United States
Environmental Protection
Agency

Office of Air Quality
Planning and Standards
Research Triangle Park, NC 27711

EPA-453/R-96-013b
October 1996

Air

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

Volume 2. Appendices A - G

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Appendix A - Median emission factors, determined from test report data, and total 1990 and total 2010 emissions, projected with the computer emission program

As explained in Chapter 3, inorganic hazardous air pollutant (HAP) emission totals were estimated through the use of emission modification factors (EMFs) in the emission factor program (EFP). The stack factors mentioned in Tables A-1 thru A-3 are medians of mean emissions for each test site and were not used in the computing estimated total emissions for inorganic HAPs. Medians were used due to a limited amount of data. These factors were extracted from the test reports and are presented for informational purposes only.

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Table A-1. Median Emission Factors, Determined from Test Report Data, and Total 1990 and Total 2010 HAP Emissions, Projected with the Emission Factor Program for Inorganic HAPs from Coal-fired Units[a]

Coal-fired units: inorganic HAPs	**Number of stack factors: PM control [b]**	**Median stack factor: PM control (lb/trillion Btu) [c]**	**Number of stack factors: PM and SO_2 control [a]**	**Median stack factor: PM and SO_2 control (lb/trillion Btu) [b]**	**Estimated total 1990 emissions (tons)**	**Estimated total 2010 emissions (tons)**
Antimony	7	1.4	4	0.13	**11**	**14**
Arsenic	21	2.9	8	0.90	**54**	**62**
Beryllium	12	0.45	5	0.14	**6.6**	**7.6**
Hydrogen chloride	15	21,000	9	1,290	**137,000**	**150,000**
Hydrogen cyanide (HCN) [d]	All HCN factors were combined	Number of factors: 5	Median factor: 28 lb/trillion Btu		**240**	**320**
Hydrogen fluoride	14	4,200	6	106	**19,500**	**25,600**
Cadmium	18	0.72	9	1.0	**1.9**	**2.3**
Chromium	22	8.4	10	4.0	**70**	**83**
Cobalt	10	2.7	6	1.0	**21**	**27**
Lead	21	4.8	9	5.8	**72**	**83**
Manganese	21	15	9	15	**180**	**232**
Mercury	20	3.9	10	3.4	**51**	**65**
Nickel	21	8.3	10	5.2	**48**	**57**
Phosphorus (P) [e]	All P factors were combined	Number of factors: 10	Median factor: 31 lb/trillion Btu		**270**	**350**
Selenium	19	62	9	8.0	**190**	**230**

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[a] Compounds are listed in the following sequence: inorganic, organic, and dioxin/furan/polycyclic aromatic hydrocarbons (PAHs). Median emission factors were determined from organic HAP concentrations at the stack, control device outlet, or boiler outlet when at least one of typically three measured flue gas concentrations was detected.

[b] Stack factors for inorganic HAPS were taken from test reports when at least one of typically three measured flue gas concentrations was detected. These factors were not used to develop the estimated emissions.

[c] Because the inorganic emissions were not directly estimated from stack factors, total emissions of inorganic HAPS projected with the computer program and from median stack factors will vary.

[d] Nationwide hydrogen cyanide emissions were determined from stack emission factors and not from EMFs.

[e] Nationwide phosphorous emissions were determined from stack emission factors and not from EMFs.

Table A-2. Median Emission Factors, Determined from Test Report Data, and Total 1990 and Total 2010 HAP Emissions, Projected with the Emission Factor Program for Inorganic HAPs from Oil-fired Units[a]

Oil-fired units: inorganic HAPs	Number of stack factors: PM control [b]	Median stack factor: PM control (lb/trillion Btu) [c]	Number of stack factors: No PM control [b]	Median stack factor: No PM control (lb/trillion Btu) [c]	Estimated total 1990 emissions (tons)	Estimated total 2010 emissions (tons)
Arsenic	2	0.32	8	5.3	**5**	**2.5**
Beryllium	2	0.33	4	0.21	**0.45**	**0.23**
Cadmium	1	0.32	9	1.6	**1.7**	**0.87**
Chromium	4	3.7	8	5.7	**4.7**	**2.4**
Cobalt	2	6.1	3	27	**20.3**	**10.3**
Hydrogen chloride	4	2,900	2	2,300	**2870**	**1456**
Hydrogen fluoride	3	230	2	140	**144**	**73**
Lead	3	2.6	8	9.0	**10.6**	**5.3**
Manganese	3	15	9	16	**9.5**	**4.8**
Mercury	3	0.24	3	0.48	**0.25**	**0.13**
Nickel	4	180	9	410	**389**	**197**
Phosphorus (P) [d]	All P factors were combined	Number of factors: 3	Median factor: 110 lb/trillion Btu		**68**	**34**
Selenium	1	1.4	8	3.8	**1.7**	**0.84**

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[a] Compounds are listed in the following sequence: inorganic, organic, and dioxin/furan/polycyclic aromatic hydrocarbons (PAHs). Median emission factors were determined from organic HAP concentrations at the stack, control device outlet, or boiler outlet when at least one of typically three measured flue gas concentrations was detected.

[b] Stack factors for inorganic HAPS were taken from test reports when at least one of typically three measured flue gas concentrations was detected. These factors were not used to develop the estimated emissions.

[c] Since the inorganic emissions were not directly estimated from stack factors, total emissions of inorganic HAPS projected with the computer program and from median stack factors will vary.

[d] Nationwide phosphorous emissions were determined from stack emission factors and not from EMFs.

Table A-3. Median Emission Factors, Determined from Test Report Data, and Total 1990 and Total 2010 HAP Emissions, Projected with the Emission Factor Program for Inorganic HAPs from Gas-fired Units[e]

Gas-fired units: inorganic HAPs	**Number of stack factors: no PM control**	**Median stack factor: no PM control (lb/trillion Btu)**	**Estimated total 1990 emissions (tons)**	**Estimated total 2010 emissions (tons)**
Arsenic	2	0.14	0.16	0.25
Cadmium	1	0.044	0.054	0.086
Chromium	2	0.96	1.2	1.9
Cobalt	1	0.12	0.14	0.23
Lead	2	0.37	0.44	0.68
Manganese	2	0.30	0.37	0.59
Mercury	2	< 0.38	0.0016	0.0024
Nickel	2	2.3	2.3	3.5
Phosphorus	1	2.2	1.3	2.0

Table A-4. Median Emission Factors, Determined from Test Report Data, and Total 1990 and Total 2010 HAP Emissions, Projected with the Emission Factor Program for Organic HAPs from Coal-fired Units

Coal-fired units: organic HAPs	Number of emission factors	Median emission factor: lb/trillion Btu	Computer program: 1990 total tons	Computer program: 2010 total tons
1,1,2-trichloroethane	1	4.7	40	53
2-chloroacetophenone	3	0.29	2.4	3.2
2,4-dinitrotoulene	3	0.015	0.13	0.17
Acetaldehyde	12	6.8	58	76
Acetophenone	7	0.68	5.8	7.7
Acrolein	6	3.3	28	37
Benzene	20	2.5	21	28
Benzylchloride	1	0.0056	0.048	0.063
Bis(2-ethylhexyl) phthalate	9	4.1	35	46
Bromoform	1	6.6	57	75

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[e] Compounds are listed in the following sequence: inorganic, organic, and dioxin/furan/polycyclic aromatic hydrocarbons (PAHs). Median emission factors were determined from organic HAP concentrations at the stack, control device outlet, or boiler outlet when at least one of typically three measured flue gas concentrations was detected.

Table A-4. (continued)

Coal-fired units: organic HAPs	Number of emission factors	Median emission factor: lb/trillion Btu	Computer program: 1990 total tons	Computer program: 2010 total tons
Carbondisulfide	8	4.3	37	48
Carbontetrachloride	2	3.3	28	37
Chlorobenzene	2	3.2	27	36
Chloroform	2	3.2	28	36
Cumene	1	0.29	2.5	3.2
Dibutylphthalate	5	2.8	24	32
Ethylbenzene	5	0.40	3.5	4.6
Ethylchloride	1	2.4	20	27
Methylchloroform	4	3.4	29	38
Ethylenedichloride	3	3.1	27	35
Formaldehyde	15	4.0	35	45
Hexane	2	0.82	6.9	9.1
Hexachlorobenzene	1	0.079	0.68	0.89
Isophorone	2	24	200	270
Methylbromide	6	0.88	7.7	10
Methylchloride	3	5.9	51	67
Methylethylketone	6	8.0	69	90
Methyliodide	1	0.40	3.4	4.5
Methylisobutylketone	3	4.9	42	53
Methylmethacrylate	1	1.1	9.3	12
Methyltertbutylether	1	1.4	12	16
Methylenechloride	5	13	110	150
n-nitrosodimethylamine	1	0.68	5.9	7.7
Naphthalene	11	0.77	6.6	8.7
m,p-cresol	2	0.68	5.8	7.6
o-cresol	3	1.7	14	19
p-cresol	1	0.95	8.2	11
Perylene	1	0.075	0.65	0.85
Pentachlorophenol	1	0.0082	0.070	0.093
Phenol	10	6.1	52	69
Phthalicanhydride	1	4.9	42	56
Propionaldehyde	4	10	89	120
Quinoline	1	0.053	0.46	0.61

Table A-4. (continued)

Coal-fired units: organic HAPs	Number of emission factors	Median emission factor: lb/trillion Btu	Computer program: 1990 total tons	Computer program: 2010 total tons
Styrene	7	3.1	27	35
Tetrachloroethylene	5	3.1	27	35
Toluene	17	3.6	31	41
Trans1,3-dichloropropene	1	4.7	40	53
Trichloroethylene	1	3.1	27	35
Vinylacetate	1	0.42	3.5	4.6
Vinylidenechloride	2	9.7	84	110
Xylenes	2	4.7	40	53
o-xylenes	5	0.82	6.9	9.1
m,p-xylenes	8	1.5	13	17
Total TEQ for 2,3,7,8-tetra-chlorodibenzo-p-dioxin	--	--	1.5 x 10^{-4}	2.0 x 10^{-4}
2,3,7,8-tetrachlorodi-benzo-p-dioxin	4	1.6 x 10^{-6}	1.4 x 10^{-5}	1.9 x 10^{-5}
1,2,3,7,8-pentachlorodi-benzo-p-dioxin	3	4.3 x 10^{-6}	3.7 x 10^{-5}	4.8 x 10^{-5}
1,2,3,4,7,8-hexachlorodi-benzo-p-dioxin	4	9.7 x 10^{-6}	8.3 x 10^{-5}	1.1 x 10^{-4}
1,2,3,6,7,8-hexachlorodi-benzo-p-dioxin	4	5.8 x 10^{-6}	5.0 x 10^{-5}	6.6 x 10^{-5}
1,2,3,7,8,9-hexachlorodi-benzo-p-dioxin	4	7.3 x 10^{-6}	6.3 x 10^{-5}	8.3 x 10^{-5}
1,2,3,4,6,7,8-heptachlorodi-benzo-p-dioxin	9	5.7 x 10^{-6}	4.9 x 10^{-5}	6.5 x 10^{-5}
Heptachlorodi-benzo-p-dioxin	6	1.1 x 10^{-4}	9.2 x 10^{-4}	1.2 x $^{10-3}$
Hexachlorodi-benzo-p-dioxin	8	2.4 x 10^{-5}	2.1 x 10^{-4}	2.7 x 10^{-4}
Octachlorodi-benzo-p-dioxin	9	5.8 x 10^{-5}	5.0 x 10^{-4}	6.6 x 10^{-4}
Pentachlorodi-benzo-p-dioxin	6	9.8 x 10^{-6}	8.5 x 10^{-5}	1.1 x 10^{-4}
Tetrachlorodi-benzo-p-dioxin	9	7.1 x 10^{-6}	6.1 x 10^{-5}	8.0 x 10^{-5}
2,3,7,8-tetrachlorodi-benzofuran	8	3.9 x 10^{-6}	3.4 x 10^{-5}	4.5 x 10^{-5}
1,2,3,7,8-pentachlorodi-benzofuran	5	2.4 x 10^{-6}	2.1 x 10^{-5}	2.8 x 10^{-5}
2,3,4,7,8-pentachlorodi-benzofuran	5	1.0 x 10^{-5}	9.0 x 10^{-5}	1.2 x 10^{-4}
1,2,3,4,7,8-hexachlorodi-benzofuran	6	1.3 x 10^{-5}	1.1 x 10^{-4}	1.5 x 10^{-4}
1,2,3,6,7,8-hexachlorodi-benzofuran	5	4.0 x 10^{-6}	3.4 x 10^{-5}	4.5 x 10^{-5}
1,2,3,7,8,9-hexachlorodi-benzofuran	4	8.5 x 10^{-6}	7.3 x 10^{-5}	9.6 x 10^{-5}
2,3,4,6,7,8-hexachlorodi-benzofuran	5	1.6 x 10^{-5}	1.4 x 10^{-4}	1.8 x 10^{-4}
1,2,3,4,6,7,8-heptachlorodi-benzofuran	8	2.0 x 10^{-5}	1.7 x 10^{-4}	2.2 x 10^{-4}
1,2,3,4,7,8,9-heptachlorodi-benzofuran	4	1.7 x 10^{-4}	1.5 x 10^{-3}	2.0 x 10^{-3}
Heptachlorodi-benzofuran	8	2.4 x 10^{-5}	2.1 x 10^{-4}	2.7 x 10^{-4}
Hexachlorodi-benzofuran	8	1.9 x 10^{-5}	1.6 x 10^{-4}	2.1 x 10^{-4}

Table A-4. (continued)

Coal-fired units: organic HAPs	Number of emission factors	Median emission factor: lb/trillion Btu	Computer program: 1990 total tons	Computer program: 2010 total tons
Octachlorodi-benzofuran	10	1.7×10^{-5}	**1.4×10^{-4}**	**1.9×10^{-4}**
Pentachlorodi-benzofuran	9	1.8×10^{-5}	**1.6×10^{-4}**	**2.1×10^{-4}**
Tetrachlorodi-benzofuran	10	1.2×10^{-5}	**1.0×10^{-4}**	**1.3×10^{-4}**
1-methylnaphthalene	2	0.0085	**0.076**	**0.10**
2-chloronaphthalene	2	0.040	**0.35**	**0.46**
2-methylnaphthalene	6	0.024	**0.20**	**0.26**
Acenaphthene	6	0.008	**0.07**	**0.09**
Acenaphthylene	5	0.0042	**0.036**	**0.047**
Anthracene	4	0.0042	**0.036**	**0.047**
Benz(a)anthracene	4	0.0021	**0.018**	**0.0024**
Benzo(a)pyrene	6	0.0010	**0.0088**	**0.012**
Benzo(e)pyrene	1	0.0012	**0.010**	**0.014**
Benzo(b)fluoranthene	1	0.0081	**0.070**	**0.092**
Benzo(b+k)fluoranthene	1	0.0016	**0.014**	**0.018**
Benzo(k)fluoranthene	1	0.0036	**0.031**	**0.040**
Benzo(g,h,i)perylene	2	0.0032	**0.028**	**0.036**
Biphenyl	1	0.34	3.1	4.0
Chrysene	4	0.0026	**0.022**	**0.030**
Dibenzo(a,h)anthracene	1	0.0003	**0.003**	**0.004**
Fluoranthene	6	0.007	**0.064**	**0.082**
Fluorene	5	0.013	**0.11**	**0.15**
Indeno(1,2,3-c,d)pyrene	2	0.0064	**0.054**	**0.072**
Phenanthrene	7	0.032	**0.31**	**0.36**
Pyrene	4	0.009	**0.081**	**0.103**

Table A-5. Median Emission Factors, Determined from Test Report Data, and Total 1990 and Total 2010 HAP Emissions, Projected with the Emission Factor Program for Organic HAPs from Oil-fired Units

Oil-fired units:organic HAPs	Number of emission factors	Median emission factor: lb/trillion Btu	Computer program: 1990 total tons	Computer program: 2010 total tons
Acetaldehyde	1	8.2	**5.0**	**2.6**
Benzene	6	1.4	**0.88**	**0.45**
Ethylbenzene	2	0.49	**0.29**	**0.15**
Formaldehyde	9	30	**19**	**9.5**
Methylchloroform	3	7.6	**4.6**	**2.4**
Methylenechloride	2	32	**20**	**10**
Naphthalene	4	0.33	**0.21**	**0.10**
Phenol	2	24	**15**	**7.5**
Tetrachloroethylene	1	0.55	**0.34**	**0.17**
Toluene	6	8.0	**4.9**	**2.5**
Vinylacetate	2	5.2	**3.2**	**1.6**
o-xylenes	1	0.84	**0.51**	**0.26**
m,p-xylenes	2	1.4	**0.82**	**0.42**
Total TEQ for 2,3,7,8-tetra-chlorodibenzo-p-dioxin	--	--	**1.1×10^{-5}**	**5.4×10^{-6}**
2,3,7,8-tetrachlorodi-benzo-p-dioxin	1	6.5×10^{-6}	**4.0×10^{-6}**	**2.0×10^{-6}**
1,2,3,7,8-pentachlorodi-benzo-p-dioxin	2	5.8×10^{-6}	**3.5×10^{-6}**	**1.8×10^{-6}**
1,2,3,4,7,8-hexachlorodi-benzo-p-dioxin	1	1.2×10^{-5}	**7.6×10^{-6}**	**3.9×10^{-6}**
1,2,3,6,7,8-hexachlorodi-benzo-p-dioxin	2	5.4×10^{-6}	**3.3×10^{-6}**	**1.7×10^{-6}**
1,2,3,7,8,9-hexachlorodi-benzo-p-dioxin	2	8.3×10^{-6}	**5.1×10^{-6}**	**2.6×10^{-6}**
1,2,3,4,6,7,8-heptachlorodi-benzo-p-dioxin	2	2.0×10^{-5}	**1.2×10^{-5}**	**6.2×10^{-6}**
Heptachlorodi-benzo-p-dioxin	2	2.0×10^{-5}	**1.2×10^{-5}**	**6.2×10^{-6}**
Hexachlorodi-benzo-p-dioxin	2	8.1×10^{-6}	**5.0×10^{-6}**	**2.5×10^{-6}**
Octachlorodi-benzo-p-dioxin	1	2.3×10^{-5}	**1.4×10^{-5}**	**7.3×10^{-6}**
Pentachlorodi-benzo-p-dioxin	2	5.8×10^{-6}	**3.5×10^{-6}**	**1.8×10^{-6}**
Tetrachlorodi-benzo-p-dioxin	2	5.7×10^{-6}	**3.4×10^{-6}**	**1.8×10^{-6}**
2,3,7,8-tetrachlorodi-benzofuran	2	4.6×10^{-6}	**2.9×10^{-6}**	**1.4×10^{-6}**
1,2,3,7,8-pentachlorodi-benzofuran	2	4.3×10^{-6}	**2.6×10^{-6}**	**1.3×10^{-6}**
2,3,4,7,8-pentachlorodi-benzofuran	2	4.8×10^{-6}	**3.0×10^{-6}**	**1.5×10^{-6}**
1,2,3,4,7,8-hexachlorodi-benzofuran	2	6.1×10^{-6}	**3.7×10^{-6}**	**1.9×10^{-6}**
1,2,3,6,7,8-hexachlorodi-benzofuran	2	3.8×10^{-6}	**2.3×10^{-6}**	**1.2×10^{-6}**
1,2,3,7,8,9-hexachlorodi-benzofuran	2	5.8×10^{-6}	**3.5×10^{-6}**	**1.8×10^{-6}**
2,3,4,6,7,8-hexachlorodi-benzofuran	1	4.8×10^{-6}	**3.0×10^{-6}**	**1.4×10^{-6}**

Table A-5. (continued)

Oil-fired units:organic HAPs	Number of emission factors	Median emission factor: lb/trillion Btu	Computer program: 1990 total tons	Computer program: 2010 total tons
1,2,3,4,6,7,8-heptachlorodi-benzofuran	1	9.4×10^{-6}	**5.7×10^{-6}**	**3.0×10^{-6}**
1,2,3,4,7,8,9-heptachlorodi-benzofuran	1	1.0×10^{-5}	**6.2×10^{-6}**	**3.2×10^{-6}**
Heptachlorodi-benzofuran	1	1.5×10^{-6}	**8.8×10^{-7}**	**4.4×10^{-7}**
Hexachlorodi-benzofuran	2	9.6×10^{-6}	**5.8×10^{-6}**	**3.0×10^{-6}**
Octachlorodi-benzofuran	1	1.0×10^{-5}	**6.2×10^{-6}**	**3.2×10^{-6}**
Pentachlorodi-benzofuran	2	7.3×10^{-6}	**4.4×10^{-6}**	**2.2×10^{-6}**
Tetrachlorodi-benzofuran	2	5.0×10^{-6}	**3.1×10^{-6}**	**1.5×10^{-6}**
2-methylnaphthalene	4	0.017	**0.010**	**0.0052**
Acenaphthene	2	0.36	**0.22**	**0.11**
Acenaphthylene	1	0.017	**0.010**	**0.0052**
Anthracene	2	0.015	**0.0093**	**0.0047**
Benz(a)anthracene	3	0.030	**0.018**	**0.0092**
Benzo(b+k)fluoranthene	2	0.033	**0.020**	**0.010**
Benzo(g,h,i)perylene	2	0.021	**0.013**	**0.0065**
Chrysene	3	0.021	**0.013**	**0.0066**
Dibenzo(a,h)anthracene	2	0.0081	**0.0050**	**0.0025**
Fluoranthene	6	0.016	**0.0097**	**0.0049**
Fluorene	5	0.021	**0.013**	**0.0065**
Indeno(1,2,3-c,d)pyrene	2	0.024	**0.014**	**0.0073**
Nitrobenzofluoranthene	1	0.015	**0.0092**	**0.0047**
Nitrochrysene/benzanthracene	1	0.016	**0.0098**	**0.0050**
Phenanthrene	9	0.025	**0.015**	**0.0077**
Pyrene	6	0.037	**0.022**	**0.011**

Table A-6. Median Emission Factors, Determined from Test Report Data, and Total 1990 and Total 2010 HAP Emissions, Projected with the Emission Factor Program for Organic HAPs from Gas-fired Units

Gas-fired units: organic HAPs	Number of emission factors	Median emission factor: lb/ trillion Btu [a]	Computer program: 1990 total tons	Computer program: 2010 total tons
Benzene	1	1.4	1.8	2.7
Formaldehyde	8	35.5	55	83
Naphthalene	2	0.70	0.66	1.0
Toluene	2	10	13	19
2-methylnaphthalene	2	0.026	0.025	0.038
Fluoranthene	1	0.0028	0.0034	0.0055
Fluorene	1	0.0026	0.0034	0.0051
1-phenanthrene	2	0.013	0.016	0.024
Pyrene	1	0.0049	0.0061	0.0094

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[a] The geometric mean of kg/10^9 cubic foot of natural gas factors were used in estimating organic emissions from gas-fired units. These median emission factors are in this table for comparison with previous tables and were not used directly to estimate organic HAPs.

Appendix B - Matrix of Electric Utility Steam-Generating Units and Emission Test Sites

Table B-1 is a matrix of utility boiler types and configurations showing each configuration's percentage of the total fossil-fuel-fired electric utility industry and the number of emission test sites analyzed in this report that fit into that category's configuration. Table B-2 shows the emission test sites whose data were used to develop this Report to Congress. Some sites are known only by their provider number because of nondisclosure agreements.

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Table B-1. Utility Boilers in the United States Organized by Unit Configuration, Fuel Type, and Emission Control Devices

Fuel	Fuel sulfur content %	Bottom type (Coal only)[a]	Particulate control[b]	Sulfur dioxide (SO_2) control type[c]	No. units	% Units	No. MWe[d]	% MWe	No. Units tested[e]
Bituminous coal	<1.5	Dry bottom	ESP	Compl fuel/SIP or NSPS	271	13.65	73385.57	14.67	2
Natural gas			None	Compl fuel/SIP or NSPS	388	19.54	67538.40	13.51	3
Bituminous coal	≥1.5 and ≤3.5	Dry bottom	ESP	Compl fuel/SIP or NSPS	274	13.80	64250.00	12.85	7
Subbituminous coal	<1.5	Dry bottom	ESP	Compl fuel/SIP or NSPS	91	4.58	40668.70	8.13	2
Fuel oil	<1.5		None	Compl fuel/SIP or NSPS	99	4.98	19948.40	3.99	6
Natural gas			None	N/A	110	5.54	18072.40	3.61	0 (Note 1)
Bituminous coal	≥1.5 and ≤3.5	Dry bottom	ESP	FGD	39	1.96	14699.90	2.94	2
Subbituminous coal	<1.5	Dry bottom	ESP	FGD	27	1.36	13536.60	2.71	2
Bituminous coal	≥1.5 and ≤3.5	Wet bottom	ESP	Compl fuel/SIP or NSPS	45	2.27	11078.70	2.22	2
Fuel oil	<1.5		ESP	Compl fuel/SIP or NSPS	45	2.27	10451.20	2.09	4
Fuel oil	<1.5		Multiclone	Compl fuel/SIP or NSPS	38	1.91	10403.55	2.08	0
Lignite	<1.5	Dry bottom	ESP	FGD	13	0.65	8744.70	1.75	1
Bituminous coal	<1.5	Dry bottom	ESP	Coal washing	24	1.21	8144.40	1.63	0 (Note 2)
Bituminous coal	<1.5	Dry bottom	ESP	FGD	18	0.91	7137.10	1.43	2
Bituminous coal	≥1.5 and ≤3.5	Dry bottom	ESP	Coal washing	11	0.55	6040.10	1.21	0 (Note 2)
Fuel oil	≥1.5 and ≤3.5		None	Compl fuel/SIP or NSPS	31	1.56	5767.40	1.15	1
Subbituminous coal	<1.5	Wet bottom	ESP	Compl fuel/SIP or NSPS	18	0.91	5731.70	1.15	3
Subbituminous coal	<1.5	Dry bottom	Fabric filter	FGD	10	0.50	5500.70	1.10	1
Bituminous coal	≥3.5	Dry bottom	ESP	FGD	11	0.55	5275.14	1.05	0
Other Types of Units Tested									
Bituminous coal	<1.5	Dry bottom	Fabric filter			< 0.55		< 1.05	2
Subbituminous coal	<1.5	AFBC	ESP			< 0.55		< 1.05	1

Table B-1. (continued).

Fuel	Fuel sulfur content %	Bottom type (Coal only)[a]	Particulate control[b]	Sulfur dioxide (SO_2) control type[c]	No. units	% Units	No. MWe[d]	% MWe	No. Units tested[e]
Subbituminous coal	<1.5	Dry bottom	Fabric filter	SDA		< 0.55		< 1.05	2
Bituminous coal	<1.5	AFBC	Fabric filter			< 0.55		< 1.05	1
Bituminous / Subbituminous coal	<1.5	Dry bottom	Fabric filter	SDA		< 0.55		< 1.05	1
Fuel oil	<1.5		SCR			< 0.55		< 1.05	1
Fuel oil	<1.5		PJFF			< 0.55		< 1.05	1
Bituminous coal	≥1.5 and ≤3.5	Dry bottom	PJFF	SDA		< 0.55		< 1.05	1
Bituminous coal	<1.5	Dry bottom	ESP/COHPAC			< 0.55		< 1.05	1
Natural gas		Combined cycle gas turbine	Water injection			< 0.55		< 1.05	1
Lignite coal	<1.5	AFBC	Fabric filter			< 0.55		< 1.05	1
Bituminous coal	≥1.5 and ≤3.5	Wet bottom	Fabric filter	WSA/SCR		< 0.55		< 1.05	1

[a] Bottom type:dry bottom =bottom ash is disposed of in a dry state / wet bottom = bottom ash is kept molten by sweeping the furnace bottom with the furnace burners after which the bottom ash is disposed of by a molten liquid removal system.
AFBC - atmospheric Fluidized bed combustor

[b] ESP = electrostatic Precipitator
Multicyclone = a series of cyclones (impact particle collectors)
PJFF = pulse jet fabric
COHPAC = compact hybrid particulate collector

[c] FGD: fluidized gas desulfurization (wet scrubber with some type of reagent)
SDA = spray drier absorber [SO_2 control]
WSA = wet sulfuric acid (system)
Coal washing: coal is washed before burning to remove both ash and sulfur.
Compl fuel/SIP or NSPS = fuel burned complies with applicable State Implementation Plan (SIP) or other local regulation or New Source Performance Standards (NSPS) requirements.
SCR = selective catalytic reduction [NO_x control]
Note 1: These units are essentially the same as the natural gas/compliance fuel/SIP or NSPS except these units do not have SIP or NSPS requiring them to control SO_2 emissions.
Note 2: From the research performed, almost all bituminous coal is washed. These utilities choose to report coal washing as a SO_2 control procedure but it could be collapsed into other unit types.

Table B-2. Utility Boiler Emission Tests

Provider[a]	Site	Fuel	Fuel sulfur	Bottom type	Control[b]	Test contractor	Reference	Report date[c]
DOE	Baldwin	Bituminous coal	Medium	Wet	ESP	Roy F. Weston	1	12/93 (P)
DOE	Boswell	Subbituminous coal	Low	Dry	FF	Roy F. Weston	2	12/93 (P)
DOE	Cardinal	Bituminous coal	Medium	Dry	ESP	Energy and Environmental Research Corp.	3	12/93 (P)
DOE	Coal Creek	Lignite coal	Low	Dry	ESP/FGD	Battelle	4	12/93 (P)
DOE	Niles	Bituminous coal	Medium	Wet	ESP	Battelle	5	12/93 (P)
DOE	Niles/NO_x	Bituminous coal	Medium	Wet	FF/SCR/WSA	Battelle	6	12/93 (P)
DOE	Springerville	Subbituminous coal	Low	Dry	SDA/FF	Southern Research Institute	7	12/93 (P)
DOE	Yates	Bituminous coal	Medium	Dry	ESP/JBR	Radian	8	12/93 (P)
NSPC	A.S. King	Subbituminous coal	Low	Wet	ESP	Interpoll	9	11/91 (P)
NSPC	Black Dog 1,3,4	Subbituminous coal	Low	Dry	ESP	Interpoll	10	1/92 (P)
NSPC	Black Dog 2	Subbituminous coal	Low	AFBC	ESP	Interpoll	11	5/92 (P)
NSPC	High Bridge 3,4,5,6	Subbituminous coal	Low	Dry	ESP	Interpoll	12	1/92 (P)
NSPC	Riverside 6,7	Subbituminous coal	Low	Dry	FF	Interpoll	13	2/92 (P)
NSPC	Riverside 8	Subbituminous coal	Low	Wet	ESP	Interpoll	14	9/92 (P)
NSPC	Sherburne 1,2	Subbituminous coal	Low	Dry	FGD	Interpoll	15,16,17	7/90, 10/91 (P)
NSPC	Sherburne 3	Subbituminous coal	Low	Dry	SDA/FF	Interpoll	18	6/90, 10/91 (P)
EPRI	Site 10	Bituminous coal	Low	AFBC	FF	Radian	19	10/92 (P)
EPRI	Site 102	Subbituminous coal	Low	Wet	ESP	Radian	20	2/93 (P)
EPRI	Site 11	Subbituminous coal	Low	Dry	ESP/FGD	Radian	21	10/92, 10/93 (P)
EPRI/SRI	Site 110	Bituminous coal	Medium	Dry	H-ESP/ C-ESP	Southern Research Institute	22	10/93 (P)

Table B-2. Continued

Provider[a]	Site	Fuel	Fuel sulfur	Bottom type	Control[b]	Test contractor	Reference	Report date[c]
EPRI/SRI	Site 110A/Low NO_x	Bituminous coal	Medium	Dry	H-ESP/ C-ESP	Southern Research Institute	22	10/93 (P)
EPRI	Site 111	Bituminous/ Subbituminous coal	Low	Dry	SDA/FF	Radian	23	1/94 (P)
EPRI	Site 112	Oil	Low	Dry	ESP	Carnot	24	12/93, 3/94 (P)
EPRI	Site 114	Bituminous coal	Medium	Wet	ESP	NA	25	5/94 (P)
EPRI	Site 115	Bituminous coal	Low	Dry	FF	Carnot	26	NA
EPRI	Site 117	Oil	Low	Dry	SCR	Carnot	27	1/94 (P)
EPRI	Site 118	Oil	Low	Dry	ESP	Carnot	28	1/94 (P)
EPRI	Site 119	Oil	Low	Dry	ESP	Carnot	29	1/94 (P)
EPRI	Site 12	Bituminous coal	Medium	Dry	ESP/FGD	Radian	30	11/92, 10/93 (P)
EPRI	Site 120	Gas		Dry	None	NA	31	NA
EPRI	Site 121	Gas		Dry	None	NA	32	NA
EPRI	Site 13	Oil		Dry	PJFF	Radian	33	2/93 (P)
EPRI	Site 14	Bituminous coal	Medium	Dry	SDA/PJFF	Radian	34	11/92 (P)
EPRI	Site 15	Bituminous coal	Medium	Dry	ESP	Radian	35	10/92 (P)
EPRI	Site 16/OFA	Bituminous coal	Medium	Dry	ESP	Radian	36	11/93 (P)
EPRI	Site 16/OFA/Low NO_x	Bituminous coal	Medium	Dry	ESP	Radian	36	11/93 (P)
EPRI	Site 18	Bituminous coal	Low	Dry	ESP/ COHPAC	Radian	37	4/93 (P)
EPRI	Site 19	Bituminous coal	Low	Dry	ESP	Radian	38	4/93 (P)
EPRI	Site 21	Bituminous coal	Medium	Dry	ESP/FGD	Radian	39	5/93 (P)
EPRI	Site 22	Bituminous coal	Low	Dry	ESP	Radian/Carnot	40	3/93, 2/94 (P)
EPRI	Site 103	Oil	Low	Dry	None	Carnot	41	3/93 (P)

Table B-2. Continued

Provider[a]	Site	Fuel	Fuel sulfur	Bottom type	Control[b]	Test contractor	Reference	Report date[c]
EPRI	Site 104	Oil	Low	Dry	None	Carnot	41	3/93 (P)
EPRI	Site 105	Oil	Low	Dry	None	Carnot	41	3/93 (P)
EPRI	Site 106	Oil	Low	Dry	None	Carnot	41	3/93 (P)
EPRI	Site 107	Oil	Low	Dry	None	Carnot	41	3/93 (P)
EPRI	Site 108	Oil	Medium	Dry	None	Carnot	41	3/93 (P)
EPRI	Site 109	Oil	Low	Dry	None	KVB - Analect	41	3/93 (P)
EPA	T. H. Wharton (Combined Cycle Gas Turbine)	Gas			Water injection	Entropy	42	9/93 (F)
EPA	Northport	Oil	Low	Dry	ESP	Entropy	43	9/93 (F)
EPA	Hammond (TNP One)	Lignite coal	Low	AFBC	FF	Entropy	44	10/93 (F)
EPA	Greens Bayou	Gas		Dry	None	Entropy	45	11/93 (F)
EPA	Kintigh	Bituminous coal	Medium	Dry	ESP/FGD	Entropy	46	12/93 (F)

[a] DOE = Department of Energy
NSPC = Northern States Power Company
EPRI = Electric Power Research Institute
SRI = Southern Research Institute

[b] ESP = *Electrostatic precipitator*
FF = Fabric filter
FGD = Flue gas desulfurization
SCR = Selective catalytic reduction [NO_x control]
WSA = Wet sulfuric acid (system)
SDA = Spray drier absorber [SO_2 control]
JBR = Jet bubbling reactor [SO_2 control]
H-ESP = Hot-side electrostatic precipitator
C-ESP = Cold-side electrostatic precipitator
PJFF = Pulse jet fabric filter
COHPAC = Compact hybrid particulate collector

[c] (P) - Preliminary Test Report
(F) - Final Test Report

Appendix B: References

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2. Preliminary draft emissions report for Boswell Energy Center - Unit 2 (Minnesota Power Company) for the Comprehensive Assessment of Toxic Emissions from Coal-fired Power Plants, prepared by Roy F. Weston, Inc., for the Department of Energy/Pittsburgh Energy Technology Center (DOE/PETC), DOE contract # DE-AC22-93PC93255, Weston project # 10016-011, Weston report # DOE017G.RP1. December 1993.

3. Preliminary draft emissions report for Cardinal Station - Unit 1 (American Electric Power) for the Comprehensive Assessment of Toxic Emissions from Coal-fired Power Plants, prepared by Energy and Environmental Research Corp. for the Department of Energy/Pittsburgh Energy Technology Center (DOE/PETC), DOE contract # DE-AC22-93PC93252. December 1993.

4. Preliminary draft emissions report for Coal Creek Station - Unit 2 (Cooperative Power Association) for the Comprehensive Assessment of Toxic Emissions from Coal-fired Power Plants, prepared by Battelle for the Department of Energy/Pittsburgh Energy Technology Center (DOE/PETC), DOE contract # DE-AC22-93PC93251. December 1993.

5. Preliminary draft emissions report for Niles Station Boiler No. 2 (Ohio Edison) for the Comprehensive Assessment of Toxic Emissions from Coal-fired Power Plants, prepared by Battelle for the Department of Energy/Pittsburgh Energy Technology Center (DOE/PETC), DOE contract # DE-AC22-93PC93251. December 1993.

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9. Results of the Air Toxic Emission Study on the No. 1 Boiler at the NSPC A.S. King Plant, prepared by Interpoll Laboratories, Inc., for NSPC, report No. 1-3304. November 1991.

10. Results of the Air Toxic Emission Study on the Nos. 1, 3, & 4 Boilers at the NSPC Black Dog Plant, prepared by Interpoll Laboratories, Inc., for NSPC, report No. 1-3451. January 1992.

11. Results of the Air Toxic Emission Study on the No. 2 Boiler at the NSPC Black Dog Plant, prepared by Interpoll Laboratories, Inc., for NSPC, report No. 2-3496. May 1992.

12. Results of the Air Toxic Emission Study on the Nos. 3, 4, 5, & 6 Boilers at the NSPC High Bridge Plant, prepared by Interpoll Laboratories, Inc., for NSPC, report No. 1-3453. January 1992.

13. Results of the Air Toxic Emission Study on the Nos. 6 & 7 Boilers at the NSPC Riverside Plant, prepared by Interpoll Laboratories, Inc., for NSPC, report No. 1-3468A. February 1992.

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42. Final electric utility combined cycle gas-fired gas turbine emission test report for T.H. Wharton Electric Generating Station (Houston Lighting and Power Company), prepared by Entropy, Inc., for the U. S. Environmental Protection Agency, Emissions Measurement Branch (EPA/EMB), EMB report No. 93-UTL-2. May 1994.

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Appendix C - Listing of Emission Modification Factors for Trace Elements Used in the Individual Boiler Analysis

Note: The following test reports were not used to develop emission modification factors (EMFs) for the reasons listed below. Northern States Power's (NSP) A.S. King unit is the same test site as the Electric Power Research Institute's (EPRI's) Site 102, and the EPA chose to use the EPRI test report. Northern States Power's Sherco unit 1 and 2 were not used to develop boiler EMFs because no coal composition data were provided. Northern States Power's Black Dog unit 1 was not used to develop boiler EMFs because tangentially-fired emissions were combined with emissions from two front-fired boilers. Finally, NSPC's High Bridge was not used to develop boiler EMFs because the test report was missing the coal feed rate during testing.

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Table C-1. Tested EMFs and Geometric Means used in the Emission Factor Program for Circulating Fluidized Bed Furnaces (coal-fired)

Unit Name	EPRI Site 10	NSP - Black Dog #2	EMF (Geometric mean)	Geometric standard deviation
ARSENIC	1.00	0.59	0.77	1.44
BERYLLIUM	0.77	0.41	0.56	1.56
CHROMIUM	0.40	0.54	0.46	1.25
COBALT	1.00		1.00	N/A
LEAD	0.49	0.36	0.42	1.24
MANGANESE	0.59	0.68	0.63	1.11
MERCURY	1.00	1.00	1.00	1.00
NICKEL	1.00	0.45	0.67	1.76
SELENIUM	1.00	0.71	0.84	1.27

Table C-2. Tested EMFs and Geometric Means used in the Emission Factor Program for Tangentially-fired, dry-bottom furnace with NO_x control (coal-fired)

Unit Name	EPRI Site 11	DOE - Coal Creek	DOE - Springerville	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY		0.01	0.03	0.02	2.17
ARSENIC	0.92	0.69	0.29	0.57	1.83
BERYLLIUM	0.79	0.35	0.87	0.62	1.64
CADMIUM	0.35	0.11	1.00	0.34	3.01
CHROMIUM	0.72	1.00	0.68	0.79	1.23
COBALT	0.92	0.61	0.73	0.74	1.23
LEAD	1.00	0.29	0.19	0.38	2.38
MANGANESE	0.98	0.59	0.72	0.75	1.30
MERCURY	0.29	0.85	1.00	0.63	1.98
NICKEL	0.25	1.00	0.70	0.56	2.07
SELENIUM	1.00	0.38	0.93	0.71	1.70

Table C-3. Tested EMFs and Geometric Means used in the Emission Factor Program for Tangentially-fired, dry-bottom furnace without NO_x control (coal-fired)

Unit Name	EPRI Site 15	DOE - Yates	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY		0.67	0.67	N/A
ARSENIC	0.60	1.00	0.77	1.44
BERYLLIUM	0.54	1.00	0.74	1.54
CADMIUM	0.01	1.00	0.11	22.68
CHROMIUM	0.58	1.00	0.76	1.47
COBALT	0.94	0.97	0.95	1.02
LEAD	1.00	1.00	1.00	1.00
MANGANESE	0.81	1.00	0.90	1.16
MERCURY	1.00	1.00	1.00	1.00
NICKEL	0.43	0.84	0.60	1.59
SELENIUM	0.70	0.70	0.70	1.01

Table C-4. Tested EMFs and Geometric Means used in the Emission Factor Program for Opposed-fired, dry-bottom furnace with NO_x control (coal-fired)

Unit Name	EPRI Site 12	EPRI Site 14	NSP - Sherburne #3	EPRI Site 111	EPRI Site 16 w/OFA and LNO_x Burners	EPRI Site 16 w/OFA	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY					0.80	1.00	0.90	1.17
ARSENIC	1.00	0.50	0.79	0.11	1.00	1.00	0.59	2.41
BERYLLIUM	1.00	0.92	0.58		0.82	1.00	0.85	1.25
CADMIUM	0.14	0.02	0.99	0.05	0.11	1.00	0.16	4.68
CHROMIUM	1.00	0.67	0.49	0.20	0.69	0.58	0.55	1.72
COBALT	1.00	1.00			0.66	0.61	0.80	1.31
LEAD	1.00	0.79	0.49		0.66	1.00	0.76	1.35
MANGANESE	1.00	0.93			0.88	0.60	0.84	1.26
MERCURY	0.74	0.74	1.00		1.00	0.64	0.81	1.22
NICKEL	0.29	0.37	0.67		0.54	0.33	0.42	1.42
SELENIUM	1.00	0.05	0.21		0.37	1.00	0.33	3.51

Table C-5. Tested EMFs and Geometric Means used in the Emission Factor Program for Front-fired, dry-bottom furnace without NO_x control (coal-fired)

Unit Name	NSP - Riverside #6-7	DOE - Cardinal	DOE - Boswell	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY	0.20	0.08	0.59	0.21	2.74
ARSENIC	0.99	0.91	0.23	0.59	2.25
BERYLLIUM	0.40	0.96	0.60	0.62	1.54
CADMIUM	0.25	1.00	1.00	0.63	2.22
CHROMIUM	1.00	0.61	1.00	0.85	1.33
COBALT		0.96	0.98	0.97	1.02
LEAD	0.19	1.00	0.42	0.43	2.27
MANGANESE	0.77	0.27	0.57	0.49	1.73
MERCURY	1.00	0.41	0.87	0.71	1.62
NICKEL	0.78	0.76	1.00	0.84	1.16
SELENIUM	1.00	0.07	0.14	0.21	4.09

Table C-6. Tested EMFs and Geometric Means used in the Emission Factor Program for Cyclone-fired, wet-bottom furnace without NO_x control (coal-fired)

Unit Name	EPRI Site 102	NSP - Riverside #8	EPRI Site 114	EPRI Site 114, 2nd Test	DOE - Niles #2	DOE - Niles #2 w/NOx Control	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY		0.61			1.00	0.92	0.82	1.30
ARSENIC	0.48	0.51	0.15	0.25	0.58	0.85	0.41	1.86
BERYLLIUM	0.04	0.08	0.15	0.15	0.26	0.30	0.13	2.09
CADMIUM	0.02	0.16	0.01	0.01	0.11	0.20	0.04	5.01
CHROMIUM	0.25	0.22	0.30	0.23	0.28	0.35	0.27	1.19
COBALT	0.21				0.20	0.33	0.24	1.31
LEAD	0.61	0.38	0.50	0.84	0.56	0.60	0.56	1.30
MANGANESE	0.33	0.13	0.20	0.18	0.15	0.19	0.19	1.37
MERCURY	1.00	1.00	0.73	0.54	1.00	1.00	0.86	1.30
NICKEL	0.30	0.12	0.72	0.31	0.29	0.39	0.31	1.76
SELENIUM	0.65	1.00	0.26	1.00	1.00	0.92	0.74	1.70

Table C-7. Tested EMFs and Geometric Means used in the Emission Factor Program for Vertically-fired, dry-bottom furnace with NO_x control (coal-fired)

Unit Name	EPRI Site 115	EMF (Geometric mean)	Geometric standard deviation
ARSENIC	0.61	0.61	N/A
BERYLLIUM	0.52	0.52	N/A
CADMIUM	0.58	0.58	N/A
CHROMIUM	0.57	0.57	N/A
COBALT	1.00	1.00	N/A
LEAD	0.38	0.38	N/A
MANGANESE	0.58	0.58	N/A
MERCURY	0.78	0.78	N/A
NICKEL	0.64	0.64	N/A
SELENIUM	0.34	0.34	N/A

Table C-8. Tested EMFs and Geometric Means used in the Emission Factor Program for Tangentially-fired, dry-bottom furnace without NO_x control (used as the factor on all units without NO_x control, which are controlled by hot-side ESPs) (coal-fired)

Unit Name	EPRI Site 110	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY	0.55	0.55	N/A
ARSENIC	0.89	0.89	N/A
BERYLLIUM	0.93	0.93	N/A
CADMIUM	1.00	1.00	N/A
CHROMIUM	1.00	1.00	N/A
COBALT	0.93	0.93	N/A
LEAD	1.00	1.00	N/A
MANGANESE	0.71	0.71	N/A
MERCURY	1.00	1.00	N/A
NICKEL	0.91	0.91	N/A
SELENIUM	0.58	0.58	N/A

Table C-9. Tested EMFs and Geometric Means used in the Emission Factor Program for Tangentially-fired, dry-bottom furnace with NO_x control (used as the factor on all units without NO_x control, which are controlled by hot-side ESPs) (coal-fired)

Unit Name	EPRI Site 110 w/NO_x control	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY	0.66	0.66	N/A
ARSENIC	0.39	0.39	N/A
BERYLLIUM	0.43	0.43	N/A
CADMIUM	0.70	0.70	N/A
CHROMIUM	1.00	1.00	N/A
COBALT	1.00	1.00	N/A
LEAD	0.36	0.36	N/A
MANGANESE	0.76	0.76	N/A
MERCURY	0.75	0.75	N/A
NICKEL	0.97	0.97	N/A
SELENIUM	0.82	0.82	N/A

Table C-10. Tested EMFs and Geometric Means used in the Emission Factor Program for Opposed-fired, dry-bottom furnace without NO_x control (coal-fired)

Unit Name	DOE - Baldwin	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY	0.32	0.32	N/A
ARSENIC	1.00	1.00	N/A
BERYLLIUM	0.44	0.44	N/A
CADMIUM	1.00	1.00	N/A
CHROMIUM	0.42	0.42	N/A
COBALT	0.34	0.34	N/A
LEAD	0.53	0.53	N/A
MANGANESE	0.20	0.20	N/A
MERCURY	0.92	0.92	N/A
NICKEL	0.43	0.43	N/A
SELENIUM	0.04	0.04	N/A

Table C-11. Tested EMFs and Geometric Means used in the Emission Factor Program for Front-fired, dry-bottom furnace without NO_x control (oil-fired)

Unit Name	EPRI Site 13, no NOx	EPRI Site 103	EPRI Site 104	EPRI Site 105	EPRI Site 107	EPRI Site 108	EPRI Site 117	EMF (Geometric mean)	Geometric standard deviation
ARSENIC	1.00	0.06	0.10	0.07	0.23	0.12	0.40	0.17	2.77
BERYLLIUM		0.02	0.02				1.00	0.07	10.64
CADMIUM	0.50	0.19	0.06	0.06	0.14	0.36	0.03	0.13	2.72
CHROMIUM	0.25	0.29	0.22	0.02	0.66	0.05	0.56	0.18	3.47
COBALT	0.82	0.32					0.42	0.48	1.62
LEAD	0.03	0.06	0.04	0.14		0.16	0.32	0.09	2.47
MANGANESE	1.00	1.00	0.57	0.16	0.45	1.00	0.21	0.51	2.14
MERCURY	1.00						1.00	1.00	1.00
NICKEL	0.89	0.69	0.28	0.40	0.74	0.90	0.76	0.62	1.57
SELENIUM		0.005		0.04	0.07	0.23	0.37	0.06	5.59

Table C-12. Tested EMFs and Geometric Means used in the Emission Factor Program for Opposed-fired, dry-bottom furnace without NO_x control (oil-fired)

Unit Name	EPRI Site 106	EPRI Site 109	EMF (Geometric mean)	Geometric standard deviation
ARSENIC	0.45	0.01	0.08	11.80
BERYLLIUM	0.02		0.02	N/A
CADMIUM	0.10	0.39	0.20	2.61
CHROMIUM	0.32	1.00	0.57	2.23
COBALT	1.00	1.00	1.00	1.00
LEAD	0.46	0.26	0.35	1.50
MANGANESE	1.00	0.80	0.89	1.17
MERCURY		0.04	0.04	N/A
NICKEL	1.00	0.79	0.89	1.18
SELENIUM	0.10	0.02	0.04	3.41

Table C-13. Tested EMFs and Geometric Means used in the Emission Factor Program for Front-fired, dry-bottom furnace with NO_x control (oil-fired)

Unit Name	EPRI Site 13	EPRI Site 118	EPRI Site 117	EMF (Geometric mean)	Geometric standard deviation
ARSENIC	0.64	0.14	1.00	0.44	2.83
BERYLLIUM		1.00	1.00	1.00	1.00
CADMIUM	0.47		1.00	0.69	1.71
CHROMIUM	0.19	0.78	1.00	0.53	2.42
COBALT	0.62	0.29	0.98	0.56	1.83
LEAD	0.08	0.57	0.97	0.35	3.73
MANGANESE	1.00	1.00	1.00	1.00	1.00
MERCURY	1.00	1.00	1.00	1.00	1.00
NICKEL	0.71	0.64	1.00	0.77	1.26
SELENIUM	0.58	0.46	1.00	0.64	1.50

Table C-14. Tested EMFs and Geometric Means used in the Emission Factor Program for Tangentially-fired, dry-bottom furnace without NO_x control (oil-fired)

Unit Name	EPRI Site 112	EMF (Geometric mean)	Geometric standard deviation
ARSENIC	1.00	1.00	N/A
BERYLLIUM	0.79	0.79	N/A
CADMIUM	0.67	0.67	N/A
CHROMIUM	0.66	0.66	N/A
COBALT	0.38	0.38	N/A
LEAD	0.26	0.26	N/A
MANGANESE	0.80	0.80	N/A
MERCURY	1.00	1.00	N/A
NICKEL	0.53	0.53	N/A
SELENIUM	1.00	1.00	N/A

Table C-15. Tested EMFs and Geometric Means used in the Emission Factor Program for Tangentially-fired, dry-bottom furnace with NO_x control (oil-fired)

Unit Name	EPRI Site 119	See Note 1	EMF (Geometric mean)	Geometric standard deviation
ARSENIC		0.44	0.44	N/A
BERYLLIUM		1.00	1.00	N/A
CADMIUM		0.69	0.69	N/A
CHROMIUM	1.00	0.62	0.79	1.40
COBALT		0.56	0.56	N/A
LEAD		0.35	0.35	N/A
MANGANESE		1.00	1.00	N/A
MERCURY		1.00	1.00	N/A
NICKEL	0.57	0.72	0.64	1.17
SELENIUM		0.64	0.64	N/A

Note 1. - Since the only source of data for this type of unit was limited to data on only two metals, it was decided to take the data from another similar unit (a front-fired, dry-bottom furnace with NOx control) along with the 2-data-point set to develop a set of geometric means. This set of geometric means is the data set in the "See Note 1" column. The geometric means of the "See Note 1" set and the 2-data-point set were derived. These means were used to represent a tangential-fired, dry-bottom furnace with NO_x control burning oil.

Table C-16. Tested EMFs and Geometric Means used in the Emission Factor Program for Fabric Filters (baghouses)

Unit Name	EPRI Site 10	EPRI Site 13 (controlling an oil-fired unit) [a]	NSP - Riverside #6-7	EPRI Site 115	DOE - Niles #2 w/NO$_x$ Control	DOE - Boswell	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY			0.03		0.01	0.06	0.02	3.42
ARSENIC	0.001	0.01	0.03	0.03	0.004	0.01	0.01	3.52
BERYLLIUM	0.004		0.06	0.003	0.001	0.01	0.01	4.55
CADMIUM			1.00	0.05	0.01	0.06	0.08	6.18
CHROMIUM		0.25	0.05	0.01	0.003	0.00	0.02	6.29
COBALT	0.004	0.001		0.01	0.001	0.01	0.003	3.10
LEAD	0.002	0.005	0.03	0.01	0.001	0.01	0.01	4.09
MANGANESE	0.02	0.15	0.05	0.01	0.01	0.00	0.01	4.58
MERCURY		1.00	1.00	0.27	0.92	0.39	0.63	1.86
NICKEL	0.002	0.001	0.20	0.05	0.001	0.01	0.01	10.57
SELENIUM			0.06	0.02	0.79	0.31	0.12	5.66
HYDROGEN CHLORIDE	Note 1						0.56	
HYDROGEN FLUORIDE	Note 1						1.00	

Note 1 - These EMFs were developed from four specific emission tests that examined HCl and HF emissions through an ESP, a baghouse, an SDA/FF, and a FGD.

Table C-17. Tested EMFs and Geometric Means used in the Emission Factor Program for Electrostatic Precipitators - hot side (located before the air preheater, controlling an coal-fired unit)

Unit Name	EPRI Site 110	EPRI Site 110 w/NO$_x$ Control	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY	0.11	0.02	0.04	3.87
ARSENIC	0.01	0.15	0.04	7.13
BERYLLIUM	0.01	0.01	0.01	1.08
CADMIUM	0.004	0.01	0.01	2.39
CHROMIUM	0.02	0.04	0.03	1.84
COBALT	0.04	0.02	0.03	1.55
LEAD	0.02	0.03	0.02	1.59
MANGANESE	0.04	0.02	0.02	1.86
MERCURY	1.00	1.00	1.00	1.00
NICKEL	0.002	0.01	0.004	3.24
SELENIUM	1.00	0.87	0.93	1.10
HYDROGEN CHLORIDE	Note 2		1.00	
HYDROGEN FLUORIDE	Note 2		1.00	

Note 2 - Because there were no data on HCl and HF emissions through an ESP attached to an oil-fired unit or a hot-side ESP attached to a coal-fired unit, the EMF was left as "1" so that all HCl and HF emissions passed through the ESP.

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[a] Although Site 13 is an oil-fired unit, the data were used here because they are a measure of baghouse performance at removing HAPs)

Table C-18. Tested EMFs and Geometric Means used in the Emission Factor Program for Electrostatic Precipitators - cold side (located after the air preheater, controlling an oil-fired unit)

Unit Name	EPRI Site 112	EPRI Site 118	EMF (Geometric mean)	Geometric standard deviation
ARSENIC	0.49	0.55	0.52	1.09
BERYLLIUM	0.23	0.10	0.16	1.76
CADMIUM	0.69		0.69	N/A
CHROMIUM	0.44	0.44	0.44	1.00
COBALT	0.25	0.08	0.14	2.27
LEAD	0.47	0.43	0.45	1.07
MANGANESE	1.00	0.83	0.91	1.14
MERCURY	0.17	0.58	0.31	2.39
NICKEL	0.27	0.07	0.14	2.50
SELENIUM		0.65	0.65	N/A
HYDROGEN CHLORIDE	Note 2		1.00	
HYDROGEN FLUORIDE	Note 2		1.00	

Note 2 - Because there were no data on HCl and HF emissions through an ESP attached to an oil-fired unit or a hot-side ESP attached to a coal-fired unit, the EMF was left as "1" so that all HCl and HF emissions passed through the ESP.

Table C-19. Tested EMFs and Geometric Means used in the Emission Factor Program for Particulate Matter Scrubber Unit - (controlling a coal-fired unit)

Unit Name	EPRI Site 125	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY	0.10	0.10	N/A
ARSENIC	0.10	0.10	N/A
BERYLLIUM	0.02	0.02	N/A
CADMIUM	0.09	0.09	N/A
CHROMIUM	0.03	0.03	N/A
COBALT	0.02	0.02	N/A
LEAD	0.03	0.03	N/A
MANGANESE	0.01	0.01	N/A
MERCURY	0.96	0.96	N/A
NICKEL	0.01	0.01	N/A
SELENIUM	1.00	1.00	N/A
HYDROGEN CHLORIDE	0.06	0.06	N/A
HYDROGEN FLUORIDE	0.09	0.09	N/A

Table C-20. Tested EMFs and Geometric Means used in the Emission Factor Program for Fluidized Gas Desulfurization Unit - (controlling a coal-fired unit)

Unit Name	EPRI Site 11	EPRI Site 12	NSP - Sherburne #1-2	NSP - Sherburne #1-2 (mercury 2nd test)	DOE - Yates	DOE - Coal Creek	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY					0.21	0.16	0.18	1.23
ARSENIC	0.49	0.23	0.04		0.08	0.76	0.20	3.29
BERYLLIUM			0.02		0.11	0.97	0.13	6.96
CADMIUM	0.85	0.62	0.16		0.75		0.50	2.16
CHROMIUM	0.51	0.40	0.03		0.18	1.00	0.26	3.72
COBALT	1.00	0.23			0.23	0.91	0.46	2.30
LEAD	1.00	0.59	0.02		0.04	0.93	0.22	6.36
MANGANESE	1.00	0.03			0.17	1.00	0.26	5.65
MERCURY	0.89	1.00	0.77	0.41	0.54	0.88	0.71	1.42
NICKEL	0.88	1.00	0.05		1.00	1.00	0.53	3.79
SELENIUM	0.17	0.27	0.17		0.26	0.60	0.26	1.66
HYDROGEN CHLORIDE	Note 1						0.21	
HYDROGEN FLUORIDE	Note 1						0.73	

Note 1 - These EMFs were developed from four specific emission tests that examined HCl and HF emissions through an ESP, a baghouse, an SDA/FF, and a FGD.

Table C-21. Tested EMFs and Geometric Means used in the Emission Factor Program for Spray Dryer Adsorber / Fabric Filter Units - (controlling a coal-fired unit)

Unit Name	NSP - Sherburne #3	NSP - Sherburne #3 (mercury 2nd test)	EPRI Site 111	EPRI Site 14	DOE - Springerville	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY	0.01				0.001	0.003	2.53
ARSENIC	0.001		0.04	0.002	0.0003	0.002	8.17
BERYLLIUM	0.000			0.01	0.0001	0.001	8.48
CADMIUM	0.08		0.13	0.19	0.001	0.04	11.55
CHROMIUM	0.03		0.03	0.005		0.02	2.91
COBALT				0.01	0.0004	0.001	6.05
LEAD	0.01			0.002	0.001	0.003	5.14
MANGANESE	0.003			0.003	0.001	0.002	1.90
MERCURY	0.54	0.46		1.00	0.98	0.70	1.50
NICKEL	0.04		0.04	0.01	0.0003	0.01	9.51
SELENIUM	0.05			0.13	0.0004	0.01	22.46
HYDROGEN CHLORIDE	Note 1					0.18	
HYDROGEN FLUORIDE	Note 1					0.18	

Note 1 - These EMFs were developed from four specific emission tests that examined HCl and HF emissions through an ESP, a baghouse, an SDA/FF, and a FGD.

Table C-22. Tested Emission Modification Factors (EMFs) and Geometric Means used in the Emission Factor Program for Electrostatic Precipitators - Cold Side (located after the air preheater, controlling coal-fired unit)

Unit Name	EPRI Site 11	EPRI Site 12	EPRI Site 15	EPRI Site 102	NSP - High Bridge 3, 4, 5, and 6	NSP - Black Dog 1, 3, and 4	NSP - Black Dog #2	NSP - Riverside #8	EPRI Site 114	EPRI Site 114, 2nd Test	DOE - Niles #2	DOE - Yates	DOE - Coal Creek	EPRI Site 16 w/OFA and LNOX Burners	EPRI Site 16 w/OFA	DOE - Cardinal	DOE - Baldwin	EMF (Geometric mean)	Geometric standard deviation
ANTIMONY					0.25		0.10	0.32			0.004	0.01	0.02	0.03	0.04	0.02	0.10	0.04	4.21
ARSENIC	0.01	0.004	0.02	0.06	0.01	0.01	0.01	0.01	0.05	0.05	0.03	0.04	0.001	0.06	0.05	0.003	0.01	0.02	3.07
BERYLLIUM	0.10	0.002	0.01	0.001	0.01	0.02	0.003	0.02	0.06	0.02	0.005	0.02	0.05	0.02	0.03	0.001	0.03	0.01	4.13
CADMIUM	0.03	0.06	0.41	0.11	0.11	0.04	0.99	0.13	0.10	0.02	0.03	0.04	0.18	0.17	0.05	0.03	0.05	0.08	2.93
CHROMIUM	0.03	0.01	0.01	0.06	0.07	0.01	0.04	0.07	0.05	0.03	0.01	0.01	0.005	0.02	0.04	0.01	0.05	0.02	2.50
COBALT	0.01	0.01	0.01	0.03							0.001	0.02	0.01	0.02	0.03	0.002	0.07	0.01	3.52
LEAD	0.04	0.03	0.01	0.02	0.02	0.05	0.66	0.04	0.06	0.06	0.003	0.02	0.002	0.03	0.08	0.01	0.06	0.03	4.03
MANGANESE	0.01	0.04	0.005	0.03	0.03	0.02	0.02	0.05	0.05	0.04	0.01	0.02	0.003	0.02	0.03	0.01	0.03	0.02	2.30
MERCURY	1.00	0.18	1.00	1.00	0.93	0.92	0.78	1.00	0.70	0.84	0.73	0.45	0.87	0.45	0.91	0.26	0.74	0.68	1.66
NICKEL	0.01	0.003	0.01	0.37	0.03	0.04	0.15	0.03	0.05	0.03	0.001	0.01	0.005	0.02	0.04	0.01	0.03	0.02	4.03
SELENIUM	0.04	0.24	0.65	0.004	0.02	0.03	0.02	0.01	1.00	1.00	0.97	0.62	0.13	1.00	0.27	0.71	1.00	0.17	6.36
HYDROGEN CHLORIDE																	Note 1	0.93	
HYDROGEN FLUORIDE																	Note 1	1.00	

Note 1 - These EMFs were developed from four specific emission tests that examined HCl and HF emissions through an ESP, a baghouse, an SDA/FF, and a FGD.

Appendix D - Discusion of the Methodology Used to Develop Nationwide Emission Totals

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D.1 INTRODUCTION

To estimate emissions of hazardous air pollutants (HAPs) from fossil-fuel-fired electric utility units (≥25 MWe), the EPA developed the emission factor program (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.

D.2 PROGRAM OPERATION

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

D.3 DATA SOURCES

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

The second data file is the emission modification factor (EMF) database. This database contains information from emissions tests conducted by the Electric Power Research Institute (EPRI), DOE, and the electric utility industry. The program first searches the input file for the type of fuel burned and the amount of fuel consumed per year in an individual unit. If the fuel type is coal, the EFP then looks for the coal's State of origin. Origin is important because the trace elements in coal are addressed by coal type (bituminous, subbituminous, and lignite) and State of origin in the U.S. Geological Survey (USGS) database, which analyzed core and channel samples (3,331 samples) of coal from the top 50 (1990 or later) economically feasible coal seams in the United States.

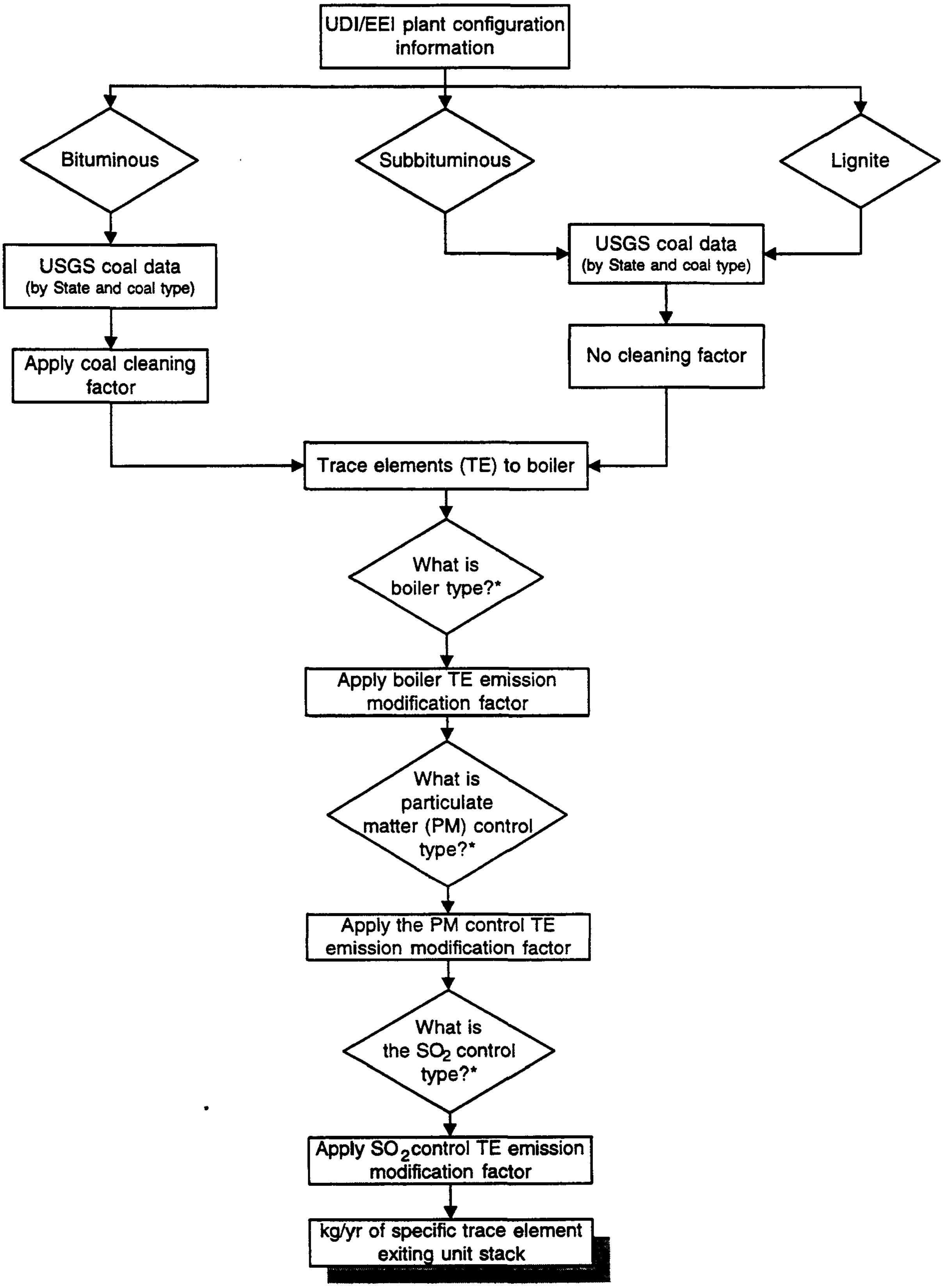


Figure D-1. Trace elements in coal.

*Taken from UDI/EEI data.

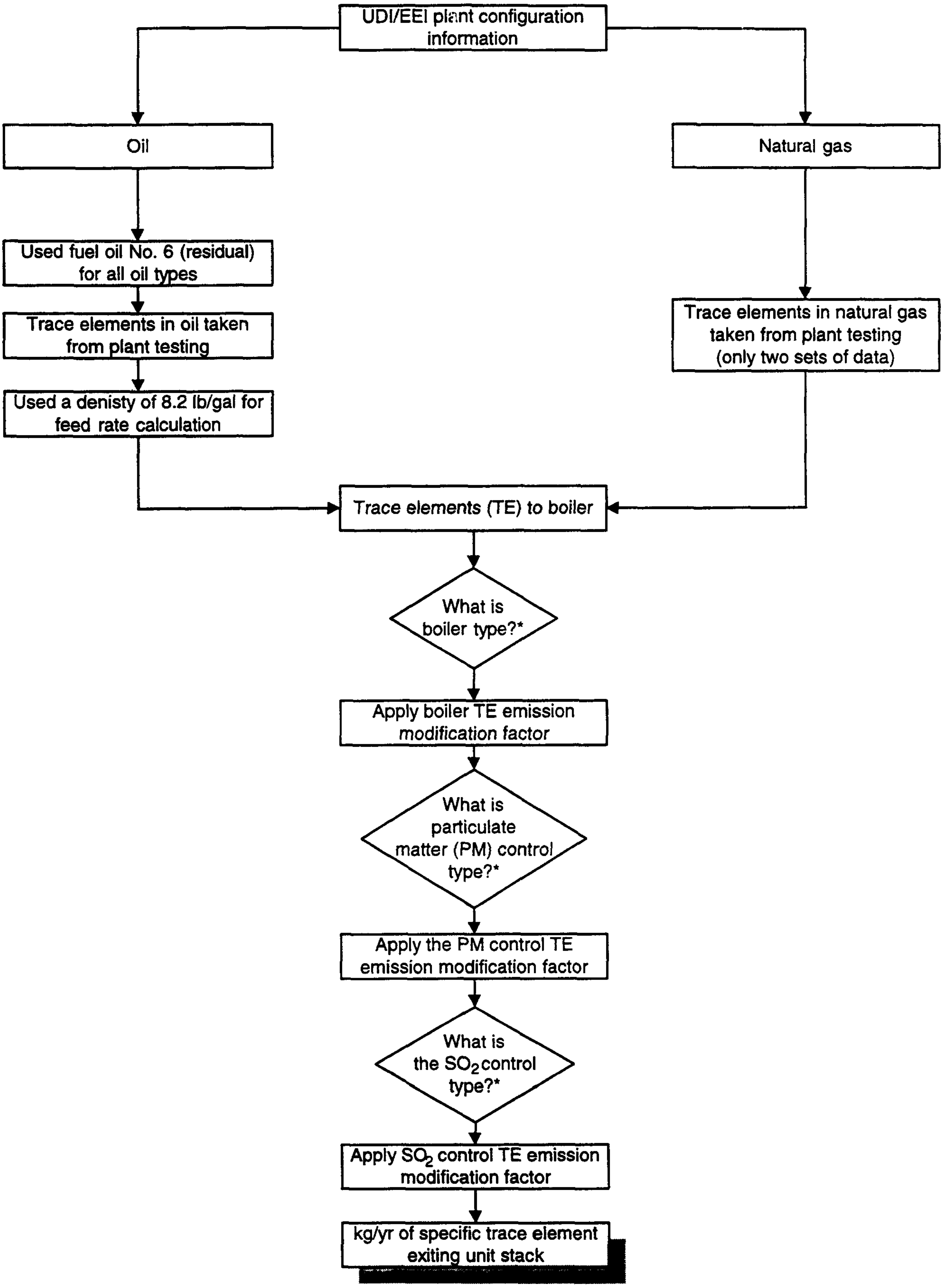
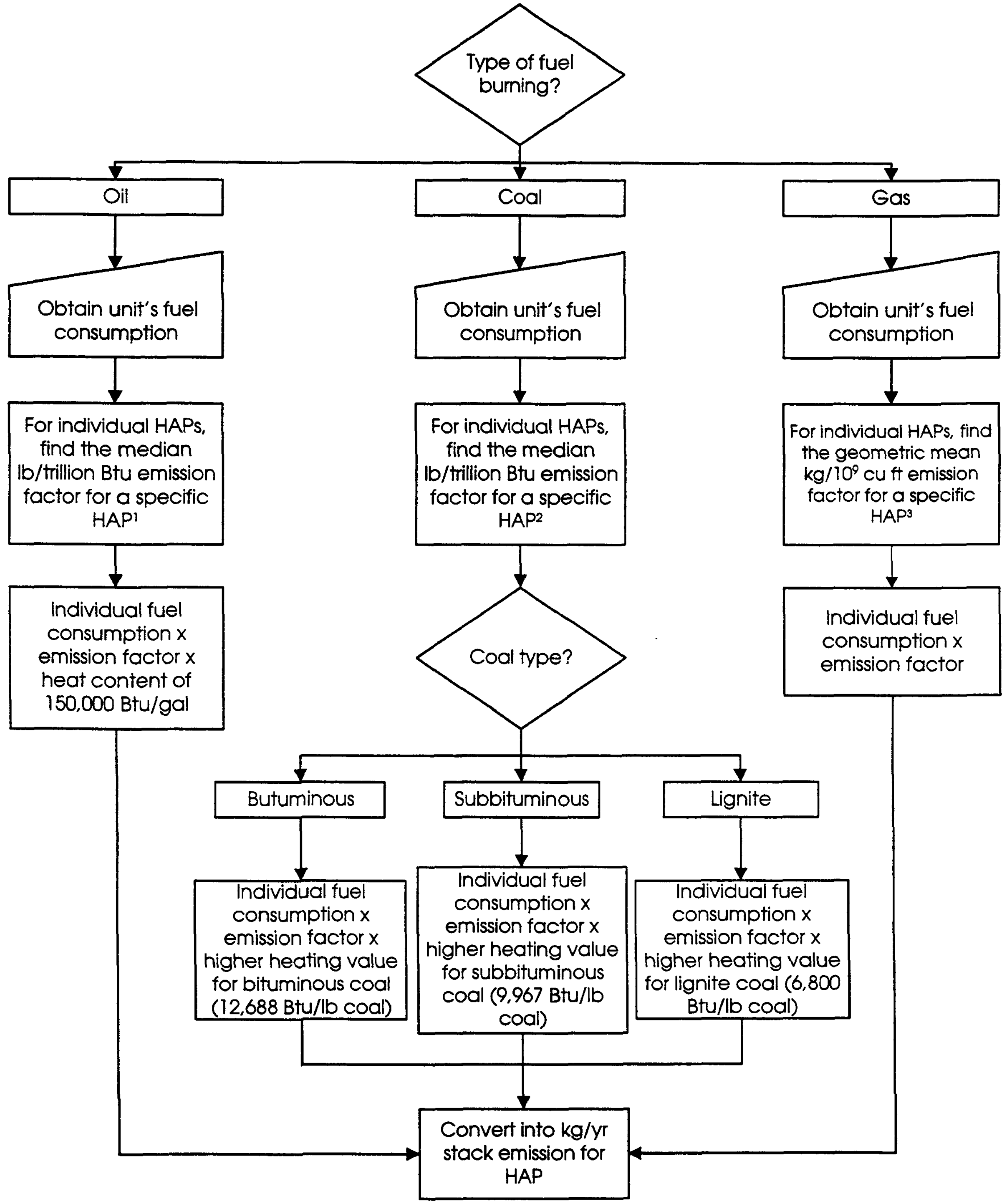


Figure D-2. Trace elements in oil and natural gas.

*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 D-3. Organic emissions.

D.4 OPERATIONAL STATUS OF BOILERS

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

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

Anthracite was disregarded as a fuel because of the limited number of units burning this type of coal.[1] Four units burning anthracite coal (in 1990) were assigned to burn bituminous coal for program computations.

Coal-fired boiler concentrations were modified for different heating values, depending on the type of coal, before being converted to a rate basis (kg/yr of individual HAP). This procedure was necessary because different coal ranks have different heating values. For example, it would require burning more lignite to achieve the same heat input to the boiler as burning bituminous coal. These values were determined as averages for each type of coal (see Table D-1).[2]

The EEI/UDI database had a number of gaps in the fuel consumption data. Some of these gaps were filled by data supplied voluntarily by the industry. To address the remaining gaps, RTI plotted the available data and fitted point-slope equations to estimate fuel consumption.[3] 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.

Capacity factors were taken from the UDI/EEI database for as many units as possible. If the above linear equation or (±5 MWe) estimating procedure were used, then the capacity factor for the

Table D-1. Average Higher Heating Values of Coal[4]

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. These excluded 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.

unit (with unknown fuel consumption) would fit an industry norm for that size unit and fuel type.

Limestone is used in circulating fluidized bed (CFB) combustors to control sulfur dioxide (SO_2). Early in the program's development, the EPA sought to address limestone's contribution to trace metal emissions. Based on the fact that limited trace metal data were available and that there were only 19 listed CFB units in the country in 1990, limestone's effect was disregarded.[5]

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.

D.5 BOILER CONFIGURATION

EPA received 51 emissions tests conducted by EPRI, DOE, and industry in time for inclusion in the EFP. Because of this limited sample, all boiler configurations, particulate, and SO_2 control types could not be sampled. To estimate the emissions from all units in the U.S., the substitution of unknown units into units with known EMFs was necessary. After studying the tested EMFs, the following patterns were observed. Coal-fired unit emissions seemed to be affected by whether the unit had a dry- or wet-bottom furnace. Oil-fired unit emissions seemed to be affected by whether or not the unit had nitrogen oxides NO_x control. Since only one type of gas-fired boiler was tested, all gas-fired units obtained their EMFs from this type of unit.[7]

One of the emission test reports that analyzed an oil-burning, tangentially fired (with NO_x control) unit contained information on two trace metals. Because this was the only unit of its kind to be tested, it was necessary to substitute the trace metal data of another similar unit (one having NO_x control) for which more data were collected. The EMFs of the oil burning, front-fired unit (with NO_x control) were averaged (by geometric mean) into the unit along with the two trace metal concentrations found in the tangentially fired boiler. Because there were organic HAP concentration numbers available for the tangentially-fired boiler, these numbers were maintained without modification.

No conventional emission testing (multimetals, volatile organic sampling train (VOST), semi-VOST) was done on combined-cycle gas turbines (CCYC). The Fourier Transform

Infrared (FTIR) system was used to test a CCYC unit for organic HAPs, but few HAPs were found. Combined-cycle gas turbines were categorized as conventional gas-fired units to address their emissions.

Testing by FTIR was also done on one example each of pulverized coal-, circulating fluidized bed-, oil-, and conventional gas-fired boilers and a combined-cycle gas turbine. However, the EPA decided not to use the data in developing estimated emissions.

Of the test reports received, three contained data that were not feasible for use in the EFP because the test contractors did not or could not test between the boiler and the particulate control device. The result was a test containing only a fuel analysis and stack emission numbers.

One EPRI emission test report (identified as EPRI Site 10) contained only one sample run instead of the normal three runs. Because only two emission test reports on CFBs (including Site 10) were available, the EPA decided to use these data.

Units were deemed dual-fuel-firing units if they fired more than 10 percent of at least one other fuel. Dual-fuel-firing emissions were modeled by splitting the dual-firing units (only oil- and gas-fired units) into two separate units with emissions exiting from the same stack. If the unit was listed as an oil-fired unit, its oil consumption rate and configuration were used to obtain its HAP emission rates for oil. The unit in question was then split into a gas-fired portion by using its gas consumption rate and changing its boiler type to the equivalent gas-fired type. This method was the most equitable way to represent dual-fuel-fired emissions, for both trace metals and organic HAPs created by either oil-fired or gas-fired boilers, respectively.

Substitution was also performed on particulate control and SO_2 control devices. Particulate control was addressed in one of six ways: electrostatic precipitator, cold-side (ESP,CS); electrostatic precipitator, hot-side (ESP,HS); electrostatic precipitator, cold-side, controlling an oil-fired unit (O-ESP,CS); fabric filter (FF); particulate scrubber; or no control.

For coal-fired units that had hot-side electrostatic precipitators (ESPs), the boiler was assigned as either HTANGNOX or HTANGNONOX depending on whether or not the unit had NO_x control. The HTANGNOX and HTANGNONOX boiler EMFs were determined from emissions testing data measured at a site that had a tangential boiler, a hot-side ESP, and a cold-side ESP; testing was conducted with and without NO_x control. Cold-side ESPs are placed after the air preheaters, while hot-side ESPs are placed before the air preheaters. The UDI/EEI database reported several units with combination HS/CS ESPs. These are units with separate ESPs before and after their air preheaters. Although one such unit was tested for HAP emissions, during the majority of its testing the cold-side ESP was turned off. Therefore, the data for this unit were used to develop hot-side ESP EMFs for the EFP. Because more data were available on ESP,CS devices, and because units controlled by HS/CS ESPs had a cold-side ESP as their last particulate matter (PM) control device, HS/CS ESPs were projected to behave like cold-side ESPs in terms of trace metal emissions. In assigning the boiler type for coal-fired units, when there was no information on whether the unit had NO_x control, it was assumed that the unit had no NO_x control and the unit was assigned HTANGNONOX boiler factors. The boiler and PM control device data were assigned in this manner for units that had hot-side ESPs since the temperature at the inlet to the hot-side ESP was approximately 700° F, whereas the temperature at the inlet to cold-side ESPs were typically around 300° F. The assignment was made to account for any effect that the approximately 700° F temperature might have on air toxic emissions.

Emission modification factors for particulate control by scrubbers were derived from data on controling trace elements by one venturi scrubber used for combined SO_2 and PM control. Particulate matter scrubbers use water only, while flue gas desulfiriztion units (FGDs) use water and a reagent (lime, limestone, etc). Although the presence of this reagent could cause the FGD to affect HAPs differently from the PM scrubbers, the EPA believes that the small number of PM scrubbers (<5) should not cause U.S. aggregate emissions to be adversely effected.

Mechanical collectors (multicyclones) are used either as precollection devices, before FFs or ESPs, or as primary collection devices for some oil-fired plants. No HAP emissions testing was done exclusively on mechanical collectors. Since mechanical collectors were projected to have little or no effect on reducing HAPs because of their ineffectiveness at removing small particles, units with only multicyclones were determined to have no control effect on HAPs in the program.

In the EFP, devices for controlling SO_2 emissions were classified as either WETSCRUB (containing all types of wet FGDs) or DRYSCRUB (containing all types of spray dryers/dry scrubbers). This substitution was necessary due to the lack of test data on a variety of wet FGD and dry scrubber types. Also, the EMFs include data from units tested with bypasses operating when using a bypass is normal operation.

D.6 STACK CHARACTERISTICS

Stack data in the UDI/EEI from some electric utility units were incomplete. Some of these gaps were due to the database reporting stack parameters from a shared stack on only one of the plant's units instead of reporting on both. The shared stack parameters were completed for these sister units. Next, an industry contractor made contact with a number of utility plants to retrieve missing stack data. This information was useful but still incomplete. The remaining gaps in stack parameter data were filled by either (1) finding a sister unit of the same configuration (and site, if possible) in order to duplicate its stack data, or (2) using the original EEI/UDI stack data to create a set of equations to estimate the relationships between stack height and gas flow, stack exit temperature, and exit velocity from stack diameter, respectively. These linear equations (point-slope) were specific to coal-, oil-, gas-, and combined cycle gas turbine-fired units. A spreadsheet procedure was developed to enter a stack height for a unit and use four separate equations to estimate the other parameters.

A few stack latitudes or longitudes not addressed in either the original EEI database or the contractor's research were found by calling the operators of the utility plants in question.

D.7 TRACE ELEMENT CONCENTRATION IN FUEL

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

Arithmetic averages of the concentrations of trace elements were determined from the USGS database by State of coal origin,[10] and the average concentrations were then used in the EFP. (Note: statewide data were not separated by coal region, and statewide averages were not weighted by coal production within the State.) Two sets of concentration data exist for coal that originated from Arizona and one set for coal that originated from Washington.[11] The two sets of Arizona data were averaged with data for Colorado, Utah, and New Mexico coal. The trace element concentrations for coals from Arizona, Louisiana, and Washington were needed for, respectively, five, one, and two utility units. Because no data were available for coal from Louisiana, data from Texas lignite coal were used to represent the concentration of trace elements in Louisiana coal.[12]

Additional data on the concentrations of the trace elements in utility coal shipments were received from ARCO Coal Company on 145 samples of Wyoming Coal and on 30 samples of bituminous Colorado coal,[13] and from the Illinois State Geological Survey (ISGS) on 34 samples of Illinois cleaned coal.[14] Arithmetic averages of the trace element concentrations provided by ARCO Coal Company and ISGS were converted to an as-received basis and used directly, without application of cleaning factors, in the EFP.[15]

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

If the fuel type was oil, the program accessed a database containing the arithmetic average of trace element concentrations in residual oil (see Figure D-2). Each concentration data point was the arithmetic average of repeated measurements, and at least one of the repeated measurements had to be a detected concentration (see discussion of nondetected data in section D.10). Because trace element data were available only on residual oil-fired units, and since 95 percent of the oil-fired units burn residual oil, all units were assumed to burn residual oil. Although densities of residual oils vary, an average density of 8.2 lb/gal was chosen for the feed rate calculation for oil. The concentration data and density were used, as shown in Equation 3 in Table D-2, to calculate a kilogram/year rate of each trace element entering the unit's oil-fired boiler. Oil-fired organic HAP exit concentration calculations included a 150,000-Btu/gallon heating value for oil.

An emission rate for each organic HAP emitted from gas-fired units was extracted from the test reports. Only two test reports analyzed organic HAPs, and a geometric mean emission rate of each

Table D-2. Computer Calculations (Inorganic HAPs)

Feed rate of trace elements to the boiler

For bituminous coal:

Equation 1

$$\text{Unit coal consumption } \frac{1{,}000 \textit{ tons coal}}{\textit{yr}} \times \frac{2{,}000 \textit{ lb}}{\textit{ton}} \times \frac{0.454 \textit{ kg}}{\textit{lb}} \times \frac{\textit{trace element ppmw (by State of coal origin)}}{1{,}000{,}000}$$

$$\times \textit{ coal cleaning factor} = \frac{\textit{kg trace element}}{\textit{yr}}$$

For lignite and subbituminous coal:

Equation 2

$$\text{Unit coal consumption } \frac{1{,}000 \textit{ tons coal}}{\textit{yr}} \times \frac{2{,}000 \textit{ lb}}{\textit{ton}} \times \frac{0.454 \textit{ kg}}{\textit{lb}} \times \frac{\textit{trace element ppmw (by State of coal origin)}}{1{,}000{,}000} = \frac{\textit{kg trace element}}{\textit{yr}}$$

For oil:

Equation 3

$$\text{Unit oil consumption } \frac{1{,}000 \textit{ bbls oil}}{\textit{yr}} \times \frac{42 \textit{ gal}}{\textit{bbl}} \times \frac{8.2 \textit{ lb}}{\textit{gal}} \times \frac{0.454 \textit{ Kg}}{\textit{lb}} \times \frac{\textit{trace element ppmw}}{1{,}000{,}000} = \frac{\textit{kg trace element}}{\textit{yr}}$$

For natural gas:

Equation 4

$$\text{Unit gas consumption } \frac{1{,}000{,}000 \textit{ cf}}{\textit{yr}} \times \frac{\textit{trace element ppmw}}{1{,}000{,}000} \times \frac{1 \textit{ atm} \times \textit{lb mole} \times {}^\circ R}{0.7302 \textit{ cf} \times \textit{atm}} \times \frac{1}{(460 + 68)^\circ R} \times \frac{\textit{trace element molecular weight (lb)}}{\textit{lb mole}}$$

$$\times \frac{0.454 \textit{ kg}}{\textit{lb}} = \frac{\textit{kg trace element}}{\textit{yr}}$$

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 (see Equation 4 in Table D-2). This result was equivalent to a stack emission because there were no PM control or SO_2 control devices on gas-fired units. The geometric mean of the concentrations were averaged and used in the gas-fired boiler calculations (see Figure D-2). The few trace elements found in the gas database were estimated by this procedure. Fuel gas density was assumed to follow the ideal gas law.

D.8 HYDROGEN CHLORIDE (HCL) AND HYDROGEN FLUORIDE (HF) CONCENTRATION IN FUEL

To obtain HCl or 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.[16] For example, for each part per million of chloride in the feed coal at one of the test sites, 0.61 lb/hr of HCl was found in the gas stream leaving the boiler and 0.00145 lb/h in the stack gas. Similarly for HF, the boiler emissions were 0.56 lb/h for each part per million of fluoride in the coal and 0.00448 lb/h in the stack. For ease of programming, the HCl and HF emissions were addressed starting with the fuel. This programming was done by multiplying the chloride and fluoride concentrations in the fuel constituents by 0.61 or 0.56, respectively. The resulting numbers allowed direct conversion into boiler emissions that could be further modified for systems with PM control or SO_2 control.

The chloride concentrations were not available for coals from the following States: Alaska, Illinois, Indiana, Iowa, Missouri, Utah, and Washington. Chloride concentrations were assigned, as shown in Table D-3, for coals originating from these States.[17]

D.9 EMISSION MODIFICATION FACTORS FOR INORGANIC HAPS

To address the partitioning of the HAP stream through the combustion and pollution control process, partitioning factors known as emission modification factors (EMFs) were developed from inorganic HAP testing data. The EMFs are fractions of the amount of a HAP compound exiting a device (boiler or air pollution control device [APCD]) divided by the amount of the same HAP compound entering that device.[18] These EMFs were averaged by taking the geometric mean of similar devices (e.g., all oil-fired tangential boilers, all cold-side ESPs). Geometric means were

Table D-3. Assigned Chloride ppmw and HCl ppmw Concentrations in Coal, by State of Coal Origin [17]

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

used because of the presence of outlying data points, the small amount of data, and the general fit of the data to a log-normal curve. These geometric means were then applied to the kg/yr feed rates entering the boiler, the effect of which either reduced or left unchanged the emissions that passed through them. Those EMFs calculated as being greater than 1.0 (i.e., more material exiting a device than entering it) are set to equal 1.0. The EMFs are based on emission test report data collected and analyzed after 1990.

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

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

Table C-1 (Appendix C) shows two sets of EMF data for the DOE Niles test site. One unit with NO_x control is in a section designated without NO_x control. This apparent anomaly occurred because the NO_x control method used, SCR, is a postcombustion NO_x control and does not effect the boiler EMFs. The data are labeled this way to identify the data obtained from a separate test report.

The HCl and HF emission factors were addressed in the fuel, therefore, all HCl and HF boiler EMFs for all fuel types, were made equal to 1 in the EFP.

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

D.10 ORGANIC AND MINERAL ACID HAPS

Organic and mineral acid HAP emissions were handled in one of two ways. The first method was used only with HCl or HF emissions and was described in section D.8. The numbers resulting from this method allowed direct conversion into boiler emissions that could be further modified for systems with PM control or SO_2 control.

Hydrochloric acid and HF EMFs for PM and SO_2 control devices were developed with data from four test reports where contractors conducted tests individually for HCl and chlorine as well as HF and fluorine, before and after each control device. The rest of the available reports showed tests only for chlorine and fluorine and estimated the fractions that were HCl or HF. In developing the HCl and HF EMFs for wet FGDs and dry scrubbers, the EPA decided to address the effect of flue gas bypass. After analyzing test data and having discussions with industry representatives, EPA decided to assume an industry average flue gas bypass of 17 percent for wet FGDs and 14 percent for dry scrubber systems. This assumption was used only in developing HCl and HF EMFs.[19] Because each of the four test sites was different than the others regarding SO_2 and PM control, the emission factors for chlorine and fluorine were maintained separately for the four system types rather than averaging them.

The second method of handling organic and mineral acid HAPs was for organics. Because they 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. Table D-4 gives the equations used to estimate organic HAP emissions from coal-, oil-, and gas-fired boilers.

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

D.11 TREATMENT OF NONDETECTED DATA IN THE DEVELOPMENT OF EMFS

In the raw data taken from the test reports, the EPA used a protocol to analyze detected and nondetected compounds in the test samples. The protocol is as follows:

- When all values for a specific compound are above the detection limit, the mean arithmetic concentration is calculated using the reported quantities.

- For results that include values both above and below the detection limit (with the detection limit shown in parentheses), one half of the detection limit is used for values below the detection limit to calculate the mean. For example:

Analytical values	Calculation	Mean value
10,12,ND(8)	(10+12+[8/2])/3	8.7 ND

The calculated mean cannot be smaller than the largest detection limit value. In the following example, the calculated mean is 2.8. This quantity is less than the largest detection limit, so the reported mean becomes ND(4).

Analytical values	Calculation	Mean value
5,ND(4),ND(3)	(5+[4/2]+[3/2])/3	ND(4)

- When all sample results are less than the detection limit, the data are not used.

D.12 MODEL CHANGES FOR ESTIMATES IN THE YEAR 2010

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

To approximate the projected increase in the use of coal, and particularly lower sulfur coals, the 2010 coal consumption was determined as follows. First the estimated overall increase in electric utility coal consumption was determined (37 percent).[20] Then, instead of using an overall percentage increase for each coal-fired unit, a factor was derived for each coal State of origin to represent the expected increase or decrease in consumption for that State's coal in 2010. The 1990 coal consumption was then multiplied by the 2010 factor, listed in Table D-5, that corresponded to the State of coal origin assigned to each unit.[21]

Table D-4. Computer Calculations (Organic HAPs)

All non-trace-element HAPs emitted from the stack

Coal-fired boiler:

Equation 1

$$\text{Unit coal consumption } \frac{1{,}000 \text{ tons coal}}{\text{yr}} \times \frac{2{,}000 \text{ lb coal}}{\text{ton coal}} \times \text{Coal HHV } \frac{\text{Btu}}{\text{lb coal}} \text{ (see Table D-4)} \times \left(\frac{1}{10^{12}}\right) \frac{\text{trillion Btu}}{\text{Btu}}$$

$$\times \text{ (Median emission factor) } \frac{\text{lb HAP}}{\text{trillion Btu}} \times \frac{0.454 \text{ kg HAP}}{\text{lb HAP}} = \frac{\text{kg HAP}}{\text{yr}}$$

Oil-fired boiler:

Equation 2

$$\text{Unit oil consumption } \frac{1{,}000 \text{ bbls oil}}{\text{yr}} \times \frac{42 \text{ gal oil}}{\text{bbl oil}} \times \text{ (HHV for residual oil) } \frac{150{,}000 \text{ Btu}}{\text{gal oil}} \times \left(\frac{1}{10^{12}}\right) \frac{\text{trillion Btu}}{\text{Btu}}$$

$$\times \text{ (Median emission factor) } \frac{\text{lb HAP}}{\text{trillion Btu}} \times \frac{0.454 \text{ kg HAP}}{\text{lb HAP}} = \frac{\text{kg HAP}}{\text{yr}}$$

Gas-fired boiler:

Equation 3

$$\text{Unit gas consumption } \frac{1{,}000{,}000 \text{ cf gas}}{\text{yr}} \times \left(\frac{1}{10^{9}}\right) \text{ billion cf} \times \text{ (Geometric mean emission factor) } \frac{\text{kg HAP}}{\text{billion cf}} = \frac{\text{kg HAP}}{\text{yr}}$$

Table D-5. Coal consumption scaling factors for 2010

State of coal origin	2010 factor[a]
Kentucky	1.27
Pennsylvania	1.23
West Virginia	1.24
Maryland	0.872
Ohio	0.872
Alabama	1.41
Louisiana	1.41
Texas	1.41
Virginia	1.41
Illinois	1
Indiana	1
Iowa	1
Kansas	1
Missouri	1
Oklahoma	1
Alaska	1.599
Arizona	1.599
Colorado	1.599
Montana	1.599
New Mexico	1.599
North Dakota	1.599
Utah	1.599
Washington	1.599
Wyoming	1.599

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[a] For each coal-fired unit, the 2010 coal consumption was determined as follows: The 1990 coal consumption was multiplied by the 2010 factor that corresponded to the State of coal origin assigned to the unit.

D.15 REFERENCES

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12. Ref. 11.

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15. Memorandum from Heath, E., RTI to Maxwell, W.H., EPA. April 29, 1994. As-shipped coal data and how gaps in the USGS and UDI database were filled for the computer emission program.

16. Memorandum from Turner, J.H., RTI, to Cole, J.D., RTI. April 19, 1994. Methodology for determining HCl and HF concentrations fom utility boilers.

17. Memorandum from Heath, E., RTI to Maxwell, W.H., EPA. May 27, 1994. USGS data gaps in chloride concentrations for seven states.

18. Memorandum from Cole, J.D., RTI, to Maxwell, W.H., EPA. March 31, 1994. Emission factor memorandum.

19. Memorandum from Cole, J.D., RTI to Maxwell, W.H., EPA. May 9, 1994. Emission modification factors for HCl and HF including FGD system bypass.

20. Economic Analysis of the Title IV Requirements of the 1990 Clean Air Act Amendments. Prepared for the U.S. EPA, Office of Air and Radiation, Acid Rain Division. Prepared by ICF Resources Incorporated. February 1994.

21. Memorandum from Heath, E., RTI to Maxwell, W.H., EPA. May 9, 1994. Proposed method to account for utilities switching to lower sulfur coals in 2010.

Appendix E - Health Effects Summaries: Overview

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This section contains summaries of health effects data for seven hazardous air pollutant (HAPs) emitted from utilities (i.e., arsenic, chromium, nickel, mercury, hydrogen chloride, hydrogen fluoride, and dioxins). Radionuclides are discussed in Chapter 9 of the interim report. All of the numbers presented in these summaries are subject to change, if EPA obtains new data in the future indicating that the risk is higher or lower than that currently being considered. For more information on health effects, readers can refer to the referenced sources at the end of Appendix E. Also, health effects information for these HAPs and other HAPs can be obtained from the EPA's Integrated Risk Information System[1] or from an EPA document titled Health Effects Notebook for Hazardous Air Pollutants.[2] Each summary, except the one for mercury, contains the following sections:

E.1 INTRODUCTION
E.2 CANCER EFFECTS
E.3 NONCANCER EFFECTS
    E.3.1 Acute (Short-Term)
    E.3.2 Chronic (Long-Term)
    E.3.3 Reproductive and Developmental

The following is a discussion of the information contained in each of these sections:

E.1 INTRODUCTION

This section presents a brief overview of the chemical, with information on its chemistry, physical properties, and major uses. If available, EPA's National Ambient Air Quality Standard (NAAQS) and/or Maximum Contaminant Level Goal (MCLG) or Maximum Contaminant Level (MCL) are also presented in this section. EPA's NAAQS are legally enforceable air standards set under the Clean Air Act Amendments of 1990; these are health-based standards with considerations such as economics and technical feasibility factored in. EPA's MCLGs are nonenforceable health goals that are set at levels at which no known or anticipated adverse health effects occur and that allow an adequate margin of safety. Maximum contaminant levels are legally enforceable drinking water standards which are set as close to the MCLGs as feasible.

E.2 CANCER EFFECTS

The results of available cancer studies in animals and/or humans are presented in this section. In addition, the EPA's cancer weight-of-evidence classification system is included. 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 of the studies. The animal studies are evaluated to decide whether biologically significant responses have occurred and whether the responses are statistically significant increases in treated versus control animals. Secondly, the human and animal evidence is combined into an overall classification. This classification is based on an analysis of both the human and animal evidence, considering the number and quality of both types of studies. 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. The result is that each chemical is placed into one of the following six categories:

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

This section also includes information on the inhalation cancer risk and oral unit cancer risk. If EPA has calculated both inhalation and oral unit cancer risk values, then this section is divided into two subsections.

The inhalation unit risk estimate (IURE) for the chemical is the estimated increased probability of a person's developing cancer from breathing air containing a concentration of 1 microgram pollutant per cubic meter ($\mu g/m^3$) of air for 70 years. The IURE is derived using mathematical models that assume a nonthreshold approach: i.e., there is some risk of cancer occurring at any level of exposure. The methods used to derive these values typically result in an "upper bound" estimate; i.e., the true risk is unlikely to exceed this value and may be lower. However, some unit risk estimates are not "upper bound" estimates but rather are based on a "maximum likelihood" estimate (e.g., arsenic).

The risk-specific dose, which is an estimate of the dose corresponding to a specified level of cancer risk, is also included. This section presents risk-specific doses corresponding to a one-in-a-million and one-in-a-hundred-thousand excess risk attributed to exposure to the chemical. This means that EPA has estimated that if an individual were to breathe air containing these concentrations of the chemical, over his or her lifetime, that person would theoretically have no more than a one-in-a-million or one-in-a-hundred-thousand increased chance of developing cancer as a direct result of breathing air containing the chemical.

If available, the oral unit cancer risk is also presented. Both the oral cancer risk and the corresponding risk-specific dose are developed for an exposure of 70 years to the chemical through the drinking water. The oral unit risk estimate (OURE) is the estimated increased risk of cancer for drinking for 70 years 2 liters/day of water that contains a concentration of 1 μg of pollutant per liter. It is expressed in units of μg/L.

E.3 NONCANCER EFFECTS

E.3.1 Acute (Short-Term)

Results from acute animal tests or acute human studies are presented in this section. Acute animal studies usually report an estimated median lethal dose (LD_{50}) or median lethal concentration (LC_{50}). This is the dose (or concentration) estimated to kill 50 percent of the experimental animals. Results from these tests are divided into the following toxicity categories:

Lethality	**Extreme**	**High**	**Moderate**	**Low**
Oral LD_{50}	< 50 mg/kg	50 to 500 mg/kg	500 to 5,000 mg/kg	>5,000 mg/kg
Dermal LD_{50}	<200 mg/kg	200 to 2,000 mg/kg	2,000 to 20,000 mg/kg	>20,000 mg/kg
Inhalation LC_{50}	<200 mg/m^3	200 to 2,000 mg/m^3	2,000 to 20,000 mg/m^3	>20,000 mg/m^3

Source: U.S. EPA. Office of Pesticide Programs, Registration and Classification Procedures, Part II. Federal Register. 40:28279.

Acute human studies usually consist of case reports from accidental poisonings. These case reports often help to define the levels at which acute toxic effects are seen in humans.

E.3.2 Chronic (Long-Term)

This section summarizes the major chronic noncarcinogenic effects seen from exposure to the chemical. Chronic animal studies usually range from 90 days to 2 years. Human studies investigating effects ranging from exposure of a few years to a lifetime are also included. In addition, subchronic studies may be included in this section. Subchronic studies are usually animal studies of several weeks to 90 days.

The Inhalation Reference Concentration (RfC) is presented in this section. The RfC is an estimate (with uncertainty spanning perhaps an order of magnitude) of the daily exposure of a chemical to the human population by inhalation (including sensitive subpopulations) that is likely to be without deleterious effects during a lifetime of exposure. The RfC is derived based on the assumption that thresholds exist for noncancer effects; i.e., there is a level below which no toxic effects would occur. 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 data base.

The RfC is not a direct or absolute estimator of risk, but rather a reference point to gauge the potential effects. Doses at or below the RfC are not likely to be associated with any adverse health effects. However, exceedance of the RfC does not imply that an adverse health effect would necessarily occur. As the amount and frequency of exposures exceeding the RfC increases, the probability that adverse effects may be observed in the human population also increases. The RfC is expressed in milligrams of pollutant per cubic meter of air (mg/m^3). If available, the Oral Reference Dose (RfD) is also presented in this section. The RfD is the oral equivalent of the RfC.

EPA's confidence in the RfC and/or RfD is also presented in this section. EPA ranks each RfC and RfD as low, medium, or high in three areas: (1) confidence in the study on which the RfC or RfD was based; (2) confidence in the data base; (3) overall confidence in the RfC or RfD. All three rankings are presented in this section.

E.3.3 Reproductive and Developmental

This section presents the results of reproductive and developmental studies on the effects of the chemical in animals and humans. Reproductive effects are those effects that adversely affect the female or the male reproductive system. Examples in the female include reduced fertility, a decrease in the survival of offspring, and alterations in the reproductive cycle. Male reproductive effects include a decrease in sperm count or an increase in abnormal sperm morphology. Developmental effects are adverse effects on the developing organism that result from exposure prior to conception (either parent), during prenatal development, or postnatally to the time of sexual maturation. Examples include altered growth, death of the developing organism, and malformations or birth defects. Reproductive and developmental effects may be observed after short-term or long-term exposure to the chemical, as some effects can be attributed to one time or short-term exposures during a critical biological cycle.

E.4 ARSENIC HEALTH EFFECTS SUMMARY

Arsenic is a naturally occurring element in the earth's crust that is usually found combined with other elements. Arsenic combined with elements such as oxygen, chlorine, and sulfur is referred to as inorganic arsenic; arsenic combined with carbon and hydrogen is referred to as organic arsenic. In this health effects summary, arsenic refers to inorganic arsenic and its associated compounds. Organic arsenic compounds, such as arsine gas, are not discussed. EPA has set a Maximum Contaminant Level (MCL) of 0.05 mg/L for inorganic arsenic.[3]

E.4.1 Cancer Effects - Arsenic

There is clear evidence that chronic exposure to inorganic arsenic in humans increases the risk of cancer. Studies 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 nonmelanoma skin cancer and bladder, liver, and lung cancer. No information is available on the risk of cancer in humans from dermal exposure to arsenic. 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.[4]

EPA has classified inorganic arsenic in Group A - Known Human Carcinogen. For arsenic, 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.[5]

E.4.1.1 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. All five studies showed excess risks of lung cancer that were related to the intensity and duration of exposure and the duration of the latency period. The estimates of unit risk obtained from the five studies were in reasonably good agreement, ranging from 1.25×10^{-3} to 7.6×10^{-3} $(\mu g/m^3)^{-1}$. Using the geometric mean of these data, EPA calculated an inhalation unit risk estimate of 4.29×10^{-3} $(\mu g/m^3)^{-1}$ (EPA).[6] Based on this unit risk estimate, EPA estimates that if an individual were to breathe air containing arsenic at 0.0002 $\mu g/m^3$[a] over his or her entire lifetime (70 years), that person would theoretically have an increased chance of one in a million of developing cancer as a direct result of breathing air containing this chemical. Similarly, EPA estimates that breathing air containing 0.002 $\mu g/m^3$ would result in an increased chance of up to one in a hundred thousand of developing cancer. EPA has high confidence in the arsenic cancer unit risk estimate for inhalation exposure 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.[7]

The Electric Power Research Institute (EPRI) has proposed a revision to EPA's IURE for inorganic arsenic. EPRI used standard EPA risk assessment methodology to recalculate the estimated risk. They calculated a new unit risk of 1.43×10^{-3} $(\mu g/m^3)^{-1}$, which is one-third the value on IRIS presented above. EPRI's risk estimate is based on updated exposure data from an epidemiology study of workers at a smelter in Tacoma, Washington, which indicated that the workers were much more highly exposed than previously thought. EPRI also used results from a recent Swedish smelter study.[8]

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[a] 0.0002 $\mu g/m^3$ (concentration corresponding to a 10^{-6} risk level) $=10^{-6}$ (risk level)/4.29×10^{-3} $(\mu g/m^3)^{-1}$ (unit risk estimate).

E.4.1.2 Oral Cancer Risk for Arsenic. To estimate the risks posed by ingesting arsenic, EPA obtained 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 artisan 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[9], they did propose that a unit risk estimate of 5×10^{-5} $(\mu g/L)^{-1}$ from oral exposure to arsenic in drinking water be used.[10]

The Taiwan cancer data have the following 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 U.S. citizens, EPA has stated "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."[11]

E.4.2 Noncancer Effects - Arsenic

E.4.2.1 Acute (Short-Term) Effects for Arsenic. Arsenic has been recognized as a human poison since ancient times, and large doses, approximately 600 µg/kg/day or higher, taken orally have resulted in death. Oral exposure to lower levels of arsenic has resulted in effects on the gastrointestinal system (nausea, vomiting); central nervous system (headaches, weakness, delirium); cardiovascular system (hypotension, shock); and the liver, kidney, and blood (anemia, leukopenia). Acute arsenic poisoning of humans, through inhalation exposure, has resulted in similar effects, including effects on the gastrointestinal system (nausea, diarrhea, abdominal pain), blood, and central and peripheral nervous system. The only effect noted from dermal (skin) exposure to arsenic in humans is contact dermatitis, with symptoms such as erythma and swelling. This effect has been noted only at high arsenic levels.[12]

Because significant information is available on the acute effects of arsenic poisoning in humans, few animal studies have been carried out. The limited available data have shown arsenic to have moderate to high acute toxicity to animals by the oral route. This is based on data showing the LD_{50} for arsenic to range between 50 and 5,000 mg/kg.[13]

E.4.2.2 Chronic (Long-Term) Effects for Arsenic. The primary effect noted in humans from chronic exposure to arsenic, through both inhalation and oral exposure, is 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. No information is available on effects in humans from chronic low-level dermal exposure to arsenic.[14]

No studies are available on the chronic noncancer effects of arsenic in animals, from inhalation or dermal exposure. Oral animal studies have noted effects on the kidney and liver.[15]

EPA has established an RfD (Reference Dose) 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.[16] This RfD was based on two studies[17] that showed that the prevalence of blackfoot disease increased with both age and dose for individuals exposed to high levels of arsenic in drinking water. This same population also displayed a greater incidence of hyperpigmentation and skin lesions. Other human studies support these findings, with several studies noting an increase in skin lesions from chronic exposure to arsenic through the drinking water. The EPA has not established a RfC for inorganic arsenic.[18]

EPA has medium confidence in the studies on which the RfD was based and in the RfD. The key studies were extensive epidemiologic reports that examined effects in a large number of people. However, doses were not well characterized, other contaminants were present, and potential exposure from food or other sources was not examined. The supporting studies suffer from other limitations, primarily the small populations studied. However, the general database on arsenic does support the findings in the key studies; this was the basis for EPA's "medium confidence" ranking of the RfD.[19]

E.4.2.3 Reproductive and Developmental. Limited information is available on the reproductive or developmental effects of arsenic in humans. The only available information consists of several studies that suggest that women who work in, or live near, metal smelters may have higher than normal spontaneous abortion rates, and their children may exhibit lower than normal birth weights. However, these studies are limited and contain significant uncertainties because they were designed to evaluate the effects of smelter pollutants in general and are not specific for arsenic.[20]

Animal studies on arsenic exposure via oral and inhalation routes have reported that arsenic at very high doses may cause death to the fetus or birth defects. No information is available on reproductive or developmental effects of arsenic in animals from dermal exposure.[21]

E.5 CHROMIUM HEALTH EFFECTS SUMMARY

Chromium is a metallic element that occurs in the environment in two major valence states: trivalent chromium (chromium III) and hexavalent chromium (chromium VI). Chromium VI compounds are much more toxic than chromium III compounds; chromium III is an essential element in humans, with a daily intake of 50 to 200 micrograms per day recommended for an adult, while chromium VI is quite toxic. However, the human body can detoxify some amount of chromium VI to chromium III. EPA has set a Maximum Contaminant Level (MCL) of 0.1 mg/L for total chromium.[22]

E.5.1 Cancer Effects for Chromium

Epidemiological studies of workers have clearly established that inhaled chromium is a human carcinogen, resulting in an increased risk of lung cancer. These studies were not able to differentiate between exposure to chromium III and chromium VI compounds. No information is available on cancer in humans from oral or dermal exposure to chromium.[23,24]

Animal studies have shown chromium VI to cause lung tumors via inhalation exposure. No studies are available that investigated cancer in animals from oral or dermal exposure to chromium VI. Chromium III has been tested in mice and rats by the oral route, with several studies reporting no increase in tumor incidence. No studies are available on cancer in animals from inhalation or dermal exposure to chromium III.[25,26]

EPA has classified chromium VI in Group A - Known Human Carcinogen.[27] Since the human studies could not differentiate between chromium III and chromium VI exposure, and only chromium VI was found to be carcinogenic in animal studies, EPA concluded that only chromium VI should be classified as a human carcinogen.[28] EPA has classified chromium III in Group D – Not Classifiable as to Human Carcinogenicity.[29]

EPA used the multistage extrapolation model, based on data from an occupational study of chromate production workers, to estimate the unit cancer risk for chromium VI. EPA calculated an IURE of 1.2×10^{-2} $(\mu g/m^3)^{-1}$.[30] Based upon this unit risk estimate, EPA estimates that if an individual were to breathe air containing chromium VI at 0.00008 $\mu g/m^3$ [b] over his or her entire lifetime, that person would theoretically have an increased chance of up to a one in one million of developing cancer as a direct result of breathing air containing this chemical. Similarly, EPA estimates that breathing air containing 0.0008 $\mu g/m^3$ would result in an increased chance of up to one in one hundred thousand of developing cancer.[31] EPA has not calculated a risk estimate from oral exposure to chromium VI[32] or from inhalation or oral exposure to chromium III.[33]

EPA has confidence in the risk estimate for chromium VI, based on the fact that the results of studies of chromium exposure are consistent across investigators and countries and because a dose response for lung tumors has been established. However, an overestimation of risk may exist due to the implicit assumption that the smoking habits of chromate workers were similar to those of the general white male population, because it is generally accepted that the proportion of smokers is higher for industrial workers than for the general population.[34]

The International Agency for Research on Cancer (IARC) has stated that there is sufficient evidence in humans for the carcinogenicity of chromium VI compounds and inadequate evidence in humans for the carcinogenicity of chromium III compounds.[35]

E.5.2 Noncancer Effects

This section presents information from human and/or animal studies on the acute (short-term), chronic (long-term), and reproductive/developmental effects of chromium VI and chromium III.

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[b] 0.00008 $\mu g/m^3$ (concentration corresponding to a 10^{-6} risk level) = 10^{-6} (risk level)/1.2×10^{-2} $(\mu g/m^3)^{-1}$ (unit risk estimate).

E.5.2.1 Acute (Short-Term) for Chromium. The respiratory tract is the major target organ for chromium VI following inhalation exposure in humans. Dyspnea, coughing, and wheezing were reported in cases in which individual inhaled very high concentrations of chromium VI. Other effects noted from acute inhalation and oral exposure to very high concentrations of chromium VI include gastrointestinal and neurological effects, while dermal exposure causes skin burns.[36]

Acute animal studies have reported chromium VI to have extreme toxicity from inhalation and oral exposure. This is based on data showing the LC_{50} for chromium VI to be less than 200 mg/m^3 and the LD_{50} to be less than 50 mg/kg. Chromium III has been shown to have moderate toxicity from oral exposure, based on LD_{50} data in the range of 500 to 5,000 mg/kg. The kidney is the major target organ for chromium VI acute toxicity in animals, with high doses resulting in kidney failure. Other target organs include the brain and the liver.[37]

E.5.2.2 Chronic (Long-Term) for Chromium. Chronic inhalation exposure to chromium VI in humans results in effects on the respiratory tract, with perforations and ulcerations of the septum, bronchitis, decreased pulmonary function, pneumonia, asthma, and nasal itching and soreness reported. Chronic exposure to high levels of chromium VI by inhalation or oral exposure may also produce effects on the liver, kidney, gastrointestinal and immune systems, and possibly the blood. Dermal exposure to chromium VI may cause contact dermatitis, sensitivity, and ulceration of the skin.[38]

Limited information is available on the chronic effects of chromium in animals. The available data indicate that, following inhalation exposure, the lung and kidney have the highest tissue levels of chromium. No effects were noted in several oral animal studies with chromium VI and chromium III.[39]

EPA has established RfD for chromium VI of 0.005 mg/kg/day, based upon a NOAEL (adjusted) of 2.4 mg/kg/day, an uncertainty factor of 500, and a modifying factor of 1.[40] This was based on a study of rats, which reported no adverse effects after their exposure to chromium VI in the drinking water for 1 year. Other studies support these findings; one study reported no significant effects in female dogs given chromium VI in the drinking water for 4 years, and a case study on humans reported no adverse health effects in a family of four who drank water for 3 years from a private well containing chromium VI at 1 mg/L.[41]

EPA has low confidence in the study on which the RfD for chromium (VI) was based and in the RfD. Confidence in the key study was ranked low due to the small number of animals tested, the small number of parameters measured, and the lack of toxic effects at the highest dose tested. The low ranking of the RfD was due to lack of high-quality supporting studies and the fact that developmental and reproductive effects are not well studied.[42]

The RfD for chromium III is 1 mg/kg/day, based upon a NOAEL (adjusted) of 1,468 mg/kg/day, an uncertainty factor of 1,000, and a modifying factor of 1.[43] This was based on no effects observed in rats fed chromium III in the diet for 2 years. EPA has low confidence in the study on which the RfD was based and in the RfD. The low ranking of the key study was due to the lack of explicit detail on study protocol and results, while the low ranking of the RfD was due to the lack of supporting data and the lack of an observed effect level in the key study.[44] EPA has not established an RfC for chromium III[45] or chromium VI.[46]

E.5.2.3 Reproductive and Developmental for Chromium. Limited information is available on the reproductive or developmental effects of chromium in humans. The only available data suggest that exposure to chromium (VI) by inhalation in women may result in complications during pregnancy and childbirth.[47]

Animal studies have not reported reproductive effects from inhalation exposure to chromium (VI). Oral studies on chromium (VI) have reported severe developmental effects in mice such as gross abnormalities and reproductive effects including decreased litter size, reduced sperm count, and degeneration of the outer cellular layer of the seminiferous tubules. No information is available on the reproductive or developmental effects of chromium (III) in humans or animals.[48]

E.6 HYDROCHLORIC ACID HEALTH EFFECTS SUMMARY

Hydrochloric acid is an aqueous solution of hydrogen chloride gas and is commercially available in several concentrations and purities. Because of impurities, commercial varieties of hydrochloric acid are generally yellow. Hydrochloric acid is used in refining metal ore, as a lab reagent, and in the removal of scale from boilers.[49]

E.6.1 Cancer Effects

Limited information is available on the possible carcinogenic effects of hydrochloric acid. No information is available on the cancer risk to humans from exposure to hydrochloric acid. The carcinogenic effects of combined and separate exposures via inhalation to formaldehyde and hydrochloric acid were investigated in a study on rats. No carcinogenic response was observed when rats were exposed only to hydrochloric acid at concentrations of 10 ppm.[50] No studies have investigated risk of cancer in animals as a result of oral or dermal exposures.

EPA has not classified hydrochloric acid with respect to potential carcinogenicity and has not estimated the unit cancer risk associated with hydrochloric acid.[51]

E.6.2 Noncancer Effects - Hydrogen Cloride

E.6.2.1 Acute (Short-Term) Effects for Hydrogen Cloride. The acute effects on humans exposed by inhalation to hydrochloric acid include coughing, choking, inflammation and ulceration of the respiratory tract, chest pain, and pulmonary edema. Oral exposure may result in corrosion of the mucous membranes, esophagus, and stomach, with nausea, vomiting, intense thirst, and diarrhea. Dermal contact with hydrochloric acid can cause burns, ulcerations, and scarring.[52]

Animals exposed to 320 parts per million (ppm) for 6 minutes suffered sensory irritation, while levels of 680 ppm or higher for 1 minute resulted in less severe effects; inhalation of air containing 6,400 mg/m^3 hydrochloric acid for 30 minutes resulted in death from laryngeal spasm, laryngeal edema, or rapidly developing pulmonary edema.[53] Acute inhalation exposure tests resulted in an LC_{50} of 1,108 ppm for exposed mice and 3,124 ppm for exposed rats (moderate to high acute toxicity). An LD_{50} of 900 mg/kg (moderate acute toxicity) was reported for rabbits exposed orally to hydrochloric acid.[54] No information is available on effects in animals from acute dermal exposure to hydrochloric acid.

E.6.2.2 Chronic (Long-Term) Effects for Hydrogen Cloride. In humans, cases of gastritis, chronic bronchitis, dermatitis, and photosensitization have been reported among individuals exposed occupationally to hydrochloric acid.[55] No other data are available specifically on the effects of long-term human exposure dermally or via inhalation or ingestion.

In animals, the only study of the effects of long-term inhalation of hydrochloric acid reported epithelial or squamous hyperplasia of the nasal mucosa, larynx, and trachea. In a 90-day inhalation study, decreased body weight gains, minimum to mild rhinitis, nasal cavity lesions, and eosinophilic globules in the epithelial lining of the nasal tissues were reported in test animals.[56] No studies are available on the long-term effects on animals from low-level oral or dermal exposures to hydrochloric acid.

EPA has established an RfC for hydrochloric acid of 0.02 mg/m^3. This concentration was based on a rate study in which hyperplasia of the nasal mucosa, larynx, and trachea were seen. An uncertainty factor of 300 was applied to an LOAEL of 6.1 mg/m^3.[57] The EPA has low confidence in the study, database, and RfC because the study used only one dose and the database did not provide any additional chronic or reproductive studies.[58]

E.6.2.3 Reproductive and Developmental for Hydrogen Cloride. No information is available on reproductive or developmental effects of hydrochloric acid in humans. In animal studies in which female rats were exposed via inhalation prior to mating and during gestation, severe dyspnea, cyanosis, and altered estrus cycles were noted in the dams; increased fetal mortality and decreased fetal weight were also reported in offspring.[59] No animal studies are available on reproductive or developmental effects of oral or dermal exposure.

E.7 HYDROGEN FLUORIDE HEALTH EFFECTS SUMMARY

Hydrogen fluoride (HF) is a colorless gas that is used in making aluminum and in making chlorofluorocarbons. HF readily dissolves in water, is present in the air or other media, and, in the dissolved form, is known as hydrofluoric acid. Air around hazardous waste sites or factories that use or produce HF may contain this chemical.[60] EPA has set a maximum contaminant level (MCL) of 4 mg/L for HF.[61]

E.7.1 Cancer Effects - Hydrogen Fluoride

A cohort of workers in Denmark exposed to hydrofluoric fumes or dust reported an increase in mortality and morbidity from respiratory cancer. Increased lung cancer rates have been reported in aluminum industry workers, although no correction was made for smoking and exposure to other chemicals. Epidemiological studies of populations exposed to fluorides through drinking water have not shown an increased risk of cancer. No data are available on cancer in humans following dermal exposure to HF.[62] No animal studies have been identified regarding the carcinogenic effects of HF. EPA has not classified HF with respect to carcinogenicity and has not estimated a unit risk for HF.[63]

E.7.2 Noncancer Effects - Hydrogen Fluoride

E.7.2.1 Acute (Short-Term) Effects for Hydrogen Fluoride. Acute (short-term) inhalation exposure to HF can cause severe respiratory damage in humans, including severe irritation and pulmonary edema. Many of the human studies regarding inhalation of HF also involved dermal exposure, making it difficult to determine which effects are specific to the inhalation route. The results of ingestion include necrosis of the esophagus and stomach with nausea, vomiting, diarrhea, circulatory collapse, and death. Severe ocular irritation and dermal burns may occur following eye or skin exposure.[64,65]

In animals, acute inhalation exposure has resulted in renal and hepatic damage. HF produces irritation of the eyes, skin, and conjunctivae in rats as a result of dermal exposure. No information was found on the effects on animals from oral exposures to HF.[66]

E.7.2.2 Chronic (Long-Term) Effects for Hydrogen Fluoride. The major health effect of chronic inhalation exposure to HF and fluoride dusts, either individually or in combination, is skeletal fluorosis.[67] Chronic inhalation exposure of humans to HF has resulted in irritation and congestion of the nose, throat, and bronchi at low levels.[68] In addition, persons exposed occupationally to HF and fluoride dusts in an aluminum smelter reported reduced expiratory volume and increased cough and sputum production. No information is available on the chronic effects of oral or dermal exposure to HF in humans.[69]

Limited information exists on the chronic effects of HF in animals. Damage to the liver, kidneys, and lungs has been observed in animals chronically exposed to HF by inhalation.[70] No information was found on the long-term effects of oral or dermal exposure in animals. EPA is reviewing the RfC and RfD for HF.[71]

E.7.2.3 Reproductive and Developmental Effects for Hydrogen Fluoride. No studies were located regarding the developmental and reproductive effects in humans from inhalation, oral, or dermal exposure to HF.[72]

Dogs exposed via inhalation to HF developed degenerative testicular changes and ulceration of the scrotum. No studies were found regarding the reproductive and developmental effects in animals from oral or dermal exposure.[73]

E.8 MERCURY HEALTH EFFECTS SUMMARY

Mercury is a naturally occurring element that exists in three forms: elemental mercury, inorganic mercury (primarily mercuric chloride), and organic mercury (primarily methyl mercury). Elemental mercury is a shiny, silver-white, odorless liquid; inorganic mercury compounds are usually white powders or crystals; and organic mercury compounds are white crystalline solids. The majority of mercury in air is elemental mercury vapor, which is released to the air by natural and industrial sources. EPA has set a Maximum Contaminant Level (MCL) of 0.002 mg/L for mercury.[74] The health effects of mercury are discussed in the draft Mercury Study Report to Congress, which is currently being reviewed by the Science Advisory Board (SAB). See Chapter 7 of this interim utility study report for information on how to obtain the draft mercury study.

E.9 NICKEL HEALTH EFFECTS SUMMARY

Nickel is a silvery-white metal that is usually found in nature as a component of silicate, sulfide, or arsenide ores. The following table presents the physical properties of some of the major forms of nickel.

The most predominant forms of nickel in the atmosphere are probably nickel sulfate, nickel oxides, and the complex oxides of nickel. Each form of nickel exhibits different physical properties. Nickel compounds may be divided into two groups: soluble and insoluble nickel compounds. The soluble compounds include nickel sulfate and nickel acetate. Insoluble compounds include nickel monoxide, metallic nickel, nickel hydroxide, nickel subsulfide, and nickel carbonyl. Most nickel is used to make stainless steel; other uses include the manufacture of batteries, electroplating baths, textile dyes, coins, spark-plugs, and machinery parts.

E.9.1 Cancer Effects - Nickel

Human studies have reported an increased risk of lung and nasal cancers among nickel refinery workers exposed to nickel refinery dust and to nickel sulfate.[75] Nickel refinery dust is defined as the "dust from pyro-metallurgical sulfide nickel matte" refineries and is a mixture of many nickel compounds, including nickel subsulfide. It is not clear which compound is carcinogenic in the nickel refinery dust.[76] No information is

Chemical Name	Formula	Description	Solubility
Metallic Nickel	Ni	Lustrous white, hard ferromagnetic metal or grey powder	Soluble in dilute nitric acids; slightly soluble in hydrochloric or sulfuric acids; insoluble in cold or hot water
Nickel Hydroxide	$Ni\ (OH)_2$	Green crystals or amorphous solid	Nearly insoluble in cold water; soluble in acid, ammonium hydroxide
Nickel Subsulfide	Ni_3S_2	Lustrous pale yellow or bronze metallic crystals	Insoluble in cold water; soluble in nitric acid
Nickel Carbonyl	$Ni\ (O)_4$	Colorless to yellow liquid	Nearly insoluble in water; soluble in ethanol, benzene, and nitric acid; insoluble in dilute acids or dilute alkali
Nickel Sulfate (anhydrous)	$NiSO_4$	Pale-green to yellow crystals	Soluble in water; insoluble in ethanol
Nickel Monoxide	NiO	Grey, black, or green powder	Insoluble in water; soluble in acid
Nickel Acetate	$Ni(OCOCH_3)_2$	Dull-green crystals	Soluble in water; insoluble in ethanol

Source: IARC 1990[35]

available on the carcinogenic effects of nickel in humans from oral or dermal exposure.[77,78]

Animal studies have reported lung tumors from inhalation exposure to the following nickel compounds and mixtures: nickel refinery dusts, nickel sulfate, nickel subsulfide, nickel carbonyl, and metallic nickel. Studies in animals have reported tumors from intramuscular and other routes of administration from exposure to nickel monoxide and nickel hydroxide. Oral animal studies have not reported tumors from exposure to nickel acetate in the drinking water. No information is available on the carcinogenic effects of nickel in animals from dermal exposure[79-82]

E.9.1.1 Cancer Effects for Nickel Refinery Dust. EPA has classified nickel refinery dust in Group A - Known Human Carcinogen. For nickel refinery dust, the Group A classification was based on an increased risk of lung and nasal cancer in humans through inhalation exposure and increased lung tumor incidences in animals.[83] The International Agency for Research on Cancer (IARC) has classified nickel refinery dust as having sufficient evidence in humans for carcinogenicity. This is based on the same information EPA used.

EPA used the additive and multiplicative extrapolation method, based on human data, to estimate the unit cancer risk for nickel refinery dust. EPA calculated an inhalation unit risk estimate of 2.4×10^{-4} $(\mu g/m^3)^{-1}$.[84] Based upon this unit risk estimate, EPA estimates that if an individual were to breathe air containing nickel refinery dust at 0.004 $\mu g/m^3$ over his or her entire lifetime (70 yrs, 24 hrs/day), that person would theoretically have an increased chance of up to one in one million of developing cancer as a direct result of breathing air containing this chemical. Similarly, EPA estimates that breathing air containing 0.04 $\mu g/m^3$ would result in an increased chance of an increased chance of up to one in one hundred thousand of developing cancer.[85]

EPA used four data sets, all from human exposure, to calculate the unit risk estimates for nickel refinery dusts. A range of incremental unit risk estimates was calculated from these data sets that were consistent with each other.[86]

E.9.1.2 Cancer Effects for Nickel Sulfate. The National Toxicology Program (NTP) has recently completed a draft report on the carcinogenic effects of nickel sulfate hexahydrate. They have concluded that there was no evidence of carcinogenic activity of nickel sulfate hexahydrate in male or female rats or male or female mice. These conclusions are based on the results of 2-year inhalation studies.[87]

The International Committee on Nickel Carcinogenesis in Man summarized the available epidemiologic data on nickel and concluded that there was strong evidence that exposure to soluble nickel (primarily nickel sulfate) was associated with an increased respiratory cancer risk.[88]

The International Agency for Research on Cancer (IARC) has classified nickel sulfate as having sufficient evidence in humans for carcinogenicity.[89] This is based on epidemiological studies that showed an increased risk of lung and nasal cancer through inhalation exposure. In addition, animal studies have reported malignant tumors in the peritoneal cavity when nickel sulfate was applied by intraperitoneal injections.[90]

E.9.1.3 Cancer Effects for Nickel Subsulfide. EPA has also classified nickel subsulfide in Group A, based upon the same studies as those that were used to classify nickel refinery dust.[91] For nickel subsulfide, EPA also used human data to estimate the unit cancer risk. EPA calculated an inhalation unit risk estimate of 4.8×10^{-4} $(\mu g/m^3)^{-1}$.[92] EPA estimates that if an individual were to breathe air containing this nickel compound at 0.002 $\mu g/m^3$ over his or her entire lifetime, that person would theoretically have an increased chance of up to one in one million chance of developing cancer as a direct result of breathing air containing this chemical. Similarly, EPA estimates that breathing air containing 0.02 $\mu g/m^3$ would result in an increased chance of up to one in one hundred thousand chance of developing cancer. EPA has also calculated unit risk estimates for nickel subsulfide from a rat inhalation study. These estimates were approximately one order of magnitude greater than those calculated from the human studies.[93]

The National Toxicology Program has recently completed a draft report on the carcinogenic effects of nickel subsulfide. They have concluded that there was clear evidence of carcinogenic activity of nickel subsulfide in male and female rats and no evidence of carcinogenic activity for male and female mice. These conclusions are based on the results of 2-year inhalation studies.[94]

IARC has classified nickel subsulfide as having sufficient evidence in humans and experimental animals for carcinogenicity.[95] The International Committee on Nickel Carcinogenesis in Man concluded that there was some evidence to suggest that exposure to nickel subsulfide presents on increased risk of lung and nasal cancer.[96]

The State of California has calculated an estimated unit risk for continuous lifetime exposure to nickel subsulfide at 1 μg Ni/m^3. This risk ranges from 2.8×10^{-3} for the maximum likelihood estimate to 3.7×10^{-3} for the upper 95 percent confidence limit. This risk estimate was based on animal data.[97]

E.9.1.4 Cancer Effects for Nickel Carbonyl. EPA has classified nickel carbonyl in Group B2 - Probable Human Carcinogen. For nickel carbonyl, this classification was based on an increase in lung tumors in animals exposed via inhalation.[98] IARC has classified nickel carbonyl as having limited evidence in experimental animals for carcinogenicity.[99] This is based on the same information as that EPA used.

EPA has not calculated an inhalation or an oral unit cancer risk estimate for nickel carbonyl, due to the lack of appropriate data. In one study, the survival rate of the animals was very low, and another study used the intravenous route of exposure.[100]

E.9.1.5 Cancer Effects for Nickel Monoxide. The NTP has recently completed a draft report on the carcinogenic effects of nickel monoxide. They have concluded that there was some evidence of carcinogenic activity of nickel monoxide in male and female rats, no evidence of carcinogenic activity in male mice, and equivocal evidence of carcinogenic activity in female mice. These conclusions are based on the results of 2-year inhalation studies.[101]

IARC has classified nickel monoxide as having sufficient evidence in experimental animals for carcinogenicity.[102] This is based on animal studies that showed an increased incidence of tumors in rats exposed via intrapleural, intramuscular, and intraperitoneal administration. The International Committee on Nickel Carcinogenesis summarized the available epidemiologic data on nickel and concluded that there was some evidence to suggest that exposure to oxidic nickel (including nickel monoxide) may result in increased lung and nasal cancer risks.[103]

E.9.1.6 Cancer Effects for Nickel Hydroxide. IARC has classified nickel hydroxide as having sufficient evidence in experimental animals for carcinogenicity.[104] This is based on animal studies that showed an increase in tumors in rats exposed via intramuscular injection.

E.9.1.7 Cancer Effects for Metallic Nickel. IARC has classified metallic nickel as having sufficient evidence in experimental animals for carcinogenicity.[105] This is based on animal studies that showed an increase in tumors from exposure via inhalation and intratracheal, intraperitoneal, and intravenous administration. The International Committee on Nickel Carcinogenesis in Man summarized the available data on nickel and concluded that the available information gave no evidence of increased respiratory cancer risks from exposure to metallic nickel.[106]

E.9.1.8 Nickel Acetate. IARC has not classified nickel acetate as to carcinogenicity.[107]

E.9.1.9 Overall Assessment for Nickel Compounds. IARC examined all of the data on nickel and stated that for an overall evaluation, it considers nickel compounds to be carcinogenic to humans and metallic nickel to be possibly carcinogenic to humans.[108]

The State of California has calculated an estimated unit risk for continuous lifetime exposure to nickel compounds at 1 $\mu g/m^3$. This risk ranges from 2.1×10^{-4} for the maximum likelihood estimate to 2.57×10^{-4} for the upper 95 percent confidence limit. This risk estimate was based on human data. They also concluded that all nickel compounds should be considered potentially carcinogenic to humans by inhalation.[109]

The American Conference of Governmental Industrial Hygienists (ACGIH) have stated that all nickel compounds should be considered as confirmed human carcinogens, based on the weight of evidence from epidemiologic studies of, or convincing clinical evidence in, exposed humans.[110]

The International Committee on Nickel Carcinogenesis in Man concluded that more than one form of nickel gives rise to lung and nasal cancer. They stated that although much of the respiratory cancer risk seen among nickel refinery workers could be attributed to exposure to a mixture of nickel oxides and sulfides, exposure to large concentrations of nickel oxides in the absence of nickel sulfides was also associated with increased lung and nasal cancer risks. In addition, there was evidence that soluble nickel exposure (such as nickel sulfate) increased the risk of these cancers. They concluded that respiratory cancer risks are primarily related to exposure to soluble nickel at concentrations greater than 1 mg/m^3 and to exposure to less soluble forms at concentrations greater than 10 mg/m^3.[111]

E.9.2 Noncancer Effects - Nickel

E.9.2.1 Acute (Short-Term) Effects for Nickel. Nickel carbonyl appears to be the most acutely toxic nickel compound. Symptoms from acute inhalation exposure in humans include headache, vertigo, nausea, vomiting, insomnia, and irritability, followed by chest pains, dry coughing, cyanosis, gastrointestinal symptoms, sweating, visual disturbances, and severe weakness. Acute oral exposure to high levels of nickel sulfate and nickel chloride in humans has resulted in vomiting, cramps, impaired vision, giddiness, headache, and cardiac arrest in humans. No information is available on the acute effects of nickel via dermal exposure in humans.[112]

The lungs and kidneys appear to be target organs for acute nickel carbonyl toxicity, via inhalation and oral exposure in animals, with pulmonary fibrosis and renal edema reported. No information is available on acute effects of nickel via dermal exposure in animals.[113] Acute animal tests, such as the LD_{50} test in rats, have shown nickel compounds to exhibit acute toxicity values ranging from low to high, based upon LD_{50} data in the range of 50 mg/kg to greater than 5,000 mg/kg. The soluble compounds, such as nickel acetate, were most toxic, and the insoluble compounds, such as metallic nickel powder, were the least toxic.[114]

E.9.2.2 Chronic (Long-Term) Effects for Nickel. Contact dermatitis is the most common effect in humans from exposure to nickel, via inhalation, oral, and dermal exposure. Cases of nickel contact dermatitis have been reported following occupational and nonoccupational exposure, with symptoms of itching of fingers, wrists, and forearms. Chronic inhalation exposure to nickel in humans also results in respiratory effects. These effects include direct respiratory effects such as asthma due to primary irritation or an allergic response and an increased risk of chronic respiratory tract infections.[115,116]

Animal studies have reported effects on the lungs, kidneys, and immune system from inhalation exposure to nickel, and effects on the respiratory and gastrointestinal systems, heart, blood, liver, kidney, and decreased body weight from oral exposure to nickel. Dermal animal studies have reported effects on the skin.[117,118]

E.9.3 Essentiality for Nickel

Nickel has been demonstrated to be an essential nutrient for some mammalian species, and it has been suggested that it may also be essential for human nutrition. A requirement for nickel has not been conclusively demonstrated in humans, and a recommended daily allowance has not been set. By extrapolation from animal data, there have been various estimates of the human daily requirement for nickel. The National Academy of Sciences estimated that a 70 kilogram person would have a daily requirement of 50 μg of nickel.[119] Other researchers have estimated requirements ranging from 30 μg to 120 μg of nickel.[120]

E.9.4 Reproductive and Developmental Effects for Nickel

No information is available regarding the reproductive or developmental effects of nickel in humans. Animal studies have reported developmental effects, such as a reduction in fetal body weight, and reproductive effects, including testicular degeneration from inhalation exposure to nickel. Oral animal studies have reported deaths in females due to pregnancy complications and a significant decrease in number of offspring per litter from exposure to nickel.[121]

E.9.5 Noncancer Health-Based Numbers for Nickel

EPA has established a Reference Dose (RfD) for nickel (soluble salts) of 0.02 mg/kg/day, based upon a NOAEL (adjusted) of 5 mg/kg/day, an uncertainty factor of 300, and a modifying factor of 1.[122] This was based on a study in rats that showed decreased body and organ weights from chronic (2-year) exposure to nickel in the diet. Other studies showed similar results, with decreased body and organ weights after exposure to nickel chloride via gavage and through the drinking water. EPA has not established a Reference Concentration (RfC) for any nickel compound.[123]

EPA has medium confidence in the RfD for nickel (soluble salts) and low confidence in the study on which it was based. The Ambrose et al.[124] study was properly designed and provided adequate toxicological endpoints; however, high mortality occurred in the controls. The database provided adequate supporting subchronic studies; this was the basis for EPA's medium confidence level in the RfD.[125]

The EPRI has recommended a RfC of 2.38 x 10^{-3} mg(Ni)/m^3 for all nickel compounds. This was based on the ACGIH Threshold Limit Value (TLV). It was translated for community exposure by scaling for exposure time differences between community and occupational exposure assumptions.[126]

Calabrese has calculated an ambient air level goal (AALG) for soluble nickel compounds of 0.36 ng (Ni)/m^3 and an AALG for insoluble nickel compounds of 7.1 ng (Ni)/m^3. An AALG is a health-based guideline based on risk assessment methodology similar to that used by EPA.[127]

The California Air Resources Board has stated that the most sensitive noncancer endpoint reported in humans is allergic sensitization, while immune suppression is the most sensitive endpoint reported in animal studies. The board has concluded that because these noncancer effects occur at concentrations greater than 3 orders of magnitude above a 24-hour maximum concentration of nickel (0.024 ng(Ni)/m^3) measured in California near an industrial source, it is unlikely that noncancer health effects would be caused by the levels of nickel compounds currently in the air.[128]

The Agency of Toxic Substances and Disease Registry[78] has recommended a minimum risk level (MRL) for intermediate duration, inhalation exposure to nickel of 9.5 x 10^{-5} mg(Ni)/m^3. They have stated that this MRL may not be protective for some hypersensitive individuals.[129] An MRL is a health-based guideline based on similar risk assessment methodology to that used by EPA.

E.9.6 Federal Regulations and Guidelines for Nickel

The Occupational Safety and Health Administration (OSHA) has established a maximum allowable level of nickel in workplace air for an 8-hour workday, 40-hour workweek of 1 mg(Ni)/m^3 for metallic nickel and insoluble compounds, and 0.1 mg(Ni)/m^3 for soluble nickel compounds.[130]

The National Institute of Occupational Safety and Health has a recommended exposure level for workplace air of 0.15 mg (Ni)/m^3 for all nickel compounds except nickel carbonyl and 7 μg (Ni)/m^3 for nickel carbonyl.[131]

The ACGIH has recommended a TLV of 0.05 mg(Ni)/m^3 for an 8-hour exposure in the workplace to all nickel compounds (elemental, insoluble, and soluble).[132]

The EPA has set a maximum contaminant level (MCL) of 0.1 mg/L for nickel. This is the maximum level allowed in drinking water.[133]

E.10 2,3,7,8-TETRACHLORODIBENZO-P-DIOXIN HEALTH EFFECTS SUMMARY

2,3,7,8-Tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) belongs to the class of compounds, chlorinated dibenzo-p-dioxins, which are referred to as dioxins. 2,3,7,8-TCDD is a colorless solid with no known odor. It does not occur naturally, nor is it intentionally manufactured by any industry, although it can be produced inadvertently in small amounts as an impurity during the manufacture of certain herbicides and germicides and has been detected in products of incineration of municipal and industrial wastes. The only present use for 2,3,7,8-TCDD is in chemical research.[134]

E.10.1 Cancer Effects - Dioxins

An increase in lung cancer risks was observed among Japanese males exposed as a result of an oil poisoning accident. Human studies have also found an association between 2,3,7,8-TCDD and soft-tissue sarcomas, lymphomas, and stomach carcinomas, although for malignant lymphomas, the increase in risk is not consistent. The increase in risk is of borderline significance for highly exposed groups and is less-significant among groups exposed to lower levels of 2,3,7,8-TCDD. Although there are problems with the studies of human effects, such as confounding factors, short follow-up period, and lack of exposure information, the overall weight of evidence from epidemiological studies suggests that the generally increased risk of cancer in humans is likely due to 2,3,7,8-TCDD.[135]

Information on the carcinogenicity of 2,3,7,8-TCDD following inhalation exposure of animals is not available. In animal studies of oral exposure to 2,3,7,8-TCDD, multisite tumors in rats and mice including the tongue, lung, nasal turbinates, liver, and thyroid have been reported. Estimates derived from human data suggest a unit risk for lung cancer of 3 x 10^{-4} to

5×10^{-4} pg/kg-day)$^{-1}$; for all cancers combined the unit risk estimate is 2×10^{-3} to 3×10^{-3} (pg/kg-day)$^{-1}$.[136]

E.10.2 Noncancer Effects – Dioxins

E.10.2.1 Acute (Short-Term) Effects for Dioxins. The acute effects on humans exposed through the spraying in Vietnam of herbicides that contained 2,3,7,8-TCDD include diarrhea, vomiting, skin rashes, fever, and abdominal pain.[137] Routes of exposure in these instances are not well defined and may include inhalation as well as oral and dermal exposures.

No information is available on effects in animals from acute inhalation exposure to 2,3,7,8-TCDD. In oral exposure studies, 2,3,7,8-TCDD is highly toxic to all laboratory animals tested even though there are large differences in species sensitivity. LD_{50} values range from 0.6 μg/kg in male guinea pigs to 5,500 μg/kg in hamsters. Other effects on animals from acute oral exposure include loss of body weight, hepatotoxicity, and decreased thymus weight.[138] Information on the effects of acute dermal exposure in animals is limited, although dermal effects have been reported.[139]

E.10.2.2 Chronic (Long Term) Effects for Dioxins. No studies are available on the inhalation toxicity of 2,3,7,8-TCDD in humans, although such exposure may have occurred in populations exposed to chemicals contaminated with 2,3,7,8-TCDD. Oral exposure of humans to chemicals contaminated with 2,3,7,8-TCDD has resulted in chloracne, immunotoxicity, hyperpigmentation, hyperkeratosis, possible hepatotoxicity, aching muscles, loss of appetite, weight loss, digestive disorders, headaches, neuropathy, insomnia, sensory changes, and loss of libido.[140]

Chloracne is the only substantiated effect in humans produced by dermal exposure to compounds contaminated with 2,3,7,8-TCDD.[141]

No information on chronic inhalation and dermal exposure is available for animals. Oral exposure to 2,3,7,8-TCDD has resulted in dermatitis, extreme loss of body weight, and effects on the liver and immune system.[142] EPA has not established an RfC or RfD for 2,3,7,8-TCDD.

E.10.2.3 Reproductive and Developmental Effects for Dioxins. Several studies have investigated the incidence of birth defects and reproductive effects in humans exposed to 2,3,7,8-TCDD through accidental releases or the spraying of

2,3,7,8-TCDD-contaminated herbicides. EPA has concluded that the data were not inconsistent with 2,3,7,8-TCDD's adversely affecting development, but as a result of the limitations of the data, these studies could not prove an association with 2,3,7,8-TCDD exposure and the observed effect. The major limitations in these human studies were the concomitant exposure to other potentially toxic chemicals, the lack of any specific quantitative data on the extent of exposure of individuals within the study group, and the lack of statistical power of the studies.[143]

No studies are available on the reproductive and developmental effects in animals caused by inhalation or dermal exposure to 2,3,7,8-TCDD.[144] In oral exposure studies, 2,3,7,8-TCDD has produced fetal anomalies, including cleft palate and hydronephrotic kidneys in mice and internal organ hemorrhage in rats, and resulted in spontaneous abortions in monkeys and decreased fetal survival.

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Appendix F: Documentation of The Inhalation Human Exposure Modeling for the Utility Study

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F.1 INTRODUCTION

The model used to calculate direct inhalation risks from (HAPs) emitted from utility boilers is the Human Exposure Model Version 1.5 (HEM). It was developed by the Pollutant Assessment Branch (PAB) of the EPA's OAQPS and was designed for screening assessments. The model is used in source ranking to assess the *relative* risks associated with exposure to different pollutants and to characterize human exposure, cancer risks, and noncarcinogenic hazards for stationary sources that emit HAPs. The HEM uses the Industrial Source Complex - Long Term Version 2 (ISCLT2) air dispersion model, updated 1990 census population data, meteorological, temperature, and mixing height databases, and chemical-specific health effects numbers (see Table F-1.)

The remainder of this technical report contains a description of ISCLT2, the population and meteorological databases, human exposure algorithms, and risk estimating methodology applied in HEM to arrive at direct inhalation risk estimates for this utility study.

F.2 ISCLT2 DISPERSION MODELING

Air dispersion modeling is used to estimate atmospheric fate and transport of pollutants from the point of emission to the location of exposure to arrive at long-term average ambient air concentrations of the pollutant. ISCLT2, the air dispersion model used in HEM, is the Agency's regulatory air dispersion model for the types of sources represented in this study. ISCLT2 is one of the primary models used to support EPA studies and regulatory programs for air pollutants. ISCLT2 uses emission parameters and meteorological data to estimate the transport and dispersion of pollutants in the atmosphere.

The ISCLT is a steady-state, Gaussian plume, atmospheric dispersion model that applies to multiple-point, area, and volume emission sources. It is designed specifically to estimate long-term ambient concentrations resulting from air emissions from these source types in a computationally efficient manner. ISCLT2 is recognized by the Guideline on Air Quality Models[1] as a preferred model for dealing with complicated sources (i.e., facilities with point, area, and volume sources) when estimating long-term concentrations (i.e., monthly or longer).

As described in the *Guideline on Air Quality Models*, the ISCLT is appropriate for modeling industrial source complexes in either rural or urban areas.[2] With this model, long-term ambient concentrations can be estimated for transport distances up to

Table F-1. Summary of HEM Features

Characterization	Single pollutant, multiple source, nationwide
Dispersion model	ISCLT2
Meteorological database	Data set from locations/years available on OAQPS TTN and from the National Weather Service
Population database	1990 Census Databases Block Level 6.9 million records
Exposure calculations	>0.5 km Interpolate air concentration to population <0.5 km Assign population to air concentration

HEM = Human Exposure Model
ISCLT2 = Industrial Source Complex Model - Long Term Version 2
OAQPS = Office of Air Quality Planning and Standards
TTN = Technology Transfer Network

50 km. The ISCLT2 incorporates separate point, area, and volume source computational algorithms for calculating ambient concentrations at user-specified locations (i.e., receptors). The locations of the receptors relative to the source locations are determined through a user-specified Cartesian coordinate reference system. For the utility study, receptors 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 kilometers from the source.

ISCLT2 source inputs vary according to source type. For the point sources in this study, the inputs include emission rate, physical stack height, stack inner diameter, stack gas exit velocity, and stack gas exit temperature.

The ISCLT2 is a sector-averaged model that uses statistical summaries of meteorological data to calculate long-term, ground-level ambient concentrations. The principal meteorological inputs to the ISCLT2 are stability array (STAR) summaries that consist of a tabulation of the joint frequency of occurrence of wind speed categories, wind-direction sectors, and Pasquill atmospheric stability categories. Other meteorological data requirements include average mixing heights for each stability class and average ambient air temperatures.

As described above, the ISCLT2 model computes long-term ambient concentrations at user-specified receptor points that occur as a result of air emissions from multiple sources. These computations are done on an emission point (stack)-by-stack basis such that the ambient concentration from each stack at each receptor is computed. Total ambient concentrations at a particular receptor are obtained by summing the contributions from each of the stacks. With Gaussian plume algorithms such as those included in the ISCLT2, the source contributions at each receptor are directly proportional to the source emission rate.

Normalized ambient concentrations for each source-receptor combination were computed such that they would correspond to a unit emission rate of 1 gram per second (g/s) for each stack in the facility. The total ambient concentration at a receptor is then computed as the sum of the contributions from each stack, where the latter are computed as the product of the normalized concentration and the desired emission rate. Mathematically, this can be expressed as follows:

$$X_i = \sum_{j=1}^{J} q_i x_{ij}$$

Where:

X_i = total ambient concentration at receptor i, µg/m^3
q_i = emission rate for stack, g/s
X_{ij} = normalized contribution from stack j to receptor i, µg/m^3
J = total number of stacks.

Thus, the principal output of the dispersion modeling is a set of normalized stack contributions, i.e., x_{ij} in the above equation for each scenario modeled.

F.2.1 Assumptions Used

For the utility study, HEM analysis flat terrain was assumed because of the lack of information. Building downwash was not considered because of the tall stacks used by the utility boilers. The assumption was made that all particles were small enough to behave as gases. All emissions from one site are assumed to originate from stacks that are collocated.

F.2.2 Model Options

Air dispersion is affected by surface roughness. The ISCLT2 model provides two regimes of surface roughness based on land classification: urban and rural. When there is no information available regarding the land classification around a particular source of interest, the air quality modeling guidelines suggest a surrogate, population density, to make a land classification determination. Because the population database which is part of the HEM model can easily provide population density estimates, this option was selected for the utility study for conducting the more detailed analyses. Initial screening analysis assumed the plant setting of "urban," which earlier sensitivity analysis indicated would maximize surrounding ambient concentrations estimates.

EPA's *Guideline on Air Quality Models*[3] distinguishes between urban and rural settings based on population density. "Urban" is defined as a population density greater than 750 people per km^2 in the area between the point source and a 3 km distance from the source; "rural" is assumed for a population density of less than 750 people per km^2.

ISCLT2 can be run in a number of different ways by changing various modeling options. For consistency in regulatory modeling applications, a set of choices has been defined as the default option. The default option set determines how the model calculates ambient air concentrations and includes:

- default stack-tip downwash calculations
- buoyancy-induced dispersion calculations
- final plume rise in all calculations
- calms processing routines
- upper-bound concentration estimates for sources influenced by building downwash from super-squat buildings
- default wind profile exponents
- default vertical potential temperature gradients.

The default option set was used in the utility study with one change. The plume rise was changed from the final plume rise option of the default selections to the use of a transitional plume rise. Plume rise accounts for how the plume behaves near the stack as a function of the momentum of release of the plume and the buoyant rising of the plume resulting from the high plume temperature in comparison to the surrounding air. The use of the transitional plume rise would be expected to produce more realistic estimates of ambient air concentrations near the stack where the maximum concentrations occur. Each of these defaults is defined further in the ISCLT2 User's Guide.[4]

F.3 HEM DATABASES

Four databases are contained in the HEM model. The meteorological database contains long-term summaries for selected locations across the country. HEM pairs plant locations with the nearest location for meteorological data contained in the database. The second database is the population database, which contains population data from the 1990 census. Ambient air concentrations of the modeled pollutant are coupled with the population numbers and location to develop nationwide exposure estimates. The two remaining databases contain estimates of ambient temperatures and mixing height.

F.3.1 Meteorological Database

The ISCLT2 meteorological database contains long-term meteorologic data, primarily from National Weather Service (NWS) airport locations, in the form of STablility ARrays (STAR summaries). STAR summaries display joint frequencies of occurrence of wind direction, wind speed, and air stability by combining these factors into a frequency distribution. HEM chooses the STAR Data set for each plant based on proximity of the plant to the location where the meteorological data were collected.[5]

The meteorological database used for the utility study contains data from hourly surface observations obtained from the OAQPS Technology Transfer System (TTN). The Support Center for Regulatory Air Models Bulletin Board System (SCRAM-BBS) contains annual data files of surface observations from 349 NWS locations (primarily airports) across the United States and its Territories for the years 1984-1989. From each location's surface observations, STAR summaries were created that encompass all available years into one long-term estimate of the location's dispersion characteristics. Figure F-1 depicts the coverage of the HEM meteorological database. The range of *averaging* years over which the data are averaged is from 1 to 6 years, with a typical average of 6 years (225 sites).

F.3.2 Population Database

The population database contains "block level" 1990 census data collected by the U.S. Census Bureau for reapportionment as specified in Public Law 94-17. It is used by the model to

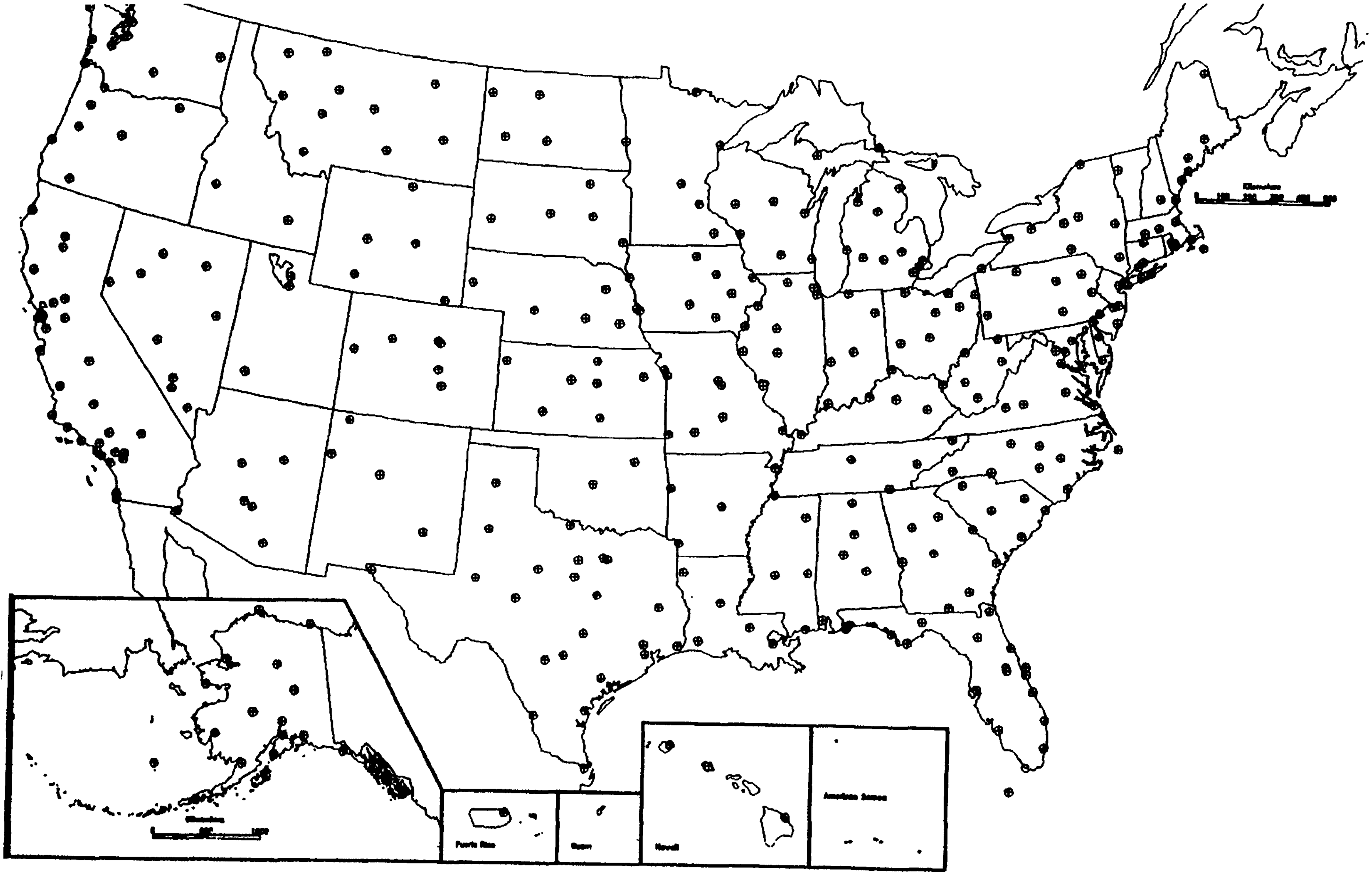

Figure F-1. **Location of STAR Data Sites** estimate the location and number of people exposed to the modeled pollutants. The 1990 population has been aggregated into 6.9 million blocks.

F.3.3 Mixing Height Database

The mixing height database is more limited in scope than the other databases mentioned above. Only 73 sites were available from the NWS for the years 1984-1989. Also, the mixing heights are calculated from observations taken once daily. Forty of the 73 sites are based on 6 years of observations.

F.3.4 Temperature Database

The temperature database provides an arithmetic average of ambient temperatures for each atmospheric stability class for each STAR site. Because the temperature was recorded for every set of wind speed and direction observations in the NWS raw data, the temperature database is similar to the meteorological database; that is, each database has the same number of sites (349), the same number of years of data to calculate the averages at each site, and the same typical number of years (6) on which averages are based. By default, the site closest to the plant is selected for air dispersion calculations and is, for this database, the nearest STAR site.

F.4 EXPOSURE ALGORITHMS

Exposure is calculated in HEM through pairing population information from the census database with modeled ambient air concentrations of each specific pollutant. The output of the dispersion model is an air concentration array around the plant. HEM calculates exposure by integrating the HAP air concentration at the population center (centroid) of the census block through interpolation of the air concentration values at the surrounding modeled points. All persons residing in the census block are treated as being exposed to the air concentration at the centroid.

F.4.1 Air Concentration - Population Pairing

ISCLT2 calculates air concentrations at user-specified receptors. For the utility study, receptors 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 receptors. Except for receptors located very close to the stack, HEM calculates exposure by interpolating the air concentration at the population centroid (the population center of the census block) between the values at the receptors surrounding the centroid. There is a linear relationship between the logarithm of the concentrations and the logarithm of the radial distances. This linear relationship is used to estimate the concentration along the radial nearest the centroid at the same distance from the stack as the centroid. The estimates are then interpolated linearly between the radials of the receptors surrounding the population centroid. Figure F-2 depicts the relationship between the receptor locations and a hypothetical block population centroid.

F.4.2 Exceptions for Population Close to Source

Within 0.5 km of the stack, the exposure is calculated differently than described above because close to the stack, the receptors are much closer together. Here, the population is estimated at the points where the air concentration is calculated, rather than the air concentrations' being estimated at the known population point. This more complicated scheme is described in detail in the HEM User's Manual.[6]

F.5 RISK CALCULATIONS

In general, long-term exposure estimates are paired with chemical-specific health benchmarks, such as inhalation unit risk estimates (IUREs), to calculate the risk to the population of developing cancer or the potential for developing other adverse health effects. Health benchmarks are input for each chemical modeled. Health benchmarks and other toxicity information are discussed in *Health Effects Summaries for the Utility Study.*[7] Risk is calculated for the exposed population on a single-pollutant basis. For carcinogens, HEM produces distributions of exposure and risk, as well as estimates of annual incidence, number of people exposed at various risk levels, and maximum individual risk (MIR). A comparison of the modeled ambient air concentration to the reference concentration is used to estimate the extent of adverse health effects for noncarcinogens. Aggregate risk associated with exposure to multiple pollutants is evaluated by adding the risks from individual pollutants.

The utility boiler HEM modeling application requires the input of chemical-specific toxicity information. HEM uses the IUREs for carcinogens to estimate cancer risks or other adverse health effects for each individual chemical according to that chemical's particular level of toxicity. The more toxic a chemical, the lower the ambient air concentration necessary to produce high risk levels.

F.5.1 Required Health Number Input

An IURE is entered in the risk calculation for each carcinogenic pollutant. The IURE represents an estimate of the increased cancer risk from a lifetime (70-year) exposure to a

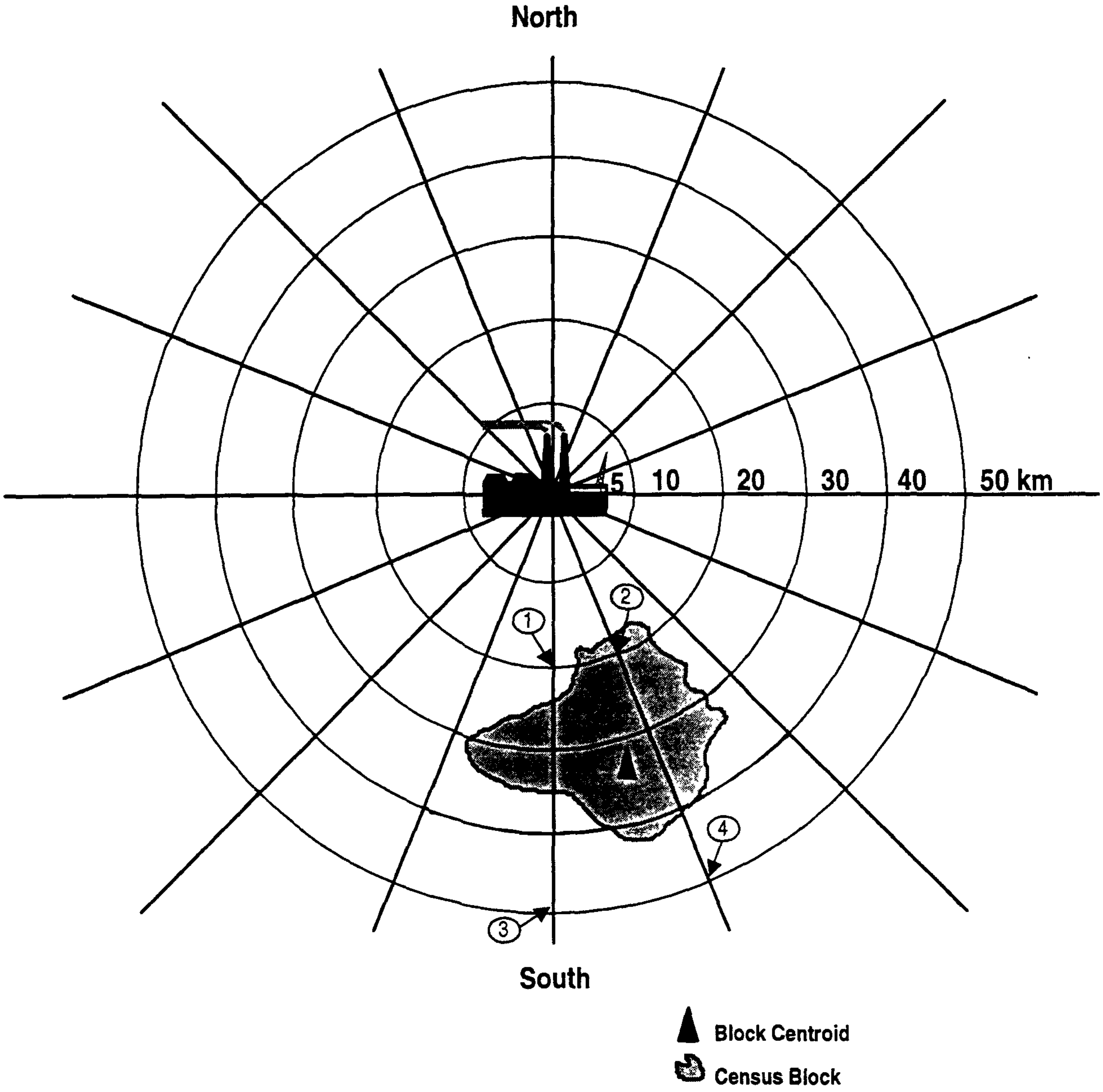

Figure F-2. The exposure algorithms interpolate between the estimated air concentrations and the population data. Air concentrations are calculated at the points where the circles and lines intersect. Poplulation is known at the block centroid locations. The concentration at the centroid is calculated based on the concentration estimated at the 4 points surrounding the centroid.

concentration of one unit of exposure. The IURE for inhalation is normally expressed as risk per $\mu g/m^3$ of air contaminant.

Hazard quotients for noncarcinogens are calculated by comparing the ambient air concentration of the pollutant with its reference concentration (RfC). The RfC is an estimate (with uncertainty spanning perhaps an order of magnitude) of the daily exposure of the human population to the chemical by inhalation (including sensitive subpopulations) that is likely to be without deleterious effects during a lifetime.

F.5.2 Risk Calculations

HEM calculates carcinogenic risk using standard EPA risk equations and assumptions. Maximum individual risk (MIR) is defined as the increased probability of an individual to develop cancer following exposure to a pollutant at the maximum modeled long-term ambient concentration assuming a lifetime of exposure. It is calculated by multiplying the estimated ambient air concentration of a HAP by the IURE.

Unlike cancer risk characterization, noncancer risks typically are not expressed as a probability of an individual suffering an adverse effect. Instead, the estimated exposure concentration is compared with a noncancer health benchmark such as an RfC. This is usually expressed as a hazard quotient. the hazard quotient is the ratio of the exposure (ambient air concentration of the pollutant) to the RfC. The RfC represents the highest protective concentration, and a ratio value greater than or equal to one would represent an exposure that may be a public health concern and should be evaluated further.

For additional information on the carcinogenic and noncarcinogenic effects of HAPs, refer to *Health Effects Summaries for the Utility Study.*[8]

F.6 ASSUMPTIONS

Simplifying assumptions are used in the HEM utility boiler analysis to enable estimation of the potential health effects due to HAP emissions from utility boilers. The following assumptions are made from HEM:

1. Direct inhalation of pollutants is the only source of exposure.
2. Average exposures are equivalent to those experienced if one constantly stayed at home; no adjustment is made

for exposure changes resulting from population movement between home, school, work, etc.

3. Homes are located at population-weighted centers (centroids) of census blocks (or at nodes of the polar grid within 0.5 km) because the locations of actual residences are not included in the database.
4. For the most exposed individuals, it is assumed that people reside at the home for their entire lifetimes (in modeling carcinogens, a lifetime is assumed to be 70 years).
5. Indoor concentrations are the same as outdoor concentrations.
6. The plant emits pollutants at the same level for the 70-year lifetime of exposure.
7. No resuspension of pollutants via dust occurs.
8. There is no population migration or growth.
9. Varying exposures that might arise as a result of differences in age, sex, health status, degree of activity, etc. do not exist.
10. Because the model does not handle complex terrain, each plant is located in flat terrain. An additional complex terrain analysis was conducted using specially-designed models.
11. The nearest meteorological location provides the most appropriate STAR, temperature, and mixing height data for the plant.
12. No pollutants are emitted from point sources other than stacks.

F.7 HEM OUTPUT

For carcinogens, HEM produces estimates of annual incidence (population risk), number of people exposed to various risk levels, and maximum individual lifetime risk. For noncarcinogens, HEM estimates the number of people exposed at various concentrations and the maximum individual concentration. These values are for individual pollutants; no summing of risks across chemicals is performed.

F.8 REFERENCES

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7. Health Effects Summaries - Overview (Apendix E of this report).

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Appendix G - Preliminary Uncertainty Analysis for the Characterization of the Human Health Risks from Direct Inhalation Exposures to Electric Utility HAP Emissions

EXECUTIVE SUMMARY

GES.1 INTRODUCTION

Section 112(n)(1)(A) of the 1990 Clean Air Act Amendments requires the U.S. Environmental Protection Agency (EPA) to report on the hazards to the public health that could reasonably occur as a result of emissions of hazardous air pollutants (HAPs) from electric utility steam generating units (utilities).

As part of the study, EPA completed an assessment of the risks to human health from direct inhalation of utility plants' HAP emissions within 50 km of each plant for the year 1990. The EPA conducted an uncertainty analysis on the estimates of risks from direct inhalation of utility HAP emissions. The need for uncertainty analysis as a part of any risk assessment and its aid in conveying results of the risk assessment is widely accepted, having been proposed in both EPA Risk Characterization Guidance[1] and the NRC Committee Report: *Science and Judgement in Risk Assessment*.[2] Furthermore, the initial risk assessment relies on a combination of point values - some conservative and some not conservative. This method yields point estimates of exposure and risk that fall at unknown percentiles of the full distributions of exposure and risk. For this reason, the EPA conducted an uncertainty analysis to determine the range of possible values and to estimate the degree of conservatism in the point estimates calculated in the original risk assessment.

The original inhalation risk assessment is presented in Chapters 4 to 6 of the interim report. However, a summary of the results are provided below to provide the reader with the risk estimates before the uncertainty analysis is presented.

GES.2 ESTIMATES OF HUMAN HEALTH RISKS FROM DIRECT INHALATION OF HAPS EMITTED BY ELECTRIC UTILITIES

The inventory of sources modeled included all coal-, oil- and gas-fired plants that have at least one boiler with a capacity of 25 MW or more: 426 coal-fired plants, 137 oil-fired plants, and 267 gas-fired plants. Risks were estimated for 12 carcinogens and four noncarcinogens.

GES.3 CARCINOGENS

Direct inhalation exposures to emissions from 24 (22 oil-fired and 2 coal-fired) plants are estimated to result in individual risks of 1 in 1,000,000 (10^{-6} or greater). Emissions from oil- and coal-fired facilities are the major contributors to these risks from direct inhalation, while all gas-fired utilities have been shown to present lower risks. The total annual cancer incidence estimated to result from direct inhalation exposure to HAP emissions within 50 km of plants was estimated as 0.56 cases/year. HAP emissions from oil-fired utilities accounted for 0.47 cases/yr or about 84 percent of the total incidence. Table G-1 summarizes the population exposed within 50 km of individual facilities at different levels of individual risk. The highest MEI across all utilities was 1 $\times 10^{-4}$ for the mixture of HAPs.

GES.4 NONCARCINOGENS

The highest HQ across all noncarcinogen HAPs across all plants was 0.12 from hydrogen chloride and 0.046 from manganese exposures from a coal-fired plant. All HQ values were less than one. These risk estimates are for inhalation exposure within 50 km alone and do not include risks associated with long-range transport or indirect exposures (e.g., ingestion of contaminated foods). The risks from indirect exposure may potentially be significant for those compounds which are environmentally persistent and have the tendency to bioaccumulate.

GES.5 DISCUSSION

The methods used up to this stage of the risk assessment incorporate a few generally conservative assumptions (i.e., more likely to overestimate rather than underestimate risk) to address some of the inherent uncertainties. The risk estimates presented in this section are intended to represent conservative estimates of risks due to inhalation exposure within 50 km of each facility. Table G-1 summarizes the risk estimates across all utilities modeled.

More information on the assumptions, models, methods, data, and uncertainties associated with the exposure and risk assessment is provided below. An uncertainty analysis was conducted and the results are used to determine the extent of conservatism in the risk analysis. The uncertainty analysis also helps identify errors of either overestimation or underestimation. Methods, models, data, and assumptions used in the analysis are identified, including their rationale and the effect of reasonable alternative assumptions on the conclusions and estimates. The uncertainty analysis focused on the three HAPs (nickel, arsenic, and chromium) which accounted for over 95 percent of total incidence due to inhalation exposure to HAPs. Since these HAPs accounted for most of the inhalation risk, an analysis of uncertainty on these three is believed to account for

Table G-1. SUMMARY OF RISK BY HAP FOR ELECTRIC UTILITY EMISSIONS

Hazardous Air Pollutant	Carcinogens									Noncarcinogen		
	MEI			Population MIR > 10^{-6}			# Plants MIR > 10^{-6}			HQ_{max}		
	Coal	Oil	Gas	Coal	Oil	Gas	Coal	Oil	Gas	Coal	Oil	Gas
Arsenic	3E-6	1E-5	2E-7	2.4K	2.4K	0	2	2	0	NA	NA	NA
Beryllium	3E-7	8E-7	--	0	0	0	0	0	0	NA	NA	NA
Cadmium	2E-7	2E-6	--	0	45	0	0	1	0	NA	NA	NA
Chromium[1]	2E-6	5E-6	--	107	2.3K	0	1	1	0	NA	NA	NA
Dioxin	5E-8	1E-7	--	0	0	0	0	0	0	NA	NA	NA
Hydrogen Chloride	NA	NA	NA	NA	0	0	NA	NA	NA	0.12	0.005	--
Lead	NA	NA	NA	NA	0	0	NA	NA	NA	0.001	0.0004	1E-7
Manganese	NA	NA	NA	NA	0	0	NA	NA	NA	0.046	0.037	--
Mercury	NA	NA	NA	0	0	0	NA	NA	NA	--	--	--
Nickel[2]	7E-7	9E-5	2E-7	0	1.7M	0	0	20	0	NA	NA	NA
n-Nitrosodimethylamine	8E-7	--	--	0	--	--	0	--	0	NA	NA	NA
Total	4E-6	1E-4	3E-7	NA	NA	NA	2	22	0	NA	NA	NA

MEI = Maximum exposed individual, which is calculated using the *highest concentration*. An individual *may* or *may* not be exposed at that point. This value may be greater than the MIR, which is calculated at the centroid of a census block.

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

NA = Not available.

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.

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

[1] Assumes that 13 percent of total chromium emitted is hexavalent chromium, the species of chromium responsible for carcinogenic potential. Trivalent chromium which would also be present is not thought to have carcinogenic potential.

[2] The nickel emitted is a mixture of various nickel compounds such as soluble nickel. This analysis assumes that all nickel emitted has the same carcinogenic potency as nickel subsulfide. See Chapter 6 for discussion of nickel risks and associated uncertanties.

a majority of the uncertainty in the overall risk estimates due to inhalation exposure.

GES.6 APPROACH TO UNCERTAINTY ANALYSIS

Uncertainty has been classified into four types (parameter uncertainty, model uncertainty, decision-rule uncertainty, and variability).[3] The first two, parameter and model uncertainty, are generally recognized 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, 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 (hence, 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 for the local inhalation risk assessment only. Long-range transport and non-inhalation exposures and risks were not assessed in this uncertainty analysis. Table G-2 briefly summarizes information regarding the parameters used in the inhalation risk estimation process. Model uncertainty was not addressed. However, these uncertainties 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 due to inhalation exposure within 50 km of plants 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

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

Table G-2. Continued

Parameter	Default Option/Assumptions (and departure from default and reason why)	Conservatism	Data (e.g., source, quality)	Uncertainty/Variability (quantitative and qualitative)	Distribution	Judgement/Strategy
DISPERSION AND EXPOSURE MODELING						
Dispersion	Gaussian plume	Unknown	Limited data on other models		Not analyzed	Beyond scope of project. Model is EPA default.
Roughness (rural v. urban)	Population density (within 3 km of plant) is assumed to be an indicator or proxy for setting of the plant, and that urban and rural are representative of surface roughness. Binary choice of urban v. rural	Unknown, may not account for values beyond model defaults	Census data on population within 3 km of facility to indicate urban.	Roughness not binary and not always attributable to population density (e.g., buildings) but other land features as well. •Urban and rural model default settings may not represent the entire range of surface roughness leading to possible extremes not addressed.	Each plant run in both modes	Population may not be an indicator of actual surface roughness. the urban and rural defaults approximate the range of uncertainty.
Terrain	Assumed to be flat terrain. Flat terrain used in gaussian plume dispersion.	Low, as shown by complex terrain analysis	Extensive data on terrain surrounding each facility. However, analysis is very exhaustive and cannot be carried out for all facilities.	Terrain effects can be significant leading to minimal dispersion and high exposure •Terrain is not a binary parameter and the degree of 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. •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. •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. •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 G-2. Continued

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

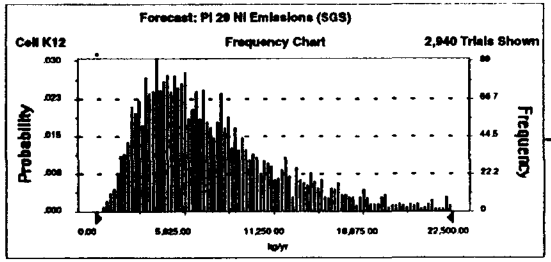
of these variables was described using an appropriate statistic (e.g., mean and standard error of means) or as a probability density function (the relative probability for discrete parameter values). The standard error of the mean (SE) for each parameter was used as the estimate of uncertainty and variability rather than the standard deviation for each parameter. The SE is considered a more appropriate statistic because the available dose-response data are based on long-term average exposures. 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. The SE of a normal distribution underestimates the range of uncertainty for small sample sizes. In this case, the t-distribution is the more appropriate distribution. Also the sample population must be representative of the total population and the underlying variability in the population. To address these concerns, adjustments were made to the observed SE to expand the estimate of uncertainty.

In general, numerical methods (i.e., Monte Carlo simulation) were then used to develop a composite uncertainty distribution by combining the individual distributions. In Monte Carlo simulations, the risk and/or model equations are repeatedly solved using randomly sampled values from the specified distributions to calculate a distribution of estimated risk values. These risk distributions were derived for both estimates of MIR and population risks and can be expressed as a probability density function or cumulative probability density function. The distribution allows the risk assessor to choose the value corresponding to the appropriate percentile in the overall distribution. For example, an exposure level or risk level can be selected which corresponds to the 95th percentile of the overall risk distribution. 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. This yields a hybrid distribution which applies to single "typical" individual from the exposed population.

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 G-1 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 (2 oil-fired and 2 coal-fired plants) were selected which appear to be 4 of the highest risk plants, based on both

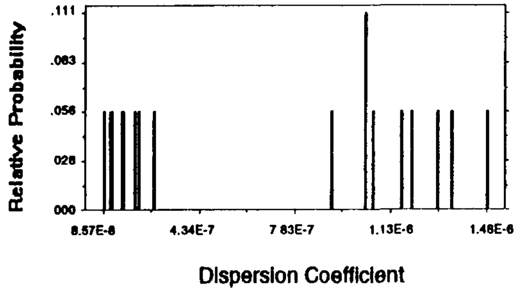
Figure G-1. DEPICTION OF COMBINING COMPONENT UNCERTAINTY DISTRIBUTIONS (i.e., EMISSIONS, DISPERSION, and EXPOSURE-RESPONSE) INTO AN OVERALL DISTRIBUTION ON UNCERTAINTY (e.g., MIR)

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

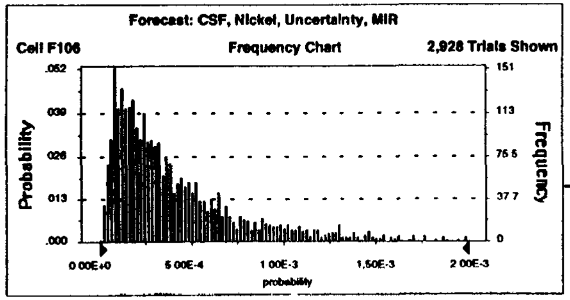


DISPERSION MODULE
Dispersion Coefficients from 18 exposure model runs
These 18 scenarios represent uncertainty in each of
three major parameters affecting dispersions and exposure:
Meteorology, Stack Parameters, and surface roughness
(include all combinations of scenarios for each parameter)

Dispersion Coefficient: Max Conc/(kg/year)

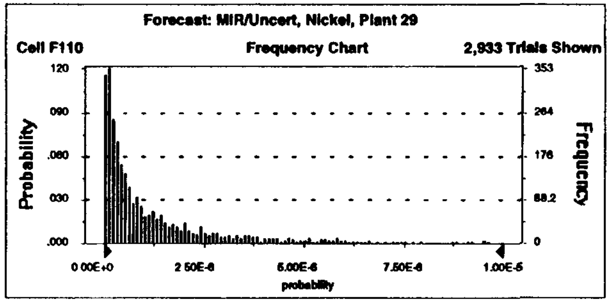


EXPOSURE-RESPONSE MODULE
Distribution of Uncertainty in IURE for MIR

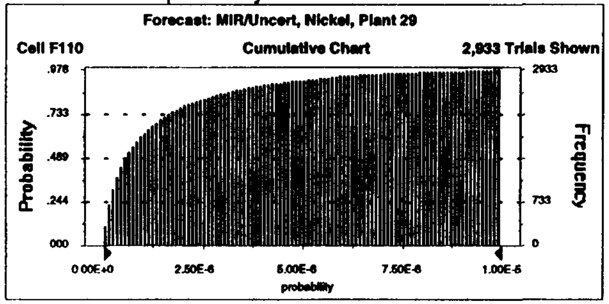


Overall distribution of MIR estimates resuting from combining the three distributions from emissions, dispersion/exposure and exposure-response (CSF) into one distribution.

Distributions can be presented as Probability Density Plots



or as Cumulative probability Plots



the estimated incidence and the maximum individual risk. Each of these plants was analyzed for arsenic, nickel, and chromium. Plant #29 is used for illustration purposes and was selected because it is the oil-fired plant contributing most to cancer incidence.

GES.7 EMISSIONS AND EXPOSURE CHARACTERIZATION

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 and the data used are described in Chapter 3. This program is based on a mass-balance concept—reducing concentrations in the fuel due to the impact of the boiler and control devices. This model was designed to use estimates of emissions from individual plants to derive a nationwide estimate of emissions. The accuracy of the model in predicting emissions for individual plants may vary significantly.

Parameters used in the emissions characterization were: fuel consumption (Coal: tons/yr, Oil: barrels/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 the amount of HAP entering the boiler to that exiting the boiler) and the air pollution control device (APCD), if present, and EMF_a. (APCD-specific factor to account for the amount of HAP entering the APCD to that exiting the APCD)

GES.7.1 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 G-3 and Figure G-2 present the summary statistics and graphical representation, respectively, of the emissions forecast predicted for Plant #29. This distribution gives some indication of the degree of uncertainty and the possible range of emissions estimates which may be experienced. The original emissions estimation program had been designed to be an unbiased estimator of emissions. The results of the uncertainty analysis tend to support this assertion. The original baseline estimates ranged from the 22nd percentile of the overall distribution (for arsenic using the SGS oil concentration data) to the 95th percentile (for chromium using the SGS oil concentration data). This broad range is attributed to the re-analysis of the oil concentration data in how non-detects were evaluated (using the probability plotting

Table G-3. Summary of Results for Monte Carlo Simulation of HAP Emissions (kg/year) from Oil-Fired Plant No. 29

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

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

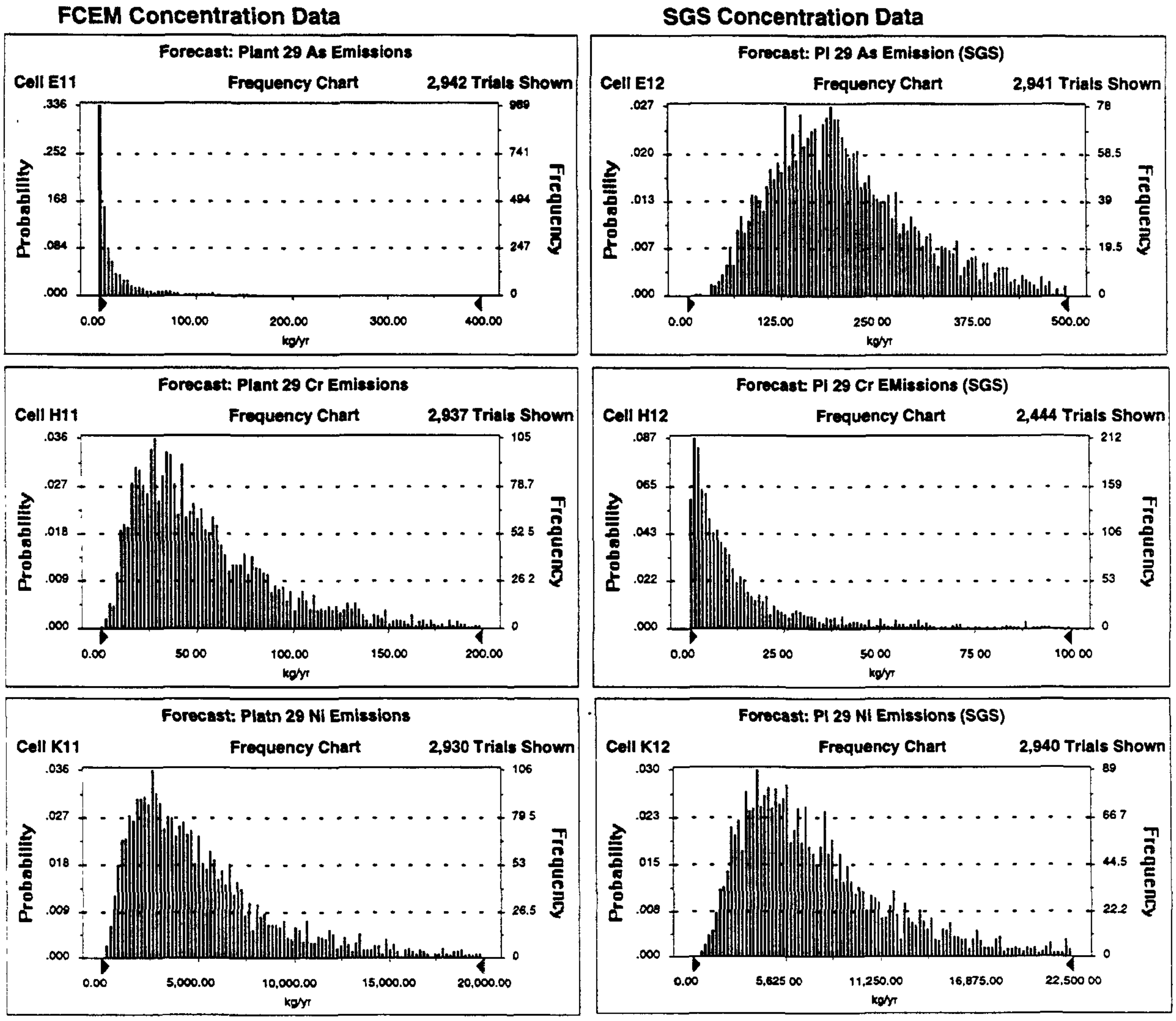
SGS = Subsequent data, trace metal analysis from samples collected for radionuclide analysis.

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

Initial Point Estimate = The estimate of emissions used in the baseline exposure assessment (Chapter 6). This value was based on the average concentration in the FCEM data.

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

Figure G-2. Summary of Results fo Monte Carlo Simulation of HAP Emissions from Oil-Fired Plant#29



routine versus treating them as half the detection limit). When the combined data set (SGS and FCEM oil concentration data) is considered the original baseline emissions estimates were correspond to the 58th percentile for arsenic, 68th percentile for nickel, and the 80th percentile for chromium. Furthermore, the 95th percentile of the simulated range of long-term average emissions estimates are within a factor of two or three of the mean and original baseline estimate and within a factor of less than six within the median.

To compare how estimates differ for the coal-fired plants, Plant #343 was selected for illustration purposes. Plant #343 was the coal-fired plant that contributed most to cancer incidence. The original point estimate of emissions ranged from the 81st to 96th percentile of the range of emissions predicted under the uncertainty analysis. This is significantly higher and may be attributed to the fact that in the original analysis coal from a single state was assumed (defined by the state whose coal was consumed most). However, for this plant the other coals used were generally of lower HAP concentrations which would tend to drive the emissions estimate lower than estimated during the baseline assessment. The 95th percentile on the simulated range of emissions ranged from 0.9 times the baseline estimate for nickel to 2.5 times the baseline estimate for arsenic. As with Plant 29 the 95th percentile of the overall range of predicted emissions typically was within a factor of two or three of the mean estimate and the original baseline estimate.

A preliminary analysis of the emissions estimation model was also conducted. Comparisons were made of test data from 19 utility boiler stacks (17 coal-fired, 2 oil-fired) against predicted emissions for the same plants using the emissions model. For each facility, the emission estimate from the model was divided by the reported value from the corresponding test report. A value of 1 meant that the model exactly predicted the test results, values lower than 1 indicated the model under-predicted emissions, while values higher than 1 indicated the model over-estimated emissions. In general, the results suggested that the model performs as expected (i.e., across a range of boilers and constituents to estimate overall emissions). The average of the ratios across all stacks and constituents was 1.0, while averages for the three HAPs are 1.4, 0.7, and 0.9 for arsenic, chromium, and nickel, 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 are found for individual boilers and constituents. The largest difference for an individual boiler estimated emissions was about 5,000 times lower than reported test results. Furthermore, the model tended to underestimate rather than overestimate emissions about 60 percent of the time within this sample of boilers. A preliminary evaluation of facilities with large differences between projected and actual emissions found that these facilities were likely to burn multiple fuel types. Petroleum coke was found to be burned at the facility with the greatest difference. Therefore, while the model appears to predict fairly well, emissions may be over- or under-estimated at individual facilities. This difference may be due to the combinations of fuel being burned since the model assumes a single fuel type. Furthermore, the plants in which the model underestimated significantly were some of the lowest emitting plants predicted by the model. Using measured rather than modeled emissions for those plants would bring their emissions more in line with the rest of the plants, but they would not become high-risk plants when compared to others. However, it should be noted that the emissions test data used in comparing modeled to measured emissions were based on short-term (and sometimes non-simultaneous) sampling. Whether these data are indicative of long-term emissions from the same stack are not known.

GES.7.2 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. In order to better estimate percentile values above 90 percent, a stochastic (Monte Carlo) approach requires large number (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 versus rural). The uncertainty analysis therefore, focused on the three parameters. The three factors being evaluated are non-linear 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 was summarized for each plant.

GES.8 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 all addressed. As with the uncertainty analysis for emissions and exposure estimates, efforts were limited to the three HAPs-- 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 cancer slope factor (CSF) focused on data, and the use of epidemiologic data (typically from workers) extrapolated to the general population.

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 risk estimations. 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 CSF for these three HAPs may differ among individuals. However, limited data indicate that the CSF differs between smokers and non-smokers and was incorporated in the analysis. Figure G-3 displays the distribution of uncertainty in the CSF for estimating both MIR and annual cancer incidence.

GES.9 RISK CHARACTERIZATION

The distribution of estimates of MIR and incidence for Plant 29 are presented in Table G-4. The results indicate that the baseline risk assessment resulted in conservative estimates of risk. It should be noted that the estimate of MIR is influenced by which of the oil concentration data sets (FCEM or SGS) are used. The original estimates of MIR for Plant 29 generated during the baseline risk assessment ranges from the 71st percentile for arsenic (using the SGS data) to the 98th percentile for chromium (also using the SGS data). Similar results were obtained for the estimates of annual cancer incidence.

For comparison, analysis of MIR and annual cancer incidence was also conducted for Plants 343, 240, and 133. In general, the results support the conclusions that the baseline assessment resulted in reasonably conservative estimates of risk while still being within the range of plausible values. For the coal-fired plants (343 and 240), the point estimates of MIR generated during the baseline assessment were associated with the 91st to 94th percentile on the overall distribution of possible MIR risk estimates. The 95th percentile (a typical high-end risk estimate) of the overall distribution is roughly twice the original risk estimate, four times the mean and within an order of magnitude to the median MIR risk estimate. For oil-fired Plant 133 the results were similar, though the uncertainty was a little more broad. The original estimates of MIR from the baseline assessment were between the 86th and 99th percentile on the estimated overall distribution of possible MIR values.

GES.10 DISCUSSION OF RESULTS

The risk estimation process used in the baseline assessment utilized a combination of parameters each with varying degrees of conservatism (the degree of over-estimation, or under-estimation). In general, the estimates of maximum individual risk (MIR) and annual cancer incidence derived in the initial 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 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

Figure G-3. Distributions of CSFs Used to Estimate MIR and Incidence Considering Uncertainty

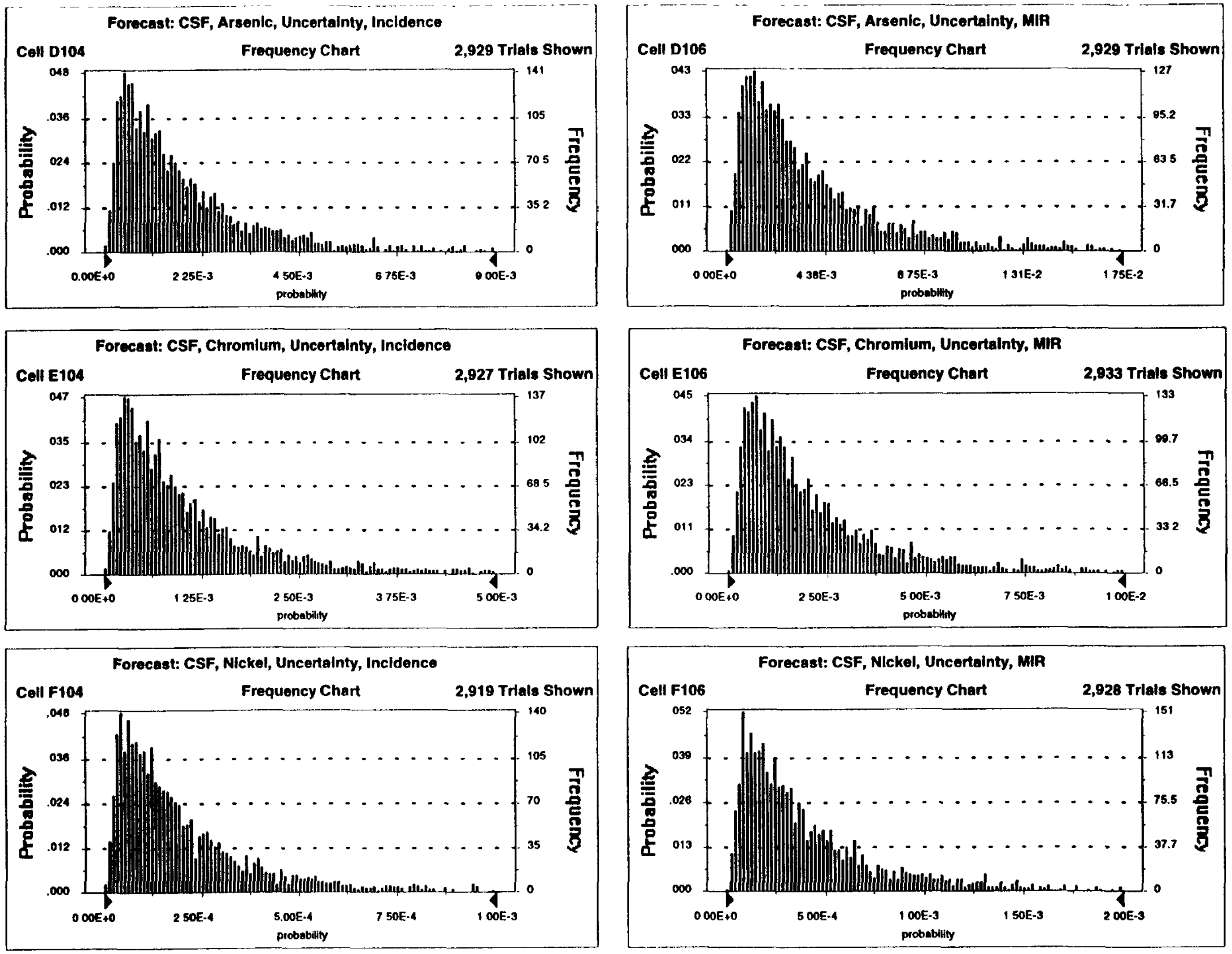


Table G-4. Distribution of MIR and Annual Incidence: Plant #29: Comparison of FCEM and SGS Concentration Data

	MIR, Plant #29 Uncertainty						Annual Incidence Plant 29 Uncertainty					
	Arsenic		Chromium		Nickel		arsenic		chromium		nickel	
	FCEM	SGS	FCEM	SGS	FCEM	SGS	FCEM	SGS	FCEM	SGS	FCEM	SGS
Mean	1E-07	6E-07	1E-07	4E-08	2E-06	3E-06	1E-03	7E-03	1E-03	4E-04	2E-02	3E-02
Initial Point Estimate	6E-07		2E-07		4E-06		8E-03		3E-03		6E-02	
(percentile)	(96)	(71)	(87)	(98)	(90)	(85)	(96)	(75)	(92)	(98)	(95)	(89)
Percentiles:												
0.0%	2E-12	1E-09	2E-10	1E-11	2E-09	6E-09	1E-07	5E-05	4E-06	3E-07	1E-04	1E-04
2.5%	3E-09	8E-09	2E-09	7E-10	2E-08	4E-08	2E-05	3E-04	4E-05	5E-06	1E-03	1E-03
5.0%	6E-09	2E-08	3E-09	1E-09	5E-08	6E-08	4E-05	6E-04	7E-05	1E-05	2E-03	2E-03
10%	1E-08	3E-08	5E-09	3E-09	9E-08	1E-07	8E-05	9E-04	1E-04	2E-05	2E-03	4E-03
25%	3E-08	7E-08	1E-08	7E-09	2E-07	3E-07	2E-04	2E-03	3E-04	5E-05	5E-03	8E-03
50%	6E-08	2E-07	4E-08	1E-08	6E-07	9E-07	4E-04	4E-03	6E-04	1E-04	1E-02	2E-02
75%	8E-08	7E-07	1E-07	2E-08	2E-06	3E-06	8E-04	8E-03	1E-03	3E-04	2E-02	4E-02
90%	2E-07	1E-06	3E-07	8E-08	4E-06	6E-06	2E-03	2E-02	3E-03	9E-04	4E-02	7E-02
95.0%	5E-07	2E-06	5E-07	1E-07	7E-06	1E-05	6E-03	2E-02	4E-03	1E-03	6E-02	1E-01
97.5%	1E-06	4E-06	7E-07	2E-07	1E-05	2E-05	1E-02	3E-02	6E-03	3E-03	9E-02	1E-01
100.0%	3E-05	2E-05	6E-06	6E-06	9E-05	7E-05	2E-01	2E-01	3E-02	3E-02	4E-01	9E-01
Ratio												
95th : mean	0.8	4.1	2.3	0.7	1.7	2.5	0.8	3.0	1.4	0.5	1.0	1.6
95th : median	8.7	10.6	12.8	9.6	11.1	10.4	14.6	5.8	7.1	13.0	5.8	5.8
95th : baseline	3.5	3.8	4.2	3.9	4.0	3.8	4.0	3.4	3.6	3.8	3.3	3.3

FCEM = Original oil concentration data, distribution defined by probability plotting technique.

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

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

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

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

deposition fraction, retention time, and exposure frequency also contributed significantly in the variability of these estimates.

These results indicate that the EPA baseline risk estimates are generally conservative estimates of risks due to inhalation esposure to HAP emissions within 50 km of plants. However, it is important to note again that this uncertainty analysis did not include an evaluation of uncertainty associated with long-range transport or multipathway exposures. This conservatism is usually appropriate given EPA's mandate of public health protection. Often there is a concern that the use of several conservative assumptions results in risk estimates which are unrealistic and beyond the range of possible risks (i.e., overly conservative). The results of the uncertainty analysis indicate that the baseline risk estimates are reasonably conservative (generally around the 90th or 95th percentile). The uncertainty analysis supports the baseline risk estimates as they are within the range of predicted risks.

However, it should be noted that this analysis has focused only on parameter uncertainty. As a result, the uncertainty presented here may underestimate the overall uncertainty. The uncertainty within those parameters evaluated may also have been constrained by the limits of the models. For example, the effect of surface roughness on dispersion is addressed by the choice of the rural or urban setting within the HEM model. However, the binary choice of urban versus rural does not fully encompass the range of surface roughness. The range of surface roughness found in the environment is broader than the range represented by the urban and rural default settings within the model. Likewise, the analysis of uncertainty in the exposure-response relationship did not consider the uncertainty related to alternative dose-extrapolation models (e.g., Weibull versus linearized multi-stage) which have been shown to differ by several orders of magnitude.

G.1 INTRODUCTION

This appendix presents the uncertainty analysis EPA has conducted on the estimates for human health risks from direct inhalation of electric utility HAP emissions within 50 km of plants. The general principles of uncertainty analysis are presented in Section G.1.1. Section G.2 of the report summarizes the risk estimates from the EPA study of direct inhalation of HAP emissions. Sections G.3 through G.5 provide the detailed uncertainty analysis for the exposure assessment, exposure-response (dose-response) assessment, and risk characterization portions of the assessment, respectively. Sections G.3 through G.5 also describe and evaluate the models, data, and methods used to conduct the analysis and their attendant uncertainties. Where possible, quantitative estimates of uncertainty have been generated. In the absence of quantitative information, a qualitative description is given.

G.1.1 The Need For Uncertainty Analysis

Unlike the other components of risk assessment, risk characterization did not have its own specific guidelines in the original 1986 EPA risk assessment guidelines. The general content of risk characterization was defined by the National Academy of Science[4] and, to a limited degree, in each EPA *Risk Assessment Guideline*.[5] However, much was left to the professional judgment of those involved in risk assessment preparation. As a result, a high degree of variability in how risk characterization has been practiced has developed.[1] Although a great deal of careful analysis and scientific judgment goes into the development of EPA risk assessments, significant information is often omitted as the results of the risk assessment are passed along to the decision-makers. Often, when risk information is presented to the ultimate decision maker and to the public, the results have been reduced to a single numerical risk value. Such an approach may not often provide sufficient information to the decision maker.

G.1.1.1 RAC Guidance on Risk Assessment. To address these concerns, the EPA's Risk Assessment Council (RAC), composed of risk assessors and risk managers from within the Agency, evaluated EPA risk assessment practices. The RAC, after careful evaluation, recommended guidance on risk assessment focusing on the risk assessment-risk management interface, risk characterization, and exposure and risk descriptors. The major elements of the guidance are summarized below.

The RAC reached several conclusions in their evaluation of EPA risk assessment practices including:

- A full and complete presentation of risk is needed, including a statement of confidence about data and methods used to develop the assessment.

- A basis for greater consistency and comparability should be developed.

- Professional judgment plays a necessary and important role in the overall statement of risk.

As practiced at EPA, the risk assessment process depends on many different kinds of scientific data (e.g., exposure, toxicity, epidemiologic), all of which are used to "characterize" the expected risk to human health or the environment. Informed use of reliable scientific data from many different sources is a central feature of the risk assessment process.

Two elements are required for full characterization of risk. First, the characterization must address qualitative and quantitative features of the assessment. That is, along with quantitative estimates of risk, full risk characterization must clearly identify all assumptions, their rationale, and the effect of reasonable alternative assumptions on the conclusions and estimates. Second, it must identify any important uncertainties in the assessment as part of a discussion on confidence in the assessment. This statement on the confidence of the assessment must identify all major uncertainties and comment on their influence in the assessment. The uncertainty statement is important for several reasons:

- Information from various sources carries different kinds of uncertainty, and knowledge of these differences is important when uncertainties are combined to characterize risk.

- Decisions must be made about expending resources to acquire additional information to reduce uncertainties.

- A clear and explicit statement of the implications and limitations of a risk assessment requires a clear and explicit statement of related uncertainties.

- Uncertainty analysis gives the decision-maker a better understanding of the implications and limitations of the assessments.

G.1.1.2 NRC Committee Report: *Science and Judgement in Risk Assessment*. Section 112(o) of the Act, required the EPA to commission a study by the National Research Council[2] on risk assessment methods used by the EPA as related to HAPs.

Recommendations developed by the NRC relevant for this uncertainty analysis include the following:

- Conduct formal uncertainty analyses, to show where additional research might resolve major uncertainties.
- Consider the limits of scientific knowledge, the remaining uncertainty, and the desire to identify errors of either overestimation or underestimation of risk.
- Consider uncertainties when ranking risks, rather than ranking risk based solely on the point estimate.

G.1.2 Background Information on Uncertainty

Uncertainty can be introduced into a risk assessment at every step in the process. It occurs because risk assessment is a complex process, requiring the integration of:

- The fate and transport of pollutants in a variable environment by processes that are often poorly understood or too complex to quantify accurately
- The potential for adverse health effects in humans as extrapolated from animal bioassays
- The probability of adverse effects in a human population that is highly variable genetically, in age, in activity level, and in life style.

Even using the most accurate data with the most sophisticated models, uncertainty is inherent in the process.

Finkel[6] classified all uncertainty into four types (parameter uncertainty, model uncertainty, decision-rule uncertainty, and variability), summarized in Table G-5. The first two, parameter uncertainty and model uncertainty, are generally recognized as major sources of uncertainty.

Parameter uncertainty occurs when parameters appearing in equations cannot be measured precisely and/or accurately either because of equipment limitations or because the quantity being measured varies spatially or temporally. Random, or sample errors, are a common source of parameter uncertainty that is especially critical for small sample sizes. More difficult to

TABLE G-5. SOURCES OF UNCERTAINTY IN RISK ASSESSMENT[a]

General Type	Specific Source of Uncertainty	Comments/Examples
Parameter uncertainty	Measurement errors	Include limitations of equipment, methodology, and human error Some processes impossible to measure exactly
	Random errors	Sampling errors Can be minimized by increasing sample size
	Systematic errors	Nonrandom errors Result of inherent flaw in data gathering processes Minimize by external peer review
Model uncertainty	Surrogate variables	Use of animal bioassays to determine effect on humans, for example
	Excluded variables	May result from model simplification or failure to recognize an important variable
	Abnormal conditions	Failure to recognize importance of episodic meteorological events, for example
	Incorrect model form	Choice of dose-response model for carcinogens, for example
Decision-rule uncertainty		More important for risk management, but need to recognize that value judgments affect choice of model and interpretation of results
Variability		Those important for health risk assessment include environmental factors, genetic variability, and lifestyle differences Even if variability is known (therefore, not in itself uncertain) it still contributes to overall uncertainty of the risk assessment

[a] Adapted from Finkel, 1990.

recognize are nonrandom or systematic errors that result from bias in sampling, experimental design, or choice of assumptions.

Model uncertainty is associated with all models used in risk assessment. These include the animal models used as surrogates for human carcinogenicity, dose-response models used in extrapolations, and the computer models used to predict the fate and transport of chemicals in the environment. The use of rodents as surrogates for humans introduces uncertainty into the risk factor since there is considerable interspecies variability in sensitivity. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in models either because of increased complexity or a lack of data on that parameter. The risk assessor needs to consider the importance of excluded variables on a case-by-case basis, because a given variable may be important in some instances and not in others. A similar problem can occur when a model that is applicable under average conditions is applied under conditions that differ from the average. Large bodies of water, for example, can cause meteorological conditions that are not adequately modeled by air dispersion models such as the Industrial Source Complex - Long Term (ISCLT) used in this analysis. Choosing the correct model form is often difficult because conflicting theories seem to explain a phenomenon equally well.

The third type, decision-rule uncertainty, is probably of more concern to risk managers. This type of uncertainty arises, for example out of the need to balance different social concerns when determining an acceptable level of risk. Finkel provides a complete discussion of decision-rule uncertainty.[3]

Variability, the fourth source of uncertainty, is often used interchangeably with the term "uncertainty," but this is not strictly correct. Variability may be tied to variations in physical and biological processes and cannot be reduced with additional research or information, although it may be known with greater certainty (e.g., age distribution of a population may be known and represented by the mean age and its standard deviation). "Uncertainty" is a description of the imperfection in knowledge of the true value of a particular parameter or its real variability in an individual or a group. In general, uncertainty is reducible by additional information-gathering or analysis activities (better data, better models), whereas real variability will not change (although it may be more accurately known) as a result of better or more extensive measurements.[7] Variability will introduce uncertainty if risks are calculated for an individual drawn at random within a population.

The degree to which all types of uncertainty need to be quantified and the amount of uncertainty that is acceptable varies with the intent of the analysis. For a screening level analysis, a high degree of uncertainty is often acceptable, provided that conservative assumptions are used to bias potential error toward protecting human health. A region-wide or nationwide study will be less uncertain than a site-specific one in determining average risks since, in the former case, it may be possible to use the average of a parameter value over many sites (which often can be estimated better than a site-specific value). However, the general analysis may be highly uncertain in defining the range of possible risks, which are influenced by site-specific conditions. In general, the more detailed or accurate the risk characterization, the more carefully uncertainty needs to be considered.

G.1.2.1 Qualitative Description of Uncertainty. Often, the sources of uncertainty can be determined, but they cannot be quantified. This can occur when a factor is known or expected to be variable, but no data are available (e.g., the amount of time people at a specific site spend out of doors). In this case, sometimes default data are available that can be useful for estimating a possible range of values. Uncertainty often arises out of a complete lack of data. A process may be so poorly understood that the uncertainty cannot be quantified with any confidence. In addition, some sources of uncertainty (such as uncertainty in theories used to deduce models) are inherently qualifications reflecting subjective modes of confidence rather than probabilistic arguments. When uncertainty can only be presented qualitatively, the possible direction and orders of magnitude of the potential error should be considered.

G.1.2.2 Quantitative Description of Uncertainty. Knowledge of experimental or measurement errors can also be used to introduce a degree of quantitative information into a qualitative presentation of uncertainty. For example, standard laboratory procedures or field sampling methods may have a known error level that can be used to quantify uncertainty. In many cases, the uncertainty associated with particular parameter values or for the estimated risks can be expressed quantitatively. A six-step process for producing a quantitative uncertainty estimate has been identified as follows[6]:

- Define the measure of risk (e.g., deaths, life-years lost, maximum individual risk (MIR), population above an "unacceptable" level of risk). More than one measure of risk may result from a particular risk assessment; however, the uncertainty should be quantified for each individually.

- Specify "risk equations" that present the mathematical relationships that express the risk measure in terms of its components. This step is used to identify the important parameters in the risk estimation process.

- Generate an uncertainty distribution for each parameter or equation component. These uncertainty distributions may be generated by the use of analogy, statistical inference techniques, or elicitation of expert opinion, or some combination of these.

- Combine the individual distributions into a composite uncertainty distribution. Monte Carlo simulation, frequently used for this step, is discussed in greater detail later in this section, and was used in this analysis.

- Recalibrate the uncertainty distributions. Inferential analysis could be used to "tighten" or "broaden" particular distributions to account for dependencies among the variables and/or to truncate the distributions to exclude extreme values.
- The output should be summarized in a manner that is clear and highlights the important risk management implications. Specific factors should be addressed including: the implication of supplanting a point estimate produced without considering uncertainty, the balance of the costs of under- or overestimating risks, unresolved scientific controversies, and implications for research.

When a detailed quantitative treatment of uncertainty is required, statistical methods are employed. Two approaches to a statistical treatment of uncertainty with regard to parameter values are described here and were used in this analysis where appropriate. The first is simply to express all variables for which uncertainty is a major concern using an appropriate statistic. For example, if a value used is from a sample (e.g., yearly emissions from a stack), both the mean and standard deviation should be presented. If the sample size is very small, it may be appropriate to give the range of sample values and use a midpoint as a best estimate in the model; or, both the smallest and largest measured value could be used to get two estimates that bound the expected true value. The appropriate statistic to use depends on the amount of data available and the degree of detail required. Uncertainties can be propagated using analytical or numerical methods.

A second approach is to use the probability distributions of major variables to propagate parameter value uncertainties through the equations used in a risk analysis. A probability distribution of expected values is developed for each parameter value. These probability distributions are typically expressed as either probability density functions (PDF) or as cumulative probability density functions (CPF). The PDF presents the relative probability for discrete parameter values while the CPF presents the cumulative probability that a value is less than or equal to a specific value. The PDF approach was used here to represent a propagate parameter uncertainty.

Uncertainties are propagated by developing a composite uncertainty distribution by combining the individual distributions with the equations used to calculate probability of cancer. Numerical methods are often employed for this phase, with Monte Carlo simulations gaining wide acceptance for this purpose. In Monte Carlo simulations, a computer program (e.g., Crystal Ball) is used to repeatedly solve the model equations under different selections of parameter values to calculate a distribution of exposure (or risk) values. Each time the equations are calculated, values are randomly sampled from the specified distributions for each parameter. The end result is a distribution of exposure (or risk). These can again be expressed as PDFs or, more appropriately, as CPFs. The distribution allows the risk assessor to choose the value corresponding to the appropriate percentile in the overall distribution. For example, an exposure level or risk level can be selected that corresponds to the 95th percentile of the overall risk distribution rather than relying on a point estimate of risk based on the 95th percentile values for each parameter. This allows the risk analyst to reflect quantitatively the confidence of that risk estimate with respect to the range of possible risks.

G.1.3 Approach to Uncertainty Analysis

This uncertainty analysis addresses only the cancer risks associated with direct inhalation of HAPs emitted from electric utilities and focused on parameter uncertainty within the models and available data. Model uncertainty was not addressed quantitatively due to limits in time and resources 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. Variability was not specifically addressed for emissions and dispersion modeling because the available data did not permit this distinction. The data presented to EPA was in different forms, some raw data while others limited to summary reports. As a result, it was not possible to identify inherent variability and the impact of sampling and analysis errors.

The overall goal of this uncertainty analysis is to estimate the range of possible estimates due to direct inhalation of nonradionuclide HAPs emitted from utilities only. Risks from indirect exposure, radionuclides, and long-range transport are addressed not addressed in this appendix. This analysis focused on parameter uncertainty and, where possible, 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 but 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 these all variables was described using an appropriate statistic (e.g., mean and standard error of means) or as a probability density function (their relative probability for discrete parameter values). The characterization of cancer risks from direct inhalation is estimated based on the long-term average concentration and exposures. Therefore, in most cases the standard error of the mean (SE) for each parameter was used as the estimate of uncertainty and variability rather than the standard deviation (SD). The SE is a more appropriate statistic given that the risk assessment is concerned with long-term average exposures and intermittent extreme values of exposure levels are not as important. However, directly using the SE from the sample population would tendency be an overconfident estimate of (i.e. too narrow a range) of uncertainty due to:

- The SE is used as an estimate of the SD of a normal distribution while the t-distribution is used to represent the sampling distribution. For small sample sizes, the t-distribution is significantly different, and approaches a normal distribution as samples sizes increase (n~30). Therefore, error is introduced by assuming a normal distribution.

- The mean of a distribution represents a long-term average only when the population samples are representative of values that would be obtained over the long-term. When data are obtained from a convenience sample, additional error is introduced due to potential nonrepresentativeness of the sample.

- The mean of a distribution represent a long-term average only if the population sample represents true variability. Measurement error must also be considered. Generally the most appropriate estimate of uncertainty in the long-term average is to use the larger of the SE or the SD due to measurement error.

Therefore, adjustments to the SE have been made to more accurately reflect uncertainty. The procedure used is described in Section G.3.1.

In general, once uncertainty was defined for each parameter, numerical methods (i.e., Monte Carlo simulation) were then used to develop a composite uncertainty distribution by combining the individual distributions. The resulting distributions provides an estimate of the range of values which can be expected and allows the risk assessor to choose the value corresponding to the appropriate percentile in the overall distribution, or to determine where the point estimate lies in the overall distribution. For example, an exposure level or risk level can be selected which corresponds to the 95th percentile of the overall risk distribution rather than relying on a point estimate of risk based on the 95th percentile values for each parameter. Since variability was not differentiated in many cases from uncertainty, they were simulated together in a one-dimensional Monte Carlo simulation yielding a hybrid distribution. This hybrid distribution applies to a single individual taken at random from the exposed population. If either variability or uncertainty dominates the analysis, then this one-dimensional approach may also be used to characterize wether a specified percentile of the population or a specified upper confidence level, respectively.

Ideally, if data and resources permitted, variability and uncertainty would be characterized separately and a two-dimensional simulation could be conducted.[8] The distributions from a two-dimensional simulation would then allow a multi-dimensional characterization of individual risks -- X level of risk (e.g., 10^{-5}) for the Yth percentile of the population (addressing variability) with Z degree of confidence (addressing uncertainty). However, the available data and resources did not allow for this approach.

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. A detailed uncertainty analysis could not be conducted on all of the electric utility plants or HAPs included in the original analysis. Therefore, a total of four plants (2 oil-fired and 2 coal-fired plants) were selected which contribute most to risk, based on both the estimated incidence and the maximum individual risk. The highest risk oil-fired and coal-fired plant were used for illustration purposes in the text. Each of these plants was analyzed for arsenic, nickel, and chromium which were shown to account for a majority of the risk (which is explained in greater detail in Section G.2).

G.2 QUANTITATIVE BASELINE ESTIMATES OF RISKS FROM DIRECT INHALATION OF ELECTRIC UTILITY HAP EMISSIONS

This section summarizes the risk estimates that were generated as part of the EPA risk assessment of direct inhalation exposures to electric utility hazardous air pollutant (HAP) emissions. The risk assessment is described in detail in

Chapters 4 to 6. The risk estimates are summarized here as a foundation and reference point for the uncertainty analysis. They are referred to throughout this document as the "baseline". Three measures of risk were estimated under the baseline case: individual risk, population risk, and noncancer risk.

G.2.1 Risks Associated With Coal-fired Electric Utility Boilers

A total of 426 coal-fired were modeled using the HEM, and 1990 emissions and population data. The HEM estimated the ambient HAP concentrations within 50 km of the plant, the population exposed, and the distribution of exposure within the population. Table G-6 summarizes the MEI risks, the number of persons exposed above individual cancer risk levels of 10^{-7} and 10^{-6}, the number of plants whose emissions result in those risk levels, and the maximum HQ for HAPs evaluated.

G.2.1.1 Individual Cancer Risk. The MEI risk was highest for arsenic (Class A, human carcinogen) at 3×10^{-6}. Table G-6 also shows that arsenic emissions from 42 plants resulted in MIRs of greater than or equal to 10^{-7}.

G.2.1.2 Population Cancer Risk. As with the MIR, arsenic and chromium are the major contributors to the total population exposed to risk levels of one in 1 million (1×10^{-6}) or more. Approximately 850 people are estimated to have a risk of 1×10^{-6} or greater from exposure to arsenic, and about 100 people have a comparable risk from exposure to chromium. The HEM also calculated the annual incidence of cancer expected for each of the chemicals. The total cancer incidence from all HAPs was estimated as 0.09 cases per year. As shown in Figure G-4, arsenic and chromium are again the major contributors and account for almost 90 percent of the estimated cancer incidence.

G.2.1.3 Noncancer Risk. The maximum HQ estimated for noncarcinogenic HAPs emitted from coal-fired plants was 0.12 for hydrogen chloride. HQ values for all other HAPs were lower.

G.2.2 Risks Associated With Oil-fired Electric Utilities

A total of 137 oil-fired plants with boilers of 25 MW or were evaluated using 1990 HAP emissions and population data. Table G-7 summarizes some of the risk values for oil-fired plants.

G.2.2.1 Individual Cancer Risk. However, there are major uncertainties in these estimates because of the speciation of nickel emissions from oil-fired utilities. In this analysis, as a conservative assumption (i.e., more likely to overestimate rather than underestimate risk), all nickel was assumed to be

Table G-6. SUMMARY OF RISK BY HAP: COAL-FIRED POWER PLANTS

Pollutant	Carcinogens					Noncarcinogen
	MEI	Population MIR > 10^{-7}	# Plants MIR > 10^{-7}	Population MIR > 10^{-6}	# Plants MIR > 10^{-6}	HQ_{max}
Arsenic	3×10^{-6}	1.7M	42	2,370	2	NA
Beryllium	3×10^{-7}	1,280	2	0	0	NA
Cadmium	2×10^{-7}	107	1	0	0	NA
Chromium[a]	2×10^{-6}	80,500	10	107	1	NA
Dioxin/furans	5×10^{-8}	0	0	0	0	NA
Hydrogen chloride	NA	NA	NA	NA	NA	0.12
Lead	NA	NA	NA	NA	NA	0.001
Manganese	NA	NA	NA	NA	NA	0.046
Mercury	NA	NA	NA	0	0	0.002
Nickel[b]	7×10^{-7}	5,100	3	0	0	NA
n-Nitrosodimethylamine	8×10^{-7}	9,150	4	0	0	NA
Total	4×10^{-6}	NA	42	NA	2	NA

MEI = Maximum exposed individual, calculated using the highest concentration. An individual may or may not be exposed at that point. This value may be greater than the MIR, which is calculated at the centroid of a census block.

MIR = Maximum Individual Risk, highest risk identified at the centroid of a census tract to which a population is assigned.

NA = Not available.

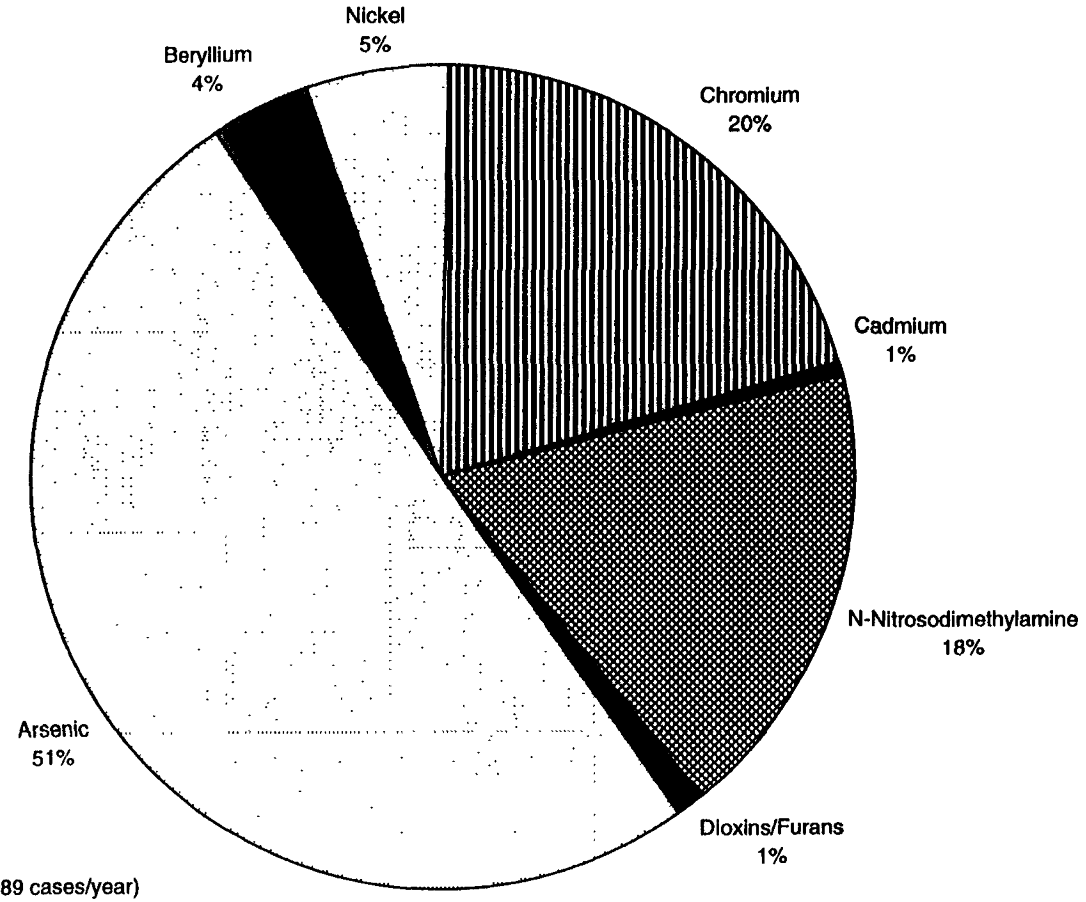
HQ = Hazard quotient, the ratio of exposure concentration to the Reference Concentration (RfC). HQ values below 1 are not expected to result in adverse effects.

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

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

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

Figure G-4. Percent Distribution of Annual Cancer Incidence Resulting from Coal-Fired Utilities



(Total Incidence = 0.089 cases/year)

These results do not include risks due to long-range transport or indirect exposure.

NOTE: These results are based on direct inhalation exposure modeling within 50 km of each facility, using the assumptions and data explained in the text. There are uncertainties associated with these numbers, also discussed in the text.

Table G-7. SUMMARY OF RISK BY HAP: OIL-FIRED POWER PLANTS

	Carcinogens					Noncarcinogen
Pollutant	MEI	Population MIR > 10^{-7}	# Plants MIR > 10^{-7}	Population MIR > 10^{-6}	# Plants MIR > 10^{-6}	HQ_{max}
Arsenic	1 x 10^{-5}	2.1M	22	2370	2	NA
Beryllium	8 x 10^{-7}	2,280	2	0	0	NA
Cadmium	2 x 10^{-6}	3,040	2	45	1	NA
Chromium	5 x 10^{-6}	257,000	10	2,280	1	NA
Dioxin/furans	1 x 10^{-7}	45	1	0	0	NA
Hydrogen chloride	NA	NA	NA	0	0	0.0055
Lead	NA	NA	NA	0	0	0.0004
Manganese	NA	NA	NA	0	0	0.037
Mercury	NA	NA	NA	0	0	NA
Nickel[b]	1 x 10^{-4}	73M	79	1.65M	20	NA
Total	1 x 10^{-4}	NA	85	NA	22	NA

MEI = Maximum exposed individual, calculated using the highest concentration. An individual may or may not be exposed at that point. This value may be greater than the MIR, which is calculated at the centroid of a census block.

MIR = Maximum Individual Risk, highest risk identified at the centroid of a census tract to which a population is assigned.

NA = Not available.

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.

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

[a] Assumes that 18 percent of total chromium emitted is hexavalent chromium, the species of chromium responsible for carcinogenic potential. Trivalent chromium, which would also be present, is not thought to have carcinogenic potential.

[b] This analysis conservatively assumes that all nickel emitted from electric utilities has the same carcinogenic potency as nickel subsulfide (the highest potency of nickel compounds tested). However, the nickel emitted is a mixture of various nickel compounds such as soluble nickel. Emissions tests have found nickel subsulfide to be present as less than 10 percent of total nickel emitted. Most nickel compounds are assumed to have carcinogenic potential though the potency is not known. If the relative potency (URE) of the mixture of nickel compounds emitted from oil-fired utilities was 50 percent of nickel subsulfide, about 23 million persons would be exposed at an MIR > 10^{-7}, and 100,000 at an MIR > 10^{-6}, if the URE were 20 percent of nickel subsulfide, about 7.5 million persons would be exposed at an MIR > 10^{-7}, and 9,930 at an MIR > 10^{-6}.

equipotent to nickel subsulfide, which has the highest cancer potency of nickel compounds evaluated by EPA. The limited speciation data indicate that less than 10 percent of all nickel emissions may be nickel subsulfide. The remainder of the nickel is a combination of various nickel compounds for which the EPA has not yet determined the carcinogenic potency.

There were three oil-fired plants with risks exceeding one in 100,000 (1 x 10^{-5}) due to nickel exposures and one plant that exceeded this level due to arsenic exposure. Nickel was responsible for MIRs that exceeded one in 10,000,000 (1 x 10^{-7}) at 79 plants and exceeded 10^{-6} at 20 plants.

G.2.2.2 Population Risk. As with the MIR, nickel and arsenic are the major contributors to the total population exposed to risk levels of one in 1,000,000 (1 x 10^{-6}) or more.

The total cancer incidence associated with HAP emissions from the 137 oil-fired plants was estimated as 0.47 cases per year. As shown in Figure G-5, nickel accounts for about 86 percent of the total annual incidence.

G.2.2.3 Noncancer Risks. The highest reported HQ resulting from oil-fired power plant emissions was 0.055 for manganese. The HQs for other HAPs and other plants were lower.

G.2.3 Risks From Gas-fired Plants

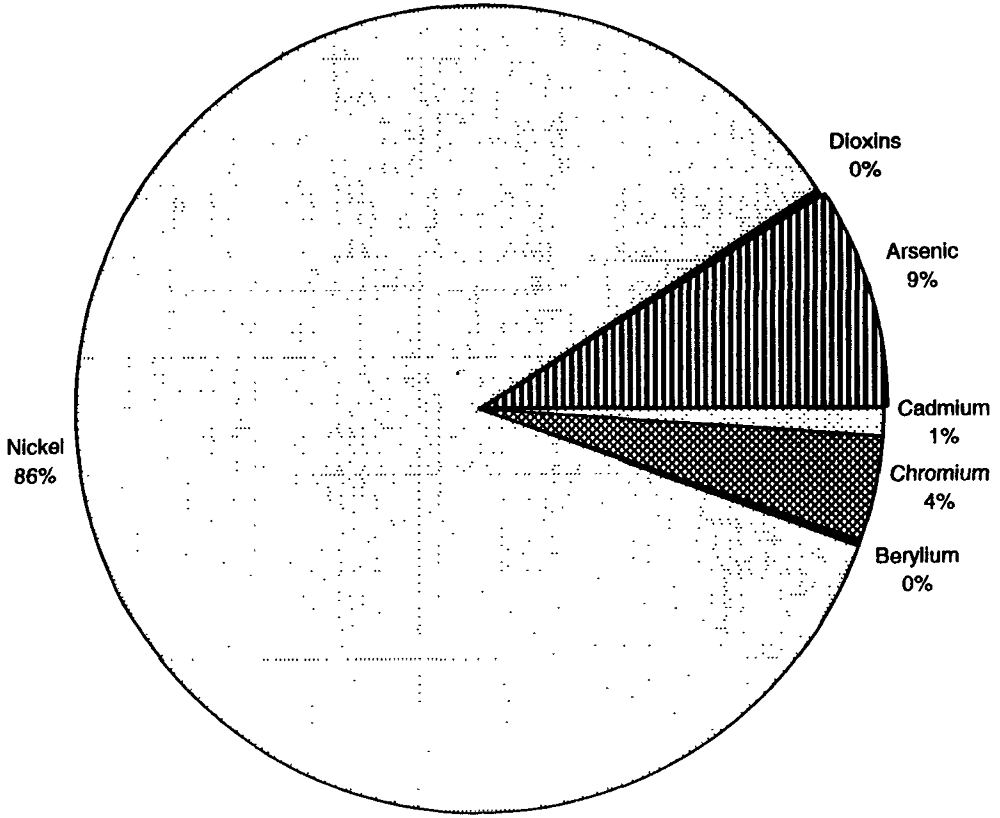
Risks were estimated for 1990 HAP emissions from 267 gas-fired facilities. Table G-8 summarizes the results. HAP emissions from only one plant resulted in risks greater than one in 10 million (10^{-7}) with only 23 persons exposed above that level. For noncarcinogens, the maximum HQ estimated was 1.32 x 10^{-7}.

G.2.4 Risks from All Electric Utilities

G.2.4.1 Individual Cancer Risk. HAP emissions from oil- and coal-fired facilities are the major contributors to risk; gas-fired utilities have been shown to present lower risks. As expected from the previous discussion, nickel, arsenic, and chromium are the major contributors to the MIR.

G.2.4.2 Population Cancer Risk. Table G-9 summarizes the population exposed to the major carcinogens at various risk levels for coal-fired and oil-fired facilities. The table shows that arsenic and chromium contribute the most to the risk from coal-fired plants compared to nickel and arsenic for oil-fired plants.

Figure G-5. Percent Distribution of Annual Cancer Incidence Resulting from HAP Emisions from Oil-Fired Electric Utilities



(Total incidence = 0.47 cases/year)

These results do not include risks due to long-range transport or indirect exposure.

NOTE: These results are based on direct inhalation exposure modeling using the assumptions and data explained in the text. There are uncertainties associated with these numbers, also discussed in the text.

TABLE G-8. SUMMARY OF RISK BY HAP: GAS-FIRED POWER PLANTS

Pollutant	Carcinogens					Noncarcinogen
	MEI	Population MIR > 10^{-7}	# Plants MIR > 10^{-7}	Population MIR > 10^{-6}	# Plants MIR > 10^{-6}	HQ_{max}
Arsenic	2 x 10^{-7}	0	0	0	0	NA
Lead	NA	NA	NA	NA	NA	1.32 x 10^{-7}
Mercury	NA	NA	NA	NA	NA	NA
Nickel[a]	2 x 10^{-7}	23	1	0	0	NA

MEI = Maximum exposed individual, calculated using the highest concentration. An individual may or may not be exposed at that point. This value may be greater than the MIR, which is calculated at the centroid of a census block.

MIR = Maximum Individual Risk, highest risk identified at the centroid of a census tract to which a population is assigned.

NA = Not available.

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.

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

Table G-9. SUMMARY OF ESTIMATED POPULATION EXPOSED AT VARIOUS LEVELS OF RISK OR GREATER

Number of Persons Exposed at Various Risk Levels or Greater by HAP							
COAL-FIRED PLANTS							
Risk Level	Arsenic	Chromium	n-Nitroso	Nickel	Beryllium	Cadmium	Dioxins/ furans
5E-6	0	0	0	0	0	0	0
2.5E-6	0	0	0	0	0	0	0
1E-6	852	107	0	0	0	0	0
5E-7	5,990	2,160	399	0	0	0	0
2.5E-7	88,800	8,630	585	947	0	0	0
1E-7	1,710,000	80,500	9,150	5,100	1,280	107	0
OIL-FIRED PLANTS							
Risk Level	Nickel	Arsenic	Chromium	Cadmium	Beryllium	Dioxins/ furans	
5.E-5	89	0	0	0	0	0	
2.5E-5	2,240	0	0	0	0	0	
1E-5	2,310	45	0	0	0	0	

Table G-9. Continued

OIL-FIRED PLANTS							
Risk Level	Nickel	Arsenic	Chromium	Cadmium	Beryllium	Dioxins/ furans	
5E-6	9,930	89	45	0	0	0	
2.5E-6	100,000	2,280	89	0	0	0	
1E-6	1,650,000	2,370	2,280	45	0	0	
5E-7	7,460,000	32,600	2,280	89	45	0	
2.5E-7	23,100,000	287,000	9,490	2,280	89	0	
1E-7	73,300,000	2,140,000	257,000	3,040	2,280	45	

Note: There may be multiple counting of population around facilities within 50 km of each other. Exposed individuals would be included in the statistics for each plant within 50

Approximately 0.56 cancer cases per year are estimated to result from HAP emissions dispersed within 50 km utilities. Figure G-6 shows the contribution of each compound to the incidence. When all of the plants are combined, nickel, arsenic, and chromium account for over 95 percent of the incidence.

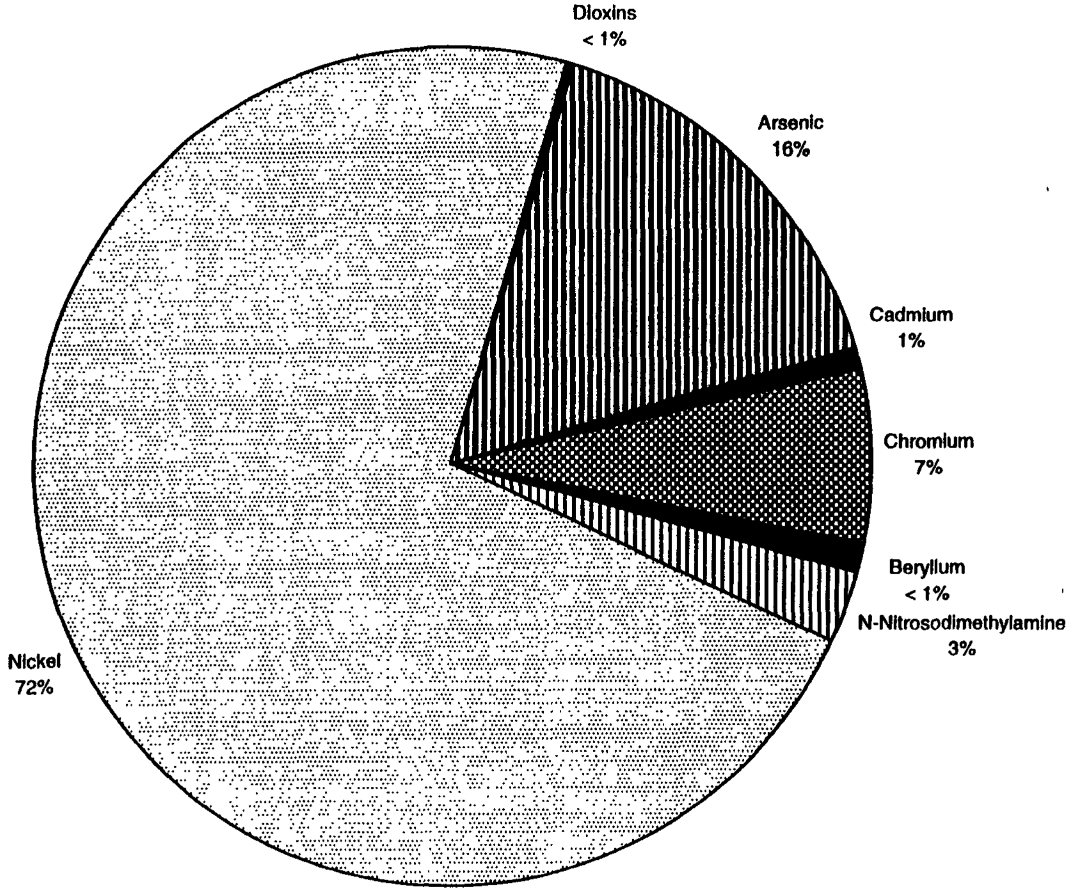
G.2.4.3 Noncancer Risks. Emissions of noncarcinogenic HAPs from utilities did not result in an estimated HAP air concentration that exceeded the RfC. The highest HQ was 0.12 for hydrogen chloride from a coal-fired plant.

G.2.5 Discussion

Direct inhalation exposure to HAPs emitted from a total of 178 utilities result in individual risk of 1 in 1,000,000 (10^{-7} or greater); exposures to emissions from 24 plants are estimated to result in individual risks of 1 in 10,000,000 (10^{-7} or greater). Emissions from oil- and coal-fired facilities are the major contributors to these risks from inhalation, while gas-fired utilities have been shown to present lower risks. The highest MEI risk accross all plants was estimated to be 1×10^{-4}.

These risk estimates are for direct inhalation exposure alone and do not include risks associated with long-range transport or indirect exposures. The risks from indirect exposure may be significant for those compounds which are environmentally persistent and have the tendency to bioaccumulate.

Figure G-6. Percent Distribution of Annual Cancer Incidence Resulting from HAP Emissions from All U.S. Electric Utilities



(Total incidence = 0.56 cases/yr)

These results do not include risks due to long-range transport or indirect exposure.

NOTE: These results are based on direct inhalation exposure modeling within 50 km of each facility, using the assumptions and data explained in the text. There are uncertainties associated with these numbers, also discussed in the text.

The risk assessment process typically uses a few conservative assumptions and default options to address some of its inherent uncertainties. However, in some cases assumptions and defualt options are used that are likely to underestimate exposures and risks. For example, the HEM inherently underestimates individual exposure in an exposed population impacted by more than one plant. The HEM calculates risks for an individual plant for the exposed population within 50 km of the plant. However, some individuals may be within 50 km of more than one power plant and would be double-counted and included in the risk distribution for each plant presumably at different individual risk levels. The risk for these individuals is the sum of the risks for each plant to which they are exposed. As a result, the HEM underestimates exposures for individuals exposed to emissions from more than one power plant. However, a modeling analysis (section 6.5 of Chapter 6) was conducted to evaluate the impact of the uncertainty associated with overlapping plumes. This analysis indicates that overlapping plumes leads to only a slight, if any, increase in exposures for the MEIs. Also, the method of double-counting used in the HEM does not affect the estimation of cancer incidence.

A risk characterization must address the uncertainties and limitations of the analysis. Therefore an uncertainty analysis evaluating each of the major models, assumptions, data, and variability was conducted on the results of this risk assessment The uncertainty analysis helps identify errors of either overestimation or underestimation. Methods, models, data, and assumptions used in the analysis are identified, including their rationale and the effect of reasonable alternative assumptions on the conclusions and estimates. These are presented in Section G.3 for exposure assessment, and Section G.4 for the exposure-response relationship. The uncertainties identified in Sections G.3 and G.4 are propagated (combined) to generate overall estimates of uncertainty in the risk estimates themselves. This process and the results are described in Section G.5.

G.3 ANALYSIS OF UNCERTAINTY OF DIRECT INHALATION EXPOSURE TO HAPs EMITTED FROM ELECTRIC UTILITIES

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

- Emissions characterization,
- Environmental fate and transport,

- Characterization of the study population, and
- Exposure calculation.

The overall exposure assessment is summarized in chapters 4 to 6 of the interim report along with a discussion of the uncertainties and limitations of this exposure assessment, including default options; models, methods, and data; uncertainty; variability; and aggregation. This section summarizes a more detailed uncertainty analysis that was conducted on the exposure assessment. The exposure assessment comprised two distinct phases: emissions characterization and exposure modeling (which includes environmental fate and transport, characterization of the study population, and exposure calculation). Emissions characterization for this utility study is described in detail in Chapter 3. The environmental fate and transport, population characterization, and exposure calculations are described in detail in Appendix F.

The baseline exposure assessment used a combination of assumptions, parameters, and input data. Some of these are considered conservative (more likely to overestimate than underestimate exposure) and others are not conservative. Any procedure that relies on a combination of point values-some conservative and some typical-yields a point estimate of exposure and risk that falls at an unknown percentile of the full distributions of exposure and risk.[7] For this reason, the degree of conservatism in the exposure calculation cannot be determined. A quantitative uncertainty analysis was conducted to identify the degree of uncertainty, degree of uncertainty, and overall confidence in the inhalation risk assessment results for the local analysis (i.e., within 50 km of plants).

The uncertainty analysis was conducted separately for the two phases of the exposure assessment identified above: emissions characterization and exposure modeling. These analyses are described in Sections G.3.1 and G.3.2, respectively.

The uncertainty analysis focuses on three HAPs (arsenic, chromium, and nickel), which contributed most to inhalation risks and is considered to be generally representative of uncertainties for the full complement of HAPs. Furthermore, a detailed uncertainty analysis could not be conducted on all of the utility plants included in the original analysis. Therefore, four plants (2 oil-fired and 2 coal-fired plants) were selected that contribute most to risk, based on both the estimated incidence and the maximum individual risk. Again, these are considered to be generally representative of other plants.

G.3.1 Emissi acterization

actor (EF) program was developed by EPA to cific emissions rates based on fuel type, fuel nfigurations, and emissions testing results. The An em program (including principles and rationale) and are described in Chapter 3. This program is based estimat alance concept: reducing concentrations in the fuel ori impact of the boiler and control devices. An emission rogram was developed and used to estimate emissions lated from the sampling based on boiler type, fuel type, ir pollution control device.

HAPs studied were grouped into two types: inorganics, which exist as trace elements, such as metals, in the fuel feed and as fluorine and chlorine, which form acid gases; and organic HAPs, which are most frequently formed during or after fuel combustion. Emissions factors were developed to predict unit-by-unit HAP emissions for all utility units in the United States. The emission factors are based on the sampled differences between the trace elements in the feed and in the exiting flue gas. The inorganic trace elements for which emission factors were estimated include antimony, arsenic, beryllium, cadmium, chromium, cobalt, lead, manganese, mercury, nickel, phosphorous, and selenium. However, the uncertainty analysis will focus on the three HAPs that contribute most (over 95 percent) to total risk: arsenic, chromium, and nickel.

G.3.1.1 Defining the Emissions Estimation Process.

The generalized equation for estimating emissions of inorganic trace elements on a plant-specific basis is as follows:

$$E = FC \times TE \times EMF_b \times EMF_a \times CF \qquad [G-1]$$

where

E = Emissions of HAP exiting the stack (kg/yr)

FC = Fuel consumption (coal: 100 ton/yr, oil: 100 bbls/y

TE = Concentration of HAP (trace metal) in fuel (ppm)

EMF_b = Boiler-specific Emissions Modification Factor t
account for the amount of HAP entering the bo
that exiting the boiler (unitless)

EMF_a = Air pollution control device (APC pecific Emissions Modification Factor to account for amount of HAP entering the APCD of that exiting t (unitless)

CF = Conversion factors to convert units o nsumption and concentration to kg/yr emission ra 0.0906, CF_{oil} = 0.15636).

In general, the parameters and steps in the em characterization can be divided into three distinct c fuel consumption, HAP concentration in the fuel, and e modifications factors (EMFs). The uncertainty was eva separately for each and is presented in Sections G.3.1. G.3.1.3, and G.3.1.4. Fuel consumption is a plant-speci parameter, while HAP concentration and EMFs are specific t HAP of concern, although which fuel type and EMFs may apply depends on the configuration of each specific plant.

G.3.1.2 <u>Fuel Consumption</u>. Fuel consumption was based on data reported by each utility in the Utility Data Institute/Edison Electric Institute (UDI/EEI) database. Where possible, 1990 data were used; if they were not available, 1989 data were used. Data in the UDI/EEI database are self-reported values but do represent the best data available. The values used for fuel consumption were plant-specific. Tables G-10a and G-10b summarize the fuel consumption data for each of the four plants included in the uncertainty analysis. The data on fuel consumption were used directly for emissions characterization. In general, the data represent a point estimate for total fuel consumption for the year 1990. This value is based on records of consumption within the plant.

The data presented in the UDI/EEI database represent annual fuel consumption rates and do not reflect the actual variation in consumption over time. The rate of fuel consumption for individual units may vary dramatically over time and between units. For example, some baseload units are used to provide rather uniform power load and may operate continuously, while others may be used to provide peak load and operate only when electric demand exceeds the output of the baseload units at that utility. Furthermore, some baseloaded plants may change production by tens or hundreds of megawatts over the course of the day, matching the electric demand. The fuel consumption data used represent yearly consumption rates, and no attempts to characterize short-term variation in fuel consumption was made for three reasons:

Table G-10a. Summary of Fuel Consumption and Plant Configurations: Oil-Fired Utilities

Plant	**Unit Name**	**Oil Consumption (1000 bbls/yr)**	**Coefficient of variation (1000 bbls/yr)**	**Boiler type**	**Particulate control**
29	1-B	292.40	5.64	O-FRONTDRYNOX	NO CONTROL
29	2-B	345.60	6.67	O-FRONTDRYNOX	NO CONTROL
29	3	703.00	13.56	O-FRONTDRYNOX	NO CONTROL
29	4-B	361.10	6.97	O-TANGDRYNONOX	NO CONTROL
29	5	1186.70	22.89	O-TANGDRYNONOX	NO CONTROL
133	3	209.40	4.04	O-FRONTDRYNOX	NO CONTROL
133	4	216.10	4.17	O-FRONTDRYNOX	NO CONTROL
133	5	418.40	8.07	O-FRONTDRYNOX	NO CONTROL
133	6	369.50	7.13	O-FRONTDRYNOX	NO CONTROL
133	7	811.70	15.66	O-TANGDRYNONOX	NO CONTROL
133	8	906.00	17.48	O-TANGDRYNONOX	NO CONTROL

Table G-10b. Summary of Fuel Consumption and lant Configurations: Coal-Fired Utilities

Plant	Unit No	Coal origin used in EFP	Coal Consumption (1000 tons/yr)	Coeff. of variation (1000 tons/yr)	Boiler type	Particulate control	SO2 Control	Fuel		Coal State(s) of origin						
343	1	IL	1,561.80	52.73	HTANGNONOX	ESP, HS	NO CONTROL	COAL	IL	68.85%	WY	23.39%	CO	6.76%	IN	1.00%
343	2	IL	1,135.60	38.34	HTANGNONOX	ESP, HS	NO CONTROL	COAL	IL	68.85%	WY	23.39%	CO	6.76%	IN	1.00%
343	3	IL	1,424.60	48.09	HTANGNONOX	ESP, HS	NO CONTROL	COAL	IL	68.85%	WY	23.39%	CO	6.76%	IN	1.00%
343	4	IL	1,044.90	35.28	HTANGNONOX	ESP, HS	NO CONTROL	COAL	IL	68.85%	WY	23.39%	CO	6.76%	IN	1.00%
240	1	PA	246.3	8.32	HTANGNONOX	ESP, HS	NO CONTROL	COAL	PA	65.10%	WV	32.62%	KY	1.68%	WY	0.61%
240	2	PA	191.2	6.45	HTANGNONOX	ESP, HS	NO CONTROL	COAL	PA	65.10%	WV	32.62%	KY	1.68%	WY	0.61%
240	3	PA	501.5	16.93	HTANGNONOX	ESP, HS	NO CONTROL	COAL	PA	65.10%	WV	32.62%	KY	1.68%	WY	0.61%
240	4	PA	517.7	17.48	HTANGNONOX	ESP, HS	NO CONTROL	COAL	PA	65.10%	WV	32.62%	KY	1.68%	WY	0.61%

- Long-term average exposure is used to calculate cancer risks minimizing the importance of short-term variations.

- Data on fuel consumption rates are not reported for yearly consumption only.

- Use of individual units as base or peak load is likely to change over time, providing for a cycle of repair and maintenance for individual units.

Uncertainty (including variability) in fuel consumption estimates can be expected to occur as a result of changes in demand for electricity (seasonally and annually), varying value of the fuel used (e.g., bituminous vs. anthracite coal), and changes in the source of fuel (e.g., State of coal origin). However, there are no data on which to estimate the uncertainty introduced by these factors and for uncertainty in fuel consumption rates. Data are not readily available on the rate of fuel consumption for individual units by month or season. The data are self-reported and to evaluate the accuracy of these data and to evaluate trends in fuel consumption over time would require significant primary data collection from individual utilities, which is beyond the scope of this study and may be restricted by guidelines established by the Office of Management and Budget. There were no data readily available which provided insight into the uncertainty or variability or uncertainty in fuel consumption. The only data available were from emissions tests conducted under optimal or steady state conditions which are not representative of uncertainty in long-term consumption. Therefore, a normal distribution with a coefficient of variation of ±5 percent was arbitrarily selected as a lower bound on the uncertainty in long-term average fuel consumption.

G.3.1.3 <u>HAP Concentration in Fuel</u>. Separate approaches to estimate HAP concentration in coal and oil were used in the emissions characterization. These were necessitated by the broad range of types and sources of coal and a lack of concentration data for fuel oil. Therefore, the uncertainty analysis was conducted, and is discussed, separately for each.

G.3.1.3.1 <u>HAP Concentration in Oil</u>. The HAP concentration data in oil are summarized in Table G-11. Limited data on trace element concentrations in oil were available. Two distinct groups of test data of trace element concentrations in residual oil have been identified. The risk assessment was conducted using one data set (FCEM) of a total of 12 different samples of residual oil and initial data analysis assumed that non-detects were at one-half the detection limit. Residual oil is assumed to

Table G-11. SUMMARY OF PARAMETER DISTRIBUTIONS: OIL-FIRED ELECTRIC UTILITIES

					Central	Tendency				
Parameter		Distribution	Unit	Min.	Mean	Geo	Max	N	SEM + SD	Std Err
	Oil/Boiler EMF									
Arsenic/Oil	frontdrynonox	beta		0.06	0.28	0.17	1.00	7	0.24	0.13
	frontdrynox	beta		0.14	0.59	0.44	1.00	3	0.34	0.25
	tangdrynonox	triangular		1.00	1.00	1.00	1.00	1		
	oppdrynonox	beta		0.01	0.23	0.08	0.45	2	0.25	0.22
Chromium/Oil										
	frontdrynonox	beta		0.02	0.29	0.18	0.66	7	0.17	0.09
	frontdrynox	beta		0.19	0.66	0.53	1.00	3	0.33	0.24
	tangdrynonox	triangular		0.66	0.66	0.66	0.66	1		
	oppdrynonox	beta		0.32	0.66	0.57	1.00	2	0.40	0.34
Nickel/Oil										
	frontdrynonox	beta		0.28	0.67	0.62	0.90	7	0.17	0.09
	frontdrynox	beta		0.64	0.79	0.77	1.00	3	0.15	0.11
	tangdrynonox	triangular		0.53	0.53	0.53	0.53	1		
	oppdrynonox	beta		0.79	0.89	0.89	1.00	2	0.19	0.11
	Oil/APCD EMF									
Arsenic	ESPcold	beta		0.49	0.52	0.52	0.55	2	0.03	0.03
Chromium	ESPcold	beta		0.44	0.44	0.44	0.44	2	0.00	0.00
Nickel	ESPcold	beta		0.07	0.17	0.14	0.27	2	0.10	0.10
OIL CONCENTRATION					log mean				log(GSD)	
Arsenic	FCEM	Lognormal		ND	-1.44			6	0.781	0.17
Chromium	FCEM	Lognormal		ND	-0.64			11	0.275	0.04
Nickel	FCEM	Lognormal		ND	1.31			14	0.309	4.44
As	As: SGS	Lognormal			-0.103			22	0.171	
Cr	Cr: SGS	Lognormal			-1.41			22	0.531	
Ni	Ni: SGS	Lognormal			1.49			22	0.24	

be representative of all oil fuel burning since 95 percent of oil-fired boilers burn residual oil, and that the 12 samples were also representative. The average concentrations taken to represent the "best estimate" were 0.699 ppm for arsenic, 0.3 ppm for chromium, and 24.37 ppm for nickel.

Subsequently a second data set (SGS) of about 22 samples of residual fuel oil were obtained which initially appeared to be similar with respect to average concentration (Baker and Robertson, 1994). Both data sets were re-analyzed using an alternative treatment of non-detects (using a probability plotting routine) and assumed to be log-normally distributed (prohibiting negative concentrations with no upper bound limit).[9] The results of the analysis are summarized in Table G-11. The results indicate that the trace element concentration differs from the averages previously calculated (using non-detects equal to half the detection limit), and that the two data sets may reflect sampling from appreciably different "population of oil samples." The distributions for nickel are roughly comparable between the two data sets, but differ dramatically for both chromium and arsenic. The difference in the two "populations" of oil samples may reflect some combination of real changes over time in residual oil HAP concentrations, seasonal differences, and/or differences in analytical chemistry. Residual fuel oil is the leftovers of the oil refining process and does vary by season (reflecting the shift of gasoline production in summer for heating oil in the winter), and with shifts in the composition of the crude and demands for different volatility and other characteristics imposed by VOC control requirements.

The major uncertainties associated with oil HAP concentration are whether there are significant differences in the oil burned throughout the country and whether differences in concentrations may occur spatially (geographic origin), temporally, or by type of oil. Since 95 percent of oil-burning units burn residual oil, use of the two residual oil data sets is reported. However, how well that residual oil data represent the universe of residual oil burned or which sample "population" is more representative is not known.

Therefore, the two data sets will be treated as two distinct cases and are assumed to be representative of the range of oil being burned by utilities. It is also assumed that the uncertainty (including variability) in the mean concentration within that data sets also represents the uncertainty in the long-term mean HAP concentration for each utility. Given that little information is available on how representative these data are of the oil being burned, and that regional differences are expected, the standard deviation (SD), rather than the SE, was used as a measure of uncertainty. The use of the SD is supported because any one individual facility may use oil which is associated with any one of the samples analyzed, and therefore, the standard deviation would more accurately reflect the uncertainty in the HAP concentration.

G.3.1.3.2 HAP Concentration in Coal. The estimation of HAP concentration in coal can be rather complex because concentrations differ among types of coal burned (bituminous, sub-bituminous, lignite, and anthracite) and the geographic origin (State) of that coal. In addition, individual plants may burn coal of multiple types and origin. For units burning coal, the risk assessment assigned each unit a single State of coal origin based on the State of origin for the majority of coal burned at that plant. The average concentrations of metallic HAPs reported in the U.S. Geological Survey [USGS] database for that State were multiplied by the amount of fuel that the unit burned in 1990. The estimation of HAP concentration is further complicated by the fact that bituminous coal is cleaned prior to shipment and burning, which reduces the trace element concentration. Therefore, an additional coal cleaning factor (CCF) (the proportion of HAP remaining after cleaning) was applied to those States with regulations requiring cleaning. The average of coal cleaning based on data from cleaning samples from Alabama, Illinois, Kentucky, and Pennsylvania was applied to bituminous coal to obtain the HAP concentration to be used in the emissions estimating equation (Equation G-1).

The major uncertainties associated with the HAP concentrations in coal include:

- Trace element concentration in coal has been shown to vary widely among States, within a State geographically, and even within a single coal seam. The use of a single-point estimate may not represent the range of concentrations (or average concentrations) that may be experienced at a single plant. Therefore, it is possible that the risk assessment may over- or underestimate actual emissions.

- The USGS database contains concentrations of trace elements in coal 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. As a result, emissions from coal-fired units may be lower than predicted, resulting in an overestimation of risk, though the degree of overestimation cannot be determined.

- Individual units may burn coal that originated from several States. These coals differ in values and trace element concentrations. For this analysis, the EF model assigns each coal-fired unit a single State of coal origin. Therefore, actual emissions from a utility may differ significantly from that estimated if alternative coals are used with higher or lower trace element concentrations.

This analysis quantitatively evaluated the uncertainty introduced with assignment to a single State of coal, and the HAP concentration within individual States, and the effectiveness of coal-cleaning. The uncertainty in relating the USGS data measured within the seam to the actual coal shipments could not be quantitatively assessed because of the lack of data.

The uncertainty analysis focused on developing an estimate of composite coal concentration for each facility that would be used in the emissions estimation equation (Equation G-1). This estimate included both an average or "best" estimate and some measure of the uncertainty around that estimate. The SE from the data were used because of the large number of samples and exhaustive sampling by the USGS of economically viable veins. This was accomplished by first defining distributions for HAP concentration in coal for each state of coal origin. Coal from nine States (Illinois, Indiana, Kentucky, New Mexico, Pennsylvania, Utah, Virginia, West Virginia, and Wyoming) was burned in the nine coal-fired plants evaluated. The HAP concentration is assumed to be log-normally distributed (which supports the physical constraint of negative concentration with no upper limit). The raw data for those States were reviewed and the average HAP concentration and SEM for each of the three HAPs were calculated for each State and are presented in Table G-12.

The raw data on coal cleaning were also reviewed and an average and SEM were calculated for each of the three HAPs. The HAP concentration and the CCF were multiplied together to estimate the HAP concentrations for those States subject to coal cleaning to estimate the HAP concentration in "cleaned" coal. The SEMs for both the CCF and HAP concentrations were used to estimate the uncertainty in the HAP concentration in "clean" coal using the following formula:

Table G-12. SUMMARY OF PARAMETER DISTRIBUTIONS FOR COAL-FIRED ELECTRIC UTILITIES

					Central	Tendency				
Parameter		Distribution	Unit	Min.	Arith	Geo	Max	N	Std Err	Std Dev
TRACE ELEM CONC										
Arsenic	State of origin									
	CO	Log normal	ppm	0.5	1.3		2.9	30		
	IN	Log normal	ppm	0.5	10.1		42.7	73	1.07	
Cleaned	IN	Log normal	ppm		5.5				0.72	
	KY	Log normal	ppm	0.1	19.1		169.0	267	1.67	
Cleaned	KY	Log normal	ppm		10.4				1.20	
	NM	Log normal	ppm	0.2	1.8		13.9	96		
	PA	Log normal	ppm	0.31	32.11		16920	527	32.10	
Cleaned	PA	Log normal	ppm		17.5				17.56	
	VA	Log normal	ppm	1	11.2		79.1	39	2.76	
Cleaned	VA	Log normal	ppm		6.1				1.57	
	WV	Log normal	ppm	0.08	10.7		237.4	431		
Cleaned	WV	Log normal	ppm		5.8				0.44	
	WY	Log normal	ppm	0.28	0.7		2.09	145	0.06	
Chromium										
	CO	Log normal	ppm	1.09	1.9		3.83	30		
	IN	Log normal	ppm	1.4	15.5		75.3	69	1.14	
Cleaned	IN	Log normal	ppm		7.9				0.81	
	KY	Log normal	ppm	3.1	16.3		86	266	0.57	
Cleaned	KY	Log normal	ppm		8.3				0.65	
	NM	Log normal	ppm	0.07	6.0		19.1	96	0.27	
	PA	Log normal	ppm	4.6	20.1		67	532	0.41	
Cleaned	PA	Log normal	ppm		10.3				0.75	
	VA	Log normal	ppm	3.8	12.5		28.2	39	0.84	
Cleaned	VA	Log normal	ppm		6.4				0.62	
	WV	Log normal	ppm	2.6	15.3		131.8	436	0.52	
Cleaned	WV	Log normal	ppm		7.8				0.61	
	WY	Log normal	ppm	0.6	2.8		10.95	145	0.07	
Nickel										
	CO	Log normal	ppm	0.9	1.3		3.64	30	0.13	
	IN	Log normal	ppm	5.3	17.9		48.14	73	1.26	
Cleaned	IN	Log normal	ppm		10.1				0.92	
	KY	Log normal	ppm	2.4	17.5		108.9	266	0.83	
Cleaned	KY	Log normal	ppm		9.9				0.74	
	NM	Log normal	ppm	0.45	4.6		14.5	93	0.26	
	PA	Log normal	ppm	3.1	20.4		223.6	547	0.65	
Cleaned	PA	Log normal	ppm		11.6				0.76	
	VA	Log normal	ppm	3.2	11.2		26.9	39	0.95	
Cleaned	VA	Log normal	ppm		6.4				0.65	
	WV	Log normal	ppm	1.7	14.3		280	437	0.76	
Cleaned	WV	Log normal	ppm		8.1				0.63	
	WY	Log normal	ppm	0.7	2.2		5.8	145	0.07	

Table G-12 (continued)

					Central	Tendency				
Parameter		Distribution	Unit	Min.	Arith	Geo	Max	N	Std Err	Std Dev
COAL CLEAN FACTOR										
	Arsenic	normal	unitless	0.23	0.54	0.51	0.97	27	0.04	0.214
	Chromium	normal	unitless	0.25	0.51	0.48	0.89	27	0.04	0.186
	Nickel	normal	unitless	0.27	0.57	0.54	0.85	27	0.03	0.168
EMF: Boiler		beta	unitless						SEM+SD	SEM
Arsenic										
	cyclwetnonox	beta	unitless	0.25	0.53	0.50	0.85	6	0.12	0.10
	frontdrynonox	beta	unitless	0.22	0.71	0.59	0.99	3	0.30	0.24
	htangnonox	triangular	unitless	0.89	0.89	0.89	0.89	1		
	oppdrynox	beta	unitless	0.11	0.73	0.59	1.00	6	0.19	0.15
	tangdrynonox	beta	unitless	0.60	0.80	0.77	1.00	2	0.41	0.20
Chromium										
	cycwetnonox	beta	unitless	0.22	0.27	0.26	0.35	6	0.02	0.02
	frontdrynonox	beta	unitless	0.61	0.87	0.85	1.00	3	0.18	0.13
	htangnonox	triangular	unitless	1.00	1.00	1.00	1.00	1		
	oppdrynox	beta	unitless	0.20	0.61	0.55	1.00	6	0.13	0.11
	tangdrynonox	beta	unitless	0.58	0.79	0.76	1.00	2	0.30	0.21
Nickel										
	cycwetnonox	beta	unitless	0.12	0.28	0.26	0.72	6	0.10	0.08
	frontdrynonox	beta	unitless	0.76	0.85	0.84	1.00	3	0.11	0.08
	htangnonox	triangular	unitless	0.91	0.91	0.91	0.91	1		
	oppdrynox	beta	unitless	0.29	0.44	0.42	0.67	5	0.09	0.07
	tangdrynonox	beta	unitless	0.44	0.64	0.60	0.84	2	0.29	0.20
EMF: Control device		beta	unitless							
Arsenic										
	Baghouse	beta	unitless	0.00	0.01	0.01	0.03	6	0.01	0.01
	ESPcold	beta	unitless	0.00	0.03	0.02	0.06	17	0.01	0.01
	ESPhot	beta	unitless	0.02	0.06	0.04	0.11	2	0.10	0.07
	wetscrub	beta	unitless	0.04	0.32	0.20	0.76	5	0.17	0.13
Chromium										
	Baghouse	beta	unitless	0.00	0.06	0.02	0.25	5	0.06	0.05
	ESPcold	beta	unitless	0.00	0.03	0.02	0.07	17	0.01	0.01
	ESPhot	beta	unitless	0.02	0.03	0.03	0.04	2	0.02	0.01
	wetscrub	beta	unitless	0.03	0.42	0.26	1.00	5	0.20	0.17
Nickel										
	Baghouse	beta	unitless	0.00	0.04	0.01	0.20	6	0.05	0.03
	ESPcold	beta	unitless	0.00	0.05	0.02	0.37	17	0.03	0.02
	ESPhot	beta	unitless	0.00	0.01	0.00	0.01	2	0.01	0.00
	wetscrub	beta	unitless	0.05	0.79	0.53	1.00	5	0.19	0.19

$$SEMcleancoal=\sqrt{(SEMhap^2 \times SEMccf^2)+(MEANhap^2 \times SEMccf^2)+(SEMhap^2 \times MEANccf^2)} \qquad [G-2]$$

The resulting average concentrations for the three HAPs in "clean" coal (Indiana, Kentucky, Pennsylvania, Virginia, and West Virginia) and the SEM for those concentrations are included in Table G-12.

$$SEMcomposite=\sqrt{(\%state1 \times SEMstate1^2)+(\%state2 \times SEMstate2^2)+(\%state3 \times SEMstate3^2)} \qquad [G-3]$$

Once HAP concentrations had been determined for each State, the next step was to estimate the average HAP concentrations in the composite coal (the overall mixture of coal) burned at each utility as well as some measure of the uncertainty related to that average. In all cases, it was assumed that each unit within a utility burned the same mixture of coal as was indicated by the data reported in the UDI/EEI database. It is likely that the mixture of coal and the relative contribution of any one state used at a facility would change over time, depending on the sulfur content, BTU heating value, cost, and availability. However, no data were available to allow for a prediction of how the mixture of coal would change with respect to the long-term average. A weighted average for HAP concentration in the composite coal was estimated for each utility using the data in Table G-10a and G-10b. The SEM of the HAP concentration per State was used to estimate the measure of uncertainty (SEM) in the average HAP concentration in the composite coal using the following formula (illustrated for plant using coal from three States): This was carried out for each of the three HAPs for each of the nine utilities being evaluated. These values and associated SEMs were used as inputs for parameter TE in the emissions estimation equation (Equation G-1).

G.3.1.4 Emissions Modification Factors. The EMFs were developed and used to represent the amount of a HAP compound entering and exiting a device (boiler or air pollution control device). The EMFs were based on the results of emissions testing conducted by the Electric Power Research Institute (EPRI), Department of Energy (DOE), EPA, and selected electric utilities. Sampling was conducted at the boiler inlet, the boiler outlet, and before and after the air pollution control devices. For each trace element in the feed, partitioning factors associated with each boiler type were developed and applied. Control device removal efficiencies were then applied to arrive at the final emission factor (the amount of a HAP emitted divided by the average amount of the same HAP entering the device). For each test site, multiple inlet and outlet samples were taken, these samples may or may not have been taken simultaneously. The EMF for each test site was calculated as the average of the inlet HAP amount divided by the average of the outlet measurement. It should be noted that the ratio of the mean inlet and outlet concentrations may differ from an average of EMFs calculated for each data point. However, raw data was not available for all test results to make a comparison of the possible errors introduced by this approach. In some cases, individual test values were reported while for others only average values were reported. Therefore, it was not possible to evaluate the true variability in the EMFs and the errors introduced through sampling and analysis.

The values for any EMF are constrained to between zero and one. A value of zero implies that the boiler and/or APCD is 100 percent efficient in removing the HAP from the exit flue gas and containing it entirely in the bottom ash of the boiler or in the APCD respectively. Values of one imply that the boiler and/or APCD have no effect on HAP removal and all that enters is emitted.

Basically, EMFs are fractions of the amount of a HAP exiting a device (boiler or air pollution control device) divided by the amount of the same HAP compound entering that device. These factors are calculated by taking the geometric mean of similar devices (e.g., oil-fired tangential boilers). Geometric means for EMF values were used in the initial risk assessment based on the limited number of samples for many of the EMFs. However, it was later decided that the arithmetic mean may be a more appropriate statistic. Therefore, for this uncertainty analysis, the mean for each EMF was used.

The major uncertainties associated with the EMFs include:

- Limited testing may have occurred at facilities where the device (boiler or APCD) was not operating in a manner representative of those of the same type.
- Data on which the EMFs have been estimated were the result of non-simultaneous sampling.
- Performance may vary significantly among similar APCDs.

- Device performance may be affected by load changes, process upsets, process modifications, environmental factors, age, maintenance, or production details.

The uncertainty analysis was limited to existing data, which vary significantly among the various devices for which EMFs were estimated. The uncertainty represented in the raw data was assumed to be representative of the overall uncertainty in the EMF. The data did not allow for a differentiation between uncertainty and variability. The raw data for each device were reviewed and the mean, geometric mean, and standard errors were calculated for each HAP. For some devices (e.g., oil tangentially fired dry without NO_x controls), only a single data point was available. To estimate the uncertainty for those EMFs, a triangular distribution was used. The single value was used as the apex (most likely) and the distribution was bound between 0 and 1.

As described above, the EMFs are constrained to values between zero and one. Using a normal distribution would allow for negative values (implying that a HAP is removed from surrounding air) and for values greater than one (implying that the HAP is created in the device), neither of which is physically possible. (Note: EMFs > 1 are possible for HAPs that are produced during combustion, but this does not apply to the trace elements.) The use of a log normal distribution prevents the occurrence of negative EMFs, but still allows for values greater than one. Therefore, the beta distribution was selected to represent the uncertainty associated with the EMFs. The beta distribution is commonly used to represent variability (or uncertainty) over a fixed range, in this case between zero and one. The value of the beta distribution lies in the variety of shapes it can assume by varying two parameters, alpha (A) and beta (B), which are explicitly defined by the available data, specifically using the mean and SEM (or SD). Alpha and beta are defined in terms of the mean and SEM as follows:

$$A = Mean \times \left[Mean \times \frac{(1-Mean)}{SE^2} - 1 \right] \qquad B = (1-Mean) \times \left[Mean \times \frac{(1-Mean)}{SE^2} - 1 \right]$$

If the two parameters are equal, the distribution is symmetrical. If $\alpha < \beta$, the distribution is said to be positively skewed (most of the values near the minimum). If $\alpha > \beta$, the distribution is said to be negatively skewed (most of the values near the maximum). The EMFs (mean) and their associated uncertainty (SEM) are presented in Table G-11 for oil-fired boilers and in Table

G-12 for coal-fired boilers. These values and their associated SEMs were used as inputs for parameter EMF_a and EMF_b in the emissions estimation equation (Equation G-1).

It should be noted that the beta distribution is defined by the existing data and does not necessarily have foundations to describe known processes. Therefore, an alternative approach was also investigated assuming that $EMF = e^{-m}$, and values of m were estimated from the available data and then described as a log-normal distribution. In general, the results were similar but, in all cases, the beta distribution yielded a greater spread in the range of possible values and uncertainty. Therefore, as a conservative approach, the beta was used in this analysis.

As described in Section G.1, directly using the SE from the sample population would tendency be an overconfident estimate of (i.e. too narrow a range) of uncertainty due to:

- The SE is used as an estimate of the SD of a normal distribution while the t-distribution is used to represent the sampling distribution. For small sample sizes, the t-distribution is significantly different, and approaches a normal distribution as samples sizes increase (n~30). Therefore, error is introduced by assuming a normal distribution.
- The mean of a distribution represents a long-term average only when the population samples are representative of values that would be obtained over the long-term. Data obtained from a convenience sample, additional error is introduced due to potential nonrepresentativeness of the sample.
- The mean of a distribution represent a long-term average only if the population sample represents true variability. Measurement error must also be considered. generally the most appropriate estimate of uncertainty in the long-term average is to use the larger of the SE or the standard deviation due to measurement error.

An estimate in the uncertainty of the SE was made to account for some of these errors and uncertainties, estimating the SD of the expected SE repeatedly sampling from the overall population. Using the method described in Hattis and Silver[27] the following steps were carried out:

- estimate mean and SD for the existing data set and assume to describe the "true" distribution of the population.

- randomly sample from a normal distribution with a "true" mean and SD the same number of samples as in the original data (repeated 2500 times).
- determine the standard error for each sample,
- calculate the SD of the sample SEs.

The measure of uncertainty used for these parameters was the initial SE + the SD of the simulated SEs and are reported in Tables G-11 and G-12.

G.3.1.5 Plant-Specific Emission Rates. Stochastic (Monte Carlo) simulation was used to determine the uncertainty in the risk estimates based on the uncertainty and variability within each of the parameters used in the emissions model. Crystal Ball (Decisioneering, Inc., Denver, CO) was used to conduct the simulation. A probability distribution that best represents the variable, its average value, and a measure of variability or uncertainty about the average value was developed for each of the parameters used in the model. The standard error of the mean (SE) for each parameter was used as the estimate of uncertainty rather than the standard deviation for each parameter. The SE is a more appropriate statistic given that the risk assessment is concerned with long-term average exposures and intermittent extreme values of exposure levels are not as important.

After the assumptions about each variable in Equation G-1 are entered into the program, a three-step process is used to generate a forecast chart containing a frequency distribution of probable values for the calculation of interest (e.g., arsenic emissions). The 3 steps are as follows:

1. Generate a random number for each variable based on its type of probability distribution, average, and measure of uncertainty.
2. Recalculate the spreadsheet using the randomly generated numbers.
3. Enter the results into the forecast frequency chart.

These three steps were repeated a total of 3,000 times to produce a frequency distribution of calculated HAP emissions values. Running the simulation 3,000 times was arbitrarily selected based on observations that the forecast curve was smooth. A quick review of data from 2,000, 2,500, 3,000 and 4,000 simulations showed little change in resulting statistics and percentile between 3,000 and 4,000 trials. It should be noted that a single utility may consist of several individual units. To estimate emissions from a given utility, each unit is modeled separately and then the emissions are totaled across all units for each simulation. The same value for trace element concentration in fuel was used for each unit within a facility based on the assumption that each unit would draw on an overall fuel supply shared by all units within a facility. The remaining parameters (FC, EMF_b, and EMF_a) were treated independently, which accounts for the fact that fuel consumption and device performance (EMF) are independent for individual units within a utility.

For illustration purposes, Figure G-7 summarizes the distributions used for each of the parameter inputs for Plant #29, the oil-fired plant that contributed most to cancer incidence. Table G-13 summarizes the distribution of predicted emissions from Plant 29 while Figure G-8 graphically presents the distributions predicted for Plant #29. This distribution gives some indication of the degree of uncertainty and the possible range of emissions estimates that may be experienced. Figure G-9 graphically presents the cumulative probability distribution for emissions from Plant 29 using the FCEM and SGS concentration data. Furthermore, Figure G-9 and Table G-13 presents a cumulative probability plot of the combined concentration data. These distributions give some indication of the degree of uncertainty and the possible range of emissions estimates which may be experienced. The original emissions estimation program had been designed to be an unbiased estimator of emissions. The results of the uncertainty analysis tend to support this assertion. The original baseline estimates ranged from the 22nd percentile of the overall distribution (for arsenic using the SGS oil concentration data) to the 95th percentile (for chromium using the SGS oil concentration data). This is attributed to the reanalysis of the two oil concentration (FCEM and SGS) data sets using the probability plotting routine rather than treating non-detects as half the detection limit used in the initial risk assessment. When the combined data set (SGS and FCEM oil concentration data) is considered the original baseline emissions estimates were correspond to the 58th percentile for arsenic, 68th percentile for nickel, and the 80th percentile for chromium. Furthermore, the 95th percentile of the simulated range of long-term average emissions estimates are within a factor of two or three of the mean and original baseline estimate and within a factor of less than six within the median.

To compare how estimates differ for the coal-fired plants, Plant #343 was selected for illustration purposes. Plant #343

Figure G-7. Assumptions Used in the Modeling Uncertainty in Emissions from Plant #29

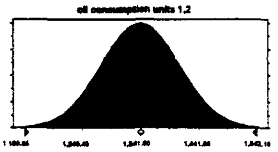
Assumption: oil consumption units 1,2

Normal distribution with parameters:

Mean	1,341.00	(=C21)
Standard Dev.	67.05	(=C27)

Selected range is from -Infinity to +Infinity

Mean value in simulation was 1,340.90



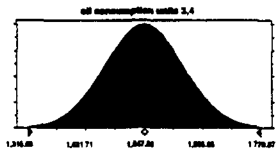
Assumption: oil consumption units 3,4

Normal distribution with parameters:

Mean	1,547.80	(=D21)
Standard Dev.	77.39	(=D27)

Selected range is from -Infinity to +Infinity

Mean value in simulation was 1,546.88



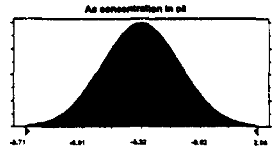
Assumption: As concentration in oil

Lognormal distribution with parameters:

Mean	-3.32	(log)
Standard Dev.	1.80	(log)

Selected range is from -Infinity to +Infinity

Mean value in simulation was 0.17



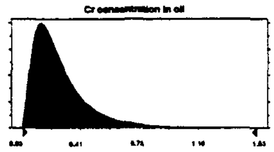
Assumption: Cr concentration in oil

Lognormal distribution with parameters:

Mean	0.28
Standard Dev.	0.20

Selected range is from 0.00 to +Infinity

Mean value in simulation was 0.29



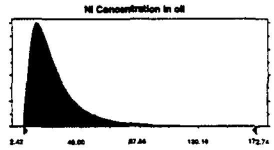
Assumption: Ni Concentration in oil

Lognormal distribution with parameters:

Mean	26.31
Standard Dev.	21.37

Selected range is from 0.00 to +Infinity

Mean value in simulation was 25.93



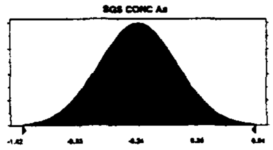
Assumption: SGS CONC As

Lognormal distribution with parameters:

Mean	-0.24	(ln=2.3*log)
Standard Dev.	0.39	(ln=2.3*log)

Selected range is from -Infinity to +Infinity

Mean value in simulation was 0.85



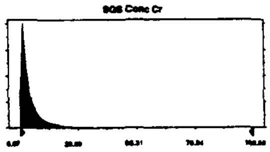
Assumption: SGS Conc Cr

Lognormal distribution with parameters:

Mean	5.74
Standard Dev.	10.68

Selected range is from 0.00 to +Infinity

Mean value in simulation was 6.02



Assumption: SGS Conc Ni

Lognormal distribution with parameters:

Mean	36.03
Standard Dev.	21.53

Selected range is from 0.00 to +Infinity

Mean value in simulation was 35.54

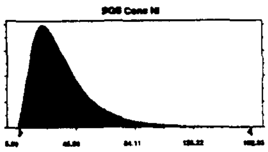


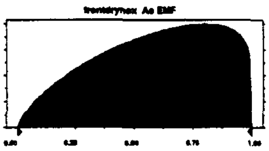
Figure G-7. Continued.

Assumption: frontdrynox As EMF

Beta distribution with parameters:

Alpha	1.69	(=C13)
Beta	1.17	(=C14)
Scale	1.00	

Selected range is from 0.00 to 1.00
Mean value in simulation was 0.59

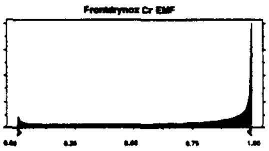


Assumption: Frontdrynox Cr EMF

Beta distribution with parameters:

Alpha	0.74	(=F13)
Beta	0.39	(=F14)
Scale	1.00	

Selected range is from 0.00 to 1.00
Mean value in simulation was 0.66

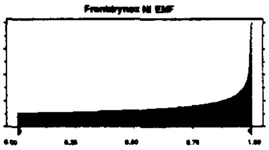


Assumption: Frontdrynox Ni EMF

Beta distribution with parameters:

Alpha	1.00	(=C13)
Beta	0.69	(=C14)
Scale	1.00	

Selected range is from 0.00 to 1.00
Mean value in simulation was 0.59

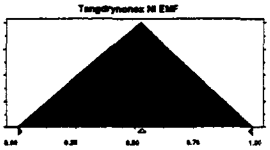


Assumption: Tangdrynonox Ni EMF

Triangular distribution with parameters:

Minimum	0.00	
Likeliest	0.53	(=J83)
Maximum	1.00	

Selected range is from 0.00 to 1.00
Mean value in simulation was 0.52

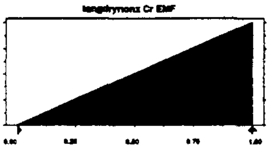


Assumption: tangdrynonx Cr EMF

Triangular distribution with parameters:

Minimum	0.00	
Likeliest	1.00	(=D23)
Maximum	1.00	

Selected range is from 0.00 to 1.00
Mean value in simulation was 0.67



Assumption: tangdrynonox Ni EMF

Triangular distribution with parameters:

Minimum	0.00	
Likeliest	0.66	(=G53)
Maximum	1.00	

Selected range is from 0.00 to 1.00
Mean value in simulation was 0.56

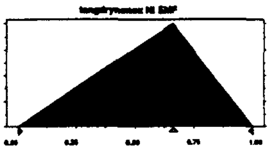


Table G-13. *Summary of Results for Monte Carlo Simulation of* HAP Emissions (kg/year) from Oil-Fired Plant#29

		As Emissions			Cr Emissions			Ni Emissions		
		FCEM	SGS	Combined	FCEM	SGS	Combined	FCEM	SGS	Combined
Mean		44	216	130	69	20	45	6423	8612	7527
Initial Point Estimate		------------	133	------------	------------	69	------------	------------	8170	------------
(Percentile)		(93)	(22)	(58)	(64)	(95)	(80)	(76)	(59)	(68)
Percentiles:	0.0%	0.0	13	0.0	4	0.2	0.2	225	507	225
(approx.)	2.5%	0.4	66	--	11	0.5	0.6	934	2001	1125
	5.0%	0.8	78	--	15	0.9	1.5	1227	2389	1462
	10%	1.6	98	2.0	20	1.6	2.1	1645	3015	2125
	25%	3.9	142	6.0	33	3.8	8.1	2731	4583	3487
	50%	9.1	197	98	54	8.7	26	4759	7010	5962
	75%	31	269	206	85	20	60	8009	10847	9562
	90%	94	358	303	135	43	106	12736	16034	14625
	95.0%	177	417	378	174	71	149	17441	20173	19060
	97.5%	322	474	455	214	117	208	22936	24544	24750
	100.0%	2177	894	2177	1087	818	1087	55170	82450	82450
Ratio										
95th : mean		4.1	1.9	2.9	2.5	3.6	3.3	2.7	2.3	2.5
95th : median		19.4	2.1	3.9	3.2	8.1	5.7	3.7	2.9	3.2
95th : baseline		1.3	3.1	2.8	2.5	1.0	2.2	2.1	2.5	2.3

FCEM = *Original oil concentration data, distribution defined by probability plotting technique.*

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

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

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

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

Figure G-8. Summary of Results fo Monte Carlo Simulation of HAP Emissions from Oil-Fired Plant#29

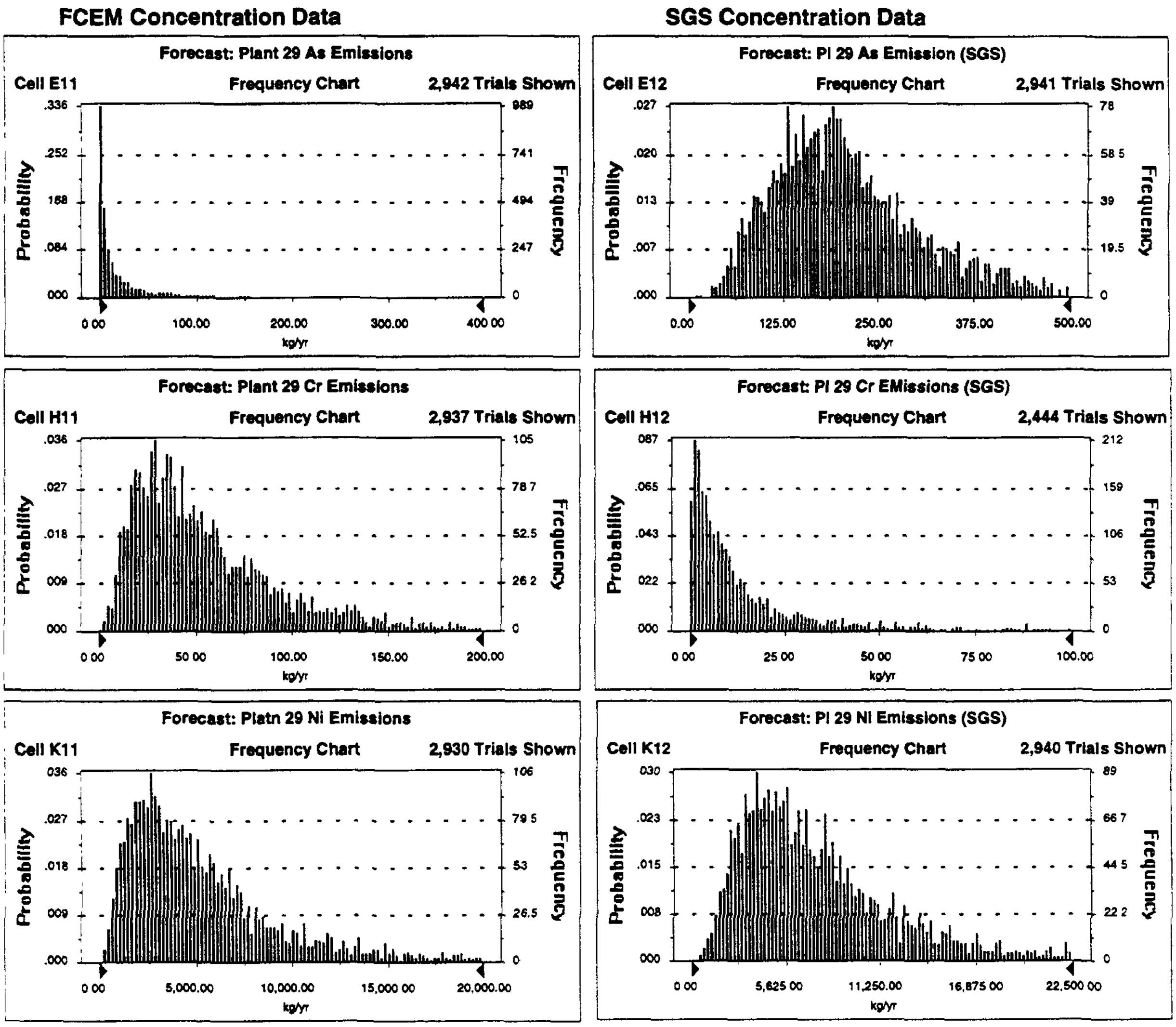
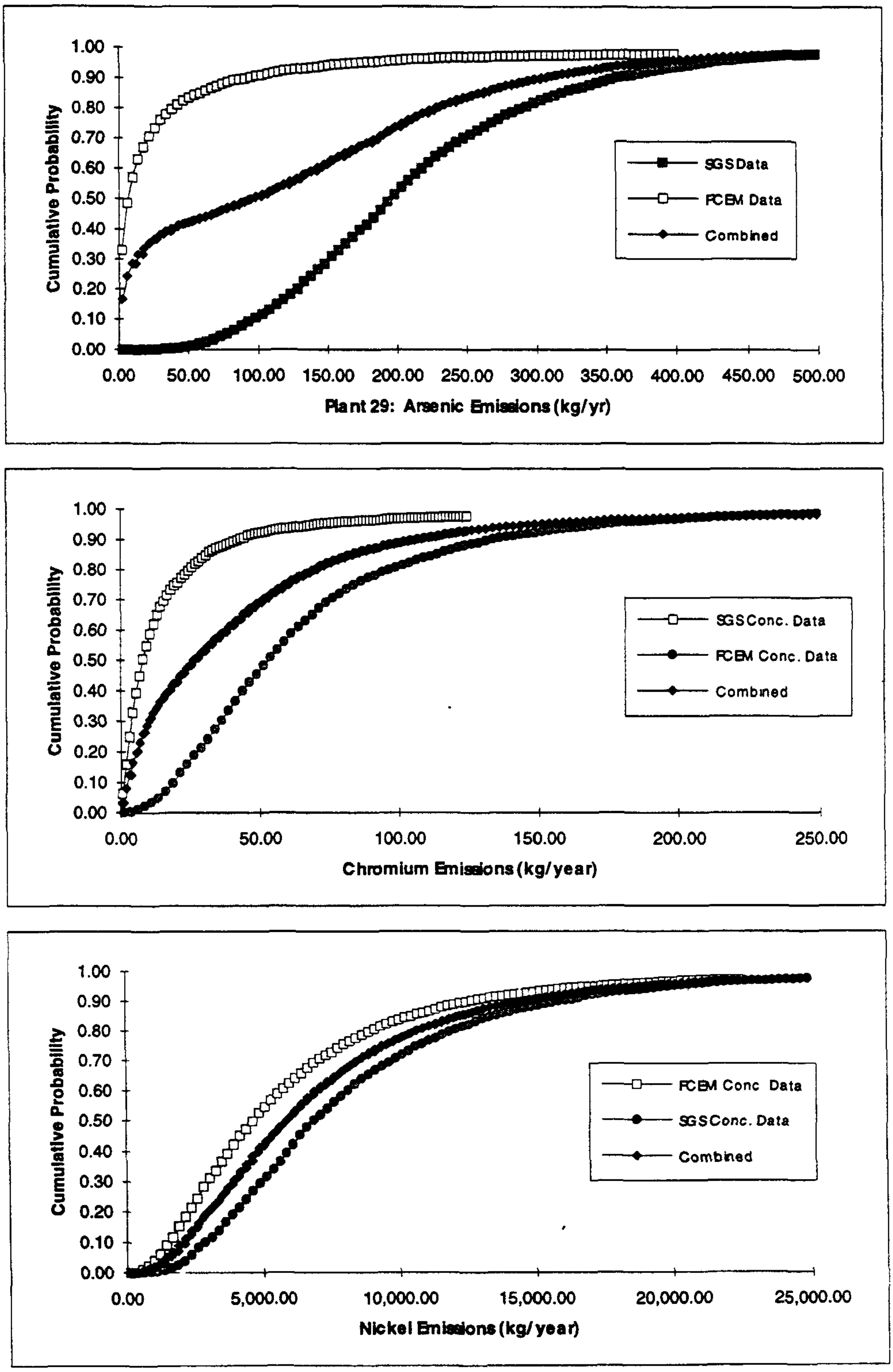


Figure G-9. Cumulative Probability of Uncertainty in Emissions from Plant #29:
Comparison of FCEM and SGS Oil Concentration Data



was the coal-fired plant that contributed most to cancer incidence. Figure G-10 graphically presents, and Table G-14 summarizes the distribution for emissions estimates resulting from the uncertainty analysis. The original point estimate of emissions ranged from the 81st to 96th percentile of the range of emissions predicted under the uncertainty analysis. This is significantly higher and may be attributed to the fact that in the original analysis coal from a single state was assumed (defined by the state whose coal was consumed most). However, for this plant the other coals used were generally of lower HAP concentrations which would tend to drive the emissions estimate lower than estimated during the baseline assessment. The 95th percentile on the simulated range of emissions ranged from 0.9 times the baseline emissions estimate for nickel to 2.5 times the baseline estimate for arsenic. As with Plant 29 the 95th percentile of the overall range of predicted emissions typically was within a factor of two or three of the mean emissions estimate and the original baseline emissions estimate.

Additional analysis was also conducted on two additional plants, Plant 133 and Plant 240, the highest oil and coal-fired plants with the highest estimated MIR, respectively. The results from this analysis are summarized in Table G-14. For Plant 133, the estimate of emissions is influenced by which of the oil concentration data sets are used. The original emissions estimate used in the baseline assessment ranged from the 32nd percentile for arsenic (using the SGS data) to the 94th percentile for arsenic (using the FCEM data) and chromium (using the SGS data). For Plant 240, the original emissions estimate used in the baseline assessment was between the 82nd and 92nd percentile on the range of predicted emissions.

G.3.2 Dispersion and Exposure

The HEM was used to conduct exposure modeling. The Industrial Source Category- Long-Term (ISCLT2) air dispersion model within HEM, was used to estimate long-term average ambient air HAP concentrations within 50 km of each source. The exposure modeling conducted for the baseline risk assessment is based on the following default assumptions:

- Dispersion occurs as predicted by a Gaussian-plume model in a rural setting in flat terrain.

- All exposures occur at one location; no adjustment is made for exposure changes resulting from population movement between home, school, work, etc.

Figure G-10. Results of Monte Carlo Simulation of HAP Emissions (kg/year) for Coal-Fired Plant #343

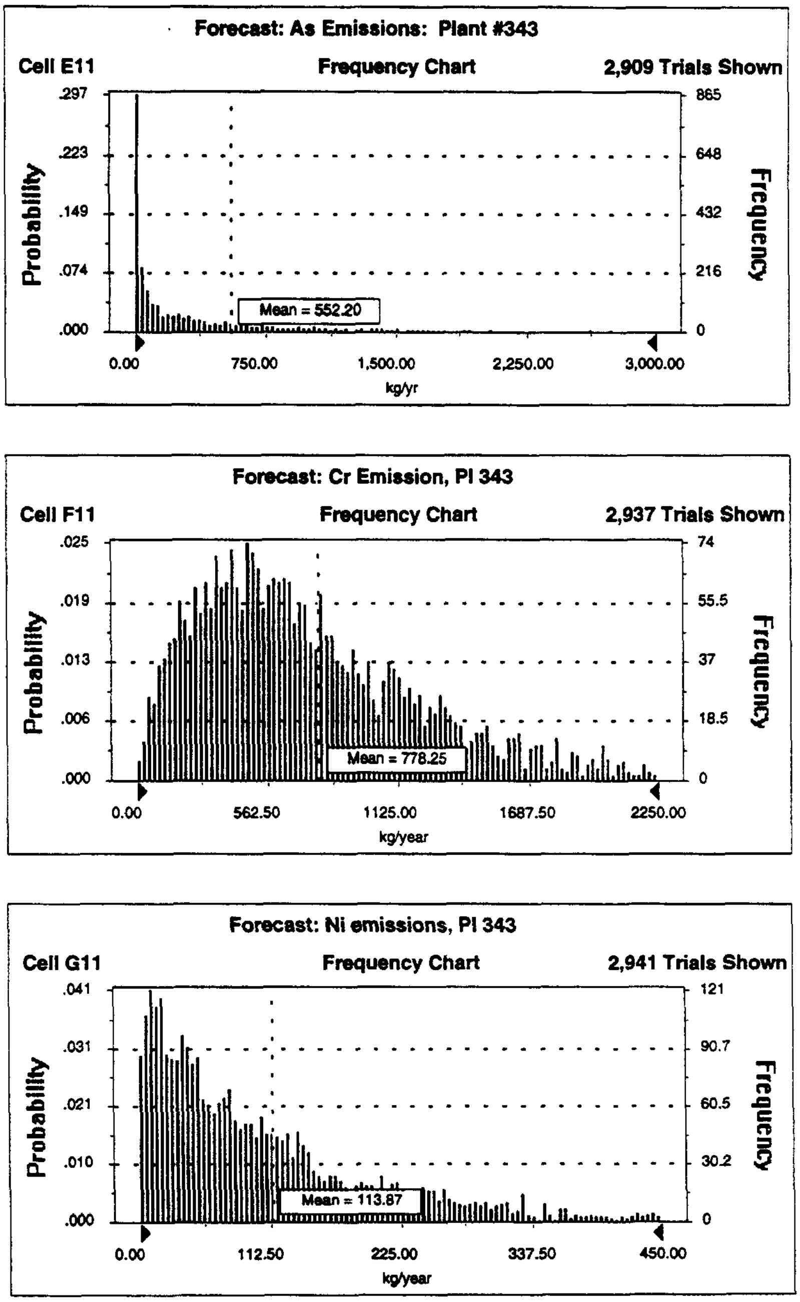


Table G-14. Summary of Results of Monte Carlo Simulation of HAP Emissions from Coal-Fired plant #343

Forecast: Emissions: Plant #343

	arsenic	chromium	nickel
Mean	552	778	114
Standard Deviation	901	540	116
Mean Std. Error	16	10	2
Original Point Estimate	969	1631	360
Percentile	81	92	96
Percentile			
0.0%	0	10	0
2.5%	2	93	4
5.0%	5	140	7
10%	10	208	13
25%	25	387	33
50%	169	653	78
75%	688	1061	151
90%	1619	1500	262
95.0%	2462	1836	334
97.5%	3198	2163	430
100.0%	8490	4623	985
Ratio			
95th : mean	4.5	2.4	2.9
95th : median	14.5	2.8	4.3
95th : baseline	2.5	1.1	0.9

Original Point Estimate = Estimate derived in the baseline assessment using EPA default values

Percentile = the percentile of the original point estimate within the Monte Carlo derived distribution

- Exposure occurs at centroids of census blocks or at user-specified location for which ambient concentrations are estimated since the locations of actual residences are not included in the database.

- People reside at the population centroids for their entire lifetimes (assumed to be 70 years for cancer risk estimation) and indoor concentrations are assumed to be the same as outdoor concentrations. (Note: the concentration at the centroid is assumed to represent a person's average lifetime exposure allowing for activity patterns between areas of higher and lower ambient concentration.)

- The power plant emits HAPs at the same level for the 70-year lifetime of exposure.

- The only source of exposure is the stack emissions dispersed in the ambient air; no resuspension of pollutants via dust is considered.

- There is no population migration or growth.

- Varying exposures that might arise as a result of differences such as age, sex, health status, and degree of activity do not exist (although variations in these with respect to issues of exposure-response are considered in Section G.4).

Dispersion modeling and exposure calculation are complex processes and nonlinear in nature. A detailed discussion of the mathematical equations, models, and data can be found in Appendix F and in several EPA reference documents.[10-14] However, the parameters and data used in the modeling can be described briefly as follows:

- Plant-specific parameters: stack heights, stack diameter, exit gas temperature, exit gas velocity, latitude and longitude (obtained from the UDI/EEI database).

- Meteorological data: STability ARrays (or STAR data summaries), joint frequencies of occurrence of windspeed, wind direction, and air stability combining into a frequency distribution (from the nearest National Weather Service location).

- Population data: 1990 U.S. Census data, on the "block level," containing 6.9 million records; for each plant, the census blocks within 50 km were identified and assigned to be located at the centroid of the census block if within 0.5 km, or to user-specified receptors where ambient air concentrations are modeled.

Some of the major uncertainties associated with the exposure modeling include:

- Meteorological conditions at the nearest STAR site may differ from the conditions near the utility site.
- The HEM assumes that exposure occurs at estimated ambient air concentrations of the pollutant. However, it has been shown that people spend a majority of their time indoors. Indoor concentrations of ambient pollutants would likely be less than those found outdoors. As a result, exposures would be less.
- Gaussian dispersion models do not accurately predict pollutant dispersion in areas of complex terrain (e.g., hills, valleys, mountains, over water), where deposition is significant, or where transformation occurs, and might not correctly predict where maximum concentrations will be realized. The uncertainty in using a Gausian plume model was not evaluated due to the lack of alternative verified models and resources.
- Surface roughness is addressed in HEM as one of two model defaults, urban or rural, using population density as a proxy. The selection of urban or rural can dramatically affect dispersion and concomitant exposure. Furthermore, these two default modes may also not cover the possible range of surface roughness which may be encountered.
- Exposure is also tied to emission rates and atmospheric processes that affect pollutant concentrations in the microenvironment where the person is exposed (e.g., infiltration of outside air indoors, transport by the prevailing wind, diffusion/transport by atmospheric turbulence, chemical and physical transformation, deposition, and reentrainment), each of which has its own underlying variability. For example, personal activities and pollutant concentrations at a specific location might change in response to outdoor temperature (e.g., open windows).

G.3.2.1 Approach to Uncertainty Analysis. Air dispersion modeling is complex and nonlinear and cannot be carried out with the use of spreadsheets (hence, cannot be used within Crystal

Ball). In addition, significant time is required to conduct the modeling and process the data for each run. A stochastic (Monte Carlo) approach requires a large number (thousands) of repetitive runs (3,000 was used for the emissions estimates) 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), surface roughness (urban versus rural), and meteorologic conditions. The uncertainty analysis, therefore, focused on the three parameters. Separate model runs were conducted for different combinations of these parameters and uncertainty was estimated by examining the range of resulting exposure estimates.

G.3.2.1.1 Plant Stack Parameters. The effective stack height or plume rise above the physical stack height will impact the degree of dispersion. In general, the greater the plume rise, the greater the degree of dispersion and area impacted; the lower the plume rise, the higher the maximum ground-level concentration with a smaller impacted area. Plume rise is a nonlinear relationship between exit gas velocity, stack diameter, exit gas temperature, windspeed, and ambient temperature. Simply, to represent the uncertainty related to effective stack height, one could calculate representative high, medium, and low values for effective stack height across all plants (e.g., ± 10 percent). It should be noted that over time a much greater degree of variation in exit velocity and stack gas temperature would be expected related to changes in plant operation. However, there were no data available to indicate how these variations would impact the long-term averages experienced at a specific plant. Ambient temperature and windspeed are the two factors that have the greatest impact on plume rise and would be a site-specific calculation. Exit gas temperature and velocity may vary according to plant utilization. No data on variations in plant operation is available. Therefore, the approach used may underestimate actual uncertainty.

G.3.2.1.2 Surface Roughness. Surface roughness influences the ground-level turbulence and dispersions and includes such things as downwash. Because the predominance of physical structures (e.g., buildings) is a measure of surface roughness, the distinction is represented by using either the urban or rural setting in HEM (Population density is used as surrogate for this parameter, hence the reference to urban versus rural.) It should be noted that similar surface roughness can occur in entirely natural settings based on terrain and vegetation. The urban and rural mode of HEM approximates the bounds of surface roughness. In actuality, surface roughness at a particular site may be somewhere in between. To address this uncertainty, HEM was run in both the urban and rural mode. It is not possible to determine where a specific site may fit on the continuum of surface roughness; therefore, the two modes are considered to be equally plausible.

G.3.2.1.3 Meteorology. Local meteorology has a great impact on dispersion. Ambient temperature, stability class, windspeed, and wind direction will all influence the degree and direction of dispersion and subsequent exposure. Meteorology can vary dramatically over a short distance and the nearest STAR site may not be representative of, or even appropriate for, the local meteorology in the vicinity of a utility. Prevailing or intervening physical features (e.g., hills or mountains, large bodies of water) may influence the prevailing wind direction and precipitation and indicate that an alternative STAR site may be more appropriate. Given the large number of utilities and STAR sites, it is not possible to determine which of the STAR sites is most appropriate, or the degree to which local meteorology may differ. Therefore, a limited number of alternative STAR sites were used to represent local meteorology.

The three factors being evaluated were nonlinear with respect to each other and required 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, HEM runs were made for 18 different combinations of dispersion parameters that may apply to a given plant. For the purposes of this uncertainty analysis, it was assumed that there was insufficient information to determine the relative correctness of each combination and, therefore, each was considered equally likely to represent the possible range of values.

G.3.2.2 Uncertainty in Exposure Modeling. For each of the four plants, HEM was run using each combination of input parameters (for a total of 18 runs) for a unit emission rate (1 kg/yr). The maximum concentration, total exposure, and number of people exposed at various concentrations were estimated for each run. 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. For illustration purposes, Figure G-11 summarizes the dispersion coefficients for estimating the maximum concentration and total exposure over the 18 different parameter combinations for Plant #29. To estimate the maximum concentration or incidence, the coefficient is multiplied by the actual emissions (kg/yr). Maximum concentration is useful in estimating the MEI risk (Max Conc x Unit Risk Estimate [URE]) and the total exposure is used to estimate cancer incidence (Exposure x URE/70 years).

The range of values of maximum concentration and exposure over the 18 combinations can be a useful measure to evaluate uncertainty. Furthermore, the uncertainty in each measure may be affected independently with regard to each parameter combination. For example, the maximum concentration may not change significantly using data from one meteorological location, however, the location where that concentration occurs and the number of exposed persons (and the resulting incidence) may change significantly due to changes in prevailing wind direction. For specific locations, the HEM is believed to predict concentrations to within a factor of two. However, for long-term estimates of maximum location (without regard to location), HEM predictions are believed to have a coefficient of variation of 10 to 40 percent, which is significantly less than the variation between individual scenarios.

G.3.2.2.1 Complex Terrain. The current version of the air dispersion model contained in the HEM is the ISCLT2 version 92062, which was designed for sources located in relatively simple or flat terrain where land elevations were below the top of the stack. The EPA conducted a screening analysis on the effect of complex terrain. Due to the recent availability of elevation data and retrieval software, a screening analysis for complex terrain around six coal- and six oil-fired higher-risk utility plants was conducted.

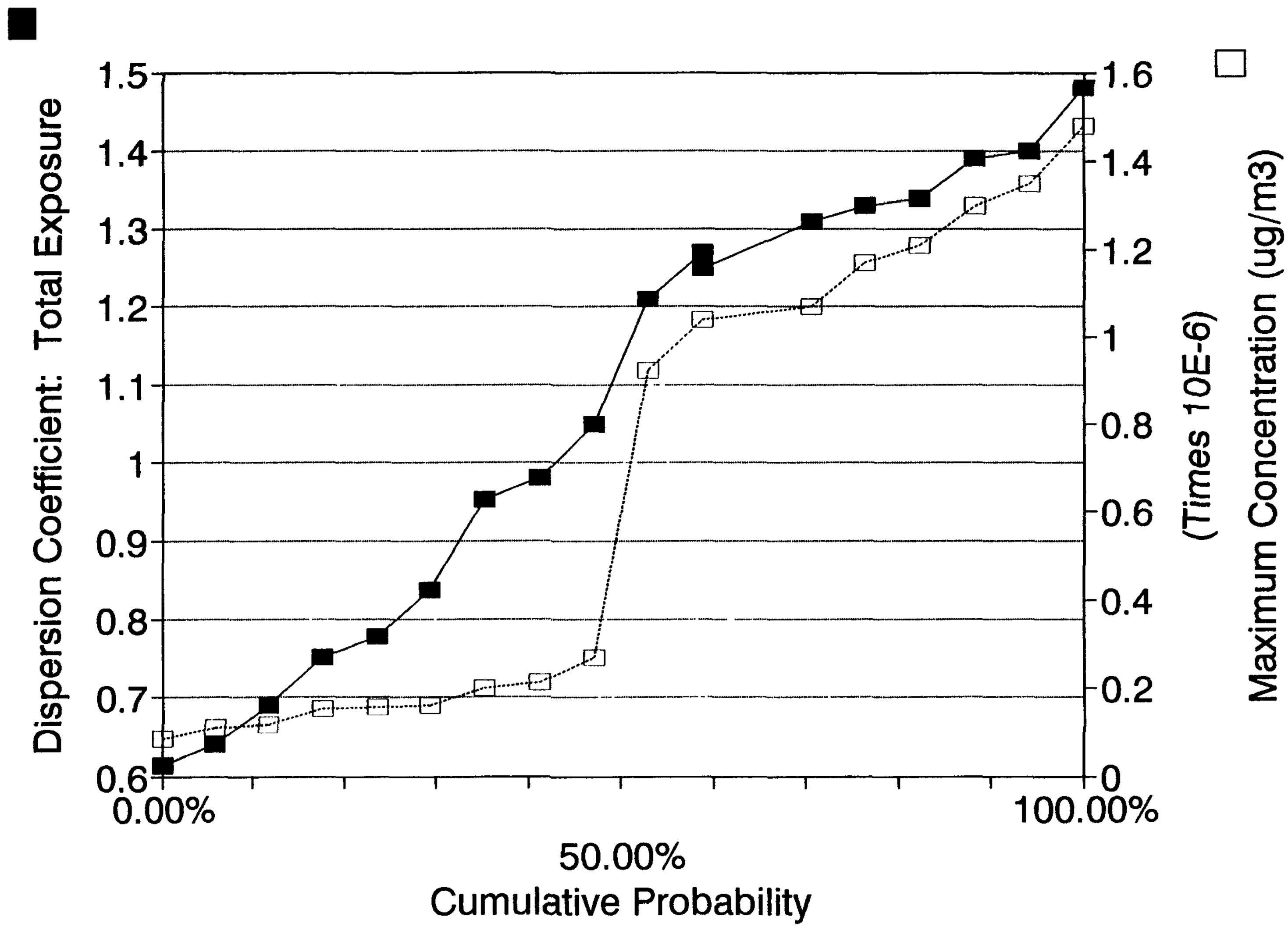


Figure G-11. Cumulative Probability Plot of HEM Dispersion Coefficients under Alternative Dispersion Scenarios

The model recommended by the Air Quality Modeling Group for use in a screening analysis for analyzing the impact of complex terrain was the Industrial Source Complex - Short Term - Draft model (ISCSTDFT), version 94340. The advantage of this model was that it contained the algorithms from the COMPLEX1 model, which is expressly designed for air dispersion when terrain features are above the tops of the stacks. The model has three air dispersion regimes. For those areas in the first regime (terrain below the stack top), the model applies the ISCST algorithms with elevated receptors. For those areas in the second regime (terrain between the top of the stack and the effective plume height), the model uses the larger of the concentrations estimated by the ISCST and the COMPLEX1 models. For the third regime (terrain above the effective plume height), the model uses the COMPLEX1 algorithms only.

Unlike the ISCLT2, the ISCSTDFT air dispersion model requires hourly meteorological observations for periods of up to 1 year. For this study, the 1984 data were used for all plants. As a preliminary study, one site was chosen and several years of meteorological data were used. For this one site, the long-term maximum concentrations (shown in Table G-15) for arsenic did not vary much from year to year; therefore, the multiple-year analysis was not conducted for other plants.

The ISCSTDFT model requires elevation estimates for each point on the polar grid system. The algorithm used to estimate the elevation of a polar grid point was to choose the maximum elevation for any of the four points, from the USGS 3-arc-second database, surrounding the polar grid point. At latitudes contained in the continental U.S., 3 arc-seconds on a map is roughly equivalent to 300 feet or 100 meters.

The ISCSTDFT model was set up to run elevation data for the polar grid system. The regulatory default mode was used, with the exception of the gradual plume rise option (see the sample outputs in the attachments). The elevation data file was created using a format that could be used directly in the air dispersion model and this greatly simplified the procedures for setting up the ISCSTDFT runs. However, source parameters, such as location and stack data were entered "by hand." The model output, the arsenic concentration profile, was then processed by a simple FORTRAN program to rearrange the data into a format suitable for exposure analysis. The rearranged arsenic concentration data were then used to run the exposure model. Exposure calculations were conducted using the exposure algorithms from the HEM.

Table G-15. Multiple Year ISCSTDFT Analysis

Year	Max. Conc. (μg/m$^{3)}$
1984	.00245
1985	.00238
1986	.00260
1987	.00244

The results of the analysis are provided in Table G-16. Although estimated maximum concentrations to which people were being exposed increased for several of the higher risk plants, the overall maximum individual risk for each source category remained at about the same level. Also, the overall population risk estimates, i.e., cancer cases per year per plant and cancer cases per source category, remained at about the same level for both source categories. Some of the estimated risks decreased because of lower terrain features. Thus, the addition of complex terrain to the analysis did not significantly change the risk assessment results. The analysis was conducted for arsenic but is representative of other toxic emissions.

G.3.3 Qualitative Discussion of Uncertainty

The results (distributions) derived in Sections G.3.1 and G.3.2 represent only a portion of the uncertainty embodied in the risk estimates. This uncertainty analysis focused only on the parameter uncertainty within the models used in the baseline analysis. Analysis was also limited to those parameters for which data or information were available. Model uncertainty has not been formally evaluated. Alternative models may yield dramatically different results.

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

TABLE G-16. RESULTS OF THE COMPLEX TERRAIN MODELING AND EXPOSURE/RISK ANALYSIS

	People		Max Risk		Total People		Annual Incidence	
Source	HEM	ISCSTDFT	HEM	ISCSTDFT	HEM	ISCSTDFT	HEM	ISCSTDFT
Coal Plants								
Plant A	399	399	1.94E-06	1.10E-06	3,980,000	3,980,000	0.0024	0.0026
Plant B	6	68	5.00E-07	4.00E-07	361,000	361,000	0.0003	0.0003
Plant C	3	69	3.61E-07	2.85E-06	683,000	683,000	0.0007	0.0010
Plant D	14	6	1.98E-07	9.26E-07	407,000	407,000	0.0002	0.0003
Plant E	5	19	1.78E-07	1.67E-07	58,100	58,100	<.0001	<.0001
Plant F	71	4	1.27E-07	1.17E-06	347,000	347,000	0.0002	0.0004
Oil Plants								
Plant G	45	45	1.38E-05	5.65E-06	836,000	836,000	0.0014	0.0013
Plant H	1	1	1.41E-06	6.41E-07	836,000	836,000	0.0002	0.0002
Plant I			8.40E-07	1.16E-06*				
Plant J			5.62E-07	9.01E-07*				
Plant K			3.12E-07	2.15E-7*				
Plant L			2.42E-07	1.72E-07*				

* Maximum risk level based on maximum concentration from the air dispersion model with no regard to actual exposure. Exposure modeling will produce a similar or lower MIR value.

that the model performs as expected (i.e., across a range of boilers and constituents to estimate overall emissions). The average of the ratios across all stacks and constituents was 1.01, while averages for the three HAPs are 1.4, 0.7, and 0.9 for arsenic, chromium, and nickel, 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 are found for individual boilers and constituents. The largest difference for an individual boiler was were estimated emissions were about 5,000 times lower than reported test results. Furthermore, the model tended to underestimate rather than overestimate emissions about 60 percent of the time within this sample of boilers. A preliminary evaluation of facilities with large differences between projected and actual emissions found that these facilities were likely to burn multiple fuel types. Petroleum coke was found to be burned at the facility with the greatest difference. Therefore, while the model appears to predict fairly well, emissions may be over- or under-estimated at individual facilities. This difference may be due to the combinations of fuel being burned since the model assumes a single fuel type. The plants in which the model underestimated significantly were some of the lowest emitting plants predicted by the model, and using measured rather than modeled emissions would bring their emissions more in line with the rest of the plants, but would not become high-risk plants when compared to others. However, it should also be noted that the emissions test data on which these comparisons were made were from short-term (often nonsimultaneous) sampling. Whether these test data reflect actual long-term average emissions from these stacks are not known.

Likewise, significant uncertainty may also be associated with the use of HEM to conduct dispersion and exposure modeling. The sensitivity analysis (described in Section G.5 below) indicated that the dispersion coefficient is a significant parameter. Therefore, it is likely that alternative models may yield dramatically different results, and that plant-specific estimates have the potential to vary from that estimated. An evaluation of model uncertainty of both the emissions factor program and HEM is indicated given their relative contribution to overall parameter uncertainty. Some studies indicated that the estimates from dispersion models may have a coefficient of variation of ± 10 to 40 percent in predicting short-term concentrations or identifying the location of maximum concentration. It is believed that with the models have greater precision in estimating long-term concentrations and exposures. A preliminary analysis was attempted to adjust the dispersion coefficients accordingly. However, the spread in the dispersion coefficients for each of the 18 scenarios was overwhelmed by the variation in the dispersion coefficients between the scenarios.

G.4 EVALUATION OF UNCERTAINTY AND VARIABILITY ANALYSIS IN THE EXPOSURE-RESPONSE ASSESSMENT FOR SELECTED HAPs

This section presents uncertainty and variability analyses for the exposure-response component of the calculation of risks from electric power generation. Uncertainty and variability of exposures were reviewed in previous sections. The goal of the present section is to estimate the variability of the quantitative relationship between exposure and the excess probability of cancer for different humans exposed to releases from electric power generation in the U.S.; and the uncertainty in the average or mean quantitative relationship between exposure and the excess probability of cancer.

There are two basic questions addressed in this section:

- *In a group of people, all exposed to the same concentration of a particular material in air as a result of electric power generation (and as provided in earlier sections of this report), what would we expect to be the variation of the excess probability of adverse health effects resulting from this exposure? Would everyone have the same probability (this is unlikely)? Would this probability of effect vary between exposed people due to differences such as the rates at which different people breathe, the length of time different people are exposed, and so on? If it does vary (which is likely), what is the cause of this variability and how can we describe the variation between people?* The answers to these questions constitute the **variability analysis** in this section. The reason for performing this analysis is to examine the degree of risk equity within groups that are exposed to the same concentration of materials in air; to identify sensitive subpopulations of people with risks larger than the average for the population; and to provide this information to risk managers for consideration in decisions. Of particular interest is the likely magnitude of the risk to the individual in the population with the maximal risk. The exposure assessment identified the maximally exposed individual (or MEI); assessment of the variability in the exposure-response relationship provides information on the variability of response within a subpopulation of individuals located at the point of the MEI.

- *In a group of people, all exposed to the same concentration of a particular material in air as a result of electric power generation, are we certain about the average (mean) excess probability of adverse health effects resulting in this population from this exposure (this is unlikely)? If we are not certain (which is likely), how can we express this uncertainty? Can we give an upper and a lower bound on where the true value of the average probability of effect is likely to be found? What are these values? Can we give a distribution of estimates of the average probability, with this distribution describing how confident we are that the true value of the average probability is any particular number (such as 10^{-6} or 10^{-5})? What is this distribution?* The answers to these questions constitute the **uncertainty**

analysis in this section. The reason for performing this analysis is to examine the confidence with which we can assert that the average probability of effect in a population for which the average exposure has already been estimated is less than any particular number.

There also are compound questions that can be addressed once answers have been supplied to the above two questions. These two answers can be combined in risk characterization (along with the assessment of variability and uncertainty in exposure from previous sections of this report) to provide answers to the questions:

- *How certain are we about the amount of variability in a population?*
- *How certain are we about the number of people whose excess probability of effect exceeds some particular value (such as 10^{-6} or 10^{-5})?*

Attention is restricted here to three HAPs which constitute in excess of 90 percent of the inhalation cancer risks from utilities in the U.S., arsenic, nickel and chromium compounds. It is assumed that the variability and uncertainty characterizing these three HAPs approximates the variability and uncertainty characterizing the entire mixture of HAPs released from utilities. This is a reasonable assumption given the degree to which these three HAPs contribute to the cancer risks. Still, it must be recognized that this uncertainty and variability analysis considers only the inhalation exposure pathway. It is likely that the indirect pathways of exposure would increase the mean value of the risk to the exposed population (perhaps also affecting the distributional characteristics, such as the SDs). The analyses reported in this section represent a lower limit on both the mean probability of effect and on the upper end of the confidence interval for this probability. In other words, both the mean and the upper limit on the confidence interval are likely to increase when the indirect exposure pathways (e.g., ingestion) are considered.

In addressing uncertainty and variability for any quantity such as the excess probability of cancer, there are two broad approaches typically used. In the first, the analysis generates an estimate of the average value in the population by using the average values for all parameters needed to calculate the risk; an estimate of the highest value by using the highest values for all parameters needed to calculate the risk; and an estimate of the lowest value by using the lowest values for all parameters needed to calculate the risk. For variability analysis, the "highest value" means the highest value actually found in the population (such as the highest probability of cancer); the "lowest value" means the lowest value actually found in that same population (such as the lowest probability of cancer). The variability analysis then is summarized by the average probability of cancer in the population; the highest probability of cancer in that population (probably in sensitive individuals); and the lowest probability of cancer in that same population (probably in insensitive individuals).

For uncertainty analysis, the "highest value" means the largest estimate of the mean probability of cancer which is consistent with existing information (in all cases in this report, the mean is taken also to be the "best estimate"); the "lowest value" means the smallest estimate of the mean probability of cancer which is consistent with existing information. The uncertainty analysis then is summarized by a "best estimate" of the average probability of cancer in the population (using the best estimate for each of the parameters needed to calculate this probability); an "upper bound" estimate of the average probability of cancer for which it can be said that it is very likely that the true value of the average probability of cancer is greater than this "upper bound" (using the upper bound estimate for each parameter); and a "lower bound" estimate for which it can be said that it is very unlikely that the true value of the mean probability of cancer is less than this "lower bound" (using the lower bound estimate for each parameter).

The problem with this first approach is that it supplies information only at three numerical values (the average, the "upper bound" and the "lower bound"), and it usually is not possible to state quantitatively the level of frequency (for variability analysis) or confidence (for uncertainty analysis) associated with the upper and lower bound estimates. A second approach that provides more information is to produce a *distribution* of values for each parameter used in calculations, each with an assigned level of probability or confidence. The result is a probability density function for the probability of cancer in the exposed population to which the risk manager may turn in asking questions such as:

- *What fraction of people have an excess probability of cancer above 10^{-6}? Above 10^{-5}?* (a question of variability).

- *With what confidence can we assert that the average probability of cancer in a population is 10^{-6}? 10^{-5}?* (a question of uncertainty).

While this second approach requires more judgment (distributions must be selected, rather than simply upper and lower bounds), it provides more complete and useful information to the risk manager. It is for this reason that the analysis in this section focuses primarily on the distributional approach, although the first approach also is examined at selected points.

It is important to note that estimates of the probability of effect require information on a number of parameters describing the biological properties of exposed people as well as the chemical and physical characteristics of the arsenic, nickel and chromium in the environment. Where appropriate, each of these components of variability and uncertainty (e.g., breathing rates, retention halftimes, etc.) are described here as quantitative probability density functions reflecting either the frequency (for variability analysis) or the degree of confidence (for uncertainty analysis) associated with any particular numerical value. This allows the uncertainty and/or variability of the separate components to be combined mathematically (here, through Monte Carlo techniques) in estimating the uncertainty and/or variability of the probability of cancer.

In some instances, however, the components are described here qualitatively. This is true primarily for the identification of sensitive subpopulations on the basis of biological characteristics, for the treatment of non-cancer endpoints, and for uncertainties introduced by the existence of different model forms for dose-response extrapolation (which is treated here semi-quantitatively). In each case, an indication is given of the potential impact of these qualitative or semi-quantitative analyses on the final estimates of the probability of cancer.

The important characteristics of risk for which uncertainty and variability of exposure-response assessment are performed in this report include:

1. Uncertainty in the Mean of the MIR. In this section, it is assumed that the highest concentration of the material in air has been identified for each source. This concentration is shown as C_{max}. It is assumed that a population is exposed to this concentration over a normal lifetime. There is uncertainty about the mean numerical value for each of the biological and environmental factors (described below) which must be

specified to convert C_{max} to an estimate of the probability of cancer. The question being addressed is: *How confident are we that the mean excess probability of cancer in a population uniformly exposed at a concentration of C_{max} is any particular value (such as 10^{-6}, 10^{-5}, etc)?*

2. Variability in the Mir for Different Individuals in a Population Located at the Point Where the Concentration Equals C_{max}. It is assumed that a population is exposed to this concentration over a lifetime. There is variability between people in the numerical value for each of the biological and environmental factors (described below) which must be specified to convert C_{max} to an estimate of the probability of cancer. As a result, there is variability in the probability of cancer for individuals within a population all exposed at the concentration C_{max}. The question being addressed is: *Within a population uniformly exposed at a concentration of C_{max}, what fraction of the people have an excess probability of cancer equal to any particular value (such as 10^{-6})?*

3. Uncertainty in the Estimate of Cancer Incidence for the Entire U.s. Population When the Average Concentration is C_{ave}. It is assumed that the U.S. population is exposed to this concentration over a lifetime. There is uncertainty about the mean value for each of the biological and environmental factors (described below) which must be described to convert C_{ave} to an estimate of the probability of cancer. The question being addressed is: *How confident are we that the excess number of cancers in a population of the size of the U.S., and exposed at a concentration of C_{ave}, is any particular value (such as 1, 0.1, etc)?*

Calculating either probability of cancer or number of cancers requires a range of assumptions (usually given as parameter values for an equation) concerning the relationship between exposure and response, as discussed in later sections. The analyses of variability and uncertainty for these assumptions are divided here into two groups of parameter values. The first contains analysis of uncertainty and variability for parameter values that are common across the three HAPs considered (arsenic, nickel and chromium). The statistical properties of the distributions describing these parameter values are the same for all three compounds. If a specific individual is above the population average for the given parameter value, this will hold true equally for all three compounds. An example of this in variability analysis is breathing rate (liters of air per minute), where if an individual has a breathing rate higher than the average for chromium, he or she also will have a higher breathing rate for nickel. An example in uncertainty analysis is that an underestimate of the average breathing rate for the population (perhaps due to sampling error) will produce an underestimate of the cancer risk for all three HAPs.

The second group contains analysis of uncertainty or variability for parameter values that differ between the three HAPs. The statistical properties of the distributions describing these parameter values are not necessarily the same for the three compounds. An example of this in variability analysis is the CSF. The degree of variability in the CSF for nickel differs from that for chromium due to differences in the data base. An example in uncertainty analysis is that the measure of uncertainty (such as the SD) for the CSF may differ between the three HAPs. If the estimate of the CSF for a HAPs such as nickel (for example) is *larger* than the true value for nickel due to sampling error, this is not inconsistent with a finding that the estimate of the CSF for chromium is *smaller* than the true value for chromium.

This distinction between uniform and compound-specific components of variability and/or uncertainty is treated here by subdividing exposure-response assessment into three separate categories: exposure times, pharmacodynamics and dose-response. *Exposure times* express the fraction of time an individual is indoors or outdoors (reflected by the mean concentration to which they are exposed, taking into account potential differences in the indoor and outdoor concentrations) and the length of residence at the point of exposure. The uncertainty and/or variability of exposure is the same for the three compounds.

Pharmacodynamics is taken here to include all considerations from exposure to target dose. It is divided conceptually into considerations related to lung intake (here, the volume of air breathed per unit time); lung uptake (the amount of the inhaled material deposited in the lung after intake, affected primarily by deposition fractions); burden (the equilibrium amount of the inhaled material in organs, affected primarily by retention half-times); biologically significant burden (the equilibrium amount of the biologically active form of the compound, affected primarily by bioactivation and/or de-activation fractions); and dose (either the amount of interaction between the biologically active form and target structures such as DNA, or the integral of the biologically significant burden over the period of exposure, often called the "area under the curve" or AUTC). For the three compounds considered here, variability for any particular pharmacodynamic property is the same for the three compounds either because the properties are identical for the three compounds (such as in the case of the volume of air breathed per unit time) or because there is insufficient information to distinguish between the compounds (such as variability and uncertainty in retention half-times). The same is true for analysis of uncertainty for these parameter values. This means the distribution describing uncertainty introduced by any particular assumption such as breathing rate, deposition fractions, retention half-times, biotransformation and dosimetry will be characterized identically for the three compounds. The distributions differ, of course, between different parameters.

The third component of exposure-response assessment, *dose-response*, is the relationship between dose and excess probability of cancer. Here, uncertainty is different for the three HAPs due to differences in the quality of epidemiologic data on which the CSF are estimated and uncertainty as to the assumption of speciation. The statistical properties of these three distributions, therefore, are different. There is little quantitative information on which to base an analysis of variability of the CSF, other than to note the effect of smoking (which is considered in the analyses which follow). This issue is treated more qualitatively through a discussion of the existence of sensitive subpopulations.

Dose-response also differs in degree, if not in kind, of uncertainty due to the need to extrapolate from high to low exposures using models that are only partially established scientifically. The issue of how to quantify uncertainty in dose-response relationships when competing biologically plausible models is one of the most highly debated areas in uncertainty analysis. This issue will be discussed semi-quantitatively in this report. The approach will be to first perform an analysis of uncertainty conditional upon adoption of the dose-response models employed in the base-line EPA risk calculations, and then to consider the potential impact of alternative scientifically-based extrapolation models.

For non-cancer endpoints, expressed in the analysis as Hazard Quotients (HQ), the treatment of uncertainty and variability is primarily qualitative. In the baseline analysis of non-cancer risks from the inhalation exposure pathway, the HQs for arsenic, nickel and chromium have not been estimated due to a lack of RfCs for these HAPs.

The highest inhalation HQ for any HAP analyzed was 0.12 for HCl. Therefore an uncertainty analysis was not performed for inhalation noncancer effects.

It must be noted, however, that the exposure assessment used in this analysis is based on an assumption of continuous exposure at a uniform ambient concentration. In real situations of exposure, the concentrations of all HAPs will fluctuate in time, rising through high values above the average and falling to low values below the average. During the excursions to high concentrations, such as might occur soon after accidental releases from a facility, the concentrations in air might exceed the time-weighted averages significantly. At the present time, however, the analysts do not have information on the distribution of concentrations in time at a single point located near a facility. Future research should focus on characterizing this source of temporal variability and its influence on the frequency of non-cancer effects in the exposed population. The same can be said for the contribution from indirect exposure pathways (which also could increase the average daily rate of intake.

A related issue is the presence of background levels of several of the HAPs (such as cadmium) in the tissues of exposed individuals. Where these background levels are close to the NOELs and LOELs, a finding that the utilities do not produce concentrations at the RfC does not ensure that non-cancer effects will fail to appear. Concentrations below the RfC could produce the effect if the background levels already are high. Future research should focus on characterizing these background levels and assessing their influence on the frequency of non-cancer effects in the exposed population.

G.4.1 The Quantitative Depiction of Uncertainty and Variability

Both uncertainty and variability analyses begin by specifying distributions for each of the parameter values appearing in equations used to estimate the probability of an effect such as cancer (the relevant equation is discussed in the following section). For uncertainty analysis, the distribution expresses the probability that the true value for the parameter (usually taken to imply the true mean for some population) is equal to any particular numerical value between a lower bound and an upper bound. For variability analysis, the distribution expresses the frequency with which the parameter has any particular numerical value within individuals in a population for which the risk is being estimated. This frequency is the same as the fraction of individuals in the population whose parameter value equals a particular numerical value.

A key problem, both conceptually and methodologically, is how these distributions should be developed. Ideally, the distributions for variability should be based on measurements of the variation of the parameter between individuals in a well specified population. Both biological and environmental parameters usually are found to be distributed lognormally, meaning the logarithms of the parameter values for individuals in the population are distributed normally. The lognormal distribution is characterized by a median, which is the numerical value of the parameter for which it may be said that 50 percent of the individuals have a parameter value which is larger and 50 percent have a value which is smaller; and by a geometric standard deviation (or GSD), which is a quantitative measure of the degree of variability. In the population, 68 percent of the numerical values are between the median divided by the GSD (the "lower bound") and the median times the GSD (the "upper bound"). In the same population, 95 percent of the numerical values are between the median divided by the square of the GSD; and the median times the square of the GSD.

Often, the GSD can be estimated directly from the available data by fitting the data to a lognormal distribution. In some cases, however, part of the measured variability is due to measurement error (properly included in uncertainty analysis) rather than variability. In such cases, the GSD of the measured distribution for variability will be larger than the true GSD. An attempt has been made in the present section to separate these two contributions to measurements of variability, primarily by selecting data in which the uncertainty in individual measurements is small compared to the standard deviation of the intersubject variability. In cases where this does not hold, the most complete approach would be to (I) estimate the uncertainty in individual measurements and (ii) de-convolve the true variability from the measured variability. That approach could not be used here since in essentially all cases of parameter values, the analysis employs ratios of parameter values in the general U.S. population over those in the epidemiological populations from which slope factors were obtained. As a result, subjective judgment was used to narrow the distribution of variability after accounting for the uncertainty in individual measurements. Throughout the present section on exposure-response, variability will be characterized by a lognormal distribution. The GSD is calculated from fitting of the data to a lognormal distribution, or from the procedure described above. The reasoning behind selection of the median and GSD are explained for each parameter value.

The situation is more complex (and contentious) for instances of uncertainty analysis. The reason is that uncertainty has several sources[2] with varying degrees of susceptibility to mathematical and statistical analysis. Measurement errors typically can be summarized by techniques of statistics, often displaying statistical properties characteristic of the normal distribution when only precision is considered. If representativeness of a sample is considered, however, it is equally valid to use a lognormal distribution since the mean values of a parameter in subpopulations usually is distributed lognormally across subpopulations. In the present analysis, a normal distribution is used when issues of precision dominate, and a lognormal distribution is used when issues of representativeness dominate.

The characteristics of these distributions can at times be estimated directly from the available data when interpreted through statistical procedures (such as calculation of the standard error of the mean based on knowledge of the standard deviation of the population of samples, which is the method used in much of the analysis of uncertainty in this report). In other instances, however, the data either are insufficient to determine the standard deviation (or GSD) by such techniques or contain sources of uncertainty not captured by standard statistical techniques. An example of the latter consideration is study design, where there may be a question as to whether the study was conducted properly, accounted for confounding influences, etc. In these instances, analysts often employ more subjective judgments of the distribution's properties. While the selection of these properties depends to some degree on examination of the available data and on past experience (and, hence, is partially objective), it also involves a judgment about factors such as study design, reliability of reporting and conceptual ambiguity which are not purely statistical issues and must be rooted in personal judgments. It is important to note that this subjective element is a part of all uncertainty analysis and does not appear only in instances where data are scarce. Such subjective judgments are noted in the present analysis when they occur. A reason is given for the judgment, although it is recognized that rational individuals may differ in their judgments.

Finally, it should be noted that throughout this analysis some parameter values have been given quantitative characterizations of uncertainty and/or variability, while others are discussed qualitatively. The Agency recognizes that it is common practice in risk analysis to attempt to quantify all sources of variability and uncertainty, even where data are insufficient to provide a statistical basis for the development of distributions. In these cases, subjective measures of variability and uncertainty typically have been used.[2] While such approaches could be used in the current analysis, the Agency has concern over the justification for subjective assignments. In many cases in this analysis, therefore, it has chosen to avoid these assignments by relying on qualitative discussions of uncertainty and variability for parameters where data are insufficient to establish distributions statistically. Reviewers of this preliminary document should not, therefore, rely solely on the quantitative aspects of the analysis provided in the various figures and tables, but should include consideration of the discussions providing qualitative characterizations of important sources of uncertainty and variability.

G.4.2 The Defining Exposure-response Relationship

Regardless of whether one is calculating the Maximum Individual Risk, the average lifetime risk for the U.S. population, the number of cancers, or the fraction of individuals with a probability of cancer above a specified numerical value, the equation relating exposure and the probability of that exposure producing cancer is:

$$LPC = C \times SF \times EF_{CF} \times ED_{CF} \times BR_{CF} \times DF_{CF} \times T_{CF} \times TF_{CF} \times D_{CF} \times SF_{CF} \times S_{CF} \qquad [G-4]$$

where

LPC = the excess lifetime probability that an individual develops cancer as a result of exposure to emissions of a compound; the values on the right-hand side of this equation must be the values for the individual or group being considered; if the values on the right-hand side are the average for a population, then LPC is the fraction of people in the population developing cancer as a result of the exposure; if the values on the right-hand side are for a MEI, then LPC is the probability of cancer for this MEI;

C = the outdoor concentration (mg/m^3) to which this individual or group is exposed; this is specified in the exposure assessment component of this analysis;

SF = the slope factor for this individual or group in units of $(mg/m^3)^{-1}$; this slope factor does not account for differences in the relationship between exposure and response that might exist in extrapolating from occupational study populations from which the slope factors originally were developed (adults under conditions of activity) to the general U.S. population of interest in this report (including all age groups and levels of activity); factors affecting the relationship between exposure and response are accounted for in the remaining terms below;

EF_{CF} = correction factor for the fact that a part of each day is spent indoors rather than outdoors; it is equal to the sum of (I) the fraction of time spent outdoors multiplied by the concentration outdoors and (ii) the fraction of time spent indoors multiplied by the concentration indoors, with this sum then divided by the concentration outdoors;

ED_{CF} = correction factor for the exposure duration for this individual or group, taken to be the ratio of the actual number of years of residence to the number of years assumed in the base-line EPA calculations (70 years);

BR_{CF} = correction factor for the breathing rate, taken as the ratio of the breathing rate in the individual or group (averaged over a 70 year lifetime with age-dependent levels of activity) to that in the population from which the slope factor was estimated (adults engaged primarily in light to heavy activity); there is no need for such a correction factor in instances where the development of the slope factor by the Agency already includes the necessary adjustment;

DF_{CF} = correction factor for the lung deposition fractions, taken as the ratio of the deposition fraction in the individual or group to that in the population from which the slope factor was estimated; there is no need for such a correction factor in instances where the development of the slope factor by the Agency already includes the necessary adjustment;

T_{CF} = correction factor for retention half-times, taken as the ratio of the retention half-time in the individual or group to that in the population from which the slope factor was estimated; only single-exponential retention functions are considered here; there is no need for such a correction factor in instances where the development of the slope factor by the Agency already includes the necessary adjustment;

TF_{CF} = "transformation fraction" correction factor for the conversion (activation or detoxification) from burden to biologically significant burden; since the relevant biotransformation information is not available for the three compounds, this is not considered in the following analysis;

D_{CF} = "dosimetry" correction factor for the conversion from biologically significant burden to dose-rate; since the relevant dosimetric information is not available for the three compounds, this is not considered in the following analysis;

SF_{CF} = correction factor for the CSF, taken as the ratio of the CSF for the individual or group to that used in the baseline risk assessment; this presumes the same speciation as in the epidemiological studies used to generate the CSF initially;

S_{CF} = correction factor for speciation, taken as the ratio of the weighted average of the CSFs for the different species (weighted by fraction of the concentration contributed by each species) in the individual or group to the weighted average in the epidemiological studies used to generate the CSF initially.

It is important to note that in all cases, the correction factors above are in the form of ratios between the general U.S. population (for which inferences are being drawn in the risk assessment) and the populations from which the CSFs were drawn (usually, epidemiological populations characterized by adult ages, relatively healthy individuals, and physiological parameters characteristic of at least light activity rather than rest). It is the uncertainty and variability of these ratios, and not of individual-specific values, that is addressed in the following sections.

G.4.3 Variability in the Exposure-response Relationship

G.4.3.1 Variability of Compound-Independent Values in Equation G-4. The values necessary for Equation G-4 that are assumed not to be a function of the compound being considered are EF_{CF}, ED_{CF}, BR_{CF}, and DF_{CF}.

Exposure Frequency (EF_{CF}): The exposure frequency assumed in the base-line calculation of the probability of cancer was 1.0, interpreted as an assumption that (I) indoor and outdoor concentrations are equal and (ii) that an individual spends all of his or her time within the assigned exposure grid-block during each year. The value of EF_{CF} is the "weighted exposure frequency". This is found by asking the following question: *if the individual spends a fraction of time indoors, where the concentration is equal to the indoor concentration, and a fraction of time outdoors, where the concentration is that used in the base-line calculation, what is the equivalent fraction of time the individual is exposed to a concentration only equal to the outdoor concentration*?

The equation for calculating EF_{CF} is:

$$EF_{CF} = f_i \times C_i + f_o \times C_o \qquad [G-5]$$

For example, if the indoor concentration is 0.2 and the outdoor concentration is 1, and if the fraction of time indoors is 0.5 (with a fraction outdoors equal to 0.5), the "weighted exposure frequency" is $0.5 \times 0.2 + 0.5 \times 1 = 0.6$. This means the actual exposure pattern (50 percent time indoors and 50 percent outdoors) produces the same level of exposure as a pattern in which 60 percent of each day was spent at the outdoor concentration (and the remainder at 0 concentration).

The median value for the ratio of the concentration indoors to that outdoors is 0.4 (EPA, 1989). This ratio will be nearly 1.0 for houses with very large rates of ventilation (so long as ventilation is not dominated by infiltration through cracks, which might act as filters for the arsenic, nickel and chromium particles); and significantly less than 0.4 for homes with very low rates of ventilation. It is assumed here that the upper 95 percent confidence limit is approximately 0.8. Since the median is 0.4, the ratio of the upper limit to the median is 2.0. The GSD then is equal to the square root of 2.0, which is 1.4. Variability of the ratio of indoor to outdoor concentrations is characterized here by a lognormal distribution with a 95 percent confidence interval of [0.2, 0.8], with a median concentration ratio of 0.4 and a geometric standard deviation of 1.4. This distribution is truncated at 0.0 and 1.0 since the ratio of concentrations cannot lie outside these bounds.

The ratio of concentrations indoors to outdoors must be supplemented by an estimate of the fraction of time spent indoors and outdoors, taken here to be given by the fractions f_I (fraction of time indoors) and f_O (fraction of time outdoors). These fractions depend on age and occupation, but the fraction of time outdoors (f_O) typically averages approximately 0.15 and the fraction indoors (f_i) 0.85.[15] The median value of the "weighted exposure frequency" is approximately (1.0 x 0.15) + (0.4 x 0.85) or 0.49, which is interpreted as 11.8 hours per day (0.49 x 24) exposed to the outdoor concentration.

Information on the variability of f_I and f_O is not available, but the fraction of time indoors cannot exceed 1.0 and probably rarely is below 0.5 (allowing for sleeping and normal household activities). This sets an upper bound on f_I of 1.0 and a lower bound of 0.5. The values of f_I and f_O for individuals clearly are completely correlated, since they must sum to 1.0 in all instances. It is assumed here, therefore, that f_I is distributed lognormally with a geometric standard deviation of 1.2 truncated at the boundaries of the interval [0.5, 1]. The value of f_O always is $1 - f_I$. The final "weighted" exposure frequency then is distributed lognormally with a median of 0.49, mean of 0.52 and a geometric standard deviation of approximately 1.4. The distribution is truncated at [0, 1].

The above distribution is representative of a first scenario in which the actual conditions in US homes is considered. As a matter of policy, however, the analysts also have chosen to examine the scenario in which the exposure frequency is 1.0 (i.e. to examine variability conditional upon the assumption that exposures always are at the outdoor concentration). In this second scenario, the mean value for EF_{CF} is 1.0 and there is no variability. Both scenarios are considered in this analysis.

Exposure Duration (ED_{CF}): The exposure duration assumed in the base-line calculations is 70 years. This is consistent with the assumption either that an individual resides the entire 70 years at the same location, or moves to grid blocks with the same average concentration. In empirical studies[16], the median length of time an individual resides in a single location is 9 years. This time is distributed approximately lognormally with a geometric standard deviation of 2.5.[16]

The use of values of ED_{CF} other than 1 is justified only for the calculation of the MIR, and only if it is assumed that ALL individuals living within the grid block characterized by the MEI move out of that grid block upon changing residence. If even one individual remains in the grid block throughout the 70 years, his or her risk is by definition the MIR. Since this does occur, there is no need to incorporate variability of ED_{CF} into the calculation of the MIR. Variability in this factor is not, therefore, considered further here, and a value of 70 years is employed in all calculations.

As to the issue of the variability of the Lifetime Probability of Cancer for the entire U.S. population, it must be noted that the exposed population exceeds 190 million, or more than 70 percent of the U.S. population. It is a reasonable assumption that exposed individuals, upon changing residences, simply change positions with another individual in the exposure field. If the individual moves out of the exposure field, a previously unexposed person moves to that location and contributes equally to the calculation of the mean Lifetime Probability of Cancer for the entire U.S. population and to the number of deaths. The authors know of no empirically-verified assumption as to the distribution of the ratio of initial air concentration (i.e. at the location from which a person moves) over the air concentration at the new place of residence for individuals who move. It is assumed here that all individuals reside in one randomly chosen location throughout their 70 year lifespan. This will tend to slightly overestimate the variation in the Lifetime Probability of Cancer between individuals in the U.S. population, but the magnitude of this overestimate cannot be determined at present due to a lack of data on the distribution of ratios of concentrations before and after migration.

Breathing Rate (BR_{CF}): Measurements of variability (Hattis and Silver, 1994), after factoring out measurement error, indicate the breathing rate is distributed lognormally in adults with a geometric SD of between 1.2 and 1.3. A geometric SD of 1.25 is assumed here as a best estimate.

Mean breathing rates in occupational populations (such as the ones on which the CSFs are based) generally are higher than the mean of 20 m^3/day for the general population (which includes consideration of the difference in breathing rates indoors and outdoors). Assuming primarily light activity in the workers, the breathing rate would be approximately 30 m^3/day. This indicates a mean value of the ratio (BR_{CF}) of 0.67. If the CSFs used in this analysis do not reflect these differences, the value of BR_{CF} would be distributed lognormally with a geometric SD of 1.25, a mean of 20/30 or 0.67, and a median of 0.67 / $\exp(\ln^2 1.25/2)$ or 0.65.

However, it is not currently clear whether the original CSFs included consideration of the different breathing rates in the two populations. Until this issue is resolved, the analysis of variability developed here includes no additional correction for breathing rates, employing a lognormal distribution with a mean of 1.0 and a geometric SD of 1.25. It is recommended that future research focus on resolving this issue.

Deposition Fraction (DF_{CF}): Deposition fractions vary between individuals due to variation in breathing rates, particle sizes and the sizes of lung passages.[17] Crawford-Brown and Hofmann[18] have estimated variability of deposition fractions for a constant particle size and found it to be distributed lognormally with a geometric SD of approximately 2 after accounting for uncertainty due to measurement errors. The differences in deposition fractions between the epidemiological study populations and the general population cannot be determined due to a lack of information on particle size distributions in the two populations. It is assumed here, therefore, that the values of DF_{CF} are distributed lognormally with a geometric SD of 2, mean of 1, and a median of $1.0 / \exp(\ln^2 2/2)$ or 0.79. Future research should focus on determining the particle size distributions of HAPs, and on the implications of this for DF_{CF}.

G.4.3.2 Variability of Compound-Specific Values in Equation G-4. The values necessary for Equation G-4 that are a function of the compound being considered, at least in the present analysis, are T_{CF}, TF_{CF}, D_{CF}, SF_{CF} and S_{CF}.

Retention Half-times (T_{CF}): Retention half-times typically are distributed lognormally[19,20], with a geometric SD of approximately 1.3 after accounting for uncertainty due to measurement error. One problem with such measurements, however, is that they often are over a short-term and conducted only once on each individual in the study population. For the current analysis, it is necessary to determine the variability in the long-term average retention characteristics in individuals (i.e. the half-time averaged over a lifetime of intake, rather than following a single acute exposure as in the experimental measurements). The information on which the analysis of Crawford-Brown[20] was based, however, includes instances of long-term retention of materials such as radium in the bone and plutonium in the lungs.

There is no information to suggest that these long-term retention half-times in the epidemiological study populations differs systematically from those in the general population, although life-time averaged retention half-times should be slightly lower in the general population due to the inclusion of young ages where the retention half-times usually are lower than adult values.[21] The effect of this age-dependence on the three compounds considered here, however, cannot be determined. Variability in T_{CF} for this analysis is assumed to be described by a lognormal distribution with a geometric standard deviation of 1.3, a mean of 1.0, and a median of 1.0 / $\exp(\ln^2 1.3/2)$ or 0.97.

Other (TF_{CF} and D_{CF}): As mentioned previously, there is no information on which to base estimates of the variability of either TF_{CF} or D_{CF} , so these parameters are set here to 1.0 with no variability. This should not introduce significant errors into the analysis of variability since the three compounds being considered are not significantly bioactivated, which is the common source of variability in these two factors when dealing with more complex chemical forms.

Slope Factor (SF_{CF}): The variability in the CSF has not been determined quantitatively for the general population. It has been noted, however, that the arsenic CSF for heavy smokers may be larger than that for never-smokers by as much as a factor of 8[22], suggesting variability in the population with respect to smoking status. The variability of CSFs for arsenic is characterized here by a lognormal distribution with a mean of 1.0 and a geometric SD of 1.5 (a factor of 5 between the upper and lower 95 percent limits). It is not possible at present, however, to provide a quantitative characterization of this variability for nickel and chromium. This variability is discussed in more detail qualitatively in a later section on sensitive subpopulations.

Speciation (S_{CF}): The variability in the speciation factor reflects the difference in CSFs for the different species of each HAPs, and the likely variation in this speciation for exposures to individuals in well defined groups. The variation will be largest for the entire U.S. population (which includes variation in the source terms that contribute to speciation), and smallest for consideration of the MIR (where the speciation within a grid block or at an exposure point will not vary as greatly between exposed individuals). These sources of variability is considered separately here for each of the three HAPs.

Arsenic: Arsenic is found primarily in oxidation states III and V. The best estimate is that 18 percent is in State III and 43 percent in State V, with the remainder in an unspecified oxidation state. At the upper bound, 33 percent is in State III and 67 percent in State V. At the lower bound, 3 percent is in State III and 19 percent in State V, with the remainder in an unspecified oxidation state.

Neither state has been shown to produce mutations[23], but there is limited evidence that both produce chromosomal changes.[24] It is not possible at this time to differentiate the tumorigenicity of these two compounds, so they are considered here to have the same CSF of 0.00429 $(\mu g/m^3)^{-1}$. Since this is the same CSF as was used in the base-line calculations, the mean for this correction factor then is 1.0. No variability is assumed since the two species are assumed to have equal CSFs, although uncertainty is treated in a later section.

Nickel: There are several forms of nickel, with soluble, sulfides and oxides being present. For nickel refinery dust (characteristic of exposures in the epidemiological population), the best estimate of the mean value of the CSF for an exposed population is 2.4×10^{-4} $(\mu g/m^3)^{-1}$. For subsulfidic dust (present in exposures of the general population), the best estimate of the mean CSF is 4.8×10^{-4} $(\mu g/m^3)^{-1}$. In exposures of the public, soluble nickel and oxides dominate, constituting between 90 and 95 percent of the total nickel. In a review by the International Committee on Nickel Carcinogenesis[25], soluble nickel was determined to be the primary carcinogenic form, being approximately a factor of 10 more potent than the less soluble forms. Recent studies in rats and mice conducted by the NTP, however, have found soluble nickel to be not carcinogenic in these species (personal communication, Hudson Bates, NIPERA, September, 1995). The soluble fraction for the general public averages approximately 60 percent of total nickel, but it is not possible to relate this accurately to the fraction in the epidemiological studies since the composition of refinery dust is generally poorly characterized. Variability of speciation, and its effect on variability of CSFs for the general population, cannot, therefore, be considered here. Since the base-line calculation employed the CSF for nickel subsulfide, the mean of this correction factor is assumed to be 1.0.

Chromium: The two compounds of interest are Cr-III+ and Cr-VI+. Initial data indicate that the primary carcinogenic form is Cr-VI+[23], although the issue of the carcinogenicity of Cr-III+ has not been resolved. Until that is resolved, it is not possible to quantitatively characterize the variability of the speciation correction factor, even if data on the degree of variability in speciation itself is available.

G.4.3.3 Summary of Variability. Each of the factors above may be incorporated into Equation G-4 to yield a determination of variability in the cancer risk for an individual at a selected level of exposure. For the present section, which deals only with exposure-response assessment, it is assumed that the concentration of each of the three HAPs is 1.0 $(\mu g/m^3)$. This will be combined with information on variability of exposures in a later section on risk characterization. A summary of the various distributions is provided in Table G-17.

Figure G-12 displays the probability density functions for the separate parameter values used in Equation G-4 for evaluating variability for both estimating incidence in which the exposure frequency correction factor has a mean of 0.52 and for estimating MIR in which the exposure frequency is set a t 1.0. Table G-17 summarizes and Figure G-13 graphically displays the resulting probability density functions for the CSF used to estimate annual incidence for exposures to arsenic, chromium, and nickel at a concentration of 1 µg/m^3 (uniform throughout the exposed population). From these, it may be seen that the 95 percent interval for the relationship between exposure and lifetime probability of cancer for arsenic is approximately 3.8×10^{-4} to 7.9×10^{-3} (the baseline calculation yields 4×10^{-3}); the 95 percent interval for the relationship between exposure and lifetime probability of cancer for nickel is approximately 4.3×10^{-5} to 8.9×10^{-4} (the baseline calculation yields 4.8×10^{-4}); and the 95 percent interval for the relationship between exposure and lifetime probability of cancer for chromium is approximately 2.2×10^{-4} to 4.4×10^{-3} (the baseline calculation yields $0.2 \times 1.2 \times 10^{-2} = 2.4 \times 10^{-3}$; the factor of 0.2 accounts for the assumption in the baseline calculation that Cr(VI) represents 20 percent of the total exposure to chromium). These confidence intervals are particularly valid for the average cancer incidence estimates.

Figure G-14 displays the resulting probability density functions for the Lifetime Probability of Cancer for exposures to arsenic, chromium, and nickel at a concentration of 1 µg/m^3 (uniform throughout the exposed population). This distribution is also summarized in Table G-17. From these, it may be seen that the 95 percent interval for the relationship between exposure and lifetime probability of cancer for arsenic is approximately 8.3×10^{-4} to 1.4×10^{-2} (the baseline calculation yields 4×10^{-3}); the 95 percent interval for the relationship between exposure and lifetime probability of cancer for nickel is approximately 9.3×10^{-5} to 1.5×10^{-3} (the baseline calculation yields 4.8×10^{-4}); and the 95 percent interval for the relationship between exposure and lifetime probability of cancer for chromium is approximately 4.6×10^{-4} to 7.5×10^{-3} (the baseline calculation yields 2.4×10^{-3}). These confidence intervals will be particularly valid for calculations of the MIR estimates.

Table G-17. Summary of Monte Carlo Simulation of Variability in CSF for Estimating MIR and Incidence

		Variability					
		Incidence			MIR		
		Arsenic	Chromium	Nickel	Arsenic	Chromium	Nickel
Mean		2.3E-03	1.3E-03	2.5E-04	4.3E-03	2.4E-03	4.8E-04
EPA Verified URE		4.3E-03	2.4E-03	4.8E-03	4.3E-03	2.4E-03	4.8E-03
(Percentile)		(89)	(89)	(89)	(64)	(64)	(64)
Percentiles	0.0%	1.4E-04	7.7E-05	1.5E-05	2.9E-04	1.6E-04	3.2E-05
	2.5%	3.9E-04	2.2E-04	4.3E-05	8.3E-04	4.6E-04	9.3E-05
	5.0%	4.8E-04	2.7E-04	5.3E-05	1.1E-03	5.9E-04	1.2E-04
	10%	6.2E-04	3.5E-04	6.9E-05	1.4E-03	7.7E-04	1.5E-04
	25%	9.6E-04	5.4E-04	1.1E-04	2.1E-03	1.2E-03	2.3E-04
	50%	1.7E-03	9.3E-04	1.9E-04	3.4E-03	1.9E-03	3.8E-04
	75%	2.8E-03	1.5E-03	3.1E-04	5.4E-03	3.0E-03	6.0E-04
	90%	4.6E-03	2.6E-03	5.2E-04	8.4E-03	4.7E-03	9.4E-04
	95.0%	6.1E-03	3.4E-03	6.8E-04	1.1E-02	6.3E-03	1.3E-03
	97.5%	7.9E-03	4.4E-03	8.8E-04	1.3E-02	7.5E-03	1.5E-03
	100.0%	2.5E-02	1.4E-02	2.8E-03	4.1E-02	2.3E-02	4.6E-03

Figure G-12. Assumptions Used in the Analysis of Variability in CSFs

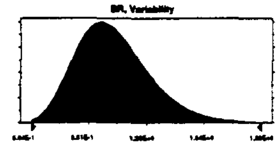
Assumption: BR, Variability

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	2.23E-01

Selected range is from 0.00E+0 to +Infinity

Mean value in simulation was 1.01E+0



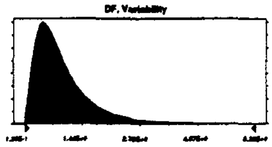
Assumption: DF, Variability

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	6.93E-01

Selected range is from 0.00E+0 to +Infinity

Mean value in simulation was 1.01E+0



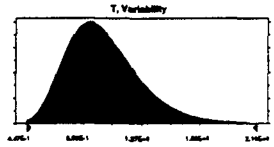
Assumption: T, Variability

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	2.62E-01

Selected range is from 0.00E+0 to +Infinity

Mean value in simulation was 1.00E+0



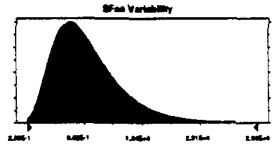
Assumption: SFas Variability

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	4.05E-01

Selected range is from 0.00E+0 to +Infinity

Mean value in simulation was 9.97E-1



Assumption: EF, Var., Incidence (Used in CSF for estimating incidence only)

Lognormal distribution with parameters:

Mean	5.20E-01
Standard Dev.	1.75E-01

Selected range is from 0.00E+0 to +Infinity

Mean value in simulation was 5.15E-1

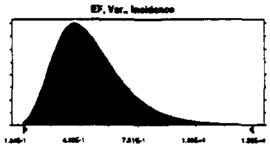


Figure G-13. Distribution of CSFs Used to Estimate Annual Incidence: Variability

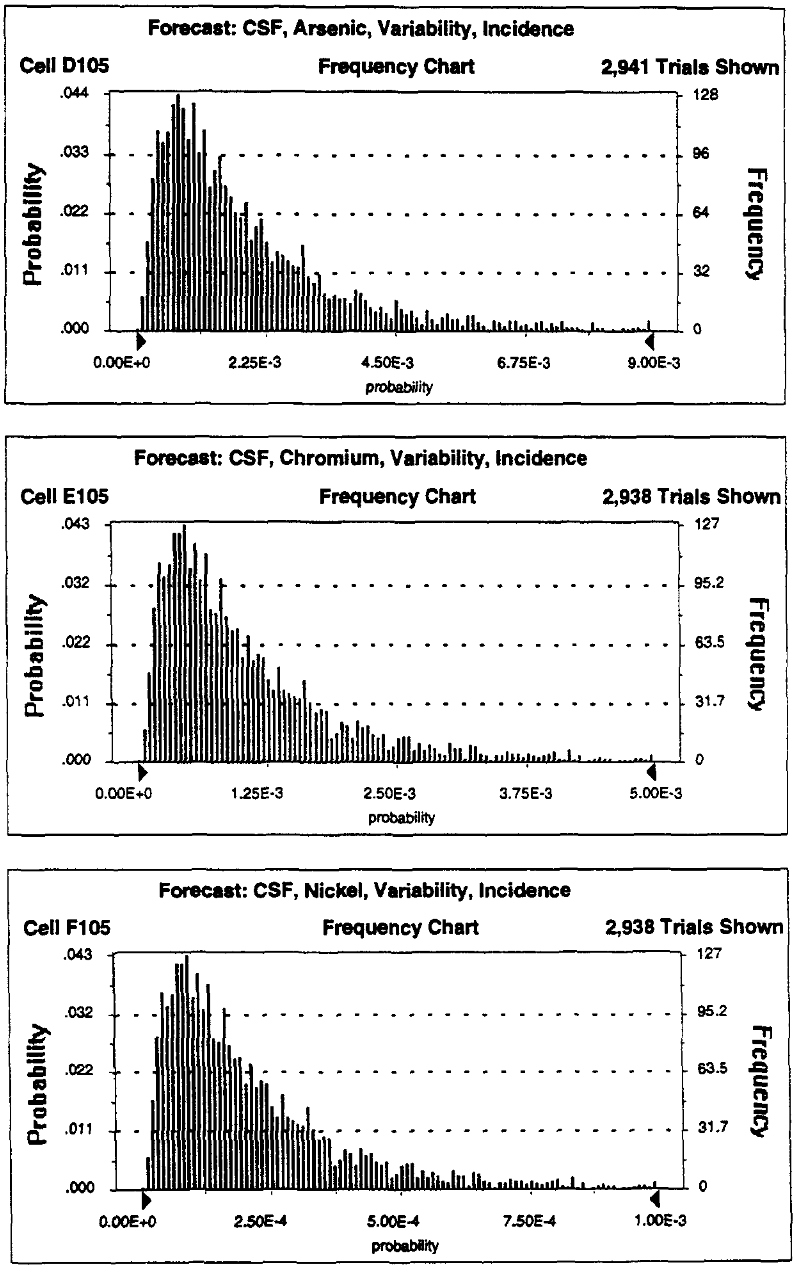
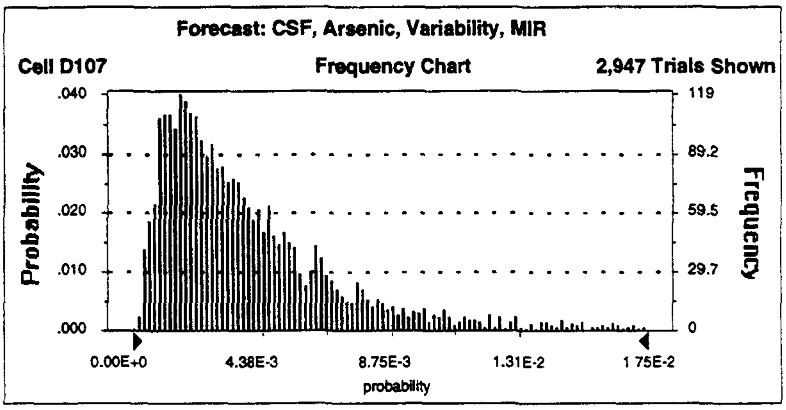
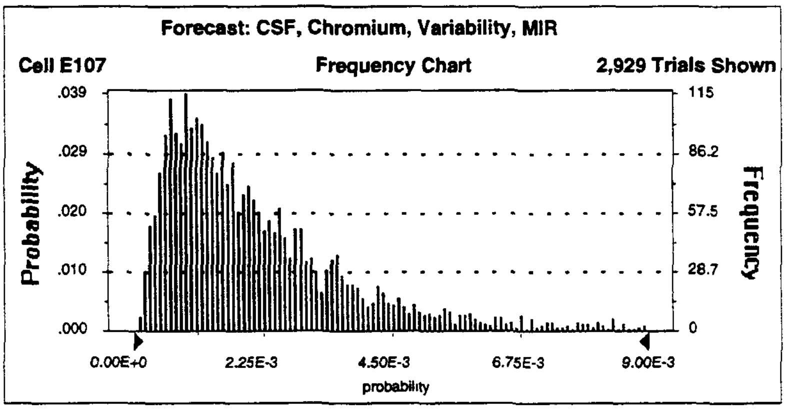
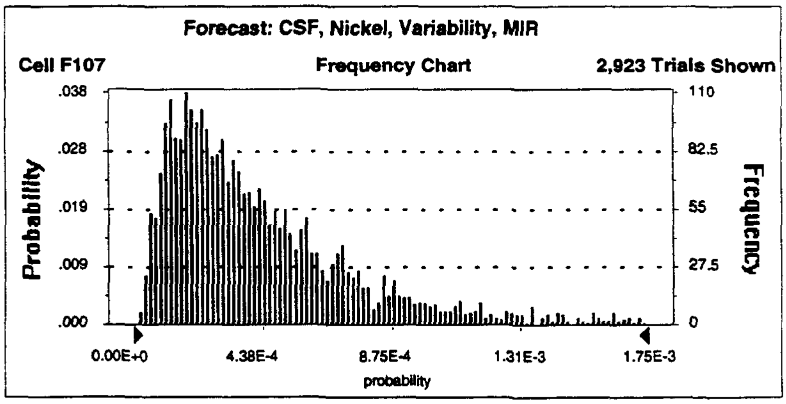


Figure G-14. Distribution of CSFs Used to Estimate MIR: Variability







G.4.4 Uncertainty in the Exposure-response Relationship

G.4.4.1 Uncertainty of Compound-Independent Values in Equation G-4. The values necessary for Equation G-4 that are assumed not to be a function of the compound being considered are EF_{CF}, ED_{CF}, BR_{CF}, and DF_{CF}.

Exposure Frequency (EF_{CF}): The exposure frequency assumed in the base-line calculation is 1.0, interpreted as an assumption that (I) indoor and outdoor concentrations are equal and (ii) that an individual spends all of his or her time within the assigned exposure grid-block. The calculation of EFCF is reviewed in section G.4.3.1 and is not repeated here.

As to the first assumption, the median value for the ratio of the concentration indoors to that outdoors is 0.4 and the mean approximately 0.42.[16] The uncertainty in this value is due primarily to measurement variability and consideration of the representativeness of the sample. The standard error of the mean is approximately 0.05. Given the sample size and the random nature of selection for the population measured, the mean value for the concentration ratio is unlikely to be less than 0.3 or greater than 0.5. A normal distribution with a SD of 0.05 is assumed here.

The ratio of concentrations indoors to outdoors must be supplemented by an estimate of the fraction of time spent indoors and outdoors, taken here to be given by terms f_I (indoors) and f_O (outdoors). These fractions depend on age and occupation, but the fraction of time outdoors typically averages approximately 0.15 and the fraction indoors 0.85.[26] This yields a mean "weighted" exposure frequency of (1.0 x 0.15) + (0.42 x 0.85) or 0.5. Information on the uncertainty of f_I and f_O is not available, but the fraction of time indoors cannot exceed 1.0 and probably rarely is below 0.5 (allowing for sleeping and normal household activities). The values of f_I and f_O for individuals clearly are completely correlated, since they must sum to 1.0 in all instances. In the absence of detailed information on which to estimate uncertainty, the subjective judgment is made here that uncertainty in f_I is characterized by a normal distribution with a mean of 0.85 and SD of 0.05, truncated at the boundary value of 1.0. The value of f_O always is $1 - f_I$. The "weighted" exposure frequency correction factors then is distributed normally with a mean of approximately 0.5 and a SD of approximately 0.05.

The above distribution is representative of a first scenario in which the actual conditions in U.S. homes is considered. However, the scenario in which the exposure frequency is 1.0 was also examined. In this second scenario, the mean value for EF_{CF} is 1.0 and there is no variability. Both scenarios are considered in this analysis. For the MEI, the second scenario is considered more appropriate.

Exposure Duration (ED_{CF}): The exposure duration assumed in the base-line calculation is 70 years. This is consistent with the assumption either that an individual resides the entire 70 years at the same location, or moves to grid blocks with the same average concentration.

The use of values of ED_{CF} other than 1 is justified only for the calculation of the MIR, and only if it is assumed that ALL individuals living within the grid block characterized by the MEI move out of that grid block upon changing residence. If even one individual remains in the grid block throughout the 70 years, his or her risk is by definition the MIR. Since this does occur, there is no need to incorporate uncertainty of ED_{CF} into the calculation of the MIR. Uncertainty in this factor is not, therefore, considered further here, and a value of 70 years is employed in all calculations.

As to the uncertainty in cancer incidence, the exposed population exceeds 190 million, or more than 70 percent of the U.S. population; therefore, it is a reasonable assumption that exposed individuals, upon changing residences, simply change positions with another individual in the exposure field. If the individual moves out of the exposure field, a previously unexposed person moves to that location and contributes equally to the calculation of the number of deaths. This change will have no effect on the estimated incidence. It is assumed here that individuals reside in one randomly chosen location throughout their 70 year lifespan, and that uncertainty in ED_{CF} is not significant.

Breathing Rate (BR_{CF}): Measurements of variability[27], corrected for measurement error, indicate the breathing rate is distributed lognormally in adults with a geometric SD of between 1.2 and 1.3. These same measurements indicate that the SE is approximately 3 percent of the mean value. This must be broadened since it does not reflect uncertainty in the effect of introducing differences in age distribution and level of activity, issues of accuracy of the measurement techniques, or issues of the representativeness of the sampled population. The subjective judgment here is that these factors should produce a distribution which is characterized by lognormality with a geometric SD of at least 1.1 for the ratio of lifetime-averaged breathing rates in the two populations, given that inclusion of age-dependence in the calculation of life-time averaged breathing rates can cause changes as large as 10 percent.

The mean breathing rates in occupational populations (such as the ones on which the CSFs used in the present analysis are based) generally are higher than the mean of 20 m^3/day for the general population. Assuming primarily light activity in the workers, the breathing rate would be approximately 30 m^3/day. As of the publication of this preliminary report, however, it is not clear whether the original CSFs included consideration of the different breathing rates in the two populations.

Deposition Fraction (DF_{CF}): Deposition fractions vary between individuals due to variation in breathing rates, particle sizes and the sizes of lung passages.[17] Crawford-Brown and Hofmann[18] have estimated variability of deposition fraction for a constant particle size and found it to be distributed lognormally with a geometric SD of approximately 2 after accounting for measurement uncertainty. The SD of the estimate of the mean deposition fraction is on the order of 10 percent of the mean when particle size is known. This must be broadened since it does not reflect uncertainty in the effect of introducing differences in age distribution and level of activity, issues of accuracy of the measurement techniques, or issues of the representativeness of the sampled population. The differences in deposition fraction between the epidemiological study populations and the general population were not included in the original development of the CSF by the Agency and cannot be determined due to a lack of information on particle size distributions in the two populations. This should broaden the distribution for uncertainty, but the degree of broadening cannot be well specified at present quantitatively. The subjective judgment here is that these factors should produce a distribution which is characterized by lognormality with a geometric SD of at least 1.25 for the ratio of lifetime-averaged deposition fractions in the two populations, given that changes in particle size within the range of values typically found in the ambient air (between 0.05 and 0.5 microns) can cause changes as large as a factor of two or more.

G.4.4.2 Uncertainty of Compound-Specific Values in Equation G-4. The values necessary for Equation G-4 that are a function of the compound being considered, at least in the present analysis, are EF_{CF}, ED_{CF}, BR_{CF}, and DF_{CF}.

Retention Half-times (T_{CF}): Retention half-times typically are distributed lognormally[19,20], with a median value of the geometric SD of approximately 1.3. There is no information to suggest that retention half-times in the epidemiological study populations differs systematically from those in the general population, although life-time averaged retention half-times should be slightly lower due to the inclusion of young ages where the retention half-times usually are lower than adult values (Crawford-Brown, 1984).

The Agency did not include a correction for retention half-times in the original analyses for the CSFs. The effect of this age-dependence on the three compounds considered here, however, cannot be determined directly due a lack of data on the effects of age on these compounds. It is assumed here that the study populations consisted primarily of adults, and that the general U.S. population contains all ages. Inclusion of younger ages into a calculation of the lifetime-averaged value of the half-time generally produces values that are within 10 percent of the adult value (based on information provided in Crawford-Brown, 1984), although it should be noted that the life-time averaged value usually is lower than the adult value. For this analysis, the subjective judgment of the analysts is that uncertainty in the estimate of the mean value of T_{CF} for the exposed population is described by a normal distribution with a mean of 1.0, and a SD of 0.1.

Other (TF_{CF} and D_{CF}): There is no information on which to base estimates of the uncertainty of either TF_{CF} or D_{CF} , so these parameters are set here to 1.0 with no uncertainty. This should not introduce significant errors into the analysis of uncertainty since the three compounds being considered are not significantly bioactivated, which is the common source of variability in these two factors when dealing with more complex chemical forms.

Slope Factor (SF_{CF}): The uncertainty in the CSF has several components. There is uncertainty due to parameter estimation for a selected extrapolation function (such as the linearized multistage model) applied to a single data set, which may be summarized by the standard error of the CSF for that data set, broadened by consideration of study design. There is uncertainty due to the existence of different data sets, which may be characterized by the standard error of the mean value of the CSF across those data sets, broadened by issues of representativeness of the sampled populations. There is uncertainty due to the inability to choose between competing exposure-response models used for extrapolation. And there is uncertainty introduced by potential differences in concurrent exposures to synergistic and/or antagonistic risk agents (such as cigarette smoke) in the general U.S. and study populations.

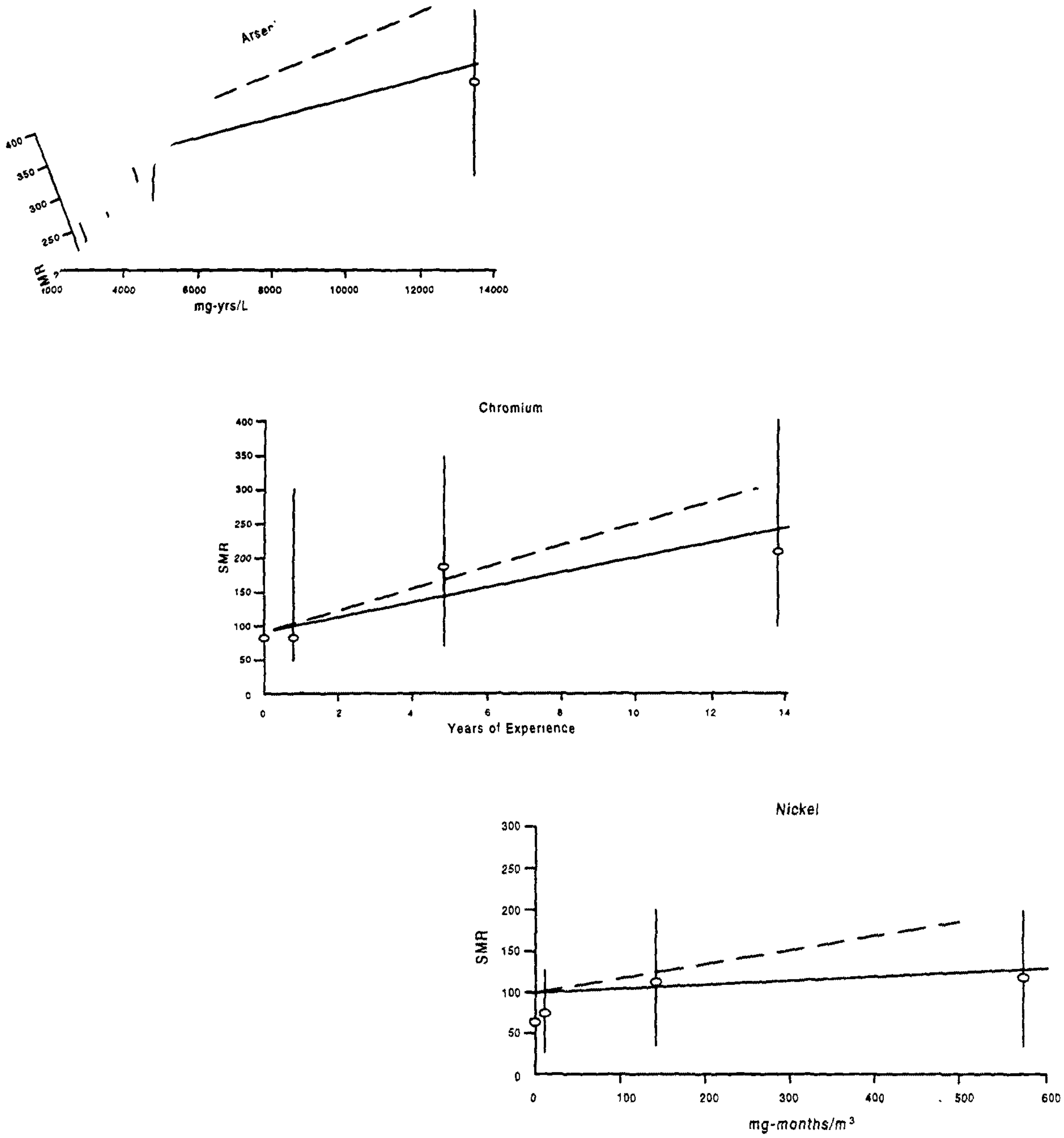
To characterize uncertainty from these sources, the authors analyzed the results of selected epidemiological studies on exposures to airborne arsenic[28], nickel[29] and chromium[30], as well as the differences in CSFs determined from different populations exposed to these compounds and from using different exposure-response functions. These uncertainties are discussed here for the three separate materials. Consideration of the uncertainty introduced by use of extrapolation functions is discussed qualitatively in a later section.

For each of the three compounds, normal distributions were converted to lognormal distributions with equivalent 68 percent confidence intervals to avoid negative values for the CSFs, and to better reflect issues of representativeness and study design (best described by lognormal distributions). The "equivalent lognormal" characteristics are described below.

Arsenic: The studies examined for the maximum likelihood estimates of the CSF varied between 1.25×10^{-3} and 7.6×10^{-3} $(\mu g/m^3)^{-1}$.[28,31,32,33] The base-line calculations of the risk from arsenic employed a mean value of 0.00429 $(\mu g/m^3)^{-1}$; the standard error of the mean for these studies is approximately 0.002 $(\mu g/m^3)^{-1}$. A recent analysis by Virens and Silvers[34] yielded a value of 0.00143 $(\mu g/m^3)^{-1}$, which is within the range of values noted above.

Uncertainty due to imprecision in the estimate of the CSF from a single data set may be approximated by examining a representative study on arsenic[28], as shown in Figure G-15. The standard error in the CSF for these data obtained from a linear model is approximately 70 percent of the mean (maximum likelihood) estimate, with a lower bound estimate that includes a CSF of 0.0. The best estimate is shown as the solid line in that figure, with the upper 68 percent function shown as the dashed line. This suggests an intra-study standard error of approximately 0.003 $(\mu g/m^3)^{-1}$. This is similar in magnitude to the standard error noted in the previous paragraph and also is similar in magnitude to the difference in maximum likelihood estimates when the linearized multistage and Weibull models are applied to the same data.

The effect of differences in smoking prevalence for the study and general U.S. populations cannot be determined quantitatively at present. The arsenic CSF for heavy smokers may be larger than that for never-smokers by as much as a factor of 8[22], but the effect on the present analysis cannot be determined rigorously at present.



This figure displays the data and model fits used in the uncertainty analyses for slope factors in the case of arsenic, nickel and chromium. In each case, the measure of effect is the SMR The value of (SMR-100 percent) is assumed to be proportional to the excess lifetime probability of cancer The open squares are the original data points from the studies cited in the text. The vertical lines around these squares are the 95 percent confidence limits, determined through Poisson statistics. The solid sloped line is the best fitting linear model as determined from least squares in which the weighting factor for each data point is the inverse of the standard error of the mean. The dashed line is the upper 68 percent confidence interval on the slope.

For each substance, the Weibull model is given by the equation:

$$(SMR\text{-}100\%)/100\% = 1 - \exp(-(a+bD^m))$$

where a, b and m are constants. The value of m is equal to 1.0 for all three substances examined here. Best-fitting values of a and b were determined from least squares analysis in which the weighting factor for each data point is the inverse of the standard error of the mean. The best-fitting linear equation and the Weibull model then were used to predict the value of (SMR-100 percent) at a dose equal to 5 percent of the maximal dose in the epidemiological studies. The ratio of these two predictions is reported in the text.

Figure G-15. Data and Model Fits for Slope Factors for Arsenic, Chromium, and Nickel.

Summarizing, the above sources of uncerta of 0.002 $(\mu g/m^3)^{-1}$ from intra-study uncertainty; of 0.003 $(\mu g/m^3)^{-1}$ from inter-study uncertainty, indicate a SD of exposure-response models; and an uncertainty d dard error exposures that cannot be characterized quantitativ g choice other than to state that it should be of a magnitud the sources noted above. Taking these considerations current the equivalent lognormal distribution has a mean of 1 sent geometric SD of 3. o

Nickel: The different studies of the CSF for nickel exposures[7] varied by a factor of approximately 5 from low around the mean estimate of 0.00048 $(\mu g/m^3)^{-1}$. This is simi the case for arsenic. If the study by Enterline and Marsh[29] shown in Figure G-15 is used to estimate uncertainty conditi on a single data set and the linearized multi-stage model, the of the CSF is approximately 80 percent of the mean, which is similar to that found for arsenic and also is similar in magnitude to the difference in maximum likelihood estimates when the linearized multistage and Weibull models are applied to the same data. Taking these considerations together, the equivalent lognormal distribution has a mean of 1.0 and a geometric SD of 3.

Chromium: Only one study[35] provides a quantitative estimate of the CSF for chromium exposures, with the other studies being restricted to length of exposure. As seen in Figure G-15, the SD of the CSF is approximately 85 percent of the mean, which is similar to that found for nickel and arsenic and also is similar in magnitude to the difference in maximum likelihood estimates when the linearized multistage and Weibull models are applied to the same data. The mean estimated by the Agency is 0.012 $(\mu g/m^3)^{-1}$ Taking these considerations together, the equivalent lognormal distribution has a mean of 1.0 and a geometric SD of 3.

Speciation (S_{CF}): The uncertainty in the influence of speciation on the CSF cannot be determined accurately since the composition in the epidemiological studies is characterized poorly. For chromium, uncertainty in S_{CF} is related directly to the uncertainty in the difference in Cr-VI+/Cr-III+ speciation fraction in the general U.S. and study populations, and the influence of this difference on the "species-averaged" CSF (i.e. the CSF weighted by the fraction of the exposure from each species). It is not possible at present to quantitatively characterize the uncertainty in the CSF for chromium. The same is true for arsenic.

The distribution for nickel is based on the recognition that the CSF for the subsulfide nickel (characteristic of general U.S.

exposures) is a factor of 2 above that for refinery nickel (characteristic of the exposures for the epidemiological population). The more recent NTP studies, however, suggest that soluble nickel may not be carcinogenic in rats and mice, which would tend to increase the confidence in CSFs below the best estimates used in the baseline analysis of risk. The CSF for the general population should be intermediate between these values, given the likely inclusion of both species of nickel to some degree. The CSF chosen in the base-line analysis, however, included only subsulfide nickel. As a conservative assumption, and based on subjective judgment (supported by the fact that the range of a factor 2 between the two species implies that the mean will not be incorrect by more than 40 percent even if one species dominates), the uncertainty for S_{CF} for nickel is given here by an equivalent lognormal distribution has a mean of 1.0 and a geometric SD of 1.2.

G.4.4.3 Summary of Uncertainty. Uncertainty in each of the factors above may be incorporated into Equation G-4 to yield a determination of uncertainty in the Lifetime Probability of Cancer for an exposed individual at a selected level of exposure. For the present section, which deals only with exposure-response assessment, it is assumed that the concentration is 1.0 ($\mu g/m^3$). This will be combined with information on uncertainty of exposures in a later section on risk characterization.

Figure G-16 displays the probability density functions for the separate parameter values used in Equation G-4 for the first scenario in which the exposure frequency correction factor has a mean of 0.52. Figure G-17 displays the resulting probability density functions for the CSF used to estimate annual incidence for exposures to arsenic, chromium, and nickel at concentrations of 1 $\mu g/m^3$ (uniform throughout the exposed population). Table G-18 summarizes the distribution. From these, it may be seen that the 95 percent interval for the relationship between exposure and lifetime probability of cancer for arsenic is approximately 2.2×10^{-4} to 8.0×10^{-3} (the baseline calculation yields 4×10^{-3}); the 95 percent interval for the relationship between exposure and lifetime probability of cancer for nickel is approximately 2.3×10^{-5} to 9.2×10^{-4} (the baseline calculation yields 4.8×10^{-4}); and the 95 percent interval for the relationship between exposure and lifetime probability of cancer for chromium is approximately 1.3×10^{-4} to 4.5×10^{-3} (the baseline calculation yields 2.4×10^{-3}). These confidence intervals will be particularly valid for calculations of the average incidence in the U.S. population.

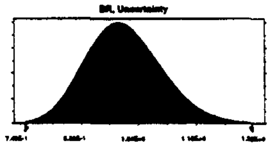
Figure G-16. Assumptions Used in the Analysis of Uncertainty in CSFs

Assumption: BR, Uncertainty

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	9.50E-02

Selected range is from 0.00E+0 to +Infinity
Mean value in simulation was 9.98E-1

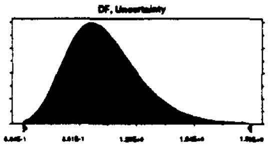


Assumption: DF, Uncertainty

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	2.23E-01

Selected range is from 0.00E+0 to +Infinity
Mean value in simulation was 1.00E+0



Assumption: T, Uncertainty

Normal distribution with parameters:

Mean	1.00E+00
Standard Dev.	1.00E-01

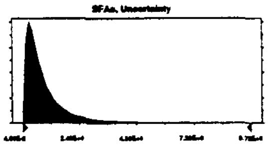
Selected range is from -Infinity to +Infinity
Mean value in simulation was 1.00E+0

Assumption: SFAs, Uncertainty

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	1.10E+00

Selected range is from 0.00E+0 to +Infinity
Mean value in simulation was 9.85E-1

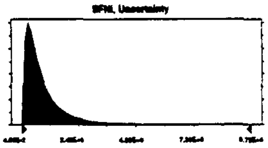


Assumption: SFNi, Uncertainty

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	1.10E+00

Selected range is from 0.00E+0 to +Infinity
Mean value in simulation was 9.85E-1

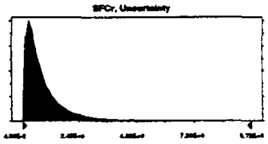


Assumption: SFCr, Uncertainty

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	1.10E+00

Selected range is from 0.00E+0 to +Infinity
Mean value in simulation was 9.90E-1

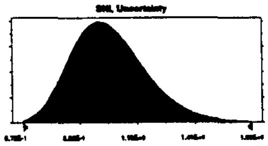


Assumption: SNi, Uncertainty

Lognormal distribution with parameters:

Mean	1.00E+00
Standard Dev.	1.82E-01

Selected range is from 0.00E+0 to +Infinity
Mean value in simulation was 1.00E+0



Assumption: EF, Uncertainty (Used for CSF to Estimate Incidence Only)

Normal distribution with parameters:

Mean	5.00E-01
Standard Dev.	5.00E-02

Selected range is from -Infinity to +Infinity
Mean value in simulation was 5.01E-1

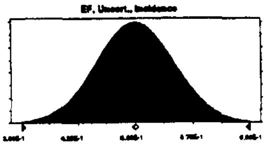
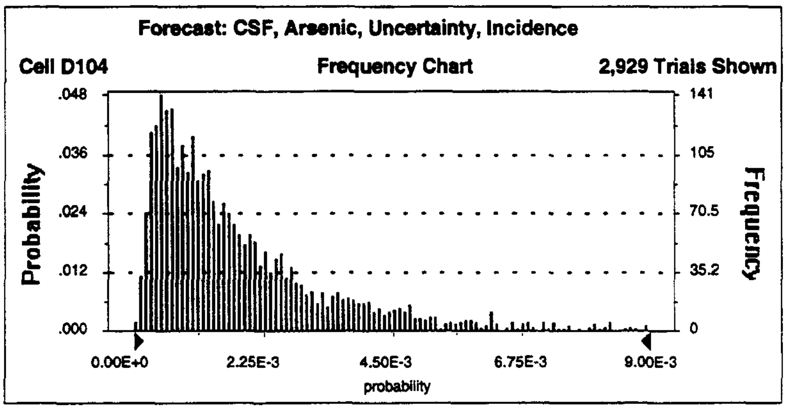
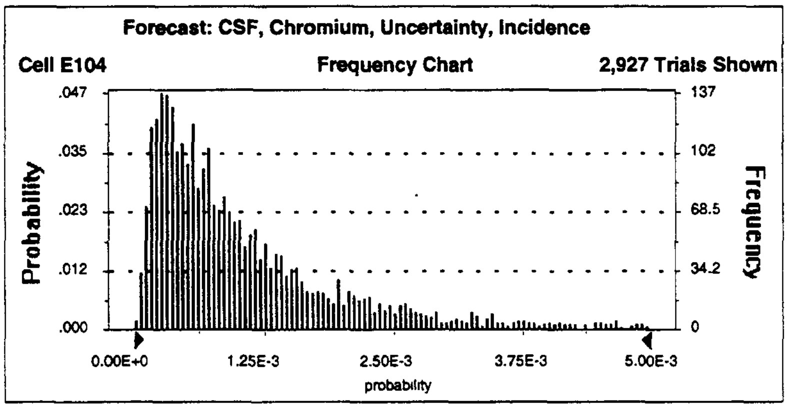


Figure G-17. Distribution of CSFs Used
to Estimate Annual Incidence: Uncertainty





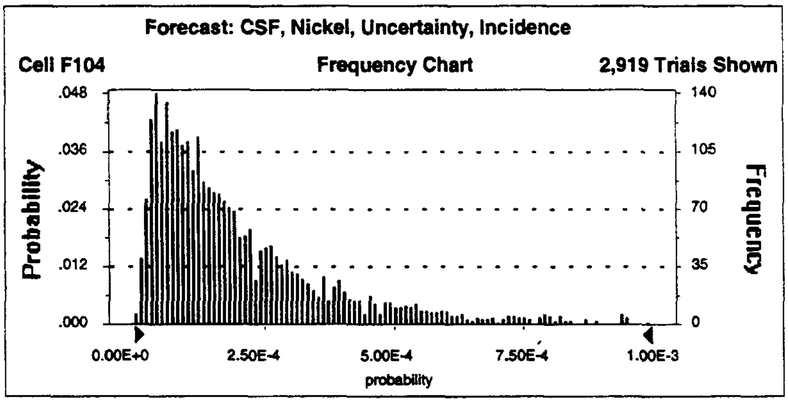


Table G-18. Summary of Monte Carlo Simulation of Uncertainty in CSF for Estimating MIR and Incidence

	Uncertainty					
	Incidence			MIR		
	Arsenic	Chromium	Nickel	Arsenic	Chromium	Nickel
Mean	2.1E-03	1.2E-03	2.4E-04	4.3E-03	2.4E-03	4.8E-04
EPA Verified URE	4.3E-03	2.4E-03	4.8E-03	4.3E-03	2.4E-03	4.8E-03
(Percentile)	(88)	(88)	(88)	(67)	(67)	(67)
Percentiles:						
0.0%	6.7E-05	3.7E-05	6.8E-06	9.1E-05	5.1E-05	8.5E-06
2.5%	2.2E-04	1.3E-04	2.3E-05	4.8E-04	2.7E-04	4.9E-05
5.0%	3.0E-04	1.7E-04	3.2E-05	6.2E-04	3.5E-04	6.7E-05
10%	4.1E-04	2.3E-04	4.5E-05	8.6E-04	4.8E-04	9.3E-05
25%	7.3E-04	4.1E-04	8.0E-05	1.5E-03	8.6E-04	1.6E-04
50%	1.4E-03	7.8E-04	1.5E-04	2.7E-03	1.5E-03	3.0E-04
75%	2.6E-03	1.4E-03	2.9E-04	5.1E-03	2.9E-03	5.7E-04
90%	4.4E-03	2.5E-03	5.1E-04	9.4E-03	5.3E-03	1.1E-03
95.0%	6.2E-03	3.5E-03	7.1E-04	1.3E-02	7.5E-03	1.4E-03
97.5%	8.1E-03	4.5E-03	9.2E-04	1.7E-02	9.7E-03	2.1E-03
100.0%	2.7E-02	1.5E-02	3.5E-03	5.2E-02	2.9E-02	5.7E-03

Figure G-18 displays the resulting probability density functions for the CSF used to estimate MIR for exposures to arsenic, chromium, and nickel at a concentration of 1 µg/m^3 (uniform throughout the exposed population). From these, it may be seen that the 95 percent interval for the relationship between exposure and lifetime probability of cancer for arsenic is approximately 4.8×10^{-4} to 1.7×10^{-2} (the baseline calculation yields 4×10^{-3}); the 95 percent interval for the relationship between exposure and lifetime probability of cancer for nickel is approximately 4.9×10^{-5} to 2.1×10^{-3} (the baseline calculation yields 4.8×10^{-4}); and the 95 percent interval for the relationship between exposure and lifetime probability of cancer for chromium is approximately 2.7×10^{-4} to 9.7×10^{-3} (the baseline calculation yields 2.4×10^{-3}). These confidence intervals will be particularly valid for calculations of the MIR.

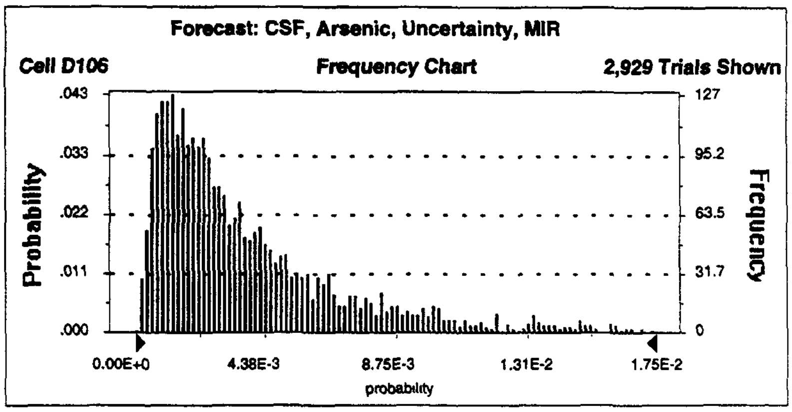
These distributions do not include detailed quantitative consideration of uncertainties treated purely qualitatively, such as the uncertainty introduced by dose-response extrapolation (for all compounds) and speciation (for chromium and arsenic). As a result, the quantitative confidence intervals and distributions presented here should be viewed as being more narrow than those that would be produced if more formal consideration were given to these other sources of uncertainty.

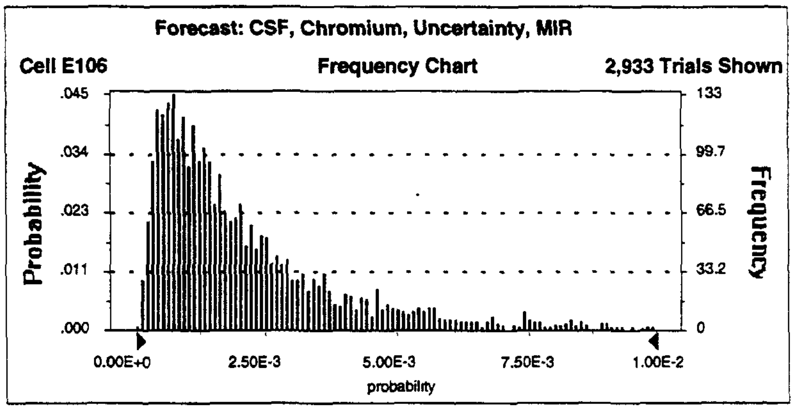
G.4.5 Susceptibility and Additional Factors Affecting Uncertainty in Exposure-Response

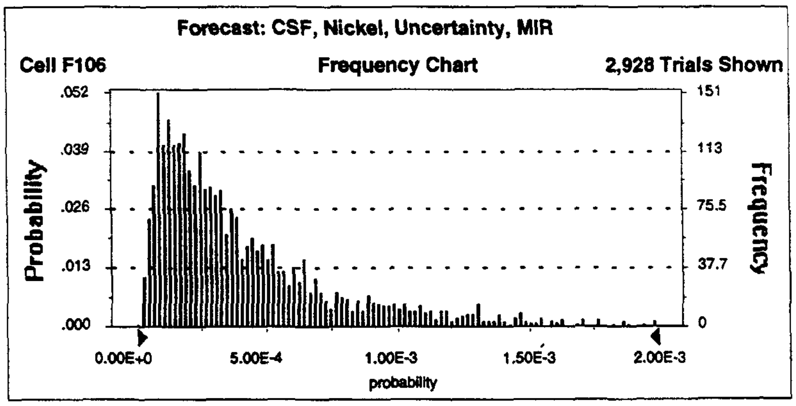
Factors leading to susceptibility for cancer induction following exposure to environmental pollutants (such as arsenic, chromium and nickel) have been reviewed recently by the National Research Council.[2] These factors relate to each of the stages of calculation employed in the present analysis: exposure frequency and duration; breathing rate; deposition fraction; retention half-time; carcinogen metabolism; and dose-response relationships. Susceptible individuals will be those:

1. At the upper end of the distribution of indoor/outdoor concentration ratios;
2. At the upper end of the distribution of fraction of time spent outdoors;
3. At the upper end of the distribution of breathing rates;
4. At the upper end of the distribution of deposition fractions;

Figure G-18.
Distribution of CSFs Used to Estimate MIR: Uncertainty







5. At the lower end of the distribution of retention half-times;

6. At the upper end of the distribution of metabolic activation rate constants (although this factor probably is of little significance for elemental compounds such as arsenic, nickel and chromium; it only is significant for chemicals requiring bioactivation);

7. Experiencing depression and/or stress;

8. With concurrent respiratory tract infections and bronchitis (disturbing pulmonary clearance or producing scarring which has been associated with promotional effects);

9. With depressed DNA repair rates (e.g. xeroderma pigmentosum;

10. Significant histories of smoking, which has been demonstrated to greatly increase susceptibility to several of the carcinogens considered here). Active smokers are likely to have a Lifetime Probability of Cancer from exposure to these pollutants which is higher than for non-smokers.

In addition to these factors which predispose individuals, and which account for intersubject variability of exposure-response relationships, it should be noted that there are both synergistic and antagonistic effects of exposures. A clear synergistic effect already has been noted in the case of smoking; others include concurrent exposures to viruses and alcoholic beverages. The CSFs employed in this study have been developed primarily from occupational studies. These studies often include concurrent exposures, which have not always been accounted for in the epidemiological analysis. As a result, it is not possible at present to quantify the effect of these concurrent exposures, or even to determine whether their presence will cause an under or over-estimation of the risks following exposure of the general U.S. population.

The effects of some of these factors on inter-individual susceptibility have been discussed in the analysis of variability in Section G.4.3. Quantitative estimates of variability in CSFs are more difficult to obtain, although it should be noted that much of the intersubject variability in this factor usually derives from differences in metabolic activation that will be relatively unimportant here. Still, the NRC[2] review indicates that variability of the CSF should be approximately lognormal with a geometric SD on the order of 2.0 to 3.0. A similar value was used in the present analysis (Section G.4.3.2.).

It must also be recognized that use of linearized extrapolation models has been established as policy within the Agency, at least until such time as other extrapolation functions have been shown to be appropriate. This determination has been based partially on the assumption that dose-response equations should be approximately linear at low doses and dose-rates. Recent research into components of carcinogenesis other than initiation, such as research on promotion, indicate that the assumption of linearity (and even the assumption of a lack of threshold) may not be correct as a general rule[36], although it might be correct in specific cases particularly when a substance acts primarily by initiation and when there is large inter-subject variability.[18,20]

The influence of alternative dose-response models, such as distributed threshold models with no linearized term applied to promoting agents and/or agents that produce significant hyperplasia, generally is to lower estimates of the probability of cancer from exposures at environmental levels below those estimated by extrapolations using the linearized multistage model. The possibility of thresholds, particularly for promotional agents, introduces the possibility that confidence intervals on the probabilities of cancer will include a probability of 0.0 for some substances. For arsenic, nickel and chromium, this possibility is lessened by findings of mutational activity typically associated with initiation. Further quantitative analysis of uncertainties introduced by alternative dose-response models is not possible at present.

In their review, the NRC[2] was divided as to the magnitude of reasonable values of susceptibility factors (defined as the ratio of the response in susceptible subpopulations to the average in the general population). Some members suggested that a factor of 10 is appropriate, with others suggesting that this was too high at present. As will be shown in Section G.5 of the present analysis, the baseline calculations of probability of cancer fall above the 90th percentile of most of the distributions. Since the geometric SD for variability is on the order of 3 in this study, the present analysis is consistent with the claim that the baseline analysis already is representative of risks to susceptible subpopulations. This claim is strengthened by the lack of application of a dose-rate effectiveness factor which accounts for the generally lowered susceptibility of individuals to a given dose when the dose-rate is lowered to levels typical in environmental exposures (although such a dose-rate effect has not been demonstrated to date for the three compounds considered here, and the "general rule" of effectiveness increasing with increasing dose-rate is not uniformly applicable across all risk agents).

G.5 EVALUATION OF UNCERTAINTY IN ESTIMATES OF RISK FROM DIRECT INHALATION OF ELECTRIC UTILITY HAP EMISSIONS

As presented in Section G.2, risks from direct inhalation of HAP emissions from utilities have been calculated. This section presents the risk equations and presents overall uncertainty and variability in the risk estimates.

G.5.1 Defining the Risk Characterization Equation

The generalized equation to estimate both individual and population risk on a plant-specific basis can be summarized as:

$$Risk = E x D x CSF \qquad [G-6]$$

where

Risk = excess probability of cancer (individual risk) or cancer cases (population risk)[a]

E = emissions (kg/yr)

D = unit dispersion coefficient per emissions of 1 kg/yr
$\mu g/m^3/kg/yr$ (= maximum concentration: individual)
($\mu g/m^3$ x persons)/kg/yr (= total exposure: population)

CSF = cancer CSF
per $\mu g/m^3$ (individual risk)
per $\mu g/m^3$ x persons (population risk).

The uncertainty associated with each parameter in this equation has been previously evaluated. Section G.3.1 addressed the uncertainty associated with E (emissions); Section G.3.2, with D (dispersion and exposure modeling), and Section G.4, with CSF (CSF, exposure-response relationship).

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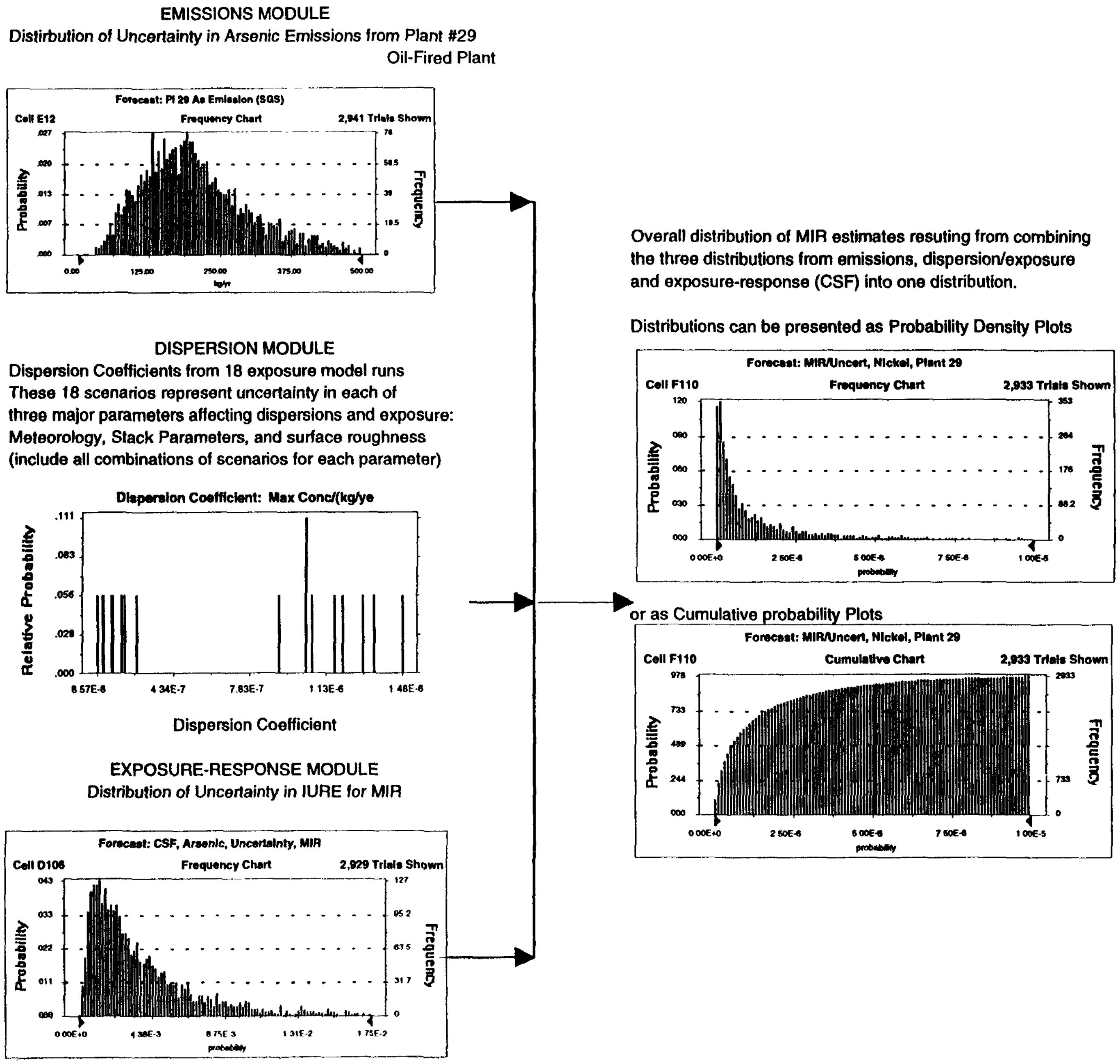
[a] To obtain annual incidence, the number of cancer cases is divided by 70 years (the assumed human lifespan) to calculate annual incidence in units of cancer cases per year.

G.5.2 Uncertainty in Risk Estimates

The overall uncertainty associated with the risk estimates can be estimated by combining the results (distributions) derived in Sections G.3.1, G.3.2, and G.4. The forecast distributions for each of these parameters were used as an input distribution to calculate individual risk to the MEI and cancer incidence. Crystal Ball was again used to conduct Monte Carlo modeling. This was done for each of the four utility plants. The process of deriving an overall distribution on risk estimates is depicted in Figure G-19, and addresses the risks associated with nickel emissions from Plant #29. Furthermore, for one oil-fired and one coal-fired plant, the simulation was conducted using distributions for all of the parameters directly so that an overall sensitivity analysis could be conducted.

G.5.2.1 Uncertainty in Individual Risk Estimates. The uncertainty in the estimates due to exposure to arsenic, chromium, and nickel emissions from Plant #29 is statistically summarized in Table G-19 and graphically depicted in Figure G-20. The original risk estimates were developed using a combination of point values – some conservative and some not conservative – which yield a point estimate of exposure and risk that falls at an unknown percentile of the full distributions of exposure and risk. The distributions presented here now give some indication of the degree of conservatism in those calculations. Results of the uncertainty analysis indicate that the baseline risk estimates are conservative, but not overly conservative. The previous point estimates of MIR are associated with about the 71st or 96th percentile on the overall distribution of possible MIR risk estimates for arsenic (depending on which oil concentration data are used), 87th or 98th percentile for chromium, and 85 or 90th percentile for nickel when the uncertainties related to emissions, dispersion and exposure modeling, and exposure-response are incorporated. Figure G-21 presents a cumulative probability plot of MIR from Plant #29 considering the uncertainty in the MIR estimate. The MIR using the FCEM, and SGS data are presented along with a combined plot assuming that each of the oil concentration data sets are equally likely. Comparison of a high-end risk descriptor (typically the 95th percentile) with a central tendency risk descriptor (median or mean) is also a useful measure of the degree of uncertainty. The 95th percentile (a typical high-end risk estimate) of the overall distribution is roughly 10 to 20 times the median MIR risk estimate for the three HAPs, indicating a geometric SD of approximately 3.8. The 95th percentile was roughly 5 times the mean MIR estimate for each of the three HAPs.

Figure G-19. DEPICTION OF COMBINING COMPONENT UNCERTAINTY DISTRIBUTIONS (*i.e.*, EMISSIONS, DISPERSION, and EXPOSURE-RESPONSE) INTO AN OVERALL DISTRIBUTION ON UNCERTAINTY (e.g., MIR)



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

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

Dispersion Coefficient

EXPOSURE-RESPONSE MODULE
Distribution of Uncertainty in IURE for MIR

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

Distributions can be presented as Probability Density Plots

or as Cumulative probability Plots

Table G-19. Distribution of MIR and Annual Incidence: Plant #29: Comparison of FCEM and SGS Concentration Data

MIR, Plant #29

Uncertainty

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

Variability

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

FCEM = Original oil concentration data, distribution defined by probability plotting technique.

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

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

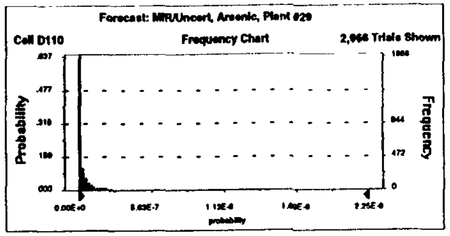
Initial Point Estimate = The estimate of emissions used in the baseline exposure assessment.
This value was based on the average concentration in the FCEM data.

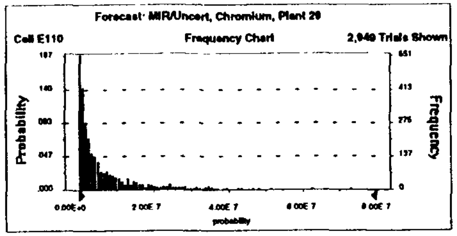
(Percentile) = The percentile of the predicted distribution corresponding to the initial point estimate.

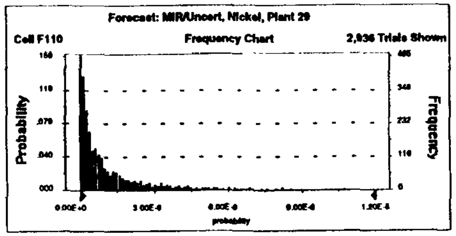
Figure G-20. Distributions of MIR: Plant #29, Comparison of FCEM and SGS Oil Concentration Data

Forecast: MIR/Uncert, Arsenic, Plant #29

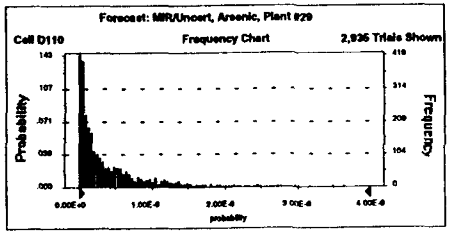
FCEM Concentration Data

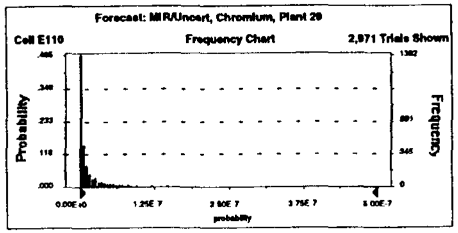


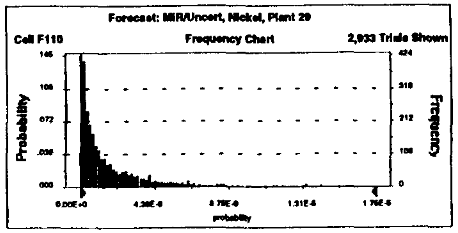




SGS Concentration Data

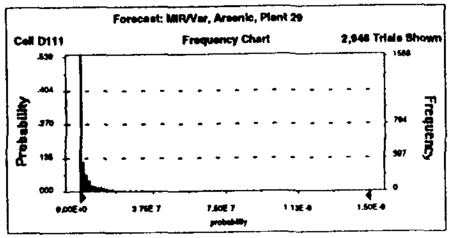


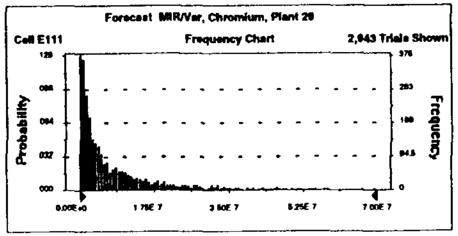


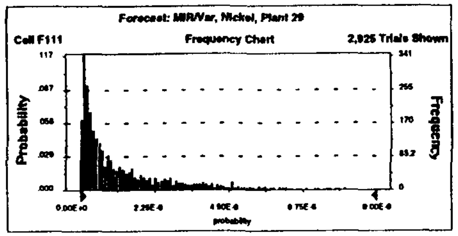


MIR/Variability, Plant 29

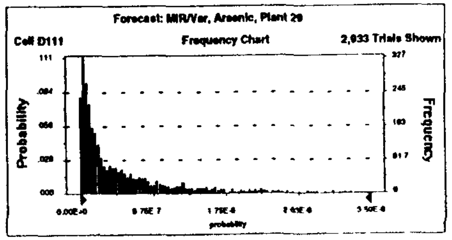
FCEM Concentration Data

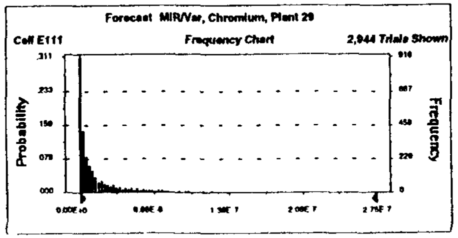






SGS Concentration Data





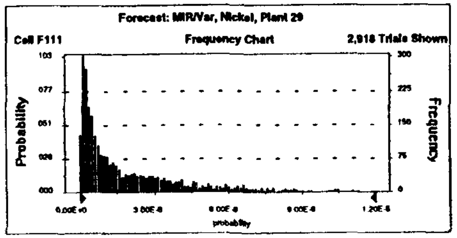
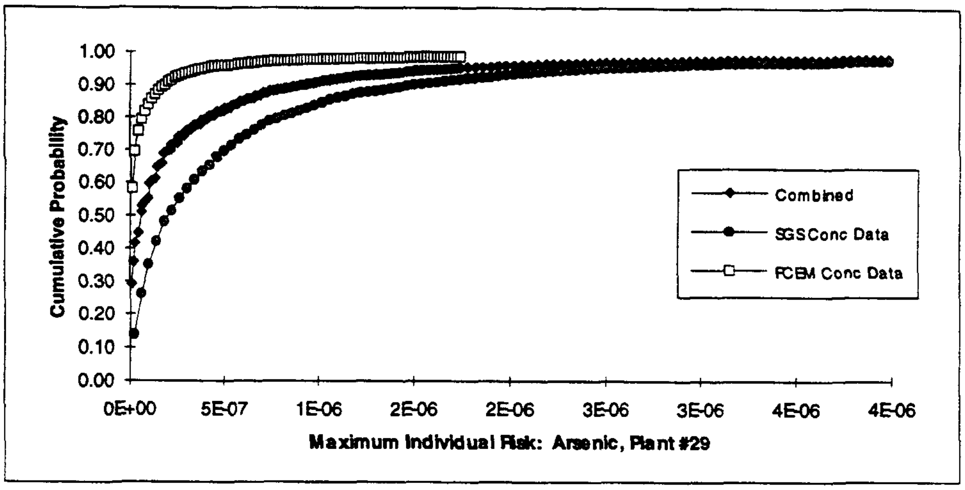
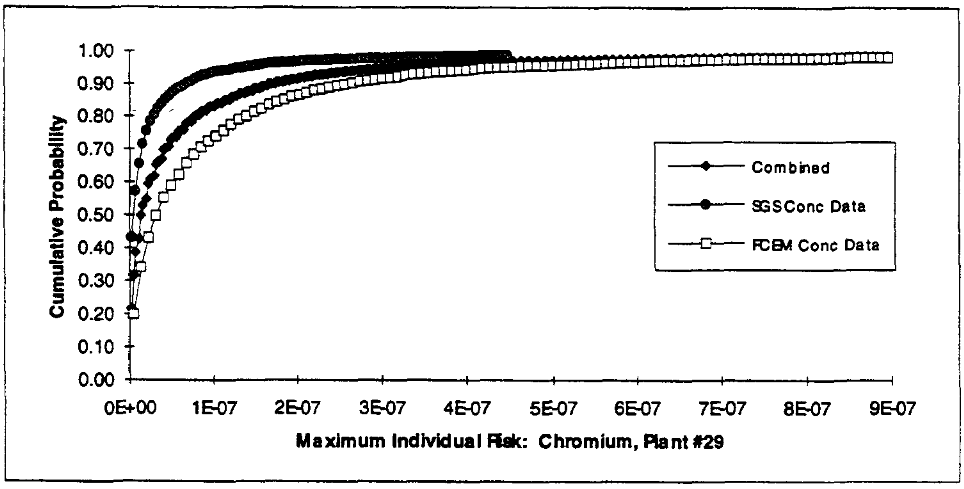
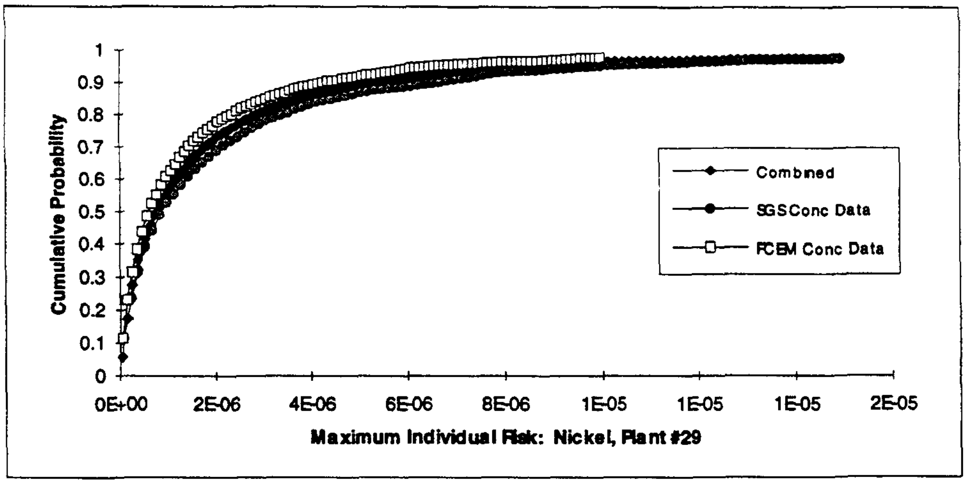


Figure G-21. Cumulative Probability Distributions of Uncertainty in MIR: Plant #29







For comparison, a summary of the uncertainty in the estimates due to exposure to arsenic, chromium, and nickel emissions from Plants #343, 133, and 240 are statistically summarized in Table G-20. In general, the results support the conclusions that the baseline assessment resulted in reasonably conservative estimates of risk while still being within the range of plausible values. For the coal-fired plants (343 and 240) the point estimates of MIR generated during the baseline assessment were associated with the 91st to 94th percentile on the overall distribution of possible MIR risk estimates. The 95th percentile (a typical high-end risk estimate) of the overall distribution is roughly twice the original risk estimate, four times the mean and within an order of magnitude to the median MIR risk estimate. For oil-fired Plant 133 the results were similar, though the uncertainty was a little more broad, most likely attributable to which of the oil concentration data sets are used. The original estimates of MIR from the baseline assessment were between the 86th and 99th percentile on the estimated overall distribution of possible MIR values.

G.5.2.2 Uncertainty in Population Risk Estimate. The process used to derive overall distributions on the risk estimates for MIR was repeated for annual cancer incidence. The distribution of possible estimates of annual cancer incidence due to emissions from Plant #29 is statistically summarized in Table G-21 and graphically depicted in Figure G-22. As with the estimates of MIR, the point estimates for annual cancer incidence associated with Plant #29 emissions generated in the risk assessment are at the upper end of the distribution (near the 95th percentile). Furthermore, the 95th percentile was about 10 times the median, and about 5 times the mean.

For comparison, the distribution of possible estimates of annual cancer incidence due to emissions from coal-fired Plant #343 is statistically summarized in Table G-22. As with the estimates of MIR, the point estimates for annual cancer incidence associated with Plant #343 are at the upper end of the distribution (near the 97th percentile). Furthermore, the 95th percentile was about 8 to 30 times the median, and about 4 times the mean.

G.5.2.3 Uncertainty in Total Annual Cancer Incidence Across Plants. The total annual cancer incidence associated with all utilities is the sum of incidence associated with individual plants. The above analysis focuses on the uncertainty in cancer risks associated with individual plants. An additional simulation was conducted to evaluate the uncertainty associated with the estimate of total annual cancer incidence across all

Table G-20. Summary of Results of Monte Carlo Simulation of Annual Cancer Incidence and MIR for Coal-Fired Plant #343

	Plant 343						Plant 240					
	Maximum Individual Risk (MIR)						Maximum Individual Risk (MIR)					
	Uncertainty			Variability			Uncertainty			Variability		
	arsenic	chromium	nickel	arsenic	chromium	nickel	arsenic	chromium	nickel	arsenic	chromium	nickel
Mean	3.1E-7	2.4E-7	6.2E-9	2.9E-7	2.3E-7	6.5E-9	1.2E-6	3.7E-7	1.1E-8	1.1E-6	3.4E-7	1.0E-8
Original Point Estimate	7.9E-7	6.6E-7	1.1E-8	7.9E-7	6.6E-7	1.1E-8	3.9E-6	1.0E-6	3.3E-8	3.9E-6	1.0E-6	3.3E-8
Percentile	92	92	93	92	92	93	94	91	92	93	92	92
Standard Deviation	1.2E-6	6.1E-7	1.6E-8	9.1E-7	3.8E-7	1.3E-8	6.4E-6	8.8E-7	2.7E-8	4.2E-6	5.8E-7	2.0E-8
Mean Std. Error	2.2E-8	1.1E-8	2.8E-10	1.7E-8	7.0E-9	2.4E-10	1.2E-7	1.6E-8	4.9E-10	7.7E-8	1.1E-8	3.6E-10
Percentile												
0.0%	8.9E-20	1.6E-10	5.6E-13	1.3E-19	5.3E-10	9.9E-13	1.1E-17	4.0E-10	1.2E-12	4.5E-17	9.2E-10	5.0E-13
2.5%	3.6E-9	4.0E-9	8.0E-11	2.4E-9	4.5E-9	7.8E-11	2.5E-8	4.5E-9	8.7E-11	1.2E-8	4.5E-9	9.2E-11
5.0%	7.2E-9	7.8E-9	1.6E-10	4.7E-9	8.4E-9	1.5E-10	5.0E-8	8.6E-9	1.7E-10	2.4E-8	8.2E-9	1.8E-10
10%	1.4E-8	1.5E-8	3.2E-10	9.5E-9	1.6E-8	3.1E-10	9.9E-8	1.7E-8	3.5E-10	4.7E-8	1.5E-8	3.7E-10
25%	3.6E-8	3.8E-8	8.0E-10	2.4E-8	3.9E-8	7.7E-10	2.5E-7	4.1E-8	8.6E-10	1.2E-7	4.0E-8	9.2E-10
50%	7.2E-8	8.6E-8	1.9E-9	4.7E-8	1.0E-7	2.4E-9	5.0E-7	1.1E-7	2.5E-9	2.4E-7	1.2E-7	3.0E-9
75%	1.9E-7	2.4E-7	5.7E-9	2.0E-7	2.5E-7	7.0E-9	7.5E-7	3.5E-7	9.6E-9	5.7E-7	3.9E-7	1.1E-8
90%	6.4E-7	5.7E-7	1.5E-8	6.9E-7	5.6E-7	1.6E-8	2.2E-6	9.4E-7	2.7E-8	2.5E-6	8.9E-7	2.8E-8
95.0%	1.3E-6	8.9E-7	2.6E-8	1.2E-6	8.4E-7	2.4E-8	4.9E-6	1.5E-6	4.4E-8	5.3E-6	1.4E-6	4.7E-8
97.5%	2.4E-6	1.3E-6	4.1E-8	2.1E-6	1.1E-6	3.7E-8	9.2E-6	2.3E-6	7.3E-8	9.2E-6	1.9E-6	6.6E-8
100.0%	2.8E-5	2.1E-5	4.4E-7	1.5E-5	6.6E-6	2.4E-7	2.4E-4	1.8E-5	3.9E-7	9.8E-5	9.1E-6	3.0E-7

Original Point Estimate = Estimate derived in the baseline assessment using EPA default values
Percentile = the percentile of the original point estimate within the Monte Carlo derived distribution

Forecast: MIR, Plant 133

	FCEM, Uncertainty			SGS Uncertainty			FCEM, variablity			SGS, Variability		
	Arsenic	chromium	nickel	arsenic	chromium	nickel	arsenic	chromium	nickel	arsenic	chromium	nickel
Mean	2.7E-6	2.5E-6	4.9E-5	1.4E-5	6.5E-7	6.8E-5	2.6E-6	2.4E-6	4.9E-5	1.4E-5	7.3E-7	6.8E-5
Original Point Estimate	2.9E-5	5.6E-6	9.5E-5	2.9E-5	5.6E-6	9.5E-5	2.9E-5	5.6E-6	9.5E-5	2.9E-5	5.6E-6	9.5E-5
(percentile)	99	89	87	88	98	82	99	89	86	87	98	80
Standard Deviation	1.9E-5	8.9E-6	1.2E-4	3.2E-5	2.6E-6	1.6E-4	1.4E-5	4.8E-6	1.0E-4	2.6E-5	2.7E-6	1.4E-4
Mean Std. Error	3.4E-7	1.6E-7	2.2E-6	5.9E-7	4.7E-8	2.9E-6	2.6E-7	8.8E-8	1.8E-6	4.7E-7	4.9E-8	2.5E-6
Percentile												
0.0%	3.9E-11	3.5E-9	4.3E-8	1.7E-8	9.1E-11	7.5E-8	3.6E-11	6.7E-9	4.1E-8	3.5E-8	3.3E-10	8.0E-8
2.5%	7.5E-8	4.9E-8	5.0E-7	1.5E-7	1.0E-8	6.6E-7	4.3E-8	2.8E-8	5.0E-7	1.6E-7	7.9E-9	8.1E-7
5.0%	1.5E-7	9.5E-8	9.6E-7	2.8E-7	2.0E-8	1.2E-6	8.7E-8	4.9E-8	9.7E-7	2.8E-7	1.5E-8	1.5E-6
10%	3.0E-7	1.9E-7	1.9E-6	5.4E-7	4.0E-8	2.4E-6	1.7E-7	9.1E-8	1.9E-6	5.3E-7	3.1E-8	3.0E-6
25%	7.5E-7	4.6E-7	4.6E-6	1.3E-6	9.9E-8	5.9E-6	4.3E-7	2.2E-7	4.7E-6	1.3E-6	7.6E-8	7.4E-6
50%	1.5E-6	9.2E-7	1.3E-5	3.9E-6	2.0E-7	1.8E-5	8.7E-7	6.9E-7	1.5E-5	4.4E-6	1.5E-7	2.3E-5
75%	2.3E-6	2.2E-6	4.3E-5	1.3E-5	4.4E-7	6.1E-5	1.3E-6	2.4E-6	5.2E-5	1.4E-5	4.6E-7	7.2E-5
90%	4.3E-6	5.9E-6	1.2E-4	3.4E-5	1.4E-6	1.7E-4	4.1E-6	6.1E-6	1.3E-4	3.6E-5	1.4E-6	1.7E-4
95.0%	9.2E-6	1.0E-5	2.1E-4	5.8E-5	2.7E-6	2.8E-4	9.4E-6	1.0E-5	2.1E-4	5.4E-5	2.7E-6	2.9E-4
97.5%	1.7E-5	1.6E-5	3.4E-4	8.5E-5	4.7E-6	4.4E-4	1.6E-5	1.5E-5	3.1E-4	8.3E-5	5.5E-6	4.3E-4
100.0%	7.8E-4	3.5E-4	2.2E-3	5.2E-4	8.1E-5	2.2E-3	4.1E-4	6.8E-5	1.6E-3	4.0E-4	5.1E-5	2.8E-3

Table G-21. Distributions of Annual Cancer Incidence Plant 29
Comparison of FCEM and SGS Data

Annual Incidence Plant 29

Uncertainty

	arsenic		chromium		nickel	
	FCEM	SGS	FCEM	SGS	FCEM	SGS
Mean	1E-03	7E-03	1E-03	4E-04	2E-02	3E-02
Initial Point Estimate	8E-03		3E-03		6E-02	
(percentile)	(96)	(75)	(92)	(98)	(95)	(89)
Percentiles:						
0.0%	1E-07	5E-05	4E-06	3E-07	1E-04	1E-04
2.5%	2E-05	3E-04	4E-05	5E-06	1E-03	1E-03
5.0%	4E-05	6E-04	7E-05	1E-05	2E-03	2E-03
10%	8E-05	9E-04	1E-04	2E-05	2E-03	4E-03
25%	2E-04	2E-03	3E-04	5E-05	5E-03	8E-03
50%	4E-04	4E-03	6E-04	1E-04	1E-02	2E-02
75%	8E-04	8E-03	1E-03	3E-04	2E-02	4E-02
90%	2E-03	2E-02	3E-03	9E-04	4E-02	7E-02
95.0%	6E-03	2E-02	4E-03	1E-03	6E-02	1E-01
97.5%	1E-02	3E-02	6E-03	3E-03	9E-02	1E-01
100.0%	2E-01	2E-01	3E-02	3E-02	4E-01	9E-01

Variability

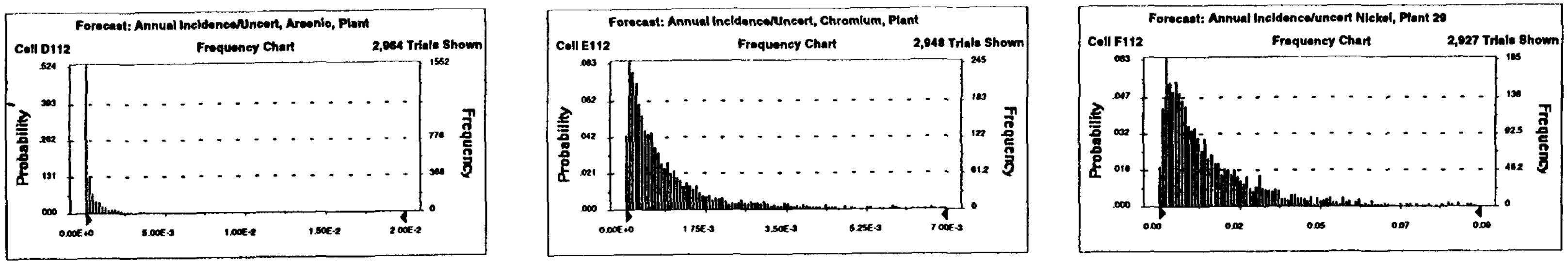
	arsenic		chromium		nickel	
	FCEM	SGS	FCEM	SGS	FCEM	SGS
Mean	1E-03	8E-03	2E-03	7E-04	2E-01	3E-01
Initial Point Estimate	8E-03		3E-03		6E-02	
(percentile)	(96)	(72)	(80)	(96)	(15)	(26)
Percentiles:						
0.0%	2E-07	8E-05	4E-06	5E-07	4E-03	5E-03
2.5%	2E-05	6E-04	1E-04	9E-06	2E-02	2E-02
5.0%	3E-05	8E-04	2E-04	2E-05	2E-02	3E-02
10%	6E-05	1E-03	3E-04	3E-05	3E-02	5E-02
25%	2E-04	2E-03	6E-04	9E-05	6E-02	9E-02
50%	3E-04	5E-03	1E-03	2E-04	1E-01	2E-01
75%	9E-04	9E-03	3E-03	6E-04	2E-01	3E-01
90%	3E-03	2E-02	5E-03	2E-03	4E-01	6E-01
95.0%	6E-03	2E-02	8E-03	3E-03	6E-01	8E-01
97.5%	1E-02	3E-02	1E-02	4E-03	8E-01	1E+00
100.0%	1E-01	2E-01	5E-02	3E-02	3E+00	8E+00

Initial Point Estimate = The estimate of emissions used in the baseline exposure assessment.
(Percentile) = The percentile of the predicted distribution corresponding to the initial point estimate.

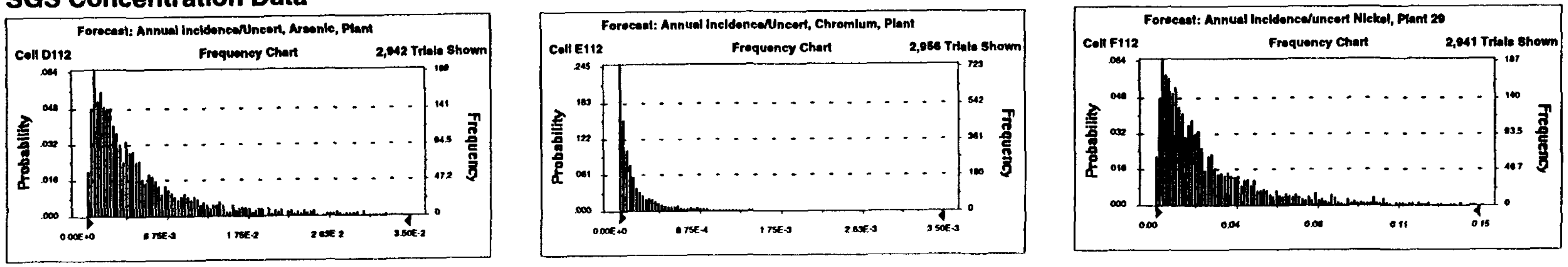
Figure G-22. Distributions of Annual Cancer Incidence: Plant 29: Comparison of SGS and FCEM Data

Forecast: Annual Incidence/Uncert, Plant 29

FCEM Concentration Data

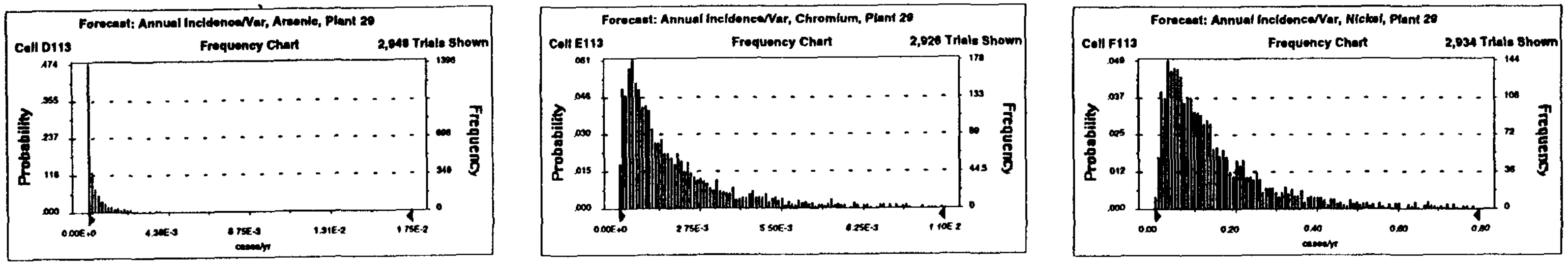


SGS Concentration Data



Forecast: Annual Incidence/Var, Plant 29

FCEM Concentration Data



SGS Concentration Data

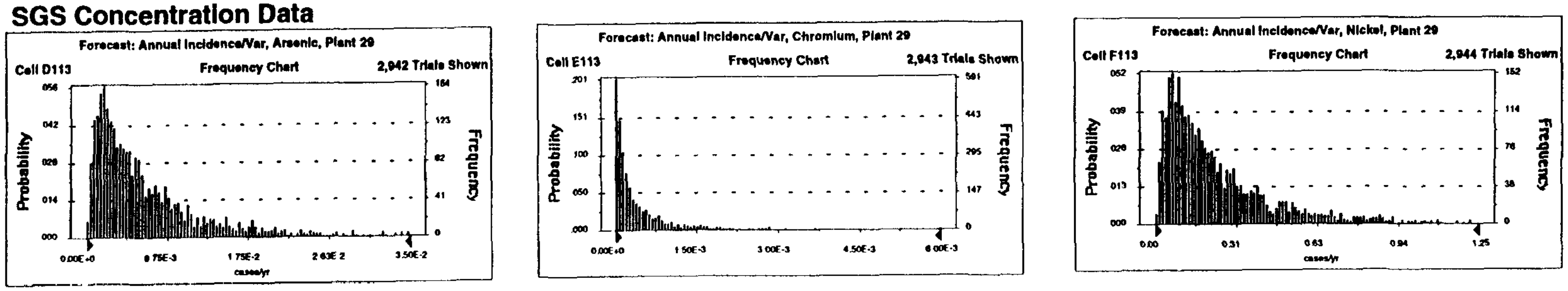


Table G-22. Summary of Results of Monte Carlo Simulation of
Annual Cancer Incidence (and MIR) for Coal-Fired Plant #343

Forecast: Annual Incidence, Plant 343 (cases/year)

	Incidence, Uncert.			**Incidence, Var**		
	arsenic	chromium	nickel	arsenic	chromium	nickel
Mean	4.5E-4	3.3E-4	9.7E-6	4.8E-4	6.3E-4	1.0E-5
Initial Point Estimate	3.1E-3	2.6E-3	1.0E-4	3.1E-3	2.6E-3	1.0E-4
(Percentile)	97	99	98	97	96	97
Standard Deviation	1.2E-3	5.6E-4	2.1E-5	1.4E-3	1.0E-3	2.0E-5
Mean Std. Error	2.2E-5	1.0E-5	3.9E-7	2.5E-5	1.9E-5	3.6E-7
Percentile						
0.0%	*6.8E-17*	*2.0E-7*	*1.2E-9*	*1.4E-16*	*1.0E-6*	*2.3E-9*
2.5%	2.9E-6	7.0E-6	1.3E-7	3.5E-6	1.5E-5	1.3E-7
5.0%	5.9E-6	1.4E-5	2.6E-7	7.0E-6	3.0E-5	2.6E-7
10%	1.2E-5	2.7E-5	5.2E-7	1.4E-5	5.7E-5	5.2E-7
25%	2.9E-5	6.4E-5	1.3E-6	3.5E-5	1.3E-4	1.4E-6
50%	6.9E-5	1.5E-4	3.6E-6	7.7E-5	3.0E-4	4.0E-6
75%	3.3E-4	3.6E-4	9.7E-6	3.7E-4	7.1E-4	1.0E-5
90%	1.1E-3	8.0E-4	2.3E-5	1.1E-3	1.5E-3	2.3E-5
95.0%	2.2E-3	1.2E-3	4.0E-5	2.2E-3	2.2E-3	3.8E-5
97.5%	3.6E-3	1.8E-3	6.0E-5	3.6E-3	3.2E-3	5.8E-5
100.0%	1.7E-2	1.0E-2	4.6E-4	2.0E-2	1.5E-2	3.2E-4

Original Point Estimate = Estimate derived in the baseline assessment using EPA default values
Percentile = the percentile of the original point estimate within the Monte Carlo derived distribution

utilities. For this analysis, the distribution of annual cancer incidence for each individual plant was sampled and summed to estimate the total cancer incidence for those four plants. Figure G-23 and Table G-23 presents these results. Again, as indicated in Table G-23, the results indicate that the original estimate from the baseline risk assessment is reasonably conservative, ranging from the 92nd to 97th percentile of the overall distribution on total cancer incidence for those four plants. Furthermore, the uncertainty in the total annual incidence is somewhat less than the uncertainty associated with individual plants. This is due to the fact that for any one plant which experiences a high distribution value it is likely that another plant would have a lower distribution value thereby more approach the central tendency. The 95th percentile of the distribution for total cancer incidence was typically within about a factor of 3 to the mean and three to six to the median, both less than would be expected for individual plants.

G.5.2.4 Sensitivity Analysis. Crystal Ball has the internal capability to conduct a sensitivity analysis to determine which parameter most influences the resulting distribution. Crystal Ball calculates sensitivity by computing contribution to variance between every assumption (parameter) and forecast (output) while the simulation is running. A sensitivity analysis was conducted for the estimation of uncertainty in MIR and incidence. For all three HAPs, the dispersion coefficient for maximum concentration contributed most to overall uncertainty followed by the CSF correction factor (SF), the EMF, and speciation correction factor (S). The trace element concentration in oil, breathing rate adjustment (BR), and oil consumption contributed least to uncertainty. Exposure frequency adjustment (EF) is the additional factor used in estimating incidence and only moderately contributed to overall uncertainty compared to the other parameters identified above.

G.5.3 Summary

The baseline risk assessment conducted by EPA on HAP emissions from utilities generated point estimates of risk. The procedure used in that study was intended to be generally conservative, more likely to overestimate than underestimate actual risk. The methods used in that analysis relied on a combination of point values, some conservative and some not conservative, yielding a point estimate of exposure and risk that falls at an unknown percentile of the full distributions of exposure and risk.

A quantitative uncertainty analysis was conducted and is presented in this Appendix. The large number of HAPs and utility

Figure G-23. Results of Monte Carlo Simulation of Estimating Annual Cancer Incidence Across Four Plants (Plants 29, 343, 240, 133)

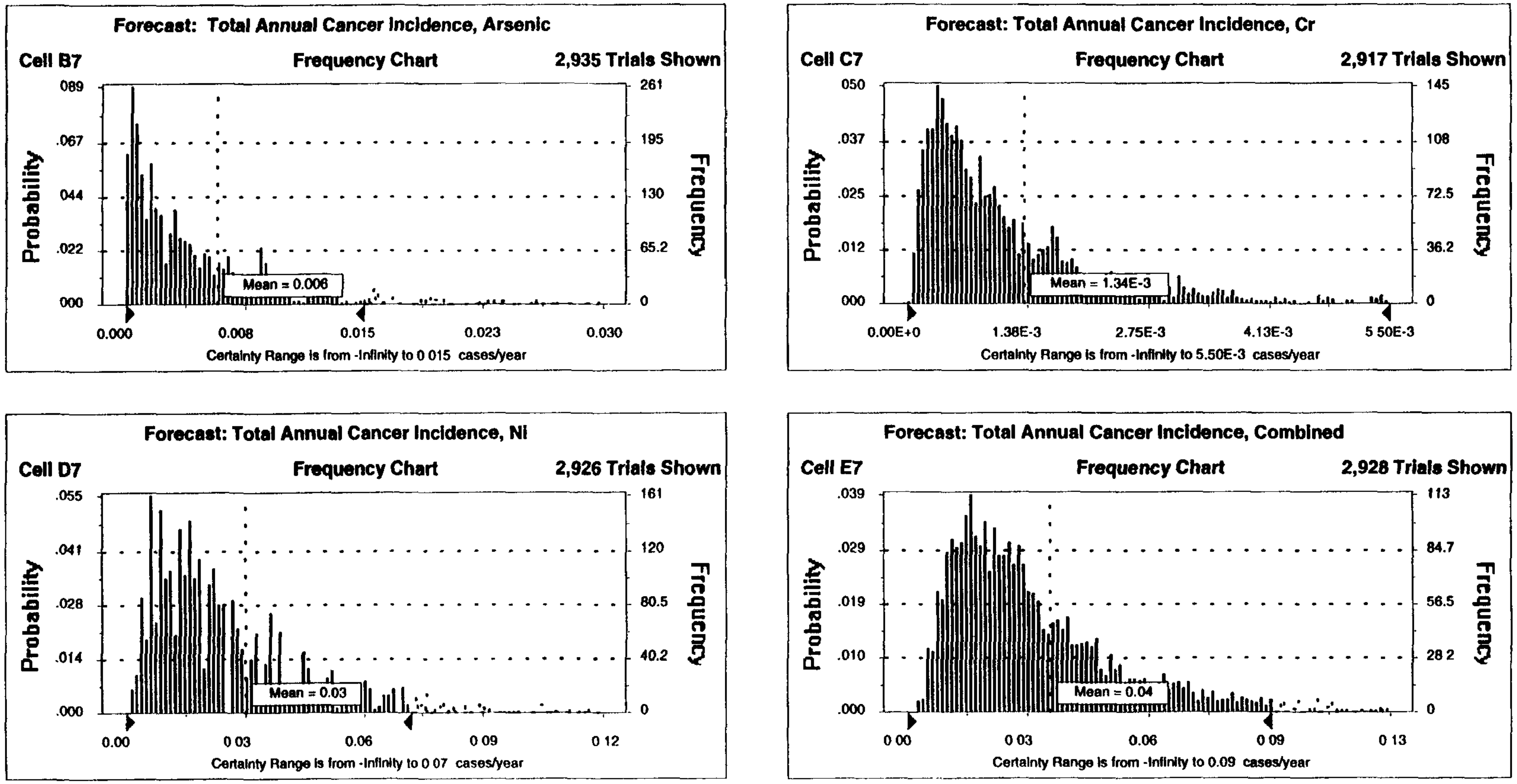


Table G-23. Summary of Monte Carlo Simulation of Estimating Total Annual Cancer Incidence Across Four Plants (Plants 29, 343, 240, 133)

Forecast: Total Annual Cancer Incidence (cases/year)
(Combined estimates from Plants, 29, 343, 240, 133)

Statistics:	Arsenic	Chromium	Nickel	Total
Mean	0.0057	0.0013	0.0302	0.0373
Standard Deviation	0.0085	0.0016	0.0315	0.0328
Mean Std. Error	0.0002	0.0000	0.0006	0.0006
Percentile				
0.0%	0.0001	0.0000	0.0018	0.0026
2.5%	0.0001	0.0001	0.0042	0.0073
5.0%	0.0002	0.0002	0.0051	0.0090
10%	0.0004	0.0003	0.0071	0.0114
25%	0.0011	0.0004	0.0118	0.0174
50%	0.0031	0.0009	0.0205	0.0281
75%	0.0069	0.0016	0.0369	0.0457
90%	0.0128	0.0030	0.0617	0.0719
95.0%	0.0196	0.0042	0.0848	0.0965
97.5%	0.0277	0.0056	0.1183	0.1241
100.0%	0.1011	0.0205	0.3426	0.3537
Baseline Estimate	0.015	0.0067	0.072	0.094
Percentile	92	97	93	95
Ratio				
95th percentile: mean	3.4	3.1	2.8	2.6
95th percentile: median	6.3	4.8	4.1	3.4
95th percentile : baseline	1.3	0.6	1.2	1.0

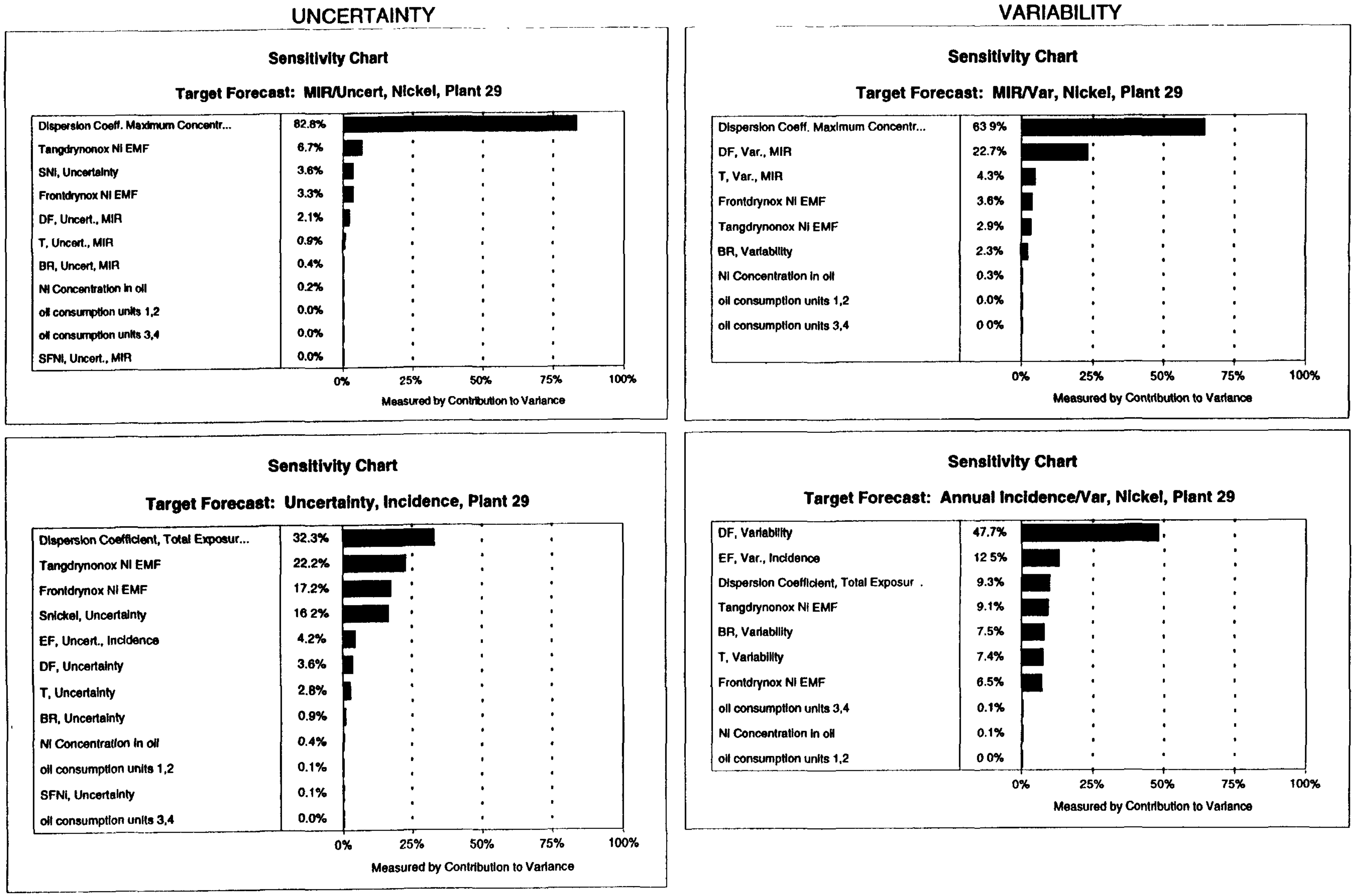
plants did not allow for an analysis of all HAPs or plants. Therefore, the analysis was limited to three HAPs (arsenic, chromium, and nickel), which accounted for over 95 percent of the total cancer incidence, and a subset of plants selected as the major contributors to incidence and which were responsible for the highest MIR. The risk assessment process was divided into three distinct components (emissions estimation, dispersion and exposure modeling, and exposure-response) and the uncertainty was analyzed independently in each. Overall distributions on the uncertainty in the risk estimates were generated by combining the uncertainties of the three components. Figure G-24 identifies the major parameters and identifies the original assumptions used in the baseline case and summarizes information on the parameter values and associated uncertainty.

The results indicate that the point estimates of risks generated in the original risk assessment were at the upper end of overall distribution, usually around the 95th percentile (and ranged from the 71st to 99th percentile). In evaluating risks, the 95th percentile is typically used as a high-end risk descriptor and the median or mean is used to describe central tendency measures of risk. The 95th percentile was about 10 times the median value on the distribution, and roughly 5 times the mean estimate of risk. A sensitivity analysis indicated that the parameters that contribute most to the overall uncertainty were the dispersion coefficient for maximum concentration followed by the CSF correction factor (SF), the EMF, and speciation correction factor (S). The trace element concentration in oil, breathing rate adjustment (BR), and oil consumption contributed least to uncertainty. Exposure frequency adjustment (EF) -- the additional factor used in estimating incidence -- only moderately contributed to overall uncertainty compared to the other parameters.

It should be noted that the uncertainty estimates presented here in this report are likely to underestimate the true overall uncertainty for several reasons. The major reasons for underestimation of uncertainty include the approach used in the analysis (focusing on parameter uncertainty), limited data, and the potential for unsuspected errors. Each is described below.

This uncertainty analysis focused on parameter uncertainty alone and did not quantify uncertainty related to model choice which in many cases (e.g., dose-response modeling) can lead to dramatic and significant differences in risk estimates. Experience indicates that the HEM and other models used by EPA to estimate dispersion and exposure are limited in their ability to estimate short-term concentrations and locations of maximum

Figure G-24. Sensitivity Analysis of Uncertainty and Variability in Estimates of MIR and Incidence: Plant 29



concentrations. However, these models are more accurate when predicting long-term average concentrations and exposure, typically with a coefficient of variation of about 10 to 40 percent. The existing range of dose-response models can result in estimates of risk which may differ by several orders of magnitude.

The uncertainty analysis did not address all parameters related to plant emissions due to constraints in available data. Existing data were reported to EPA in various forms, ranging from summary reports to raw data. Therefore, the available data did not allow for the separate evaluation of uncertainty and variability, given that sampling and analysis error could not be quantified. Furthermore, data did not support the simulation of many factors related to plant emissions, particularly in fuel consumption. The rate of fuel consumption, and the fuel mix consumed is expected to vary dramatically over time related to plant operations and electricity demand. This will have a significant impact on the pattern of emissions and flue gas parameters (i.e., exit velocity and flue gas temperature). The type of fuel consumed would be expected to be influenced by sulfur content, seasonal demand, cost, and availability. These factors may result in a higher degree of variation in emissions and subsequent exposures than predicted. However, while these factors would result in high degrees of short-term variation it is unknown how this may impact the long-term average emissions.

The risk assessment and uncertainty analysis focused on the cancer risks associated with direct inhalation of HAP emissions within 50 km of plants. Additional risks and concomitant uncertainties also exist for indirect exposures and long-range transport, and re-entrainment of and subsequent exposure to deposited particulates. This analysis was also limited to long-term average exposures and did not account for the expected short-term variations in exposure which may influence noncancer risks, and total dose to carcinogens which are dose rate dependent.

Evidence also indicates that commonly used 95 percent bounds for normal and lognormal distributions of variables are very sensitive to the underestimation of the true uncertainty and typically do not address unsuspected errors. Empirical probability distributions of the normalized deviations of the measured quantities from the true values (in several data sets from fields of nuclear physics, energy, and population projections) do not follow the usually implied normal distribution. These distributions are better described by an exponential distribution.[37] If much of the uncertainty comes from systematic errors, the usual justification for the normal distribution between observed deviations of measured quantities from the true values does not apply.

Unsuspected errors are quite common, and new measurements are often far from the previous values. Environmental measurements are rarely repeated with the same samples, and it is hard to estimate how widespread the unaccounted errors are in routinely collected data. One can hedge against unsuspected uncertainties by multiplying the reported uncertainty range by a default safety factor or to solve for the exponential factor to describe the observed deviations. This safety factor or exponential factor can be used to expand the confidence intervals. This approach has been used in evaluating large number of data sets related to nuclear physics, and energy projections[37] and suggested for use in the environmental field.[27] One reviewer suggested using this approach to expand the estimates of uncertainty for this study.[7] However, the exponential approach has not yet been documented to apply to environmental measurements or models, and to test the applicability would require greater amounts of data than are available for this project. Therefore, EPA recognizes the potential for unsuspected errors contributing to the overall uncertainty, but feels as though at present there is insufficient information to support the estimation of the proper exponential function to be used to expand our confidence intervals.

EPA recognizes the potential for unsuspected errors in the data and its associated representation of uncertainty. The manner in which the data were collected and reported can contribute to these errors. The emissions test data were developed by EPRI in cooperation with the EPA and the Department of Energy. Often these test data were limited in number and did not encompass the entire universe of facilities with regards to plant configurations, age, and operation and maintenance. Furthermore, with a single short-term test it is not possible to use conventional methods to estimate or subtract out short-term variability and measurement errors from the long-term variability of a parameter (e.g., emissions) Also it is not possible to estimate the possible effects of realistic variations in fuels and plant operating characteristics that could affect long-term emissions. Data are often collected under relatively favorable conditions in these respects and, therefore, may not entirely reflect the full variability of important operating parameters that would be seen over the lifetime of these facilities.[7]

In conclusion, the uncertainty analysis presented here supports the original risk estimates. Given, EPA's mandate of protecting public health, EPA risk assessments are intended to yield conservative estimates of risk (more likely to overestimate than underestimate). This is typically accomplished with the use of conservative assumptions. The original estimates were shown to be realistically conservative ranging from the 71st to 98th percentile of the overall distributions of risk predicted from the uncertainty analysis. Past criticisms state that the use of compounded conservative assumptions can lead to risk estimates beyond the range of expected values. However, the uncertainty analysis indicates that the original estimates for this study are well within the range of possible values.

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TECHNICAL REPORT DATA

(Please read Instructions on reverse before completing)

1 REPORT NO EPA-453/R-96-013b	2.	3. RECIPIENT'S ACCESSION NO
4 TITLE AND SUBTITLE Study of Hazardous Air Pollutant Emissions from Electric Utility Steam Generating Units -- Interim Final Report Volume 2. Appendices A - G		5. REPORT DATE October 1996
		6. PERFORMING ORGANIZATION CODE
7 AUTHOR(S)		8 PERFORMING ORGANIZATION REPORT NO.
9 PERFORMING ORGANIZATION NAME AND ADDRESS U.S. Environmental Protection Agency Emission Standards Division/Air Quality Strategies and Standards Division Office of Air Quality Planning and Standards Research Triangle Park, NC 27711		10. PROGRAM ELEMENT NO.
		11. CONTRACT/GRANT NO.
12 SPONSORING AGENCY NAME AND ADDRESS		13 TYPE OF REPORT AND PERIOD COVERED
		14. SPONSORING AGENCY CODE

15 SUPPLEMENTARY NOTES

16 ABSTRACT

This report has been prepared pursuant to section 112(n)(1)(A) of the Clean Air Act, and provides the Congress and the public with information regarding the emissions, fate, and transport of utility HAPs. The primary components of this report are: (1) a description of the industry; (2) an analysis of emissions data; (3) an assessment of hazards and risks due to inhalation exposures to numerous HAPs (excluding mercury); (4) an assessment of risks due to multipathway (inhalation plus non-inhalation) exposure to radionuclides; and (5) a discussion of alternative control strategies. The assessment for mercury includes a description of emissions, deposition estimates, control technologies, and a dispersion and fate modeling assessment which includes predicted levels in various media based on modeling from four representative utility plants using hypothetical scenarios. The EPA plans to publish a final report at a later date which will include (1) a more complete assessment of the exposures, hazards, and risks; (2) conclusions, as appropriate and feasible, regarding the significance of the risks and impacts to public health; and (3) a determination as to whether regulation of utility HAPs is appropriate and necessary.

17 KEY WORDS AND DOCUMENT ANALYSIS

a DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Air Pollution Atmospheric Dispersion Modeling Electric Utility Steam Generating Units Hazardous Air Pollutants/Air Toxics	Air Pollution Control	

18 DISTRIBUTION STATEMENT Release Unlimited	19. SECURITY CLASS *(Report)* Unclassified	21 NO. OF PAGES 258
	20. SECURITY CLASS *(Page)* Unclassified	22 PRICE

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