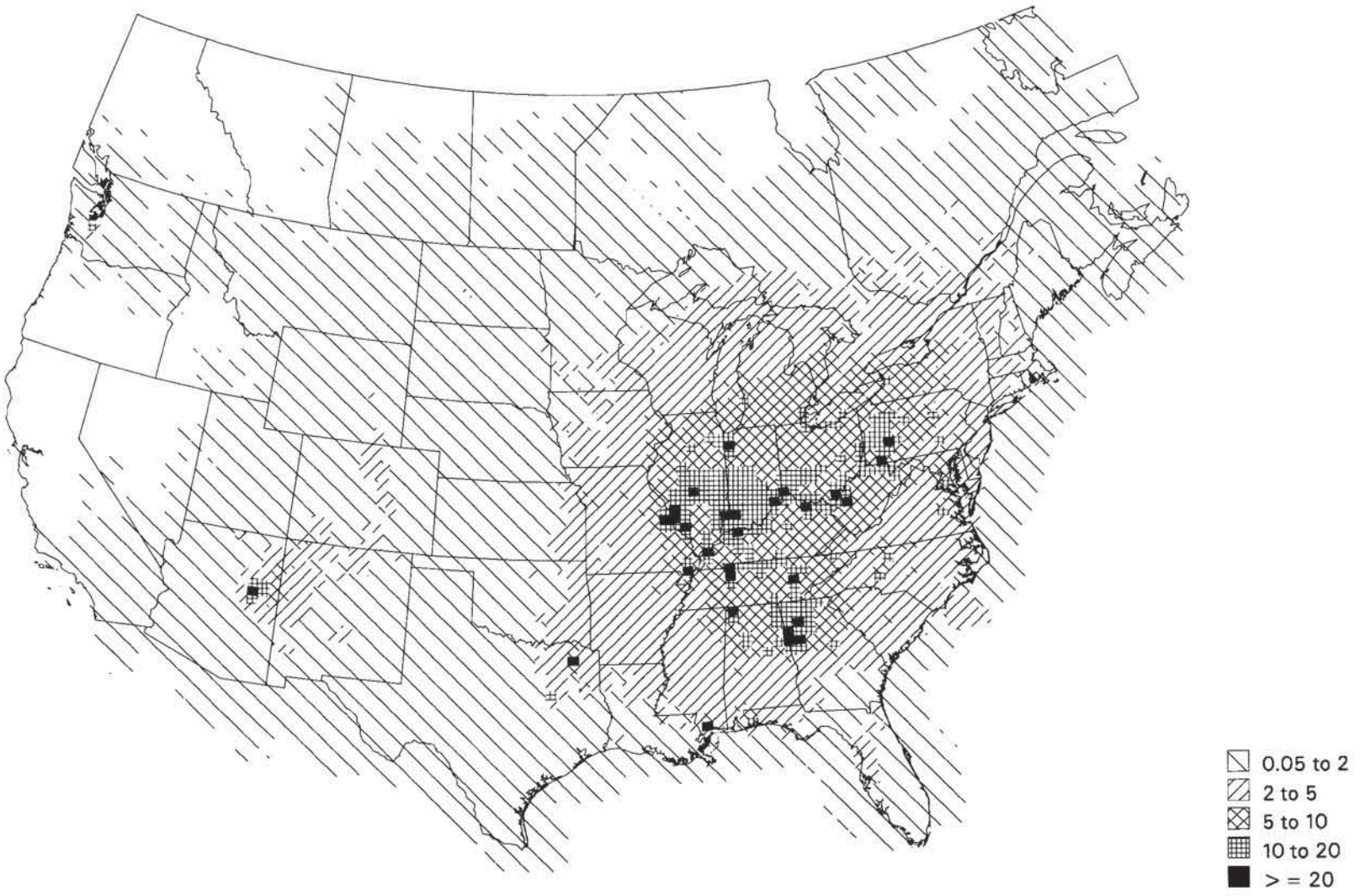



Figure 8-4. Predicted Lead Wet and Dry Deposition from Coal Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$



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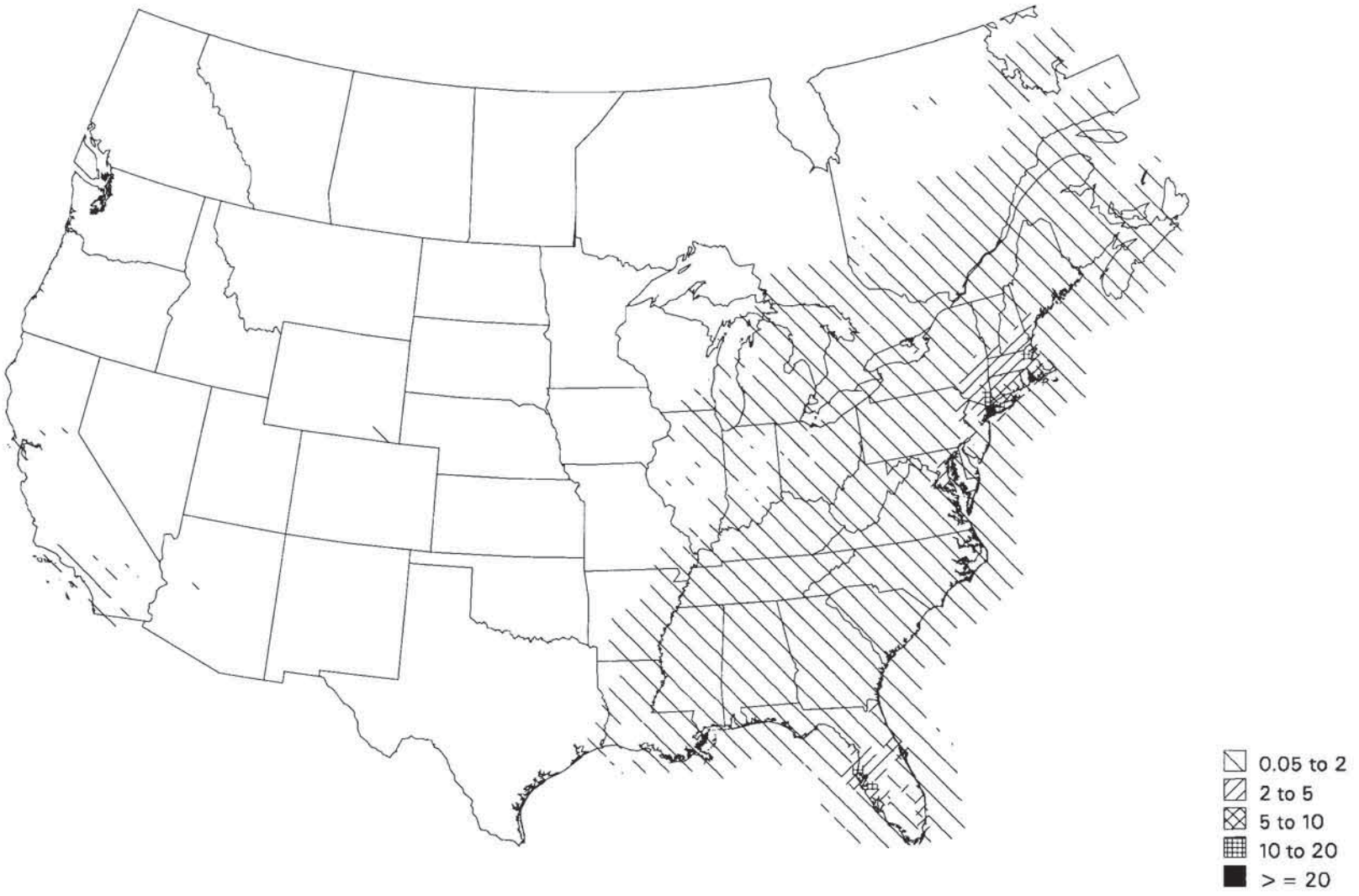



Figure 8-5. Predicted Lead Wet and Dry Deposition from Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$



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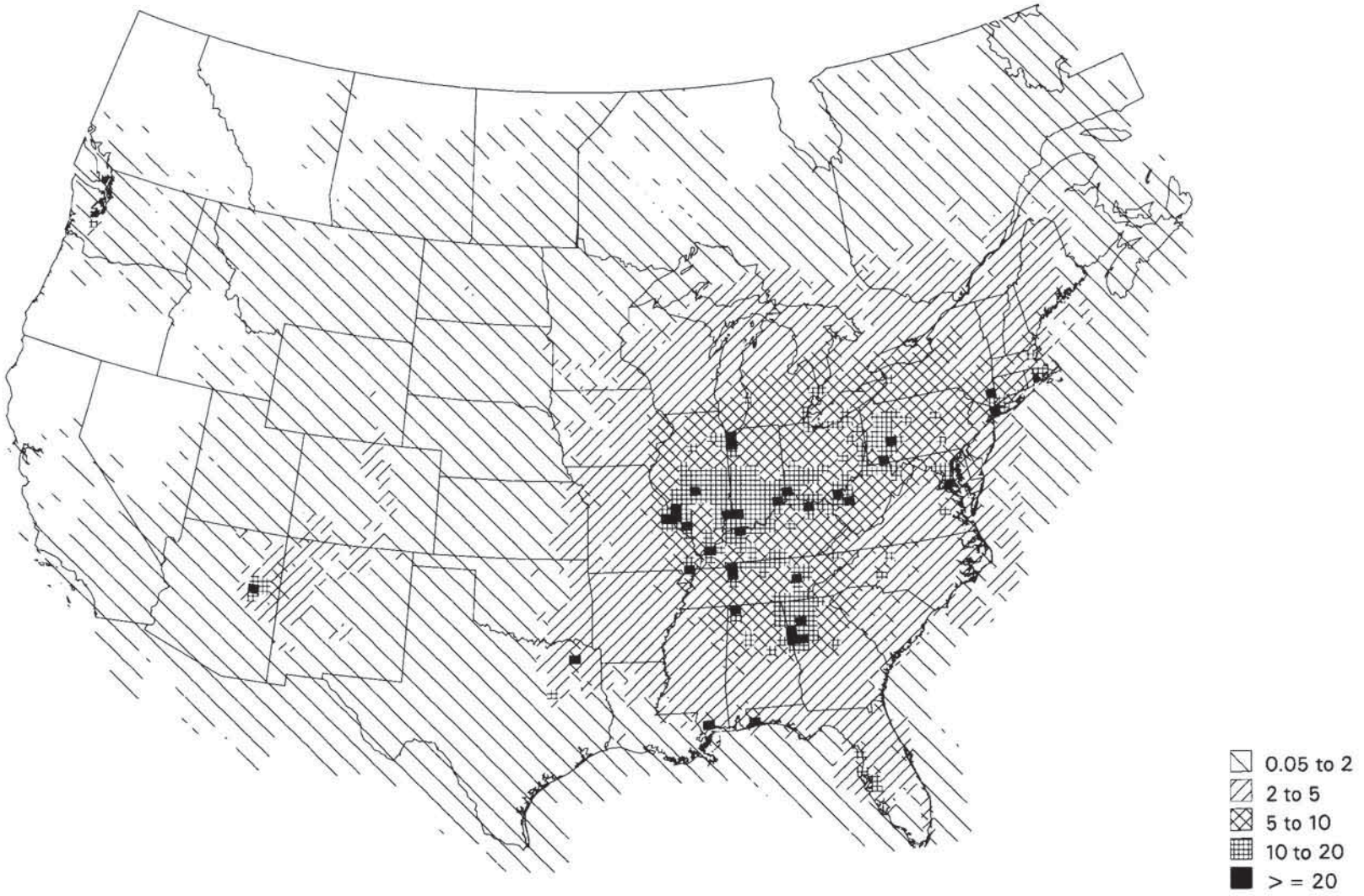



Figure 8-6. Predicted Lead Wet and Dry Deposition from Coal and Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$



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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 that cadmium bioaccumulates in all levels of the food chain.[6] Table 8-2 presents concentrations of cadmium in various foods.

The highest background levels of cadmium are found in soils.[6] 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 $1 \times 10^{-6}$ mg/m$^3$ in remote areas to $4 \times 10^{-5}$ in urban areas. Concentrations of cadmium in surface water, groundwater, and drinking water are typically less than 1 $\mu$g/L. In a study conducted in 27 U.S. cities, 12 food groups were tested and cadmium was detected in nearly all samples. As shown in Table 8-2, 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 at levels up to thousands of times higher than the levels of cadmium found in the water. In the United States, the adult intake of cadmium attributable to diet is estimated to be approximately 30 $\mu$g/d. Assuming a gastrointestinal absorption of 5 to 10 percent, the amount of cadmium absorbed from diet is approximately 1 to 3 $\mu$g/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 $\mu$g/cigarette. A cadmium exposure and absorption level of 1 to 3 $\mu$g/d can result from smoking one pack of cigarettes per day. Based on these data, the ATSDR states that, for the general nonsmoking population, the highest levels of exposure to cadmium are most likely to occur through consumption of food.[6] 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 was undertaken to estimate the steady-state bovine milk biotransfer factors (i.e., the rate at which the compounds are

Table 8-2. Concentration of Cadmium in Various Food Products[6]

| 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 |

transferred to milk) for six metals: arsenic, cadmium, chromium, lead, mercury, and nickel.[3] 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. The bioconcentration factor estimated for cadmium is $1.3 \times 10^{-6}$ L/kg. To put this value into perspective, the author points out the estimated biotransfer factor of TCDD is $2.6 \times 10^{-2}$ 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 from 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. Deposition rates were estimated for cadmium using the RELMAP model. The methods and model are described in detail in chapter 6 (section 6.6). Figures 8-7 through 8-9 show the results of the cadmium RELMAP modeling.

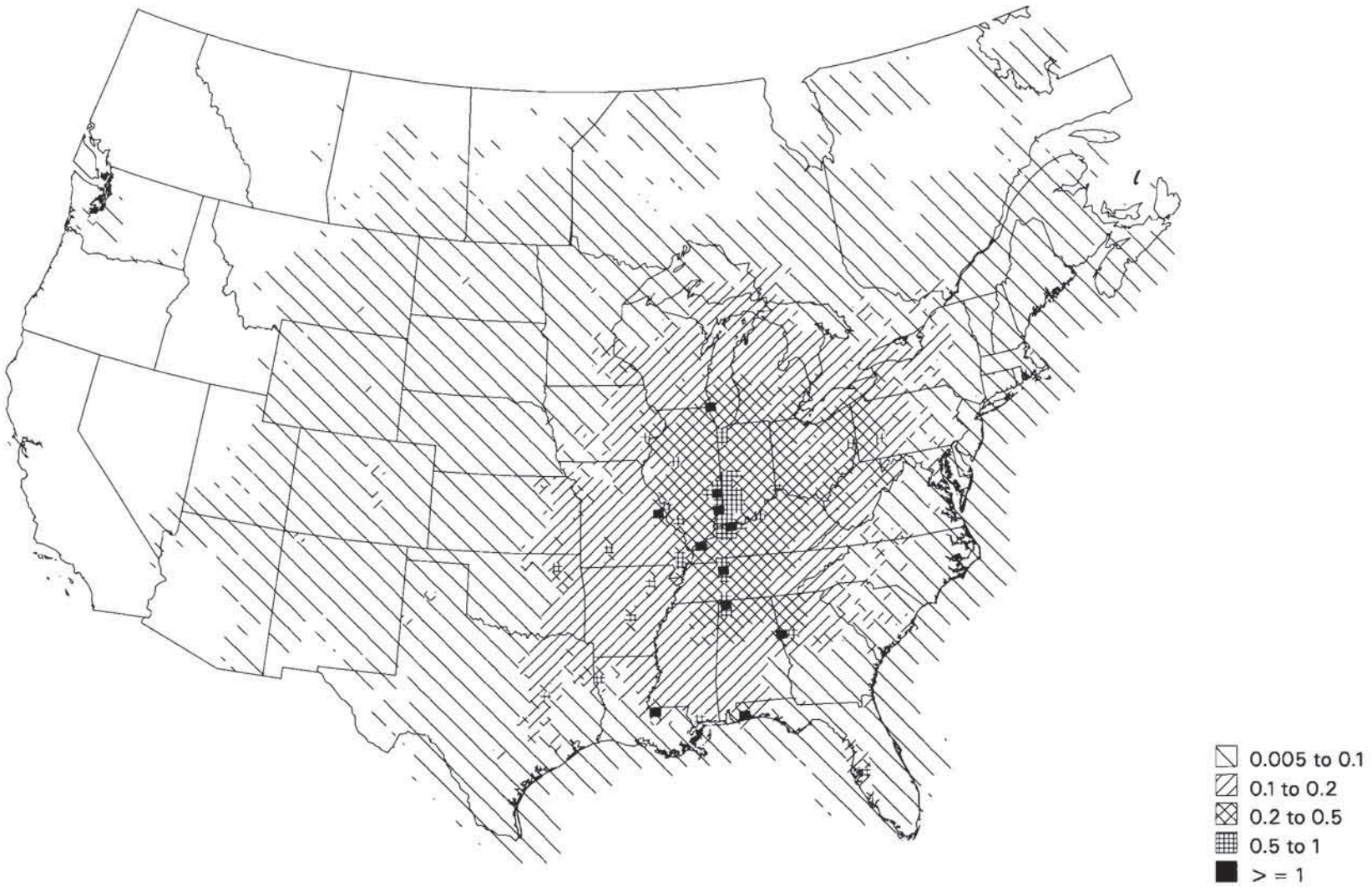



Figure 8-7. Predicted Cadimum Wet and Dry Deposition from Coal Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$

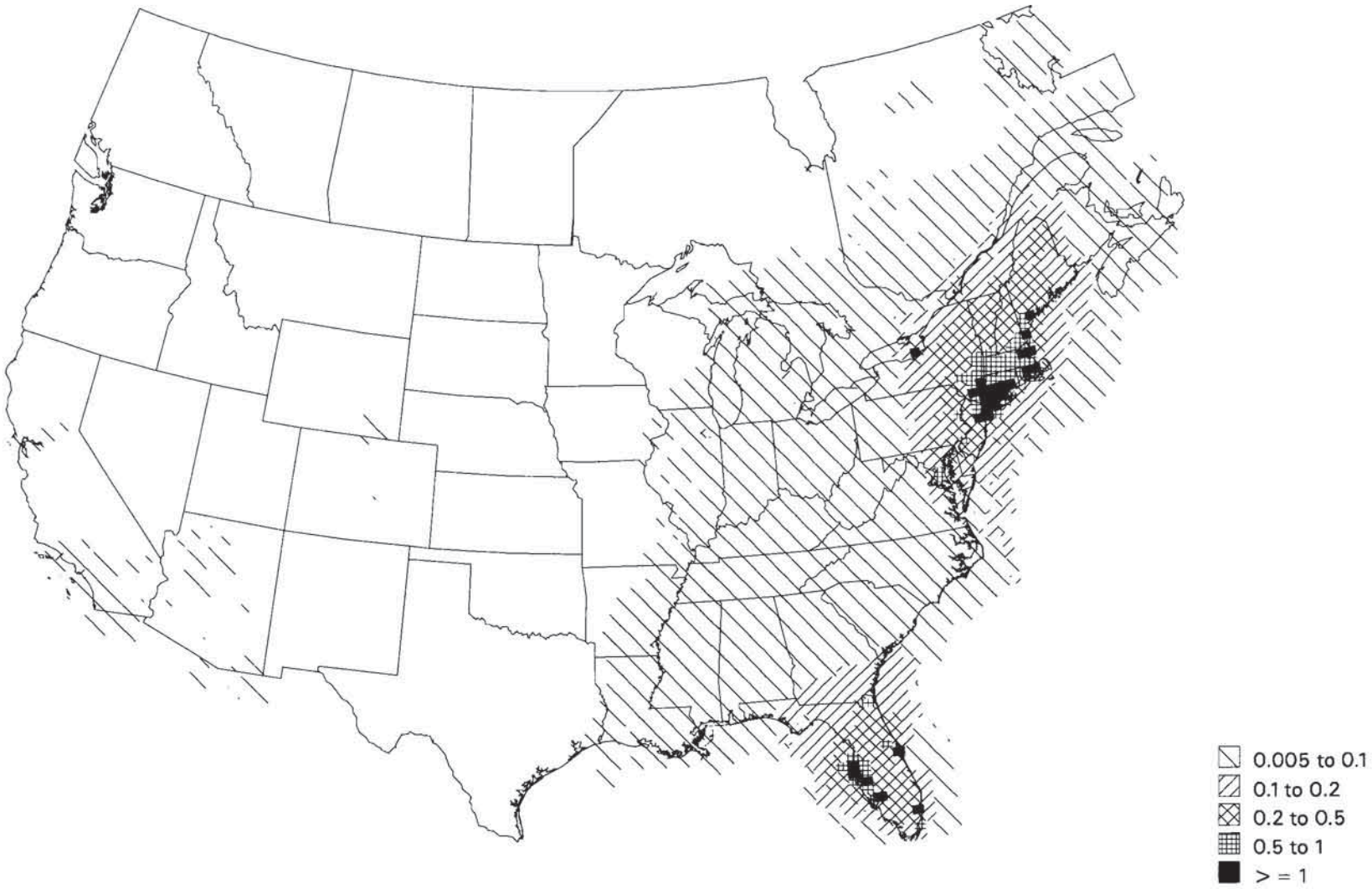



Figure 8-8. Predicted Cadimum Wet and Dry Deposition from Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: µg/m²/yr




Figure 8-9. Predicted Cadimum Wet and Dry Deposition from Coal and Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$



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## 8.4 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 appropriate in future studies. Due, in part, to low emissions of these HAPs from utilities relative to other anthropogenic sources, a quantitative assessment of noninhalation exposure to lead and/or cadmium has not been given as high a priority as arsenic, radionuclides, dioxins, and mercury for multipathway assessment. However, 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 appropriate in future studies to more comprehensively evaluate the impacts of emissions of lead and cadmium from utilities.

## 8.5 REFERENCES


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3. Stevens, J. B. Disposition of toxic metals in the agricultural food chain. 1. Steady-state bovine milk biotransfer factors. *Environmental Science and Technology.* Volume 25, No. 7. 1991. pp. 1289-1294.

4. Bolger, P. M., Yess, N. J., Gunderson, E. L., Troxell, T. C., and Carrington, C. D. Identification and reduction of sources of dietary lead in the United States. *Food Additives and Contaminants,* Volume 13, No. 1. 1996. pp. 53-60.

5. U.S. Environmental Protection Agency. *1990 Emissions Inventory of Forty Section 112 (k) Pollutants.* September 1997.

6. Agency for Toxic Substances and Disease Registry. *Toxicology Profile for Cadmium.* TP-92/06. U.S. Public Health Service. Atlanta, GA. 1993.

# 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 Generating 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 that 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 for 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 human activities, particularly 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, 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 earlier reports, including the Background Information Document supporting the decision not to regulate radionuclide emissions from coal-fired boilers issued in 1989.[4] The EPA report summarized in this chapter 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 means 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, in bituminous coal with an average content of 1.24 ppm uranium and 2.18 ppm thorium 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.

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

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 |

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. Analysis 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, estimated 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 is 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 report, estimates of radionuclide emissions and associated human health risks are based on fossil-fired boiler 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 EPA report.[5] 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.

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 |

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. 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 oil-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

The Clean Air Act Assessment Package-1993 (CAP-93) is the most recent version of a computer model used for population dose and risk assessment for radionuclide air emissions. For a given facility,

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 | $2.3 \times 10^{0}$ | $1.3 \times 10^{2}$ | $1.1 \times 10^{-1}$ | $1.5 \times 10^{0}$ | $2.6 \times 10^{-2}$ | $1.8 \times 10^{-1}$ |
| Th-234 | $1.2 \times 10^{0}$ | $1.3 \times 10^{-2}$ | $1.1 \times 10^{-1}$ | $7.7 \times 10^{-1}$ | $2.6 \times 10^{-2}$ | $1.8 \times 10^{-1}$ |
| Pa-234m | $1.2 \times 10^{0}$ | $1.3 \times 10^{-2}$ | $1.1 \times 10^{-1}$ | $7.7 \times 10^{-1}$ | $2.6 \times 10^{-2}$ | $1.8 \times 10^{-1}$ |
| Pa-234 | $1.2 \times 10^{0}$ | $1.3 \times 10^{-2}$ | $1.1 \times 10^{-1}$ | $7.7 \times 10^{-1}$ | $2.6 \times 10^{-2}$ | $1.8 \times 10^{-1}$ |
| U-234 | $2.3 \times 10^{0}$ | $2.5 \times 10^{-2}$ | $2.1 \times 10^{-1}$ | $1.5 \times 10^{0}$ | $4.9 \times 10^{-2}$ | $3.4 \times 10^{-1}$ |
| Th-230 | $1.2 \times 10^{0}$ | $4.9 \times 10^{-2}$ | $4.1 \times 10^{-1}$ | $7.7 \times 10^{-1}$ | $9.5 \times 10^{-2}$ | $6.7 \times 10^{-1}$ |
| Ra-226 | $1.7 \times 10^{0}$ | $2.9 \times 10^{-2}$ | $2.6 \times 10^{-1}$ | $1.2 \times 10^{0}$ | $5.7 \times 10^{-2}$ | $4.3 \times 10^{-1}$ |
| Rn-222 | $3.0 \times 10^{2}$ | $2.5 \times 10^{3}$ | $3.8 \times 10^{2}$ | $2.0 \times 10^{2}$ | $4.9 \times 10^{3}$ | $6.2 \times 10^{2}$ |
| Po-218 | $5.6 \times 10^{0}$ | $3.1 \times 10^{-2}$ | $2.7 \times 10^{-1}$ | $3.8 \times 10^{0}$ | $6.0 \times 10^{-2}$ | $4.4 \times 10^{-1}$ |
| Pb-214 | $5.6 \times 10^{0}$ | $3.1 \times 10^{-2}$ | $2.7 \times 10^{-1}$ | $3.8 \times 10^{0}$ | $6.0 \times 10^{-2}$ | $4.4 \times 10^{-1}$ |
| Bi-214 | $1.2 \times 10^{0}$ | $3.1 \times 10^{-2}$ | $2.7 \times 10^{-1}$ | $7.7 \times 10^{-1}$ | $6.0 \times 10^{-2}$ | $4.4 \times 10^{-1}$ |
| Po-214 | $5.6 \times 10^{0}$ | $3.1 \times 10^{-2}$ | $2.7 \times 10^{-1}$ | $3.8 \times 10^{0}$ | $6.0 \times 10^{-2}$ | $4.4 \times 10^{-1}$ |
| Pb-210 | $5.6 \times 10^{0}$ | $3.1 \times 10^{0}$ | $2.7 \times 10^{1}$ | $3.8 \times 10^{0}$ | $6.0 \times 10^{0}$ | $4.4 \times 10^{1}$ |
| Bi-210 | $1.2 \times 10^{0}$ | $3.1 \times 10^{0}$ | $2.7 \times 10^{1}$ | $7.7 \times 10^{-1}$ | $6.0 \times 10^{0}$ | $4.4 \times 10^{1}$ |
| Po-210 | $5.6 \times 10^{0}$ | $3.1 \times 10^{0}$ | $2.7 \times 10^{1}$ | $3.8 \times 10^{0}$ | $6.0 \times 10^{0}$ | $4.4 \times 10^{1}$ |
| Th-232 | $7.1 \times 10^{-1}$ | $2.1 \times 10^{-2}$ | $1.8 \times 10^{1}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{-2}$ | $3.0 \times 10^{-1}$ |
| Ra-228 | $1.0 \times 10^{0}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{0}$ | $7.1 \times 10^{-1}$ | $9.1 \times 10^{-1}$ | $6.7 \times 10^{0}$ |
| Ac-228 | $7.1 \times 10^{-1}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{0}$ | $4.7 \times 10^{-1}$ | $9.1 \times 10^{-1}$ | $6.7 \times 10^{0}$ |
| Th-228 | $7.1 \times 10^{-1}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{0}$ | $4.7 \times 10^{-1}$ | $9.1 \times 10^{-1}$ | $6.7 \times 10^{0}$ |
| Ra-224 | $1.0 \times 10^{0}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{0}$ | $7.1 \times 10^{-1}$ | $9.1 \times 10^{-1}$ | $6.7 \times 10^{0}$ |
| Rn-220 | $1.6 \times 10^{2}$ | $5.7 \times 10^{-1}$ | $8.4 \times 10^{0}$ | $1.1 \times 10^{2}$ | $1.1 \times 10^{0}$ | $1.4 \times 10^{1}$ |
| Po-216 | $3.5 \times 10^{0}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{0}$ | $2.4 \times 10^{0}$ | $9.1 \times 10^{-1}$ | $6.7 \times 10^{0}$ |
| Pb-212 | $3.5 \times 10^{0}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{0}$ | $2.4 \times 10^{0}$ | $9.1 \times 10^{-1}$ | $6.7 \times 10^{0}$ |
| Bi-212 | $7.1 \times 10^{-1}$ | $4.7 \times 10^{-1}$ | $4.1 \times 10^{0}$ | $4.7 \times 10^{-1}$ | $9.1 \times 10^{-1}$ | $6.7 \times 10^{0}$ |
| Tl-208 | $2.1 \times 10^{-1}$ | $1.4 \times 10^{-1}$ | $1.2 \times 10^{0}$ | $1.4 \times 10^{-1}$ | $2.7 \times 10^{-1}$ | $1.9 \times 10^{0}$ |
| K-40 | $7.8 \times 10^{0}$ | $6.2 \times 10^{-3}$ | $5.2 \times 10^{-3}$ | $5.3 \times 10^{0}$ | $1.2 \times 10^{-2}$ | $8.5 \times 10^{-3}$ |

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. Primary model parameters for plume dispersion and depletion are based on available 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 of CAP-93).

From plume dispersion and plume depletion calculations, the CAP-93 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 $3.9 \times 10^{-4}$ 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 exposed to radon.[1,5] 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 $1.3 \times 10^{5}$ 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 $3.6 \times 10^{-4}$ 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.

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

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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.[6] The revised estimates yield a nominal value of $5.1 \times 10^{-4}$ fatal cancers per rad for uniform whole body exposure to low-LET radiation and $2.2 \times 10^{-4}$ 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.

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 (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 about 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.

## 9.2 RADIONUCLIDE UNCERTAINTY ANALYSIS

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.[7] As in almost all assessments of environmental health risk, the risk estimates were based on modeling rather than direct measurements of exposure and risk; therefore, 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. 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; 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 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 $10^{-4}$.

## 9.3 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., $1 \times 10^{-4}$ to $1 \times 10^{-5}$).

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 risk attributable to radionuclide emissions from electric utility steam generating units (SGUs) (includes coal-, oil-, and gas-fired utilities )is less than 1 cancer death per year (i.e., $3.36 \times 10^{-1}$ 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 that of arsenic, which was modeled with the RELMAP (see chapter 6), 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 $1 \times 10^{-5}$ due to multipathway exposures to radionuclide emissions from utilities. The highest MIR value of $3 \times 10^{-5}$ 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

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 |
|---|---|---|---|---|
| $1 \times 10^{0}$ to $1 \times 10^{-1}$ | 0 | 0 | 0 | 0 |
| $1 \times 10^{-1}$ to $1 \times 10^{-2}$ | 0 | 0 | 0 | 0 |
| $1 \times 10^{-2}$ to $1 \times 10^{-3}$ | 0 | 0 | 0 | 0 |
| $1 \times 10^{-3}$ to $1 \times 10^{-4}$ | 0 | 0 | 0 | 0 |
| $1 \times 10^{-4}$ to $1 \times 10^{-5}$ | 1,027 | $1.3 \times 10^{-5}$ | $1.92 \times 10^{-4}$ | $1.92 \times 10^{-4}$ |
| $1 \times 10^{-5}$ to $1 \times 10^{-6}$ | 95,745 | $2.2 \times 10^{-6}$ | $3.06 \times 10^{-3}$ | $3.26 \times 10^{-3}$ |
| Less than $1 \times 10^{-6}$ | 196,000,000 | $1.2 \times 10^{-7}$ | $3.32 \times 10^{-1}$ | $3.36 \times 10^{-1}$ |

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 | $3 \times 10^{-5}$ | 5 | 3,340 | | | | |
| Plant #247 | $3 \times 10^{-5}$ | 4 | 900 | | | | |
| Plant #60 | $2 \times 10^{-5}$ | 4 | 3,160 | | | | |
| Plant #301 | $2 \times 10^{-5}$ | 2 | 750 | 3 | 262 | | |
| Plant #251 | $2 \times 10^{-5}$ | 4 | 1,540 | | | | |
| Plant #406 | $2 \times 10^{-5}$ | 4 | 2,777 | | | | |
| Plant #256 | $2 \times 10^{-5}$ | 3 | 1,728 | | | | |
| Plant #17 | $2 \times 10^{-5}$ | | | | | 2 | 1,112 |
| Plant #133 | $2 \times 10^{-5}$ | 2 | 1,135 | | | 2 | 66 |
| Plant #318 | $1 \times 10^{-5}$ | 6 | 1,100 | | | | |
| Plant #672 | $1 \times 10^{-5}$ | 8 | 1,965 | | | | |
| Plant #668 | $1 \times 10^{-5}$ | 7 | 2,304 | | | | |
| Plant #82 | $1 \times 10^{-5}$ | | | | | 2 | 804 |
| Plant #207 | $1 \times 10^{-5}$ | | | | | 2 | 558 |
| Plant #253 | $1 \times 10^{-5}$ | 3 | 2,052 | | | | |
| Plant #489 | $1 \times 10^{-5}$ | 4 | 1,872 | | | | |
| Plant #651 | $1 \times 10^{-5}$ | | | | | 6 | 372 |

MIR = maximum individual risk expressed as lifetime fatal cancer risk

two facilities are identified as exclusively oil-fired plants. 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 $1 \times 10^{-5}$. The MEI risk due to inhalation exposure to radionuclides from the second highest risk oil-fired utility is estimated to be $3 \times 10^{-6}$. The other 135 oil-fired utilities and all coal-fired utilities are estimated to pose cancer risks less than $1 \times 10^{-6}$ due to inhalation exposure to radionuclides.

Background Radiation Exposures

The risks due to exposure to radionuclide emissions from utilities are substantially lower than the risks due to exposure to natural background radiation. As shown in Tables 9-6 and 9-7 the average exposure to natural background radiation (excluding radon progeny) for the U.S. population has been estimated to be roughly about 100 millirems (mRems) per year.[8,9]

Background radiation exposure can come from internal or external sources. External sources include cosmic (extraterrestrial) and terrestrial (radionuclides in soil and rock). Internal sources include inhaled and ingested radionuclides retained in the body, with inhaled radon progeny treated as a separate problem. Radiation from consumer products and fallout from weapons tests make minor contributions to background. Average doses are listed in Table 9-6.

Background radon exposure is assessed on the basis of exposure to its progeny. The estimate is based on the distribution of short half-life radioactive progeny of radon in the inspired air. Progeny estimates must be calculated for each environment independently. The values in Table 9-7 are average population values and do not reflect the ranges possible.

Risks of background radiation exposure in the United States can be calculated using the average annual dose from external and internal sources and the average annual radon exposures and risk conversion factors from appropriate references.

The age averaged lifetime risk of fatal cancer associated with the average annual dose of 100 mrem (Table 9-6) is $5.7 \times 10^{-5}$.[10] Continual lifetime exposure at 100 mrem/y yields a risk of $4.3 \times 10^{-3}$. The lifetime risk of fatal lung cancer associated with the average annual exposure of 0.257 WLM (Table 9-7) is 5.8 $10^{-5}$.[11] Continuous lifetime exposure (about 75 years) at 0.257 WLM/y yields a risk of $4.4 \times 10^{-3}$.

## Table 9-6. Average Background Radiation Doses (effective dose equivalent excluding inhaled radon progeny)

| Source | Annual Dose (mrem) |
|---|---|
| External | |
| Cosmic[12] | 28.4 |
| Terrestrial[13] | 23.6 |
| Internal[14] | |
| Ingested and inhaled | 39 |
| Fallout[14] | 1 |
| Building materials and consumer products[14] | 8 |
| Total | 100 |

Notes:

An individual is expected to spend 87.9% of his/her time indoors (estimates of fraction of time in the indoor and outdoor environments were adapted from data in the *Draft Exposure Factors Handbook*, U.S. EPA, Washington, D.C., 1997.)

For terrestrial sources, the indoor dose rate in air is 0.8 times the outdoor value and the effective dose equivalent (mrem) is 0.7 times the absorbed dose (mrad) in air.[13]

The range of background exposure in the United States from the sources listed in Table 9-6 is from about 75 mrem/year to about 200 mrem/year.

## Table 9-7. Average Annual Background Exposures Due to Radon Progeny

| Source | Annual Exposure (WLM) |
|---|---|
| Residence[11] | 0.242 |
| Outdoors | 0.015 |
| Total | 0.257 |

Notes:

WLM = working level month

This radon estimate is incomplete since there is no estimate of the average exposure level inside structures other than residences. The estimated average exposure and risk can only be higher than listed here.

The estimates of average radon concentration used were: 1.25 pCi/l in residences and 0.3 pCi/l outdoors.

The estimated fraction of time in the outdoor environment (0.121) was adapted from data in the Draft Exposure Factors Handbook, U.S. EPA, Washington, D.C., 1997. The equilibrium fraction was assumed to be: 0.5 indoors and 0.8 outdoors.

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 $1 \times 10^{-5}$ 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.

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# 10.0 SCREENING LEVEL ASSESSMENT OF MULTIPATHWAY EXPOSURES AND RISKS TO ARSENIC EMISSIONS

## 10.1 PURPOSE AND SCOPE

Though arsenic is a naturally occurring element that is found in environmental media (air, water, and soil), as well as in biota, it is also released to the environment by anthropogenic sources, including fossil-fuel-fired electric utility plants (i.e., utilities). Since arsenic compounds are known to cause health effects in humans from inhalation and ingestion exposures, and since arsenic has been found at relatively high concentrations in animals and plants that are food sources for humans, the potential impact of arsenic emissions from utilities on human health was evaluated. Human exposure to arsenic through multiple exposure pathways was assessed. Though inhalation exposure assessments have been performed for arsenic emissions from many types of sources, few assessments have examined non-inhalation exposures to arsenic emissions from anthropogenic sources. Specific objectives of this analysis were: (1) to assess the magnitude of the contribution of arsenic emissions from utilities to concentrations in environmental media and biota; (2) to assess potential human exposure to arsenic emissions through multiple exposure pathways, including ingestion; (3) to identify dominant pathways of potential exposure to arsenic emitted from coal-fired and oil-fired utilities, and (4) to characterize potential human health risks from exposure to arsenic emissions from utilities.

### 10.1.1 Rationale and Usefulness of Model Plant Approach

Arsenic is generally present as a low-level contaminant in coal and oil. During combustion arsenic is volatilized from coal and oil and released to the atmosphere. For this assessment, it was not possible to model the emission impact of every utility plant. Consequently, the actual arsenic emission data and facility characteristics for any specific source were not modeled. Instead, a model plant approach was used to represent actual sources. The model plants were designed to characterize the arsenic emission rates and the atmospheric release processes exhibited by typical facilities in each of the four source classes considered. The modeled facilities were not designed to exhibit extreme sources (e.g., facilities with the highest arsenic emission rates) but rather to serve as representatives of the combustion source class.

In taking the model plant approach, it was realized that there would be a great deal of uncertainty about the predicted fate and transport of arsenic and about the ultimate estimates of exposure. The uncertainty can be divided into modeling uncertainty and parameter uncertainty. Parameter uncertainty can be further subdivided into uncertainty and variability depending on the level to which a particular model parameter is understood. A limited quantitative analysis of uncertainty is presented. It is also hoped that the direction of future research can be influenced toward reducing the identified uncertainties that significantly impact key results.

For the assessment, a series of fate, transport, and exposure models were used to predict arsenic concentrations in environmental media, pertinent biota, and arsenic contact with humans. An effort was made to estimate the amount of receptor contact with arsenic as well as the oxidative state and form of arsenic contacted. No attempt was made to estimate an internal dose.

Three models were used to predict environmental arsenic concentrations and exposure: the Regional Lagrangian Model of Air Pollution (RELMAP), the Industrial Source Complex Short-Term Version 3 (ISCST3), and the Indirect Exposure Model 2 (IEM2).

## 10.2 BACKGROUND INFORMATION ON ARSENIC

### 10.2.1 Forms of Arsenic in the Environment

Arsenic has five electrons in its outer shell; hence, it has four possible oxidation states: +5, +3, 0, and -3. Arsenic is rarely found in the environment as a free element. The two primary valence states of arsenic are the trivalent state, which is denoted by $As^{3+}/As^{3-}$ or As(III), and the pentavalent state, which is denoted by $As^{5+}$ or As(V). Arsenic in each of these valence states forms both organic and inorganic compounds (an organic arsenic compound is one in which the arsenic atom is covalently attached to at least one carbon atom). Arsenic compounds are typically classified in two different manners: (1) according to the oxidation state of arsenic ($As^{3-}$, $As^{3+}$, and $As^{5+}$), or (2) according to whether or not arsenic is in the organic form. Table 10-1 shows common arsenic compounds and their classification by valence state and organic/inorganic form.

As(III) is more mobile and soluble than the other common form, As(V).[1] Organic arsenic is present in most soils[2] and is found most often in the environment in combination with oxygen, chlorine, and sulfur. Inorganic arsenic occurs naturally in geologic formations, where its most common form is arsenopyrite (FeAsS).

Figure 10-1 presents a generalized scheme for the geochemical cycling of arsenic through various compartments of the environment. The atmosphere is a major conduit for arsenic emitted from anthropogenic sources to the balance of the cycle via the wet and dry deposition process.[3] Dry and wet deposition from the atmosphere onto soils may be followed by movement through soils either into groundwater or surface water. Passage of arsenic into surface waters may then be followed by further transfer to sediments.

The existence of chemical and biochemical transformations that occur within the cycle makes the environmental cycling of arsenic more complicated. Trivalent arsenic in the atmosphere can undergo oxidation to the pentavalent state. Such conversion can also occur in aerated surface waters. On the other hand, pentavalent arsenic in an aqueous medium which is somewhat acidic is an oxidant, and, in the presence of oxidizable material, will react to form trivalent arsenic.[4]

Table 10-1. Common Arsenic Compounds, and Classification by Valence State and Organic/Inorganic[5,6]

| Type | As(III) | As(V) |
|---|---|---|
| Inorganic | Arsenopyrite<br>Arsenite<br>Arsenous acid<br>Arsenic trioxide | Arsenate<br>Arsenic acid<br>Arsenic pentoxide |
| Organic | Arsenobetaine<br>Arsenocholine<br>Tetramethylarsonium ion | Monomethylarsonic acid<br>Dimethylarsinic acid<br>Trimethylarsine oxide<br>Dimethylarsinylethanol |

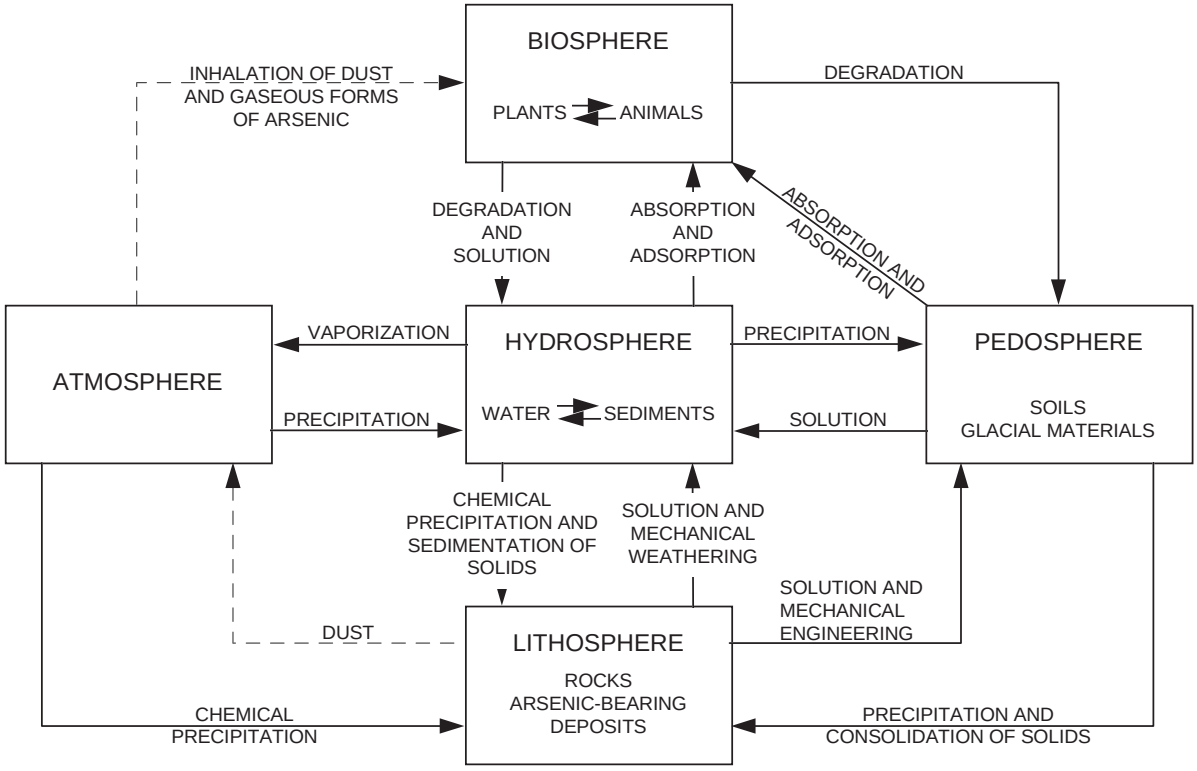



Figure 10-1. The generalized geochemical cycle for arsenic.[8,9]

10.2.2 Sources of Arsenic

The primary source of arsenic emissions in commercial processes is as a by-product of the treatment of copper, lead, cobalt, and gold ores. It is used in the production of commercial products such as agricultural products, wood preservatives, animal feed additives, medicine, ceramics, and glass, and in the dying and printing processes.[9] Estimates of the impact of anthropogenic activities that release arsenic on the global arsenic cycle vary widely, but the impacts on the local arsenic cycle can be significant.

As noted in Table 10-2, hazardous waste incineration is the most significant source of anthropogenic emissions of arsenic to air for the United States. Hazardous waste incineration is estimated to result in 1,742 tons of arsenic emissions annually, 88 percent of the total national atmospheric arsenic emissions from anthropogenic sources. Industrial source combustion, followed by utility combustion, are the second and third highest emitters of arsenic, respectively. Industrial source combustors emit an estimated 67 tons of arsenic annually, and utilities emit approximately 65 tons of arsenic each year.

10.2.3 Arsenic in the Atmosphere

Arsenic air concentrations in unimpacted areas are generally below several nanograms per cubic meter. However, values near some combustion facilities can be significantly higher, as evidenced by measurements taken outside of the United States. Reported values in the literature are provided in Table 10-3.

Two studies document initial results from monitoring for wet and dry deposition of selected trace elements at two Maryland shoreline sites in close proximity to Chesapeake Bay waters.[10,11] Both sites are located in rural settings but receive trace metal inputs from the Baltimore-Washington DC metro area and the Ohio Valley. The Ohio Valley is heavily industrialized, and sources in the Baltimore and Washington DC area include several coal and oil-fired utilities as well as other non-utility sources. Measured deposition rates for arsenic are summarized in Table 10-4.

10.2.4 Arsenic in Water

Arsenic concentrations in most U.S. rivers and lakes range from less than 10 to over 1000 $\mu$g/L.[12] A summary of measured values found in precipitation, groundwater, lakes, rivers, and oceans is provided in the EPA draft Screening Level Multipathway Exposure Analysis for Arsenic.[13]

It is expected that As(V) will dominate fresh water bodies that are approximately at equilibrium, and that a large fraction of the arsenic carried to the water body will be deposited in the benthic sediment.[4] As(V) is significantly more soluble in water than As(III) (e.g., 630 g of arsenic pentoxide will dissolve in 100 g of water, compared to only about 2 g for arsenic tridoxide at typical ambient temperatures).[4]

Table 10-2. National Arsenic Atmospheric Emission Estimates by Source Category[14]

| Facility type | Tons/year | Percent |
|---|---|---|
| Hazardous waste incineration | 1,742 | 88 |
| Industrial combustion | 67 | 3.4 |
| Utility combustion | 65 | 3.3 |
| Primary copper smelting | 47 | 2.4 |
| Commercial/institutional combustion | 32 | 1.6 |
| Copper ores, mining | 13 | 0.65 |
| Pressed and blown glass and glassware manufacturing | 6.1 | 0.31 |
| Primary nonferrous metals | 6.0 | 0.31 |
| Landfill waste gas flares | 5.9 | 0.3 |
| Secondary nonferrous metals | 1.6 | 0.081 |
| Turbines - distillate oil | 0.81 | 0.041 |
| Agricultural chemicals manufacturing | 0.47 | 0.023 |
| Sewage sludge incineration | 0.41 | 0.021 |
| Wood preserving | 0.41 | 0.021 |
| Medical waste incineration | 0.21 | 0.011 |
| Municipal waste combustion | 0.20 | 0.010 |
| Industrial inorganic chemicals manufacturing | 0.20 | 0.010 |
| Other anthropogenic sources | 0.46 | 0.029 |
| Total | 1,989 | 100 |

Table 10-3. Reported Arsenic Air Concentrations

| Type | Mean (ng/m$^3$) | Max (ng/m$^3$) | Reference |
|---|---|---|---|
| Remote areas | <21 | | Eisler 1994 |
| Kaarvatn, N.Europe | 0.5 | 1 | Pacyna et al. 1989 |
| Nordmoen, N. Europe | 1.2 | 2 | |
| Prestebaake, N. Europe | 0.5 | 1.8 | |
| Chesapeake Bay (Wye, Elms) | 0.69, 0.625 | 1.96, 1.56 | Wu et al. 1994 |
| Near coal-fired power plant (Czech) | 19000-69000 | | Eisler 1994 |
| Urban areas | <160 | | |

Table 10-4. Measured Arsenic Deposition Rates[a]

| Location | Arsenic dry deposition rate ($\mu g/m^2/yr$) | Arsenic wet deposition rate ($\mu g/m^2/yr$) | Reference |
|---|---|---|---|
| Lake Michigan | 35 | - | Wu et al. 1994 |
| Chesapeake Bay | 54-150 | - | Wu et al. 1994 |
| Chesapeake Bay | - | 45-52 | Scudlark et al. 1994 |

[a] These deposition rates are the result of emissions from a combination of natural and anthropogenic sources. Utilities are only one of the many source categories that may be contributing to the deposition rates.

### 10.2.5 Arsenic in Sediments

Arsenic concentrations in sediments of various rivers and harbors in the United States have been reported by Brannon and Patrick (1987) as 1.0 to 8.8 $\mu$g/g dry weight, and much higher concentrations have been observed at contaminated sites. Sediments often act as a sink for arsenic in natural systems and can hold high concentrations of arsenic. At contaminated sites, such as Sugar Creek, arsenic concentrations ranging from 4,470 to 66,700 $\mu$g/g dry weight were measured.[4] A summary of measured sediment concentrations is provided in the EPA draft *Screening Level Multipathway Exposure Analysis for Arsenic*.[13]

### 10.2.6 Arsenic in Soil

The majority of soils contain levels of arsenic that vary between 1 and 5 mg/kg, with much higher concentrations being observed around anthropogenic sources and areas with high background sources. Arsenic also ends up in soil as a result of the historical direct applications of arsenical pesticides (which are being phased out in some areas and are prohibited in others), landfarming of sewage sludge, deposition from air pollution, and as waste materials from industrial processes. Soil concentrations have been found to decrease rapidly with distance from an elevated point source and have been reported to decline to background concentrations within 8 to 16 km of the source.[9]

Ambient soil concentrations given in Bowen range from 0.1 to 40 mg/kg, with a mean of 6;[15] and Vinogradov[16] estimated a range of 1 to 50 mg/kg with a mean of 5.[16] A range in the United States was given by Stater, et al. of 1 to 20 mg/kg, with a mean of 7.5 mg/kg (n=52),[17] and by Conner and Shacklette as 1.6 to 72 mg/kg, with a mean of 7.5 mg/kg (n=1215).[18] Wood and Duda[19] compiled background arsenic concentrations in soils sampled from the southeastern United States (NC, SC, GA, and AL) and found that mean arsenic concentrations ranged from 0.69 to 13.3 mg/kg.[19] Cullen and Reimer report that the average arsenic concentration in the continental crust is 3 $\mu$g/g.[20]

There are few data on either the speciation of arsenic in soil or the determination of the fraction of arsenic in soil that is inorganic.

Changes in season, temperature, pH, and relative reducing/oxidizing conditions will initiate kinetically slow conversions between As(III) and As(V) species. In aerobic soils, As(V) is the dominant species, while As(III) may dominate in reduced soils (such as those that are temporarily flooded or groundwater wells). In very reduced soils (such as swamps), arsenic can be found in its elemental state, arsine, or as methylated arsenic. The results in Takamatsu et al.[21] and Bombach et al.[22] indicate that generally less than 20 percent of the arsenic in soil is As (V)[21,22]

The bioavailability of arsenic in soils contaminated either through smelter operation or mine tailings has been assessed.[23] In a recent study, immature swine were dosed with the contaminated soils (immature swine were selected because of the similarity of their gastrointestinal (GI) tracts to those of human children). The absorption from the GI tract into the bloodstream was monitored. The relative (mean) bioavailability of soil arsenic was 78 percent (C.I. = 56-111 percent) and the absolute (mean) was 52 percent (C.I. = 44-61 percent).

10.2.7 Arsenic in Terrestrial Plants

Arsenic does not readily translocate to the shoots of plants and is found mostly in the roots. In general, arsenic uptake values are low. Studies have found that, to achieve 1 mg/kg of arsenic in fresh weight plants, a soil concentration of 200-300 mg/kg is necessary.[24,25-27] Measured concentrations in a variety of food and other plants are shown in Table 10-5.

10.2.8 Arsenic in Aquatic Plants

Arsenic is not a major contaminant of aquatic plants, except in severe cases of pollution. Highest reported concentrations in aquatic plants are the result of mine and smelter wastes, reaching levels as high as 1,450 mg/kg dry weight.[28] Arsenic concentrations in some untreated areas have been measured as ranging from 1.4 to 13 mg/kg dry weight.[28]

10.2.9 Arsenic in Terrestrial Animals

Some studies have found arsenic can accumulate in meat/dairy products with most arsenic compounds accumulating in the liver and kidney.[29,30] However, reported concentrations are rare. Table 10-6 shows values reported in the literature.

10.2.10 Arsenic in Fish

Arsenic does not appear to bioaccumulate or bioconcentrate in freshwater finfish. However, it does appear to bioconcentrate in freshwater bivalves. Lower trophic level organisms generally appear to have higher concentrations of arsenic than predatory or omnivorous aquatic and marine species.

Table 10-5. Measured Arsenic Concentrations in Plants

| Terrestrial plant | Concentration (dry weight mg/kg) [mean, (range)] | Reference |
|---|---|---|
| *Edible plants* | | |
| Lettuce | 0.056 (0.012-0.68)[a] | Wiersma et al. 1986 |
| | 0.72* | Wiersma et al. 1986 (Barudi and Bielig 1980)[31] |
| | 0.076 (0 - 0.20)[a] | Wiersma et al. 1986 (Jelinek and Corneliussen 1977) |
| Spinach | 0.056 (0.024 - 0.12)[a] | Wiersma et al. 1986 |
| | 1.16[a] | Wiersma et al. 1986 (Barudi and Bielig 1980) |
| Tomato | 0.004 (0.0012 - 0.012)[a] | Wiersma et al. 1986 |
| | 0.4[a] | Wiersma et al. 1986 (Barudi and Bielig 1980) |
| Carrot | 0.12 (0.024 - 0.44)[a] | Wiersma et al. 1986 (Jelinek and Corneliussen 1977) |
| Potato | 0.064 (0.012 - 0.20)[a] | Wiersma et al. 1986 |
| | 0.04 (0 - 1.16)[a] | Wiersma et al. 1986 (Jelinek and Corneliussen 1977) |
| Wheat | 0.24 (0.024 - 1.44)[a] | Wiersma et al. 1986 |
| | < 0.24[a] | Wiersma et al. 1986 (Varo et al. 1980)[32] |
| Barley | 0.32 (0.024 - 1.88)[a] | Wiersma et al. 1986 |
| Oats | 0.92 (0.44 - 2.72)[a] | Wiersma et al. 1986 |
| | <0.24 (0.16 - 0.4)[a] | Varo et al. 1980 |
| Apple | 0.08 (0.008 - 1.08)[a] | Wiersma et al. 1986 |
| | 0.12 (<0.04 - 1.92)[a] | Reinhard 1974 |
| *Impacted/arsenic-treated areas* | | |
| Alfalfa from Montana smelter area | (2.0-28.4)[a] | Jenkins 1980[28] |
| Various species from impacted soils (80 mg/kg) | (<0.2-5.8) | Merry et al. 1986[33] |
| *Grass/fodder crops* | | |
| Colonial bent grass on low arsenic soil | (0.3-3) | Jenkins 1980 |
| Grasses on non-treated areas | (0.1-0.9) | NRC 1977 |
| Scotch heather on low-arsenic soil | 0.3 | Jenkins 1980 |
| Grass (soil concentration 1-38 $\mu$g/g dry weight, mean 11 $\mu$g/g) | 1.4 (0.36 - 5.56)[a] | Wiersma et al. 1986 |
| Silage maize (soil concentration 1-110 $\mu$g/g dry weight, mean 10 $\mu$g/g) | 0.88 (0.28 - 2.36) | Wiersma et al. 1986 |
| Sugar beet crowns & leaves (soil concentration 1-36 $\mu$g/g dry weight, mean 14 $\mu$g/g) | 2.56 (0.44 - 8.0) | Wiersma et al. 1986 |

[a] Converted from freshweight values assuming water content of 0.8.
[b] These values are for the Netherlands. A range of mean soil concentrations of 5 - 12 $\mu$g/g dry weight is reported.

Table 10-6. Measured Arsenic Concentrations in Meat and Other Animal Products

| Meat product | Wet weight ($\mu$g/g) | Assumed moisture content | Dry weight ($\mu$g/g) | Reference |
|---|---|---|---|---|
| Beef | 0.005 ±0.001 | 0.615[a] | 0.013 ±0.003 | Vreman et al. 1986 |
| Beef Liver | 0.008 ±0.001 | 0.70[a] | 0.027 ±0.003 | Vreman et al. 1986 |
| Milk | <0.001 | 0.87[a] | <0.003 | Vreman et al. 1986 |
| Pork | <0.1 | 0.615[b] | <0.3 | Jelinek and Corneliussen 1977[c] |
| Poultry | 0 - 0.5 | 0.615[b] | 0 - 1.30 | Jelinek and Corneliussen 1977[c] |
| Eggs | 0 - 0.2 | ? | | Jelinek and Corneliussen 1977[c] |
| Lamb | 0.002 ±0.001 | 0.67 | 0.006 ±0.003 | van der Veen and Vreman 1986 |

[a] Baes et al. 1984
[b] Assumed same as beef.
[c] Detection limit of 0.1 $\mu$g/g. Measurement is for $As_2O_3$.

The species of arsenic present in fish appear to be somewhat variable. Marine organisms can convert inorganic arsenic into organic arsenic species, with arsenobetaine being both the major form identified in fish and the suspected metabolic endpoint for arsenic in the marine environment. Arsenobetaine, or "fish-arsenic," an organic arsenic species, is a relatively nontoxic form of arsenic. There is little evidence of human toxicity due to arsenobetaine exposure. Arsenocholine and inorganic arsenic species have also been identified as forms of arsenic in fish. Arsenocholine may be more toxic than arsenobetaine, but these results are questionable. Inorganic arsenicals are rapidly converted to less toxic organic forms of arsenic in fish.[34] The primary arsenic form in marine fish is arsenobetaine, which comes from arsenosugars present in algae and not from the formation of trimethylarsine oxide.

Table 10-7 shows a summary of measured freshwater fish concentrations of total arsenic. Even though shellfish and other marine foods contain the greatest arsenic concentrations, much of the arsenic present in fish and shellfish exists in the less toxic organic form.[35] In shellfish, 0-2.9 percent of the total arsenic is generally measured to be in inorganic forms; in finfish, 0-9.5 percent is generally measured as inorganic.

Table 10-7. Total Arsenic Concentrations in Freshwater Fish in the United States

| Freshwater fish | Concentration range (Wet weight mg/kg) | Reference |
|---|---|---|
| Bass - muscle | 0.0 - 0.51 | Jenkins 1980[28] |
| Bluegill- nationwide- whole | 0.05-0.4 | Jenkins 1980; Wiener et al. 1984[36] |
| Catfish - native- muscle | 0.0-0.3 | Jenkins 1980 |
| Catfish - cultured- muscle | 0.2-3.1 | Jenkins 1980 |
| Common carp - nationwide - muscle | 0.0-0.2 | Jenkins 1980 |
| Northern pike - northern U.S. - muscle | <0.01- 0.1 | Jenkins 1980 |
| Coho salmon- USA- muscle | 0.07 - 0.17 | Jenkins 1980 |
| Atlantic salmon - muscle oil | 0.8-3.1 | Jenkins 1980 |
| Rainbow trout- all tissues | <0.4 | Jenkins 1980 |
| Various species - USA, 1976 - 1984 – | 0.14 - 2.9 | Lima et al. 1984[37]; Schmitt & Brumbaugh |
| White sucker - Muscle | 0.03-0.13 | Jenkins 1980 |
| Lake trout- nationwide- whole | 0.06-0.68 | Jenkins 1980 |

### 10.2.11 Speciation of Arsenic in Food Products

Though total arsenic levels have been measured in a variety of food products, limited data are available on concentrations of particular arsenic species in food products. The chemical forms of arsenic in foods are varied and complex. Table 10-8 lists reported percentages of inorganic arsenic in foods. The form found in fish and marine foods was discussed above. Other food products, such as meats, rice, and cereals, contain higher percentages, and often higher total amounts, of inorganic arsenic.

### 10.2.12 Arsenic Near Anthropogenic Sources

Increased arsenic concentrations have been measured in environmental media surrounding anthropogenic atmospheric emission sources, including copper and lead smelters. The measured data indicate an elevation of arsenic concentrations that is related to the emission of arsenic and subsequent deposition to the terrestrial environment. Table 10-9 shows a summary of values. Many of these data were collected at sources that operated prior to the use of modern particulate control technology, and the facilities are not of the type considered in this report.[38] However, they are included because the data indicate that some arsenic sources can have an observable impact on their local environment.

## 10.3 SUMMARY OF MODELS AND APPROACH

In order to address the objectives of this study, a series of modeling efforts were performed. The Regional Lagrangian Model of Air Pollution (RELMAP) model was used to model the long-range transport of arsenic from coal- and oil-fired utilities. A separate effort involved modeling the atmospheric dispersion and deposition of arsenic on a local scale (within 50 km), using the Industrial Source Complex

## Table 10-8. Percentage of Inorganic Arsenic Compared to Total Arsenic in Selected Foods

| Food | Percent Inorganic As |
|---|---|
| Milk and dairy products | 75% |
| Meat-- beef and pork | 75% |
| Poultry | 65% |
| Saltwater fish[a] | 0% |
| Freshwater fish (0-9.5 percent)[a] | 10% |
| Cereals | 65% |
| Rice | 35% |
| Vegetables | 0.5% |
| Potatoes | 10% |
| Fruits | 10% |

[a] The values presented for fish are not widely accepted.

Source: Weiler (1987) in Borum and Abernathy (1994) as cited in IF Kaiser, 1996.

## Table 10-9. Environmental Concentrations near Facilities[38]

| Facility | Value(s) | Reference |
|---|---|---|
| *Air (ng/m$^3$)* | | |
| Former USSR | 500-1900 | Pershagen, G. and Vahter, M. 1979 |
| Texas | max. 1400 | Pershagen, G. and Vahter, M. 1979 |
| Tacoma, Washington | max. 1500 | Pershagen, G. and Vahter, M. 1979 |
| Romania | max. 1600 | Pershagen, G. and Vahter, M. 1979 |
| Germany | 900-1500 | Pershagen, G. and Vahter, M. 1979 |
| *Drinking water ($\mu$g/L)* | | |
| Mexico, from plant producing $As_2O_3$ | 4000-6000 | Pershagen, G. and Vahter, M. 1979 |
| Japan, near factory producing As sulfide | 3000 | Pershagen, G. and Vahter, M. 1979 |
| *Dust (mg/kg or $\mu$g/g)* | | |
| Tacoma, WA, near smelter | 1300 (remote from smelter 70) | NRC 1977 |
| *Sediments (mg/kg dry weight, $\mu$g/g)* | | |
| Near sewer outfall | 35 | NRCC 1978 |
| *Soil (mg/kg or $\mu$g/g) dry weight* | | |
| Tacoma, WA, near smelter | Max. 380 | Pershagen, G. and Vahter, M. 1979 |
| Japan | max. 2470 | Pershagen, G. and Vahter, M. 1979 |
| *Fish, near smelter (water arsenic 2.3-2.9 $\mu$g/L), $\mu$g/g freshweight* | | |
| Total arsenic | 0.05 - 0.24 | Norin et al. 1985. |
| Inorganic arsenic | 0.01 - 0.02 | Norin, H., M. Vahter, A. Christakopoulos, and M. Sandstroem 1985. |

Source: Eisler (1994)

Short-Term Model 3 (ISCST3). Finally, using the predicted air concentrations and deposition rates, the Indirect Exposure Model 2 (IEM2) model was used to predict environmental concentrations and subsequent exposure for several hypothetical exposure scenarios.

10.3.1 Source Classes Considered and Model Plant Approach

Four different arsenic source facilities were considered in the local-scale aspect of this study: three coal-fired utilities (a small, a medium, and a large facility) and an oil-fired utility. A list of these facilities and their emission rates is provided in Table 10-10. Details about these facilities and how the parameterizations for the model plants were performed are provided in Appendix A of the *EPA Draft Screening Level Multipathway Exposure Analysis for Arsenic*.[13]

10.3.2 Atmospheric Transport Modeling

10.3.2.1 Local Scale Modeling. The ISCST3 was used to estimate the atmospheric dispersion and deposition of emitted arsenic within a 50 km radius of the facilities.[39,40] The ISCST3 uses hourly meteorological data to estimate the ambient air concentrations of an emitted pollutant, as well as the wet and dry deposition rates. The dry deposition of the air pollutant is calculated, based on particle size, atmospheric conditions, and gravitational settling velocities. Wet deposition of the pollutant is based on scavenging coefficients that depend on particle size and the precipitation rate.

For each of the four facilities considered, ISCST3 was run in both a humid and an arid site, with receptors placed at 16 directions around the facility and 30 distances between 200 m and 50 km, for a total of 480 receptors. Simple (i.e., flat) terrain was assumed. Values at other locations were estimated using linear interpolation, as discussed in Appendix A of *EPA Draft Screening Level Multipathway Exposure Analysis for Arsenic*.[13] Area-averaged air concentrations and deposition rates were estimated, for use in the exposure modeling.

10.3.2.2 Regional Transport Modeling. Long-range transport was modeled for arsenic. The methods, model, and approach are described in detail in chapter 6 (section 6.6). Figures 10-2 through 10-4 show the results of the arsenic RELMAP modeling.

10.3.3 Indirect Exposure Modeling

Because of its chemical and physical characteristics, arsenic emitted to the atmosphere may be transported to other environmental media (i.e., soil or water), thus allowing noninhalation exposures to arsenic to occur. The IEM2 was used to predict the terrestrial and aquatic fate and transport of arsenic deposited in the region of interest, as well as the human exposure to arsenic. The IEM2 calculated arsenic concentrations in watershed soils; these concentrations were then used in calculating concentrations in various food plants. The waterbody component of the IEM2 model calculated arsenic concentrations in surface water and in aquatic organisms. In

Table 10-10. Summary of Model Plants and Emission Rates Used for the Assessment

| Model plant | Stack/source | Total arsenic emissions | |
|---|---|---|---|
| | | kg/yr | g/s |
| Large coal-fired utility boiler | Stack 1 | 85.0 | 2.70e-3 |
| | Stack 2 | 74.0 | 2.35e-3 |
| | Stack 3 | 98.0 | 3.11e-3 |
| Medium coal-fired utility boiler | Stack 1 | 32.0 | 1.01e-3 |
| | Stack 2 | 30.0 | 9.51e-4 |
| Small coal-fired utility boiler | Stack 1 | 15.4 | 4.88e-4 |
| | Stack 2 | 8.8 | 2.79e-4 |
| Oil-fired utility boiler | Stack 1 | 14.6 | 4.63e-4 |
| | Stack 2 | 18.8 | 5.96e-4 |

addition, the IEM2 model calculated human exposures for selected exposure scenarios through multiple exposure routes, including food consumption, water ingestion, and inhalation.

10.3.3.1 Description of Exposure Scenarios. Three basic exposure scenarios were considered: a subsistence farmer (adult and child), a subsistence fisher (adult and child), and a pica child. These scenarios were considered because they represent possible high-end scenarios for exposure to arsenic. Table 10-11 summarizes the exposure pathways considered for each of these scenarios.

Table 10-12 shows the default values for the scenario-independent parameters for both the child and adult receptors, and Table 10-13 shows the default values for the scenario-dependent exposure parameters (the technical bases for these values are provided in Appendix B of the EPA Draft *Screening Level Multipathway Exposure Analysis for Arsenic*).[13]

The subsistence farmer scenario consists of a subsistence farmer and child who consume elevated levels of locally-grown food products. It was assumed that each farm was located on approximately 10 acres. The subsistence farmer was assumed to raise livestock and to consume home-grown animal tissue and animal products, including chickens, eggs, beef, and dairy products. All chicken feed was assumed to be derived from non-local sources (and is, therefore, not contaminated with arsenic). For bovine consumption of contaminated feed, 100 percent of the hay and corn used for feed was assumed to be from the affected area. It was also assumed that the drinking water for the subsistence farmer comes from rainwater collected in cisterns. Though rainwater collected in cisterns may not be the primary source of drinking water for most farmers, some are still expected to use rainwater collected in cisterns as the primary source. Since rainwater is likely to have the highest arsenic levels due to arsenic emissions from anthropogenic sources, this assumption is consistent with the high-end exposure scenario for this screening level assessment.

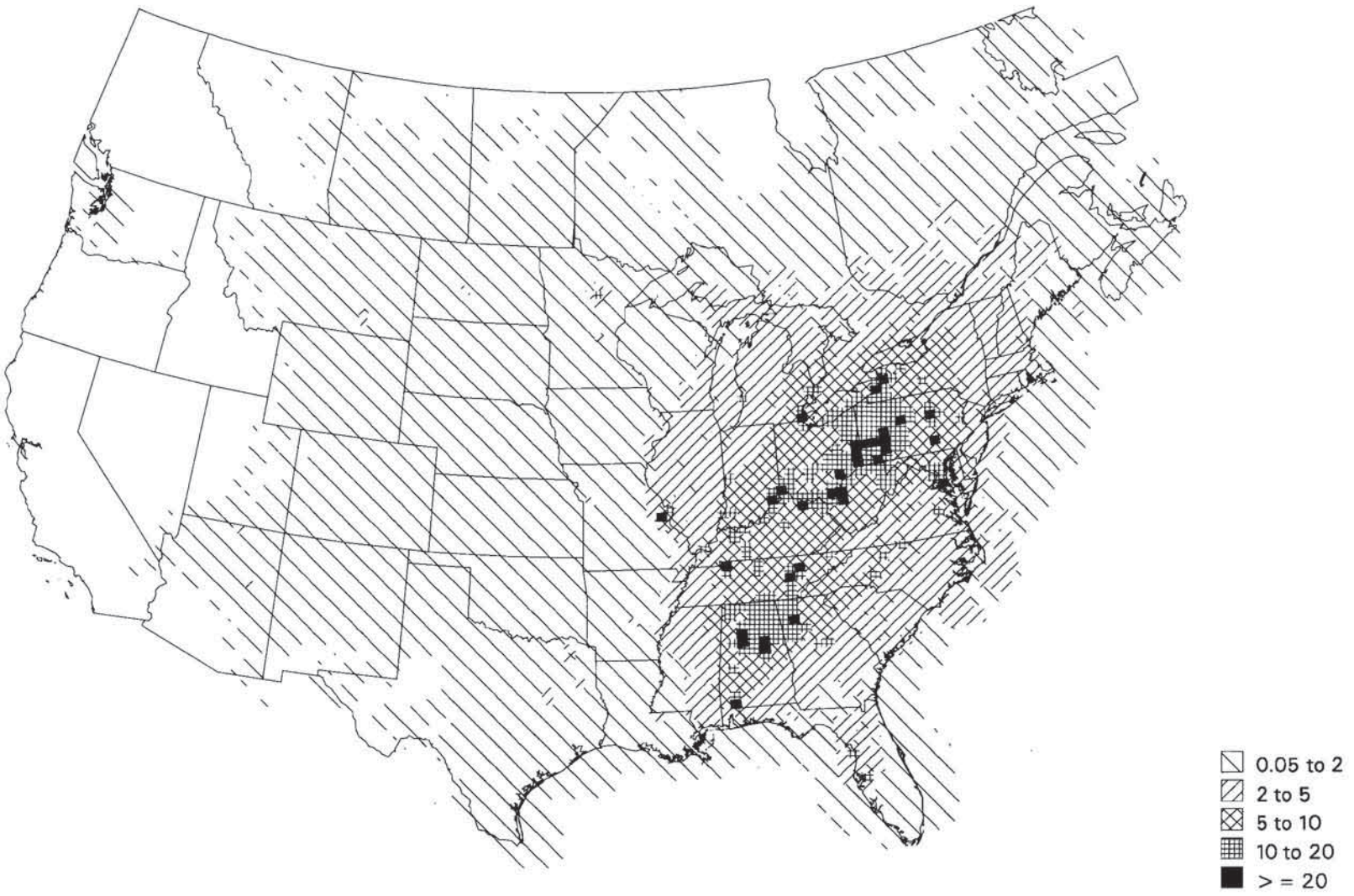



Figure 10-2. Predicted Arsenic Wet and Dry Deposition from Coal Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$



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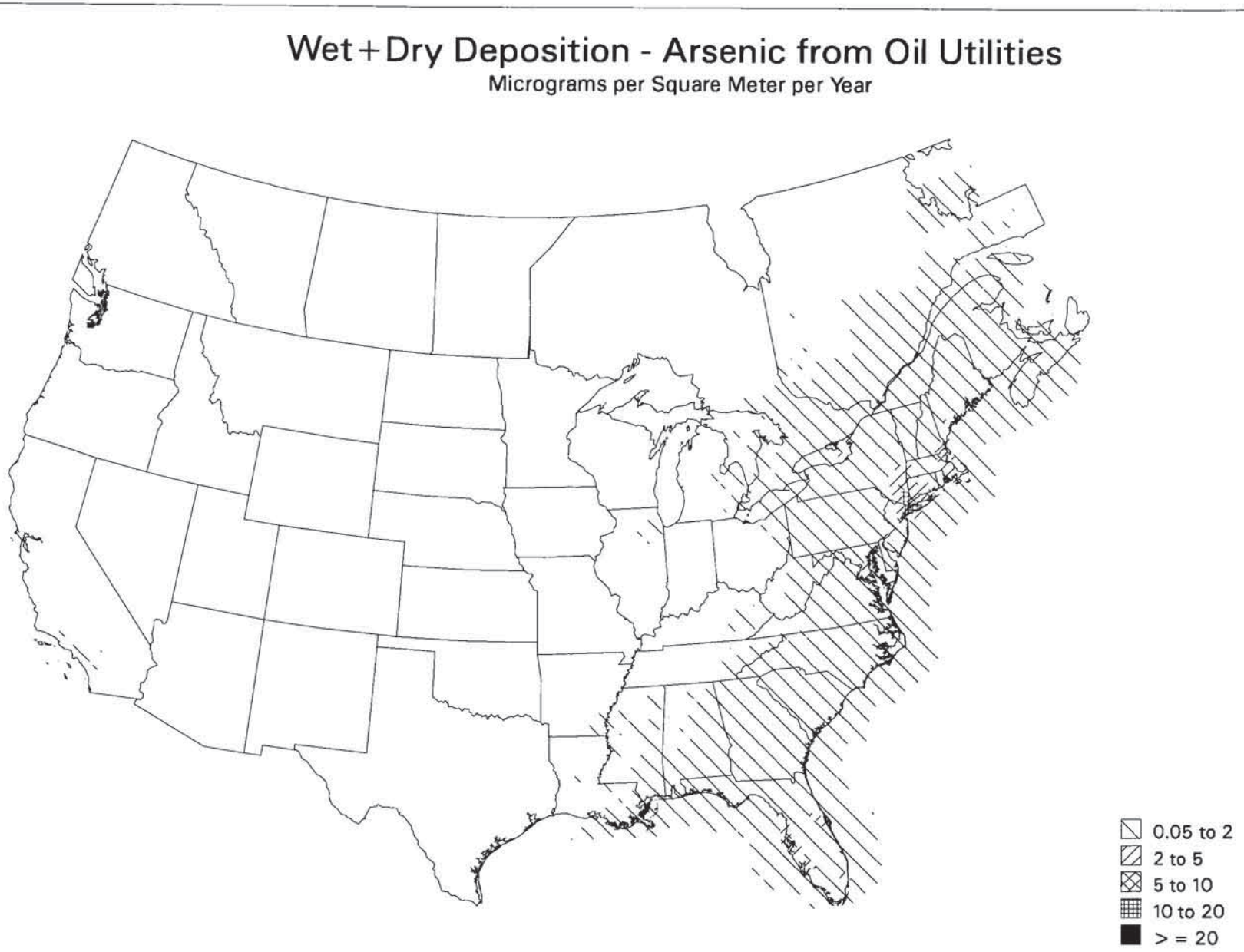



Figure 10-3. Predicted Arsenic Wet and Dry Deposition from Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$

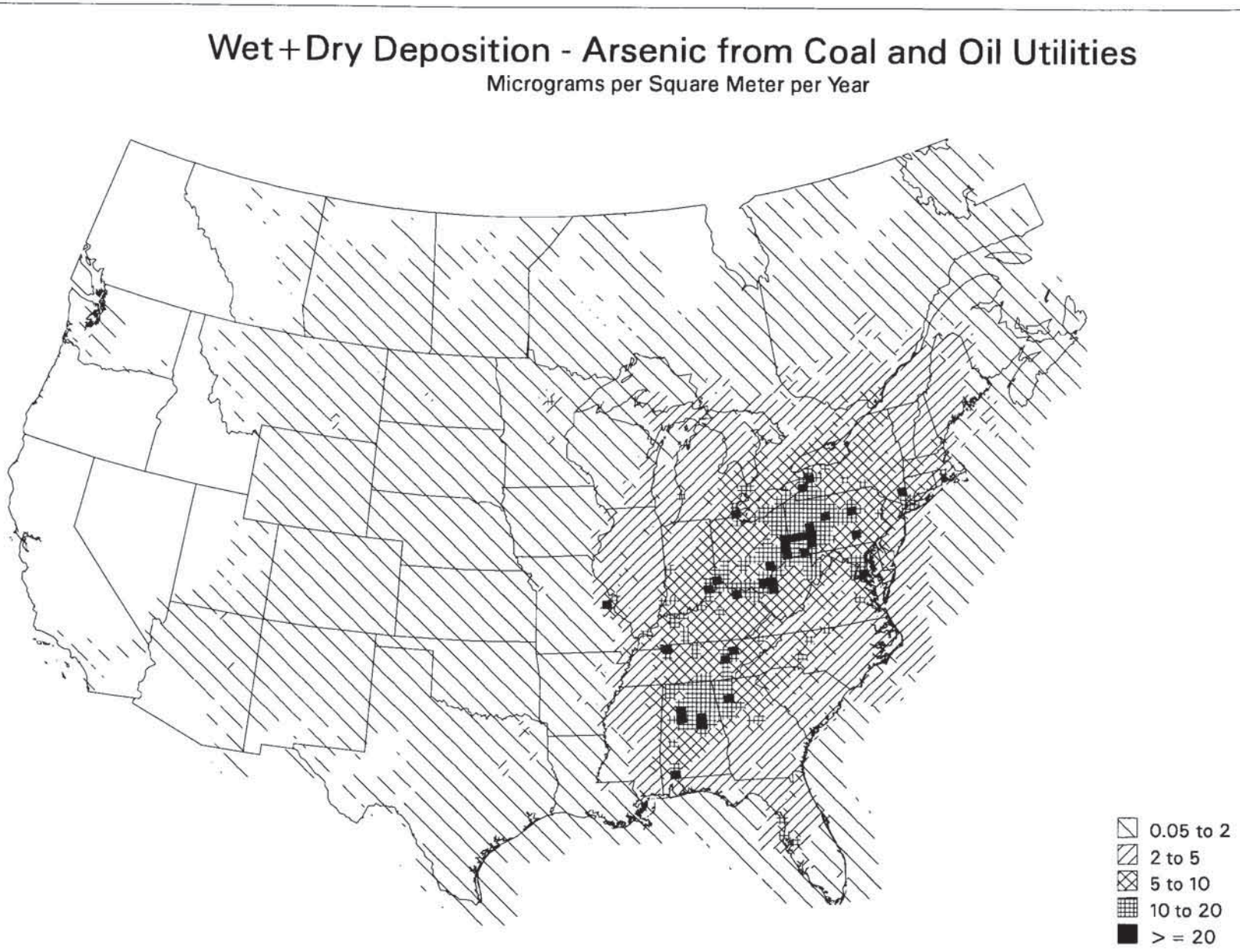



Figure 10-4. Predicted Arsenic Wet and Dry Deposition from Coal and Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: $\mu g/m^2/yr$



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Table 10-11. Summary of Human Exposure Scenarios Considered

| Exposure route | Subsistence farmer | | Pica child | Subsistence fisher | |
|---|---|---|---|---|---|
| | Adult | Child | | Adult | Child |
| Air inhalation | X | X | X | X | X |
| Soil Ingestion | X | X | X | X | X |
| Animal ingestion | X | X | | | |
| Vegetable ingestion | X | X | | X | X |
| Fish ingestion | | | | X | X |
| Water ingestion | X | X | | X | X |

Blank = Pathway not considered; X = Pathway considered.

Table 10-12. Default Values of Scenario-Independent Exposure Parameters

| Parameter[a] | Value[b] | |
|---|---|---|
| | Adult | Child |
| Body weight (kg) | 70 | 17 |
| Exposure duration (years) | 30 | 18 |
| Inhalation rate ($m^3$/day) | 20 | 16 |
| Vegetable consumption rates (g/kg body weight/day) | | |
| Leafy vegetables | 0.028 | 0.008 |
| Grains and cereals | 1.87 | 3.77 |
| Legumes | 0.381 | 0.666 |
| Potatoes | 0.17 | 0.274 |
| Fruits | 0.57 | 0.223 |
| Fruiting vegetables | 0.064 | 0.12 |
| Animal product consumption rates (g/kg body weight/day) | | |
| Beef (excluding liver) | 0.341 | 0.553 |
| Beef liver | 0.066 | 0.025 |
| Dairy | 0.599 | 2.04 |
| Pork | 0.169 | 0.236 |
| Poultry | 0.111 | 0.214 |
| Eggs | 0.093 | 0.073 |
| Lamb | 0.057 | 0.061 |
| Soil ingestion rates (g/day) | 0.1 | Scenario-dependent |
| Water ingestion rate (L/day) | 2 | 1 |

[a] All human consumption rates except for soil and water are reported as dry weight.
[b] See Appendix B of the EPA draft Screening Level Multipathway Exposure Analysis for details regarding these parameter values.

Table 10-13. Values for Scenario-Dependent Exposure Parameters[a]

| Parameter | Subsistence farmer | | Pica Child | Subsistence fisher[b] | |
|---|---|---|---|---|---|
| | Adult | Child | | Adult | Child |
| Finfish ingestion rates (g/day) | NA | NA | NA | 60 | 20 |
| Soil Ingestion rate (g/day) | 0.1 | 0.2 | 7.5 | 0.1 | 0.2 |
| Contact time for inhalation (hr/day) | 24 | 24 | 24 | 24 | 24 |
| Contact fractions (unitless)[c] | | | | | |
| Animal products | 1 | 1 | NA | NA | NA |
| Leafy vegetables | 1 | 1 | NA | 0.058 | 0.058 |
| Grains and cereals | 1 | 1 | NA | 0.667 | 0.667 |
| Legumes | 1 | 1 | NA | 0.8 | 0.8 |
| Potatoes | 1 | 1 | NA | 0.225 | 0.225 |
| Fruits | 1 | 1 | NA | 0.233 | 0.233 |
| Fruiting vegetables | 1 | 1 | NA | 0.623 | 0.623 |
| Root vegetables | 1 | 1 | NA | 0.268 | 0.268 |
| Drinking water[d] | 1 | 1 | NA | 1 | 1 |

[a] See Appendix B of the EPA draft *Screening Level Multipathway Exposure Analysis for Arsenic* for more details regarding these values.
[b] The subsistence fisher scenario does not consider consumption of freshwater bivalves or marine organisms.
[c] Contact fraction is the fraction of the total consumption of a food product from the study site.
[d] The source of the contaminated drinking water is different for the subsistence farmer and fisherperson scenarios.

NA - Not considered to be applicable to this assessment.

For the urban high-end scenario, a pica child was defined as consuming 7.5 g of soil per day at the location of maximum deposition of arsenic. Estimates of the rate of soil ingestion by pica children range from 5 g/day to 50 g/day. The data of Calabrese et al. (1989) show a range of 5-10 g/day. A value of 7.5 grams was considered appropriate to represent this subpopulation of children.

The subsistence fisher scenario consisted of a subsistence fisher and child whose fish and water consumption scenarios were associated with the hypothetical lake setting. The high-end fish consumer scenario represented an individual who was assumed to ingest large amounts of locally-caught fish, as well as home-grown garden produce (plant ingestion parameters identical to the rural home gardener scenario) and drinking water from the affected lake. These

consumption scenarios were thought to represent identified fish-consuming subpopulations in the United States. No commercial distribution of locally caught fish was assumed. Fish consumption rates for the fish-consuming subpopulations were derived from the Columbia River Inter-Tribal Fish Commission Report.[41]

All arsenic was assumed to be of the inorganic form when estimating exposure, except for fish consumption. Based on the estimates of 0.5 to 75 percent inorganic arsenic in a variety of food products (see Table 10-8), particularly percentages of 65-75 percent for meat, dairy products, grains, and cereals, a conservative estimate of 100 percent was selected, since the assessment is being performed as a screening-level assessment. For freshwater fish, for which more measurements have been taken, it is assumed that 10 percent of the arsenic is inorganic arsenic. This is also a conservative estimate, since lower levels of inorganic arsenic are generally measured in fish, but it is appropriate to select a conservative number for this screening-level assessment.

10.3.3.2 Description of Waterbody/Watershed Configuration. The watershed was assumed to be a circular region with a radius of 3.5 km, and the waterbody was assumed to be a small circular lake with a radius of 0.9 km (i.e., the ratio of watershed to waterbody is 15). Three different locations of the lake within the watershed were considered; these locations are shown in Figure 10-5. The area-averaged values for the watersheds and waterbodies were calculated at five distances (0.2, 2, 5, 10 and 25 km; where "distance" is the distance between the closest point on the watershed and the facility) and for 16 directions around each facility; thus, there are a total of 240 different watershed/waterbody configurations per facility.

10.3.4 Determination of Background Values

To assist in determining the reasonableness of the IEM2 model parameterization, model runs were performed using background soil concentrations, air concentrations, and deposition rates. Of these parameters, the soil concentration was found to be the most important because it was critical to the estimation of arsenic concentrations in plants and to the estimation of the flux to the water body. In this section, the values assumed for arsenic background soil concentrations, air concentrations, and deposition rates are discussed.

In this assessment, background arsenic concentrations were defined as the natural arsenic levels in the soil and air that would exist without any anthropogenic input of arsenic. Determination of such a background level presents a formidable challenge: it is difficult to quantify; there can be considerable variability in what constitutes background; and, at present, there is little guidance on how background should be determined. However, due to the naturally high concentrations of arsenic in many soils, it was considered

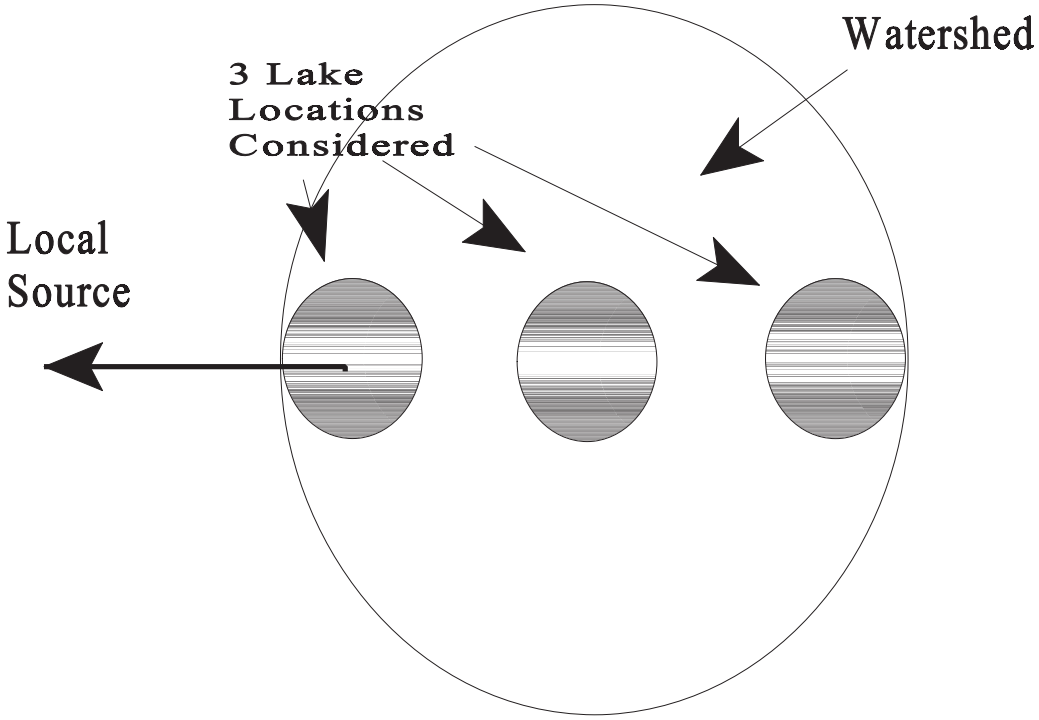



Figure 10-5. Location of waterbody considered within watershed.

critical that such background concentrations be addressed, even if only in a simple, screening-level manner. It is hoped that this will serve as a starting place for future arsenic assessments.

Natural and anthropogenic arsenic soil concentrations show a wide range of variation. Even crustal levels of arsenic can range from 0.1 to several hundred mg/kg. Therefore, it is difficult to establish typical ambient levels of arsenic in soil without respect to a particular geological formation and geographical area, and general comparisons must be made with caution. For the purpose of this modeling exercise, a background soil concentration of 3 $\mu$g/g was assumed. This is the average arsenic concentration for the continental crust reported in Cullen and Reimer.[20]

Chilvers and Peterson[7] reported that the current natural levels of arsenic in the atmosphere account for approximately 60 percent of the total atmospheric load, due primarily to volcanic sources and biological methylation. Based on this, the mean air concentration of 0.7 ng/m$^3$, reported in Wu et al.[10] for the Chesapeake Bay and assumed to be representative for a relatively unimpacted area, was multiplied by 0.6 to obtain a background arsenic air concentration of 0.4 ng/m$^3$. It is important to note that the ultimate impact of the air concentration in this case is minimal, as the assumed soil concentration drives the concentrations in biota and the water body. The dry deposition rate was assumed to be 60 $\mu$g/m$^2$/yr (=0.6*100; see Table 10-4),[10] and the wet deposition rate was assumed to be 30 $\mu$g/m$^2$/yr (=0.6*50; see Table 10-4).[42] Few data were located on the speciation of arsenic in the atmosphere. Andreae reports that the As(V):As(III) ratio was 2:1 in rainfall,[43] and so it was assumed that this same relationship holds for the background dry deposition rate and air concentration.

## 10.4 MODELING RESULTS

The following is a presentation of the results of this assessment. Included is a discussion of the air modeling results, a comparison of these results with measured data, and the potential impact of these on hypothetical receptors.

### 10.4.1 Air Modeling Results/Comparison with Measured Data

10.4.1.1 Local Scale Modeling. The fraction of arsenic emissions predicted to be deposited within 50 km of the source are presented in Tables 10-14 and 10-15, representing the two different meteorological regimes that were modeled. In general, less than 50 percent of the emissions are predicted to be deposited within 50 km, and facilities with shorter stacks have a higher fraction of emissions deposited within 50 km of the facility. Except for the large coal-fired utility, dry deposition is greater than wet deposition. This is the opposite of what has been concluded for regional models (e.g., the regional results presented in this report and those in Alcamo et al.)[44]

Table 10-14. Fraction of Arsenic Emissions Predicted to Be Deposited Within 50 km in an Arid Site

| Boiler type | Stack[a] | Release height (m) | Fraction of emissions deposited within 50 km | | |
|---|---|---|---|---|---|
| | | | Total | Dry | Wet |
| Oil-fired utility boiler | Stack 1 | 87 | 13% | 12% | 1% |
| | Stack 2 | 83 | 14% | 13% | 1% |
| Large coal-fired utility boiler | Stack 1 | 198 | 5% | 4% | 1% |
| | Stack 2 | 191 | 5% | 4% | 1% |
| | Stack 3 | 173 | 6% | 5% | 1% |
| Medium coal-fired utility boiler | Stack 1 | 147 | 7% | 6% | 1% |
| | Stack 2 | 122 | 9% | 8% | 1% |
| Small coal-fired utility boiler | Stack 1 | 84 | 14% | 13% | 1% |
| | Stack 2 | 79 | 14% | 13% | 1% |

[a] Size distribution for stacks is mass based.

Table 10-15. Fraction of Arsenic Emissions Predicted to Be Deposited Within 50 km in a Humid Site

| Boiler type | Stack[a] | Release height (m) | Fraction of emissions deposited within 50 km | | |
|---|---|---|---|---|---|
| | | | Total | Dry | Wet |
| Oil-fired utility boiler | Stack 1 | 87 | 21% | 15% | 6% |
| | Stack 2 | 83 | 21% | 16% | 6% |
| Large coal-fired utility boiler | Stack 1 | 198 | 10% | 4% | 7% |
| | Stack 2 | 191 | 11% | 4% | 7% |
| | Stack 3 | 173 | 12% | 5% | 7% |
| Medium coal-fired utility boiler | Stack 1 | 147 | 14% | 7% | 7% |
| | Stack 2 | 122 | 16% | 10% | 7% |
| Small coal-fired utility boiler | Stack 1 | 84 | 23% | 16% | 7% |
| | Stack 2 | 79 | 24% | 17% | 7% |

[a] Size distribution for stacks is mass based.

For the large coal-fired utility, the high stacks result in a significantly lower fraction deposited by dry deposition. The predicted wet deposition is not as sensitive to stack height and, therefore, does not change.

10.4.1.2 Regional Scale Modeling. RELMAP was used to model the long-range transport of arsenic. Modifications to RELMAP for atmospheric arsenic simulation were based on the assumption that all arsenic emissions are in particulate form. Only the field mode of RELMAP was used in this assessment (see Section 6.6 of Chapter 6 for description of field mode and other details of the RELMAP). The RELMAP regional-scale air modeling results are presented in Table 10-16.

Table 10-16. RELMAP Air Modeling Results

| Result | Western United States (>90° longitude) | | Eastern United States (>90° longitude) | |
|---|---|---|---|---|
| | 50th percentile | 90th percentile | 50th percentile | 90th percentile |
| Arsenic air concentration (ng/$m^3$) | 9.00e-04 | 6.10e-03 | 1.40e-02 | 3.90e-02 |
| Arsenic dry deposition rate ($\mu$g/$m^2$/yr) | 9.80e-03 | 8.50e-02 | 7.40e-02 | 7.60e-01 |
| Arsenic wet deposition rate ($\mu$g/$m^2$/yr) | 7.70e-02 | 7.70e-01 | 1.90e+00 | 7.40e+00 |

## 10.4.2 Indirect Exposure Modeling

10.4.2.1 Comparison of Predicted Values with Measured Values. Table 10-17 compares the results for the eastern and western sites. The more humid climate of the eastern site results in slightly more runoff, and, therefore, more influx of arsenic into the water body. A critical factor in the water body calculations is the soil-water partition coefficient. The value used here is 100 L/kg (see Appendix B of the EPA draft *Screening Level Multipathway Exposure Analysis for Arsenic for Explanation*)[13] and is based on a combination of curve fits to more sophisticated partitioning modeling and other literature data. The surface water concentrations are essentially inversely proportional to the soil-water partition coefficient. The surface water and sediment concentrations are within the range of values reported for lakes in Section 2. It is noted that the sediment concentrations are not used for any subsequent modeling.

To assist in determining the reasonableness of the model predictions, typical values for input parameters were used in the model, and then the predicted concentrations were compared with measured values. The predicted plant and animal concentrations are shown in Table 10-18. The predicted concentrations are comparable with those presented in (Tables 10-5, 10-6, and 10-7), although the values are not consistently at one end of the range for all food types. For example, the predicted beef values are slightly higher (by about 6 ng/g), while most of the other food types are in the middle or lower end of the range of concentrations reported in the literature for what are assumed to be "unimpacted" values. Most of the arsenic in the beef (60 percent) is predicted to come from the ingestion of grain/forage/silage, while the rest is predicted to come from the ingestion of soil.

The fish and bivalve concentrations are calculated from the arsenic water concentration. The difference in predicted values is due entirely to the difference in bioconcentration factors used (1 L/kg for finfish, 350 L/kg for bivalves).

Table 10-17. Predicted Surface Water and Benthic Sediment Concentrations for the Hypothetical Water Bodies

| Concentrations[a] | Eastern site | Western site |
|---|---|---|
| Total arsenic water concentration ($\mu$g/L) | 14 | 9 |
| Percent of arsenic dissolved | 97 | 90 |
| Predicted suspended sediment concentration (mg/L) | 0.5 | 2.2 |
| Total arsenic benthic sediment concentration (µg/g dry weight) | 27 | 17 |

[a] Assuming arsenic air concentration of 0.4 ng/m$^3$, deposition rate of 90 µg/m$^2$/yr, and soil concentration of 3 µg/g.

Table 10-18. Modeled Arsenic Concentrations[a]

| Biota | Modeled concentration ($\mu$g/g dry weight) | Background concentration range ($\mu$g/g dry weight)[b] |
|---|---|---|
| Grain | 0.181 | 0.024 - 2.72 |
| Fruits | 0.002 | 0.008 - 1.92 |
| Fruiting vegetables | 0.001 | |
| Leafy vegetables | 0.009 | 0.056 - 1.16 |
| Beef | 0.022 | 0.01 - 0.016 |
| Dairy | 0.001 | < 0.003 |
| Pork | 0.008 | < 0.3 |
| Poultry | 0.008 | 0 - 1.30 |
| Freshwater fish (fresh weight) | 0.013 | 0 - 3.1 |
| Freshwater bivalves (fresh weight) | 4.7 | |

[a] Assuming a background soil concentration (3 $\mu$g/g), air concentration (0.4 ng/m$^3$), and deposition rate (90 $\mu$g/m$^2$/yr).
[b] Background concentration ranges were taken from data in Tables 5 and 6.

The predicted exposures for the hypothetical receptors are shown in Tables 10-19 (total arsenic) and 10-20 (inorganic arsenic). The intake of total arsenic is predicted to be dominated by the ingestion of grains for all scenarios in which grains are consumed, except for the adult subsistence fisher. In the case of the adult subsistence fisher, most of the exposure to arsenic is predicted to be from the ingestion of freshwater bivalves. Because only 10 percent of the total arsenic in fish is assumed to be inorganic, ingestion of grains is predicted to be the dominant food ingestion exposure to inorganic arsenic, even for the adult subsistence fisher scenario. Exposure to arsenic for the pica child scenario, in which only exposure to arsenic through soil ingestion is considered, is predicted to be approximately twice as large as that for the subsistence farmer child.

## Table 10-19. Predicted Total Arsenic Exposure for Hypothetical Receptors[a]

| Scenario | Results for Total As | | % for Receptor | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Eastern site (humid) | Total inhalation intake (mg/kg/day) | Total ingestion intake (mg/kg/day) | Water | Grains | Legumes | Potatoes | Root_ vegetables | Fruits | Fruiting_ vegetables | Leafy_ vegetables | Beef | Beef_ liver | Dairy | Pork | Poultry | Eggs | Lamb | Soil_ ingestion | Fish | Bivalves |
| Subsistence Farmer Adult | 1E-07 | 4E-04 | 0 | 94 | 1 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| Subsistence Farmer Child | 4E-07 | 7E-04 | 0 | 91 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 |
| Pica Child | 4E-07 | 1E-03 | | | | | | | | | | | | | | | | | | |
| Subsistence Fisher Adult | 1E-07 | 5E-04 | 0 | 44 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 52 |
| Subsistence Fisher Child | 4E-07 | 5E-04 | 0 | 89 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 3 | 0 |
| **Western site (arid)** | | | | | | | | | | | | | | | | | | | | |
| Subsistence Farmer Adult | 1E-07 | 4E-04 | 1 | 93 | 1 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| Subsistence Farmer Child | 4E-07 | 8E-04 | 1 | 91 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 |
| Pica Child | 4E-07 | 1E-03 | | | | | | | | | | | | | | | | | | |
| Subsistence Fisher Adult | 1E-07 | 4E-04 | 1 | 56 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 40 |
| Subsistence Fisher Child | 4E-07 | 5E-04 | 2 | 89 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 2 | 0 |

[a] Assuming background arsenic soil concentration of 3 $\mu$g/g, air concentration of 0.4 ng/m$^3$, and deposition rate of 90 $\mu$g/m$^2$/yr.



## Table 10-20. Predicted Total Inorganic Arsenic Exposure for Hypothetical Receptors[a]

| Scenario | Results for Inorganic As | | % for Receptor | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Eastern site (humid) | Total inhalation intake (mg/kg/day) | Total ingestion intake (mg/kg/day) | Water | Grains | Legumes | Potatoes | Root_ vegetables | Fruits | Fruiting_ vegetables | Leafy_ vegetables | Beef | Beef_ liver | Dairy | Pork | Poultry | Eggs | Lamb | Soil_ ingestion | Fish | Bivalves |
| Subsistence Farmer Adult | 1E-07 | 4E-04 | 0 | 94 | 1 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| Subsistence Farmer Child | 4E-07 | 7E-04 | 0 | 91 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 |
| Pica Child | 4E-07 | 1E-03 | | | | | | | | | | | | | | | | | | |
| Subsistence Fisher Adult | 1E-07 | 3E-04 | 0 | 86 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 10 |
| Subsistence Fisher Child | 4E-07 | 5E-04 | 0 | 91 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 |
| **Western site (arid)** | | | | | | | | | | | | | | | | | | | | |
| Subsistence Farmer Adult | 1E-07 | 4E-04 | 1 | 93 | 1 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| Subsistence Farmer Child | 4E-07 | 8E-04 | 1 | 91 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 |
| Pica Child | 4E-07 | 1E-03 | | | | | | | | | | | | | | | | | | |
| Subsistence Fisher Adult | 1E-07 | 3E-04 | 2 | 89 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 6 |
| Subsistence Fisher Child | 4E-07 | 5E-04 | 2 | 90 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 |

[a] Assuming background arsenic soil concentration of 3 $\mu$g/g, air concentration of 0.4 ng/m$^3$, and deposition rate of 90 $\mu$g/m$^2$/yr.

10.4.2.2 Combined Results

10.4.2.2.1 *Contribution of individual and regional sources to total concentrations*. One of the objectives of this analysis was to assess the magnitude of the contribution of arsenic emissions from the four types of utility boilers, as compared to concentrations in environmental media and biota. This objective was addressed by calculating the concentrations in media and biota with and without an estimate of background, and by including the regional contribution of nationwide utilities. These results are shown in Tables 10-21 through 10-25.

When background levels are included and watershed area-averaged media concentration values are used, concentrations in media are usually dominated by background estimates. This is because the concentrations in most media are strongly dependent on the arsenic soil concentration, and the build-up of arsenic in soil, due to deposition of arsenic from the facilities over 30 years, is predicted to be only a fraction of typical background soil concentrations. The large coal-fired utility burner (LCUB) was predicted to account for slightly more than 50 percent of the total deposition on the watershed considered, but the build-up in the soil after 30 years is still predicted to be less than 5 percent of the total soil concentration. The facilities' contribution to the watershed air concentration is small relative to the assumed background concentration of 0.7 $ng/m^3$.

As mentioned above, none of the facilities is predicted to contribute more than 10 percent to the total area-averaged arsenic soil concentration in the watershed after a 30-year build-up period. The LCUB has the largest contribution. A possibly important uncertainty, though, is that it is not known if the deposited arsenic from any of the facilities considered is more bioavailable than that typically found naturally in the soil. If the deposited arsenic is significantly more bioavailable, then the contribution to the total exposure could be more than the predicted level.

The concentrations in grain were also examined because the ingestion of grain is predicted to be a dominant pathway for exposure to arsenic. The contribution of each facility to the arsenic concentration in grain followed the same pattern as that for soil, because the grain was predicted to accumulate arsenic mostly from the soil.

It is important to note that the regional contribution of utilities, as estimated by the RELMAP model, is generally less than 10 percent of the total background concentration or deposition. However, this contribution is larger than that for some types of facilities considered on the local scale when the upper 90th percentile is used.

## Table 10-21. Watershed Air Concentration

| Watershed air concentration (µg/m$^3$)[a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 2.2E-06 | 4.0E-04 | 99 | 4.4E-04 | 91 | 9 |
| Medium coal-fired utility boiler | 1.4E-06 | 4.0E-04 | 100 | 4.4E-04 | 91 | 9 |
| Small coal-fired utility boiler | 2.3E-06 | 4.0E-04 | 99 | 4.4E-04 | 91 | 9 |
| Oil-fired utility boiler | 2.0E-06 | 4.0E-04 | 100 | 4.4E-04 | 91 | 9 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 1.5E-06 | 4.0E-04 | 100 | 4.1E-04 | 98 | 1 |
| Medium coal-fired utility boiler | 1.1E-06 | 4.0E-04 | 100 | 4.1E-04 | 98 | 1 |
| Small coal-fired utility boiler | 1.9E-06 | 4.0E-04 | 100 | 4.1E-04 | 98 | 1 |
| Oil-fired utility boiler | 1.9E-06 | 4.0E-04 | 100 | 4.1E-04 | 98 | 1 |

[a] Edge of watershed, 200 m from source

## Table 10-22. Watershed Deposition Rate

| Watershed deposition rate (µg/m$^2$/yr)[a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 1.2E+02 | 2.1E+02 | 42 | 2.2E+02 | 41 | 4 |
| Medium coal-fired utility boiler | 3.1E+01 | 1.2E+02 | 74 | 1.3E+02 | 70 | 6 |
| Small coal-fired utility boiler | 1.4E+01 | 1.0E+02 | 86 | 1.1E+02 | 80 | 7 |
| Oil-fired utility boiler | 1.5E+01 | 1.1E+02 | 86 | 1.1E+02 | 79 | 7 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 1.6E+01 | 1.1E+02 | 85 | 1.1E+02 | 85 | 1 |
| Medium coal-fired utility boiler | 4.8E+00 | 9.5E+01 | 95 | 9.6E+01 | 94 | 1 |
| Small coal-fired utility boiler | 3.9E+00 | 9.4E+01 | 96 | 9.5E+01 | 95 | 1 |
| Oil-fired utility boiler | 3.5E+00 | 9.4E+01 | 96 | 9.4E+01 | 95 | 1 |

[a] Edge of watershed 200 m from source

## Table 10-23. Surface Water Concentration

| Surface water concentration (µg/L)[a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 3.1E-01 | 1.4E+01 | 97 | 1.4E+01 | 97 | 1 |
| Medium coal-fired utility boiler | 7.8E-02 | 1.4E+01 | 98 | 1.4E+01 | 98 | 1 |
| Small coal-fired utility boiler | 3.4E-02 | 1.4E+01 | 99 | 1.4E+01 | 99 | 1 |
| Oil-fired utility boiler | 3.8E-02 | 1.4E+01 | 99 | 1.4E+01 | 98 | 1 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 1.0E-01 | 9.4E+00 | 96 | 9.4E+00 | 96 | 3 |
| Medium coal-fired utility boiler | 2.9E-02 | 9.3E+00 | 97 | 9.3E+00 | 97 | 3 |
| Small coal-fired utility boiler | 1.9E-02 | 9.3E+00 | 97 | 9.3E+00 | 97 | 3 |
| Oil-fired utility boiler | 1.8E-02 | 9.3E+00 | 97 | 9.3E+00 | 97 | 3 |

[a] Edge of watershed 200 m from source.

## Table 10-24. Untilled Soil Concentration

| Untilled soil concentration (µg/g)[a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 4.7E-02 | 3.1E+00 | 97 | 3.1E+00 | 97 | 1 |
| Medium coal-fired utility boiler | 1.2E-02 | 3.0E+00 | 98 | 3.0E+00 | 98 | 1 |
| Small coal-fired utility boiler | 5.4E-03 | 3.0E+00 | 99 | 3.0E+00 | 99 | 1 |
| Oil-fired utility boiler | 5.8E-03 | 3.0E+00 | 99 | 3.0E+00 | 99 | 1 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 1.5e-02 | 3.1e+00 | 97 | 3.1e+00 | 97 | 3 |
| Medium coal-fired utility boiler | 4.8E-03 | 3.1E+00 | 97 | 3.1E+00 | 97 | 3 |
| Small coal-fired utility boiler | 3.8E-03 | 3.1E+00 | 97 | 3.1E+00 | 97 | 3 |
| Oil-fired utility boiler | 3.5E-03 | 3.1E+00 | 97 | 3.1E+00 | 97 | 3 |

[a] Edge of watershed 200 m from source.

Table 10-25. Grain Concentration

| Grain (µg/g dry weight)[a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 6.4E-04 | 1.8E-01 | 99 | 1.8E-01 | 99 | 0 |
| Medium coal-fired utility boiler | 1.6E-04 | 1.8E-01 | 100 | 1.8E-01 | 100 | 0 |
| Small coal-fired utility boiler | 7.3E-05 | 1.8E-01 | 100 | 1.8E-01 | 100 | 0 |
| Oil-fired utility boiler | 7.9E-05 | 1.8E-01 | 100 | 1.8E-01 | 100 | 0 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 7.2E-05 | 1.8E-01 | 100 | 1.8E-01 | 100 | 0 |
| Medium coal-fired utility boiler | 2.2E-05 | 1.8E-01 | 100 | 1.8E-01 | 100 | 0 |
| Small coal-fired utility boiler | 1.8E-05 | 1.8E-01 | 100 | 1.8E-01 | 100 | 0 |
| Oil-fired utility boiler | 1.6E-05 | 1.8E-01 | 100 | 1.8E-01 | 100 | 0 |

[a] Edge of watershed 200 m from source.

10.4.2.2.2 *Contribution of individual and regional sources to inorganic arsenic exposure*. The predicted exposures to inorganic arsenic are summarized in Tables 10-26 through 10-33 for the hypothetical exposure scenarios considered. Tables 10-26 through 10-29 represent the ingestion exposure pathway for the hypothetical exposure scenarios, and Tables 10-30 through 10-33 represent the inhalation exposure pathway for the exposure scenarios. In general, less than 10 percent of the total inorganic arsenic exposure through ingestion of food products is predicted to be attributable to any of the local sources, for the hypothetical scenarios considered. This is a reflection of the individual sources' contribution to the total soil concentration, because the concentrations in media and biota, and, hence, exposure to these media and biota, is strongly dependent on the soil concentration. The exception to the trend is the pica child scenario, in which a pica child is assumed to be exposed at the location of maximum deposition. In this case, the LCUB is predicted to contribute up to approximately 40 percent of the total inorganic exposure. The other facilities are predicted to contribute less than 15 percent. While this scenario is considered to be rare, it does indicate that is possible for some of the types of facilities considered to result in exposures comparable to background exposure.

Table 10-26. Inorganic Arsenic Intake via Ingestion for Pica Child

| Inorganic arsenic intake (mg/kg/day) for pica child[a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility +background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 7.3E-04 | 2.1E-03 | 64 | 2.1E-03 | 64 | 1 |
| Medium coal-fired utility boiler | 1.8E-04 | 1.5E-03 | 87 | 1.5E-03 | 87 | 1 |
| Small coal-fired utility boiler | 7.7E-05 | 1.4E-03 | 93 | 1.4E-03 | 93 | 1 |
| Oil-fired utility boiler | 8.8E-05 | 1.4E-03 | 93 | 1.4E-03 | 93 | 1 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 5.3E-05 | 1.4E-03 | 93 | 1.4E-03 | 93 | 3 |
| Medium coal-fired utility boiler | 3.5E-05 | 1.4E-03 | 95 | 1.4E-03 | 95 | 3 |
| Small coal-fired utility boiler | 1.6E-05 | 1.4E-03 | 96 | 1.4E-03 | 96 | 3 |
| Oil-fired utility boiler | 1.7E-05 | 1.4E-03 | 96 | 1.4E-03 | 96 | 3 |

[a] Receptor point based on location of maximum deposition.

Table 10-27. Inorganic Arsenic Intake via Ingestion for Subsistence Farmer Adult

| Inorganic arsenic intake (mg/kg/day) for subsistence farmer adult[a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 5.4E-06 | 3.7E-04 | 98 | 3.7E-04 | 98 | 0 |
| Medium coal-fired utility boiler | 1.3E-06 | 3.6E-04 | 99 | 3.6E-04 | 99 | 0 |
| Small coal-fired utility boiler | 5.8E-07 | 3.6E-04 | 100 | 3.6E-04 | 99 | 0 |
| Oil-fired utility boiler | 6.3E-07 | 3.6E-04 | 100 | 3.6E-04 | 99 | 0 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 2.3E-05 | 3.9E-04 | 94 | 3.9E-04 | 94 | 0 |
| Medium coal-fired utility boiler | 5.2E-06 | 3.7E-04 | 98 | 3.7E-04 | 98 | 0 |
| Small coal-fired utility boiler | 1.8E-06 | 3.7E-04 | 99 | 3.7E-04 | 99 | 0 |
| Oil-fired utility boiler | 2.3E-06 | 3.7E-04 | 99 | 3.7E-04 | 99 | 0 |

[a] Edge of watershed 200 m from source.

## Table 10-28. Inorganic Arsenic Intake via Ingestion for Subsistence Farmer Child

| Inorganic arsenic intake (mg/kg/day) for subsistence farmer child [a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 1.0E-05 | 7.5E-04 | 98 | 7.5E-04 | 98 | 0 |
| Medium coal-fired utility boiler | 2.5E-06 | 7.5E-04 | 99 | 7.5E-04 | 99 | 0 |
| Small coal-fired utility boiler | 1.1E-06 | 7.4E-04 | 100 | 7.5E-04 | 99 | 0 |
| Oil-fired utility boiler | 1.2E-06 | 7.5E-04 | 100 | 7.5E-04 | 99 | 0 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 4.6E-05 | 8.0E-04 | 94 | 8.0E-04 | 94 | 0 |
| Medium coal-fired utility boiler | 1.0E-05 | 7.6E-04 | 98 | 7.6E-04 | 98 | 0 |
| Small coal-fired utility boiler | 3.7E-06 | 7.5E-04 | 99 | 7.5E-04 | 99 | 0 |
| Oil-fired utility boiler | 4.7E-06 | 7.5E-04 | 99 | 7.6E-04 | 99 | 0 |

[a] Edge of watershed 200 m from source.

## Table 10-29. Inorganic Arsenic Intake via Ingestion for Subsistence Fisher Adult

| Inorganic arsenic intake (mg/kg/day) for subsistence fisher adult [a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 5.0E-06 | 2.7E-04 | 98 | 2.7E-04 | 98 | 0 |
| Medium coal-fired utility boiler | 1.3E-06 | 2.6E-04 | 99 | 2.6E-04 | 99 | 0 |
| Small coal-fired utility boiler | 5.3E-07 | 2.6E-04 | 99 | 2.6E-04 | 99 | 1 |
| Oil-fired utility boiler | 5.8E-07 | 2.6E-04 | 99 | 2.6E-04 | 99 | 1 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 2.3E-05 | 2.8E-04 | 91 | 2.8E-04 | 91 | 1 |
| Medium coal-fired utility boiler | 5.3E-06 | 2.6E-04 | 97 | 2.6E-04 | 97 | 1 |
| Small coal-fired utility boiler | 1.9E-06 | 2.6E-04 | 99 | 2.6E-04 | 99 | 1 |
| Oil-fired utility boiler | 2.4E-06 | 2.6E-04 | 99 | 2.6E-04 | 99 | 1 |

[a] Edge of watershed 200 m from source.

## Table 10-30. Inorganic Arsenic Intake via Ingestion for Subsistence Fisher Child

| Inorganic arsenic intake (mg/kg/day) for subsistence fisher child [a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 9.3E-06 | 5.1E-04 | 98 | 5.1E-04 | 98 | 0 |
| Medium coal-fired utility boiler | 2.3E-06 | 5.0E-04 | 99 | 5.0E-04 | 99 | 0 |
| Small coal-fired utility boiler | 9.7E-07 | 5.0E-04 | 99 | 5.0E-04 | 99 | 0 |
| Oil-fired utility boiler | 1.1E-06 | 5.0E-04 | 99 | 5.0E-04 | 99 | 0 |
| *Western site (arid)* | | | | | | |
| Large Coal-fired utility boiler | 4.7E-05 | 5.5E-04 | 91 | 5.5E-04 | 91 | 0 |
| Medium coal-fired utility boiler | 1.1E-05 | 5.1E-04 | 97 | 5.1E-04 | 97 | 1 |
| Small coal-fired utility boiler | 3.7E-06 | 5.1E-04 | 99 | 5.1E-04 | 99 | 1 |
| Oil-fired utility boiler | 4.8E-06 | 5.1E-04 | 99 | 5.1E-04 | 99 | 1 |

[a] Edge of watershed 200 m from source.

## Table 10-31. Inorganic Arsenic Intake via Inhalation for Pica Child

| Inorganic arsenic intake via inhalation (mg/kg body wt./day) for pica child [a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 4.1E-21 | 3.8E-07 | 100 | 4.1E-07 | 91 | 9 |
| Medium coal-fired utility boiler | 1.7E-18 | 3.8E-07 | 100 | 4.1E-07 | 91 | 9 |
| Small coal-fired utility boiler | 1.7E-16 | 3.8E-07 | 100 | 4.1E-07 | 91 | 9 |
| Oil-fired utility boiler | 8.9E-16 | 3.8E-07 | 100 | 4.1E-07 | 91 | 9 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 4.9E-18 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |
| Medium coal-fired utility boiler | 8.6E-24 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |
| Small coal-fired utility boiler | 6.8E-19 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |
| Oil-fired utility boiler | 1.9E-18 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |

[a] Receptor point based on location of maximum deposition.

## Table 10-32. Inorganic Arsenic Intake via Inhalation for Subsistence Farmer Adult and Subsistence Fisher Adult

| Inorganic arsenic intake via inhalation (mg/kg body wt./day) for subsistence farmer and fisher adults [a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 6.2E-10 | 1.1E-07 | 99 | 1.3E-07 | 91 | 9 |
| Medium coal-fired utility boiler | 4.1E-10 | 1.1E-07 | 100 | 1.3E-07 | 91 | 9 |
| Small coal-fired utility boiler | 6.4E-10 | 1.1E-07 | 99 | 1.3E-07 | 91 | 9 |
| Oil-fired utility boiler | 5.7E-10 | 1.1E-07 | 100 | 1.3E-07 | 91 | 9 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 0.0E+00 | 1.1E-07 | 100 | 1.2E-07 | 98 | 2 |
| Medium coal-fired utility boiler | 4.3E-23 | 1.1E-07 | 100 | 1.2E-07 | 98 | 2 |
| Small coal-fired utility boiler | 1.3E-18 | 1.1E-07 | 100 | 1.2E-07 | 98 | 2 |
| Oil-fired utility boiler | 4.7E-18 | 1.1E-07 | 100 | 1.2E-07 | 98 | 2 |

[a] Edge of watershed 200 m from source.

## Table 10-33. Inorganic Arsenic Intake via Inhalation for Subsistence Farmer Child and Subsistence Fisher Child

| Inorganic arsenic intake via inhalation (mg/kg body wt./day) for subsistence farmer and fisher children [a] | | | | | | |
|---|---|---|---|---|---|---|
| **Facility** | **Facility only** | **Facility + background** | **% Background** | **Facility + background + RELMAP90th** | **% Background** | **% RELMAP** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired utility boiler | 2.0E-09 | 3.8E-07 | 99 | 4.2E-07 | 91 | 9 |
| Medium coal-fired utility boiler | 1.4E-09 | 3.8E-07 | 100 | 4.1E-07 | 91 | 9 |
| Small coal-fired utility boiler | 2.1E-09 | 3.8E-07 | 99 | 4.2E-07 | 91 | 9 |
| Oil-fired utility boiler | 1.9E-09 | 3.8E-07 | 100 | 4.2E-07 | 91 | 9 |
| *Western site (arid)* | | | | | | |
| Large coal-fired utility boiler | 0.0E+00 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |
| Medium coal-fired utility boiler | 1.4E-22 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |
| Small coal-fired utility boiler | 4.2E-18 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |
| Oil-fired utility boiler | 1.5E-17 | 3.8E-07 | 100 | 3.8E-07 | 98 | 2 |

[a] Edge of watershed 200 m from source.

## 10.5 HAZARD IDENTIFICATION AND DOSE-RESPONSE FOR ARSENIC

### 10.5.1 Introduction

Inhalation exposure to inorganic arsenic in humans has been strongly associated with lung cancer.[13] Human oral exposure to inorganic arsenic has been associated with an increased risk of several types of cancer in humans, including skin, bladder, liver, and lung cancer.[13,45] Oral exposure to inorganic arsenic has also been associated with noncancer effects, including effects to the central nervous system, cardiovascular system, gastrointestinal tract, liver, kidney, and blood.[44] Appendix E of this report contains more information on health effects of arsenic. Chapter 4 contains general information on terminology, definitions, and guidelines related to risk assessments. A short summary of the health effects of arsenic is presented here.

Arsenic in both the trivalent (+3) and pentavalent (+5) oxidation states may form both inorganic and organic compounds. Both trivalent and pentavalent species of inorganic arsenic may be taken up by humans from the gut (as can organic forms); these forms may be found in urine (after exposure), along with monomethylated arsenic and dimethylated arsenic. When ingested by humans, pentavalent forms are reduced to trivalent arsenic which is then subject to methylation-forming monomethyl- and dimethyl arsenic. Trivalent forms appear to be more toxic than pentavalent forms. Methylation was thought to result in detoxification; however, this assumption has been called into question recently. Despite its toxicity, there is some (weak) evidence to indicate that arsenic is an essential element in the human diet.[5]

The toxicological data on organic arsenic are limited. In fish and shellfish, organic arsenic is absorbed through the gastro-intestinal tract and excreted rapidly (60-70 percent within a few days) in the urine.[5] However, since the inorganic arsenic forms appear to be of primary toxicological significance in humans, the following discussion of health effects of arsenic and the risk characterization focus only on inorganic arsenic.

### 10.5.2 Cancer Effects of Arsenic

There is clear evidence that chronic exposure to inorganic arsenic in humans increases the risk of cancer. Various studies of humans have reported that inhalation of arsenic results in an increased risk of lung cancer. In addition, ingestion of arsenic has been associated with an increased risk of non-melanoma skin cancer and bladder, liver, and lung cancer. Animal studies have not clearly associated arsenic exposure, via ingestion exposure, with cancer. No studies have investigated the risk of cancer in animals as a result of inhalation or dermal exposure.[46]

EPA has classified inorganic arsenic in Group A - Known Human Carcinogen. The Group A classification was based on the increased incidence in humans of lung cancer through inhalation exposure and the

increased risk of skin, bladder, liver, and lung cancer through drinking water exposure.[45]

10.5.2.1 Inhalation Cancer Risk for Arsenic. EPA used the absolute-risk linear extrapolation model to estimate the inhalation unit risk for inorganic arsenic. Five studies on arsenic-exposed copper smelter workers were modeled for excess cancer risk. Using the geometric mean of these data, EPA calculated an inhalation unit risk estimate (IURE) of $4.29 \times 10^{-3}$ per ($\mu g/m^3$). The IURE is an upper-bound estimate of the increased probability of a person developing cancer from breathing air containing a concentration of 1 $\mu$g/m3 of air for 70 years.[47] EPA has high confidence in the arsenic cancer IURE because the studies examined a large number of people, the exposure assessments included air measurements and urinary arsenic measurements, and lung cancer incidence was significantly increased over expected values.[45] The inhalation cancer slope factor is $1.5 \times 10^{+1}$ per (mg/kg/day). The inhalation slope factor is an upper bound estimate of the increased risk of developing cancer due to an average inhalation intake of 1 mg/kg/day of arsenic over a lifetime (70 years).[46]

10.5.2.2 Oral Cancer Risk for Arsenic. To estimate the risks posed by ingesting arsenic, EPA obtained data in Taiwan concerning skin cancer incidence, age, and level of exposure via drinking water. In 37 villages that had obtained drinking water for 45 years from artesian wells with various elevated levels of arsenic, 40,421 individuals were examined for hyperpigmentation, keratosis, skin cancer, and blackfoot disease (gangrene of the extremities caused by injury to the peripheral vasculature). The local well waters were analyzed for arsenic, and the age-specific cancer prevalence rates were found to correlate with both local arsenic concentrations and age (duration of exposure). Based on these data, although EPA has not presented the calculations for the oral unit risk estimate for arsenic, they did propose that a unit risk estimate of $5 \times 10^{-5}$ $(\mu g/L)^{-1}$ from oral exposure to arsenic in drinking water be used.[45] This equates to an oral cancer slope factor of 1.5E + 00 per (mg/kg/day). The oral cancer slope factor is an upper bound estimate of the increased risk of developing cancer due to an average oral intake of 1 mg/kg/day of arsenic over a lifetime (70 years).[45,46]

The Taiwan cancer data have several limitations: (1) the water was contaminated with substances such as bacteria and ergot alkaloids, in addition to arsenic; (2) total arsenic exposure was uncertain because of intake from the diet and other sources; (3) early deaths from blackfoot disease may have led to an underestimate of prevalence; and (4) there was uncertainty concerning exposure durations. Due to these limitations, and also because the diet, economic status, and mobility of individuals in Taiwan are different from those of most United States citizens, EPA has stated that "the uncertainties associated with ingested inorganic arsenic are such that estimates could be modified downwards as much as an order of magnitude, relative to risk estimates associated with most other carcinogens."[45]

### 10.5.3 Noncancer Effects of Arsenic

10.5.3.1 Chronic (Long-Term) Effects for Arsenic. The primary noncancer effects noted in humans from chronic exposure to arsenic, through both inhalation and oral exposure, are effects on the skin. The inhalation route has resulted primarily in irritation of the skin and mucous membranes (dermatitis, conjunctivitis, pharyngitis, and rhinitis), while chronic oral exposure has resulted in a pattern of skin changes that include the formation of warts or corns on the palms and soles along with areas of darkened skin on the face, neck, and back. Other effects noted from chronic oral exposure include peripheral neuropathy, cardiovascular disorders, liver and kidney disorders, and blackfoot disease.[45]

EPA has established an RfD for inorganic arsenic of 0.0003 mg/kg/day, based on a NOAEL (adjusted to include arsenic exposure from food) of 0.0008 mg/kg/day, an uncertainty factor of 3, and a modifying factor of 1.[45] The EPA has not established a RfC for inorganic arsenic.[45]

## 10.6 RISK CHARACTERIZATION

In this section, the information on hazard, dose-response, and exposure are combined to characterize the potential risks due to arsenic emissions from the model utility plants. As stated above, inorganic arsenic is the form of arsenic considered to be of primary concern for causing adverse health effects. Therefore, risks and hazards have been estimated only for the inorganic arsenic.

Increased cancer risk for each hypothetical person, under each hypothetical scenario, for four different model plants placed in two locations has been estimated. The general method of estimating the increased risk of cancer due to multipathway exposure to inorganic arsenic, is to multiply the predicted intakes (mg/kg/day) from the multipathway exposure modeling by the EPA's Oral Cancer Potency Factor for inorganic arsenic (1.51E+00 per [mg/kg]/day) and then to multiply by 30/70 to adjust for the assumed duration of exposure. The potency factor is based on lifetime exposure (i.e., 70 years). For all of the exposure scenarios, except for the pica child, it is assumed that the hypothetical person is exposed for 30 years. A 30-year exposure assumption is considered appropriate for most exposure scenarios considered in this analysis. However, for the pica child, it is assumed that exposure only occurs for 7 years (i.e., duration adjustment factor was 7/70), since it highly unlikely that the pica child would continue pica behavior for more than 7 years. (Note: the 7-year assumption may be high for pica behavior; however, other exposures that may occur as the child gets older such as through food consumption were not considered. Therefore, the 7-year assumption may be a reasonable assumption for this screening assessment.) For the subsistence fisher child and subsistence farmer child, a 30-year exposure is assumed because it is quite possible that the child will continue these consumption behaviors as they become adults. The doses

may decrease some because of the changes in body size, however, the consumption rate is also expected to increase as they become adults.

A 30-year exposure duration was assumed for the inhalation risk calculations in this multipathway arsenic assessment. To estimate the increased cancer risk due to inhalation exposure the predicted intakes via inhalation (in mg/kg/day) were multiplied by the inhalation cancer slope factor (1.5E+01 per [mg/kg]/day) and then multiplied by 30/70 to adjust for the assumed duration of exposure.

To estimate the noncancer hazard, the predicted intakes are compared to the EPA's oral RfD (3E-04 mg/kg/day). The estimated intakes in mg/kg/day, were divided by the RfD to calculate a hazard quotient (HQ). Therefore, a HQ greater than 1.0 indicates exceedance of the RfD.

It is important to remember that this is a screening level multipathway analysis. The analysis is based on the use of model plants placed in hypothetical locations, as well as the use of hypothetical exposure scenarios and various assumptions, to predict exposure to arsenic emitted by the model plant. The results give some indication of the potential hazards and risks that may occur due to emissions from an electric utility plant. However, the results are not applicable to any particular plant. There are uncertainties and limitations to the analyses and results. The risk estimates could be overestimates or underestimates of the true risk for any particular facility. There are uncertainties and variabilities in the modeling inputs and results, exposure estimates, cancer potency estimates, RfD, and overall risk estimates. The risk estimates presented below are predicted values based on hypothetical scenarios and are intended for screening purposes only. The results are shown in Tables 10-34 through 10-41.

### 10.6.1 Discussion of Cancer Risk Assessment Results

This analysis of multipathway exposures to arsenic emissions is a screening analysis. Thus, these quantitive exposure and risk results, because of the many modeling and analytic uncertainties, are very uncertain and do not, therefore, conclusively demonstrate the existence of health risks of concern associated with exposures to utility emissions either on a national scale or from any actual individual utility. The lack of measured data around these sources preclude a comparison with modeled results. These results do suggest that exposures and risks of concern cannot at present be ruled out and that there is a need for development of additional scientific information to evaluate whether risk levels of concern may exist.

The cancer risks due to multipathway exposures to inorganic arsenic from utility emissions alone (no background) are estimated to be no greater than approximately $1 \times 10^{-4}$ (for pica child) based on this screening level analysis. The large coal-fired utility at the eastern humid site was estimated to pose this highest risk for the pica child. Considering background exposures alone, the risk for the pica child is estimated to be as high as $3 \times 10^{-4}$.

Table 10-34. Inorganic Arsenic Intake, Predicted Cancer Risk, and Noncancer Hazard Quotient (HQ) for Pica Child[a]

| | Facility only | | | Facility +background | | | Facility + background + RELMAP90th | | |
|---|---|---|---|---|---|---|---|---|---|
| Electric utility model Facility | Inorganic arsenic intake (mg/kg/d) | Estimated increased cancer risk | Non-cancer HQ | Inorganic arsenic intake (mg/kg/d) | Estimated increased cancer risk | Non-cancer HQ | Inorganic arsenic intake (mg/kg/d) | Estimated increased cancer risk | Non-cancer HQ |
| *Eastern site (humid)* | | | | | | | | | |
| Large coal-fired boiler | 7.3E-04 | 1E-04 | 2.4 | 2.1E-03 | 3E-04 | 7 | 2.1E-03 | 3E-04 | 7 |
| Medium coal-fired boiler | 1.8E-04 | 3E-05 | 0.6 | 1.5E-03 | 2E-04 | 5 | 1.5E-03 | 2E-04 | 5 |
| Small coal-fired boiler | 7.7E-05 | 1E-05 | 0.3 | 1.4E-03 | 2E-04 | 5 | 1.4E-03 | 2E-04 | 5 |
| Oil-fired boiler | 8.8E-05 | 1E-05 | 0.3 | 1.4E-03 | 2E-04 | 5 | 1.4E-03 | 2E-04 | 5 |
| *Western site (arid)* | | | | | | | | | |
| Large coal-fired boiler | 5.3E-05 | 8E-06 | 0.2 | 1.4E-03 | 2E-04 | 5 | 1.4E-03 | 2E-04 | 5 |
| Medium coal-fired boiler | 3.5E-05 | 5E-06 | 0.1 | 1.4E-03 | 2E-04 | 5 | 1.4E-03 | 2E-04 | 5 |
| Small coal-fired boiler | 1.6E-05 | 2E-06 | 0.05 | 1.4E-03 | 2E-04 | 5 | 1.4E-03 | 2E-04 | 5 |
| Oil-fired boiler | 1.7E-05 | 3E-06 | 0.06 | 1.4E-03 | 2E-04 | 5 | 1.4E-03 | 2E-04 | 5 |

[a] Receptor point based on location of maximum deposition.

Table 10-35. Inorganic Arsenic Intake, Predicted Cancer Risk, and Noncancer Hazards for Subsistence Farmer Adult[a]

| | Facility only | | | Facility + background | | | Facility + background + RELMAP90th | | |
|---|---|---|---|---|---|---|---|---|---|
| Electric utility model facility | Inorganic arsenic intake (mg/kg/d) | Estimated increased cancer risk | Non-cancer HQ | Inorganic arsenic intake (mg/kg/d) | Estimated increased cancer risk | Non-cancer HQ | Inorganic arsenic intake (mg/kg/d) | Estimated increased cancer risk | Non-cancer HQ |
| *Eastern site (humid)* | | | | | | | | | |
| Large coal-fired boiler | 5.4E-06 | 4E-06 | 0.03 | 3.7E-04 | 3E-04 | 1.2 | 3.7E-04 | 3E-04 | 1.2 |
| Medium coal-fired boiler | 1.3E-06 | 1E-06 | 0.004 | 3.6E-04 | 2E-04 | 1.2 | 3.6E-04 | 2E-04 | 1.2 |
| Small coal-fired boiler | 5.8E-07 | 4E-07 | 0.002 | 3.6E-04 | 2E-04 | 1.2 | 3.6E-04 | 2E-04 | 1.2 |
| Oil-fired boiler | 6.3E-07 | 5E-07 | 0.003 | 3.6E-04 | 2E-04 | 1.2 | 3.6E-04 | 2E-04 | 1.2 |
| *Western site (arid)* | | | | | | | | | |
| Large coal-fired boiler | 2.3E-05 | 1E-05 | 0.08 | 3.9E-04 | 3E-04 | 1.3 | 3.9E-04 | 3E-04 | 1.3 |
| Medium coal-fired boiler | 5.2E-06 | 4E-06 | 0.02 | 3.7E-04 | 3E-04 | 1.2 | 3.7E-04 | 3E-04 | 1.2 |
| Small coal-fired boiler | 1.8E-06 | 1E-06 | 0.006 | 3.7E-04 | 3E-04 | 1.2 | 3.7E-04 | 3E-04 | 1.2 |
| Oil-fired boiler | 2.3E-06 | 1E-06 | 0.008 | 3.7E-04 | 3E-04 | 1.2 | 3.7E-04 | 3E-04 | 1.2 |

[a] Edge of watershed 200 m from source.

Table 10-36. Inorganic Arsenic Intake, Predicted Cancer Risk, and Noncancer Hazards for Subsistence Farmer Child[a]

| | Facility only | | | Facility + background | | | Facility + background + RELMAP90th | | |
|---|---|---|---|---|---|---|---|---|---|
| **Electric utility model facility** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated Increased cancer risk** | **Non-cancer HQ** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** |
| *Eastern site (humid)* | | | | | | | | | |
| Large coal-fired boiler | 1.0E-05 | 1E-05 | 0.03 | 7.5E-04 | 5E-04 | 2.5 | 7.5E-04 | 5E-04 | 2.5 |
| Medium coal-fired boiler | 2.5E-06 | 2E-06 | 0.008 | 7.5E-04 | 5E-04 | 2.5 | 7.5E-04 | 5E-04 | 2.5 |
| Small coal-fired boiler | 1.1E-06 | 1E-06 | 0.004 | 7.4E-04 | 5E-04 | 2.5 | 7.5E-04 | 5E-04 | 2.5 |
| Oil-fired boiler | 1.2E-06 | 1E-06 | 0.004 | 7.5E-04 | 5E-04 | 2.5 | 7.5E-04 | 5E-04 | 2.5 |
| *Western site (arid)* | | | | | | | | | |
| Large coal-fired boiler | 4.6E-05 | 3E-05 | 0.15 | 8E-04 | 6E-04 | 2.7 | 8.0E-04 | 6E-04 | 2.7 |
| Medium coal-fired boiler | 1.0E-05 | 1E-05 | 0.03 | 7.6E-04 | 5E-04 | 2.5 | 7.6E-04 | 5E-04 | 2.5 |
| Small coal-fired boiler | 3.7E-06 | 3E-06 | 0.01 | 7.5E-04 | 5E-04 | 2.5 | 7.5E-04 | 5E-04 | 2.5 |
| Oil-fired boiler | 4.7E-06 | 3E-06 | 0.02 | 7.5E-04 | 5E-04 | 2.5 | 7.5E-04 | 5E-04 | 2.5 |

[a] Edge of watershed 200 m from source.

Table 37. Inorganic Arsenic Intake, Predicted Cancer Risk, and Noncancer Hazards for Subsistence Fisher Adult[a]

| | Facility only | | | Facility + background | | | Facility + background + RELMAP90th | | |
|---|---|---|---|---|---|---|---|---|---|
| **Electric utility model facility** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** |
| *Eastern site (humid)* | | | | | | | | | |
| Large coal-fired boiler | 5.0E-06 | 4E-06 | 0.02 | 2.7E-04 | 2E-04 | 0.9 | 2.7E-04 | 2E-04 | 0.9 |
| Medium coal-fired boiler | 1.3E-06 | 1E-06 | 0.004 | 2.6E-04 | 2E-04 | 0.9 | 2.6E-04 | 2E-04 | 0.9 |
| Small coal-fired boiler | 5.3E-07 | 4E-07 | 0.002 | 2.6E-04 | 2E-04 | 0.9 | 2.6E-04 | 2E-04 | 0.9 |
| Oil-fired boiler | 5.8E-07 | 4E-07 | 0.002 | 2.6E-04 | 2E-04 | 0.9 | 2.6E-04 | 2E-04 | 0.9 |
| *Western site (arid)* | | | | | | | | | |
| Large coal-fired boiler | 2.3E-05 | 1E-05 | 0.08 | 2.8E-04 | 2E-04 | 0.9 | 2.8E-04 | 2E-04 | 0.9 |
| Medium coal-fired boiler | 5.3E-06 | 4E-06 | 0.02 | 2.6E-04 | 2E-04 | 0.9 | 2.6E-04 | 2E-04 | 0.9 |
| Small coal-fired boiler | 1.9E-06 | 1E-06 | 0.006 | 2.6E-04 | 2E-04 | 0.9 | 2.6E-04 | 2E-04 | 0.9 |
| Oil-fired boiler | 2.4E-06 | 2E-06 | 0.008 | 2.6E-04 | 2E-04 | 0.9 | 2.6E-04 | 2E-04 | 0.9 |

[a] Edge of watershed 200 m from source.

Table 10-38. Inorganic Arsenic Intake, Predicted Cancer Risk, and Noncancer Hazards for Subsistence Fisher Child[a]

| | Facility only | | | Facility + background | | | Facility + background + RELMAP90th | | |
|---|---|---|---|---|---|---|---|---|---|
| **Electric utility model facility** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** | **Inorganic Arsenic Intake (mg/kg/d)** | **Estimated increased cancer risk** | **Non-cancer HQ** |
| *Eastern site (humid)* | | | | | | | | | |
| Large coal-fired boiler | 9.3E-06 | 7E-06 | 0.03 | 5.1E-04 | 4E-04 | 1.7 | 5.1E-04 | 4E-04 | 1.7 |
| Medium coal-fired boiler | 2.3E-06 | 2E-06 | 0.008 | 5.0E-04 | 4E-04 | 1.7 | 5.0E-04 | 4E-04 | 1.7 |
| Small coal-fired boiler | 9.7E-07 | 7E-07 | 0.003 | 5.0E-04 | 4E-04 | 1.7 | 5.0E-04 | 4E-04 | 1.7 |
| Oil-fired boiler | 1.1E-06 | 1E-06 | 0.004 | 5.0E-04 | 4E-04 | 1.7 | 5.0E-04 | 4E-04 | 1.7 |
| *Western site (arid)* | | | | | | | | | |
| Large coal-fired boiler | 4.7E-05 | 3E-05 | 0.16 | 5.5E-04 | 4E-04 | 1.7 | 5.5E-04 | 4E-04 | 1.7 |
| Medium coal-fired boiler | 1.1E-05 | 1E-05 | 0.04 | 5.1E-04 | 4E-04 | 1.7 | 5.1E-04 | 4E-04 | 1.7 |
| Small coal-fired boiler | 3.7E-06 | 3E-06 | 0.01 | 5.1E-04 | 4E-04 | 1.7 | 5.1E-04 | 4E-04 | 1.7 |
| Oil-fired boiler | 4.8E-06 | 3E-06 | 0.02 | 5.1E-04 | 4E-04 | 1.7 | 5.1E-04 | 4E-04 | 1.7 |

[a] Edge of watershed 200 m from source.

Table 10-39. Inorganic Arsenic Intake via Inhalation and Predicted Cancer Risks for Pica Child[a]

| | Facility only | | Facility + background | | Facility + background + RELMAP90th | |
|---|---|---|---|---|---|---|
| **Electric utility model facility** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired boiler | 4.1E-21 | 3E-20 | 3.8E-07 | 3E-06 | 4.1E-07 | 3E-06 |
| Medium coal-fired boiler | 1.7E-18 | 1E-17 | 3.8E-07 | 3E-06 | 4.1E-07 | 3E-06 |
| Small coal-fired boiler | 1.7E-16 | 1E-15 | 3.8E-07 | 3E-06 | 4.1E-07 | 3E-06 |
| Oil-fired boiler | 8.9E-16 | 7E-15 | 3.8E-07 | 3E-06 | 4.1E-07 | 3E-06 |
| *Western site (arid)* | | | | | | |
| Large coal-fired boiler | 4.9E-18 | 4E-17 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |
| Medium coal-fired boiler | 8.6E-24 | 7E-23 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |
| Small coal-fired boiler | 6.8E-19 | 6E-18 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |
| Oil-fired boiler | 1.9E-18 | 1E-17 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |

[a] Receptor point based on location of maximum deposition.

Table 10-40. Inorganic Arsenic Intake via Inhalation for Subsistence Farmer Adult and Subsistence Fisher Adult and Predicted Cancer Risks[a]

| | Facility only | | Facility + background | | Facility + background + RELMAP90th | |
|---|---|---|---|---|---|---|
| **Electric utility model facility** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired boiler | 6.2E-10 | 4E-09 | 1.1E-07 | 1E-06 | 1.3E-07 | 1E-06 |
| Medium coal-fired boiler | 4.1E-10 | 3E-09 | 1.1E-07 | 1E-06 | 1.3E-07 | 1E-06 |
| Small coal-fired boiler | 6.4E-10 | 5E-09 | 1.1E-07 | 1E-06 | 1.3E-07 | 1E-06 |
| Oil-fired boiler | 5.7E-10 | 4E-09 | 1.1E-07 | 1E-06 | 1.3E-07 | 1E-06 |
| *Western site (arid)* | | | | | | |
| Large coal-fired boiler | 0.0E+00 | 0.0 | 1.1E-07 | 1E-06 | 1.2E-07 | 1E-06 |
| Medium Coal-fired boiler | 4.3E-23 | 3E-22 | 1.1E-07 | 1E-06 | 1.2E-07 | 1E-06 |
| Small coal-fired boiler | 1.3E-18 | 1E-17 | 1.1E-07 | 1E-06 | 1.2E-07 | 1E-06 |
| Oil-fired Boiler | 4.7E-18 | 3E-17 | 1.1E-07 | 1E-06 | 1.2E-07 | 1E-06 |

[a] Edge of watershed 200 m from source.

Table 10-41. Inorganic Arsenic Intake via Inhalation for Subsistence Farmer Child and Predicted Cancer Risks[a]

| | Facility only | | Facility + background | | Facility + background + RELMAP90th | |
|---|---|---|---|---|---|---|
| **Electric utility model facility** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** | **Inorganic arsenic intake (mg/kg/d)** | **Estimated increased cancer risk** |
| *Eastern site (humid)* | | | | | | |
| Large coal-fired boiler | 2.0E-09 | 1E-08 | 3.8E-07 | 3E-06 | 4.2E-07 | 3E-06 |
| Medium coal-fired boiler | 1.4E-09 | 1E-08 | 3.8E-07 | 3E-06 | 4.1E-07 | 3E-06 |
| Small coal-fired boiler | 2.1E-09 | 1E-08 | 3.8E-07 | 3E-06 | 4.2E-07 | 3E-06 |
| Oil-fired boiler | 1.9E-09 | 1E-08 | 3.8E-07 | 3E-06 | 4.2E-07 | 3E-06 |
| *Western site (arid)* | | | | | | |
| Large coal-fired boiler | 0.0E+00 | 0 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |
| Medium coal-fired boiler | 1.4E-22 | 1E-21 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |
| Small coal-fired boiler | 4.2E-18 | 3E-17 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |
| Oil-fired boiler | 1.5E-17 | 1E-16 | 3.8E-07 | 3E-06 | 3.8E-07 | 3E-06 |

[a] Edge of watershed 200 m from source) and predicted cancer risks.

The cancer risks due to multipathway exposures to inorganic arsenic from utility emissions alone (no background) are estimated to be no greater than approximately $1 \times 10^{-4}$ (for pica child) based on this screening level analysis. The large coal-fired utility at the eastern humid site was estimated to pose this highest risk for the pica child. Considering background exposures alone, the risk for the pica child is estimated to be as high as $3 \times 10^{-4}$.

The estimated cancer risks for the subsistence farmer adult and subsistence fisher adult are estimated to be as high as $1 \times 10^{-5}$ and the estimated cancer risks for the subsistence farmer child and subsistence fisher child are estimated to be as high as $3 \times 10^{-5}$. In all scenarios, it was the large coal-fired utility that was estimated to pose the greatest multipathway risks, the medium coal-fired utility was estimated to pose the next highest risks, and the small coal-fired utility and the oil-fired utility were estimated to present very similar magnitudes of exposures and risks. Background exposures were estimated to dominate the exposures and risks in all scenarios. Including background exposures increases the risks by approximately 2 to 200 times, depending on the receptor and plant scenario analyzed.

Inhalation risks are predicted to be significantly lower than the risks from multipathway exposures. The highest inhalation risks due to the model utility plant emissions only (no background) were estimated to be $1 \times 10^{-8}$ for the hypothetical receptors in this analysis. The RELMAP results contributed slightly (approximately 9 percent) to the total inhalation exposures and risks. Similar to multipathway exposures, the inhalation exposures to background were estimated to be substantially higher than the inhalation exposures due to utilities emissions based on this model plant analysis. However, it is important to recall that the estimated risks for inhalation exposures to arsenic from the HEM analyses, described in chapter 6 (in which every actual plant [684 plants] in the United States was modeled), were estimated to be up to $3 \times 10^{-6}$, which is about 300 times higher than the inhalation risks predicted in this model plant analysis. This may be due to several factors. One likely factor is that many more plants were modeled (684 actual plants, including some outliers instead of 4 model plants) for the HEM analyses. Also, in the HEM analyses, 349 meteorological sites were included, whereas only 2 meteorological stations were included in the multipathway analysis. In addition, 70-year exposures were assumed for the HEM analyses; 30-year exposures were the primary assumption used for the multipathway analysis. Moreover, the multipathway analysis was intended to predict potential high-end exposures due to multipathway exposures; therefore, the distances from plant were chosen that would likely result in high-end multipathway exposures. The distances chosen for this multipathway analysis were mainly based on the locations of predicted high deposition rates, and not based on high air concentrations. The 200-meter distance from stack used for most of the scenarios is estimated to be a location of high deposition; however, this distance is not likely to be the distance of highest air concentrations. The distances of high air concentrations are likely

to be much further from the stack. Therefore, it is not expected that this model plant analysis would predict inhalation exposures and risks as high as the HEM analysis.

10.6.2 Discussion of the Noncancer Risk Assessment Results

The highest predicted intake of arsenic due to utility emissions only was 7E-04 mg/kg/day (predicted for the pica child/large coal-fired utility/eastern site scenario), which is 2.4 times higher than the RfD (3E-04 mg/kg/day) and equates to an HQ of 2.4. All other scenarios had predicted HQs less than 1.0 when considering utility emissions only. When background is included, the HQs range from 0.9 to 7. These results suggest that adverse noncancer effects due to utility emissions alone could possibly be of concern for the pica child. For all other scenarios analyzed, it appears that utility emissions alone are not likely to be of concern for noncancer effects. However, exposures to current background levels of arsenic, as well as exposures to current background levels combined with utility emissions of arsenic, could be a potential concern for adverse noncancer effects.

10.7 CONCLUSIONS

10.7.1 Contribution of Arsenic Emissions from Utilities to Concentrations in Environmental Media and Biota.

The facilities considered were found to contribute less than 10 percent of the total predicted arsenic concentrations in soil, water and all biota when a background soil concentration of 3 $\mu$g/g and air concentration of 0.7 ng/m$^3$ were assumed, and area-averaged deposition rates were used for a watershed 200 m downwind from the facility. Soil and predicted water concentrations are dominated (e.g., >95 percent) by background when background is included.

10.7.2 Determination of Dominant Pathways of Potential Exposure to Anthropogenic Arsenic Emissions

Using the hypothetical exposure scenarios discussed above, it was found that exposure to inorganic arsenic through ingestion was mainly through the ingestion of grain. This result holds true whether or not anthropogenic sources are considered because there is not sufficient evidence to support the assumption that the anthropogenic arsenic varied chemically from that already in the environment. The highest dietary exposures to total arsenic for adults not consuming marine organisms or freshwater bivalves are the result of consumption of freshwater fish and grains.

Exposure to inorganic arsenic through the ingestion of fish was not predicted to be a major pathway of exposure because there is considerable evidence that little of the total arsenic in fish tissue is inorganic arsenic. For the subsistence fisher scenario considered, exposure to inorganic arsenic from grain was larger than that from fish, even though a high fish consumption rate was assumed.

Exposure to arsenic through the ingestion of unusually high amounts of soil (e.g., pica) was shown to be of potential concern. Dominant exposure for the pica child is from background soil concentrations of arsenic, when background levels are considered.

Since this is a screening level analysis, further detailed analyses are needed to better characterize the risks posed.

## 10.8. UNCERTAINTIES AND LIMITATIONS

### 10.8.1 Limitations and Uncertainties for the Multipathway Exposure Modeling

The following are uncertainties and limitations associated with this assessment that may affect the results.

If a location is such that the background concentrations are significantly lower than those assumed in the assessment, then the contribution of any particular individual source to the total arsenic concentrations and subsequent exposure will be larger than that determined here. However, this will also reduce the total arsenic exposure. If the background concentrations are significantly higher than that assumed here, then although the contribution of a source to the total is less, the additional contribution may result in the exceedance of certain toxic thresholds. At this time, there is little guidance on the determination of or incorporation of background concentrations in an assessment. For chemicals which have a threshold of action, inclusion of an appropriate estimate of background is critical in the determination of the impacts to public health.

There are few data on the speciation or bioavailability of arsenic in environmental media or anthropogenic emissions. In this assessment, it was assumed that the arsenic emitted from utilities is similar to that reported in the literature for various media. If the form of the emitted arsenic is significantly different, then this can have important impacts on the assessment results and conclusions.

The bioavailability of anthropogenic arsenic is assumed to be the same as the bioavailability of arsenic that in the environment in this assessment. If this is not true, then this could impact the conclusions reached. More research is needed on the bioavailability of anthropogenic arsenic.

An important uncertainty when identifying a dominant pathway is the range of pathways which are considered. For example, not considered here was exposure to arsenic through the ingestion of tap or well water. Inclusion of this pathway would increase the primary or background exposure (depending on the level of exposure), and hence reduce the relative contribution of any particular source.

### 10.8.2 Limitations and Uncertainties for the Risk Characterization

As discussed in section 10.5 above and in Appendix E, there are limitations in the health effects data for arsenic. For example,

although human data are available on the carcinogenicity of arsenic, there are no animal studies available on the carcinogenicity of arsenic from inhalation exposure and the animal ingestion studies have not clearly shown an association between arsenic ingestion exposure and cancer.

The oral cancer potency factor (1.5 per mg/kg/day) is a reasonable estimate based on available information. However, the available toxicity data are limited and uncertain; hence the cancer potency estimate is somewhat uncertain. Also, because of differences in genetics, diet, lifestyle, stress management, and a number of other host factors, the human population is likely to have considerable variation in individual sensitivities to developing cancer per unit of exposure. Therefore, although the oral potency factor may be a reasonable estimate, it is not likely to accurately represent the potency for all humans. Many humans may be either more or less sensitive to developing cancer, than predicted by the oral potency factor.

The human exposure estimates presented above are uncertain due to modeling uncertainties (described above), and also because of human variability and uncertainties associated with several factors such as breathing rates, food consumption rates, activity patterns, body size, metabolic differences, and other factors. Variation and uncertainty in meteorology, environmental fate of arsenic, bioaccumulation rates, and other factors also contribute to the overall uncertainty in the risks estimates. Therefore, the risk estimates presented above should be viewed as reasonable estimates for a screening level assessment, but should also be viewed as containing significant uncertainties. Further analyses are needed to more fully assess the potential risks due to arsenic exposure from utilities.

## 10.9 RESEARCH NEEDS

To improve the multimedia, multipathway exposure assessment for arsenic, additional data on the speciation of arsenic in air, soil, and water are needed. Better information on arsenic bioavailability is also needed. In addition, research to improve the understanding of watershed dynamics would benefit this study, as well as other efforts to model pollutant fate and transport. Additional health effects data are also needed, especially for ingestion exposure route, for cancer and noncancer effects, to better characterize the hazards of arsenic exposures.

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# 11.0 A MULTIPATHWAY SCREENING-LEVEL ASSESSMENT FOR DIOXINS/FURANS

## 11.1 INTRODUCTION

Polychlorinated dibenzo-p-dioxin (PCDD) and dibenzofuran (PCDF) emissions were identified as a priority for multipathway exposure analysis (see chapter 5). PCDDs and PCDFs, which will be referred to collectively as dioxins, are ubiquitous in the environment.[1] 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 (congeners) are probable human carcinogens.[1] 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.[1]

The occurrence of dioxin and dioxin-like compounds in the environment appears to be primarily the result of human activities.[1,2] The national estimated loading of these compounds from identified sources into the environment is approximately 12,000 g toxicity equivalents (TEQ)/yr. When this loading of 12,000 g TEQ/yr is compared to annual loadings of other hazardous air pollutants (HAPs), it appears to be relatively low. However, small quantities of dioxin emissions can be of concern because of the high toxicity and persistence, and tendency to bioaccumulate.[1] 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 95 g TEQ/yr, dioxin emissions from utilities represent about 1 percent of total dioxin emissions. However, there are substantial uncertainties associated with the dioxin emissions estimates due to several factors including the following: emissions test data were available from only 13 utility units; data were not collected from all types of facilities (e.g., no data were collected from units with hot-side ESPs); and many measurements were below the minimum detection limit. Therefore, the estimated emissions for the model plants described below are also uncertain. The actual dioxin emissions could be greater than, or less than, the predicted emissions presented in this chapter and other chapters of this report.

There are several hypotheses on the chemistry and conditions for the formation of PCDDs and PCDFs, including (1) contaminated feed stock, (2) formation from precursors, and (3) formation *de novo*.[1] The contaminated feedstock theory suggests that PCDDs and PCDFs in the feed material survive the combustion process to become emissions. However, this theory is not thought to be the principle explanation for PCDD/PCDF formation. The second theory states that the formation of PCDDs/PCDFs from precursors that are structurally related to PCDDs/PCDFs occurs through thermal breakdown and rearrangement during the combustion process. Precursors referred to in this theory include

polychlorinated biphenyls, chlorinated phenols, and chlorinated benzenes. This process is believed to occur when the precursor condenses and adsorbs to the surface of fly ash particles in the temperature range of 250-450° C. The third theory on the chemistry and conditions of PCDD/PCDF formation states that creation is from moieties that bear little structural resemblance to the PCDDs/PCDFs. Such compounds may include petroleum products, chlorinated plastics, non-chlorinated plastics, cellulose, lignin, coke, coal, particulate carbon, and hydrogen chloride gas. This *de novo* formation is believed to occur in the same temperature range used in the precursor theory, but occurs as a reaction between a chlorine donor and another molecule forms a chemical intermediate that then serves as a precursor in the formation of PCDDs and PCDFs.

Emission data for various congeners of dioxins and furans were obtained from 10 coal- and 2 oil-fired units using EPA Reference Method 23. Eleven of these 12 tests resulted in one or more congeners being identified at blank-corrected values above the minimum detection level (MDL) for each individual test (there were usually three replicate runs per test). Of the 188 sets of replicate analyses for the various congeners, 109 (58 percent) had data values above the MDL for all replicates. The dioxin/furan testing conducted at utility boilers typically made use of field blanks, laboratory blanks, calibration spiking, and strict measurement criteria during the analyses.

As a larger portion of the dioxin/furan data sets were below the MDL compared to other analyzed constituents, a summary of the protocol for dealing with non-detect values is presented here. (See Appendix D for further details.) Consistent with the methodology used for other constituents, for those data sets containing at least one value above the MDL, a run average was obtained by using one-half the MDL for those values below the MDL in the averaging analysis. If all three data values for a data set were below the MDL, that data set was not used in the analyses. Using this approach resulted in a nationwide dioxin emissions estimate of 95 g/yr TEQ for utilities.

There are other methodologies for handling non-detect values in analyses. Reference Method 23, written for use on MWCs, indicates that non-detects are to be handled as zero in the calculations. Use of the Method 23 approach results in a nationwide dioxin/furan emissions estimate for utility plants of 58 g/yr TEQ, which is roughly 2 times lower than the nationwide estimate of 95 g/yr TEQ reported above. The EPA believes that the approach taken (i.e., using one-half the MDL in the analyses) is appropriate for this analysis.

It has been hypothesized that the primary mechanism by which dioxins enter the terrestrial food chain is through atmospheric deposition.[1] The PCDDs and PCDFs 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 be

attributed to their stability under most environmental conditions and also to the great number of sources located throughout the United States. 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, PCDDs and PCDFs remain adsorbed to particulate matter, 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 sedimentation and, ultimately, burial of sediments.

Once PCDDs and PCDFs are deposited and make their way into various media and into biota, they are available for human uptake through ingestion. The draft Dioxin Reassessment report 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/day.[3] This value is more than 500-fold higher than the EPA's 1985 risk-specific dose of 0.006 pg TEQ/kg body weight/day associated with an upper-bound risk of 1 in a million ($1 \times 10^{-6}$) and is several hundredfold higher than the revised risk-specific dose estimates presented in the draft Dioxin Reassessment report.[1] An inhalation exposure assessment for utility emissions was performed for priority HAPs using the human exposure model (HEM), as described in chapter 6 of this report. The cancer risk to the maximum exposed individual in that assessment was estimated to be $1 \times 10^{-7}$.

As part of the draft dioxin reassessment, methodologies for conducting site-specific indirect exposure modeling are presented.[3] 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 the 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 dioxins 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.[1] This is of significance since the uptake of plants by foraging animals, including cattle, 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/day, 20 to 60 times higher than the estimated range for background exposure to adults (1 to 3 pg of TEQ/kg/day).

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., fish, 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.

Dioxins were identified (in chapter 5) as high priority for multipathway risk assessment. This chapter presents a screening-level assessment of the multipathway exposures and risks associated with

dioxins. The local-scale dioxin and furan emissions estimates used in this screening-level assessment were from four model plant fossil-fuel-fired utility boilers, which were developed for this analysis. The cumulative effects of long-range transport of utility boiler PCDD/PCDF emissions are also considered in the screening-level analysis. The model plants were developed to represent actual utility coal- and oil-fired boilers, and were designed to characterize potential emission rates and atmospheric release processes for average large and small boilers of each fuel type. The assessment presented in this report used long-range and local atmospheric transport and deposition modeling and direct and indirect multipathway exposure modeling to predict cancer risks from inhalation and ingestion of dioxin- and furan-contaminated air, water, soil, and food.

Polychlorinated dioxins and furans may also cause noncancer health effects in humans. Developmental, reproductive, and immune system endpoints have been reviewed as part of the dioxin reassessment effort.[1] However, reference concentrations have not been established by EPA for use in human risk assessment. As a result, only the cancer endpoints have been examined in this study. Additionally, the breast milk exposure pathway for infants has been recognized as a significant source of exposure, as described above. However, quantitative assessment of this pathway is not included in this analysis.

This analysis of noninhalation exposures to dioxin emissions is a screening analysis. Thus, these quantitative exposure and risk results, because of the many modeling and analytic uncertainties, are very uncertain and do not conclusively demonstrate the existence of health risks of concern associated with exposures to utility emissions either on a national scale or from any actual individual utility. The lack of measured data around these sources precludes a comparison with modeled results. These results do suggest that exposures and risks of concern cannot at present be ruled out and that there is a need for development of additional scientific information to evaluate whether risk levels of concern may exist.

EPA's Industrial Source Complex Short Term, Version 3 (ISCST3) model was used to estimate individual congener ambient air concentrations and wet/dry deposition rates associated with emissions from each of the model plants.[4] The ISCST3 uses Gaussian dispersion to calculate air concentration and deposition in the local area (i.e., within 50 km of the emissions source). Particle size, atmospheric conditions and gravitational settling velocities are used to predict dry deposition. Precipitation rate and particle size are used to estimate scavenging coefficients which are used to predict wet deposition.

Long-range transport of dioxins emitted from multiple power plants is also a potential cause of increased population risk. Therefore, long-range modeling of utility dioxin emissions using the RELMAP model are summarized below for consideration in the screening-level multipathway exposure assessment.

After the air concentrations and deposition rates were modeled, multipathway exposures were estimated using a modified version of the Indirect Exposure Methodology (IEM) - Spreadsheet provided by EPA Office of Research and Development. The IEM methodology uses site-specific exposure scenarios and dispersion model predictions of air concentrations and deposition rates to calculate pollutant concentrations in vegetation, soil, water, and the aquatic and terrestrial food chain. The methodology then calculates human exposure from data on ingestion and dermal contact.

The following section (section 11.2) summarizes the RELMAP long-range dispersion modeling for dioxins. Section 11.3 describes the methods and data used in this risk assessment, including emissions sources, local dispersion modeling, and multipathway exposure and risk modeling. In section 11.4 the results of the risk assessment are presented for the different scenarios and pathways considered. Section 11.5 describes how the results were tested for their sensitivity to changes in various assumptions about the model parameters. The final section (section 11.6) provides conclusions on the analysis, limitations, and uncertainties.

## 11.2 LONG-RANGE TRANSPORT MODELING

Long-range atmospheric transport modeling was performed for the continental United States using the RELMAP model. RELMAP modeling allows consideration of emissions contributions from multiple sources on a national scale. More detailed description of the RELMAP model, methods, and uncertainties can be found in chapter 6 of this report. The methods and emissions sources used in the RELMAP modeling of dioxins were patterned after the RELMAP modeling described here in chapter 6. The dioxin RELMAP results described here consider the combined TEQ emissions from all U.S. coal- and oil-fired utilities.

The RELMAP model was used to generate air concentration, wet deposition, and dry deposition results for dioxin emissions for the contiguous 48 states. The results of all three analyses indicate that long-range transport provides higher air concentrations and deposition rates for dioxins in the Eastern United States, with the maximum for each falling in the general region of the Ohio River Valley.

Figure 11-1 presents the RELMAP results for dioxin air concentration. Long-range transport of emissions generated air concentrations ranging from less than 10 to a maximum of 316 attograms/$m^3$, with an average of 19 attograms/$m^3$. Figure 11-2 provides the RELMAP wet deposition results. The results show that long-range transport accounted for wet depositions ranging from less than one to a maximum of 93 picograms per square meter per year (pg/$m^2$/yr). The overall average wet deposition for Figure 11-2 was 25 pg/$m^2$/yr. The dry deposition model results (Figure 11-3) are slightly less than the wet deposition results. The dry deposition ranged from less than one to a maximum deposition of 82 pg/$m^2$/yr, with an average of 24 pg/$m^2$/yr.

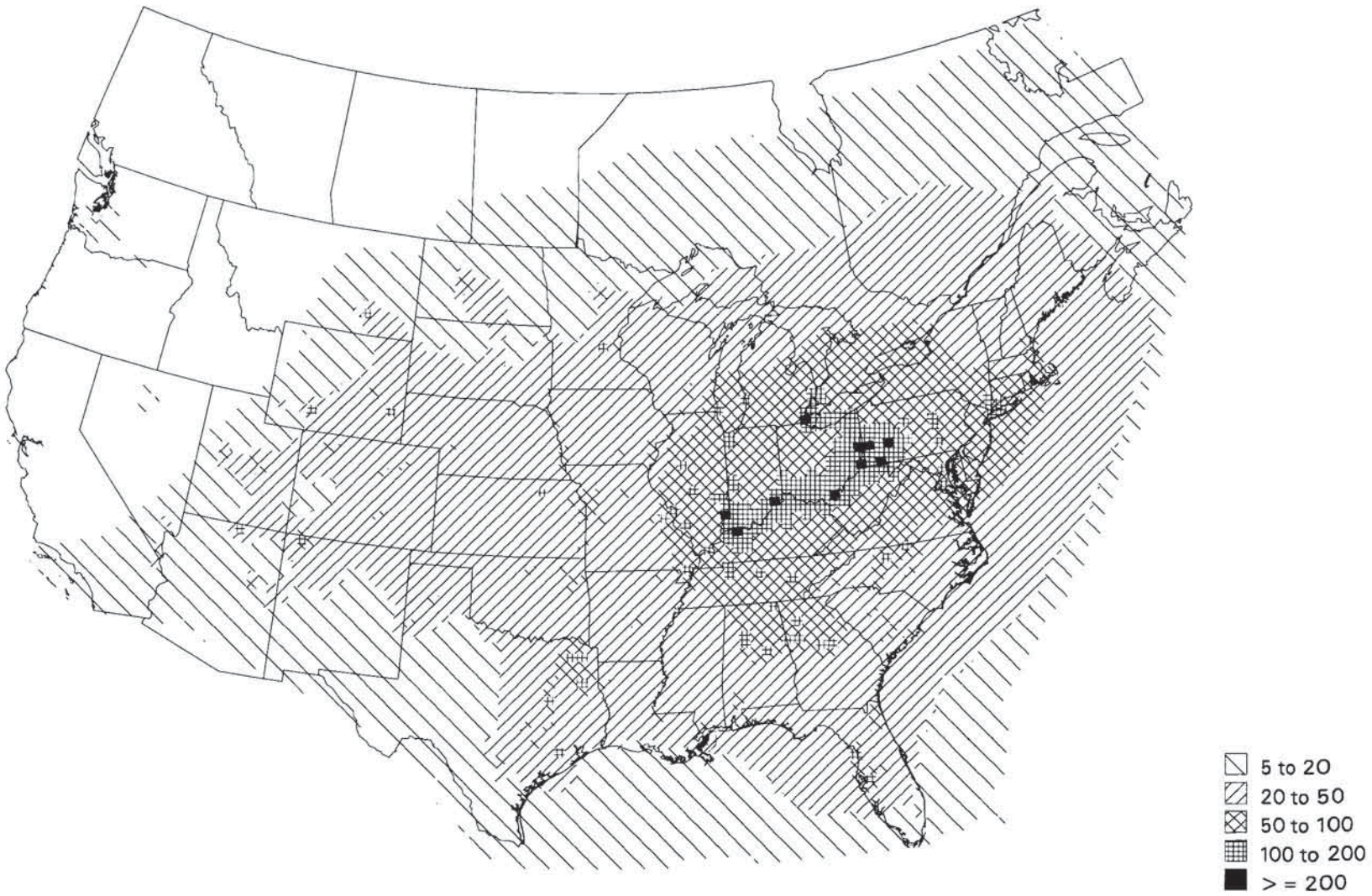



Figure 11-1. Results of the RELMAP Modeling Analysis from 1990 Emissions Estimates for Total Dioxin (incl TEQ factors) from Coal and Oil Utilities: Predicted Air Concentration of Total Dioxin, Units: attograms ($10^{-18}$ grams)/$m^3$



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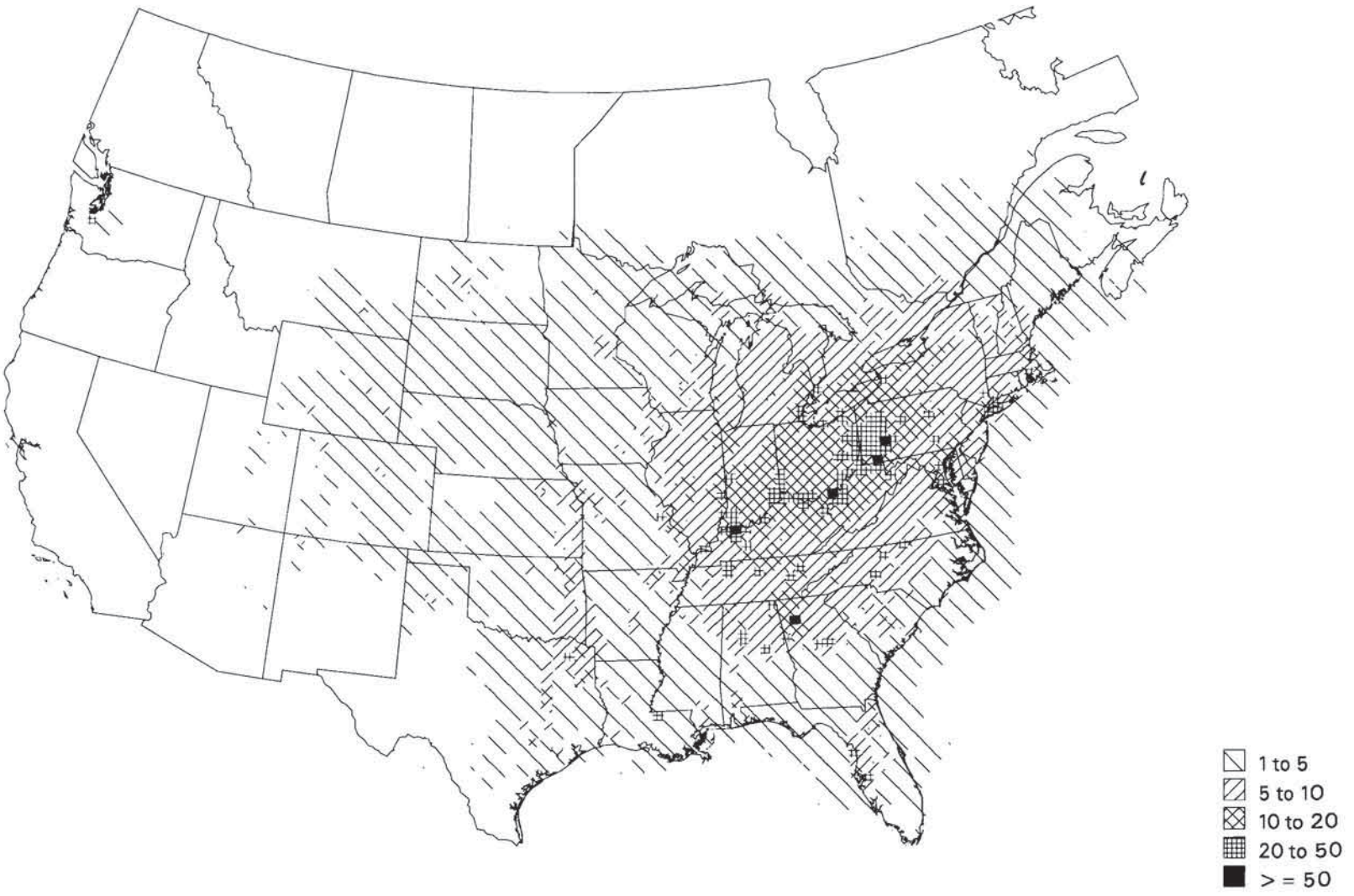



Figure 11-2. Predicted Total Dioxin (incl TEQ factors) Wet Deposition from Coal and Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: picograms ($10^{-12}$ grams)/$m^3$



Exhibit 2024
Page 400

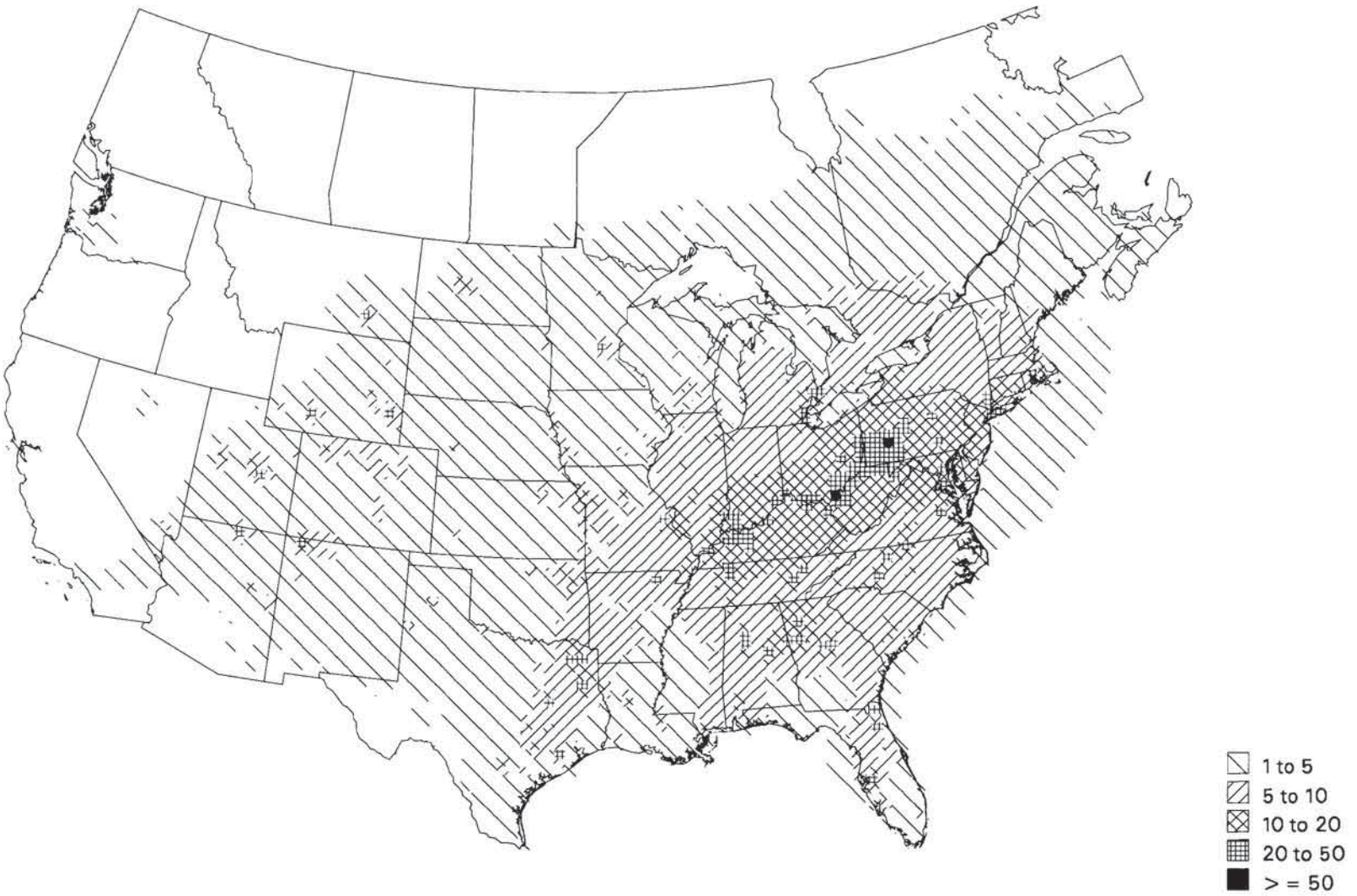



Figure 11-3. Predicted Total Dioxin (incl TEQ factors) Dry Deposition from Coal and Oil Utilities Based on 1990 Emissions Estimates as Modeled with RELMAP, Units: picograms ($10^{-12}$ grams)/$m^3$



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## 11.3 RISK ASSESSMENT METHODOLOGY

### 11.3.1 Emissions Sources

A model plant approach was used to describe local-scale dioxin emissions from utility boilers. Four model plants were selected, one each for: large coal-fired, small coal-fired, large oil-fired, and small oil-fired. The selection of large and small model plants represents an upper- and lower-end depiction of dioxin emissions. The dioxin emissions rates for the model plants were developed based on dioxin emissions estimates from the EPA database of 426 coal-fired plants and 145 oil-fired plants. The 426 coal-fired and 137 oil-fired plants were ranked by 2,3,7,8-TCDD TEQ emissions (kg/y) and then divided into thirds. The large model plants represent the average dioxin emissions for the upper ranking third of the plants for each fuel type (e.g., emissions for the large coal-fired utility model plant are the average dioxin emissions for the 142 largest emitting coal-fired utility plants), while the small model plants represent the average dioxin emissions from the lower third of the plants for each fuel type. The resulting dioxin and furan congener-specific emissions rates and process parameters for each for each model plant are shown in Tables 11-1 and 11-2.[5]

The emissions information for each of the four model plants was applied for two locations. One location (case #1) was modeled using the emissions from each of the model plants combined with high-end meteorology and terrain conditions. Albany, New York, was selected to be the case #1 meteorological location, as this location has previously been shown to have high-end meteorology for the purpose of dispersion modeling. Specifically, Albany's geographic location in the Hudson River Valley limits wind flow patterns, which generally limits the dilution effects of varied wind flow patterns. Complex terrain was combined with the Albany meteorological data to complete the case #1 application of the model plant data. Similarly, all model plant data were applied to more average meteorology and simple terrain. Springfield, Missouri, was selected as a location for average meteorological conditions (case #2). The application of two locations and terrain conditions to the model plant data gives a total of eight model plants considered in the assessment (Table 11-3).

### 11.3.2 Local Air Dispersion Modeling

The EPA's ISCST3 dispersion model was used to predict the local-scale (<50 km from the source) atmospheric dispersion of model plant dioxin emissions. ISCST3 is a Gaussian plume model that has the capacity to model wet and dry depletion with corresponding annual deposition rates, as well as ambient air concentrations. A more detailed description of the ISCST3 model may be found in the other multipathway assessments of this report (Chapters 7 and 10), or in other EPA publications.[4] The resulting annual depositions and ambient concentrations are used as inputs to the IEM-spreadsheet model for the purpose of multipathway exposure and risk analysis.

Table 11-1. Congener-Specific Emissions Rates for Model Plants (kg/yr)

| Congener[a] | Large coal | Large oil | Small coal | Small oil |
|---|---|---|---|---|
| PCDDs: | | | | |
| 2378 | 1.24e-05 | 1.82e-04 | 1.67e-04 | 8.71e-06 |
| 12378 | 3.22e-04 | 1.61e-04 | 4.34e-05 | 7.70e-06 |
| 123478 | 7.32e-04 | 3.46e-04 | 9.87e-05 | 1.66e-05 |
| 123789 | 5.54e-04 | 2.33e-04 | 7.46e-05 | 1.12e-05 |
| 123678 | 4.39e-04 | 1.52e-04 | 5.92e-05 | 7.28e-06 |
| 1234678 | 4.34e-04 | 5.54e-04 | 5.85e-05 | 2.66e-05 |
| Octa | 4.41e-03 | 6.47e-04 | 5.94e-04 | 3.10e-05 |
| PCDFs: | | | | |
| 2378 | 2.99e-04 | 1.28e-04 | 4.02e-05 | 6.15e-06 |
| 23478 | 7.88e-04 | 1.34e-04 | 1.06e-04 | 6.44e-06 |
| 12378 | 1.85e-04 | 1.21e-04 | 2.50e-05 | 5.80e-06 |
| 123478 | 9.77e-04 | 1.69e-04 | 1.32e-04 | 8.09e-06 |
| 123678 | 3.02e-04 | 1.07e-04 | 4.07e-05 | 5.14e-06 |
| 123789 | 6.40e-04 | 1.62e-04 | 8.63e-05 | 7.79e-06 |
| 234678 | 1.23e-03 | 1.33e-04 | 1.65e-04 | 6.38e-06 |
| 1234678 | 1.49e-03 | 2.62e-04 | 2.01e-04 | 1.26e-05 |
| 1234789 | 1.31e-02 | 2.82e-04 | 1.77e-03 | 1.35e-05 |
| Octa | 1.25e-03 | 2.82e-04 | 1.68e-04 | 1.35e-05 |

PCDDs = polychlorinated dibenzo-p-dioxins; PCDFs = polychlorinated dibenzofurans

[a] Congener: Numbers represent chlorine placement on the molecule.
Data in this table provided by EPA for this analysis. [5]

Table 11-2. Process Parameters for Model Plants

| Model plant | Stack height (m) | Stack diameter (m) | Exit velocity (m/s) | Exit temp. (K) |
|---|---|---|---|---|
| Large coal-fired (800 MW) | 211.8 | 8.290 | 28 | 412.6 |
| Small coal-fired (232 MW) | 119.5 | 4.938 | 22 | 399.3 |
| Large oil-fired (506 MW) | 157.0 | 6.584 | 26 | 434.8 |
| Small oil-fired (174 MW) | 70.10 | 3.962 | 19 | 419.8 |

MW = megawatt
Data in this table provided by EPA for this analysis. [5]

Table 11-3. Naming Scheme for Eight Model Plants

| | |
|---|---|
| LCH | **L**arge **C**oal-Fired Plant, **H**igh-End Meteorology and Complex Terrain |
| SCH | **S**mall **C**oal-Fired Plant, **H**igh-End Meteorology and Complex Terrain |
| LOH | **L**arge **O**il-Fired Plant, **H**igh-End Meteorology and Complex Terrain |
| SOH | **S**mall **O**il-Fired Plant, **H**igh-End Meteorology and Complex Terrain |
| LCC | **L**arge **C**oal-Fired Plant, **C**entral-tendancy Meteorology and Simple Terrain |
| SCC | **S**mall **C**oal-Fired Plant, **C**entral-tendancy Meteorology and Simple Terrain |
| LOC | **L**arge **O**il-Fired Plant, **C**entral-tendancy Meteorology and Simple Terrain |
| SOC | **S**mall **O**il-Fired Plant, **C**entral-tendancyMeteorology and Simple Terrain |

The ISCST3 model requires inputs related to the processes and location of modeled facilities. The ISCST3 dispersion modeling requires various emissions source process parameters for dispersion and deposition modeling, including stack height, stack diameter, exit velocity, and exit temperature. The value of each of these inputs for each of the model plants is presented in Table 11-2.[5] Other process-specific dispersion parameters include particle size distributions, mass fraction of particles, and wet and dry scavenging coefficients (Table 11-4). These data were obtained for utility boilers from AP-42,[6] *Perry's Chemical Engineering Handbook*,[7] and the *User's Guide for the Industrial Source Complex Dispersion Models*,[4] respectively. Additional emissions source information, such as meteorology, complex terrain, and land use factors, was varied between case #1 and case #2 (Table 11-5). The ISCST3 meteorological inputs were five years of data from the Solar and Meteorological Surface Observation Network (SAMSON) for each of the locations considered in this analysis.[8] The Albany, New York, terrain information was obtained through the United States Geological Survey.[9]

ISCST3 modeling was completed using the "default" model options specified in the *Guidelines on Air Quality Models.*[10] A unit emissions rate (1 g/s) was used in modeling the ambient air concentrations and wet and dry deposition rates. A polar array of receptors was used, with receptors placed 22.5 degrees apart on concentric circles 500 m (for case #1) and 5,000 m (for case #2) from the source. The case #1 and case #2 receptor distances were patterned after prevous dioxin indirect exposure assessments.[3,11] The receptor distances from the source were patterned after these studies because experience has shown that, in general, exposure is increased closer to the source, and these locations were believed to be reasonable distances for potential exposure scenarios. The locations were tested for sensitivity regarding the modeled exposure/risk results (see section 11.5.3, below), and determined to be adequate for this analysis.

Table 11-4. Process-Specific Depletion Parameters

| Parameter | Value | | | | |
|---|---|---|---|---|---|
| *Gas depletion* | | | | | |
| Gas Scavenging coefficient ($s^{-1}$) | $1.8 \times 10^{-4}$ | | | | |
| *Particle depletion* | | | | | |
| Particle size distribution (microns) | 15 | 10 | 6 | 2.5 | 1 |
| Mass fraction per particle size: coal-fired [a] | .005 | 0.09 | 0.20 | 0.23 | 0.43 |
| Mass fraction per particle size: oil-fired [a] | 0.31 | 0.115 | 10.5 | 10.5 | 36.5 |
| Particle density ($g/cm^3$) [a] | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 |
| Particle scavenging coefficient ($s^{-1}$) | $6.6 \times 10^{-4}$ | $6.6 \times 10^{-4}$ | $4.2 \times 10^{-4}$ | $1.8 \times 10^{-4}$ | $4.3 \times 10^{-5}$ |

[a] Particle size distributions by mass taken from AP-42 [6] data for dry bottom boilers burning bituminous coal and residual oil-burning utility boilers. Particle density estimated for bituminous coal [7] and confirmed with EPA/ORD.

Table 11-5. Other Emissions Source Information

| Setting type | Case #1 | Case #2 |
|---|---|---|
| ***Location Data*** | | |
| Surface/upper air | Albany, NY/Albany, NY | Springfield, MO/Monett, MO |
| Anemometer height (m) | 10.00 | 6.10 |
| ***Preprocessor Inputs*** | | |
| Land use within 5 km | Suburban/rural | Suburban/rural |
| Minimum Monin-Obukov length (m) | 2.00 | 2.00 |
| Roughness height (m) | 0.50 | 0.33 |
| Noon-time albedo (fraction) | 0.28 | 0.22 |
| Bowen ratio (fraction) | 0.50 | 0.51 |
| Net radiation absorbed in ground (fraction) | 0.15 | 0.15 |
| Anthropogenic heat flux ($watts/m^2$) | 0.00 | 5.40 |

ISCST3 was used to generate a wet and dry deposition rate, as well as an ambient air concentration for both particle and vapor phase emissions. To determine the separate particle and vapor fractions of the emissions, each model plant case was run twice: once for particle phase emissions and once for vapor phase emissions. A particle modeling run was performed for each model plant using the particle information in Table 11-4, with the output options of wet and dry deposition and air concentration. A vapor, or gas phase, modeling run was also performed for each model plant by omitting particle

information and allowing only air concentration and wet depletion outputs. It was assumed that gaseous emissions behaved like small particles for the purpose of gas scavenging and wet deposition modeling. Dry deposition of vapors was approximated using a dry deposition rate of 0.2 cm/s, which was estimated from previous EPA research[12] and cited in the secondary aluminum smelter indirect exposure analysis for dioxins.[11] For each of the eight model plant cases, the maximum vapor and particle air concentration and wet and dry deposition rates were converted from the unit emissions results to congener specific values using the model plants' individual dioxin congener emissions rates (Table 11-1). The vapor and particle congener specific values were then multiplied by the appropriate vapor/particle fraction for ambient conditions.[3] Additionally, particle air concentrations were modified to reflect only particles less than approximately 10 microns, based on the initial mass fraction in Table 11-4, for the inhalation exposure.

### 11.3.3 Exposure Modeling and Risk Calculation

Exposure and risk modeling were performed using the IEM-Spreadsheet. The spreadsheet model is based on the *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*[13] and its Addendum.[14] The IEM-spreadsheet model uses the dispersion modeled annual air concentrations and wet and dry deposition rates to estimate direct and indirect exposure and risks associated with dioxin model plant emissions. Direct exposure and risks are determined from the inhalation of airborne emissions particles and vapors. Indirect (non-inhalation) exposure and risks are those associated with human contact with the dioxins that have accumulated in the environment. The model assesses indirect exposures and risks by using a simplified mass balance approach to determine the fate and concentrations of emissions in environmental media including surface soils, surface water, fish, vegetation, and domestic animals. Using estimates of human contact, these environmental media concentrations are translated to the indirect (non-inhalation) human exposures and the related cancer risk values. The different routes or pathways of indirect human contact considered by the IEM-spreadsheet model include dermal contact with soil and ingestion of water, soil, fish, plants, and domestic animals. A flow chart diagram of the processes involved in the multipathway analysis and the IEM-spreadsheet model are presented in Figure 11-4.

The version of the IEM-spreadsheet used in this assessment was developed for dioxins by the EPA Office of Research and Development (EPA/ORD).[15] The spreadsheet was modified, as advised by ORD, to be more consistent with the current assumptions and methods used by ORD in their work on dioxins. The spreadsheet was modified with a single soil loss rate constant, a biota sediment accumulation factor (BSAF), and an increased no-till depth of soil incorporation, to suit the recommendations of EPA/ORD. The single soil loss rate constant has a value of 0.0277 $y^{-1}$, which is based on a new estimate of a 25-year

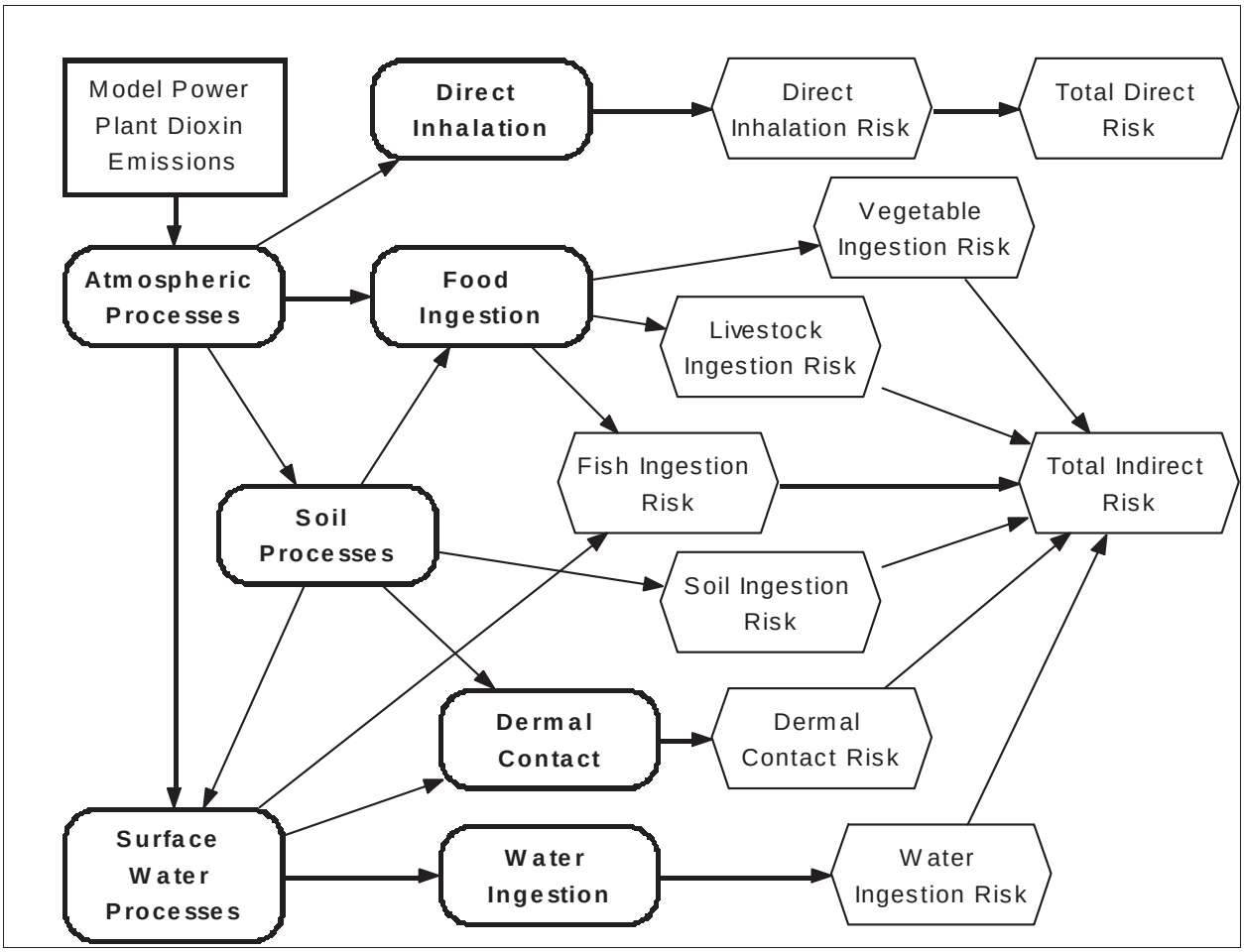



Figure 11-4. Flow Chart of Multipathway Processes

half-life for dioxins in surface soils.[15] The fish accumulation pathway was adjusted to use BSAF calculations because dioxins are "super-hydrophobic," and thus are not detected in the water column even in highly contaminated environments where the aquatic life show high tissue concentrations.[3] In the IEM-spreadsheet model, the hydrophobic compounds are assumed to be adsorbed to the hydrophobic components of aquatic sediments. The BSAF is the preferred method of calculation because it bases the concentration in fish on the concentration in adsorbed to bottom sediments, thus providing a more accurate relationship between aquatic environment concentrations and fish tissue concentrations. The BSAF values used in this assessment were based on those in *Estimating Exposure to Dioxin-Like Compounds*[3] and additional data from the *Great Lakes Water Quality Initiative Technical Support Document for the Procedure to Determine Bioaccumulation Factors*,[16] provided by EPA/ORD.[15] The depth of no-till soil incorporation was increased based on new data,[17] which was recommended by EPA/ORD.[15] In addition to the above stated

modifications, the animal bioaccumulation methodology was modified to reflect *Estimating Exposure to Dioxin-Like Compounds*[3] due to the availability of data from this document. Also, the original version of IEM-Spreadsheet did not calculate direct inhalation exposures and risk, so the model was modified to calculate direct inhalation risk, per the *Addendum to the IEM Methodology*.[14]

Aside from the parameters and recommended changes described above, all other IEM-spreadsheet inputs were chosen specifically for this study. Each input parameter, its modeled value, and the source of the data are presented in Appendix G-1 of this report. Every attempt was made to keep the parameters used in this chapter consistent with those used in the multipathway assessments for mercury and arsenic (chapters 7 and 10 of this report). This includes the description of the watershed, which was assumed to have the same characteristics as the hypothetical watershed described in the mercury and arsenic chapters. The IEM-spreadsheet inputs related to human exposure were obtained from the *Mercury Study Report to Congress*[18] due to the parallel observed between the mercury multipathway analysis in the *Mercury Study Report to Congress* and the mercury multipathway analysis in Chapter 7. For parameters specific to dioxin and furan congeners, the preferred source of input data was *Estimating Exposure to Dioxin-Like Compounds*.[3] Additional site-specific information was obtained for parameters related to the assumed model plant locations (see section 11.3.2 above for location descriptions); and for parameters not specified by the sources listed above, general defaults were obtained from the methodology supporting the IEM-spreadsheet.[13,14]

The indirect exposure methodology and input parameters described above were used to estimate exposures to dioxin and furans. The case #2 exposure and risk modeling scenarios used the results of the dispersion modeling (see section 11.3.2 above) and exposure assumptions related to case #2 receptors. The case #2 receptors were based on a resident adult and child scenario. The case #2 scenarios were used in modeling exposure and cancer risks through the direct inhalation, soil dermal contact, soil ingestion, home-grown vegetable ingestion, and surface-supplied drinking water ingestion pathways. The case #1 exposure and risk modeling scenarios employed the results of the case #1 dispersion modeling (see section 11.3.2 above) and exposure assumptions related to case #1 receptors. The case #1 receptors were based on subsistence fisher and subsistence farmer scenarios. These case #1 receptors were considered for all the pathways described for the case #2 receptors. Additionally, the subsistence farmer scenario considered ingestion of home-grown animal products and the subsistence fisher scenario considered ingestion of locally caught fish. The case #1 scenarios also used different vegetable ingestion contact fractions. The vegetable ingestion contact fractions represent a subsistence farmer, a rural gardener, and urban gardener[18] for the subsistence farmer, subsistence fisher, and residents, respectively. Table 11-6 presents a summary of the scenarios and pathways considered in this assessment, and Table 11-7 presents all 16 hypothetical scenarios included in this assessment.

Table 11-6. Summary of Receptor Scenarios and Pathways

| Receptor scenario | Pathways considered |
|---|---|
| Adult resident<br>(used for case #2 modeling scenarios only) | Direct inhalation, soil dermal, soil ingestion, vegetable ingestion, and drinking water ingestion |
| Child resident<br>(used for case #2 modeling scenarios only) | Direct inhalation, soil dermal, soil ingestion, vegetable ingestion, and drinking water ingestion |
| Subsistence fisher<br>(used for case #1 modeling scenarios only) | Direct inhalation, soil dermal, soil ingestion, vegetable ingestion, drinking water ingestion, and fish ingestion |
| Subsistence farmer<br>(used for case #1 modeling scenarios only) | Direct inhalation, soil dermal, soil ingestion, vegetable ingestion, drinking water ingestion, and animal product ingestion |

Table 11-7. The 16 Hypothetical Scenarios Included in the Screening Level, Model Plant, Dioxin Multipathway Exposure and Risk Assessment.

| Scenario | Description |
|---|---|
| LCH-fisher | Subsistence fisher living near large coal-fired plant, using case #1 modeling assumptions and inputs |
| LOH-fisher | Subsistence fisher living near large oil-fired plant, using case #1 modeling assumptions and inputs |
| SCH-fisher | Subsistence fisher living near small coal-fired plant, using case #1 modeling assumptions and inputs |
| SOH-fisher | Subsistence fisher living near small oil-fired plant, using case #1 modeling assumptions and inputs |
| LCH-farmer | Subsistence farmer living near large coal-fired plant, using case #1 modeling assumptions and inputs |
| LOH-farmer | Subsistence farmer living near large oil-fired plant, using case #1 modeling assumptions and inputs |
| SCH-farmer | Subsistence farmer living near small coal-fired plant, using case #1 modeling assumptions and inputs |
| SOH-farmer | Subsistence farmer living near small oil-fired plant, using case #1 modeling assumptions and inputs |
| LCC-resident | Adult resident living near large coal-fired plant, using case #2 modeling assumptions and inputs |
| LOC-resident | Adult resident living near large oil-fired plant, using case #2 modeling assumptions and inputs |
| SCC-resident | Adult resident living near small coal-fired plant, using case #2 modeling assumptions and inputs |
| SOC-resident | Adult resident living near small oil-fired plant, using case #2 modeling assumptions and inputs |
| LCC-child | Child resident living near large coal-fired plant, using case #2 modeling assumptions and inputs |
| LOC-child | Child resident living near large oil-fired plant, using case #2 modeling assumptions and inputs |
| SCC-child | Child resident living near small coal-fired plant, using case #2 modeling assumptions and inputs |
| SOC-child | Child resident living near large oil-fired plant, using case #2 modeling assumptions and inputs |

## 11.4 DISPERSION, EXPOSURE, AND RISK RESULTS

This section presents the relationship between dioxin emissions from the model plant stack and human cancer endpoints. Dispersion modeling provided estimates of ambient air concentrations and wet and dry deposition rates. The IEM-spreadsheet modeling estimated dioxin concentrations in environmental media (i.e., soil, water, plant, animal, and fish). The IEM-spreadsheet also estimated human exposure to these environmental concentrations to approximate the corresponding human exposure through various pathways (see Table 11-6) in terms of lifetime averaged daily doses (LADDs) and cancer risk estimates. The exposure and risk results of the local-scale and long-range modeling were compared.

The dispersion modeling results for each model plant at both the case #1 and case #2 receptor distances are presented in Appendix G-2. The data in Appendix G-2 are the maximum vapor and particle phase ambient air concentrations and wet/dry deposition rates for each model plant, under both case #1 and case #2 conditions (see sections 11.3.1 and 11.3.2 above for description). The dispersion modeling results indicate that the case #1 exposures stem entirely from wet deposition, with the exception of very low air concentration and dry deposition associated with the small oil-fired model plant. It is likely that the air concentration and dry deposition rate are zero because the modeled plume of emissions has not yet settled to ground level at this distance. From this it can be concluded that the modeled case #1 receptors, located at 500 m, probably are not at the highest direct inhalation exposure location because receptors are actually too close to the very tall stacks. However, the 500 m distance has much higher wet deposition rates, which may account for higher indirect exposures. Following the observation of wet deposition versus plume impaction, the sensitivity of the results to the receptor distance from the stack was analyzed (see section 11.5.3 below).

The environmental media dioxin concentrations resulting from the mass balance performed by the IEM-spreadsheet are presented in Appendix G-3. The concentrations are reported for surface soil, surface water, whole fish, and plants and animals produced on contaminated lands. The results indicate that the bioaccumulated concentrations in organisms (fish, plants, and animals) exceed those that persist in soil and water. This is a reasonable result, given the sequestration of the lipophilic dioxin and furan compounds in the organisms' tissues.

The congener-specific LADDs for each of the exposure pathways and scenarios modeled in this chapter are detailed in Appendix G-4. The LADDs were calculated by the IEM-spreadsheet model, using the media concentrations described above and the input parameters related to human contact and exposure (Appendix G-1). The greatest LADD is seen in the subsistence fisher fish consumption pathway, with animal product and vegetable consumption following closely. This observation is consistent with the modeled media concentrations because each of

these pathways involves consumption of an organism that has bioaccumulated dioxins from the environment.

Background exposures to dioxins in the United States have been estimated.[2] The estimated exposures range from 0.8 pg/day for soil ingestion to 37.0 pg/day for ingestion of beef and veal. The background exposure from fish ingestion is listed as 7.8 pg/day, and the background exposure from inhalation is 0.8 pg/day. After converting the units on the modeled LADD exposures in Appendix G-3, the corresponding maximum modeled exposures (which do not take into account background) are 2.1 pg/day for soil ingestion, 140 pg/day for animal product ingestion, 420 pg/day for fish ingestion, and 0.0014 pg/day for direct inhalation. In each of the indirect pathways, the modeled exposures exceed background.

The congener-specific results of cancer risk calculation for each pathway and each scenario are presented in Appendix G-5. The cancer risk values were calculated using a slope factor for dioxin of 0.156 kg-d/ng, and the Toxicity Equivalency Factors (TEFs) for 2,3,7,8-dioxin congeners (see Appendix G-1). This slope factor was adopted from *Estimating Exposure to Dioxin-Like Compounds*;[3] the derivation of the factor is described in the report *Risk Analysis of TCDD Contaminated Soil*.[19] The calculated risks show the same pattern as the LADDs, with the highest risks generally associated with fish consumption. A summary of the predicted dioxin TEQ cancer risks for each hypothetical scenario is provided in Table 11-8. Note that the zero direct risks in the case #1 exposure scenarios are due to the selection of 500 m as the receptor location. As described above, at this location the ambient air concentrations are zero, but the deposition is near to maximum.

The TEQ cancer risks are greater for the combined indirect (non-inhalation) exposures than the direct (inhalation) exposures in every modeled scenario. In all cases, the indirect risks are at least an order of magnitude larger. These results demonstrate the need to consider indirect risks for environmental pollution that is persistent in the environment, such as dioxins. This is especially evident in the case #1 analyses where the direct risk is zero (because vapor and particle air concentrations are zero), yet the indirect risks from the multipathway exposures based on wet deposition are as high as $2 \times 10^{-4}$.

The highest risk, predicted to be $2 \times 10^{-4}$, is to the subsistence fisher hypothetical scenario from the indirect (non-inhalation) pathways for the large coal-fired model plant. Given the high level of risk predicted for the subsistence fisher, this value is explored further in a sensitivity analysis (see section 11.5.2 below). The maximum direct (inhalation) risk for this model plant was $4 \times 10^{-10}$, which is orders of magnitude less than the risk predicted for the maximum exposed individual considered by the direct inhalation assessment in chapter 6 of this report. This large difference in inhalation risk estimates may be due to the selection of receptor distance in the model plant analysis.

Table 11-8. Summary of Predicted Cancer Risks from the Screening Level Multipathway Assessment for Model Plants, for 16 Hypothetical Scenarios

| Scenario[a] | Predicted cancer risk due to direct inhalation exposure[b] | Predicted risk due to indirect (non-inhalation) exposure[b] |
|---|---|---|
| LCH-fisher | 0 | 1E-04 to 2E-04 |
| LOH-fisher | 0 | 5E-05 to 1E-04 |
| SCH-fisher | 0 | 3E-05 to 6E-05 |
| SOH-fisher | 1E-13 | 3E-06 to 6E-06 |
| LCH-farmer | 0 | 1E-05 to 3E-05 |
| LOH-farmer | 0 | 1E-05 to 2E-05 |
| SCH-farmer | 0 | 5E-06 to 1E-05 |
| SOH-farmer | 1E-13 | 5E-07 to 1E-06 |
| LCC-resident | 2E-10 | 2E-08 to 5E-08 |
| LOC-resident | 2E-10 | 1E-08 to 2E-08 |
| SCC-resident | 4E-10 | 1E-08 to 3E-08 |
| SOC-resident | 7E-11 | 3E-9 to 6E-09 |
| LCC-child | 1E-10 | 1E-08 to 2E-08 |
| LOC-child | 1E-10 | 1E-08 to 2E-08 |
| SCC-child | 2E-10 | 1E-08 to 2E-08 |
| SOC-child | 5E-11 | 1E-09 to 2E-09 |

[a] See Tables 11-3, 11-6, and 11-7 and text for description of scenarios and associated assumptions, methods and inputs.

[b] The risks are presented as a range. For each scenario, the higher predicted risks are based on results of the modeling method, assumptions, and data inputs as described in this chapter. The lower predicted risks for each scenario are based on the assumption that the emissions estimates for the model plants (Table 11-1) could possibly be over estimated by a factor of 2 (see discussion in section 11.1); therefore, it is predicted (based on the assumption that the exposure and risk calculations are reasonably linear) that the risks for the hypothetical scenarios could be about 2 times lower than those predicted using inputs and methods described in this chapter if the emissions rates using in the calculations were 2 times lower than those shown in Table 11-1.

The predicted risks presented in Table 11-8 are screening level estimates based on model plants and hypothetical scenarios. The results do not apply to any actual utility plant. There are substantial uncertainties in these predicted risks because of numerous assumptions and data inputs with varying levels of uncertainty. For example, the emissions estimates for the model plants are quite uncertain and may be overestimated, or possibly underestimated (see

section 11.1 for discussion of emissions data uncertainty). Additional uncertainties and variability in the assessment and data inputs are discussed in section 11.5.

The multipathway exposures and risks resulting from the local-scale ISCST3 dispersion modeling results (described in the preceding paragraphs) were compared to those resulting from the RELMAP modeling of long-range transport (Appendix G-6). The comparison was made for dioxin TEQ rather than the congener specific method used in the local-scale modeling, due to the limited data available from RELMAP. To obtain exposure and risk estimates, the average RELMAP data for air concentration, and wet and dry deposition were used with the IEM - spreadsheet model (thus paralleling what was done for the local-scale ISCST3 dispersion result). It was assumed that the air concentration results of the RELMAP model (Figure 11-1) were inhalable for the purpose of inhalation exposure. A similar comparison using background air concentration and deposition rates was not preformed for this study due to limited data.

The comparison shows that in most cases the local-scale emissions exposures are a greater percentage of the total exposures than the long-range transport exposures. The importance of the local-scale emissions generally decreases as distance from the facility increases. Thus, long-range transport was more influential in the case #2 modeling scenarios, for which local-scale modeling was at a greater distance from the facility. Long-range transport was most influential in the direct inhalation pathway and the vegetable ingestion pathway because the RELMAP modeled air concentrations were relatively large portions of the total TEQ air concentration.

## 11.5 UNCERTAINTY AND SENSITIVITY ANALYSIS

The models and assumptions used in this study have been the subject of significant review and revision as part of other EPA studies.[20] The reviews have identified limitations of the methods and the data sets from which the parameters are derived. Reviewers have also cautioned EPA against misuse of the indirect exposure methodology, including both the dispersion and the intermedia transfer modeling components. Every effort has been made in the present study to use exposure models and parameter values consistent with the latest EPA studies of dioxin and mercury risks.

Specifically, the review of IEM methodology performed by the Science Advisory Board (SAB)[20] made several recommendations that were considered in this analysis. The SAB recommended accounting for the cumulative impacts of several sources on the regional or National level. The national RELMAP modeling results included in sections 2.0 and 4.0 of this chapter respond to this recommendation. The SAB cited lack of validation results as a scientific uncertainty in the IEM methodology. The SAB recommends sensitivity analysis as a means of determining the importance of input values. Sensitivity analyses were performed for this assessment, and are described in the following

subsections. However, the SAB notes problems in the model with conservation of mass and chemical thermodynamics that were not corrected in the model for this application. It should be noted that these issues and others add uncertainty to the mass balance estimation of media concentrations performed by the IEM spreadsheet, as well as the corresponding modeled indirect exposures and risks.

### 11.5.1 Model Elasticity

A brief, initial sensitivity analysis was performed for selected parameters employed in the risk modeling exercise. Using professional judgement, a limited number of parameters were chosen for their perceived importance in the model. Additionally, the chosen parameters are known to have uncertainty in their value, generally due to limited or variable empirical data. The selected parameters were individually varied from 90 percent to 110 percent of their initial value, while all other parameter values were held constant. The resulting changes in the modeled risks for the pathway(s) impacted were analyzed. All sensitivity analyses were performed using data from the large coal-fired model plant because it demonstrated the largest indirect exposures and risks.

The bioconcentration factor (BCF), or biota sediment accumulation factor (BSAF), for fish was analyzed in the fisher scenario, as it is recognized that fish consumption is an important factor for estimating risks to the subsistence fisher, and is the overall driver of the risk assessment. There is a limited set of data available for consideration in the selection of BSAF;[3,16] and thus BSAF was analyzed for impact on the fish consumption pathway by percentage variation of the modeling input value.

Similarly, the BCFs for animal products (beef, chicken, dairy, and eggs) were analyzed for the animal product consumption pathway of the subsistence farmer scenario, as animal product consumption provides significant risk in many of the modeled scenarios. Again, there are limited data available for the bioconcentration of dioxin-like compounds in animal products,[3,21] and no true distribution of data could be determined for analysis. Thus, in order to explore the impact of changes in the BCFs, percentage changes in the modeling input value were used.

The overall soil loss rate constant, which was recommended for dioxin exposure modeling by EPA/ORD, was analyzed by expanding the number of pathways considered in the adult resident scenario. The soil loss rate constant was chosen because it influences the exposure media concentrations in several pathways, including soil dermal exposure, soil ingestion, plant ingestion, animal product ingestion, water ingestion, and fish consumption. There was believed to be uncertainty in the value of this parameter because the value recommended by the EPA has changed from 0.0693 $y^{-1}$ [3] to 0.0277 $y^{-1}$.[15] Again, due to the unavailability of a distribution for the soil loss rate constant, the sensitivity analysis was performed by percent change in the value.

The results of this sensitivity analysis indicate that the IEM-spreadsheet model is responsive to each of the parameters addressed in this sensitivity analysis in differing amounts. None of the tested cases create a change in risk greater than ten percent as a result of the ten percent change in the tested parameter. The bioconcentration and accumulation factors (Figure 11-5) show exactly a ten percent change in modeled risk due to the ten percent change in the input value, while the changes in risk, due to the ten percent change in the soil loss rate constant, vary from 3.7 percent to 0.2 percent for the different pathways considered (Figure 11-6).

These results indicate that none of the tested parameters will create a greater percentage change in risk than the percentage change in the input value. This conclusion is not unreasonable given the general linearity of the model and the large number of input parameters and equations playing into the calculated risks.

### 11.5.2 Fish Consumption Pathway Sensitivity Analysis

In consideration of the risk analysis results, it was observed that the fish consumption pathway has the highest cancer risks. Thus, a more extensive sensitivity analysis was undertaken for the input parameters involved in the fish consumption pathway. The inputs into the fish consumption pathway were considered, and five parameters were chosen for this analysis: fish consumption rate of the subsistence fisher, the fish lipid correction term, the BSAF, the fraction organic carbon in the bottom sediments, and the organic carbon partition coefficient ($K_{oc}$). The available data for each of these parameters was examined and high and low values were selected, as described below. The IEM-spreadsheet was then used to hold all other parameters constant while individually considering each of the high and low values for the fish consumption pathway input variables. This analysis was performed using the large coal-fired model plant and the subsistence fisher scenario because this combination generated the greatest risk in the entire assessment.

The fish consumption rate for the subsistence fisher scenario in the risk modeling exercise was 0.86 g fish/kg body weight/day. This value was calculated from the 60 g fish/d recommendation of the *Mercury Study Report to Congress*.[18] This value is based on the weighted mean of the data from a 1994 study by the Columbia River Inter-Tribal Commission (as presented in the EPA Exposure Factors Handbook).[22,23] The units of g fish/d were converted by assuming a 70 kg body weight. It was also assumed that, as a subsistence population, the fishers caught all of the consumed fish locally. High and low consumption rates taken from the Columbia River data set have the values of 0.04 g fish/kg body weight/day ($10^{th}$ percentile) and 2.0 g fish/kg body weight/day ($90^{th}$ percentile).

The fish lipid correction term is used as a correction for the edible portion of fish lipid, as the BSAFs are developed for whole fish lipid concentration. The fish lipid correction term in the risk modeling exercise was 0.07, as recommended in *Estimating Exposure to Dioxin-Like*

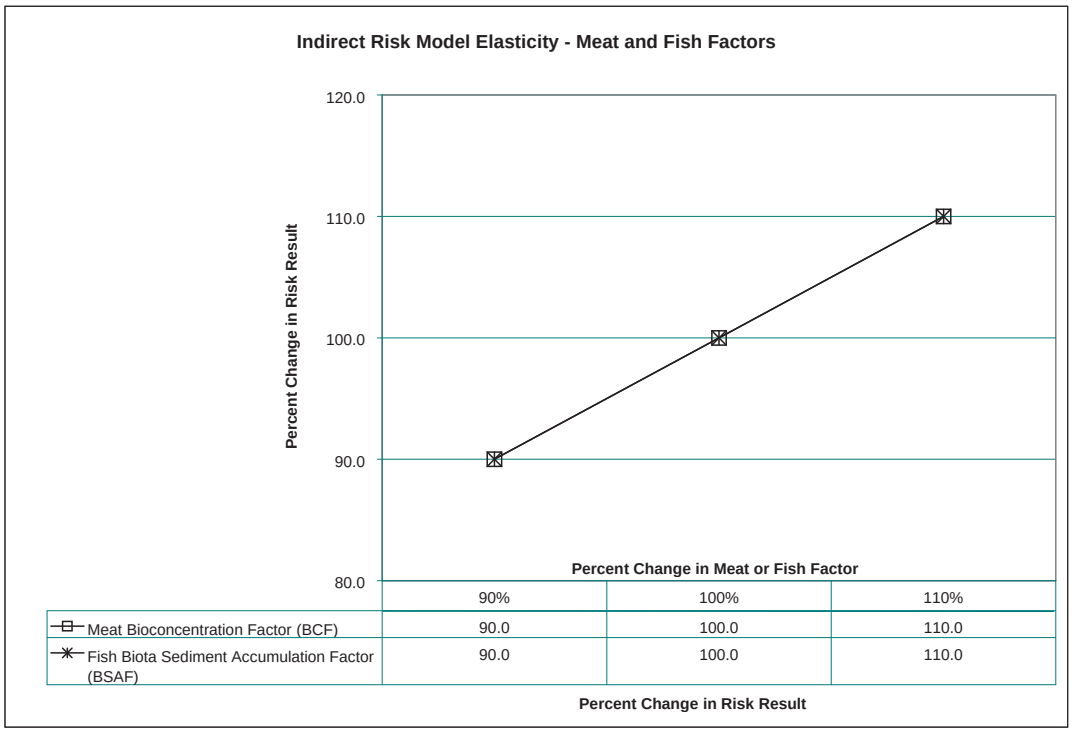



Figure 11-5. Risk Model Sensitivity to Changes in Meat and Fish Factors

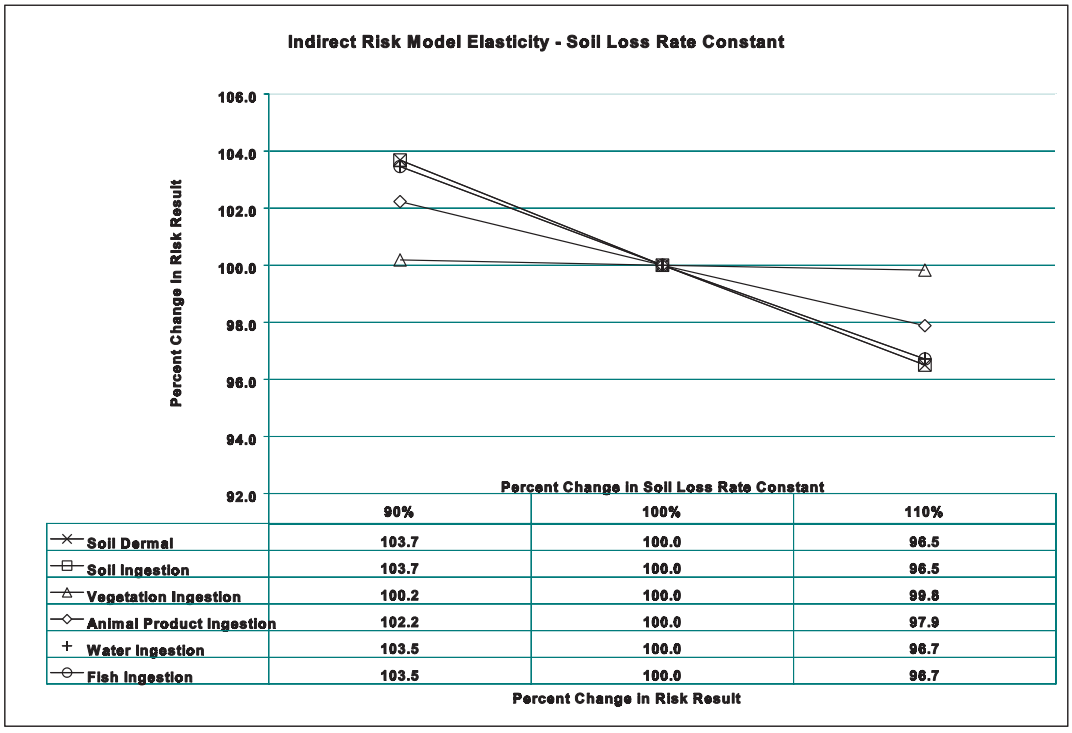



Figure 11-6. Risk Model Sensitivity to Changes in Soil Loss Rate Constant

*Compounds*.[3] This document notes that fish lipid content generally varies from 5 percent to 20 percent, and thus these values were used as the high and low sensitivity analysis values for this parameter.

The BSAFs used in the risk modeling exercise were the congener specific values used in the demonstration of methodology in chapter 5 of *Estimating Exposure to Dioxin-Like Compounds*.[3] The limited set of empirical values for dioxin congener BSAFs[3,16] was considered in the determination of the high and low values for sensitivity analysis. The highest and lowest empirical values for each congener were chosen to use as the high and low values in this sensitivity analysis.

The fraction of organic carbon in the sediment used in the risk modeling exercise was 0.03, as recommended by *Estimating Exposure to Dioxin-Like Compounds*.[3] This value was assumed to be more than the soil organic carbon fraction (0.01) and less than the organic carbon fraction of suspended solids (0.05). Thus, the assumed values for the soil and suspended solids organic carbon fractions were used as the low and high values for variation in sediment organic carbon content.

The $K_{oc}$ was considered in this sensitivity analysis because it directly affects the concentration of dioxins and furans adsorbed to the benthic sediments, which is used in the fish pathway. The concentration of each congener that is adsorbed to the benthic sediments is dependent on the sediment-water partition coefficient, and this coefficient was determined by estimation using $K_{oc}$. The initial values for $K_{oc}$ were obtained from *Estimating Exposure to Dioxin-Like Compounds*.[3] The $K_{oc}$ for each congener was varied, using the high and low empirical values from Mackay,[24] and assumptions where data was unavailable. The calculation of the adsorbed sediment concentrations is complex and involves many input parameters; therefore, the only attempt at analyzing sensitivity to the adsorbed sediment concentration was through the analysis of $K_{oc}$.

The results of the fish consumption pathway sensitivity analysis are presented in Figure 11-7 and Table 11-9. The results indicate that none of the single changes to low-end parameter values decreased the fish ingestion risks below $9 \times 10^{-6}$. However, the high assumption for BSAF raised the modeled cancer risks to as high as $1 \times 10^{-3}$, or one in one thousand for the subsistence fisher.

### 11.5.3 Sensitivity Analysis of Plume Impaction

As previously described in section 11.3, it was recognized that the risks described for the case #1 farmer and fisher scenarios are due to only wet deposition concentrations. The zero values for dry deposition and air concentration at the case #1 location imply that plume impaction has not occurred at 500 m. Thus, an analysis was performed to identify areas of plume impaction, with the idea that potential exposures may exceed even the case #1 modeling scenarios.

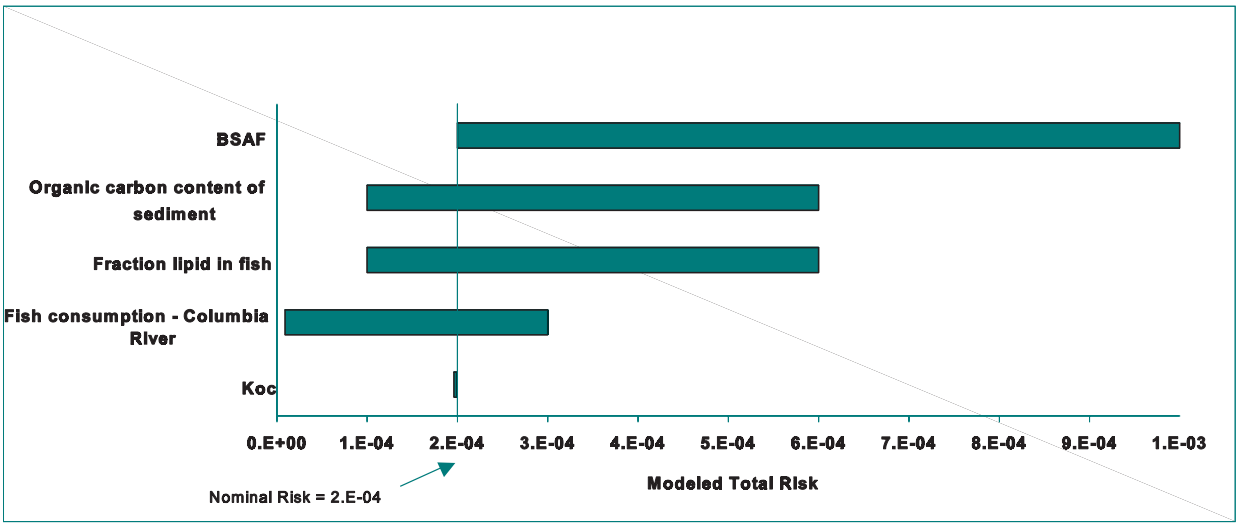



Figure 11-7. Sensitivity of Predicted Risk to the Subsistence Fisher to Changes in Parameter Values

Table 11-9. Fish Consumption Pathway Sensitivity Analysis Inputs and Results

| *Parameter* | *Parameter Value* | | | *Risk Result* | | |
|---|---|---|---|---|---|---|
| | Low | Nominal | High | Low | Nominal | High |
| BSAF (unitless) | congener specific | congener specific | congener specific | 2e-04 | 2e-04 | 1e-03 |
| Organic carbon content of sediment (fraction) | 1e-02 | 3e-02 | 5e-02 | 6e-04 | 2e-04 | 1e-04 |
| Fraction lipid in fish (fraction) | 5e-02 | 7e-02 | 2e-01 | 1e-04 | 2e-04 | 6e-04 |
| Fish consumption - Columbia River (kg/kg/d) | 4e-05 | 2e-03 | 2e-03 | 9e-06 | 2e-04 | 3e-04 |
| $K_{oc}$ (L/kg) | congener specific | congener specific | congener specific | 2e-04 | 2e-04 | 2e-04 |

BSAF = biota sediment accumulation factor

ISCST3 dispersion modeling for each of the case #1 model plants was run again with receptors placed every 1000 m on a polar grid out to a distance of 25,000 m. Flat terrain was used in the dispersion remodeling because it was realized that variation in terrain would influence the location of plume impaction and maximum concentration. The resulting data has not been corrected for congener specific emissions rates (unit emissions are used), and it has not been proportioned for vapor and particle phase fractions. However, it

still serves the analysis for determination of maximum values. Table 11-10 presents the highest values for each model plant in terms of wet and dry deposition and air concentration of both particles and vapors. (Note that dry deposition of vapors is not included in this table, as it is not an ISCST3 modeled value [see section 11.3.2.]) The location of each value in Table 11-10 is indicated by x and y coordinates in meters. Table 11-11 presents modeled concentrations at 2.5 km, 10 km, and 25 km, as are reported in the mercury multipathway analysis (chapter 7). The locations in Table 11-11 are described in terms of degrees from the polar grid origin. For this analysis the polar origin was North with degrees progressing clockwise.

Due to the disparate locations of the maximum modeled concentrations for each modeled plant, it is difficult to determine where the single site of maximum risk would be. It is likely that the maximum risks would be observed at the 50 m location because that is the location of maximum wet deposition. In all modeled cases, wet deposition represents a significantly larger share, by an order of magnitude or greater, of the environmental concentration than either dry deposition or air concentration. This data does not indicate that a re-analysis of the model plant risks is necessary because the effort required to determine the exact location of a maximum and average exposure scenario in a multipathway analysis is beyond the scope of this screening-level analysis.

## 11.6 SUMMARY OF RESULTS

The results of this assessment indicate that the predicted cancer risks associated with case #2 exposures to dioxin emissions from the model plants are not greater than $1 \times 10^{-7}$ for the model adult and child resident scenarios. This is the case for both the direct and the indirect exposure pathways. However, risks of $1 \times 10^{-6}$ or above are predicted for the indirect pathways of the case #1 scenarios. In particular, the subsistence fisher scenario results in a predicted risk of $2 \times 10^{-4}$. When factors influencing the magnitude of the risk to the subsistence fisher were examined in a sensitivity analysis, it appeared that the assumptions about the BSAF had the greatest impact on the predicted risk.

Taken one at a time, several other uncertainty factors have the potential to raise the risk estimate for the subsistence fisher above $2 \times 10^{-4}$; these include fish consumption rates, the organic carbon content of sediment, and the fraction of fish tissue that is lipid. This assessment has been conducted using model plant data and two geographic locations selected to represent two potential scenarios. The location of the case #1 receptors was arbitrarily placed at a point of high wet deposition. This assumption was made for the purpose of providing a conservative estimate of the case #1 risk.

The case #1 scenario represents a sub-population that has the potential to receive high exposures due to relatively high ingestion rates, and assumed to be in a location for which the meteorology would

Table 11-10. Maximum Dispersion Modeling Locations and Concentrations

| Model plant phase | Wet deposition location in m (x,y) | Wet deposition ($g/m^2/y$) | Dry deposition location in m (x,y) | Dry deposition ($g/m^2/y$) | Air concentration location in m (x,y) | Air concentration ($\mu g/m^3$) |
|---|---|---|---|---|---|---|
| LCH, vapor | 0, -500 | 0.07430 | - | - | 0, 9000 | 0.00050 |
| LCH, particle | 0, -500 | 0.09158 | Several | 0.00026 | Several | 0.00046 |
| SCH, vapor | 0, -500 | 0.08372 | - | - | Several | 0.00344 |
| SCH, particle | 0, -500 | 0.10274 | 8314, -3444 | 0.00210 | -3826, 9238 | 0.00302 |
| LOH,vapor | 0, -500 | 0.07846 | - | - | Several | 0.00105 |
| LOH, particle | 0, -500 | 0.14166 | Several | 0.00092 | Several | 0.00084 |
| SOH, vapor | 0, -500 | 0.09083 | - | - | -2296, 5543 | 0.00951 |
| SOH, particle | 0, -500 | 0.16233 | 4619, -1813 | 0.00997 | -1913, 4619 | 0.00792 |

also tend to result in high exposures for given rates of release of dioxins. The results of this analysis can be considered a potential representative example for utility boilers, yet they do not represent a specific plant or location. The analysis should be considered an analysis of hypothetical scenarios used for demonstration.

This analysis of noninhalation exposures to dioxin emissions is a screening analysis. Thus, these quantitative exposure and risk results, because of the many modeling and analytic uncertainties, are very uncertain and do not, therefore, conclusively demonstrate the existence of health risks of concern associated with exposures to utility emissions either on a national scale or from any actual individual utility. The lack of measured data around these sources precludes a comparison with modeled results. These results do suggest that exposures and risks of concern cannot at present be ruled out and that there is a need for development of additional scientific information to evaluate whether risk levels of concern may exist.

Table 11-11. Dispersion Modeling Concentrations at Specified Distances

| Model plant, phase | Wet deposition location (°) | Wet deposition ($g/m^2/y$) | Dry deposition location (°) | Dry deposition ($g/m^2/y$) | Air concentration location (°) | Air Concentration ($\mu g/m^3$) |
|---|---|---|---|---|---|---|
| 2.5 km concentrations | | | | | | |
| LCH, vapor | 180 | 0.01342 | - | - | 360 | 0.00015 |
| LCH, particle | 180 | 0.01414 | 360 | 0.00005 | 360 | 0.00014 |
| SCH, vapor | 180 | 0.01406 | - | - | 360 | 0.00040 |
| SCH, particle | 180 | 0.01535 | 360 | 0.00088 | 360 | 0.00196 |
| LOH, vapor | 360 | 0.01406 | - | - | 360 | 0.00040 |
| LOH, particle | 360 | 0.01970 | 360 | 0.00030 | 360 | 0.00036 |
| SOH, vapor | 180 | 0.01577 | - | - | 360 | 0.00681 |
| SOH, particle | 180 | 0.02141 | 112.5 | 0.00768 | 360 | 0.00607 |
| 10 km concentrations | | | | | | |
| LCH, vapor | 180 | 0.00217 | - | - | 360 | 0.00049 |
| LCH, particle | 337.5 | 0.00121 | 337.5, 360 | 0.00019 | 360 | 0.00045 |
| SCH, vapor | 180 | 0.00225 | - | - | 337.5 | 0.00344 |
| SCH, particle | 180 | 0.01890 | 112.5 | 0.00207 | 337.5 | 0.00302 |
| LOH, vapor | 180 | 0.00221 | - | - | 360 | 0.00100 |
| LOH, particle | 180 | 0.00211 | 112.5 | 0.00084 | 360 | 0.00083 |
| SOH, vapor | 180 | 0.00233 | - | - | 337.5 | 0.00852 |
| SOH, particle | 180 | 0.00210 | 112.5 | 0.00675 | 337.5 | 0.00679 |
| 25 km concentrations | | | | | | |
| LCH, vapor | 180 | 0.00046 | - | - | 337.50 | 0.00046 |
| LCH, particle | 180 | 0.00037 | 112.5 | 0.00026 | 135 | 0.00039 |
| SCH, vapor | 180 | 0.00047 | - | - | 337.5 | 0.00256 |
| SCH, particle | 180 | 0.00036 | 112.5 | 0.00099 | 337.5 | 0.00205 |
| LOH, vapor | 180 | 0.00046 | - | - | 337.5 | 0.00103 |
| LOH, particle | 180 | 0.00036 | 112.5 | 0.00080 | 337.5 | 0.00079 |
| SOH, vapor | 180 | 0.00047 | - | - | 360 | 0.00444 |
| SOH, particle | 180 | 0.00033 | 337.5 | 0.00205 | 360 | 0.00355 |

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# 12.0 LITERATURE REVIEW ON THE POTENTIAL IMPACTS OF HYDROGEN CHLORIDE AND HYDROGEN FLUORIDE EMISSIONS

## 12.1 OVERVIEW

The information presented in this chapter and in Appendix H was collected to expand the EPA's knowledge of the potential impacts of HCl and HF emissions from utilities. The details and references are presented in Appendix H. This chapter presents a summary of the findings. The EPA is updating its current state of knowledge of potential health impacts; atmospheric chemistry (e.g., half-life, impacts on the acid rain phenomenon); potential human exposure through pathways other than direct inhalation; and possible ecological harm. The EPA's goal is to understand the potential impacts from HCl and HF emissions to any and all health and environmental areas. This chapter is not intended to provide a detailed, comprehensive treatise on the above subject area; rather, it is designed to provide general technical information that will identify possible problem areas that may call for additional, more detailed research.

Published evidence for potential impacts of HCl and HF was evaluated from a wide variety of sources. Overall, there is extensive information available on the toxicology of these two pollutants; however, literature pertaining specifically to HF and HCl atmospheric chemistry is relatively scarce, especially that pertaining to fine particulate matter and acid rain. Literature on HCl and HF from sources outside the United States and pertaining to emissions sources other than utilities has also been evaluated.

This chapter is organized so that the findings for HCl are presented first, followed by the findings for HF. Within each section, evidence from the literature for transport and transformation through atmospheric, terrestrial, and aquatic processes is presented first, followed by evidence for impacts on human health; vegetation; and wild, domestic, and aquatic animals.

## 12.2 SUMMARY OF FINDINGS

This chapter provides a synopsis of the information of interest found during the literature review on potential impacts (e.g., acid rain, fine particulate matter, visibility, and toxicity to various plant and animal species) of hydrogen chloride and hydrogen fluoride emissions from utilities.

### 12.2.1 Hydrogen Chloride

12.2.1.1 HCl Emissions and Formation. Utilities emit a substantial amount of the anthropogenic atmospheric emissions of HCl in the United States.[1] As shown in chapter 3 of this report, utilities were estimated to emit 146,000 tpy of HCl in the United States in 1990. Other important sources of HCl are industrial coal combustion and solid waste combustion. Natural sources of HCl emissions include

volcanic activity, marine plants and microorganisms, and land plant combustion.

According to available information, ambient concentrations of HCl in the United State ranged from none detected to 4 $\mu g/m^3$. Rural sites can be expected to be at the low end of this range and urban sites are likely to be toward the high end. Human health effects are discussed in Appendix E.

HCl can be formed several ways in the atmosphere. Anthropogenic chlorocarbons can react with OH radicals to produce small amounts of HCl. Nonanthropogenic HCl can be formed from deliquescent sea-salt in the marine environment. HCl can also be created or destroyed through the interaction between fogwater and aerosols. Although information on HCl formation by-products is scarce, available sources indicate that reactions generating HCl can produce the following by-products in the atmosphere: $NaNO_3$, $Na_2SO_4$, hydrocarbon radicals, and $NH_3$.

12.2.1.2 HCl Atmospheric Processes. The atmospheric lifetime of HCl is estimated to be between 1 and 5 days. HCl is a highly reactive gas that is removed from the atmosphere via wet and dry deposition.

12.2.1.3 HCl Atmospheric Transport. In general, because of its high solubility, HCl will be removed from the atmosphere much faster than $SO_2$ or $NO_2$ and will be deposited in close proximity to the emissions source. However, conditions exist under which $SO_2$ is subject to further transport. One study found evidence that HCl-enriched plumes, believed to originate from coal-burning utilities, reached a rural site two days after emissions release. Evidence was found that HCl may affect the atmospheric chemistry of mercury, and thus the toxicity of mercury emissions from utilities. HCl emissions are believed to contribute to some limited degree to the formation of atmospheric acidity and acid rain. In addition, HCl appears to indirectly contribute to some limited degree, to fine PM and visibility problems. However, there are significant uncertainties as to the extent of the impacts due to HCl emissions in these areas. Further research and evaluation are needed to determine if, and to what extent HCl contributes to acid rain, fine PM, and visibility issues.

12.2.1.4 HCl Terrestrial Processes. Information on the terrestrial behavior of HCl is scarce. The references found discuss the evidence that HCl can lower pH to the point that $SO_2$ oxidation is delayed, possibly altering the spatial deposition of acid species, and that gaseous HCl damages limestone.

12.2.1.5 HCl Aquatic Processes. The chemistry of ubiquitous chlorinated compounds in natural waters is affected by a number of factors that determine the persistence and toxicity of interim species. Chloride cycling in watersheds was found to be more complex than previously thought. The traditional view has been that atmospherically deposited chloride is rapidly transported. Further

research of chloride cycling would be needed to gain a better understanding of the aquatic processes.

12.2.1.6 HCl Human Health Impacts. Evidence of local irritation to the upper respiratory tract by HCl was found, and long-term exposure may cause tooth erosion. The WHO concluded in a review on HCl that there are no mutagenic, carcinogenic, or teratogenic effects related to HCl. Appendix E contains more information on health effects.

12.2.1.7 HCl Vegetation Impacts. Atmospheric and leaf chloride levels were found to be closely correlated. Atmospheric chloride can concentrate in and cause damage to foliar tissues. High chloride concentrations resulted in tissue death in coastal vegetation.

12.2.1.8 HCl Terrestrial Animal Impacts. HCl that reaches plants and soil via wet and dry deposition is then available for uptake by animals. The adverse health symptoms of HCl exposure in animals include eye, nasal, and respiratory tract irritation, with the respiratory tract being the primary target.

12.2.1.9 HCl Aquatic Animal Impacts. Toxicity of chlorinated compounds to aquatic biota varies widely. One study reported the 24-hour $LC_{50}$ for one marine species as 0.0018 mg/L, and the $LC_{50}$ for one sensitive freshwater species as 0.003 mg/L.

12.2.2 Hydrogen Fluoride

12.2.2.1 HF Emissions and Formation. Anthropogenic sources are responsible for most atmospheric fluoride. Anthropogenic emissions of hydrogen fluoride (HF) originate from coal combustion and the aluminum, phosphate, and steel-making industries. As shown in chapter 3, utilities are estimated to emit approximately 19,600 tpy of HF nationwide. Volcanoes are the primary natural sources of HF. Ocean spray, fires, and dust from soil and rock weathering contribute relatively minor amounts of fluoride to the atmosphere. Measured concentrations of HF at one monitoring station in the United States ranged between 1 $\mu g/m^3$ and 8 $\mu g/m^3$. Atmospheric concentrations of fluoride in remote rural areas are reported to be approximately 0.1 $\mu g/m^3$, which is at the limit of detection. Estimates of the relative proportions of gaseous to particulate emissions of industrial fluoride vary. Fluoride particulates range from distinct minerals to alumina with HF adsorbed to its surface, and particle diameters range from <0.1 µm to approximately 10 µm. Volcanic emissions are usually not predominately HF, but rather other fluoride-containing compounds that react in the atmosphere to form HF.

12.2.2.2 HF Atmospheric Processes. HF is described as moderately persistent in the atmosphere, with an estimated lifetime of approximately 1 to 5 days. Wet and dry deposition are the primary routes of HF removal from the atmosphere. HF atmospheric reaction products are primarily fluoride salts. HF does not biodegrade.

12.2.2.3 HF Atmospheric Transport. Fluoride emissions from utilities are transported on a regional scale. Fluoride has been used as an atmospheric tracer because the fluoride to sulfur oxide ratio is relatively constant in the coal-fired utilities examined, providing a characteristic utility emissions fingerprint. Evidence was found that coal-fired utility emissions contribute to measured concentrations of atmospheric fluoride at distances of up to 500 km. Another study found that elevated levels (18-21 ppm) of fluorides emitted from a utility could be detected in grape leaves at distances of up to 37 km. However, it has been reported that the measurement of fluoride transport and deposition have problems of accuracy due to limitations of analytical methods.

12.2.2.4 HF Terrestrial Processes. Fluoride is lost from the various surfaces on which it is deposited and leaves ecosystems at a rapid rate. The volatilization pathway as a route of fluoride export from ecosystems needs further investigation. Soil can be both a sink and source of fluoride, but fluoride is not usually available or labile in soils. Soluble fluoride-containing process water and leachate of phosphogypsum were shown to dissolve much of the fine soil clay fraction, as well as the smectite of the coarse clay fraction of soils. Several conditions are reported to facilitate rapid uptake of water-soluble fluoride and transport. The natural buffering capacity of soils or water, or dilution can reduce acidity added by the presence of HF. Sulfate and fluoride were found to slightly retard aluminum's mobility through soils.

12.2.2.5 HF Aquatic Processes. Fluoride is a major component of seawater, and natural and anthropogenic fluoride may accumulate in waterbody sediments. In freshwaters with pH greater than 5, fluoride is mainly present as fluoride ion.

12.2.2.6 HF Human Health Impacts. Adverse effects of fluoride on human health include dental fluorosis, gastric disturbances, reductions in urinary concentrating ability, skeletal fluorosis, and even death. Optimally fluoridated water has been shown not to be associated with a detectable risk of cancer in humans. Fluoride exposure is not associated with birth defects, and there is no indication that organ systems are affected by chronic, low-level fluoride exposure. Genotoxicity studies have yielded contradictory results. There is disagreement about whether the increased prevalence of dental fluorisis observed in the United States since the 1940s is a toxic effect. Crippling skeletal fluorosis has not been and is not a public health problem in the United States. Beneficial effects of high fluoride regimens in reducing osteoporosis have not been demonstrated. Further epidemiological studies are required to determine whether or not an association exists between various levels of fluoride in the drinking water and bone fractures.[2] Appendix E contains more information on health effects of HF.

12.2.2.7 HF Vegetation Impacts. Inherent differences in the resistance of some tropical tree species to fluoride may be related to

their capacity to accumulate aluminum. Although some plant species are tolerant of elevated fluoride levels, the storage of large amounts of fluoride in plant tissues may present a risk to ecosystems. Plant uptake of fluoride was found to be limited to the smaller, water-soluble, and labile fractions. The major pathway of fluoride to plants is atmospheric deposition.

Atmospheric fluoride is capable of injuring certain plant species at lower concentrations than any other air pollutant. However, most plant species are relatively resistant to fluoride. No morphological damage to lichen species exposed to high concentrations of fluoride was found. Fluoride was found to be the most important pollutant contributing to vegetation damage in one section of a tropical rainforest.

12.2.2.8 HF Terrestrial Animal Impacts. Two studies on the toxicity of fluoride to several rodent species found the animals to exhibit visible incisor lesions after fluoride ingestion. Bone fluoride loads in four predatory bird species were found to be greater in males of all species examined and higher than average in more industrial regions.

Conflicting information was found concerning whether fluoride accumulates in food chains. Variations in fluoride concentrations within plant organs can result in animal species with differing feeding niches ingesting different amounts of fluoride. Contamination of foliage with soil may constitute an important route of fluoride transfer to large herbivores in situations where soil has been treated with phosphate fertilizer or exposed to substantial airborne deposition of fluoride.

12.2.2.9 HF Aquatic Animal Impacts. Biomagnification in aquatic animals is reported to be negligible to very slight. Two trout species were demonstrated to be more resistant to fluoride than freshwater benthic macroinvertebrates. Diatoms appear to be tolerant of, and stimulated to grow by, high fluoride concentrations; the ecological significance of this is uncertain. Limited evidence exists for fluoride-containing effluent effects on both abundance and diversity of estuarine/marine organisms at relatively low fluoride levels.

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# 13.0 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 potential health risks. 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 emission quantities of the compounds).

## 13.1 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 are 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.

### 13.1.1 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 potential concern with fuel switching is whether or not it will increase HAP emissions, due to potentially increased concentrations of trace elements in the fuel and different fly ash characteristics that impact effective PM and HAP control with existing APCDs (e.g., ESPs).

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 13-1 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 13-1, 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 13-1. 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 | |

[a] The coal data listed in Table 13-1 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 13-1.

[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 dat 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.

13.1.1.1 Switching to Natural Gas Combustion. As shown in Table 13-1, 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.

13.1.1.2 Switching from Coal to Residual Oil Combustion. As shown in Table 13-1, 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.

13.1.1.3 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 13-1 shows the relationship between the concentrations of mercury and sulfur in 153 samples of coal shipments.[2] As shown in Figure 13-1, 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 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

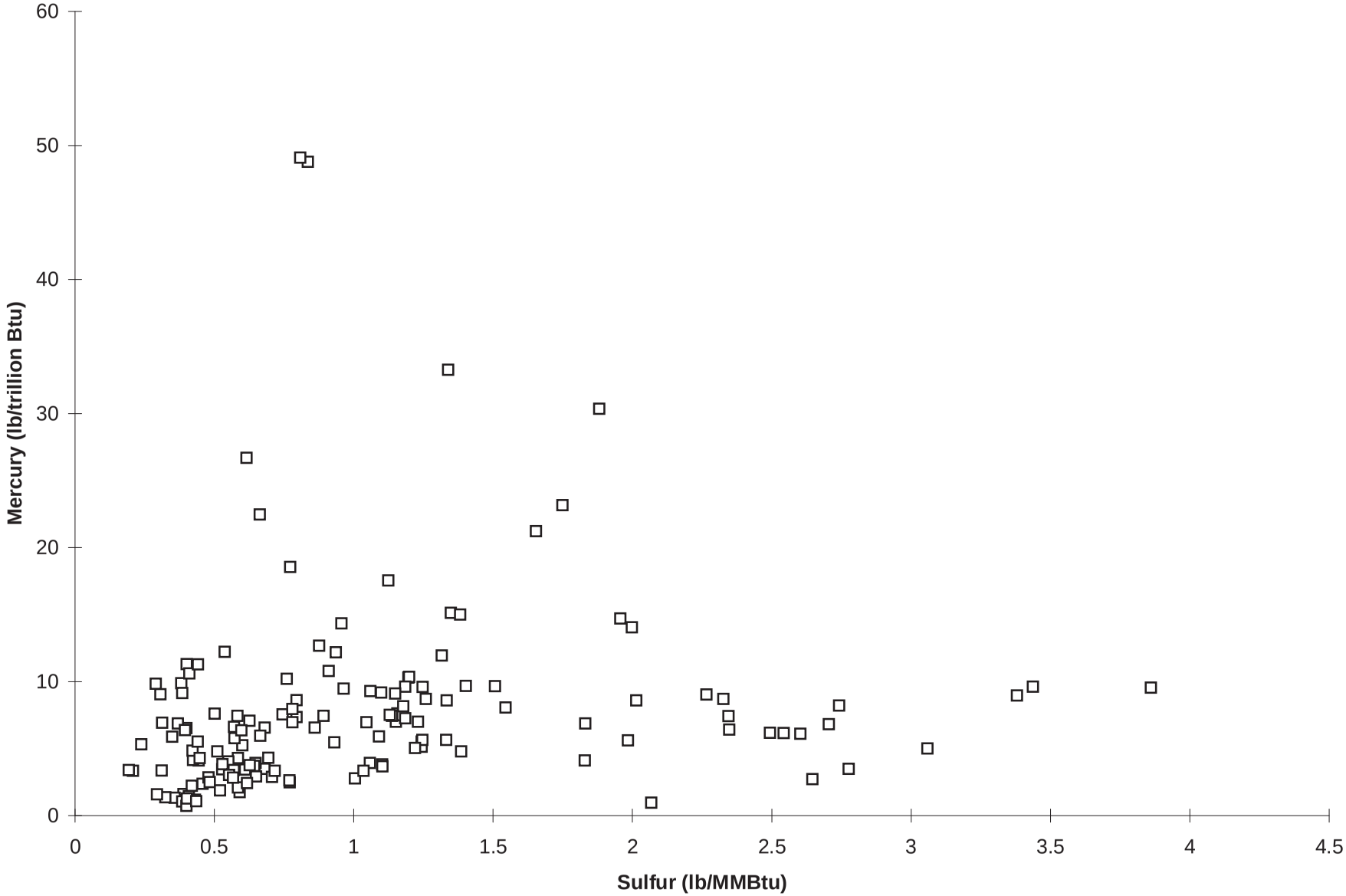



Figure 13-1. Relation between the concentrations of mercury and sulfur in 153 samples of coal shipments.[2]

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 would be the possibility of changing the distribution of the elemental and oxidized forms of mercury that could affect mercury control with existing control devices. Another factor is that the blending of two different coals might change the higher heating value of the resulting mixture, with subsequent effects on the quantity of fuel required for combustion.

The qualitative effects of switching to lower-sulfur-containing coals on other metallic HAPs are examined in Figure 13-2(a-g) through plots of the average concentrations of each HAP, excluding mercury, with sulfur content in coal. As shown in Figure 13-2(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.

### 13.1.2 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. 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 or by surface-based floatation).[5] In the process of removing the mineral matter, coal cleaning generates solid refuse that contains trace elements; the solid refuse must be disposed of properly. Any coal cleaning liquid wastes will also contain trace elements, but the liquid wastes may be properly clarified and then recycled.

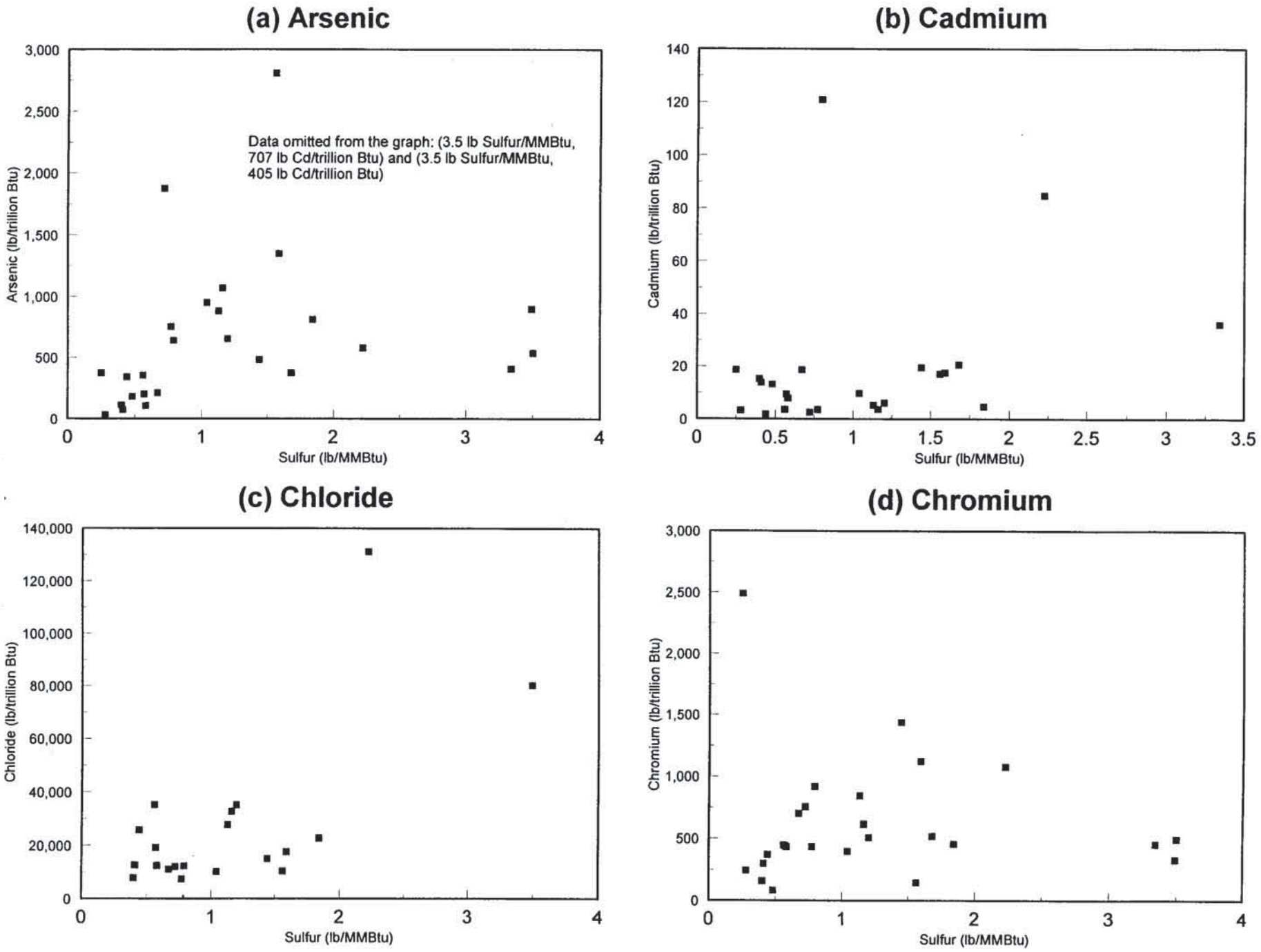
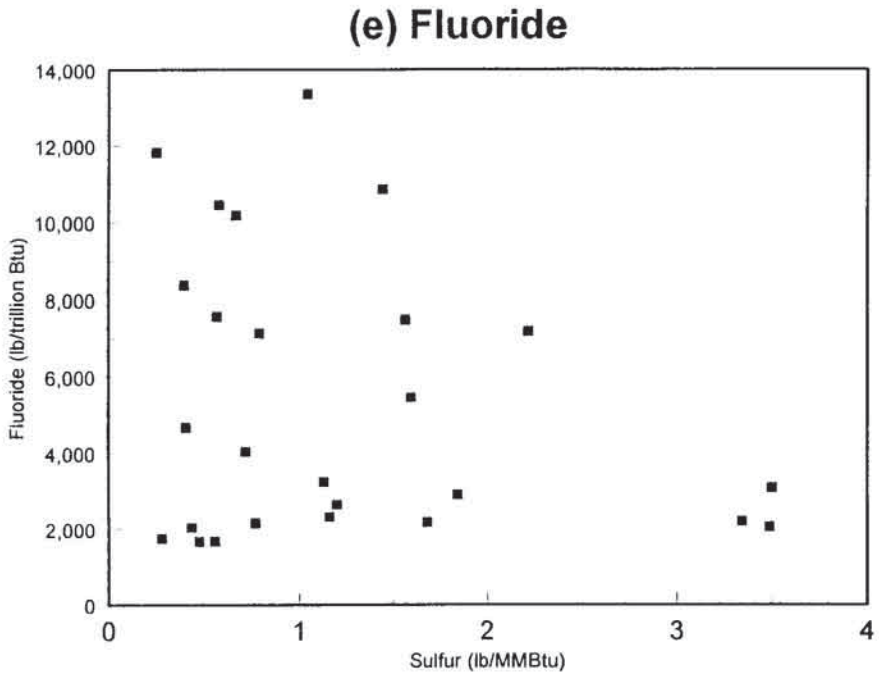
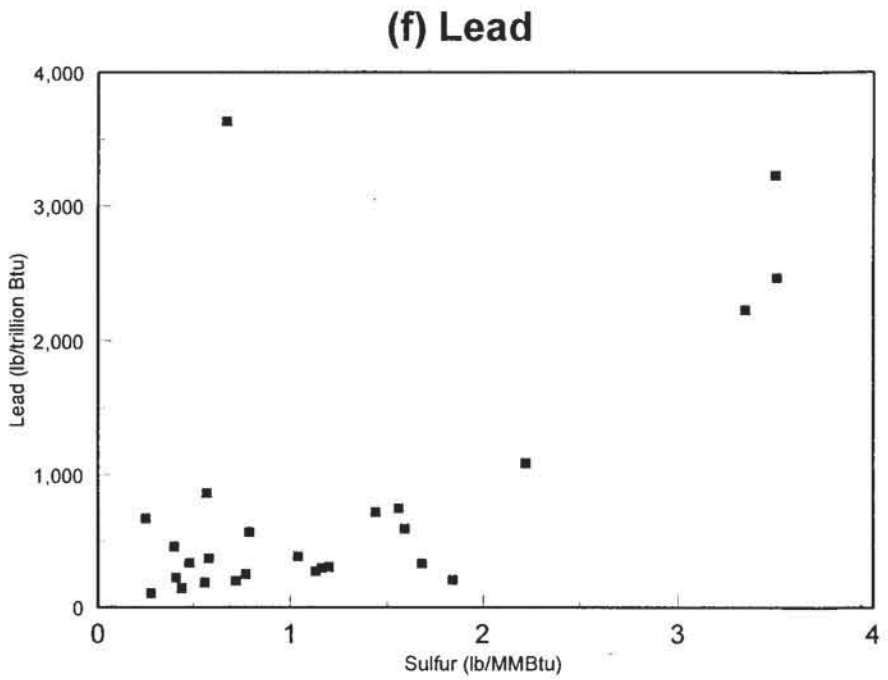



Figure 13-2 (a-g). Relation between concentration of selected trace elements and sulfur in modified USGS data.



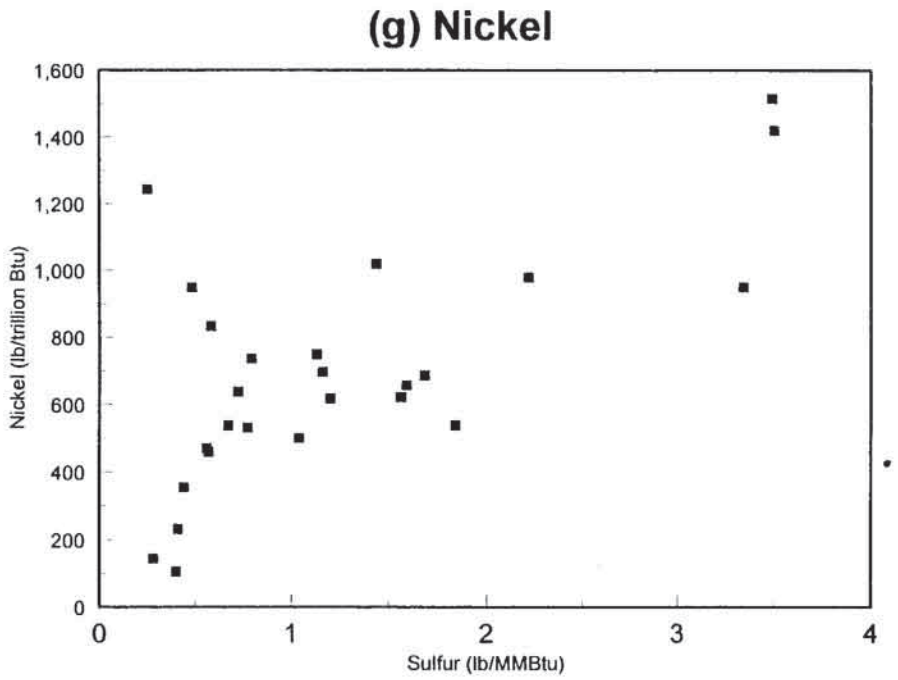




Figure 13-2 (a-g). continued.



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Table 13-2 lists the limited amount of available data on trace element reductions achieved through conventional coal cleaning. In Table 13-2, 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 13-2, 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 13-2, 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]

The average trace element reductions, listed in Table 13-2 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 13-2, 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. Additional information on advanced coal cleaning is provided in section 13.6.1.

# Table 13-2. 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 |

[a] Negative percentages are listed as entries. However, averages were determined with negative percentages treated as zero removal.

Table 13-2. (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.

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.

### 13.1.3 Coal Gasification

Coal gasification converts coal to a syngas-fuel 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 (IGCC) technology 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 13-3, 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.

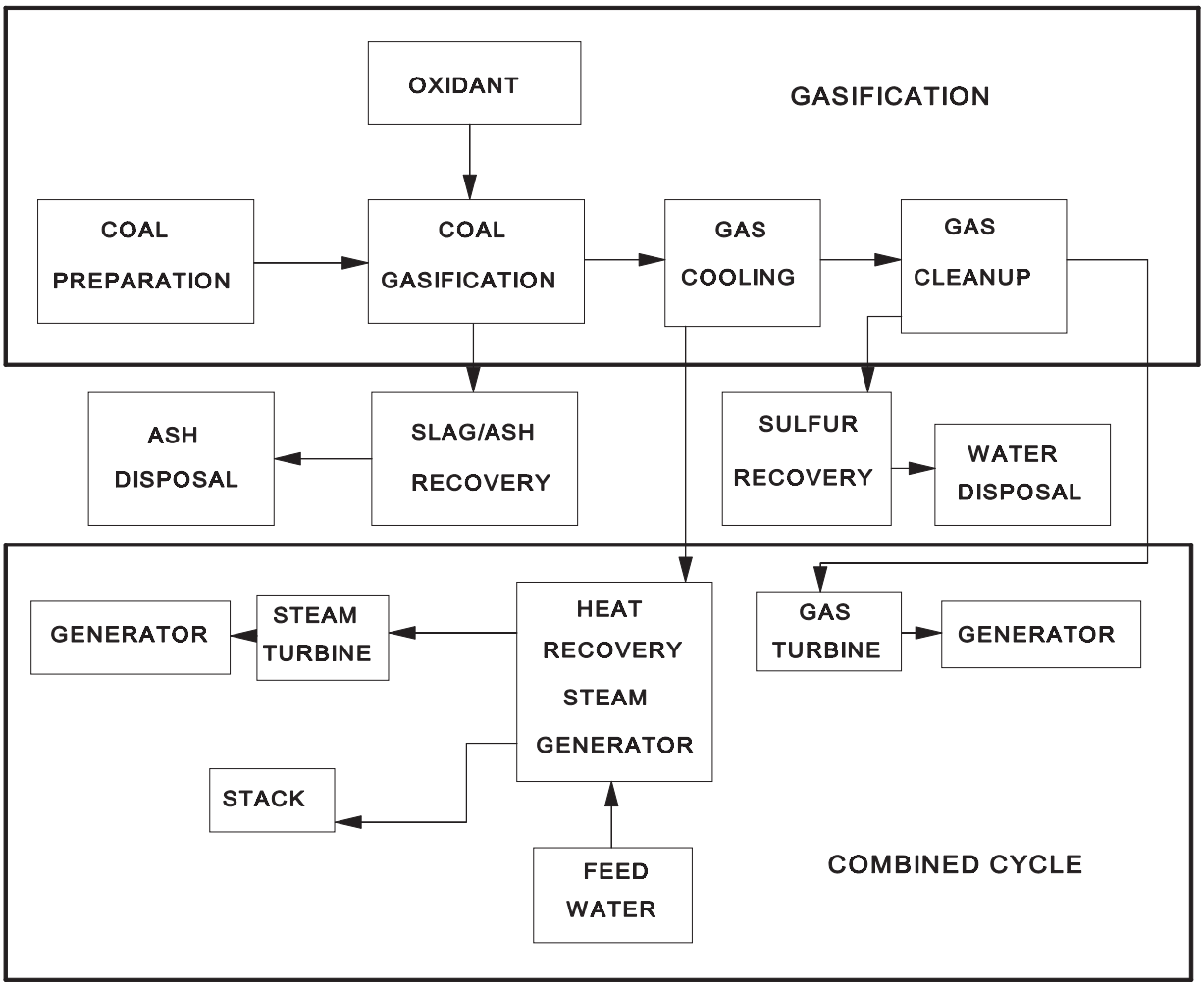



Figure 13-3. Coal gasification combined cycle technology.

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 13-3.[10] The hot gas from the gasifier was treated in a moving bed with zinc titanate sorbent,

## 13.2 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 the recent emission testing on utility units provided a significant amount of information on the generation and control of trace metals but considerably less information on organic HAPs, trace metals are used

Table 13-3. Emissions from an Air-Blown, Fixed-Bed Gasifier

| Trace metal | Emissions to flare, μg/Nm$^3$ | Emissions from turbine simulator, μg/Nm$^3$ | Total air, μg/Nm$^3$ |
|---|---|---|---|
| **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 |

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.[11] 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.[12] 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 13-4, there appears to be a trend toward reductions in HAP emissions through the addition of $NO_x$ control. However, this trend is neither uniform

Table 13-4. 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) [13-16]

| 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.



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(see arsenic, beryllium, and cadmium percentage change between Sites 110 and 114) nor universal (see Site 16 compared to 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 13-5) 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 13-5. 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, lead, and mercury seemed to be emitted in higher amounts by wet bottom units while beryllium, cadmium, chromium, and manganese seemed to be emitted in higher amounts by dry bottom units (see Figure 13-4a). When units firing only bituminous coal were analyzed, the same effect was observed (see Figure 13-5a). 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 13-6a). 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 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.

**Table 13-5. Comparison of wet bottom vs. dry bottom electric utility boilers metallic HAP emissions, trace element removal, and trace metal concentrations in feed coal** [13,14,16-33]

| Compound | | EPRI Site 114 (without $NO_x$ 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) | DOE Boswell | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Bottom Type** | | WET | WET | WET | WET | WET | WET | WET | | | | |
| **Furnace Type** | | Cyclone | Cyclone | Cyclone | Cyclone | Cyclone | Cyclone | Cyclone | | | | |
| | Fuel type | BIT | BIT | BIT | BIT | SUB | SUB | SUB | | Total bituminous & subbituminous | Bituminous | Subbituminous |
| | Boiler emissions (lb/trillion Btu) | | | | | | | | | Average emissions | Average emissions | Average emissions |
| Arsenic | | 137.2 | 151.4 | 2349.3 | 1583.3 | 32.4 | 51.4 | 34.4 | | 620 | 1055 | 39 |
| Beryllium | | 39.9 | 36.2 | 55.6 | 41.3 | 5.0 | 3.1 | 11.7 | | 28 | 43 | 7 |
| Cadmium | | 17.4 | 20.4 | 4.8 | 2.6 | 2.1 | 1.9 | 11.0 | | 9 | 11 | 5 |
| Chromium | | 241.8 | 164.6 | 434.0 | 366.1 | 54.7 | 146.0 | 446.7 | | 265 | 302 | 216 |
| Lead | | 1365.7 | 1007.2 | 637.5 | 593.0 | 56.3 | 152.0 | 176.5 | | 570 | 901 | 128 |
| Manganese | | 392.1 | 375.2 | 420.4 | 322.5 | 66.4 | 599.0 | 6278.5 | | 1208 | 378 | 2315 |
| Mercury | | 6.5 | 4.5 | 21.3 | 19.7 | 3.0 | 6.4 | 4.7 | | 9 | 13 | 5 |
| | Boiler 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 | 0.23 | | 56% | 54% | 59% |
| Beryllium | | 0.15 | 0.15 | 0.30 | 0.26 | 0.08 | 0.04 | 0.60 | | 77% | 79% | 76% |
| Cadmium | | 0.01 | 0.01 | 0.20 | 0.11 | 0.16 | 0.02 | 1.00 | | 79% | 92% | 61% |
| Chromium | | 0.30 | 0.23 | 0.35 | 0.28 | 0.22 | 0.25 | 1.00 | | 63% | 71% | 51% |
| Lead | | 0.50 | 0.84 | 0.60 | 0.56 | 0.38 | 0.61 | 0.42 | | 44% | 38% | 53% |
| Manganese | | 0.20 | 0.18 | 0.19 | 0.15 | 0.13 | 0.33 | 0.57 | | 75% | 82% | 66% |
| Mercury | | 0.73 | 0.54 | 1.00 | 1.00 | 1.00 | 1.00 | 0.87 | | 12% | 18% | 4% |
| | Trace metal concentrations in feed coal (microgram/gram) | | | | | | | | | Average trace metal concentrations in feed coal (microgram/gram) | Average trace metal concentrations in feed coal (microgram/gram) | Average trace metal concentrations in feed coal (microgram/gram) |
| Arsenic | | 12.0 | 8.1 | 34.0 | 33.3 | 0.7 | 1.3 | 1.4 | | 13.0 | 21.9 | 1.1 |
| Beryllium | | 3.6 | 3.3 | 2.3 | 1.9 | 0.7 | 0.9 | 0.2 | | 1.8 | 2.8 | 0.6 |
| Cadmium | | 39.0 | 47.0 | 0.3 | 0.3 | 0.2 | 1.0 | 0.1 | | 12.5 | 21.7 | 0.4 |
| Chromium | | 11.0 | 9.4 | 15.3 | 16.0 | 2.7 | 7.0 | 3.2 | | 9.2 | 12.9 | 4.3 |
| Lead | | 37.0 | 16.0 | 13.0 | 13.0 | 1.6 | 3.0 | 3.9 | | 12.5 | 19.8 | 2.8 |
| Manganese | | 26.5 | 27.0 | 26.7 | 25.3 | 5.5 | 22.0 | 102.0 | | 33.6 | 26.4 | 43.2 |
| Mercury | | 0.12 | 0.11 | 0.26 | 0.21 | 0.03 | 0.06 | 0.05 | | 0.12 | 0.17 | 0.05 |

BIT = bituminous
SUB = subbituminous
SNOX = wet sulfuric acid-selective catalytic reduction process

**Table 13-5. Continued**

| Compound | | EPRI Site 110 (w/LNB and offset air) | EPRI Site 110 (w/LNB) | DOE Yates | EPRI Site 15 | DOE Cardinal | EPRI Site 116 | EPRI Site 12 | EPRI Site 14 | EPRI Site 115 | EPRI Site 16 OFA | EPRI Site 16 OFA/LNB |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Bottom Type** | | DRY | DRY | DRY | DRY | DRY | DRY | DRY | DRY | DRY | DRY | DRY |
| **Furnace Type** | | Tangential | Tangential | Tangential | Tangential | Opposed | Front | Opposed | Opposed | Vertical | Opposed | Opposed |
| | Fuel type | BIT | BIT | BIT | BIT | BIT | BIT | BIT | BIT | BIT | BIT | BIT |
| | Boiler emissions (lb/trillion Btu) | | | | | | | | | | | |
| Arsenic | | 64.2 | 141.7 | 393.1 | 598.9 | 1043.2 | 61.5 | 498.4 | 230.4 | 38.2 | 1789.0 | 1805.9 |
| Beryllium | | 45.3 | 43.9 | 90.1 | 45.8 | 113.0 | 4.1 | 58.6 | 30.1 | 17.5 | 108.3 | 130.4 |
| Cadmium | | 120.9 | 448.8 | 23.8 | 7.4 | 32.4 | 0.6 | 33.8 | 5.3 | 4.0 | 10.9 | 21.1 |
| Chromium | | 743.0 | 745.1 | 2798.8 | 1164.3 | 976.0 | 71.8 | 1252.7 | 735.6 | 87.5 | 934.3 | 845.8 |
| Lead | | 503.2 | 878.8 | 688.6 | 811.3 | 623.0 | 23.9 | 381.2 | 115.2 | 167.1 | 437.3 | 351.2 |
| Manganese | | 999.7 | 1021.8 | 2055.9 | 1752.6 | 1350.8 | 71.8 | 1723.6 | 1152.2 | 334.3 | 745.4 | 890.9 |
| Mercury | | 0.02 | | 12.4 | | 1.7 | 2.2 | 7.6 | 0.8 | 1.7 | 7.1 | 10.7 |
| | Boiler emission modification factors | | | | | | | | | | | |
| Arsenic | | 0.89 | 0.39 | 1.00 | 0.60 | 0.91 | 0.70 | 1.00 | 0.50 | 0.61 | 1.00 | 1.00 |
| Beryllium | | 0.93 | 0.43 | 1.00 | 0.54 | 0.96 | 0.35 | 1.00 | 0.92 | 0.52 | 1.00 | 0.82 |
| Cadmium | | 1.00 | 0.70 | 1.00 | 0.01 | 1.00 | 0.12 | 0.14 | 0.02 | 0.58 | 1.00 | 0.11 |
| Chromium | | 1.00 | 1.00 | 1.00 | 0.58 | 0.61 | 0.27 | 1.00 | 0.67 | 0.57 | 0.58 | 0.69 |
| Lead | | 1.00 | 0.36 | 1.00 | 1.00 | 1.00 | 0.26 | 1.00 | 0.79 | 0.38 | 1.00 | 0.66 |
| Manganese | | 0.71 | 0.76 | 1.00 | 0.81 | 0.27 | 0.21 | 1.00 | 0.93 | 0.58 | 0.60 | 0.88 |
| Mercury | | 0.66 | | 1.00 | | 0.41 | 0.97 | 0.74 | 0.74 | 0.78 | 0.64 | 1.00 |
| | Trace metal concentrations in feed coal (microgram/gram) | | | | | | | | | | | |
| Arsenic | | 2.0 | 1.9 | 2.3 | | 12.9 | 5.0 | 6.2 | 6.3 | 0.5 | 17.0 | 23.0 |
| Beryllium | | 1.3 | 0.6 | 1.1 | 1.1 | 1.3 | 0.7 | 0.7 | 0.5 | 0.2 | 1.4 | 2.2 |
| Cadmium | | 2.1 | 1.7 | 0.3 | 8.0 | 0.1 | 0.3 | 3.4 | 3.0 | 0.1 | 0.1 | 2.6 |
| Chromium | | 9.0 | 5.5 | 24.8 | 26.0 | 18.0 | 15.0 | 17.0 | 15.0 | 1.1 | 22.0 | 17.0 |
| Lead | | 17.0 | 5.0 | 8.0 | 4.0 | 6.8 | 5.3 | 2.4 | 2.0 | 2.1 | 5.1 | 7.3 |
| Manganese | | 16.0 | 17.0 | 23.4 | 28.0 | 57.3 | 19.0 | 18.0 | 17.0 | 4.2 | 17.0 | 14.0 |
| Mercury | | 0.08 | | 0.08 | | 0.05 | 0.13 | 0.14 | 0.24 | 0.02 | 0.15 | 0.14 |

BIT = bituminous
SUB = subbituminous
SNOX = wet sulfuric acid-selective catalytic reduction process

**Table 13-5. Continued**

| Compound | | EPRI Site 11 | EPRI Site 111 | DOE Springerville | NSP Sherburne 3 | NSP Riverside 6,7 | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Bottom Type** | | DRY | DRY | DRY | DRY | DRY | | | | |
| **Furnace Type** | | Tangential | Opposed | Tangential | Opposed | Front | | | | |
| | Fuel type | SUB | SUB (BIT Mixed) | SUB | SUB | SUB | | Total bituminous & subbituminous | Bituminous | Subbituminous |
| | Boiler emissions (lb/trillion Btu) | | | | | | | Average emissions | Average emissions | Average emissions |
| Arsenic | | 124.1 | 5.8 | 81.0 | 64.4 | 47.1 | | 437 | 606 | 64 |
| Beryllium | | 9.9 | | 93.4 | 21.8 | 9.4 | | 55 | 62 | 34 |
| Cadmium | | 58.7 | 16.6 | 172.4 | 7.0 | 10.3 | | 61 | 64 | 53 |
| Chromium | | 248.2 | | 607.0 | 253.6 | 326.7 | | 786 | 941 | 359 |
| Lead | | 143.1 | | 91.0 | 525.4 | 236.1 | | 398 | 453 | 249 |
| Manganese | | 1406.5 | | 5627.6 | 4978.2 | 480.0 | | 1639 | 1100 | 3123 |
| Mercury | | | | 5.4 | 6.0 | 4.3 | | 5 | 5 | 5 |
| | Boiler 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 | | 0.92 | 0.11 | 0.29 | 0.79 | 0.99 | | 27% | 22% | 38% |
| Beryllium | | 0.79 | | 0.87 | 0.58 | 0.40 | | 24% | 23% | 34% |
| Cadmium | | 0.35 | 0.05 | 1.00 | 0.99 | 0.25 | | 39% | 48% | 47% |
| Chromium | | 0.72 | | 0.68 | 0.49 | 1.00 | | 23% | 27% | 28% |
| Lead | | 1.00 | | 0.19 | 0.49 | 0.19 | | 30% | 23% | 53% |
| Manganese | | 0.98 | | 0.72 | 0.00 | 0.77 | | 24% | 29% | 38% |
| Mercury | | | | 1.00 | 1.00 | 1.00 | | 19% | 23% | 0% |
| | Trace metal concentrations in feed coal (microgram/gram) | | | | | | | Average trace metal concentrations in feed coal (microgram/gram) | Average trace metal concentrations in feed coal (microgram/gram) | Average trace metal concentrations in feed coal (microgram/gram) |
| Arsenic | | 1.6 | 0.6 | 1.5 | 0.7 | 0.4 | | 5.5 | 7.7 | 1.0 |
| Beryllium | | 0.2 | | 1.1 | 0.3 | 0.2 | | 0.9 | 1.0 | 0.4 |
| Cadmium | | 2.0 | 4.0 | 0.5 | 0.1 | 0.4 | | 1.8 | 2.0 | 1.4 |
| Chromium | | 4.1 | | 9.2 | 4.4 | 2.6 | | 12.7 | 15.5 | 5.1 |
| Lead | | 1.4 | | 5.0 | 9.1 | 10.5 | | 6.1 | 5.9 | 6.5 |
| Manganese | | 17.0 | | 80.8 | | 5.4 | | 23.9 | 21.0 | 34.4 |
| Mercury | | | | 0.04 | 0.02 | 0.001 | | 0.09 | 0.11 | 0.02 |

BIT = bituminous
SUB = subbituminous
SNOX = wet sulfuric acid-selective catalytic reduction process

**Figure 13-4a. Average boiler emissions**

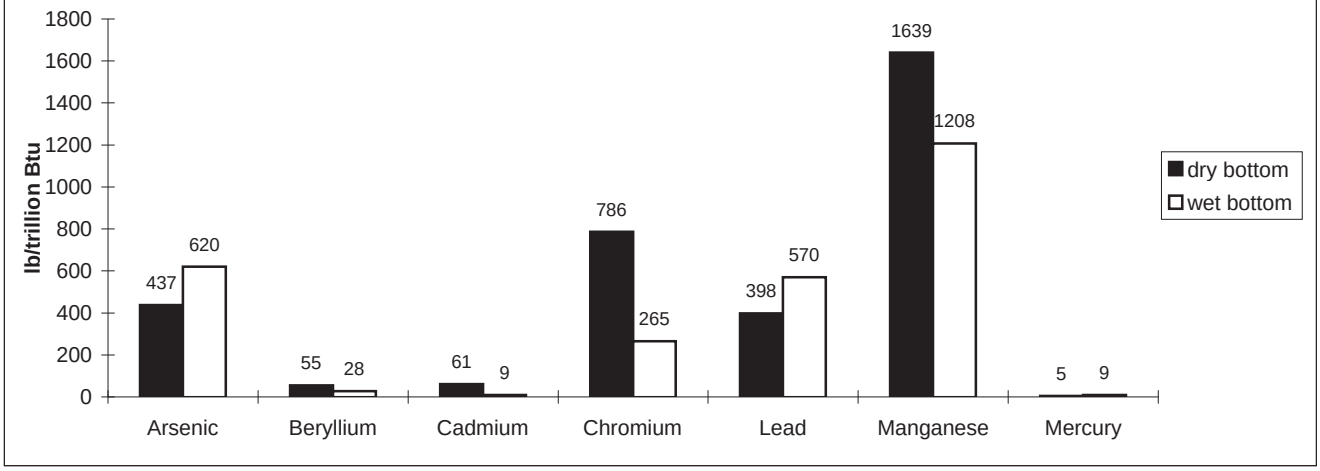



**Figure 13-4b. Average trace metal removal by boiler**

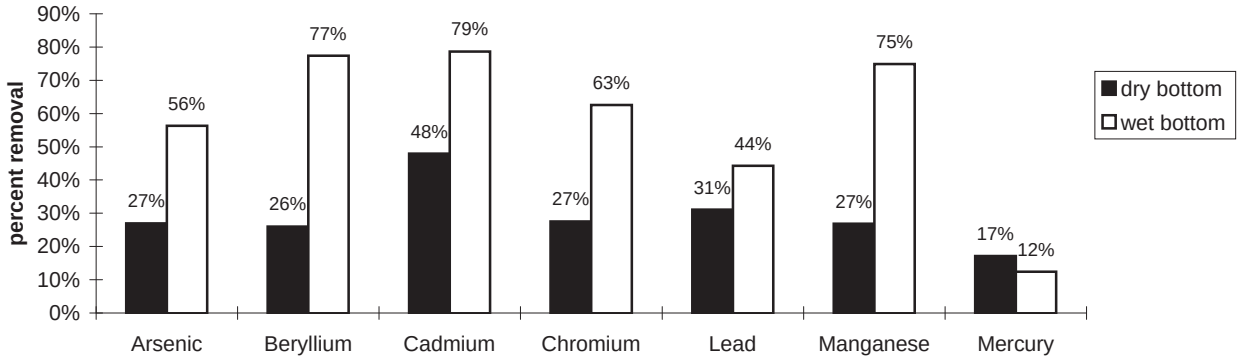



**Figure 13-4c. Average trace metal concentration in feed coal**

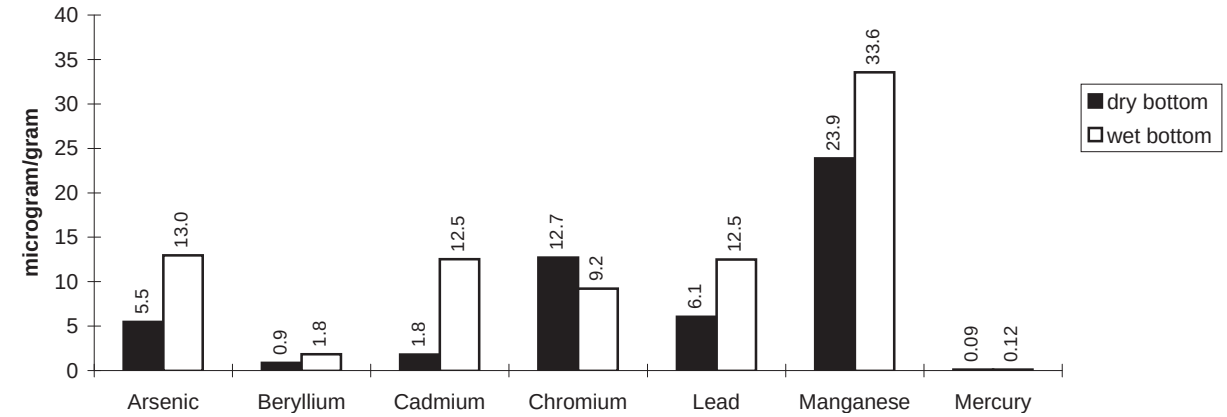



Figure 13-4 (a-c). Average coal-fired boiler emissions, trace metal removal, and average trace element concentration in feed coal vs. bottom type (bituminous and subbituminous coal)

Note: Data taken from Table 13-5.

**Figure 13-5a. Average boiler emissions**

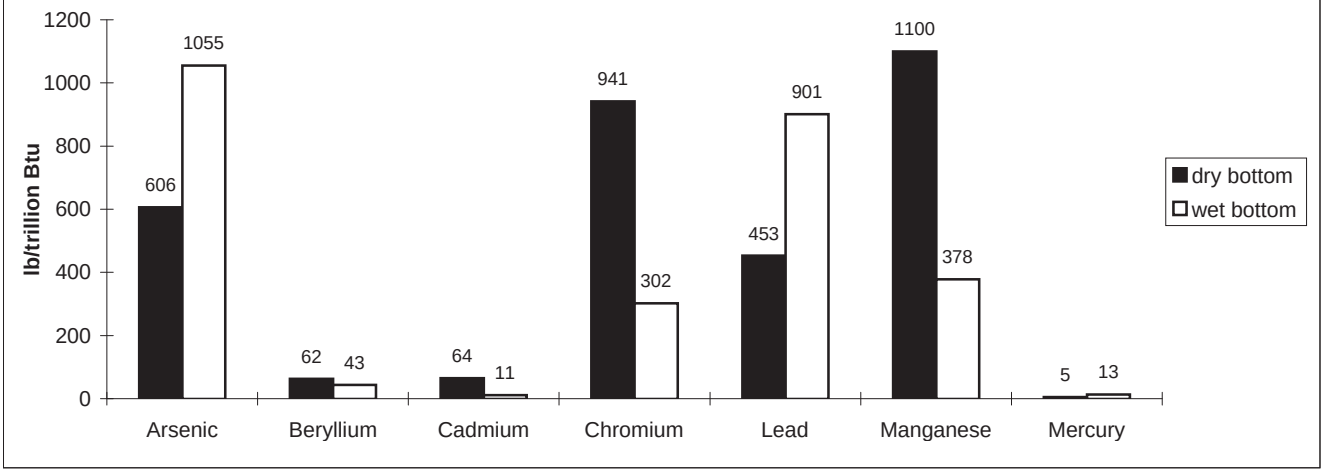



**Figure 13-5b. Average trace metal removal by boiler**

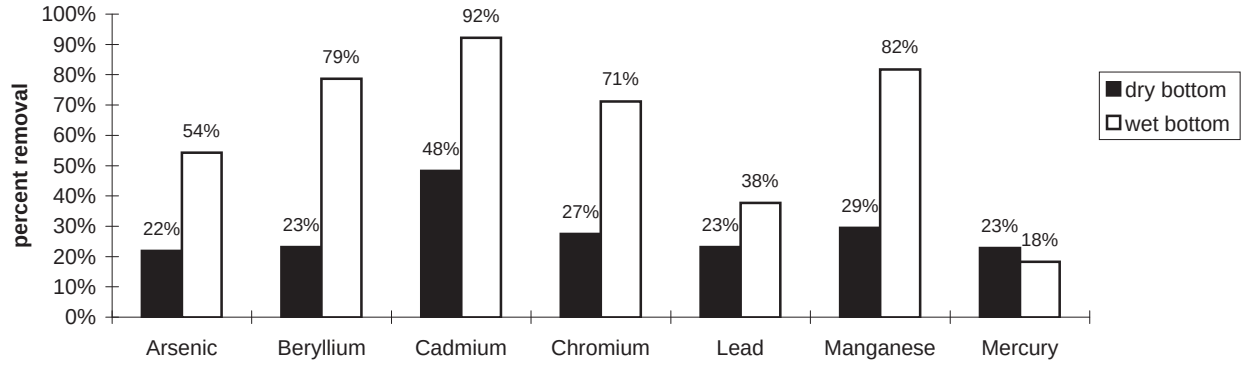



**Figure 13-5c. Average trace metal concentration in feed coal**

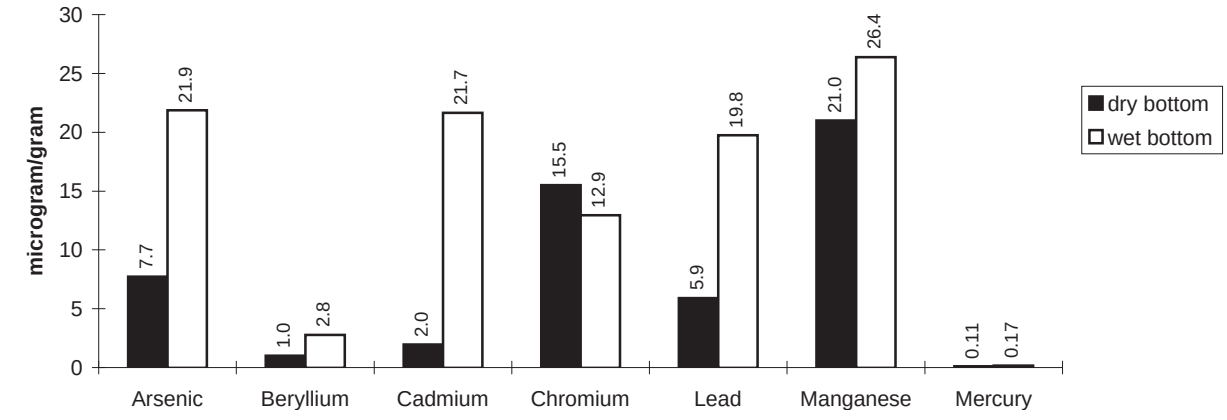
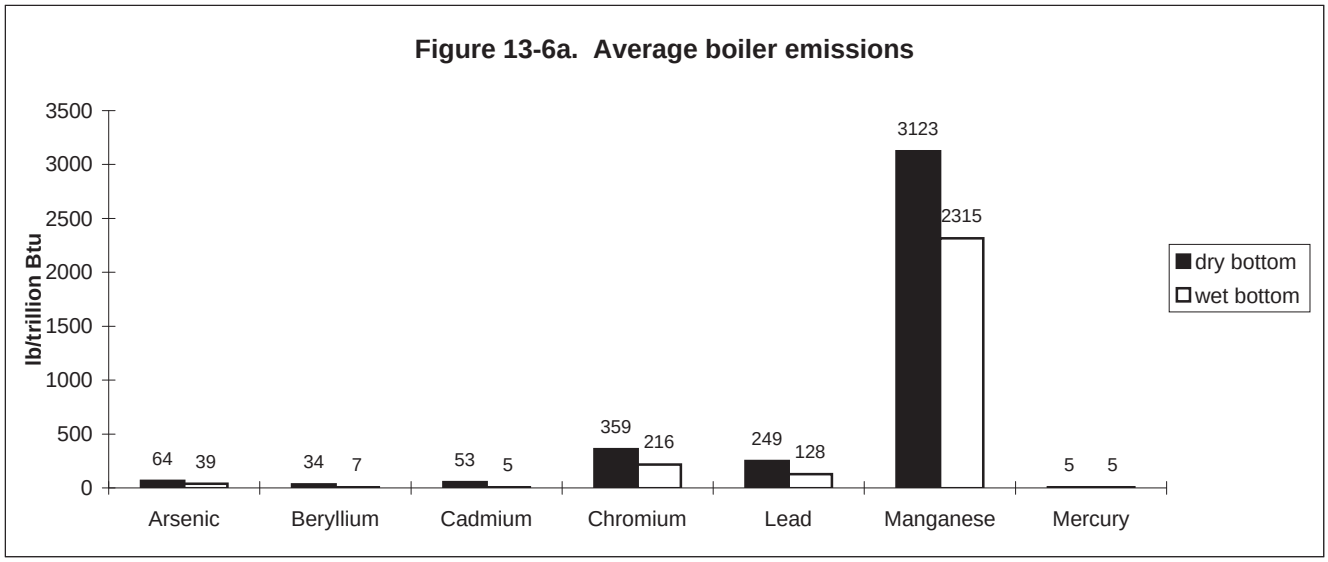
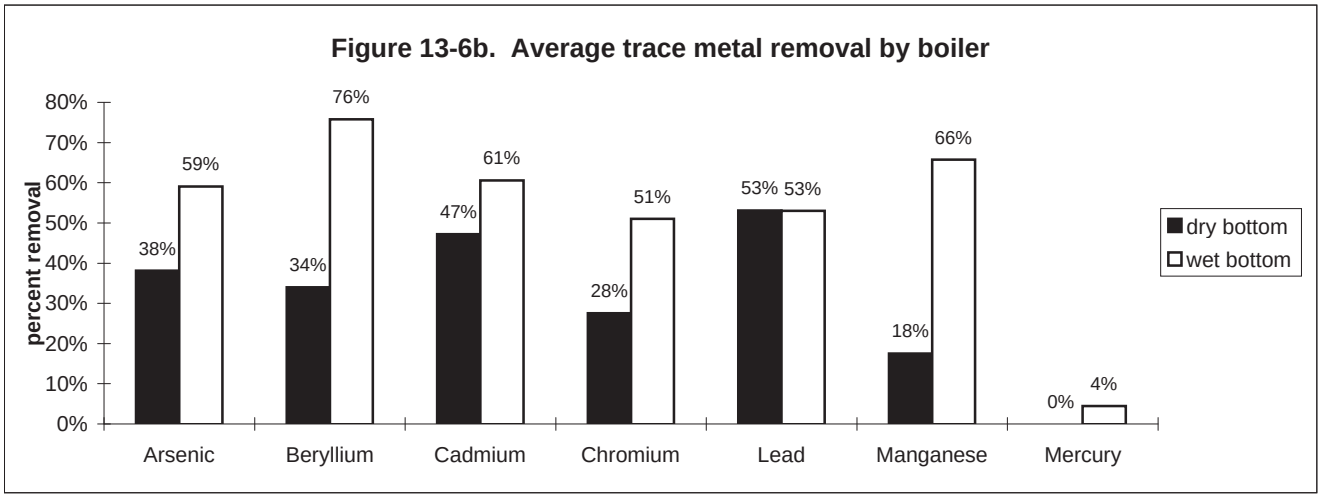



Figure 13-5 (a-c). Average coal-fired boiler emissions, trace metal removal, and average trace element concentration in feed coal vs. bottom type (bituminous coal-fired only)

Note: Data taken from Table 13-5.



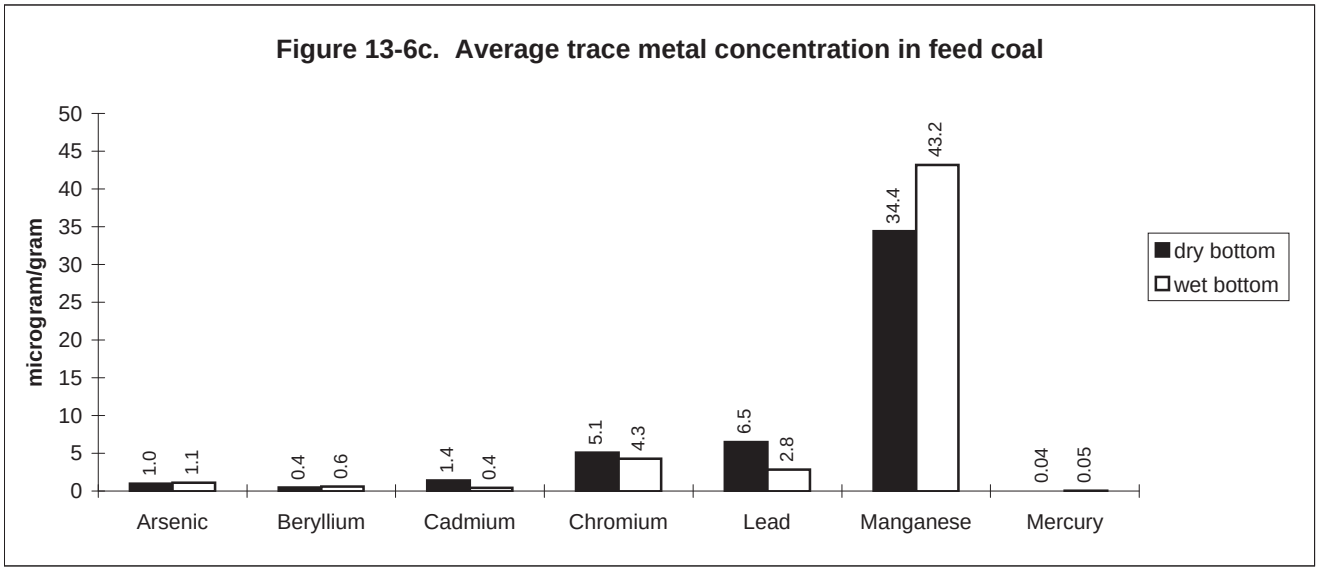




Figure 13-6 (a-c). Average coal-fired boiler emissions, trace metal removal, and average trace element concentration in feed coal vs. bottom type (subbituminous coal-fired only)

Note: Data taken from Table 13-5.

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 13-4(a and b), 13-5(a and b), and 13-6(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 situation did not occur in Figures 13-4(c), 13-5(c), or 13-6(c) because these data were taken directly from the coal feed without modification. The result of this methodology 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.

## 13.3 POSTCOMBUSTION CONTROL

To comply with various local, State, and Federal requirements, utilities routinely use postcombustion technologies for the control of PM and $SO_2$. The following sections assess how different APCDs affect removal of selected HAPs from fossil-fuel-fired electric utility flue gas.

### 13.3.1 Particulate Phase Controls

Figures 13-7 through 13-14 and Tables 13-6 through 13-9 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 3.4.6). 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 particulate form are readily controlled by PM control devices. These HAPs include arsenic, beryllium, cadmium, chromium, lead, and manganese. Table 13-10 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.

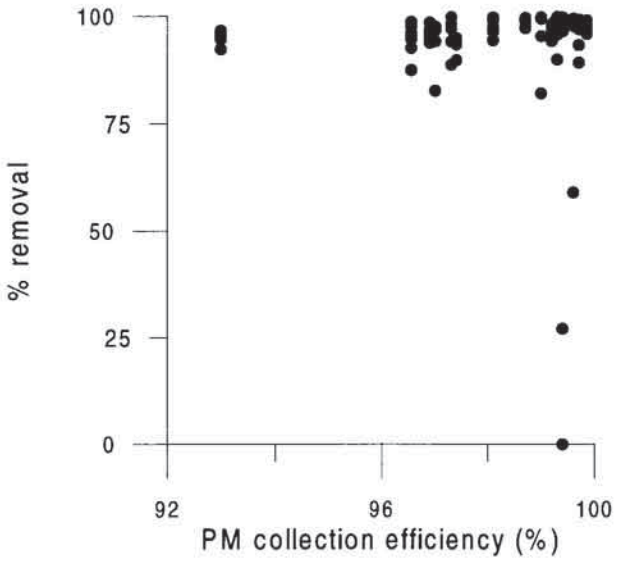



Figure 13-7. Removal of Metallic HAPs by Electrostatic Precipitators (Cold-side, Coal) (Includes, Arsenic, Beryllium, Cadmium, Chromium, Lead, and Manganese)

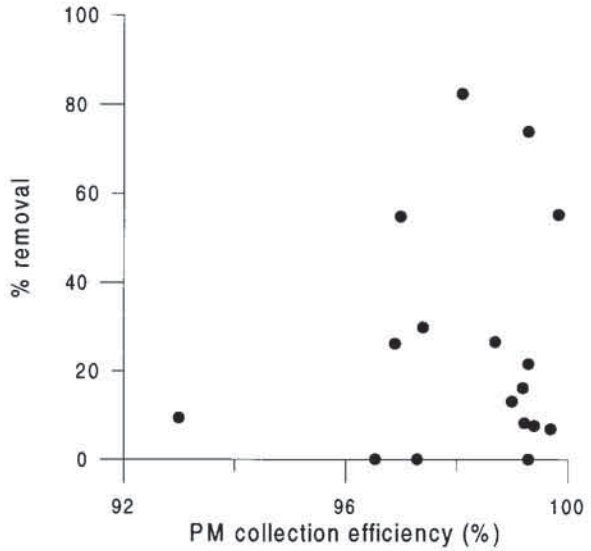



Figure 13-8. Removal of,mercury by electrostatic precipitators (cold-side, coal)

| | Mean | Median | Sdev | Min | Max | Count |
|---|---|---|---|---|---|---|
| Arsenic | 98 | 99 | 2.1 | 94 | >99 | 18 |
| Beryllium | 94 | 98 | 17 | 27 | >99 | 18 |
| Cadmium | 80 | 92 | 31 | 0 | 98 | 18 |
| Chromium | 97 | 97 | 2.3 | 93 | >99 | 18 |
| Lead | 93 | 97 | 15 | 34 | >99 | 18 |
| Manganese | 98 | 97 | 1.5 | 95 | >99 | 18 |
| Mercury | 25 | 16 | 26 | 0 | 82 | 17 |

14, 16, 17, 19-21, 23, 25, 26, 32, 34-38

Table 13-6. Descriptive Statistics for HAP Removal Efficiencies Shown in Figures 13-7 and 13-8



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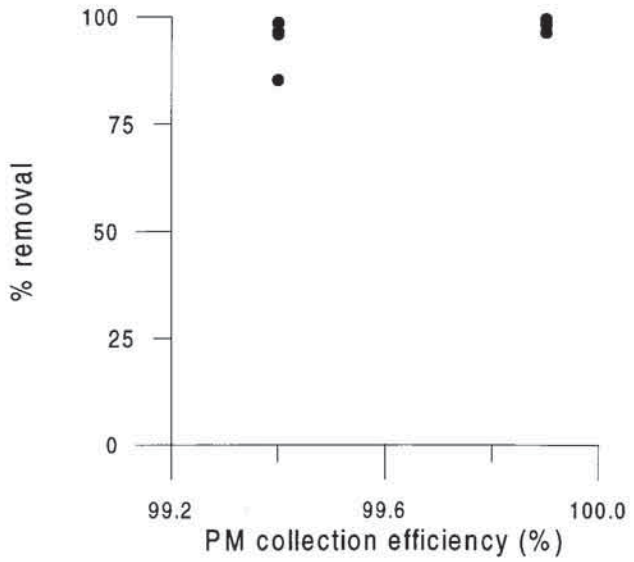



Figure 13-9. Removal of Metallic HAPs by Electrostatic Precipitators (Hot-side, Coal) (Includes, Arsenic, Beryllium, Cadmium, Chromium, Lead, and Manganese)

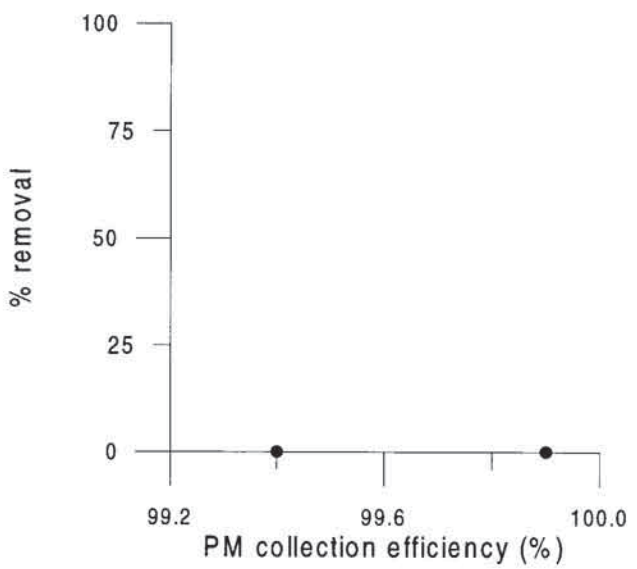



Figure 13-10. Removal of Mercury by Electrostatic Precipitators (Hot-side, Coal)

| | Mean | Median | S Dev | 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 13-7. Descriptive Statistics for HAP Removal Efficiencies Shown in Figures 13-9 and 13-10





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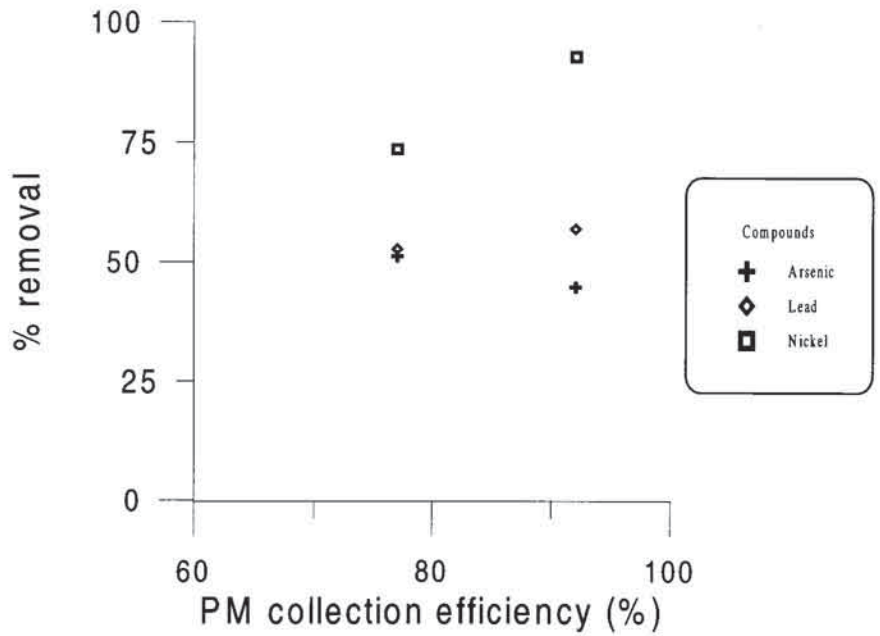



Figure 13-11. Removal of Metallic HAPs by an Electrostatic Precipitator (Oil)(Includes, Arsenic, Lead, and Nickel)

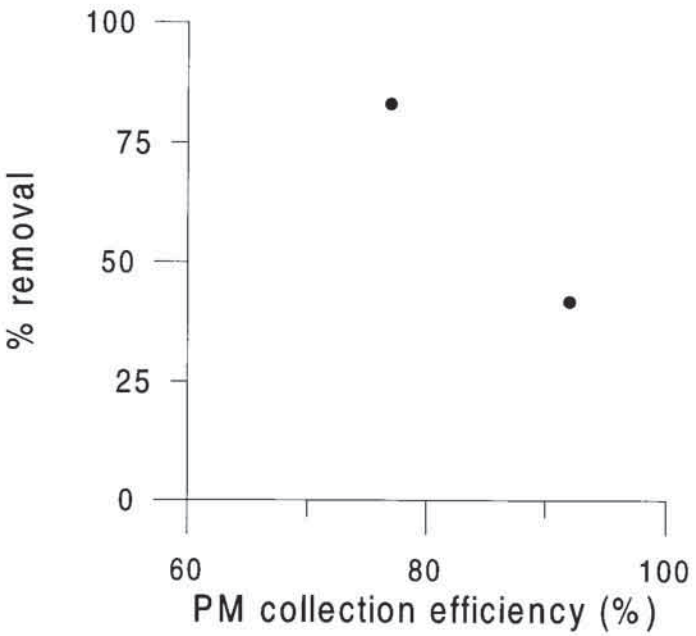



Figure 13-12. Removal of Mercury by an Electrostatic Precipitator (Oil)

| | 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 |

39, 40

Table 13-8. Descriptive Statistics for HAP Removal Efficiencies Shown in Figures 13-11 and 13-12



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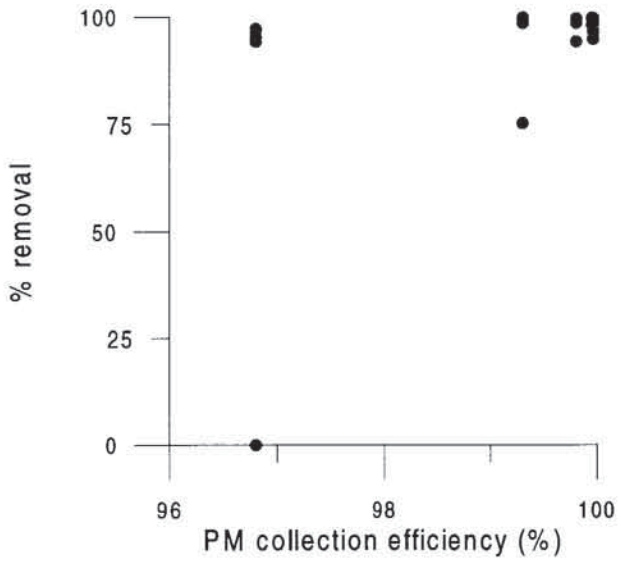



Figure 13-13. Removal of Metalic HAPs by a Fabric Filter (Coal) (Includes, Arsenic, Beryllium, Cadmium, Chromium, Lead, and Manganese)

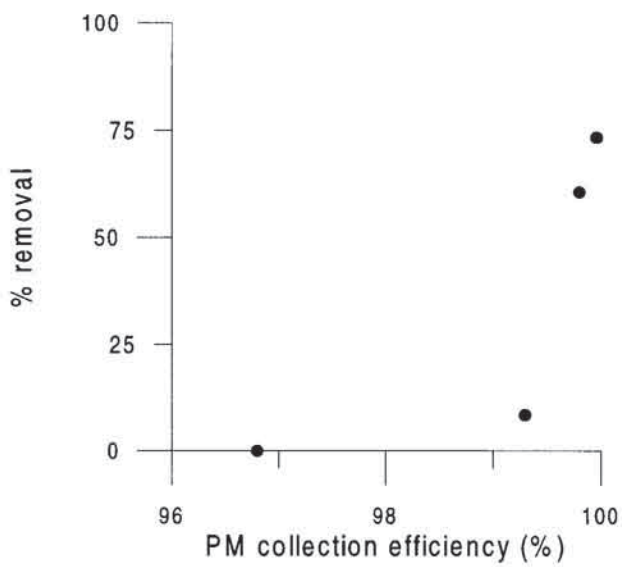



Figure 13-14. Removal of Mercury by a Fabric Filter (Coal)

| | Mean | Median | Sdev | Min | Max | Count |
|---|---|---|---|---|---|---|
| Arsenic | 99 | 99 | 1.4 | 97 | >99 | 5 |
| 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 | 1.3 | 97 | >99 | 5 |
| Manganese | 98 | 99 | 1.8 | 95 | >99 | 5 |
| Mercury | 36 | 34 | 37 | 0 | 73 | 4 |

18, 27-29, 41

Table 13-9. Descriptive Statistics for HAP Removal Efficiencies Shown in Figures 13-13 and 13-14



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Table 13-10. 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) | 108 | 89 |
| ESP (hot-side) | 12 | 92 |
| FF | 33 | 89 |

ESP = electrostatic precipitator
FF = fabric filter
HAP = hazardous air pollutant

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. Mercury removals and current investigations on the control of mercury are further discussed in section 13.6.

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 in MWC systems, dioxins and furans can be formed in the particulate-laden flue gas stream upstream of or within the associated PM control equipment at temperatures in the range of 480° to 1,020° F (250° to 550° C).[42-44] Units that are equipped with hot-side ESPs (ESPs upstream of the air preheater operating at temperatures in the range of 600 to 750°F [316° to 400° C]) 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.

### 13.3.2 Vapor Phase Controls

Figures 13-15 through 13-18 and Tables 13-11 and 13-12 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 eight data sites at which four of the eight 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.

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.

### 13.3.3 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 15 percent and 14 percent bypass, respectively. 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 15 percent bypass was estimated to remove approximately 80 percent of the HCl and approximately 29 percent of the HF; and an SDA/FF with 14 percent bypass was estimated to remove approximately 82 percent of each acid gas.[47] 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.

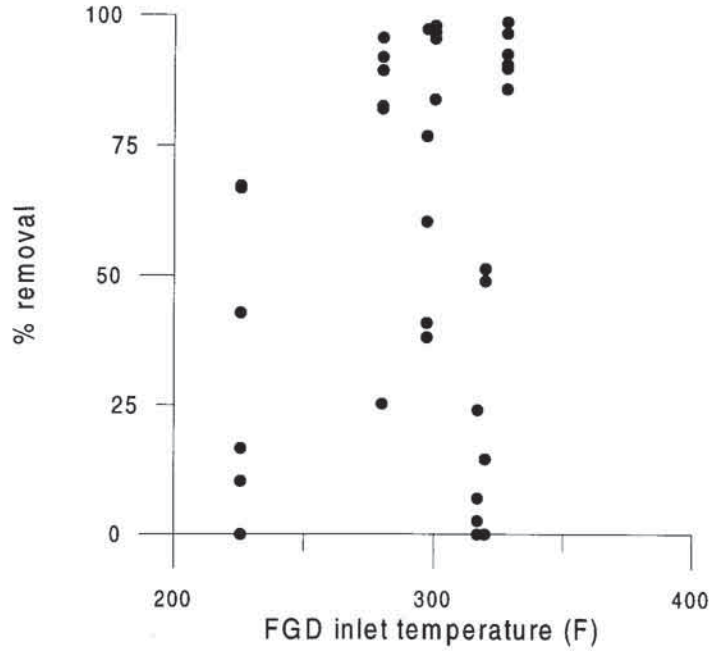



Figure 13-15. Removal of Metallic HAPs by an FGD (Coal) (Includes, Arsenic, Beryllium, Cadmium, Chromium, Lead, and Manganese)

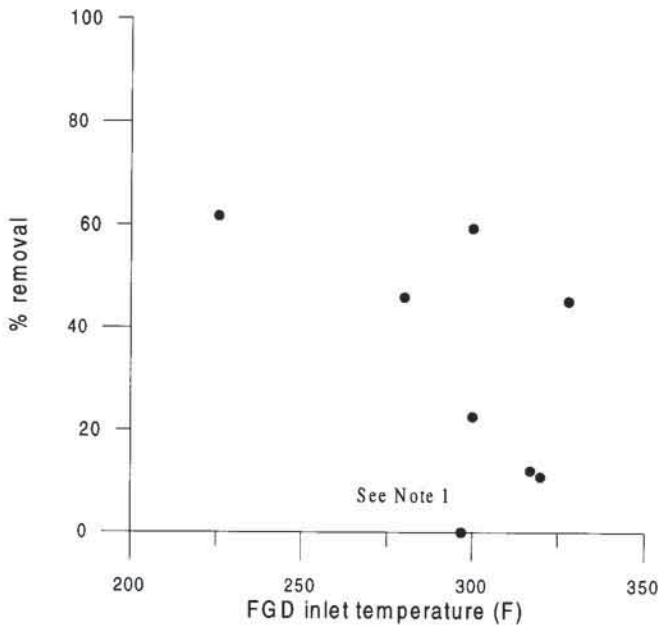



Figure 13-16. Removal of Mercury by an FGD (Coal)

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 13-16. (The ESP for this site was given an "82" percent removal.)

| | Mean | Median | Sdev | Min | Max | Count |
|---|---|---|---|---|---|---|
| Arsenic | 69 | 79 | 26.0 | 24 | 96 | 8 |
| Beryllium | 73 | 87 | 36 | 3 | 98 | 6 |
| Cadmium | 38 | 25 | 35 | 0 | 90 | 7 |
| Chromium | 55 | 57 | 35.8 | 0 | 97 | 8 |
| Lead | 57 | 61 | 39 | 0 | 98 | 8 |
| Manganese | 50 | 68 | 47.4 | 0 | 99 | 7 |
| Mercury | 31 | 23 | 23 | 0 | 62 | 9 |

21, 23, 25, 38, 45, 46, 48-50

Table 13-11. Descriptive Statistics for HAP Removal Efficiencies Shown in Figures 13-15 and 13-16



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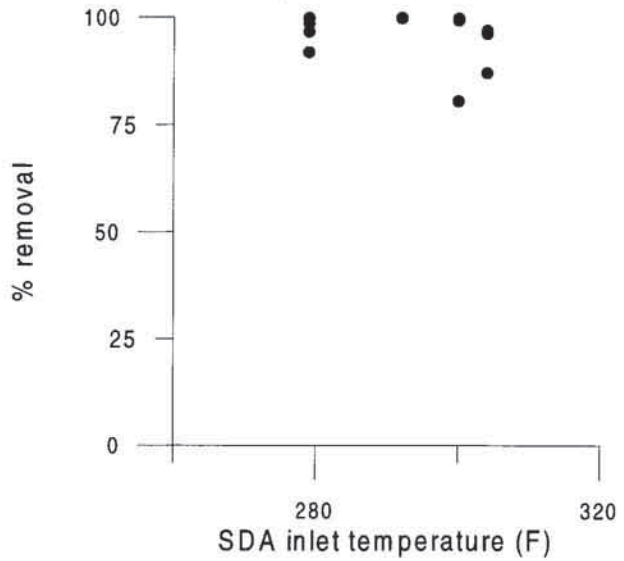



Figure 13-17. Removal of Metallic HAPs by a Spray Dryer Adsorber/fabric Filter (Coal) (Includes, Arsenic, Beryllium, Cadmium, Chromium, Lead, and Manganese)

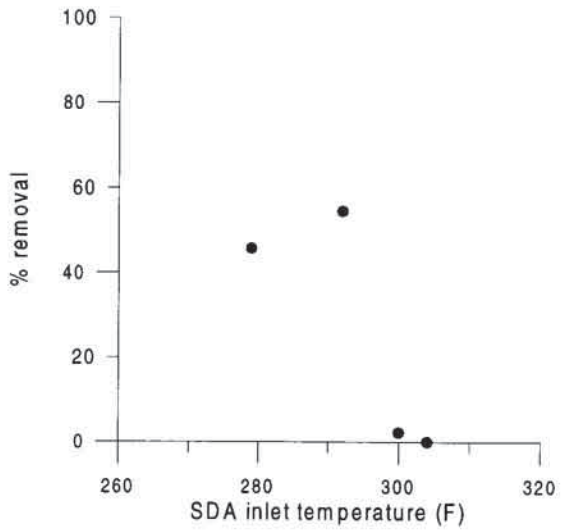



Figure 13-18. Removal of Mercury by a Spray Dryer Adsorber/Fabric Filter (Coal)

| | 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 |

22, 30, 31, 45

Table 13-12. Descriptive Statistics for HAP Removal Efficiencies Shown in Figures 13-17 and 13-18



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13.3.4 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 13.6.1.2), 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.

13.4 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.[51] 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.[52]

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.[53] Other assessments indicate that with realistic investments in research and development

in both crop production and generation techniques, biomass could be competitive (without subsidies) in niche markets within 5 years (with high energy prices, which are not currently forecast) and within 10 years in larger markets.[54]

Another method would be to switch to a source of renewable energy (e.g., wind, solar), 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.)

Future potential electrical transmission equipment could include the development and use of superconductive power lines, which could 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).[55]

## 13.5 POLLUTANT TRADEOFFS

### 13.5.1 HAP Increase/Decrease

The various strategies for limiting HAP emissions, discussed in sections 13.1 through 13.4, have different effects in controlling air emissions of all HAPs. Table 13-13 presents the qualitative effects of the different control strategies on air emissions.

Table 13-13 provides a comparison of HAP removal effectiveness of different existing and alternative control strategies. As shown in Table 13-13, 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 13.3.1, 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.

Table 13-13. 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 | | | | | |

[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.

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.

13.5.2 Water/Solid Waste Considerations

Coal cleaning can produce a variety of waste problems.[56] The process creates a liquid waste containing fine coal particles and inorganic elements and compounds dissolved from the ash in 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 groundwater can occur from coal wastepiles or storage piles if water is allowed to infiltrate them. Contaminants such as iron, manganese, and heavy metals (such as cadmium and silver) may leach from the wastes.[56]

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.[57] In addition to removing sulfur oxides, regenerable processes generate a usable product from the sludge, such as gypsum, which 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.[58] 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.[59]

Changes in all elements of the alternative controls, from conservation to technology improvements, can reduce the amount of waste produced by the utility industry.

## 13.6 AVAILABLE CONTROL TECHNOLOGY AND STRATEGIES FOR MERCURY CONTROL

Typical mercury removal efficiencies for conventional emission controls are discussed in section 13.3. Conventional controls are generally inconsistent in their effectiveness, and range from 0 to more than 83 percent removal (based on emissions testing at full-scale utility boilers).

Strategies for further reducing mercury emissions from electric power generation include demand reduction to decrease overall fossil fuel use, use of other forms of generation (e.g., nuclear power, biomass), switching to fuels having less mercury (e.g., natural gas), improving the mercury removal efficiency of conventional controls, and adding controls that remove mercury more effectively than conventional controls. The mercury content in utility flue gas ranges from 0.2 $\mu$g/dry standard cubic meter ($\mu$g/dscm) to 25 $\mu$g/dscm at 7 percent oxygen and standard conditions of 20° C and 1 atmosphere, while utility flue gas flow rates may range from 10,000 to 4,000,000 dscm/min.[60] 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.

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. There is other evidence that the chloride content of the coal impacts on the form and suitability of mercury removal by conventional control systems. This evidence has led to research on improving mercury removal from utility flue gas streams.

This section briefly reviews one pre-combustion technology (coal cleaning) and three classes of post-combustion technologies (enhancement of existing APCDs, carbon injection, and novel techniques). Additional information may be found in Appendix I.

### 13.6.1 Pre-Combustion Strategies

As mentioned in section 13.1.2, 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.[63] Advanced coal cleaning techniques are being

investigated for improved mercury removal potential. Like conventional cleaning techniques, the advanced cleaning techniques cannot be considered a mercury control technique for all coals at this time.

Advanced coal cleaning methods such as selective agglomeration and column froth flotation have the potential to increase the amount of mercury removed by conventional cleaning alone. Bench-scale studies indicate that the combination of conventional and advanced coal cleaning techniques removed from 40 to 82 percent of the mercury contained in eight samples of raw coal.[61,62]

Advanced cleaning methods, such as column froth flotation, are starting to emerge. Microcel™ is a type of column froth flotation available through ICF Kaiser and Control International. The company is the exclusive licensee for the technology in the coal fields east of the Mississippi River and has sold units for commercial operation in Virginia, West Virginia, and Kentucky. Ken-Flote™ is another type of column froth flotation cell.

## 13.6.2 Post-Combustion Strategies

### 13.6.2.1 Impact of Fuels and Temperature on Mercury Emissions

Fuels and the temperature of the flue gas can have a significant impact on the quantity of mercury emissions in the flue gas from a boiler and on the ability of control systems to remove the mercury.

13.6.2.1.1 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[63] in one of three forms: (1) elemental, (2) ionic, or (3) organic.

In the early 1990s, test methods were developed to quantify the species of mercury present in utility flue gas. Since that time, more accurate speciation test methods (e.g., Ontario Hydro, EPA draft 101B) have been developed. Mercury speciation testing indicates that the distribution of ionic mercury, most likely mercuric chloride ($HgCl_2$), and elemental mercury each varied in the sampled coal-fired utility flue gas.

Preliminary test results suggest that the chloride concentration in the coal and the type of coal (e.g., bituminous, subbituminous, or lignite) may be associated with a particular speciation of mercury in the flue gas, but more data are needed to verify these associations. Specifically, higher concentrations of ionic mercury were associated with tested coals containing high chloride concentrations (0.1 to 0.3 weight percent),[64-66] while 149° C (300° F) flue gas from tested

subbituminous coals appeared to contain approximately ten times the percentage of elemental mercury as compared with flue gas from tested bituminous coals.[67-69] 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.[70]

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. Analysis of two samples of flue gas suggests that mercury is predominantly in the elemental form when the fuel is oil.

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. Wet scrubbing of the flue gas may result in increased ionic mercury removal.

The scrubbing affinity for ionic mercury has been observed in pilot-scale studies. Preliminary results from tests of pilot-scale wet and dry scrubbers indicate that at least 90 percent of the ionic mercury was captured, while the removal of elemental mercury ranged from 0 to approximately 70 percent.[65,66,71-73] These preliminary test results suggest that a scrubbing system will have a low mercury removal efficiency if it treats flue gas from a boiler that fires subbituminous coal (containing relatively more elemental mercury) and a high mercury removal efficiency if it treats flue gas from a boiler that fires bituminous coal (containing relatively more ionic mercury).[74] Full-scale studies are needed to verify these observations.

Speciation of mercury is important in planning control strategies but is still under investigation. 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.

13.6.2.1.2 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.[60] Mercury is found predominantly in the vapor phase in utility flue gas.[63] 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[63] 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 increased from 27 percent to 51 percent as the temperature of the flue gas decreased from 107° C (225° F) to 96° C (205° F).[75]

13.6.2.2 Effectiveness of Mercury Capture Effectiveness of Existing Pollution Control Technology. Typical mercury removal efficiencies for conventional emission controls are discussed in section 13.3. Since conventional controls are generally inconsistent in their effectiveness, with a range from 0 to more than 83 percent removal (based on emissions testing at full-scale utility boilers), conventional controls cannot be considered a mercury control strategy for all electric utility plants at this time. Due to the limited effectiveness of conventional controls, research continues on ways to improve mercury capture by conventional emission control devices, sorbent injection, and the development of novel techniques. In order to develop low cost post-combustion mercury strategies for full-scale utility operation, fundamental research must continue on the chemistry and interactions of flue gas constituents, fly ash, and mercury-species encountered at various flue gas conditions across the utility industry.

The following sections briefly describe research on enhancing mercury removal with existing control technologies, activated carbon injection, and development of novel mercury control technologies.

13.6.2.3 Enhancement of Existing Control Devices.

13.6.2.3.1 Enhanced Particulate Control. Several approaches to improving the capture of fine particles and mercury with existing control devices are being investigated. Studies are underway on enhancing particulate control with a new Asea Brown Boveri (ABB) precharger, a wet ESP, flue gas cooling and humidification, and an advanced power supply (the ABB Switched Integrated Rectifier). Flue gas cooling and humidification tests are currently in progress. This approach shows promise in improving the collection of particulate-bound mercury, and may also cause vapor-phase mercury to condense on particulate matter and be captured in the ESP.[76] Research is also underway on optimizing the mercury removal capability of conventional pollution control technologies.[77]

13.6.2.3.2 Enhanced Wet Scrubbers. Several approaches to improving the capture of total and elemental mercury are being investigated. These include optimizing the liquid-to-gas ratio, wet FGD tower design, and improved elemental mercury removal with scrubbing liquid additives and catalysts. Brief descriptions of these approaches follow.

Liquid-to-gas ratio. The liquid-to-gas (L/G) ratio of a wet FGD system is dictated by the desired removal efficiency of $SO_2$ and impacts the removal efficiency of oxidized mercury. In general, high efficiency (95 percent $SO_2$ removal) systems are designed with L/G ratios of 120 gal/1000 acf to 150 gal/1000 acf. In two separate pilot-scale studies, increasing the L/G ratio from approximately 40 gal/1000 acf to approximately 125 gal/1000 acf increased the removal efficiency of oxidized mercury from 90 percent to 99 percent.[73,78] Test data suggest that increasing the L/G ratio did not affect the removal efficiency of elemental mercury, which was close to zero percent.[79]

Wet FGD tower design. Most of the existing U.S. wet FGD systems have open spray tower or tray tower designs.[80] Recent research has shown that tray tower designs are more effective in removing oxidized mercury from boiler flue gas than tower designs. In one study of wet FGD systems, where the composition of the flue gas was mostly oxidized mercury, the tray tower design removed from 85 to 95 percent of the total mercury, whereas the open spray tower design removed from 70 to 85 percent of the total mercury.[81]

Improved elemental mercury removal. Since two studies noted higher concentrations of elemental mercury in the outlet of a wet FGD system compared to the inlet concentrations of elemental mercury,[82,83] research is in progress on ways to convert and capture elemental mercury. Research currently is investigating scrubbing liquid additives and catalysts to convert elemental to ionic mercury.

Several scrubbing liquid additives that combine strong oxidizing properties with relatively high vapor pressures are being investigated as techniques to enhance the capture of mercury in a wet scrubber. Of three halogen (i.e., chlorine, bromine, and iodine) solutions tested to date, the chlorine solution appears to remove the most elemental mercury in the presence of $SO_2$ and NO. Further testing of these and possibly other oxidizing reagents is planned.[84]

Due to a much higher solubility compared to elemental mercury, oxidized mercury is readily removed in a wet scrubber. Investigations are underway on the conversion of vapor-phase elemental mercury to more soluble $Hg^{++}$ in bench and pilot-scale studies. The effect of flue gas temperature and residence time on the oxidation potential of a number of catalysts and coal-based fly ashes is being studied. To date, pilot-scale tests of three iron-based catalysts, a carbon, a bituminous, and lignite fly ash have shown the carbon-based catalyst to be the most effective in converting elemental mercury to $Hg^{++}$. Further testing of the carbon catalysts is planned.[85]

13.6.2.4 Activated Carbon Injection. Activated carbon (AC) injection is considered a potential control technology for mercury

emitted from electric utilities, since a form of this technology has been successfully demonstrated on medical waste incinerators and municipal waste combustors.[86-89]

The level of mercury control that might be achieved with AC injection into 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 13-14, 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.

13.6.2.4.1 Factors Affecting Mercury Removal Efficiency. Preliminary data from various pilot- and bench-scale studies suggest that factors besides the optimum amount of AC that is injected may affect mercury removal. These factors are temperature, the speciation of the vapor-phase mercury and type of activated carbon[67] injected into the flue gas, and flue gas composition.

Temperature. A pilot-scale study of AC injection upstream of an FF suggests that mercury removal efficiencies and the required amount of AC injection were apparently temperature dependent.[75] In reducing the temperature from approximately 116°C (240°F) to approximately 93°C (200°F), the mercury removal increased from approximately 80 percent (with an injection rate of approximately 3,500 $\mu$g carbon/$\mu$g of inlet mercury) to approximately 98 percent (with an injection rate of approximately 155 $\mu$g carbon/$\mu$g of inlet mercury). (The high mercury removal percentages suggest that flue gas contained mostly ionic mercury.)

These test results suggest that more mercury is removed and less carbon is 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

Table 13-14. Comparison of Typical Uncontrolled Flue Gas Parameters at Utilities and MWCs[a]

| Uncontrolled flue gas parameters | Coal-fired utility boiler [60,90] | Oil-fired utility boiler [91,92] | MWC[b] [93,94] |
|---|---|---|---|
| Temperature (°C) | 121 - 177 | 121 - 177 | 177 - 299 |
| Mercury content ($\mu$g/dscm) | 1 - 25 | 0.2 - 2 | 400 - 1,400 |
| Chloride content ($\mu$g/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 |

MWC = municipal waste combustion

[a] Standard conditions are 0° C and 1 atmosphere.
[b] Moisture content in the MWC flue gas was assumed to be 13.2 percent.

96° C (205° F) and still avoid acid condensation, provided low-sulfur coals (less than about 1 weight percent sulfur) are burned.[95] 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.

Speciation of mercury. The effectiveness of AC injection in recovering different forms of mercury is still under investigation. The available data indicate iodide- and sulfur-impregnated AC are needed for significant elemental mercury removal.

Studies without AC injection at a pilot-scale SDA/ESP system in Denmark and a full- and pilot-scale SDA/FF system indicated that essentially all of the ionic mercury was removed (with greater than 97 percent removal efficiencies) while essentially none of the elemental mercury was removed (with 0 to approximately 3 percent removal efficiencies).[65] Studies indicated that the removal of elemental mercury was increased to approximately 50 percent with AC injection ahead of the SDA/ESP and SDA/FF and to approximately 93 percent with injection of iodide- and sulfur-impregnated AC ahead of two pilot-scale test systems (a SDA/FF system and the University of North Dakota Energy and Environmental Research Center (UNDEERC) system consisting of a boiler and baghouse).[65,96] Pilot-scale testing at the UNDEERC system indicated that the percent-removal of elemental mercury with lignite-based AC was temperature dependent.[97]

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, and to characterize, mercury speciation in utility flue gas.

Flue gas composition. Flue gas components such as sulfur oxides, water, and chlorine compounds can affect the mercury removal efficiency of carbon. A recent bench-scale study investigated the effects of $SO_2$ and HCl on the adsorption of elemental mercury and mercuric chloride ($HgCl_2$) by a lignite-based activated carbon.[98] Removing $SO_2$ from the flue gas increased the equilibrium adsorption capacity for elemental mercury by a factor of about 5.5 compared to 3.5 for mercuric chloride. Removing HCl from the flue gas did not affect the equilibrium adsorption capacity of the carbon for mercuric chloride; however, it did prevent the carbon from adsorbing elemental mercury. With no HCl in the gas, the carbon adsorption capacity for mercuric chloride was larger than that for elemental mercury. Other carbons may not be affected by the presence of HCl and $SO_2$ if the mercury adsorption mechanism is different.

Research continues on the chemistry and interactions of flue gas, fly ash, and mercury species. This fundamental research is needed at various flue gas conditions encountered across the utility industry in order to develop low cost mercury strategies for full-scale utility operation. Thus, while AC injection shows promise as a mercury control technology, more data and research are needed to understand the factors that affect mercury removal.

13.6.2.5 Emerging Technologies for Controlling Mercury Emissions from Utilities. Research continues on developing potential technologies for mercury emission reduction from utility plants. This research is aimed at either the addition of some type of sorbent technology or novel technology for mercury control. Emerging technologies are described below.

13.6.2.5.1 Sorbent technology. 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.

Sulfur-impregnated carbon. 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.[65]

Iodide-impregnated carbon. 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 pilot-scale studies, iodide-impregnated carbon achieved approximately 99 percent mercury removal.[65,99]

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 (with certain coal types and 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.[100]

Chloride-impregnated carbon. 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.[108] Although the amount is small, chloride-impregnated AC injection would introduce additional chlorine (a HAP) into the flue gas stream.

Sorbalit. 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.[93] 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.[101]

Other sorbents. Numerous studies are underway to develop other sorbents as economical alternatives to activated carbon.[102,103] Some of the sorbents under investigation include volcanic pumice, sulfur- and iodide-impregnated carbons, several proprietary sorbents, high-carbon fly ash, Darco FGD (an activated carbon derived from lignite), an activated carbon prepared from a bituminous coal, steam-activated lignite, thermal-activated bituminous coal, chemical-activated hardwood, iodine impregnated steam-activated coconut shell, and sulfur-impregnated steam-activated bituminous coal.[104]

Sorbent Technologies is marketing a sorbent called Mercsorbent.[105] The company claims that the sorbent is effective in removing elemental mercury at high temperatures typical of utility flue gas, and is unaffected by common co-existing flue gases, such as $SO_2$, HCl, and $H_2O$. Mercsorbent can be used for sorbent injection or it can be used as a coating on an FF. A bench-scale duct-injection system at Sorbent Technologies facilities is now being used to test Mersorbent with this

approach. The company is also scheduled to demonstrate the sorbent at the refuse incinerator in Fort Dix, New Jersey; prior compliance sampling at this facility suggests that a significant amount of its mercury is in the elemental form. A coal-fired boiler or slipstream is also being sought for a test of the new sorbent material.

Sodium sulfide injection. 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.[106] 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.[107] 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.

Carbon with circulating fluidized beds (CFBs). Another potential process for the reduction of mercury emissions is the use of AC in a CFB.[63] 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.[108]

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.

In the United States, Environmental Elements Corporation is developing a CFB that promotes agglomeration of fine particulate matter, allowing for its capture in an ESP. In addition, activated carbon is added to the fluid bed to adsorb mercury vapor. High residence time, due to the recirculation of the particles, allows for effective utilization of the carbon. Water sprayed within the circulating bed further promotes the removal of mercury. Results from bench-scale testing indicate that mercury was significantly reduced when passed through the fluidized bed of fly ash and activated carbon. Based on these tests, a carbon-to-mercury usage was determined for the system that compares favorably with other sorbent-based mercury control techniques. There are plans to install a pilot unit and test at Public Service Electric and Gas's Mercer Station.[109]

13.6.2.5.2 Novel technologies. Additional potential processes for controlling mercury emissions include advanced coal cleaning, a condensing heat exchanger, gold sorbent technology, other sorbent injection processes, a corona reactor, and mercury amalgamation. These technologies are briefly described below.

Condensing heat exchanger. Based on condensing heat exchanger technology, McDermott Technology (formerly Babcock & Wilcox) is developing an integrated flue gas treatment system for recovering waste heat and removing $SO_2$, $SO_3$, particles, and trace elements from coal combustion flue gas. The condensing heat exchanger is a two-pass, counter-flow shell and tube heat exchanger. The hot flue gas enters the top and flows downward through the first cooling stage, across a horizontal transition region, then upward through the second cooling stage. An alkali reagent is sprayed from the top of the second stage to aid in the removal of $SO_2$. Testing of the technology is being conducted at McDermott Technology's research facility in Alliance, Ohio. Preliminary results indicate that total mercury removal across both stages of the condensing heat exchanger is about 62 percent when firing a blend of Ohio coals. Additional testing is planned on two other bituminous coals.[110]

Gold sorbent technology. ADA Technologies has begun development and testing of a process, called Mercu-RE, for the removal and recovery of vapor-phase mercury from coal-fired utility boilers. The process takes mercury from flue gases and produces liquid, elemental mercury with no secondary wastes. Noble metals are used to adsorb mercury at typical flue gas temperatures. The mercury is then thermally desorbed.

Preliminary results from laboratory tests indicate that a gold-coated monolith captured virtually all of the elemental mercury injected into a simulated flue gas. Pressure drop through the monolith was low, which is critical to full-scale use. Further testing of the gold monoliths will include repeated sorption and desorption cycles. This phase will be followed by testing on a pilot-scale coal combustion flue gas at Consol's research facility in Library, Pennsylvania.[111]

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

stream. The solids are mixed with material collected in a baghouse to produce the $SO_2$ scrubbing reagent for the spray dryer.

Application of the E-LIDS™ system, when firing an Ohio bituminous coal in the Clean Environment Development Facility (CEDF) at the Alliance Research Center of McDermott Technology, Incorporated, has shown efficient emissions control performance. Sulfur dioxide emissions generated from firing the nominal 3-percent sulfur coal were reduced by more than 99 percent to less than 0.10 lbs $SO_2/10^6$ Btu. Total mercury emissions were reduced from an uncontrolled level of 17.6 $\mu$g/dscm to less than 0.2 $\mu$g/dscm for an average total removal efficiency of greater than 98 percent. The measured performance confirmed earlier results obtained in the 5 x $10^6$ Btu/hr small boiler simulator (SBS) facility. Mercury measurements upstream of the dry scrubber indicated that both the limestone injection and operation of the spray dryer/baghouse system at close to the saturation temperature contributed to the observed total mercury emissions reduction. The furnace limestone injection alone reduced mercury emissions to an average of 3.1 $\mu$g/dscm.[112]

Corona reactor. Environmental Elements Corporation is developing a process for mercury control through DOE's Small Business Innovative Research program. The first concept utilizes an intense corona discharge to convert $Hg^0$ to mercuric oxide. The process also produces $SO_3$ to serve as a conditioner for high-resistivity fly ash. A corona discharge in coal combustion flue gas will produce oxidizing radicals, such as OH and atomic oxygen. Bench-scale results indicate that the corona reactor, operating at relatively low power levels and short residence time, yielded high elemental mercury vapor oxidation. The mercuric oxide, in the form of solid particles, was removed using conventional particulate control technology. The corona reactor may also convert mercuric chloride to mercuric oxide, allowing for its capture as well. The system is currently being tested on a slipstream at Alabama Power's Plant Miller.[113]

Mercury amalgamation. There are plans to investigate the interaction of mercury with metals such as zinc, silver, tin, and cadmium. Mercury has been shown to amalgamate with certain metals. Both experimental and modeling efforts are planned to determine the suitability of metals for the capture of mercury.[114]

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111. Ref. 76, p. 5.

112. Ref. 73

113. Ref. 76, p. 5.

114. Ref. 76, p. 7.

## 14.0 SUMMARY OF RESULTS, TECHNICAL FINDINGS, AND RESEARCH NEEDS

The following summary of results, technical findings, and research needs is based on this study and the currently available scientific data.

### 14.1 INDUSTRY GROWTH AND HAP EMISSIONS

1. Utility units emit a significant number of the 189 HAPs included on the section 112(b) list, although in most cases they are responsible for very small percentages of total anthropogenic emissions. Coal-fired units emit the largest number of utility-originated 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 34 percent of the United States anthropogenic airborne emissions of mercury but no more than 4 percent of other measured HAPs.

2. Under the assumptions made in this Report, HAP emissions are predicted to increase during the period 1990-2010. Over this period, utility coal consumption is estimated to increase by approximately 29 percent, oil consumption is estimated to decrease by approximately 48 percent, and natural gas consumption is expected to increase by about 61 percent. Coal combustion accounts for the increase in HAP emissions.

3. Actions anticipated to be taken by the utility industry to comply with other provisions of the Act (e.g., acid rain, NAAQS revisions) or with other initiatives (e.g., electricity restructuring, global warming abatement) may impact HAP emissions.

4. The emission test program provided valid and representative results for the purpose of this study. 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 helped to establish these protocols.

### 14.2 INHALATION RISK ASSESSMENT

5. For the vast majority of the more than 196 million persons living within 50 km of any utility unit, the lifetime cancer risk due to inhalation exposure to HAP emissions is likely to be less than $1 \times 10^{-6}$.

6. Out of 426 coal-fired plants, EPA's modeling indicates that only 2 coal-fired plants pose high-end excess lifetime cancer risks greater than 1 x $10^{-6}$ due to inhalation exposure to HAP emissions. For no plant does the inhalation MIR exceed 4 x $10^{-6}$. More than 50 percent of the inhalation cancer risk is attributable to arsenic. The average inhalation MIR across all plants is estimated to be roughly 1/10th to 1/100 as large as the highest inhalation MIR. Central tendency inhalation risks for all exposed individuals are predicted to be approximately one to three orders of magnitude lower than the highest inhalation MIR. The population cancer incidence due to inhalation exposure to coal-fired utility HAP emissions, considering the results of both the local and long-range transport modeling, is estimated to be no greater than 1.3 cases per year nationwide.

7. For the year 2010, the cancer MIRs from coal-fired utilities are not expected to be significantly different. However, due to uncertainties about future demand, industry operations, government regulation, etc., the EPA has low confidence in this projection.

8. Out of 137 oil-fired plants, EPA's modeling indicates that only 11 oil-fired plants pose high-end excess lifetime cancer risks greater than 1 x $10^{-6}$ due to inhalation exposure to HAP emissions. For no plant does the inhalation MIR exceed 6 x $10^{-5}$. More than 50 percent of the inhalation cancer risk is attributable to nickel and the assumption that nickel emissions are 50 percent as carcinogenic as nickel subsulfide. The average inhalation MIR across all plants is estimated to be roughly 1/10th to 1/100th as large as the highest inhalation MIR. Central tendency inhalation risks for all exposed individuals are predicted to be roughly two to three orders of magnitude lower than the highest inhalation MIR. Population cancer incidence due to inhalation exposure to oil-fired utility HAP emissions (considering local and long-range transport) is estimated to be no greater than 0.5 cases per year nationwide.

9. For the year 2010, the cancer MIRs from oil-fired utilities are predicted to be approximately 30 to 50 percent lower than the 1990 estimates. The EPA has good confidence in this projection because of the well-established trend of declining oil use by utilities.

10. Based on the quantitative parameter uncertainty analysis conducted for the inhalation risk assessment, the EPA estimates that the high-end inhalation risk estimates presented in this report are conservative (i.e., more likely to be overestimating rather than underestimating the true MIR). 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") inhalation MIR estimates are predicted to be roughly 2 to 10 times lower than the high-end MIR estimates. However, there are limitations to the uncertainty analysis and limitations in available data and the range of uncertainty is, most likely, larger than estimated by this study.

11. 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), residence time, activity patterns, and other uncertainties and variabilities, that were not quantitatively assessed.

12. With regard to noncancer effects, the highest hazard quotient for any HAP, considering both short- and long-term exposures, is approximately 0.1 for HCl from coal-fired utilities. That is, the highest exposure concentration for HCl was estimated to be 10 times lower than the Inhalation Reference Concentration (RfC) for HCl. The highest hazard index for all HAPs is about 0.2.

## 14.3 MERCURY

13. Mercury cycles in the environment as a result of natural and human (anthropogenic) activities. Most of the mercury in the atmosphere is elemental mercury vapor, which circulates in the atmosphere for up to a year, and hence can be widely dispersed and transported thousands of miles from sources of emissions. Even after it deposits, mercury commonly is emitted back to the atmosphere to be re-deposited elsewhere. The modeling of long-range transport of mercury suggests that about one-third of United States utility emissions is deposited within the lower 48 States. The remaining two-thirds are transported outside of United States borders where it diffuses into the global reservoir. Finally, predictions of the ISC3 and RELMAP models indicate that most of the mercury emitted by utilities is transported further than 50 km from the emission source.

14. The analysis of mercury fate and transport completed for this study, as well as the analyses completed in the EPA's 1997 Mercury Study Report to Congress, in conjunction with available scientific knowledge, supports a plausible link between mercury emissions from anthropogenic combustion and

industrial sources and mercury concentrations in air, soil, water and sediments. The critical variables contributing to this linkage are these:

- the species of mercury that are emitted from the sources, with $Hg^0$ mostly contributing to concentrations in ambient air and $Hg^{2+}$ mostly contributing to concentrations in soil, water, and sediments;
- the overall amount of mercury emitted from a combustion source;
- the watershed soil loss rates, including reduction and erosion;
- the water body loss rates, including outflow, reduction, and settling; and
- the climate conditions.

In addition, the analysis of mercury fate and transport supports a plausible link between mercury emissions from anthropogenic combustion and industrial sources and methylmercury concentrations in freshwater fish. However, these fish methylmercury concentrations also result from existing background concentrations of mercury (which may consist of mercury from natural sources, as well as mercury which has been re-emitted from the oceans or soils) and deposition from the global reservoir (which includes mercury emitted by other countries). Given the current scientific understanding of the environmental fate and transport of this element, it is not possible to quantify how much of the methylmercury in fish consumed by the United States population is contributed by current United States emissions relative to other sources of mercury (such as natural sources and re-emissions from the global pool). The critical variables contributing to the plausible link include:

- the species of mercury that are emitted, with emitted divalent mercury mostly depositing into local watershed areas and, to a lesser extent the atmospheric conversion of elemental mercury to divalent species which are deposited over greater distances;
- the overall amount of mercury emitted from a source;
- the watershed soil loss rates, including reduction and erosion;

- the water body loss rates, including outflow, reduction, and settling;

- the extent of mercury methylation in the water body;

- the extent of food web bioaccumulation in the water body; and

- the climate conditions.

From the analysis of deposition and on a comparative basis, the deposition of $Hg^{2+}$ close to an emission source is greater for receptors in elevated terrain (i.e., terrain above the elevation of the stack base) than from receptors located in flat terrain (i.e., terrain below the elevation of the stack base). The critical variables are parameters that influence the plume height, primarily the stack height and stack exit gas velocity.

## 14.4 DIOXINS AND ARSENIC

15. Based on a screening level multipathway analysis, the highest estimated individual risks due to utility arsenic emissions are predicted to be no higher than $1 \times 10^{-4}$ and are probably much lower for the vast majority of the exposed population. The increased cancer risks due to multipathway exposures to arsenic emissions, based on screening level model-plant analysis, using hypothetical scenarios, were predicted to range from less than $4 \times 10^{-7}$ up to $1 \times 10^{-4}$. The highest predicted risk (i.e., $1 \times 10^{-4}$) was for a hypothetical scenario pica child assumed to be living at the point of maximum deposition, which is considered an upper bound, conservative scenario. When the risk from background exposure ($2 \times 10^{-4}$) is added to the maximum risk from utility exposure, the risk for the pica child is estimated to be up to $3 \times 10^{-4}$. Background exposures were estimated to dominate the exposures and risk. There are substantial uncertainties associated with this screening level arsenic multipathway assessment, and the results do not apply to any specific existing utility plant. Further assessment is needed to more fully evaluate the risks due to arsenic emissions from utilities.

16. Based on a screening level multipathway assessment for dioxins, total modeled lifetime cancer risks related to indirect exposure to dioxins, based on model-plant analyses, are predicted to range from $1 \times 10^{-10}$ to $2 \times 10^{-4}$. The results of the analyses indicate that the exposures and risks due to fish consumption are the highest of all pathways considered. In all scenarios, the noninhalation (e.g., ingestion) exposures were predicted to be at least one order of magnitude larger than the inhalation exposures

and modeled exposures exceed the background exposures for dioxins. There are substantial uncertainties associated with this dioxin screening level multipathway assessment and the results do not apply to any specific existing utility plant. Further assessment is needed to more fully evaluate the risks due to dioxin emissions from utilities.

## 14.5 RADIONUCLIDE ANALYSIS

17. For the vast majority of the more than 196 million persons living within 50 km of any utility unit, the lifetime fatal cancer risk due to radionuclide emissions is less than $1 \times 10^{-6}$.

18. The highest MIR to any individual, within a 50-km radius, resulting from multipathway exposure to radionuclide emissions from utility units is estimated to be up to $3 \times 10^{-5}$, and 17 of the 684 plants were estimated to potentially pose an MIR greater than $1 \times 10^{-5}$.

19. 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/year.

20. The quantitative uncertainty analysis performed for the radionuclide analysis indicates that the population risk estimates are central values of the true probability distribution.

## 14.6 ALTERNATIVE CONTROL STRATEGIES

21. There are a number of alternative control strategies that are effective, based on the data obtained for this report, in controlling some of the HAPs emitted from utility units. These strategies are summarized below.

    - Conversion of coal- and oil-fired units to natural gas firing effectively eliminates emission of HAPs.

    - 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).

    - 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.

    - 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. Advanced coal cleaning technologies show promise in reducing mercury from coal from approximately 30 to greater than 60 percent. Further research is needed in methods of effecting greater trace metal removals during coal cleaning and in assessing the various impacts of these methods.

- 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.

- 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.

- 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. Research is underway to further enhance fine particle removal, including trace metallic HAPs, from these control devices. Organic HAPs do not appear to be well controlled by PM control devices but these compounds are generally found near the detection limit. 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.

- 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. Research is underway to further enhance the mercury removal capability of FGD systems. 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.

- Add-on technologies for the control of mercury have not been demonstrated on utility units in the United States. 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.

- Pollution prevention methods (i.e., 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.

## 14.7 AREAS FOR FURTHER RESEARCH AND ANALYSIS

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 is needed.

### 14.7.1 Emissions Data for Dioxins

Emissions data for dioxins 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).

### 14.7.2 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 the health effects associated with those different forms would be of value. Further evaluation of chromium speciation is also needed.

### 14.7.3 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. Arsenic and dioxins are two HAPs identified as priority for further multipathway assessment.

### 14.7.4 Long-range Transport Exposures

Uncertainties remain regarding long-range transport of HAPs. Further modeling and evaluation could be helpful to assess the impacts of long-range transport of HAPs from utilities.

### 14.7.5 Mercury Issues

There are numerous areas regarding mercury that may need further research, study, or evaluation. A number of potential areas for further study include the following:

- review the estimates of the levels of exposure to mercury associated with subtle neurological endpoints;

- quantify and/or evaluate the relationship between a change in United States mercury emissions and the resulting change in methylmercury levels in fish;

- evaluate actual consumption patterns and estimates of the methylmercury exposure of the subpopulations of concern;

- gather additional data on the mercury content of various types of coal;

- establish improved methods for measuring mercury concentrations in water;

- study the occupational, dietary, and behavioral factors that affect mercury exposures for people who are determined to be exposed above a threshold of concern;

- study the public health and environmental benefits that would be expected by reducing mercury emissions from utilities;

- evaluate and/or research control technologies or pollution prevention options that are available, or will be available, that could potentially reduce mercury emissions and what are the costs, economic impacts, and feasibility of those options;

- evaluate how other regulations, programs, and activities (e.g., acid rain program, electricity restructuring, NAAQS, climate change) affect mercury emissions;

- gather additional data on mercury emissions (e.g., how much is emitted from various types of utility units, how much is divalent vs elemental mercury, and how do factors such as control device, fuel type, and plant configuration affect emissions and speciation); and

- study how much mercury is emitted from natural sources and past anthropogenic sources.

14.7.6 Projections to the Year 2010

There are significant uncertainties and unknowns in the emissions and risk projections made to the year 2010 (e.g., impact of industry restructuring; impact of State efforts to regulate restructuring; impact of any climate change initiatives). Further research and evaluation in this area is needed.

14.7.7 Ecological Risks

The effects of HAPs on wildlife, endangered species, and terrestrial and aquatic ecosystems were not evaluated in this study. Although not mandated by section 112(n)(1)(A), further evaluation of

ecological risks due to HAP emissions would be needed to fully evaluate the impacts of utility HAP emissions.

14.7.8 Criteria Pollutant and Acid Rain Programs

Further evaluation is needed to assess the impacts of the Acid Rain and Criteria Pollutant programs (e.g., impact of revisions to the PM-fine and ozone NAAQS; impact of Ozone Transport Assessment Group [OTAG] activities) on HAP emissions.

14.7.9 Short-term Emissions

A limited assessment of short-term exposures was completed. However, further evaluation of short-term releases, especially high-end, peak releases, could be useful to fully assess the potential impacts to public health due to emissions of HAPs (particularly HCl and HF) from utilities.

## TECHNICAL REPORT DATA

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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 67 HAPs; (4) assessments of risks due to multipathway (inhalation plus non-inhalation) exposures to four HAPs (radionuclides, mercury, arsenic, and dioxins); and (5) a discussion of alternative control strategies. The assessment for mercury in this report includes a description of 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) based on modeling from four representative utility plants using hypothetical scenarios. The EPA has not evaluated human or wildlife exposures to mercury emissions from utilities in this report. 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.

17. KEY WORDS AND DOCUMENT ANALYSIS

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