United States
Environmental Protection
Agency

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

EPA-453/R-98-004b
February 1998

Air

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

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Appendix A - Median Emission Factors, Determined from Test Report Data, and Total 1990, 1994, and 2010 Emissions, Projected with the Emission Factor Program

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

Coal-fired units: inorganic HAPs	Estimated total 1990 emissions (tons)	Estimated total 1994 emissions (tons)	Estimated total 2010 emissions (tons)
Antimony	7.95	7.98	9.93
Arsenic	60.93	55.81	70.61
Beryllium	7.13	7.93	8.20
Hydrogen chloride	143,000	134,000	155,000
Hydrogen cyanide (HCN) [a]	240.66	250.8	318.31
Hydrogen fluoride	19,500	23,100	25,700
Cadmium	3.33	3.15	3.82
Chromium	73.27	61.60	87.43
Cobalt	21.21	22.67	27.08
Lead	75.47	61.77	86.89
Manganese	163.97	167.72	219.02
Mercury	45.80	51.34	59.74
Nickel	58.05	52.04	68.65
Phosphorus (P) [b]	270.74	331.41	358.09
Selenium	153.83	183.68	213.21

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

[b] 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

Oil-fired units: inorganic HAPs	Estimated total 1990 emissions (tons)	Estimated total 1994 emissions (tons)	Estimated total 2010 emissions (tons)
Arsenic	**5.02**	**3.51**	**2.54**
Beryllium	**0.46**	**0.40**	**0.23**
Cadmium	**1.71**	**1.09**	**0.86**
Chromium	**4.74**	**3.91**	**2.40**
Cobalt	**20.41**	**15.48**	**10.31**
Hydrogen chloride	**2860**	**2100**	**1450**
Hydrogen fluoride	**143**	**284**	**73**
Lead	**10.58**	**8.92**	**5.35**
Manganese	**9.28**	**7.30**	**4.70**
Mercury	**0.25**	**0.19**	**0.13**
Nickel	**392.83**	**322.37**	**198.17**
Phosphorus (P) [a]	**67.25**	**50.89**	**34.10**
Selenium	**1.65**	**1.42**	**0.84**

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[a] 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

Gas-fired units: inorganic HAPs	Estimated total 1990 emissions (tons)	Estimated total 1994 emissions (tons)	Estimated total 2010 emissions (tons)
Arsenic	**0.15**	**0.18**	**0.25**
Cadmium	[a]	[a]	[a]
Chromium	[a]	[a]	[a]
Cobalt	**0.14**	**0.15**	**0.23**
Lead	**0.43**	**0.47**	**0.68**
Manganese	[a]	[a]	[a]
Mercury	**0.0015**	**0.0017**	**0.0243**
Nickel	**2.19**	**2.42**	**3.49**
Phosphorus	**5.65**	**6.23**	**8.98**

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[a] The emission factors are not available for this compound, but the compound was detected in one or more tests. Some of these compounds encompass a group of compounds, and although the total is not available, some members of the group may be presented elsewhere.

Table A-4. Median Emission Factors, Determined from Test Report Data, and Total 1990, Total 1994 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 (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
1,1,2-trichloroethane	1	4.70	40.39	1	4.70	42.10	1	4.70	53.43
2-chloroacetophenone	3	0.29	2.50	3	0.29	2.60	3	0.29	3.30
2,4-dinitrotoulene	3	0.015	0.130	3	0.015	0.130	3	0.015	0.180
Acetaldehyde	12	6.75	58.01	12	4.85	43.44	12	6.75	76.74
Acetophenone	7	0.68	5.84	8	0.91	8.15	7	0.68	7.73
Acrolein	6	3.25	27.93	7	3.3	29.56	6	3.25	36.95
Benzene	20	2.50	21.48	22	2.50	22.40	20	2.50	28.42
Benzyl chloride	1	0.006	0.050	1	0.006	0.050	1	0.006	0.060
Bis(2-ethylhexyl) phthalate	9	4.10	35.24	9	4.10	36.73	9	4.10	46.61
Bromoform	1	6.60	56.73	1	6.60	59.11	1	6.60	75.03
Carbon disulfide	8	4.30	36.96	9	2.90	25.97	8	4.30	48.88
Carbon tetrachloride	2	3.25	27.93	2	3.25	29.11	2	3.25	36.95
Chlorobenzene	2	3.18	27.34	2	3.18	28.48	2	3.18	36.15
Chloroform	2	3.20	27.50	3	3.00	26.87	2	3.20	36.38
Cumene	1	0.29	2.50	1	0.29	2.60	1	0.29	3.30
Dibutyl phthalate	5	2.8	24.07	4	2.55	22.84	5	2.8	31.83
Ethyl benzene	5	0.41	3.52	6	0.41	3.63	5	0.41	4.66
Ethyl chloride	1	2.40	20.63	1	2.40	21.49	1	2.40	27.28
Methyl chloroform	4	3.42	29.35	4	2.10	18.81	4	3.42	38.82

Table A-4. (Continued)

Coal-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
Ethylene dichloride	3	3.10	26.64	3	3.10	27.76	3	3.10	35.24
Formaldehyde	15	4.00	34.38	14	3.25	29.11	15	4.00	45.47
Hexane	2	0.83	7.10	3	1.50	13.43	2	0.83	9.38
Hexachlorobenzene	1	0.08	0.68	1	0.08	0.70	1	0.08	0.90
Isophorone	2	24.0	206.28	2	24.0	214.97	2	24.0	272.83
Methyl bromide	6	0.89	7.65	6	2.245	20.11	6	0.89	10.12
Methyl chloride	3	5.90	50.71	4	5.5	49.27	3	5.90	67.07
Methyl ethyl ketone	6	8.00	68.76	7	5.00	44.78	6	8.00	90.95
Methyl iodine	1	0.40	3.44	2	5.02	44.97	1	0.40	4.54
Methyl isobutyl ketone	3	4.90	42.12	3	4.90	43.89	3	4.90	55.70
Methyl methacrylate	1	1.10	9.45	1	1.10	9.86	1	1.10	12.51
Methyl tert-butyl ether	1	1.40	12.03	1	1.40	12.54	1	1.40	15.92
Methylene chloride	5	13.0	111.8	7	10.0	89.6	5	13.0	147.8
n-nitrosodimethylamine	1	0.68	5.84	1	0.68	6.09	1	0.68	7.73
Naphthalene	11	0.77	6.62	12	0.32	2.87	11	0.77	8.76
m,p-cresol	2	0.675	5.80	2	0.675	6.05	2	0.675	7.68
o-cresol	3	1.7	14.61	3	1.7	15.22	3	1.7	19.33
p-cresol	1	0.95	8.16	1	0.95	8.51	1	0.95	10.80
Perylene	1	0.075	0.65	1	0.075	0.67	1	0.075	0.86
Pentachlorophenol	1	0.008	0.07				1	0.008	0.09
Phenol	10	6.1	52.43	10	6.1	54.64	10	6.1	69.34

Table A-4. (Continued)

Coal-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
Phthalic anhydride	1	4.9	42.12	1	4.9	43.89	1	4.9	55.70
Propionaldehyde	4	10.35	88.96	4	10.35	92.71	4	10.35	117.66
Quinoline	1	0.053	0.45	1	0.053	0.47	1	0.053	0.61
Styrene	7	3.1	26.64	8	2.4	21.49	7	3.1	35.24
Tetrachloroethylene	5	3.1	26.64	5	3.1	27.76	5	3.1	35.24
Toluene	17	3.6	30.94	19	3.3	29.56	17	3.6	40.92
Trans-1,3-dichloropropene	1	4.7	40.39	1	4.7	42.10	1	4.7	53.43
Trichloroethylene	1	3.1	26.64	1	3.1	27.76	1	3.1	35.24
Vinyl acetate	1	0.42	3.61	1	0.42	3.76	1	0.42	4.77
Vinylidene chloride	2	9.7	83.37	1	6.7	59.69	2	9.7	110.28
Xylenes	2	4.65	39.96	4	3.05	27.32	2	4.65	52.87
o-xylenes	5	0.81	6.96	5	0.81	7.26	5	0.81	9.21
m,p-xylenes	8	1.45	12.46	9	1.2	10.75	8	1.45	16.49
Total TEQ for 2,3,7,8-tetra-chlorodibenzo-p-dioxin	17		9.7×10^{-5}	17		1.2×10^{-4}	17		1.1×10^{-4}
2,3,7,8-tetrachlorodi-benzo-p-dioxin	4	1.5×10^{-6}	1.3×10^{-5}	6	2.6×10^{-6}	2.3×10^{-5}	4	1.5×10^{-6}	1.8×10^{-5}
1,2,3,7,8-pentachlorodi-benzo-p-dioxin	3	2.8×10^{-6}	2.4×10^{-5}	3	2.8×10^{-6}	2.5×10^{-2}	3	2.8×10^{-6}	3.1×10^{-5}
1,2,3,4,7,8-hexachlorodi-benzo-p-dioxin	4	5.9×10^{-6}	5.1×10^{-5}	4	5.9×10^{-6}	5.3×10^{-5}	4	5.9×10^{-6}	6.7×10^{-5}
1,2,3,6,7,8-hexachlorodi-benzo-p-dioxin	4	6.6×10^{-6}	5.6×10^{-5}	4	6.6×10^{-6}	5.9×10^{-5}	4	6.6 x 10-6	7.5×10^{-5}

Table A-4. (Continued)

Coal-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
1,2,3,7,8,9-hexachlorodi-benzo-p-dioxin	**4**	**7.9 x 10^{-6}**	**6.7 x 10^{-5}**	**5**	**4.2 x 10^{-6}**	**3.7 x 10^{-5}**	**4**	**7.9 x 10-6**	**8.9 x 10^{-5}**
1,2,3,4,6,7,8-heptachlorodi-benzo-p-dioxin	**8**	**4.2 x 10^{-6}**	**3.6 x 10^{-5}**	**10**	**7.4 x 10^{-6}**	**6.6 x 10^{-5}**	**8**	**4.2 x 10-6**	**4.7 x 10^{-5}**
Heptachlorodi-benzo-p-dioxin	**6**	**7.6 x 10^{-5}**	**6.5 x 10^{-4}**	**6**	**7.6 x 10^{-6}**	**6.8 x 10^{-4}**	**6**	**7.6 x 10^{-5}**	**8.6 x 10^{-4}**
Hexachlorodi-benzo-p-dioxin	**7**	**2.7 x 10^{-5}**	**2.3 x 10^{-4}**	**7**	**2.7 x 10^{-5}**	**2.4 x 10^{-4}**	**7**	**2.7 x 10^{-5}**	**3.0 x 10^{-4}**
Octachlorodi-benzo-p-dioxin	**10**	**3.6 x 10^{-5}**	**3.1 x 10^{-4}**	**11**	**2.6 x 10^{-5}**	**2.3 x 10^{-4}**	**10**	**3.6 x 10^{-5}**	**4.1 x 10^{-4}**
Pentachlorodi-benzo-p-dioxin	**6**	**8.0 x 10^{-6}**	**6.9 x 10^{-5}**	**6**	**8.0 x 10^{-6}**	**7.1 x 10^{-5}**	**6**	**8.0 x 10-6**	**9.1 x 10^{-5}**
Tetrachlorodi-benzo-p-dioxin	**9**	**8.8 x 10^{-6}**	**6.8 x 10^{-5}**	**9**	**8.8 x 10^{-6}**	**7.1 x 10^{-5}**	**9**	**8.8 x 10-6**	**9.1 x 10^{-5}**
2,3,7,8-tetrachlorodi-benzofuran	**1**	**4.4 x 10^{-6}**	**3.8 x 10^{-5}**	**9**	**4.2 x 10^{-6}**	**3.7 x 10^{-5}**	**1**	**4.4 x 10-6**	**5.0 x 10^{-5}**
1,2,3,7,8-pentachlorodi-benzofuran	**1**	**4.6 x 10^{-6}**	**3.9 x 10^{-5}**	**5**	**4.6 x 10^{-6}**	**4.1 x 10^{-5}**	**1**	**4.6 x 10-6**	**5.2 x 10^{-5}**
2,3,4,7,8-pentachlorodi-benzofuran	**1**	**4.8 x 10^{-6}**	**4.2 x 10^{-5}**	**6**	**7.6 x 10^{-6}**	**6.8 x 10^{-5}**	**1**	**4.8 x 10-6**	**5.5 x 10^{-5}**
1,2,3,4,7,8-hexachlorodi-benzofuran	**1**	**7.9 x 10^{-6}**	**6.8 x 10^{-5}**	**6**	**7.3 x 10^{-6}**	**6.5 x 10^{-5}**	**1**	**7.9 x 10^{-6}**	**8.9 x 10^{-5}**
1,2,3,6,7,8-hexachlorodi-benzofuran	**1**	**4.0 x 10^{-6}**	**3.4 x 10^{-5}**	**8**	**4.0 x 10^{-6}**	**3.6 x 10^{-5}**	**1**	**4.0 x 10^{-6}**	**4.5 x 10^{-5}**
1,2,3,7,8,9-hexachlorodi-benzofuran	**4**	**6.8 x 10^{-6}**	**5.8 x 10^{-5}**	**5**	**6.3 x 10-6**	**5.6 x 10^{-5}**	**4**	**6.8 x 10-6**	**7.7 x 10^{-5}**
2,3,4,6,7,8-hexachlorodi-benzofuran	**5**	**1.2 x 10^{-5}**	**1.0 x 10^{-4}**	**6**	**1.4 x 10^{-5}**	**1.3 x 10^{-4}**	**5**	**1.2 x 10^{-5}**	**1.4 x 10^{-4}**
1,2,3,4,6,7,8-heptachlorodi-benzofuran	**8**	**5.7 x 10^{-6}**	**4.9 x 10^{-5}**	**9**	**7.1 x 10-6**	**6.4 x 10^{-5}**	**8**	**5.7 x 10^{-6}**	**6.5 x 10^{-5}**

Table A-4. (Continued)

Coal-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
1,2,3,4,7,8,9-heptachlorodi-benzofuran	**4**	**1.8×10^{-5}**	**1.6×10^{-4}**	**5**	**8.3 x 10-6**	**7.4×10^{-5}**	**4**	**1.8×10^{-5}**	**2.1×10^{-4}**
Heptachlorodi-benzofuran	**9**	**1.9×10^{-5}**	**1.6×10^{-4}**	**9**	**1.9×10^{-5}**	**1.7×10^{-4}**	**9**	**1.9×10^{-5}**	**2.2×10^{-4}**
Hexachlorodi-benzofuran	**8**	**2.1×10^{-5}**	**1.8×10^{-4}**	**8**	**2.1×10^{-5}**	**1.9×10^{-4}**	**8**	**2.1×10^{-5}**	**2.4×10^{-4}**
Octachlorodi-benzofuran	**9**	**1.7×10^{-5}**	**1.4×10^{-4}**	**10**	**1.9×10^{-5}**	**1.7×10^{-4}**	**9**	**1.7×10^{-5}**	**1.9×10^{-4}**
Pentachlorodi-benzofuran	**10**	**1.2×10^{-5}**	**1.0×10^{-4}**	**10**	**1.2×10^{-5}**	**1.1×10^{-4}**	**10**	**1.2×10^{-5}**	**1.3×10^{-4}**
Tetrachlorodi-benzofuran	**10**	**1.1×10^{-5}**	**9.8×10^{-5}**	**10**	**1.1×10^{-5}**	**1.0×10^{-4}**	**10**	**1.1×10^{-5}**	**1.3×10^{-4}**
1-methylnaphthalene	**3**	**0.011**	**0.098**	**5**	**0.011**	**0.102**	**3**	**0.01**	**0.132**
2-chloronaphthalene	**3**	**0.040**	**0.352**	**2**	**4.6×10^{-4}**	**0.004**	**3**	**0.04**	**0.462**
2-methylnaphthalene	**6**	**0.032**	**0.275**	**9**	**0.03**	**0.275**	**6**	**0.032**	**0.363**
Acenaphthene	**8**	**0.013**	**0.108**	**9**	**0.017**	**0.154**	**8**	**0.013**	**0.143**
Acenaphthylene	**7**	**0.004**	**0.036**	**10**	**0.0057**	**0.052**	**7**	**0.004**	**0.047**
Anthracene	**7**	**0.004**	**0.039**	**10**	**0.005**	**0.047**	**7**	**0.004**	**0.052**
Benz(a)anthracene	**6**	**0.002**	**0.018**	**9**	**0.004**	**0.040**	**6**	**0.002**	**0.024**
Benzo(a)pyrene	**8**	**0.001**	**0.009**	**7**	**0.0015**	**0.013**	**8**	**0.001**	**0.012**
Benzo(e)pyrene	**4**	**0.001**	**0.012**	**6**	**0.002**	**0.022**	**4**	**0.001**	**0.016**
Benzo(b)fluoranthene	**1**	**0.008**	**0.069**	**2**	**0.005**	**0.048**	**1**	**0.008**	**0.092**
Benzo(b+k)fluoranthene	**3**	**0.004**	**0.036**	**4**	**0.004**	**0.039**	**3**	**0.004**	**0.048**
Benzo(k)fluoranthene	**1**	**0.004**	**0.031**	**3**	**0.001**	**0.011**	**1**	**0.004**	**0.040**
Benzo(g,h,i)perylene	**4**	**0.002**	**0.019**	**6**	**0.002**	**0.017**	**4**	**0.002**	**0.025**
Biphenyl	**4**	**0.18**	**1.562**	**6**	**0.073**	**0.649**	**4**	**0.18**	**2.068**

Table A-4. (Continued)

Coal-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
Chrysene	6	0.003	0.022	9	0.006	0.054	6	0.003	0.030
Dibenzo(a,h)anthracene	2	0.001	0.011	5	7.0×10^{-4}	0.006	2	0.001	0.015
Fluoranthene	8	0.016	0.132	10	0.025	0.220	8	0.016	0.176
Fluorene	7	0.013	0.110	10	0.015	0.132	7	0.013	0.154
Indeno(1,2,3-c,d)pyrene	4	0.003	0.028	8	0.002	0.021	4	0.003	0.037
Phenanthrene	7	0.032	0.275	10	0.072	0649	7	0.032	0.363
Pyrene	7	0.012	0.106	10	0.013	0.121	7	0.012	0.143

Table A-5. Median Emission Factors, Determined from Test Report Data, and Total 1990, Total 1994 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 (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
Acetaldehyde	**1**	**8.2**	**5.0**	**1**	**8.2**	**3.8**	**1**	**8.2**	**2.5**
Benzene	**6**	**1.40**	**0.86**	**7**	**1.00**	**0.46**	**6**	**1.40**	**0.43**
Ethylbenzene	**2**	**0.49**	**0.30**	**2**	**0.49**	**0.23**	**2**	**0.49**	**0.15**
Formaldehyde	**9**	**30.0**	**18.4**	**11**	**20.0**	**9.3**	**9**	**30.0**	**9.3**
Methyl chloroform	**3**	**7.6**	**4.6**	**3**	**7.6**	**3.5**	**3**	**7.6**	**2.4**
Methylene chloride	**2**	**32.25**	**17.93**	**2**	**32.25**	**14.92**	**2**	**32.25**	**10.01**
Naphthalene	**4**	**0.34**	**0.21**	**10**	**0.61**	**0.28**	**4**	**0.34**	**0.10**
Phenol	**2**	**24.30**	**14.86**	**2**	**24.30**	**11.24**	**2**	**24.30**	**7.54**
Tetrachloroethylene	**1**	**0.55**	**0.34**	**1**	**0.55**	**0.25**	**1**	**0.55**	**0.18**
Toluene	**6**	**8.00**	**4.90**	**7**	**8.40**	**3.88**	**6**	**8.00**	**2.49**
Vinyl acetate	**2**	**5.15**	**3.15**	**2**	**5.15**	**2.39**	**2**	**5.15**	**1.60**
o-xylenes	**1**	**0.84**	**0.52**	**1**	**0.84**	**0.39**	**1**	**0.84**	**0.26**
m,p-xylenes	**2**	**1.35**	**0.83**	**2**	**1 .35**	**0.63**	**2**	**1.35**	**0.42**
Total TEQ for 2,3,7,8-tetra-chlorodibenzo-p-dioxin	**17**		**7.0 x 10^{-5}**	**17**		**8.8 x 10-6**	**17**		**3.0 x 10^{-6}**
2,3,7,8-tetrachlorodi-benzo-p-dioxin		[a]			[a]			[a]	

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[a] After a check was made of nondetect procedures used in 1990 and 2010 emissions, this emission factor did not pass the criteria for detection. Therefore, no 1994 emission number is available for this compound. Some of these compounds encompass a group of compounds, and although the total is not available, some members of the group may be presented elsewhere.

Table A-5. (Continued)

Oil-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
1,2,3,7,8-pentachlorodi-benzo-p-dioxin	**2**	**4.0×10^{-6}**	**2.4×10^{-6}**	**2**	**4.0×10^{-6}**	**1.9×10^{-6}**	**2**	**4.0×10^{-6}**	**1.2×10^{-6}**
1,2,3,4,7,8-hexachlorodi-benzo-p-dioxin	**2**	**9.9×10^{-6}**	**6.1×10^{-6}**	**2**	**9.9×10^{-6}**	**4.6×10^{-6}**	**2**	**9.9×10^{-6}**	**3.1×10^{-6}**
1,2,3,6,7,8-hexachlorodi-benzo-p-dioxin	**1**	**8.2×10^{-6}**	**5.0×10^{-6}**	**1**	**8.2×10^{-6}**	**3.8×10^{-6}**	**1**	**8.2×10^{-6}**	**2.5×10^{-6}**
1,2,3,7,8,9-hexachlorodi-benzo-p-dioxin	**1**	**9.6×10^{-6}**	**5.9×10^{-6}**	**1**	**9.6×10^{-6}**	**4.5×10^{-6}**	**1**	**9.6×10^{-6}**	**3.0×10^{-6}**
1,2,3,4,6,7,8-heptachlorodi-benzo-p-dioxin	**1**	**5.9×10^{-5}**	**3.6×10^{-5}**	**1**	**5.9×10^{-5}**	**2.7×10^{-5}**	**1**	**5.9×10^{-5}**	**1.8×10^{-5}**
Heptachlorodi-benzo-p-dioxin	**1**	**1.2×10^{-4}**	**7.3×10^{-5}**	**1**	**1.2×10^{-4}**	**5.5×10^{-5}**	**1**	**1.2×10^{-4}**	**3.7×10^{-5}**
Hexachlorodi-benzo-p-dioxin	**2**	**8.2×10^{-5}**	**5.0×10^{-5}**	**2**	**8.2×10^{-5}**	**3.8×10^{-5}**	**2**	**8.2×10^{-5}**	**2.5×10^{-5}**
Octachlorodi-benzo-p-dioxin	**2**	**1.4×10^{-4}**	**8.4×10^{-5}**	**2**	**1.4×10^{-4}**	**6.4×10^{-5}**	**2**	**1.4×10^{-4}**	**4.3×10^{-5}**
Pentachlorodi-benzo-p-dioxin	**2**	**8.0×10^{-5}**	**4.9×10^{-5}**	**2**	**8.0×10^{-5}**	**3.7×10^{-5}**	**2**	**8.0×10^{-5}**	**2.5×10^{-5}**
Tetrachlorodi-benzo-p-dioxin	**1**	**1.0×10^{-4}**	**6.3×10^{-5}**	**1**	**1.0×10^{-4}**	**4.8×10^{-5}**	**1**	**1.0×10^{-4}**	**3.2×10^{-5}**
2,3,7,8-tetrachlorodi-benzofuran	**1**	**6.7×10^{-6}**	**4.1×10^{-6}**	**1**	**6.7×10^{-6}**	**3.1×10^{-6}**	**1**	**6.7×10^{-6}**	**2.1×10^{-6}**
1,2,3,7,8-pentachlorodi-benzofuran	**1**	**8.2×10^{-6}**	**5.0×10^{-6}**	**1**	**8.2×10^{-6}**	**3.8×10^{-6}**	**1**	**8.2×10^{-6}**	**2.5×10^{-6}**
2,3,4,7,8-pentachlorodi-benzofuran	**1**	**5.9×10^{-6}**	**3.6×10^{-6}**	**1**	**5.9×10^{-6}**	**2.7×10^{-6}**	**1**	**5.9×10^{-6}**	**1.8×10^{-6}**
1,2,3,4,7,8-hexachlorodi-benzofuran	**1**	**9.6×10^{-6}**	**5.9×10^{-6}**	**1**	**9.6×10^{-6}**	**4.5×10^{-6}**	**1**	**9.6×10^{-6}**	**3.0×10^{-6}**
1,2,3,6,7,8-hexachlorodi-benzofuran	**1**	**4.4×10^{-6}**	**2.8×10^{-6}**	**1**	**4.4×10^{-6}**	**2.1×10^{-6}**	**1**	**4.4×10^{-6}**	**1.4×10^{-6}**

Table A-5. (Continued)

Oil-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
1,2,3,7,8,9-hexachlorodi-benzofuran		[a]			[a]			[a]	
2,3,4,6,7,8-hexachlorodi-benzofuran	**1**	**2.8×10^{-6}**	**1.7×10^{-6}**	**1**	**2.8×10^{-6}**	**1.3×10^{-6}**	**1**	**2.8×10^{-6}**	**8.7 x 10-7**
1,2,3,4,6,7,8-heptachlorodi-benzofuran	**1**	**2.0×10^{-5}**	**1.2×10^{-5}**	**1**	**2.0×10^{-5}**	**9.3×10^{-6}**	**1**	**2.0×10^{-6}**	**6.2×10^{-6}**
1,2,3,4,7,8,9-heptachlorodi-benzofuran		[a]			[a]			[a]	
Heptachlorodi-benzofuran	**1**	**2.4×10^{-5}**	**1.5×10^{-5}**	**1**	**2.4×10^{-5}**	**1.1×10^{-5}**	**1**	**2.4×10^{-5}**	**7.6×10^{-6}**
Hexachlorodi-benzofuran	**1**	**2.3×10^{-5}**	**1.4×10^{-5}**	**1**	**2.3×10^{-5}**	**1.1×10^{-5}**	**1**	**2.3×10^{-5}**	**7.1×10^{-6}**
Octachlorodi-benzofuran	**1**	**2.1×10^{-5}**	**1.3×10^{-5}**	**1**	**2.1×10^{-5}**	**9.6×10^{-6}**	**1**	**2.1×10^{-5}**	**6.4×10^{-6}**
Pentachlorodi-benzofuran	**1**	**4.0×10^{-5}**	**2.4×10^{-5}**	**1**	**4.0×10^{-5}**	**1.9×10^{-5}**	**1**	**4.0×10^{-5}**	**1.2×10^{-5}**
Tetrachlorodi-benzofuran	**1**	**1.0×10^{-4}**	**6.3×10^{-5}**	**1**	**1.0×10^{-4}**	**4.8×10^{-5}**	**1**	**1.0×10^{-4}**	**3.2×10^{-5}**
2-methylnaphthalene	**4**	**0.017**	**0.010**	**5**	**0.027**	**0.013**	**4**	**0.017**	**0.005**
Acenaphthene	**2**	**0.358**	**0.220**	**4**	**0.080**	**0.037**	**2**	**0.358**	**0.110**
Acenaphthylene	**1**	**0.017**	**0.010**	**1**	**0.017**	**0.008**	**1**	**0.017**	**0.005**
Anthracene	**2**	**0.015**	**0.009**	**3**	**0.013**	**0.006**	**2**	**0.015**	**0.005**
Benz(a)anthracene	**3**	**0.030**	**0.018**	**3**	**0.030**	**0.014**	**3**	**0.030**	**0.09**

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[a] After a check was made of nondetect procedures used in 1990 and 2010 emissions, this emission factor did not pass the criteria for detection. Therefore, no 1994 emission number is available for this compound. Some of these compounds encompass a group of compounds, and although the total is not available, some members of the group may be presented elsewhere.

Table A-5. (Continued)

Oil-fired units: organic HAPs	Number of emission factors (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
Benzo(b+k)fluoranthene	**2**	**0.033**	**0.020**	**2**	**0.033**	**0.015**	**2**	**0.033**	**0.010**
Benzo(g,h,i)perylene	**2**	**0.021**	**0.013**	**2**	**0.024**	**0.011**	**2**	**0.021**	**0.006**
Chrysene	**3**	**0.021**	**0.013**	**3**	**0.021**	**0.010**	**3**	**0.021**	**0.007**
Dibenzo(a,h)anthracene	**2**	**0.008**	**0.005**	**2**	**0.010**	**0.005**	**2**	**0.008**	**0.003**
Fluoranthene	**6**	**0.016**	**0.010**	**7**	**0.024**	**0.011**	**6**	**0.016**	**0.005**
Fluorene	**5**	**0.021**	**0.013**	**8**	**0.018**	**0.009**	**5**	**0.021**	**0.006**
Indeno(1,2,3-c,d)pyrene	**2**	**0.024**	**0.014**	**2**	**0.024**	**0.011**	**2**	**0.024**	**0.007**
Nitrobenzofluoranthene	**1**	**0.015**	**0.009**	**1**	**0.015**	**0.007**	**1**	**0.015**	**0.005**
Nitrochrysene/benzanthracene	**1**	**0.016**	**0.010**	**1**	**0.016**	**0.007**	**1**	**0.016**	**0.005**
Phenanthrene	**9**	**0.025**	**0.015**	**11**	**0.036**	**0.017**	**9**	**0.025**	**0.008**
Pyrene	**6**	**0.037**	**0.022**	**8**	**0.010**	**0.004**	**6**	**0.037**	**0.011**

Table A-6. (REVISED) 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 (1990)	Median emission factor: lb/trillion Btu (1990)	Computer program: 1990 total tons	Number of emission factors (1994)	Median emission factor: lb/trillion Btu (1994)	Computer program: 1994 total tons	Number of emission factors (2010)	Median emission factor: lb/trillion Btu (2010)	Computer program: 2010 total tons
Benzene	1	1.4	1.72	1	1.	1.89	1	1.4	2.73
Formaldehyde	8	29.0	35.62	8	29.	39.23	8	29.0	56.58
Naphthalene	2	0.67	0.83	2	0.67	0.90	2	0.67	1.31
Toluene	2	10.2	12.53	2	10.2	13.79	2	10.2	19.90
2-methylnaphthalene	2	0.026	0.031	2	0.026	0.035	2	0.026	0.050
Fluoranthene	1	0.003	0.003	1	0.0028	0.004	1	0.003	0.005
Fluorene	1	0.003	0.003	1	0.0026	0.004	1	0.003	0.005
Phenanthrene	2	0.013	0.016	2	0.013	0.017	2	0.013	0.025
Pyrene	1	0.005	0.006	1	0.005	0.007	1	0.005	0.010

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. The matrix was then used only as a guide to gather data on the largest number of unit configurations possible with the available resources by targeting the most prevalent unit types. It should be noted that the totals in Table B-1 were taken from the 1991 EEI Power Statistics Database and do not correlate with the 1994 industry statistics given in Chapter 2.

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.

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
Subbituminous coal	<1.5	Dry bottom	Fabric filter	SDA		< 0.55		< 1.05	2

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]
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)
DOE	Paradise	Costain coal	Medium	Wet	ESP	Southern Research Institute	9	5/95 (F)
NSPC	A.S. King	Subbituminous coal	Low	Wet	ESP	Interpoll	10	11/91 (P)
NSPC	Black Dog 1,3,4	Subbituminous coal	Low	Dry	ESP	Interpoll	11	1/92 (P)
NSPC	Black Dog 2	Subbituminous coal	Low	AFBC	ESP	Interpoll	12	5/92 (P)
NSPC	High Bridge 3,4,5,6	Subbituminous coal	Low	Dry	ESP	Interpoll	13	1/92 (P)
NSPC	Riverside 6,7	Subbituminous coal	Low	Dry	FF	Interpoll	14	2/92 (P)
NSPC	Riverside 8	Subbituminous coal	Low	Wet	ESP	Interpoll	15	9/92 (P)
NSPC	Sherburne 1,2	Subbituminous coal	Low	Dry	FGD	Interpoll	16 17 18	7/90, 10/91 (P)
NSPC	Sherburne 3	Subbituminous coal	Low	Dry	SDA/FF	Interpoll	19	6/90, 10/91 (P)
EPRI	Site 10	Bituminous coal	Low	AFBC	FF	Radian	20	10/92 (P)
EPRI	Site 101	Subbituminous coal	Low	Dry	FF/FGD	Radian	21	10/94 (P)
EPRI	Site 102	Subbituminous coal	Low	Wet	ESP	Radian	22	2/93 (P)
EPRI	Site 11	Subbituminous coal	Low	Dry	ESP/FGD	Radian	23	10/92, 10/93 (P)
EPRI/SRI	Site 110	Bituminous coal	Medium	Dry	H-ESP/ C-ESP	SRI	24	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	SRI	24	10/93 (P)
EPRI	Site 111	Bituminous/ Subbituminous coal	Low	Dry	SDA/FF	Radian	25	1/94 (P)
EPRI	Site 112	Oil	Low	Dry	ESP	Carnot	26	12/93, 3/94 (P)
EPRI	Site 113	Oil	Medium	Dry	Multicyclone	Carnot	27	3/94 (P)
EPRI	Site 114	Bituminous coal	Medium	Wet	ESP	NA	28	5/94 (P)
EPRI	Site 115	Bituminous coal	Low	Dry	FF	Carnot	29	NA
EPRI	Site 116	Bituminous coal	Medium	Dry	ESP	Radian	30	10/94 (P)
EPRI	Site 117	Oil	Low	Dry	SCR	Carnot	31	1/94 (P)
EPRI	Site 118	Oil	Low	Dry	ESP	Carnot	32	1/94 (P)
EPRI	Site 119	Oil	Low	Dry	ESP	Carnot	33	1/94 (P)
EPRI	Site 12	Bituminous coal	Medium	Dry	ESP/FGD	Radian	34	11/92, 10/93 (P)
EPRI	Site 120	Gas		Dry	None	NA	35	NA
EPRI	Site 121	Gas		Dry	None	NA	36	NA
EPRI	Site 125	Bituminous coal	Medium	Wet	ESP	SRI	37	8-95 (P)
EPRI	Site 13	Oil		Dry	PJFF	Radian	38	2/93 (P)
EPRI	Site 14	Bituminous coal	Medium	Dry	SDA/PJFF	Radian	39	11/92 (P)
EPRI	Site 15	Bituminous coal	Medium	Dry	ESP	Radian	40	10/92 (P)
EPRI	Site 16/OFA	Bituminous coal	Medium	Dry	ESP	Radian	41	11/93 (P)
EPRI	Site 16/OFA/Low NO_x	Bituminous coal	Medium	Dry	ESP	Radian	41	11/93 (P)
EPRI	Site 18	Bituminous coal	Low	Dry	ESP/ COHPAC	Radian	42	4/93 (P)
EPRI	Site 19	Bituminous coal	Low	Dry	ESP	Radian	43	4/93 (P)
EPRI	Site 20	Lignite coal	Medium	Dry	ESP/FGD	Radian	44	3/94 (P)

Table B-2. (continued)

Provider[a]	Site	Fuel	Fuel sulfur	Bottom type	Control[b]	Test contractor	Reference	Report date[c]
EPRI	Site 21	Bituminous coal	Medium	Dry	ESP/FGD	Radian	45	5/93 (P)
EPRI	Site 22	Bituminous coal	Low	Dry	ESP	Radian/Carnot	46	3/93, 2/94 (P)
EPRI	Site 103	Oil	Low	Dry	None	Carnot	47	3/93 (P)
EPRI	Site 104	Oil	Low	Dry	None	Carnot	47	3/93 (P)
EPRI	Site 105	Oil	Low	Dry	None	Carnot	47	3/93 (P)
EPRI	Site 106	Oil	Low	Dry	None	Carnot	47	3/93 (P)
EPRI	Site 107	Oil	Low	Dry	None	Carnot	47	3/93 (P)
EPRI	Site 108	Oil	Medium	Dry	None	Carnot	47	3/93 (P)
EPRI	Site 109	Oil	Low	Dry	None	KVB - Analect	47	3/93 (P)
EPA	T. H. Wharton (Combined Cycle Gas Turbine)	Gas			Water injection	Entropy	48	9/93 (F)
EPA	Northport	Oil	Low	Dry	ESP	Entropy	49	9/93 (F)
EPA	Hammond (TNP One)	Lignite coal	Low	AFBC	FF	Entropy	50	10/93 (F)
EPA	Greens Bayou	Gas		Dry	None	Entropy	51	11/93 (F)
EPA	Kintigh	Bituminous coal	Medium	Dry	ESP/FGD	Entropy	52	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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11. Results of the Air Toxic Emission Study on the Nos. 1, 3, & 4 Boilers at the NSC Black Dog Plant, prepared by Interpoll Laboratories, Inc., for NSPC, report No. 1-3451. January 1992.

12. 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.

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

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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.

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	EPRI Site 110 w/NO_x control	DOE - Coal Creek	DOE - Springerville	EMF (Geometric mean)	Geometric standard deviation
Antimony		0.66	0.01	0.03	0.06	4.74
Arsenic	0.92	0.39	0.69	0.29	0.52	1.11
Beryllium	0.79	0.43	0.35	0.87	0.57	1.08
Cadmium	0.35	0.70	0.11	1.00	0.41	1.42
Chromium	0.72	1.00	1.00	0.68	0.84	1.02
Cobalt	0.92	1.00	0.61	0.73	0.80	1.02
Lead	1.00	0.36	0.29	0.19	0.37	1.21
Manganese	0.98	0.76	0.59	0.72	0.75	1.02
Mercury			0.85	1.00	0.92	1.00
Nickel	0.25	0.97	1.00	0.70	0.64	1.18
Selenium	1.00	0.82	0.38	0.93	0.74	1.08

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	EPRI Site 110	DOE - Yates	EMF (Geometric mean)	Geometric standard deviation
Antimony		0.55	0.67	0.61	1.01
Arsenic	0.60	0.89	1.00	0.81	1.02
Beryllium	0.54	0.93	1.00	0.80	1.04
Cadmium	0.01	1.00	1.00	0.23	8.72
Chromium	0.58	1.00	1.00	0.84	1.03
Cobalt	0.94	0.93	0.97	0.95	1.00
Lead	1.00	1.00	1.00	1.00	1.00
Manganese	0.81	0.71	1.00	0.83	1.01
Mercury		0.66	1.00	0.81	1.02
Nickel	0.43	0.91	0.84	0.69	1.06
Selenium	0.70	0.58	0.70	0.66	1.00

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 (Note 1)	0.74	1.00		1.00	0.64	0.81	1.22
Nickel	0.29	0.37	0.67	0.02	0.54	0.33	0.24	2.25
Selenium	1.00	0.05	0.21		0.37	1.00	0.33	3.51

Note 1 - This EMF was obtained from the mercury retest.

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 - Boswell	1990 EMF (Geometric mean)	1990 Geometric standard deviation	EPRI Site 116	1994 Geometric Mean	1994 Geometric Standard Deviation
Antimony	0.20	0.59	0.34	1.16	0.12	0.24	1.93
Arsenic	0.99	0.23	0.48	1.30	0.7	0.54	1.76
Beryllium	0.40	0.60	0.49	1.02	0.35	0.44	1.08
Cadmium	0.25	1.00	0.50	1.27	0.12	0.31	3.18
Chromium	1.00	1.00	1.00	1.00	0.27	0.65	1.77
Cobalt		0.98	0.98	1.00	0.3	0.54	1.42
Lead	0.19	0.42	0.28	1.08	0.26	0.28	1.16
Manganese	0.77	0.57	0.66	1.01	0.21	0.45	1.58
Mercury	1.00	0.87	0.93	1.00	0.97	0.94	1.01
Nickel	0.78	1.00	0.88	1.00	0.23	0.56	1.86
Selenium	1.00	0.14	0.37	1.63	0.47	0.40	2.71

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	DOE - Niles #2	DOE - Niles #2 w/NOx Control	DOE- Baldwin	EMF (Geometric mean)	Geometric standard deviation
Antimony		0.61		1.00	0.92	0.32	0.65	1.02
Arsenic	0.48	0.51	0.15	0.58	0.85	1.00	0.51	1.20
Beryllium	0.04	0.08	0.15	0.26	0.30	0.44	0.16	1.38
Cadmium	0.02	0.16	0.01	0.11	0.20	1.00	0.10	3.79
Chromium	0.25	0.22	0.30	0.28	0.35	0.42	0.29	1.02
Cobalt	0.21			0.20	0.33	0.34	0.26	1.03
Lead	0.61	0.38	0.50	0.56	0.60	0.53	0.52	1.01
Manganese	0.33	0.13	0.20	0.15	0.19	0.20	0.19	1.04
Mercury	1.00	1.00	0.73	1.00	1.00	0.92	0.93	1.01
Nickel	0.30	0.12	0.72	0.29	0.39	0.43	0.33	1.15
Selenium	0.65	1.00	0.26	1.00	0.92	0.04	0.43	1.96

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 Cyclone-fired, Wet-bottom Furnace With NO_x Control (Coal-fired)

Unit Name	EPRI Site 114, NO_x	EMF (Geometric mean)	Geometric standard deviation
Antimony			N/A
Arsenic	0.25	0.25	N/A
Beryllium	0.15	0.15	N/A
Cadmium	0.01	0.01	N/A
Chromium	0.23	0.23	N/A
Cobalt			N/A
Lead	0.84	0.84	N/A
Manganese	0.18	0.18	N/A
Mercury	0.54	0.54	N/A
Nickel	0.31	0.31	N/A
Selenium	1.00	1.00	N/A

Table C-9. 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 - Cardinal	EMF (Geometric mean)	Geometric standard deviation
Antimony	0.08	0.08	N/A
Arsenic	0.91	0.91	N/A
Beryllium	0.96	0.96	N/A
Cadmium	1.00	1.00	N/A
Chromium	0.61	0.61	N/A
Cobalt	0.96	0.96	N/A
Lead	1.00	1.00	N/A
Manganese	0.27	0.27	N/A
Mercury	0.41	0.41	N/A
Nickel	0.76	0.76	N/A
Selenium	0.07	0.07	N/A

Table C-10. 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-11. 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

Note: This EMF suggests that this boiler type does a good job of controlling/reducing mercury from the fuel oil being burned. Nothing in the test report suggests that there were any problems encountered with this test sample so EPA chose to leave it in the EFP. It should be noted that all other tested oil-fired boilers had mercury EMFs of 1.0.

Table C-12. 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	1990 EMF (Geometric mean)	1990 Geometric standard deviation	EPRI Site 113	1994 EMF (Geometric mean)	1994 Geometric standard deviation
Arsenic	0.64	0.14	1.00	0.44	2.83	0.21	0.37	1.95
Beryllium		1.00	1.00	1.00	1.00		1.00	1.00
Cadmium	0.47		1.00	0.69	1.71	1.00	0.78	1.27
Chromium	0.19	0.78	1.00	0.53	2.42	1.00	0.62	1.68
Cobalt	0.62	0.29	0.98	0.56	1.83	0.40	0.52	1.58
Lead	0.08	0.57	0.97	0.35	3.73	0.37	0.36	2.18
Manganese	1.00	1.00	1.00	1.00	1.00	0.93	0.98	1.02
Mercury	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Nickel	0.71	0.64	1.00	0.77	1.26	0.40	0.65	1.32
Selenium	0.58	0.46	1.00	0.64	1.50	1.00	0.72	1.30

Table C-13. 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-14. 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-15. Tested EMFs and Geometric Means Used in the Emission Factor Program for Fabric Filters (baghouses)

Unit Name	EPRI Site 10	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		0.005	0.06	0.02	1.66
Arsenic	0.001	0.03	0.03	0.004	0.009	0.01	2.20
Beryllium	0.004	0.06	0.003	0.001	0.01	0.01	2.51
Cadmium		1.00	0.05	0.01	0.06	0.08	3.47
Chromium		0.05	0.013	0.003	0.004	0.01	1.84
Cobalt	0.004		0.007	0.001	0.006	0.004	1.36
Lead	0.002	0.03	0.007	0.001	0.01	0.01	2.70
Manganese	0.018	0.05	0.005	0.005	0.003	0.01	1.66
Mercury		1.00	0.27	0.92	0.39	0.56	1.17
Nickel	0.002	0.20	0.05	0.001	0.007	0.01	11.26
Selenium		0.06	0.02	0.79	0.31	0.12	3.09
Hydrogen Chloride	Note 1					0.56	
Hydrogen Fluoride	Note 1					1.00	

Note 1 - These EMFs were developed from emission tests that examined HCl and HF emissions through a baghouse.

Table C-16. 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.

Table C-17. 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		0.648	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-18. 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-19. Tested EMFs and Geometric Means Used in the Emission Factor Program for Flue 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	1990 EMF (Geometric mean)	1990 Geometric standard deviation	EPRI Site 20	EPRI Site 101	DOE - Paradise	1994 (Geometric mean)	1994 Geometric standard deviation
Antimony					0.21	0.16	0.18	1.23			0.28	0.21	1.26
Arsenic	0.49	0.23	0.04		0.08	0.76	0.20	3.29	0.19	0.57	0.14	0.21	2.59
Beryllium			0.02		0.11	0.97	0.13	6.96	0.15	0.33	0.03	0.12	3.77
Cadmium	0.85	0.62	0.16		0.75		0.50	2.16	1.00	0.83	0.10	0.47	2.35
Chromium	0.51	0.40	0.03		0.18	1.00	0.26	3.72	0.47	0.90	0.07	0.28	3.20
Cobalt	1.00	0.23			0.23	0.91	0.46	2.30	0.24	0.41	0.10	0.34	2.15
Lead	1.00	0.59	0.02		0.04	0.93	0.22	6.36	0.45	0.33	0.09	0.22	4.02
Manganese	1.00	0.03			0.17	1.00	0.26	5.65	0.32	1.00	0.01	0.21	5.03
Mercury	0.89 (Note 1)	1.00 (Note 2)	0.77	0.41	0.54	0.88	0.71	1.42	0.80	0.38	0.55	0.66	1.40
Nickel	0.88	1.00	0.05		1.00	1.00	0.53	3.79	0.78	1.00	0.01	0.37	5.31
Selenium	0.17	0.27	0.17		0.26	0.60	0.26	1.66	0.21	0.47	0.71	0.31	1.62
Hydrogen Chloride						Note 3	0.21			Note 3	0.20		
Hydrogen Fluoride						Note 3	0.73			Note 3	0.71		

Note 1 - This EMF was obtained from the mercury retest.

Note 2 - This EMF was obtained from the mercury retest. Because emissions testing was done at the inlet to the ESP and the outlet of the FGD only, EPA chose to give all of the mercury removal potential to the ESP (see Table C-21).

Note 3 - See section 3.4.7. for explanation of HCl and HF EMF Development.

Table C-20. 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	0.0003			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 - See section 3.4.7. for explanation of HCl and HF EMF Development.

Table C-21. 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
Antimony					0.25		0.10	0.32			0.004
Arsenic	0.01	0.004	0.02	0.06	0.01	0.01	0.01	0.01	0.05	0.05	0.03
Beryllium	0.10	0.002	0.01	0.001	0.01	0.02	0.003	0.02	0.06	0.02	0.005
Cadmium	0.03	0.06	0.41	0.11	0.11	0.04	0.99	0.13	0.10	0.02	0.03
Chromium	0.03	0.01	0.01	0.06	0.07	0.01	0.04	0.07	0.05	0.03	0.01
Cobalt	0.01	0.01	0.01	0.03							0.001
Lead	0.04	0.03	0.01	0.02	0.02	0.05	0.66	0.04	0.06	0.06	0.003
Manganese	0.01	0.04	0.005	0.03	0.03	0.02	0.02	0.05	0.05	0.04	0.01
Mercury	1.00	0.18 (Note 1)		1.00	0.93	0.92	0.78	1.00	0.70	0.84	0.73
Nickel	0.01	0.003	0.01	0.37	0.03	0.04	0.15	0.03	0.05	0.03	0.001
Selenium	0.04	0.24	0.65	0.004	0.02	0.03	0.02	0.01	1.00	1.00	0.97
Hydrogen Chloride											
Hydrogen Fluoride											

Unit Name	DOE - Yates	DOE - Coal Creek	EPRI Site 16 w/OFA and LNOX Burners	EPRI Site 16 w/OFA	DOE - Cardinal	DOE - Baldwin	1990 EMF (Geometric Mean)	1990 Geometric Standard Deviation	EPRI Site 116	1994 EMF (Geometric Mean)	1994 Geometric Standard Deviation
Antimony	0.01	0.02	0.03	0.04	0.02	0.10	0.04	4.21	0.12	0.05	3.68
Arsenic	0.04	0.001	0.06	0.05	0.003	0.01	0.02	3.07	0.01	0.01	3.06
Beryllium	0.02	0.05	0.02	0.03	0.001	0.03	0.01	4.13	0.73	0.02	4.34
Cadmium	0.04	0.18	0.17	0.05	0.03	0.05	0.08	2.93	1	0.09	3.10
Chromium	0.01	0.005	0.02	0.04	0.01	0.05	0.02	2.50	0.003	0.02	2.55
Cobalt	0.02	0.01	0.02	0.03	0.002	0.07	0.01	3.52	0.29	0.02	4.15
Lead	0.02	0.002	0.03	0.08	0.01	0.06	0.03	4.03	0.002	0.02	3.99
Manganese	0.02	0.003	0.02	0.03	0.01	0.03	0.02	2.30	0.03	0.02	2.14
Mercury	0.45	0.87	0.45	0.91	0.26	0.74	0.67	1.13	0.92	0.68	1.13
Nickel	0.01	0.005	0.02	0.04	0.01	0.03	0.02	4.03	0.11	0.02	3.83
Selenium	0.62	0.13	1.00	0.27	0.71	1.00	0.17	6.36	0.32	0.02	6.04
Hydrogen Chloride						Note 1	0.93		Note 1	0.94	
Hydrogen Fluoride						Note 1	1.00		Note 1	1.00	

Note 1 - See Note 2 in Table C-19.

Note 1 - These EMFs were developed from emission tests that examined HCl and HF emissions through an ESP.

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

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 and 1994 editions) and an extract from Production Costs, U.S. Gas Turbine and Combined-Cycle Power Plants (for 1994 estimates). 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.

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 and 1996 edition addressing 1994 data). Only units that

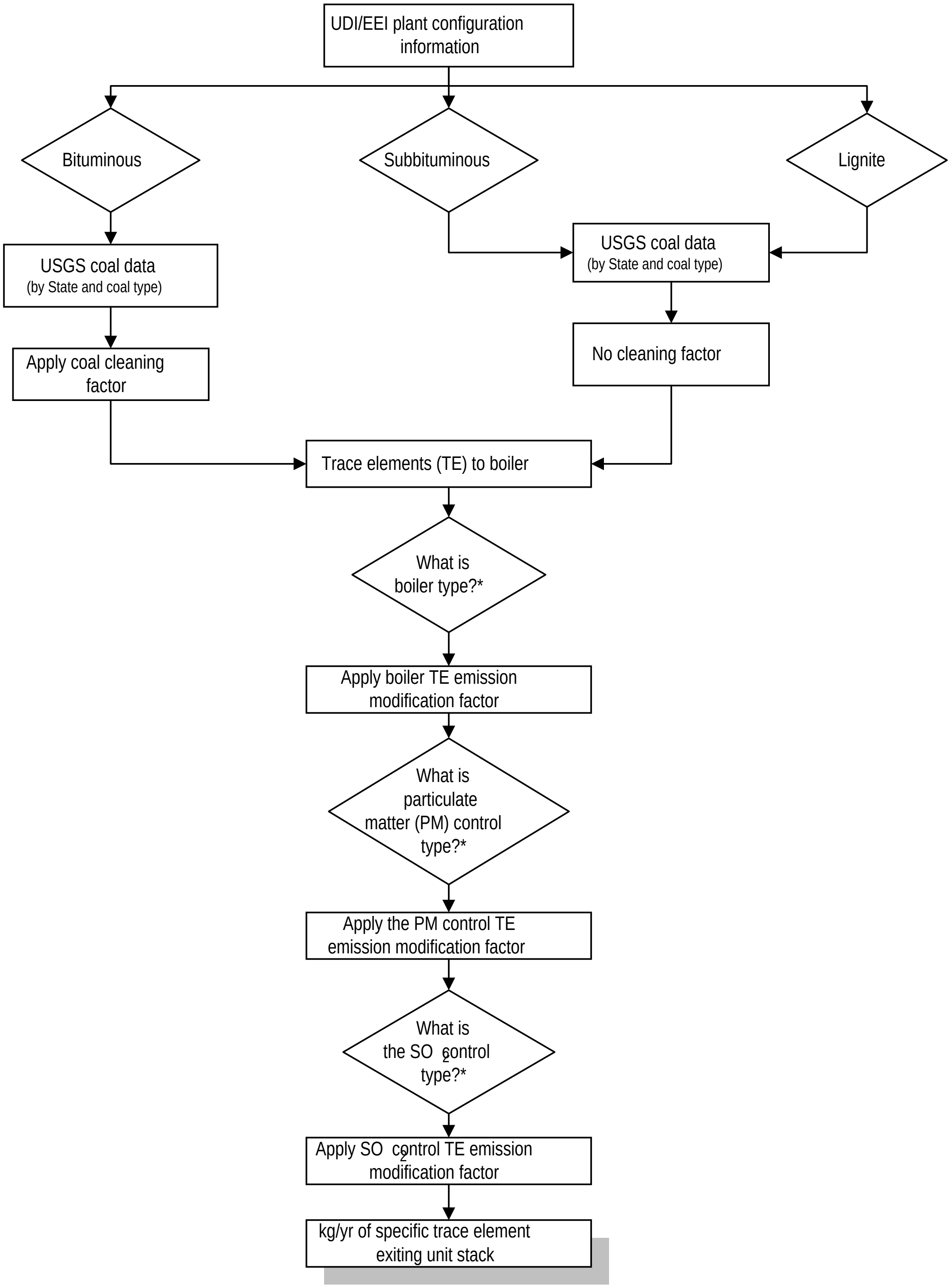


Figure D-1. Trace elements in coal.

*Taken from UDI/EEI data.

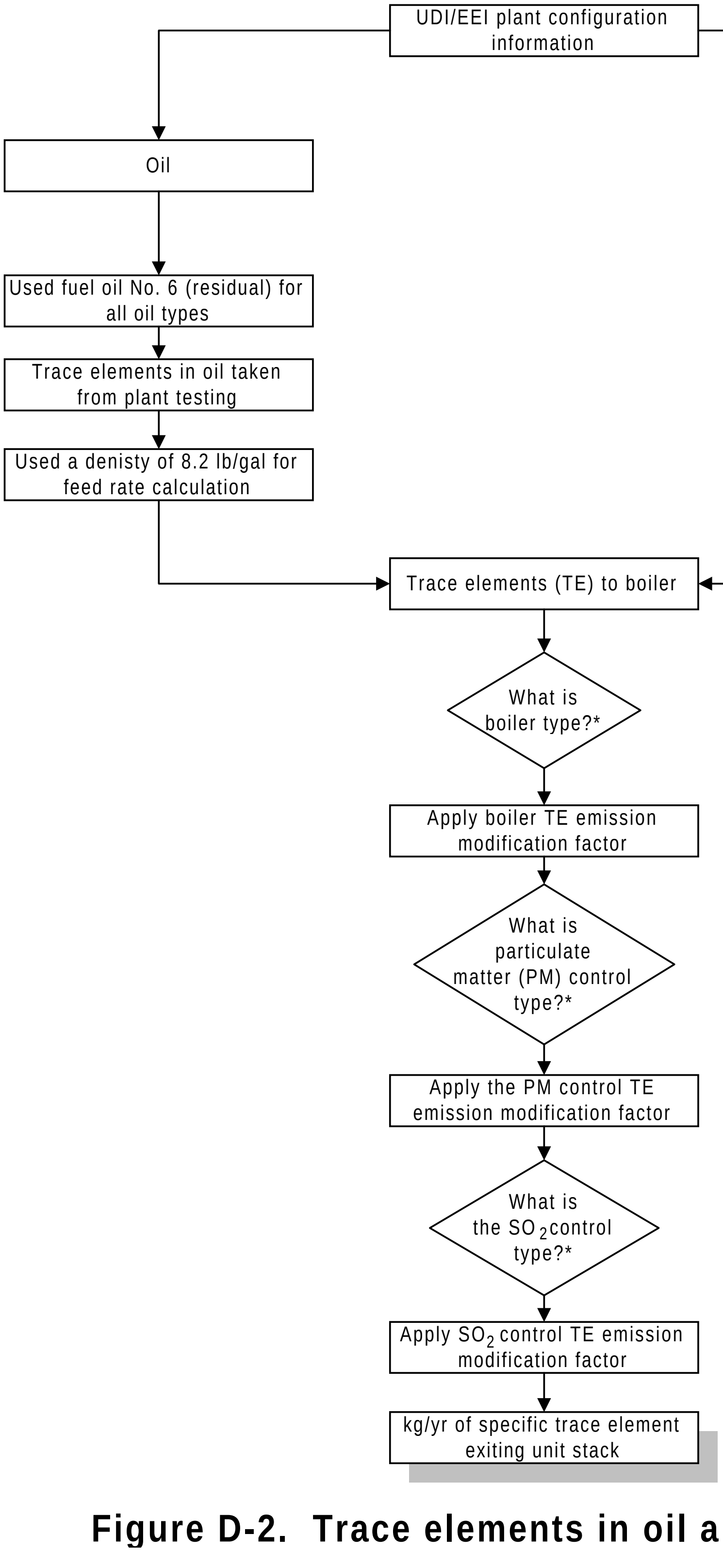
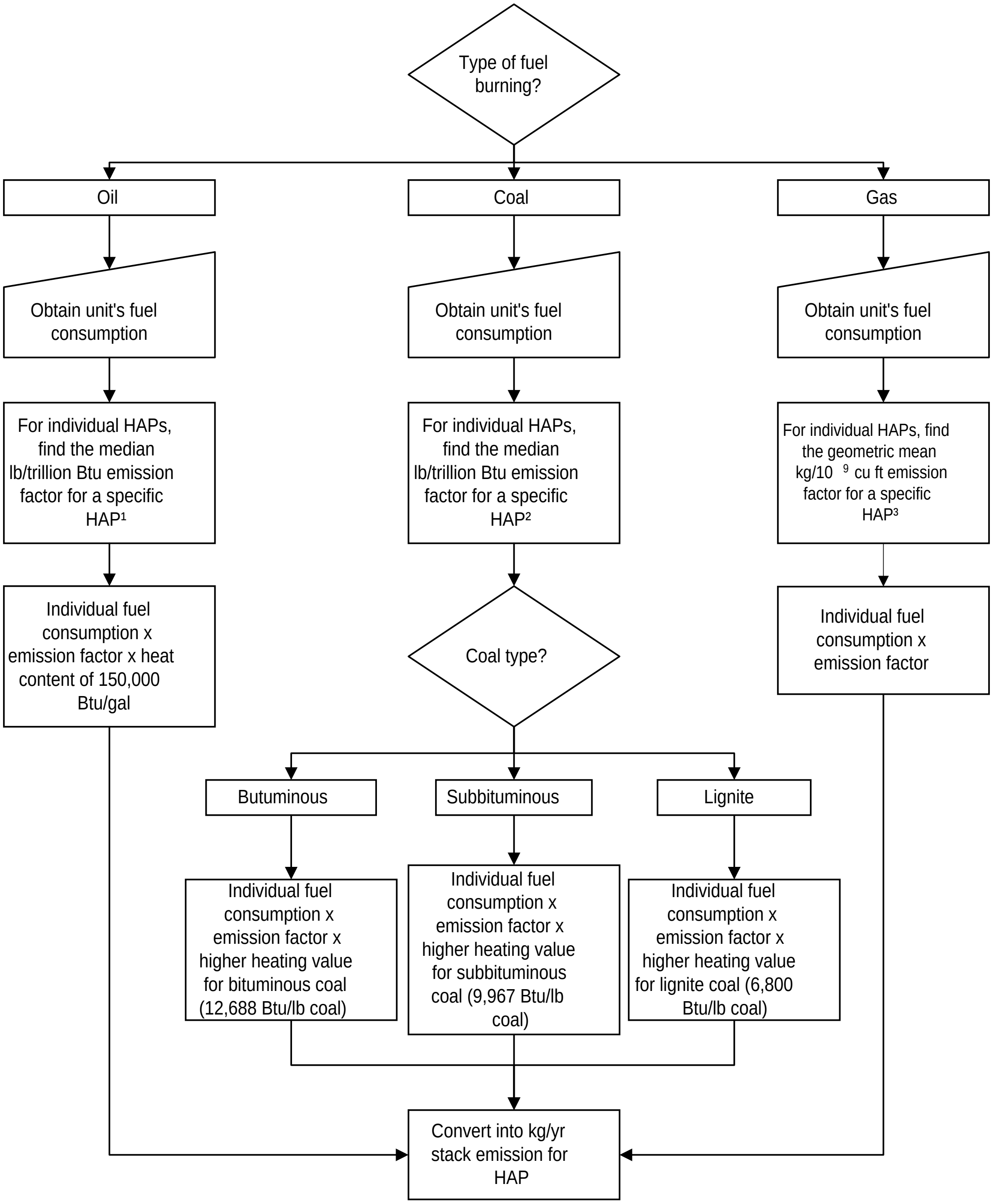


Figure D-2. Trace elements in oil and natural gas.

*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.

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 1990 EEI/UDI Power Statistics database but were actually on indefinite standby and thus did not emit any HAPs. These units were excluded from the nationwide emissions totals in Appendix A. Other units listed on indefinite standby (i.e., no fuel burned) were excluded from 1994 emission estimates.

Only coal-fired, oil-fired, and natural gas-fired units were included in the EFP. This decision was made because units using these fuels make up an overwhelming majority of the fossil-fuel–fired electric utility units with a capacity >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 and 1994) were assigned to burn bituminous coal for program computations.

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, EPA plotted the available data and fitted point-slope equations to estimate fuel consumption.[2] These equations involved plotting nameplate megawatts (modified to take into account the unit's capacity factor) against fuel usage. If the fuel usage and the unit capacity factor in 1990 were not given, 1989 fuel consumption data were used. If 1989 data were not available, the geometric mean of the 1980–1988 EEI fuel consumption data was used. When all other options had been tried unsuccessfully, an average fuel consumption of units rated within ±5 MW of the unit with unknown fuel usage was used. Similar problems in the 1994 UDI/EEI database were solved by using 1990 data where possible and by similar methods to those stated above when not possible.

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 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 for 1990 and 1994.[3]

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.[4] The State of origin used in the EFP was the State that contributed the highest percentage of the unit's coal. Coal consumption by State for each utility was found in volumes of *Cost and Quality of Fuels for Electric Utility Plants* for 1990[5] and 1994.[6]

D.5 BOILER CONFIGURATION

The EPA received 51 emissions tests conducted by EPRI, DOE, and industry in time for inclusion in the EFP for 1990 emission estimates. A further seven reports were available for the 1994 emission estimates. Because of this limited sample, not all boiler configurations, particulate, and SO_2 control types could be sampled. To estimate the emissions from all units in the United States, 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. The Fourier Transform Infrared (FTIR) system was used to test a combined-cycle gas turbine 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, four 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.

For 1990 emission estimates, 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 were 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 considered the most equitable way at the time to represent dual-fuel-fired emissions, for both trace metals and organic HAPs created by either oil-fired or gas-fired boilers, respectively.

For 1994 estimates, where units fired more than one type of fuel, emissions were modeled for each fuel. If the unit was listed as a coal-fired unit, its coal consumption rate and configuration were used to obtain its HAP emission rate for coal. Similarly, if the unit also fired oil, its oil consumption rate and equivalent boiler configuration were used to obtain its HAP emission rate for oil. If gas were also fired, the unit’s gas consumption rate and equivalent gas-fired boiler type were used obtain its HAP emission rate for gas.

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); ESP, hot-side (ESP,HS); ESP, cold-side, controlling an oil-fired unit (O-ESP,CS); fabric filter (FF); particulate scrubber; or no 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 were 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 TANGDRYNONOX 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. Table D-1 shows the boiler substitutions and associated PM control devices.

Emission modification factors for particulate control by scrubbers were derived from data on controlling trace elements by one venturi scrubber used for combined SO_2 and PM control. Particulate matter scrubbers use water only, while flue gas desulfirization 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, i.e., emissions from bypassed gas are included in the EFP results.

D.6 STACK CHARACTERISTICS

Stack data for 1990 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

Table D-1. Boiler Substitutions Used in the Emission Factor Program (EFP)

Facility Boiler	Boiler Used in EFP	Associated Control Device
CFBDRYNONOX	CFBDRYNOX	BAGHOUSE
CFBDRYNOX	CFBDRYNOX	BAGHOUSE
COMCYCLNONOX	G-FRONTDRYNONOX	NO CONTROL
COMCYCLNOX	G-FRONTDRYNONOX	NO CONTROL
COMCYCLNOX	O-FRONTDRYNONOX	NO CONTROL
CYCLWETNONOX	CYCLWETNONOX	ESP, CS
CYCLWETNONOX	TANGDRYNONOX	ESP, HS
CYCLWETNOX	CYCLWETNONOX	BAGHOUSE
FRONTDRYNONOX	FRONTDRYNONOX	BAGHOUSE
FRONTDRYNONOX	TANGDRYNONOX	ESP, HS
FRONTDRYNOX	FRONTDRYNONOX	BAGHOUSE
FRONTDRYNOX	TANGDRYNONOX	ESP, HS
FRONTWETNONOX	CYCLWETNONOX	BAGHOUSE
FRONTWETNONOX	TANGDRYNONOX	ESP, HS
OPPODRYNONOX	TANGDRYNOX	ESP, HS
OPPODRYNONOX	OPPODRYNOX	BAGHOUSE
OPPODRYNOX	TANGDRYNOX	ESP, HS
OPPODRYNOX	OPPODRYNOX	BAGHOUSE
OPPOWETNONOX	TANGDRYNONOX	ESP, HS
OPPOWETNONOX	OPPOWETNONOX	ESP, CS
OPPOWETNOX	TANGDRYNONOX	ESP, HS
OPPOWETNOX	OPPOWETNONOX	BAGHOUSE
REARDRYNONOX	FRONTDRYNONOX	ESP, CS
REARDRYNONOX	TANGDRYNONOX	ESP, HS
STOKDRYNONOX	CYCLWETNONOX	ESP, CS
TANGDRYNONOX	TANGDRYNONOX	ESP, HS
TANGDRYNONOX	TANGDRYNONOX	BAGHOUSE
TANGDRYNOX	TANGDRYNOX	ESP, HS
TANGDRYNOX	TANGDRYNOX	BAGHOUSE
TANGWETNONOX	CYCLWETNONOX	ESP, CS
TANGWETNOX	TANGDRYNONOX	ESP, HS
TANGWETNOX	OPPOWETNONOX	BAGHOUSE
UNKNOWN	TANGDRYNONOX	ESP, CS
VERTDRYNONOX	VERTDRYNOX	BAGHOUSE
VERTWETNONOX	CYCLWETNONOX	ESP, CS
G-CYCLWETNONOX	G-FRONTDRYNONOX	NO CONTROL
G-CYCLWETNOX	G-FRONTDRYNONOX	NO CONTROL
G-FRONTDRYNONOX	G-FRONTDRYNONOX	NO CONTROL
G-FRONTDRYNOX	G-FRONTDRYNONOX	NO CONTROL
G-FRONTWETNONOX	G-FRONTDRYNONOX	NO CONTROL
G-HORZDRYNONOX	G-FRONTDRYNONOX	NO CONTROL
G-OPPODRYNONOX	G-FRONTDRYNONOX	NO CONTROL

(continued)

Table D-1. (Continued)

Facility Boiler	Boiler Used in EFP	Associated Control Device
G-OPPODRYNOX	G-FRONTDRYNONOX	NO CONTROL
G-REARDRYNONOX	G-FRONTDRYNONOX	NO CONTROL
G-REARDRYNOX	G-FRONTDRYNONOX	NO CONTROL
G-REARDRYNOX	G-FRONTDRYNONOX	NO CONTROL
G-TANGDRYNONOX	G-FRONTDRYNONOX	NO CONTROL
G-TANGDRYNOX	G-FRONTDRYNONOX	NO CONTROL
G-TANGWETNONOX	G-FRONTDRYNONOX	NO CONTROL
G-TANGWETNONOX	G-FRONTDRYNONOX	NO CONTROL
G-UNKNOWN	G-FRONTDRYNONOX	NO CONTROL
G-UNKNOWNDRYNONOX	G-FRONTDRYNONOX	NO CONTROL
G-VERTDRYNONOX	G-FRONTDRYNONOX	NO CONTROL
G-VERTWETNONOX	G-FRONTDRYNONOX	NO CONTROL
O-CYCLDRYNONOX	O-FRONTDRYNONOX	NO CONTROL
O-CYCLEDRYNOX	O-FRONTDRYNONOX	NO CONTROL
O-CYCLWETNONOX	O-FRONTDRYNONOX	O-ESP, CS
O-CYCLWETNOX	O-FRONTDRYNONOX	NO CONTROL
O-FRONTDRYNONOX	O-FRONTDRYNONOX	NO CONTROL
O-FRONTDRYNOX	O-FRONTDRYNOX	NO CONTROL
O-FRONTWETNONOX	O-FRONTDRYNONOX	NO CONTROL
O-FRONTWETNOX	O-FRONTDRYNOX	O-ESP, CS
O-OPPODRYNONOX	O-OPPODRYNONOX	NO CONTROL
O-OPPODRYNOX	O-FRONTDRYNOX	NO CONTROL
O-OPPOWETNONOX	O-FRONTDRYNONOX	NO CONTROL
O-REARDRYNONOX	O-FRONTDRYNONOX	NO CONTROL
O-REARDRYNOX	O-FRONTDRYNOX	O-ESP, CS
O-TANGDRYNONOX	O-TANGDRYNONOX	NO CONTROL
O-TANGDRYNOX	O-TANGDRYNOX	NO CONTROL
O-TANGWETNONOX	O-FRONTDRYNONOX	NO CONTROL
O-UNKNOWNDRYNONOX	O-FRONTDRYNONOX	NO CONTROL
O-VERTDRYNONOX	O-FRONTDRYNONOX	O-ESP, CS

Notes to Table D-1: The following conventions are used for naming boilers in this table. The name describes primary fuel used, firing type, bottom type, and presence or absence of nitrogen oxides control.

Fuel: No prefix = coal
G- = gas
O- = oil
CFB boilers are coal fired

Boiler firing type:

CFB	Circulating fluidized bed
COMCYC	Combined cycle
CYC	Cyclone
FRONT	Front
HORZ	Horizontal
OPPO	Opposed
REAR	Rear
TANG	Tangential
UNKNOWN	Firing type not specified in Utility Data Institute database (UDI)
VERT	Vertical

Boiler bottom type:

DRY	Dry ash
WET	Molten ash

Nitrogen oxides control:

NONOX	No nitrogen oxides control
NOX	Nitrogen oxides control by any of several means as specified in the UDI database

Control device:

BAGHOUSE	Fabric filter
ESP, CS	Cold-side electrostatic precipitator (after the air preheater)
ESP, HS	Hot-side electrostatic precipitator (before the air preheater)
O-ESP, CS	Cold-side electrstatic precipitator applied to an oil-fired boiler
NO CONTROL	Particulate controls not applied to this boiler (usually gas or oil fired)

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.

Because only 1990 estimated emissions were used for risk analysis, missing stack data or latitude/longitude data for 1994 were not addressed.

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] Two exceptions were bituminous coals from Illinois and Colorado, for which analyses were on an as-shipped basis representative of the coal to be fired. Tables at the end of this appendix (D-8 and D-9) list trace element concentrations in fuel and coal cleaning factors, respectively, as used in the EFP.

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 five, one, and two utility units, respectively. 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] In summary, USGS data were used for all States with the following exceptions: two sets of USGS data for Arizona coals were averaged with USGS data for Colorado, Utah, and New Mexico coals; Texas lignite data were substituted for Louisiana coal; Arco data were used for Wyoming coals and for Colorado bituminous coals; and ISGS data were used for Illinois coals.

For a unit that burned bituminous coal, the kilogram/year (kg/yr) feed rate of trace elements to the boiler was determined from the average trace element concentration in the coal, a coal cleaning factor, and the annual fuel consumption rate. No coal cleaning factors were applied to lignite and subbituminous coals (see Equations 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.12). 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 kg/yr 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 for gas-fired units analyzed organic HAPs, and a geometric mean emission rate of each observed organic HAP was used. This rate in kilogram HAP/10^9 cubic feet was then multiplied by the unit’s gas consumption to obtain a kilogram HAP/year stack emission rate of each specific HAP (see Equation 4 in Table D-1). 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.

Total quantities burned for each type of fuel (coal, gas, or oil) and each type of boiler (as shown in Table D-1) are shown in Table D-3. Coal consumption is further quantified by coal rank.

D.8 HYDROGEN CHLORIDE AND HYDROGEN FLUORIDE CONCENTRATION IN FUEL

To obtain hydrogen chloride (HCl) or hydrogen fluoride (HF) emissions from the boiler, emission factors were derived by performing mass balances for chloride and fluoride, then converting these balances to the equivalent levels of HCl or HF throughout the boiler system.[16] For example, for each lb/hr of chloride in the feed coal at

Table D-2. Computer Calculations (Inorganic HAPs)

Feed rate of trace elements to the boiler

For bituminous coal:

Equation 1

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

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

For lignite and subbituminous coal:

Equation 2

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

For oil:

Equation 3

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

For natural gas:

Equation 4

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

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

Table D-3. Fuel Consumption by Type of Boiler

Facility Boiler	Fuel Consumed[a]	Primary Fuel
CFBDRYNONOX	1,157	Coal
CFBDRYNOX	1885	Coal
COMCYCLNONOX	166,767	Gas
CYCLWETNONOX	56,723	Coal
CYCLWETNOX	3,098	Coal
FRONTDRYNONOX	5,713	Coal
FRONTDRYNOX	59,686	Coal
FRONTWETNONOX	10,045	Coal
OPPODRYNONOX	121,708	Coal
OPPODRYNOX	138,237	Coal
OPPOWETNONOX	6,217	Coal
OPPOWETNOX	29,590	Coal
REARDRYNONOX	733	Coal
STOKDRYNONOX	731	Coal
TANGDRYNONOX	217,266	Coal
TANGDRYNOX	138,085	Coal
TANGWETNONOX	1,546	Coal
TANGWETNOX	16,606	Coal
UNKNOWN	93	Coal
VERTDRYNONOX	5,008	Coal
VERTWETNONOX	792	Coal
G-CYCLWETNONOX	10,245	Gas
G-CYCLWETNOX	789	Gas
G-FRONTDRYNONOX	278,662	Gas
G-FRONTDRYNOX	269,047	Gas
G-FRONTWETNONOX	4,262	Gas
G-HORZDRYNONOX	190	Gas
G-OPPODRYNONOX	409,885	Gas
G-OPPODRYNOX	662,119	Gas
G-REARDRYNONOX	758	Gas
G-REARDRYNOX	29,747	Gas
G-TANGDRYNONOX	416,979	Gas
G-TANGDRYNOX	221,779	Gas
G-TANGWETNONOX	6,287	Gas
G-UNKNOWNDRYNONOX	1,337	Gas
G-VERTDRYNONOX	139	Gas
O-CYCLDRYNONOX	898	Oil
O-CYCLWETNONOX	228	Oil
O-CYCLWETNOX	792	Oil
O-FRONTDRYNONOX	2,772	Oil
O-FRONTDRYNOX	35,929	Oil
O-FRONTWETNONOX	182	Oil

(continued)

Table D-3. (Continued)

Facility Boiler	Fuel Consumed[a]	Primary Fuel
O-FRONTWETNOX	134	Oil
O-OPPODRYNONOX	7,967	Oil
O-OPPODRYNOX	6,589	Oil
O-OPPOWETNONOX	766	Oil
O-REARDRYNONOX	1,407	Oil
O-REARDRYNOX	1,287	Oil
O-TANGDRYNONOX	32,654	Oil
O-TANGDRYNOX	7,304	Oil
O-TANGWETNONOX	9,226	Oil

[a] Coal in thousands of tons per year, gas in millions of cubic feet per year, oil in thousands of barrels per year. Quantities do not include gas and oil used as starting or temporary fuels in boilers that normally burn other fuels as the primary fuel.

Note: Nationwide total fuel consumption from these boilers is, in the units given above:

Coal		815,135	(18.0×10^{15} Btu)
	Bituminous	405,013	(10.3×10^{15} Btu)
	Subbituminous	330,978	(6.6×10^{15} Btu)
	Lignite	79,128	(1.1×10^{15} Btu)
Gas		2,708,342	(2.84×10^{15} Btu)
Oil		146,148	(0.92×10^{15} Btu)

one of the test sites, 0.63 lb/hr of HCl was found in the gas stream leaving the boiler. Similarly for HF, the boiler emissions were 0.64 lb/hr for each lb/hr of fluoride in the coal. For ease of programming, the HCl and HF emissions were addressed starting in the fuel. This programming was done by multiplying the chloride and fluoride concentrations in the fuel constituents by 0.63 or 0.64, respectively. The resulting numbers allowed direct conversion into boiler emissions that could be further modified for systems with PM control or SO_2 control. For the 1990 emission estimates, before obtaining further test reports, the factors were 0.61 for HCl and 0.56 for HF.

The chloride concentrations were not available for coals from the following States: Alaska, Illinois, Indiana, Iowa, Missouri, Utah, and Washington. Chloride concentrations were assigned, as shown in Table D-4, for coals originating from these States.[17]

D.9 EMISSION MODIFICATION FACTORS FOR INORGANIC HAPS

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.

To address the partitioning of the HAP stream through the combustion and pollution control process, partitioning factors

Table D-4. 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 0.63 =	34.0
Illinois	1,136 x 0.63 =	715.7
Indiana	1,033 x 0.63 =	650.8
Iowa	1,498 x 0.63 =	943.7
Missouri	1,701 x 0.63 =	1,071.6
Utah	220 x 0.63 =	138.6
Washington	104 x 0.63 =	65.5

(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 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 generally segregated into only coal-, oil-, or gas-fired bins.

The EFP itself uses EMFs to partition the emissions as they proceed from the fuel through the unit to the stack exit as follows. The average concentrations of metallic HAPs in an individual fuel by State (based on USGS data) were multiplied by the amount of fuel that the unit burned in 1990 and 1994. 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 occurs 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.

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

D.10 ACID GAS HAPS

The method used with HCl or HF emissions allowed direct conversion from coal chlorine or fluorine content into boiler emissions, as described in section D.8, that could be further modified for systems with PM control or SO_2 control.

Hydrochloric acid and HF EMFs for PM and SO_2 control devices were developed with data from test reports in which contractors conducted tests individually for HCl, chlorine, HF, and fluorine before and after each control device. These tests were in contrast to the remaining tests for which HCl and HF values were estimated or omitted rather than measured.

The next steps after obtaining amounts of HCl or HF leaving the boiler were to construct EMFs for the PM control device, then for the SO_2 control device. Using chlorine as an example, the measured amount of HCl entering the PM control device (in kg/yr with suitable conversion factors) was compared with the measured amount of HCl leaving the PM control device. Using these two quantities, an EMF was formed as described in section D.9.

In the final step, EMFs were formed for HCl and HF through the SO_2 control device based on the measured mass of HCl or HF entering that device (leaving the PM control device) and the mass measured at the exit of the SO_2 control device. However, a modification was required to account for flue-gas bypass around the SO_2 control device. A portion of the flue gas is bypassed to maintain SO_2 removal at the minimum permitted amount. This action is used as a means of reducing energy required to reheat the flue gas for effective plume rise from the stack. In developing the HCl and HF EMFs for wet FGDs and dry scrubbers, the effect of flue gas bypass was treated by analyzing utility test data from the four plants (of eight tested) that used bypasses, reviewing municipal waste incinerator results that showed a typical HCl or HF removal efficiency of 95 percent, and having discussions with industry representatives. Based on the 95 percent removal efficiency coupled with the measured values for quantity of flue gas bypassed, an industry average effective value for flue gas bypass in 1994 was estimated. The value was assumed to be 15 percent (17 percent for 1990 data) for wet FGDs and 14 percent (for 1990 and 1994 data) for dry scrubber systems. These assumptions were used only in the development of HCl and HF EMFs.[19] Future wet FGDs are not expected to use flue-gas bypass in normal operation.

D.11 ORGANIC HAPS

Because organic HAPs were not always tested at the entrance and exit of each control device in the emissions testing, all organic HAP emissions were addressed by examining the test data and determining the concentration of a particular HAP exiting the stack. Organic HAP concentrations were obtained from emission test reports. Table D-5 gives the equations used to estimate organic HAP emissions from coal-, oil-, and gas-fired boilers.

Organic stack emissions from coal-fired boilers were first determined on an emission factor basis (lb/trillion Btu) to account for different coal heating values, then converted to a rate basis (kg/yr of individual HAP). This procedure was necessary because different coal ranks had different heating values. For example, it would require burning more lignite to achieve the same heat input to the boiler as burning bituminous coal. These values were determined as averages for each type of coal (see Table D-6).[20]

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. Where nondetected data were used (in about 40 percent of the individual congener test series), the same procedure as for EMFs (described below) was followed to establish a calculated mean for the (usually) three test values. For each individual organic HAP observed in testing, a median concentration was obtained to represent the average value of the usually small and scattered data set. For example, of the coal-fired boilers tested for dioxins/furans, 12 had detected values for one or more of the congeners. 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.

Table D-5. Computer Calculations (Organic HAPs)

All non-trace-element HAPs emitted from the stack

Coal-fired boiler:

Equation 1

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

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

Oil-fired boiler:

Equation 2

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

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

Gas-fired boiler:

Equation 3

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

Table D-6. Average Higher Heating Values of Coal[21]

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.

D.12 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.13 MODEL CHANGES FOR ESTIMATES IN THE YEAR 2010

Emission estimates for 2010 were derived from the same basic 1990 model described above. However, changes to input files were made to accommodate expected changes in fuel usage, 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).[22] 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-7, that corresponded to the State of coal origin assigned to each unit.[23]

Tables D-8a,b,c and D-9 list trace element concentrations in fuel and coal cleaning factors, respectively, as used in the EFP.

Table D-7. 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.

Table D-8a. Trace Element Concentrations in Coal

State	Coal type	Compound	Concentration, ppmw
AK	Subbituminous	ANTIMONY	1.90
		ARSENIC	3.00
		BERYLLIUM	0.50
		CADMIUM	0.15
		CHROMIUM	20.00
		COBALT	5.00
		CHLORINE	53.93
		FLUORINE	95.00
		LEAD	5.40
		MANGANESE	88.00
		MERCURY	0.07
		NICKEL	10.00
		SELENIUM	1.60
AL	Bituminous	ANTIMONY	1.82
		ARSENIC	53.00
		BERYLLIUM	1.88
		CADMIUM	0.06
		CHROMIUM	22.80
		COBALT	8.20
		CHLORINE	380.00
		FLUORINE	127.00
		LEAD	7.00
		MANGANESE	41.00
		MERCURY	0.19
		NICKEL	17.50
		SELENIUM	1.88
AR	Lignite	ANTIMONY	1.17
		ARSENIC	4.30
		BERYLLIUM	2.40
		CADMIUM	0.29
		CHROMIUM	16.90
		COBALT	6.00
		CHLORINE	142.00
		FLUORINE	63.00
		LEAD	9.80
		MANGANESE	119.00
		MERCURY	0.25
		NICKEL	11.80
		SELENIUM	5.00

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
AZ	Subbituminous	ANTIMONY	0.47
		ARSENIC	2.10
		BERYLLIUM	1.10
		CADMIUM	0.10
		CHROMIUM	4.60
		COBALT	2.10
		CHLORINE	200.00
		FLUORINE	79.00
		LEAD	9.00
		MANGANESE	27.00
		MERCURY	0.07
		NICKEL	4.80
		SELENIUM	1.50
CO	Bituminous	ANTIMONY	0.91
		ARSENIC	1.34
		BERYLLIUM	0.36
		CADMIUM	0.18
		CHROMIUM	1.89
		COBALT	1.03
		CHLORINE	92.97
		FLUORINE	98.78
		LEAD	5.44
		MANGANESE	10.83
		MERCURY	0.07
		NICKEL	1.25
		SELENIUM	0.87
CO	Subbituminous	ANTIMONY	0.35
		ARSENIC	1.03
		BERYLLIUM	0.84
		CADMIUM	0.08
		CHROMIUM	4.10
		COBALT	1.60
		CHLORINE	118.00
		FLUORINE	99.00
		LEAD	3.50
		MANGANESE	32.00
		MERCURY	0.14
		NICKEL	7.90
		SELENIUM	0.89

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
IA	Bituminous	ANTIMONY	2.30
		ARSENIC	12.00
		BERYLLIUM	1.88
		CADMIUM	14.00
		CHROMIUM	12.10
		COBALT	10.00
		CHLORINE	1498.36
		FLUORINE	77.00
		LEAD	68.00
		MANGANESE	259.00
		MERCURY	0.19
		NICKEL	31.00
		SELENIUM	3.60
IL	Bituminous	ANTIMONY	0.82
		ARSENIC	6.78
		BERYLLIUM	1.31
		CADMIUM	0.98
		CHROMIUM	12.66
		COBALT	3.19
		CHLORINE	1136.07
		FLUORINE	84.14
		LEAD	24.51
		MANGANESE	33.74
		MERCURY	0.08
		NICKEL	12.74
		SELENIUM	1.72
IN	Bituminous	ANTIMONY	1.40
		ARSENIC	10.10
		BERYLLIUM	2.82
		CADMIUM	0.49
		CHROMIUM	15.40
		COBALT	5.20
		CHLORINE	1032.79
		FLUORINE	65.00
		LEAD	10.90
		MANGANESE	38.00
		MERCURY	0.11
		NICKEL	17.90
		SELENIUM	2.17

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
KS	Bituminous	ANTIMONY	0.85
		ARSENIC	25.00
		BERYLLIUM	1.47
		CADMIUM	10.00
		CHROMIUM	10.10
		COBALT	15.00
		CHLORINE	2500.00
		FLUORINE	64.00
		LEAD	111.00
		MANGANESE	160.00
		MERCURY	0.19
		NICKEL	41.00
		SELENIUM	2.70
KY	Bituminous	ANTIMONY	1.13
		ARSENIC	19.10
		BERYLLIUM	3.17
		CADMIUM	0.16
		CHROMIUM	16.30
		COBALT	6.60
		CHLORINE	1139.00
		FLUORINE	86.00
		LEAD	10.60
		MANGANESE	32.00
		MERCURY	0.15
		NICKEL	17.50
		SELENIUM	3.83
LA	Lignite	ANTIMONY	0.82
		ARSENIC	3.70
		BERYLLIUM	1.90
		CADMIUM	0.15
		CHROMIUM	11.40
		COBALT	3.30
		CHLORINE	115.00
		FLUORINE	83.00
		LEAD	5.50
		MANGANESE	141.00
		MERCURY	0.19
		NICKEL	7.80
		SELENIUM	6.00

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
MD	Bituminous	ANTIMONY	0.81
		ARSENIC	26.00
		BERYLLIUM	2.01
		CADMIUM	0.14
		CHROMIUM	26.70
		COBALT	11.00
		CHLORINE	914.00
		FLUORINE	107.00
		LEAD	10.00
		MANGANESE	13.00
		MERCURY	0.42
		NICKEL	22.00
		SELENIUM	3.80
MO	Bituminous	ANTIMONY	1.60
		ARSENIC	10.00
		BERYLLIUM	2.01
		CADMIUM	0.80
		CHROMIUM	12.20
		COBALT	6.70
		CHLORINE	1701.64
		FLUORINE	60.00
		LEAD	67.00
		MANGANESE	99.00
		MERCURY	0.17
		NICKEL	23.00
		SELENIUM	4.20
MT	Bituminous	ANTIMONY	0.69
		ARSENIC	7.00
		BERYLLIUM	0.52
		CADMIUM	0.08
		CHROMIUM	3.10
		COBALT	1.50
		CHLORINE	80.00
		FLUORINE	104.00
		LEAD	3.00
		MANGANESE	37.00
		MERCURY	0.09
		NICKEL	3.90
		SELENIUM	0.70

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
MT	Lignite	ANTIMONY	0.92
		ARSENIC	18.00
		BERYLLIUM	1.04
		CADMIUM	0.11
		CHROMIUM	0.94
		COBALT	0.80
		CHLORINE	67.00
		FLUORINE	159.00
		LEAD	4.80
		MANGANESE	68.00
		MERCURY	0.12
		NICKEL	4.00
		SELENIUM	0.72
MT	Subbituminous	ANTIMONY	0.69
		ARSENIC	7.00
		BERYLLIUM	0.52
		CADMIUM	0.08
		CHROMIUM	3.10
		COBALT	1.50
		CHLORINE	80.00
		FLUORINE	104.00
		LEAD	3.00
		MANGANESE	37.00
		MERCURY	0.09
		NICKEL	3.90
		SELENIUM	0.70
ND	Lignite	ANTIMONY	0.58
		ARSENIC	8.40
		BERYLLIUM	0.82
		CADMIUM	0.11
		CHROMIUM	7.00
		COBALT	2.70
		CHLORINE	110.00
		FLUORINE	34.00
		LEAD	3.73
		MANGANESE	86.00
		MERCURY	0.13
		NICKEL	4.10
		SELENIUM	0.79

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
NM	Subbituminous	ANTIMONY	1.07
		ARSENIC	1.80
		BERYLLIUM	2.70
		CADMIUM	0.16
		CHROMIUM	6.00
		COBALT	2.65
		CHLORINE	95.00
		FLUORINE	87.00
		LEAD	31.00
		MANGANESE	45.00
		MERCURY	0.06
		NICKEL	4.60
		SELENIUM	1.94
OH	Bituminous	ANTIMONY	0.81
		ARSENIC	23.20
		BERYLLIUM	2.39
		CADMIUM	0.12
		CHROMIUM	14.30
		COBALT	0.90
		CHLORINE	719.00
		FLUORINE	92.00
		LEAD	7.30
		MANGANESE	28.30
		MERCURY	0.22
		NICKEL	14.90
		SELENIUM	3.80
OK	Bituminous	ANTIMONY	0.69
		ARSENIC	24.00
		BERYLLIUM	0.86
		CADMIUM	0.10
		CHROMIUM	15.00
		COBALT	6.20
		CHLORINE	267.00
		FLUORINE	77.00
		LEAD	10.00
		MANGANESE	74.00
		MERCURY	0.17
		NICKEL	17.00
		SELENIUM	1.80

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
PA	Bituminous	ANTIMONY	1.23
		ARSENIC	32.10
		BERYLLIUM	2.45
		CADMIUM	0.10
		CHROMIUM	20.10
		COBALT	7.90
		CHLORINE	1096.00
		FLUORINE	78.00
		LEAD	10.80
		MANGANESE	23.50
		MERCURY	0.29
		NICKEL	20.40
		SELENIUM	3.55
TX	Lignite	ANTIMONY	0.82
		ARSENIC	3.70
		BERYLLIUM	1.90
		CADMIUM	0.15
		CHROMIUM	11.40
		COBALT	3.30
		CHLORINE	115.00
		FLUORINE	83.00
		LEAD	5.50
		MANGANESE	141.00
		MERCURY	0.19
		NICKEL	7.80
		SELENIUM	6.00
UT	Bituminous	ANTIMONY	0.23
		ARSENIC	0.89
		BERYLLIUM	0.61
		CADMIUM	0.08
		CHROMIUM	7.70
		COBALT	2.70
		CHLORINE	219.67
		FLUORINE	57.00
		LEAD	3.90
		MANGANESE	8.00
		MERCURY	0.04
		NICKEL	4.10
		SELENIUM	2.00

(continued)

Table D-8a. (Continued)

State	Coal type	Compound	Concentration, ppmw
VA	Bituminous	ANTIMONY	0.93
		ARSENIC	11.00
		BERYLLIUM	1.66
		CADMIUM	0.05
		CHROMIUM	12.50
		COBALT	6.30
		CHLORINE	930.00
		FLUORINE	74.00
		LEAD	5.80
		MANGANESE	19.00
		MERCURY	0.14
		NICKEL	11.20
		SELENIUM	2.70
WA	Subbituminous	ANTIMONY	0.30
		ARSENIC	1.50
		BERYLLIUM	1.10
		CADMIUM	0.11
		CHROMIUM	0.70
		COBALT	4.70
		CHLORINE	103.28
		FLUORINE	14.00
		LEAD	2.80
		MANGANESE	41.00
		MERCURY	0.06
		NICKEL	7.90
		SELENIUM	0.40
WV	Bituminous	ANTIMONY	0.93
		ARSENIC	10.60
		BERYLLIUM	2.78
		CADMIUM	0.10
		CHROMIUM	15.30
		COBALT	7.20
		CHLORINE	1216.00
		FLUORINE	58.00
		LEAD	7.20
		MANGANESE	19.10
		MERCURY	0.16
		NICKEL	14.20
		SELENIUM	3.97

(continued)

State	Coal type	Compound	Concentration, ppmw
WY	Subbituminous	ANTIMONY	0.73
		ARSENIC	0.69
		BERYLLIUM	0.18
		CADMIUM	0.13
		CHROMIUM	2.82
		COBALT	0.87
		CHLORINE	118.30
		FLUORINE	43.70
		LEAD	2.07
		MANGANESE	5.65
		MERCURY	0.08
		NICKEL	2.17
		SELENIUM	0.51

Table D-8b. Trace Element Concentrations in Fuel Oil (for 1994 estimates)

Trace Element	Concentration in Oil, ppmw
Arsenic	0.306
Beryllium	0.027
Cadmium	0.020
Chromium	0.31
Cobalt	1.63
Chlorine	131
Fluorine	17.5
Lead	1.41
Manganese	0.35
Mercury	0.0092
Nickel	26
Selenium	0.095

Table D-8c. Trace Element Concentrations in Gas

Trace Element	Concentration in gas mg/m^3
Arsenic	0.000963
Cobalt	0.100
Lead	0.100
Mercury	0.0000024
Nickel	0.0500

Table D-9. Coal Cleaning Factors for Bituminous Coals Used in the Emission Factors Program[a]

Constituent	Cleaning factor
Antimony	0.715
Arsenic	0.554
Beryllium	0.711
Cadmium	0.624
Chromium	0.512
Cobalt	0.537
Chlorine	0.496
Fluorine	0.496
Lead	0.449
Manganese	0.382
Mercury	0.790
Nickel	0.568
Selenium	0.745

[a] Applying the cleaning factors to United States Geographical Survey (USGS) constituent concentrations for bituminous coals from the States named below results in new, lower constituent concentrations (modified USGS concentrations), which are used in the emission factors program.

Note: States to which applied: AL, IA, IN, KS, KY, MD, MO, OH, OK, PA, UT, VA, WV

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Appendix E - Health Effects Summaries: Overview

Appendix E 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

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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).

risk assessment methodology to recalculate the estimated risk. They calculated a new unit risk of 1.43 x 10^{-3} (μg/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]

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 x 10^{-5} (µg/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

μg/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 μg/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.

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,

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

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 Chloride

E.6.2.1 Acute (Short-Term) Effects for Hydrogen Chloride. 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 Chloride. 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 Chloride. 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]

ln 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. The health effects of mercury and mercury compounds are summarized in chapter 7 of this Utility HAP Study Report and are discussed in greater length and detail in Volume V of the Mercury Study Report to Congress.[74]

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. Table E-1 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 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[79,80,81,82]

E.9.1.1 Cancer Effects for Nickel Refinery Dust. U.S. Environmental Protection Agency 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 U.S. Environmental Protection Agency used.

Table E-1. Physical Properties of Some Forms of Nickel

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]

U.S. Environmental Protection Agency used the additive and multiplicative extrapolation method, based on human data, to estimate the unit cancer risk for nickel refinery dust. U.S. Environmental Protection Agency calculated an inhalation unit risk estimate of 2.4 x 10^{-4} (μg/m$^{3)-1}$.[84] Based upon this unit risk estimate, U.S. Environmental Protection Agency estimates that if an individual were to breathe air containing nickel refinery dust at 0.004 µg/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, U.S. Environmental Protection Agency estimates that breathing air containing 0.04 µg/m^3 would result in an increased chance of an increased chance of up to one in one hundred thousand of developing cancer.[85]

U.S. Environmental Protection Agency 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. U.S. Environmental Protection Agency 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, U.S. Environmental Protection Agency also used human data to estimate the unit cancer risk. U.S. Environmental Protection Agency calculated an inhalation unit risk estimate of 4.8×10^{-4} $(\mu g/m^3)^{-1}$.[92] U.S. Environmental Protection Agency 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, U.S. Environmental Protection Agency 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. U.S. Environmental Protection Agency 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. U.S. Environmental Protection Agency 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 U.S. Environmental Protection Agency used.

U.S. Environmental Protection Agency 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

U.S. Environmental Protection Agency 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. U.S. Environmental Protection Agency has not established a Reference Concentration (RfC) for any nickel compound.[123]

U.S. Environmental Protection Agency 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. 1976 study was properly designed and provided adequate toxicological endpoints; however, high mortality occurred in the controls.[124] The database provided adequate supporting subchronic studies; this was the basis for U.S. Environmental Protection Agency'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 U.S. Environmental Protection Agency.[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 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 U.S. Environmental Protection Agency.

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 U.S. Environmental Protection Agency 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 3×10^{-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)$^{-1}$ (U.S. Environmental Protection Agency[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] U.S. Environmental Protection Agency 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. U.S. Environmental Protection Agency 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

F.1 INTRODUCTION

The model used to calculate direct inhalation risks from hazardous air pollutants (HAPs) emitted from utility boilers is the Human Exposure Model Version 1.5 (HEM 1.5). It was developed by the Pollutant Assessment Branch (PAB) of the U.S. Environmental Protection Agency's (EPA) Office of Air Quality Planning and Standards (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 1.5 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 1.5, 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 G*uideline on Air Quality Models*, the ISCLT is appropriate for modeling industrial source complexes in either rural or urban areas.[1] With this model, long-term ambient concentrations can be estimated for transport distances up to 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,

Table F-1. Summary of HEM 1.5 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

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 1.5 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*[2] 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 radius 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. Instead of the final plume rise option of the default selections, a transitional plume rise was used. 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*.[3]

F.3 HEM DATABASES

Four databases are contained in the HEM 1.5 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 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 1.5 chooses the STAR data set for each plant based on proximity of the plant to the location where the meteorological data were collected.[4]

The meteorological database used for the utility study contains data from hourly surface observations obtained from the OAQPS Technology Transfer Network (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 1.5 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 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. Of the 73 sites, 40 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 1.5 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 1.5 calculates exposure by integrating the HAP air concentration at

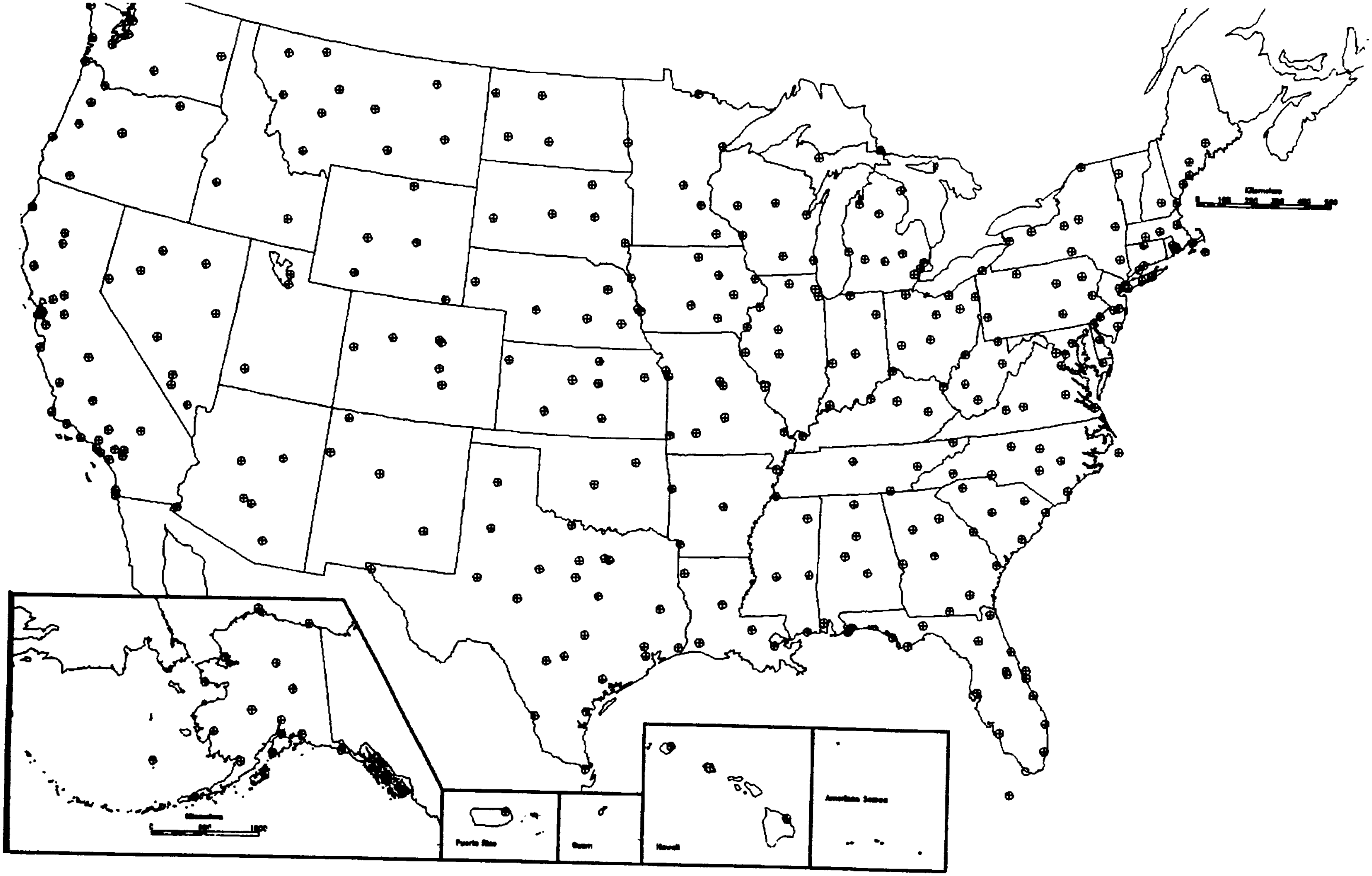

Figure F-1. **Location of STAR Data Sites** the population center (centroid) of the census block through interpolation of the air concentration values at the surrounding modeled points. All persons residing in the census block are treated as being exposed to the air concentration at the centroid.

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 1.5 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.[5]

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 Appendix E (*Health Effects Summaries: Overview*). Risk is calculated for the exposed population on a single-pollutant basis. For carcinogens, HEM 1.5 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.

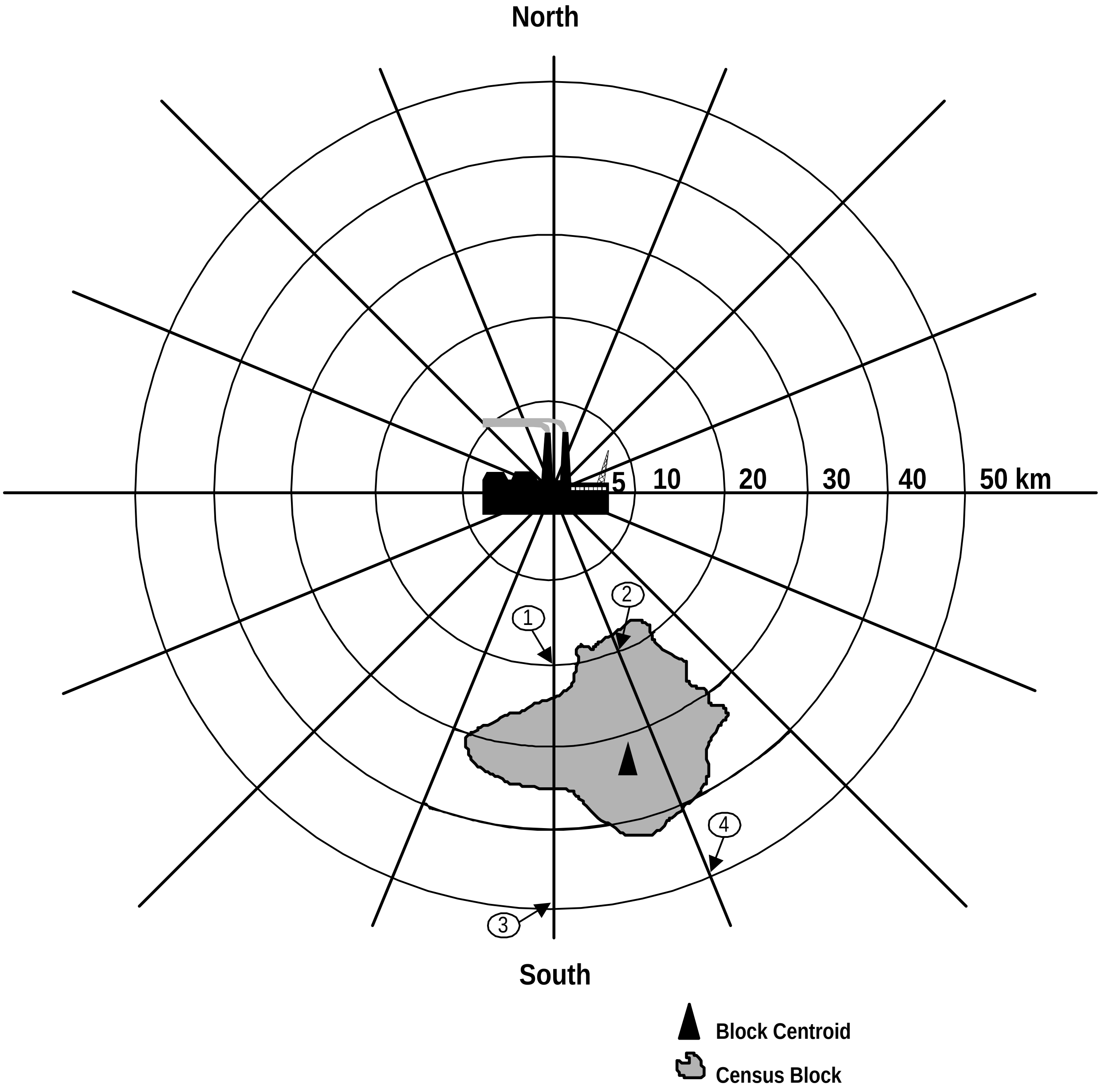

Figure F-2. The exposure algorithms interpolate between the estimated air concentrations and the population data. Air concentrations are calculated at the points where the circles and lines intersect. Population is known at the block centroid locations. The concentration at the centroid is calculated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

The utility boiler HEM modeling application requires the input of chemical-specific toxicity information. HEM 1.5 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 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 1.5 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 Appendix E (*Health Effects Summaries: Overview*).

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 1.5:

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 1.5 OUTPUT

For carcinogens, HEM 1.5 produces estimates of annual incidence (population risk), number of people exposed to various risk levels, and maximum individual lifetime risk. For noncarcinogens, HEM 1.5 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

1. U.S. Environmental Protection Agency. *Guideline on Air Quality Models (Revised)*. EPA-450/2-78-027R. Office of Air Quality Planning and Standards, Research Triangle Park, NC. 1993, revised 1995. pp. 4-4.

2. Ref. 1, pp. 8-10.

3. U.S. Environmental Protection Agency. *User's Guide for the Industrial Source Complex (ISC2) Dispersion Models, Volume 1 - User's Instructions*. EPA-450/4-92-008a. Office of Air Quality Planning and Standards, Research Triangle Park, NC. 1992.

4. Ref. 3, pp. 3-58 to 3-60

5. U.S. Environmental Protection Agency. *User's Manual for the Human Exposure Model (HEM)*. EPA-450/5-86-001. Office of Air Quality Planning and Standards, Research Triangle Park, NC. 1986. pp. 2-12 to 2-19.

Appendix G — Data Tables for Dioxin Multipathway Assessment

Appendix G-1

IEM Model Input Parameters
For Dioxin Screening Level
Multipathway Assessment
(Chapter 11)

Table G-1. Scenario Dependent Exposure/Risk Modeling Parameters

Parameter	Subsistence fisher	Subsistence farmer	Adult resident	Child resident	Source of data
Exposure duration, years	30	30	30	18	EPA, 1996c
Distance from source, meters	500	500	5000	5000	EPA, 1994a; EPA, 1997b
Air temperature, celsius	9.14	9.14	13.1	13.1	SAMSON, 1993
Runoff, cm/y	50	50	25	25	Gerhaghty, 1973
Wind speed, m/s	4.38	4.38	4.19	4.19	SAMSON, 1993
Reduction in meat conc. due to fattening prior to slaughter	1.0	N/A	N/A	N/A	EPA, 1994a
Body weight, kg	70	70	70	17	EPA, 1996c
Inhalation rate, m^3/day	20	20	20	16	EPA, 1996c
Plant consumption, g/kg body weight/day					
grains	1.87	1.87	1.87	3.77	EPA, 1996c
legumes	0.381	0.381	0.381	0.666	EPA, 1996c
potatoes	0.170	0.170	0.170	0.274	EPA, 1996c
root vegetables	0.024	0.024	0.024	0.036	EPA, 1996c
fruits	0.57	0.57	0.57	0.223	EPA, 1996c
fruiting vegetables	0.064	0.064	0.064	0.120	EPA, 1996c
leafy vegetables	0.0281	0.0281	0.0281	0.008	EPA, 1996c
Animal consumption, g/kg body weight/d					
beef	N/A	0.341	N/A	N/A	EPA, 1996c
dairy	N/A	0.599	N/A	N/A	EPA, 1996c
poultry	N/A	0.111	N/A	N/A	EPA, 1996c
eggs	N/A	0.073	N/A	N/A	EPA, 1996c
Soil ingestion, g/d	0.1	0.1	0.1	0.2	EPA, 1996c

Table G-1. (Continued)

Parameter	Subsistence fisher	Subsistence farmer	Adult resident	Child resident	Source of data
Water consumption, L/kg body weight/d	0.0285	0.0285	0.0285	0.02	EPA, 1996c
Fish ingestion, kg/kg body weight/d	0.000857	N/A	N/A	N/A	EPA, 1996c
Dermal soil contact time, hr/day	1.5	8.33	1.5	5.6	EPA, 1996a
Dermal area exposed to soil, cm^2	5000	5000	5000	850	EPA, 1996a; EPA, 1990
Dermal soil cont. amount, g/ cm^2	0.02	0.02	0.02	1.0	EPA, 1990
Fraction contaminated contact					
grains	0.667	1.0*	0.195	0.195	EPA, 1996c
legumes	0.8	1.0*	0.5	0.5	EPA, 1996c
potatoes	0.225	1.0*	0.031	0.031	EPA, 1996c
root vegetables	0.268	1.0*	0.073	0.073	EPA, 1996c
fruits	0.233	1.0*	0.076	0.076	EPA, 1996c
fruiting vegetables	0.623	1.0*	0.317	0.317	EPA, 1996c
leafy vegetables	N/A	1.0*	N/A	N/A	EPA, 1996c
beef	N/A	1.0*	N/A	N/A	EPA, 1996c
dairy	N/A	1.0*	N/A	N/A	EPA, 1996c
eggs	N/A	1.0*	N/A	N/A	EPA, 1996c
poultry	N/A	1.0*	N/A	N/A	EPA, 1996c

*Indicates a value not taken from the stated source, rather assumed appropriate for the scenario.

Table G-1a. Scenario Independent, Dioxin Congener Independent Parameters

Input parameter	Input value	Source of current value
Ideal gas constant, L-atm/mole-K	8.21e-02	EPA, 1996d
Air density, g/cm3	1.19e-03	EPA, 1994a
Solids density, kg/L, or g/cm3	2.65e+00	EPA, 1993
Drag coefficient	1.10e-03	EPA, 1993
Von Karman's coefficient	7.40e-01	EPA, 1993
Boundry thickness	4.00e+00	EPA, 1993
Interception fraction, unitless		
grains	0.00e+00	EPA, 1996d
legumes	8.00e-03	EPA, 1996d
potatoes	0.00e+00	EPA, 1996d
root vegetables	0.00e+00	EPA, 1996d
fruits	5.00e-02	EPA, 1996d
fruiting vegetables	5.00e-02	EPA, 1996d
leafy vegetables	1.50e-01	EPA, 1996d
forage	4.70e-01	EPA, 1996d
silage	4.40e-01	EPA, 1996d
Plant surface loss coefficient, yr-1	1.80e+01	EPA, 1993
Fraction wet deposition adhering	6.40e-01	EPA, 1994a
Length of plants' exposure, yrs		
grains	0.00e+00	EPA, 1996d
legumes	1.23e-01	EPA, 1996d
potatoes	0.00e+00	EPA, 1996d
root vegetables	0.00e+00	EPA, 1996d
fruits	1.23e-01	EPA, 1996d
fruiting vegetables	1.23e-01	EPA, 1996d
leafy vegetables	1.57e-01	EPA, 1996d
forage	1.23e-01	EPA, 1996d
silage	1.23e-01	EPA, 1996d
Yield of plants, kg dw/m2		

Table G-1a. (Continued)

Input parameter	Input value	Source of current value
grains	3.00e-01	EPA, 1996d
legumes	1.04e-01	EPA, 1996d
potatoes	4.80e-01	EPA, 1996d
root vegetables	3.34e-01	EPA, 1996d
fruits	1.07e-01	EPA, 1996d
fruiting vegetables	1.07e-01	EPA, 1996d
leafy vegetables	1.77e-01	EPA, 1996d
forage	3.10e-01	EPA, 1996d
silage	8.40e-01	EPA, 1996d
Surface Area Volume to Whole Plant Volume Correction		
grains	5.00e-01	EPA, 1994a
legumes	1.00e-02	EPA, 1994a
	2.50e-01	EPA, 1994a
root vegetables	2.50e-01	EPA, 1994a
fruits	1.00e-02	EPA, 1994a
fruiting vegetables	1.00e-02	EPA, 1994a
leafy vegetables	1.00e-02	EPA, 1994a
forage	5.00e-01	EPA, 1994a
silage	5.00e-01	EPA, 1994a
Feed (grain & silage) and grass (forage) diet fractions, unitless		
beef-grain & silage	4.80e-01	EPA, 1994a
beef-forage	4.80e-01	EPA, 1994a
dairy-grain & silage	9.00e-01	EPA, 1994a
dairy-forage	8.00e-02	EPA, 1994a
poultry & eggs-silage	8.50e-01	Lorber, 1997
poultry & eggs-forage	5.00e-02	Lorber, 1997
Soil diet fraction; unitless		
beef	4.00e-02	EPA, 1994a
dairy	2.00e-02	EPA, 1994a

Table G-1a. (Continued)

Input parameter	Input value	Source of current value
poultry & eggs	1.00e-01	Lorber, 1997
Bioavialibility of contaminant in soil, unitless	6.50e-01	EPA, 1994a
Total soil loss constant, yr-1	2.77e-02	Lorber, 1997
Fraction dermally absorbed, unitless	3.00e-02	EPA, 1994a
Cancer potency factor, kg-d/mg	1.56e+05	EPA, 1994a
Fish lipid content correction term	7.00e-02	EPA, 1994a
Fraction org. C in bottom sediment, unitless	3.00e-02	EPA, 1994a
Averaging time, days	2.56e+04	EPA, 1994a
Land area of fallout, km^2	3.73e+01	EPA, 1996d
Impervious surface area, km^2	3.73e+00	10% Land Area Fallout
Effective diameter of cont. area, m	2.00e+02	Lorber, 1997
Soil bulk density, g/cm^3	1.40e+00	EPA, 1996d
Watershed depth of incorporation, cm (no till)	2.00e+00	Brzuzy, 1995
Mixing depth for soil tillage, cm	2.00e+01	EPA, 1996d
Soil vol. water content, mL/cm^3	2.00e-01	EPA, 1993
Erosivity factor, kg/km^2-yr	1.60e+02	EPA, 1996d
Erodibility factor, t/ac	3.00e-01	EPA, 1996d
Topographic factor, unitless	2.00e-01	EPA, 1996d
Cover management factor, unitless	1.00e-01	EPA, 1996d
Support practice factor, unitless	1.00e+00	EPA, 1996d
Sediment delivery ratio to water body	2.00e-01	EPA, 1993 equation rounded to equal EPA, 1997d
Pollutant enrichment factor	2.00e+00	EPA, 1996d
Water body surface area, km^2	2.49e+00	EPA, 1996d
Water column volume, m^3	1.24e+07	EPA, 1996d
Annual volumetric flow rate, m^3/yr	1.00e+07	EPA, 1993
Average current velocity, m/sec	7.50e-01	EPA, 1993
Average water temperature, (°C)	1.50e+01	EPA, 1993
Suspended solids concentration, mg/L	5.00e+00	EPA, 1993
Benthic sediment concentration, kg/L	1.00e+00	EPA, 1996d

Table G-1a. (Continued)

Input parameter	Input value	Source of current value
Upper benthic sediment depth, m	2.00e-02	EPA, 1993
Lifetime, years	7.00e+01	EPA, 1994a
Exposure time, hours/day	1.80e+01	EPA, 1994a
Exposure frequency, days/yr	3.65e+02	Max conservative estimate

Table G-1b. Scenario Independent, Dioxin Congener Dependent Parameters

Input parameter	CDDs 2378	12378	123478	123789	123678	1234678	Octa	CDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Source
Toxicity equivalency factor	1	0.5	0.1	0.1	0.1	0.01	0	0.1	0.5	0.05	0.1	0.1	0.1	0.1	0.01	0.01	0	EPA, 1994a
Molecular weight, g/mole	3.22 e+02	3.56 e+02	3.91 e+02	3.91 e+02	3.91 e+02	4.25 e+02	4.61 e+02	3.06 e+02	3.40 e+02	3.40 e+02	3.75 e+02	3.75 e+02	3.75 e+02	3.75 e+02	4.09 e+02	4.09 e+02	4.45 e+02	EPA, 1997b
Henry's constant, atm-m3/mole	1.60 e-05	2.60 e-06	1.20 e-05	1.20 e-05	1.20 e-05	7.50 e-06	7.00 e-09	8.60 e-06	6.20 e-06	6.20 e-06	1.40 e-05	6.10 e-06	1.00 e-05	1.00 e-05	5.30 e-05	5.30 e-05	1.90 e-06	EPA, 1994a
Soil water part. coeff., ml/g (L/kg)	2.70 e+04	2.70 e+04	3.80 e+05	1.20 e+05	1.20 e+05	9.80 e+05	2.40 e+05	2.10 e+04	5.10 e+04	3.80 e+04	1.20 e+05	1.20 e+05	1.20 e+05	1.20 e+07	4.90 e+05	4.90 e+05	3.90 e+06	EPA, 1994a
Susp. sed-water part. coeff., L/kg	1.35 e+05	1.35 e+05	1.90 e+06	6.00 e+05	6.00 e+05	4.90 e+06	1.20 e+06	1.05 e+05	2.55 e+05	1.90 e+05	6.00 e+05	6.00 e+05	6.00 e+05	6.00 e+07	2.45 e+06	2.45 e+06	1.95 e+07	EPA, 1994a
Benthic sed-water part. coeff., L/kg	8.10 e+04	8.10 e+04	1.14 e+06	3.60 e+05	3.60 e+05	2.94 e+06	7.20 e+05	6.30 e+04	1.53 e+05	1.14 e+05	3.60 e+05	3.60 e+05	3.60 e+05	3.60 e+07	1.47 e+06	1.47 e+06	1.17 e+07	EPA, 1994a
Plant-soil bioconcentration factor																		
grains	5.60 e-03	5.60 e-03	1.21 e-03	1.21 e-03	2.33 e-03	7.01 e-04	1.59 e-03	6.50 e-03	3.88 e-03	4.59 e-03	2.33 e-03	2.33 e-03	2.33 e-03	2.33 e-03	1.05 e-03	1.05 e-03	3.1 8e-04	EPA, 1993
legumes	5.60 e-03	5.60 e-03	1.21 e-03	1.21 e-03	2.33 e-03	7.01 e-04	1.59 e-03	6.50 e-03	3.88 e-03	4.59 e-03	2.33 e-03	2.33 e-03	2.33 e-03	2.33 e-03	1.05 e-03	1.05 e-03	3.18 e-04	EPA, 1993
potatoes	3.92 e+03	3.92 e+03	3.02 e+04	3.02 e+04	1.26 e+04	6.27 e+04	2.11 e+04	3.23 e+03	6.42 e+03	5.13 e+03	1.26 e+04	1.26 e+04	1.26 e+04	1.26 e+04	3.64 e+04	3.64 e+04	1.80 e+05	EPA, 1993
root vegetables	3.92 e+03	3.92 e+03	3.02 e+04	3.02 e+04	1.26 e+04	6.27 e+04	2.11 e+04	3.23 e+03	6.42 e+03	5.13 e+03	1.26 e+04	1.26 e+04	1.26 e+04	1.26 e+04	3.64 e+04	3.64 e+04	1.80 e+05	EPA, 1993
fruits	5.60 e-03	5.60 e-03	1.21 e-03	1.21 e-03	2.33 e-03	7.01 e-04	1.59 e-03	6.50 e-03	3.88 e-03	4.59 e-03	2.33 e-03	2.33 e-03	2.33 e-03	2.33 e-03	1.05 e-03	1.05 e-03	3.18 e-04	EPA, 1993
fruiting vegetables	5.60 e-03	5.60 e-03	1.21 e-03	1.21 e-03	2.33 e-03	7.01 e-04	1.59 e-03	6.50 e-03	3.88 e-03	4.59 e-03	2.33 e-03	2.33 e-03	2.33 e-03	2.33 e-03	1.05 e-03	1.05 e-03	3.18 e-04	EPA, 1993
leafy vegetables	5.60 e-03	5.60 e-03	1.21 e-03	1.21 e-03	2.33 e-03	7.01 e-04	1.59 e-03	6.50 e-03	3.88 e-03	4.59 e-03	2.33 e-03	2.33 e-03	2.33 e-03	2.33 e-03	1.05 e-03	1.05 e-03	3.18 e-04	EPA, 1993
forage	5.60 e-03	5.60 e-03	1.21 e-03	1.21 e-03	2.33 e-03	7.01 e-04	1.59 e-03	6.50 e-03	3.88 e-03	4.59 e-03	2.33 e-03	2.33 e-03	2.33 e-03	2.33 e-03	1.05 e-03	1.05 e-03	3.18e-0 4	EPA, 1993
silage	5.60 e-03	5.60 e-03	1.21 e-03	1.21 e-03	2.33 e-03	7.01 e-04	1.59 e-03	6.50 e-03	3.88 e-03	4.59 e-03	2.33 e-03	2.33 e-03	2.33 e-03	2.33 e-03	1.05 e-03	1.05 e-03	3.18 e-04	EPA, 1993
Air-to-plant biotransfer factor																		
grains	6.55 e+04	2.39 e+05	5.20 e+05	5.20 e+05	5.20 e+05	1.00 e+06	2.36 e+06	4.57 e+04	9.75 e+04	9.75 e+04	1.62 e+05	1.62 e+05	1.62 e+05	1.62 e+05	8.30 e+06	8.30 e+06	2.28 e+06	Lorber 1995

Table G-1b. (Continued)

Input parameter	CDDs 2378	12378	123478	123789	123678	1234678	Octa	CDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Source
legumes	6.55 e+04	2.39 e+05	5.20 e+05	5.20 e+05	5.20 e+05	1.00 e+06	2.36 e+06	4.57 e+04	9.75 e+04	9.75 e+04	1.62 e+05	1.62 e+05	1.62 e+05	1.62 e+05	8.30 e+06	8.30 e+06	2.28 e+06	Lorber 1995
fruits	6.55 e+04	2.39 e+05	5.20 e+05	5.20 e+05	5.20 e+05	1.00 e+06	2.36 e+06	4.57 e+04	9.75 e+04	9.75 e+04	1.62 e+05	1.62 e+05	1.62 e+05	1.62 e+05	8.30 e+06	8.30 e+06	2.28 e+06	Lorber 1995
fruiting vegetables	6.55 e+04	2.39 e+05	5.20 e+05	5.20 e+05	5.20 e+05	1.00 e+06	2.36 e+06	4.57 e+04	9.75 e+04	9.75 e+04	1.62 e+05	1.62 e+05	1.62 e+05	1.62 e+05	8.30 e+06	8.30 e+06	2.28 e+06	Lorber 1995
leafy vegetables	6.55 e+04	2.39 e+05	5.20 e+05	5.20 e+05	5.20 e+05	1.00 e+06	2.36 e+06	4.57 e+04	9.75 e+04	9.75 e+04	1.62 e+05	1.62 e+05	1.62 e+05	1.62 e+05	8.30 e+06	8.30 e+06	2.28 e+06	Lorber 1995
forage	6.55 e+04	2.39 e+05	5.20 e+05	5.20 e+05	5.20 e+05	1.00 e+06	2.36 e+06	4.57 e+04	9.75 e+04	9.75 e+04	1.62 e+05	1.62 e+05	1.62 e+05	1.62 e+05	8.30 e+06	8.30 e+06	2.28 e+06	Lorber 1995
silage	6.55 e+04	2.39 e+05	5.20 e+05	5.20 e+05	5.20 e+05	1.00 e+06	2.36 e+06	4.57 e+04	9.75 e+04	9.75 e+04	1.62 e+05	1.62 e+05	1.62 e+05	1.62 e+05	8.30 e+06	8.30 e+06	2.28 e+06	Lorber 1995
Bioconcentration ratio of contaminant, unitless																		
beef	5.76	5.55	2.69	2.99	2.32	0.48	0.69	1.25	4.13	0.97	3.12	2.67	2.72	2.37	0.55	1.32	0.27	EPA, 1994a
dairy	5.76	5.55	2.69	2.99	2.32	0.48	0.69	1.25	4.13	0.97	3.12	2.67	2.72	2.37	0.55	1.32	0.27	EPA, 1994a
poultry	8.8	6.8	3.6	2.4	5.6	1.4	0.3	3.1	7.4	18	4.8	5.3	4.1	2.1	1	0.9	0.3	Steph. 1995
eggs	7.8	6	5.4	4.5	10.2	4.8	4.3	2.7	7.8	20.5	7.4	8.2	6.2	3	3.1	2.2	1.4	Steph. 1995

Appendix G-2

Dispersion Modeling Results (Chapter 11) for Dioxin Congener Specific Emission Rates

Table G-2. Dioxin Congener Specific Annual Average Dispersion Modeling Results for Model Plants

ISCST3 Dispersion Modeling Results																	
Congener	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
LCH - fisher & farmer - 500 m																	
Particles																	
Inhalable air conc. (µg/m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet dep. (g/m2)	1.62e-11	6.92e-10	1.98e-09	1.58e-09	1.22e-09	1.24e-09	1.28e-08	2.52e-10	1.60e-09	3.12e-10	2.67e-09	8.24e-10	1.65e-09	3.32e-09	4.15e-09	3.73e-08	3.63e-09
Dry dep. (g/m2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Vapors																	
Air conc. (µg/m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet dep. (g/m2)	1.61e-11	1.97e-10	1.21e-10	2.61e-11	4.14e-11	2.05e-11	0	5.00e-10	5.57e-10	1.83e-10	1.38e-10	4.27e-11	1.66e-10	2.03e-10	1.40e-10	6.17e-10	0
Dry dep. (g/m2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LOH - fisher & farmer - 500 m																	
Particles																	
Inhalable air conc. (µg/m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet dep. (g/m2)	3.68e-10	5.35e-10	1.45e-09	1.03e-09	6.55e-10	2.44e-09	2.91e-09	1.67e-10	4.21e-10	3.15e-10	7.14e-10	4.52e-10	6.48e-10	5.56e-10	1.13e-09	1.24e-09	1.27e-09
Dry dep. (g/m2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Vapors																	
Air conc. (µg/m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet dep. (g/m2)	2.49e-10	1.04e-10	6.03e-11	1.16e-11	1.51e-11	2.76e-11	0	2.26e-10	1.00e-10	1.26e-10	2.52e-11	1.60e-11	4.43e-11	2.32e-11	2.61e-11	1.40e-11	0
Dry dep. (g/m2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SCH - fisher & farmer - 500 m																	
Particles																	
Inhalable air conc. (µg/m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet dep. (g/m2)	2.45e-10	1.05e-10	2.99e-10	2.38e-10	1.85e-10	1.87e-10	1.94e-09	3.80e-11	2.42e-10	4.72e-11	4.04e-10	1.25e-10	2.50e-10	5.00e-10	6.29e-10	5.65e-09	5.47e-10
Dry dep. (g/m2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Vapors																	
Air conc. (µg/m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table G-2. (Continued)

ISCST3 Dispersion Modeling Results																	
Congener	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
Wet dep. (g/m2)	2.44e-10	3.00e-11	1.83e-11	3.96e-12	6.29e-12	3.11e-12	0	7.58e-11	8.44e-11	2.79e-11	2.10e-11	6.48e-12	2.52e-11	3.07e-11	2.13e-11	9.40e-11	0
Dry dep. (g/m2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SOH - fisher & farmer - 500 m																	
Particles																	
Inhalable air conc. (µg/m3)	1.07e-15	1.55e-15	4.21e-15	2.99e-15	1.91e-15	7.11e-15	8.45e-15	4.86e-16	1.23e-15	9.17e-16	2.07e-15	1.32e-15	1.89e-15	1.62e-15	3.30e-15	3.61e-15	3.68e-15
Wet dep. (g/m2)	2.02e-11	2.93e-11	7.95e-11	5.65e-11	3.60e-11	1.34e-10	1.60e-10	9.18e-12	2.32e-11	1.73e-11	3.91e-11	2.49e-11	3.57e-11	3.05e-11	6.23e-11	6.81e-11	6.95e-11
Dry dep. (g/m2)	1.24e-15	1.81e-15	4.90e-15	3.48e-15	2.22e-15	8.27e-15	9.83e-15	5.66e-16	1.43e-15	1.07e-15	2.41e-15	1.53e-15	2.20e-15	1.88e-15	3.84e-15	4.20e-15	4.28e-15
Vapors																	
Air conc. (µg/m3)	1.52e-15	6.35e-16	3.68e-16	7.10e-17	9.23e-17	1.69e-16	0	1.38e-15	6.13e-16	7.72e-16	1.54e-16	9.78e-17	2.72e-16	1.42e-16	1.60e-16	8.56e-17	0
Wet dep. (g/m2)	1.38e-11	5.77e-12	3.35e-12	6.45e-13	8.39e-13	1.53e-12	0	1.26e-11	5.56e-12	7.02e-12	1.40e-12	8.88e-13	2.47e-12	1.29e-12	1.45e-12	7.78e-13	0
Dry dep. (g/m2)	9.58e-17	4.00e-17	2.32e-17	4.48e-18	5.82e-18	1.06e-17	0	8.73e-17	3.86e-17	4.87e-17	9.71e-18	6.17e-18	1.71e-17	8.93e-18	1.01e-17	5.40e-18	0
LCC - resident & child - 5000 m																	
Particles																	
Inhalable air conc. (µg/m3)	4.68e-14	2.00e-12	5.71e-12	4.55e-12	3.53e-12	3.57e-12	3.70e-11	7.27e-13	4.63e-12	9.00e-13	7.70e-12	2.38e-12	4.78e-12	9.59e-12	1.20e-11	1.08e-10	1.05e-11
Wet dep. (g/m2)	5.20e-13	2.22e-11	6.35e-11	5.06e-11	3.93e-11	3.97e-11	4.11e-10	8.08e-12	5.14e-11	1.00e-11	8.56e-11	2.65e-11	5.31e-11	1.07e-10	1.33e-10	1.20e-09	1.17e-10
Dry dep. (g/m2)	3.89e-14	1.66e-12	4.75e-12	3.79e-12	2.94e-12	2.97e-12	3.08e-11	6.05e-13	3.85e-12	7.49e-13	6.41e-12	1.98e-12	3.97e-12	7.98e-12	9.98e-12	8.96e-11	8.72e-12
Vapors																	
Air conc. (µg/m3)	1.08e-13	1.33e-12	8.12e-13	1.76e-13	2.78e-13	1.38e-13	0	3.37e-12	3.75e-12	1.23e-12	9.29e-13	2.87e-13	1.12e-12	1.37e-12	9.45e-13	4.15e-12	0
Wet dep. (g/m2)	6.68e-13	8.20e-12	5.02e-12	1.09e-12	1.72e-12	8.50e-13	0	2.08e-11	2.32e-11	7.61e-12	5.74e-12	1.78e-12	6.90e-12	8.44e-12	5.84e-12	2.57e-11	0
Dry dep. (g/m2)	6.82e-15	8.37e-14	5.12e-14	1.11e-14	1.76e-14	8.68e-15	0	2.12e-13	2.36e-13	7.77e-14	5.86e-14	1.81e-14	7.04e-14	8.61e-14	5.96e-14	2.62e-13	0
LOC - resident & child - 5000 m																	
Particles																	
Inhalable air conc. (µg/m3)	1.39e-12	2.02e-12	5.46e-12	3.87e-12	2.47e-12	9.21e-12	1.10e-11	6.29e-13	1.59e-12	1.19e-12	2.69e-12	1.71e-12	2.44e-12	2.10e-12	4.26e-12	4.69e-12	4.78e-12
Wet dep. (g/m2)	9.63e-12	1.40e-11	3.79e-11	2.69e-11	1.72e-11	6.39e-11	7.61e-11	4.37e-12	1.10e-11	8.26e-12	1.87e-11	1.18e-11	1.70e-11	1.46e-11	2.96e-11	3.25e-11	3.32e-11
Dry dep. (g/m2)	2.29e-12	3.32e-12	8.98e-12	6.37e-12	4.07e-12	1.51e-11	1.81e-11	1.04e-12	2.62e-12	1.96e-12	4.43e-12	2.81e-12	4.02e-12	3.45e-12	7.02e-12	7.71e-12	7.87e-12

Table G-2. (Continued)

ISCST3 Dispersion Modeling Results																	
Congener	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
Vapors																	
Air conc. (µg/m3)	3.40e-12	1.42e-12	8.22e-13	1.58e-13	2.06e-13	3.76e-13	0	3.08e-12	1.36e-12	1.72e-12	3.44e-13	2.18e-13	6.05e-13	3.16e-13	3.56e-13	1.91e-13	0
Wet dep. (g/m2)	1.01e-11	4.21e-12	2.43e-12	4.68e-13	6.11e-13	1.11e-12	0	9.14e-12	4.04e-12	5.11e-12	1.02e-12	6.45e-13	1.79e-12	9.36e-13	1.05e-12	5.67e-13	0
Dry dep. (g/m2)	2.14e-13	8.96e-14	5.18e-14	9.97e-15	1.30e-14	2.37e-14	0	1.94e-13	8.60e-14	1.09e-13	2.17e-14	1.37e-14	3.81e-14	1.99e-14	2.24e-14	1.21e-14	0
SCC - resident & child - 5000 m																	
Particles																	
Inhalable air conc. (µg/m3)	4.25e-12	1.82e-12	5.19e-12	4.13e-12	3.21e-12	3.24e-12	3.36e-11	6.59e-13	4.19e-12	8.20e-13	7.01e-12	2.16e-12	4.34e-12	8.67e-12	1.09e-11	9.80e-11	9.50e-12
Wet dep. (g/m2)	7.24e-12	3.10e-12	8.85e-12	7.05e-12	5.48e-12	5.53e-12	5.73e-11	1.12e-12	7.15e-12	1.40e-12	1.20e-11	3.69e-12	7.40e-12	1.48e-11	1.86e-11	1.67e-10	1.62e-11
Dry dep. (g/m2)	3.79e-12	1.62e-12	4.63e-12	3.69e-12	2.87e-12	2.89e-12	2.99e-11	5.88e-13	3.74e-12	7.31e-13	6.26e-12	1.93e-12	3.87e-12	7.74e-12	9.73e-12	8.75e-11	8.47e-12
Vapors																	
Air conc. (µg/m3)	9.87e-12	1.21e-12	7.43e-13	1.60e-13	2.55e-13	1.26e-13	0	3.07e-12	3.42e-12	1.13e-12	8.51e-13	2.63e-13	1.02e-12	1.24e-12	8.64e-13	3.81e-12	0
Wet dep. (g/m2)	9.44e-12	1.16e-12	7.10e-13	1.53e-13	2.43e-13	1.20e-13	0	2.93e-12	3.27e-12	1.08e-12	8.14e-13	2.51e-13	9.75e-13	1.19e-12	8.26e-13	3.64e-12	0
Dry dep. (g/m2)	6.23e-13	7.65e-14	4.68e-14	1.01e-14	1.61e-14	7.93e-15	0	1.94e-13	2.16e-13	7.12e-14	5.37e-14	1.66e-14	6.44e-14	7.83e-14	5.45e-14	2.40e-13	0
SOC - resident & child - 5000 m																	
Particles																	
Inhalable air conc. (µg/m3)	6.35e-13	9.23e-13	2.50e-12	1.78e-12	1.13e-12	4.22e-12	5.02e-12	2.89e-13	7.30e-13	5.45e-13	1.23e-12	7.82e-13	1.12e-12	9.61e-13	1.96e-12	2.14e-12	2.19e-12
Wet dep. (g/m2)	4.76e-13	6.92e-13	1.87e-12	1.33e-12	8.49e-13	3.17e-12	3.76e-12	2.17e-13	5.47e-13	4.09e-13	9.24e-13	5.87e-13	8.42e-13	7.21e-13	1.47e-12	1.61e-12	1.64e-12
Dry dep. (g/m2)	1.13e-12	1.64e-12	4.44e-12	3.16e-12	2.01e-12	7.51e-12	8.93e-12	5.14e-13	1.30e-12	9.69e-13	2.19e-12	1.39e-12	2.00e-12	1.71e-12	3.48e-12	3.81e-12	3.89e-12
Vapors																	
Air conc. (µg/m3)	1.60e-12	6.67e-13	3.87e-13	7.46e-14	9.70e-14	1.77e-13	0	1.45e-12	6.43e-13	8.11e-13	1.62e-13	1.03e-13	2.85e-13	1.49e-13	1.68e-13	8.99e-14	0
Wet dep. (g/m2)	4.77e-13	1.99e-13	1.16e-13	2.23e-14	2.90e-14	5.30e-14	0	4.35e-13	1.92e-13	2.43e-13	4.83e-14	3.07e-14	8.53e-14	4.45e-14	5.02e-14	2.69e-14	0
Dry dep. (g/m2)	1.01e-13	4.20e-14	2.44e-14	4.70e-15	6.12e-15	1.12e-14	0	9.17e-14	4.06e-14	5.12e-14	1.02e-14	6.48e-15	1.80e-14	9.38e-15	1.06e-14	5.67e-15	0

Appendix G-3

IEM Modeled Dioxin Congener Specific Environmental Media Concentrations

Table G-3. IEM Modeled Dioxin Congener Specific Environmental Media Concentrations

	PCDDs							PCDFs										
Scenario	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total
LCH - fisher & farmer																		
Soil concentration, mg/g	2.3e-11	6.5e-10	1.5e-09	1.2e-09	9.2e-10	9.1e-10	9.3e-09	5.5e-10	1.6e-09	3.6e-10	2.0e-09	6.3e-10	1.3e-09	2.6e-09	3.1e-09	2.8e-08	2.6e-09	5.7e-08
Water concentration, mg/L	3.7e-12	1.0e-10	1.1e-10	1.0e-10	7.9e-11	6.2e-11	7.0e-10	9.9e-11	1.8e-10	4.7e-11	1.8e-10	5.4e-11	1.1e-10	1.7e-10	2.2e-10	1.9e-09	1.7e-10	4.3e-09
Whole Fish concentration, mg/kg	3.7e-08	1.0e-06	1.1e-06	8.4e-07	6.6e-07	8.3e-08	1.7e-08	8.6e-07	2.5e-06	5.8e-07	1.5e-06	4.5e-07	9.5e-07	1.9e-06	2.8e-07	2.5e-06	4.8e-09	1.5e-05
Plant concentration, µg/g																		
grains	1.3e-11	3.6e-10	1.8e-10	1.4e-10	2.1e-10	6.4e-11	1.5e-09	3.6e-10	6.1e-10	1.7e-10	4.8e-10	1.5e-10	3.1e-10	6.0e-10	3.3e-10	2.9e-09	8.4e-11	8.4e-09
legumes	9.2e-11	2.5e-09	5.3e-09	4.0e-09	3.3e-09	3.1e-09	3.3e-08	2.2e-09	5.9e-09	1.4e-09	7.3e-09	2.3e-09	4.7e-09	9.2e-09	1.1e-08	9.5e-08	8.9e-09	2.0e-07
potatoes	8.5e-11	2.3e-09	3.0e-09	7.3e-09	2.4e-09	1.5e-09	2.0e-08	2.1e-09	4.9e-09	1.2e-09	5.4e-09	1.7e-09	3.5e-09	6.7e-11	5.8e-09	5.1e-08	3.0e-09	1.2e-07
root vegetables	8.5e-11	2.3e-09	3.0e-09	7.3e-09	2.4e-09	1.5e-09	2.0e-08	2.1e-09	4.9e-09	1.2e-09	5.4e-09	1.7e-09	3.5e-09	6.7e-11	5.8e-09	5.1e-08	3.0e-09	1.2e-07
fruits	4.9e-10	1.4e-08	3.1e-08	2.4e-08	1.9e-08	1.9e-08	1.9e-07	1.1e-08	3.3e-08	7.5e-09	4.2e-08	1.3e-08	2.7e-08	5.3e-08	6.4e-08	5.6e-07	5.4e-08	1.2e-06
fruiting vegetables	4.9e-10	1.4e-08	3.1e-08	2.4e-08	1.9e-08	1.9e-08	1.9e-07	1.1e-08	3.3e-08	7.5e-09	4.2e-08	1.3e-08	2.7e-08	5.3e-08	6.4e-08	5.6e-07	5.4e-08	1.2e-06
leafy vegetables	9.3e-10	2.6e-08	6.0e-08	4.6e-08	3.6e-08	3.6e-08	3.6e-07	2.2e-08	6.2e-08	1.4e-08	8.0e-08	2.5e-08	5.2e-08	1.0e-07	1.2e-07	1.1e-06	1.0e-07	2.2e-06
forage	1.7e-09	4.6e-08	1.0e-07	7.8e-08	6.3e-08	6.1e-08	6.3e-07	4.0e-08	1.1e-07	2.5e-08	1.4e-07	4.3e-08	9.0e-08	1.8e-07	2.1e-07	1.8e-06	1.7e-07	3.8e-06
silage	5.5e-10	1.5e-08	3.5e-08	2.7e-08	2.1e-08	2.1e-08	2.1e-07	1.3e-08	3.6e-08	8.4e-09	4.7e-08	1.5e-08	3.0e-08	5.9e-08	7.2e-08	6.3e-07	6.0e-08	1.3e-06
Animal concentrations, µg/g																		
beef	6.7e-09	1.8e-07	2.0e-07	1.7e-07	1.0e-07	2.1e-08	3.1e-07	3.4e-08	3.2e-07	1.7e-08	3.1e-07	8.1e-08	1.7e-07	2.9e-07	8.2e-08	1.7e-06	3.3e-08	4.1e-06
dairy	3.0e-09	7.9e-08	8.9e-08	7.5e-08	4.6e-08	9.4e-09	1.4e-07	1.5e-08	1.4e-07	7.7e-09	1.4e-07	3.6e-08	7.7e-08	1.3e-07	3.7e-08	7.8e-07	1.5e-08	1.8e-06
poultry	1.8e-08	3.9e-07	4.8e-07	2.5e-07	4.5e-07	1.1e-07	2.5e-07	1.5e-07	1.0e-06	5.7e-07	8.6e-07	2.9e-07	4.8e-07	4.7e-07	2.7e-07	2.2e-06	6.9e-08	8.3e-06
eggs	1.6e-08	3.4e-07	7.2e-07	4.6e-07	8.3e-07	3.8e-07	3.5e-06	1.3e-07	1.1e-06	6.5e-07	1.3e-06	4.6e-07	7.2e-07	6.8e-07	8.5e-07	5.3e-06	3.2e-07	1.8e-05
LOH - fisher & farmer																		
Soil concentration, mg/g	4.5e-10	4.7e-10	1.1e-09	7.5e-10	4.9e-10	1.8e-09	2.1e-09	2.9e-10	3.8e-10	3.2e-10	5.4e-10	3.4e-10	5.0e-10	4.2e-10	8.4e-10	9.1e-10	9.2e-10	1.3e-08
Water concentration, mg/L	7.0e-11	7.3e-11	7.8e-11	6.5e-11	4.2e-11	1.2e-10	1.6e-10	5.2e-11	4.3e-11	4.2e-11	4.6e-11	2.9e-11	4.3e-11	2.7e-11	5.9e-11	6.4e-11	6.0e-11	1.1e-09

Table G-3. (Continued)

	PCDDs							PCDFs										
Scenario	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total
Whole fish concentration, mg/kg	7.1e-07	7.4e-07	7.9e-07	5.4e-07	3.5e-07	1.6e-07	3.8e-09	4.5e-07	6.1e-07	5.1e-07	3.9e-07	2.5e-07	3.6e-07	3.1e-07	7.6e-08	8.3e-08	1.7e-09	6.3e-06
Plant concentration, µg/g																		
grains	2.5e-10	2.6e-10	1.3e-10	9.1e-11	1.1e-10	1.3e-10	3.4e-10	1.9e-10	1.5e-10	1.5e-10	1.3e-10	7.9e-11	1.2e-10	9.8e-11	8.8e-11	9.6e-11	2.9e-11	2.4e-09
legumes	1.8e-09	1.8e-09	3.8e-09	2.6e-09	1.7e-09	6.1e-09	7.4e-09	1.1e-09	1.4e-09	1.2e-09	1.9e-09	1.2e-09	1.8e-09	1.5e-09	2.9e-09	3.2e-09	3.1e-09	4.5e-08
potatoes	1.6e-09	1.7e-09	2.2e-09	4.7e-09	1.3e-09	2.9e-09	4.6e-09	1.1e-09	1.2e-09	1.1e-09	1.4e-09	8.9e-10	1.3e-09	1.1e-11	1.6e-09	1.7e-09	1.1e-09	3.0e-08
root vegetables	1.6e-09	1.7e-09	2.2e-09	4.7e-09	1.3e-09	2.9e-09	4.6e-09	1.1e-09	1.2e-09	1.1e-09	1.4e-09	8.9e-10	1.3e-09	1.1e-11	1.6e-09	1.7e-09	1.1e-09	3.0e-08
fruits	9.4e-09	9.7e-09	2.2e-08	1.5e-08	1.0e-08	3.7e-08	4.3e-08	6.0e-09	7.9e-09	6.7e-09	1.1e-08	7.0e-09	1.0e-08	8.7e-09	1.7e-08	1.9e-08	1.9e-08	2.6e-07
fruiting vegetables	9.4e-09	9.7e-09	2.2e-08	1.5e-08	1.0e-08	3.7e-08	4.3e-08	6.0e-09	7.9e-09	6.7e-09	1.1e-08	7.0e-09	1.0e-08	8.7e-09	1.7e-08	1.9e-08	1.9e-08	2.6e-07
leafy vegetables	1.8e-08	1.8e-08	4.3e-08	2.9e-08	1.9e-08	7.0e-08	8.3e-08	1.1e-08	1.5e-08	1.3e-08	2.1e-08	1.3e-08	2.0e-08	1.6e-08	3.3e-08	3.6e-08	3.6e-08	4.9e-07
forage	3.2e-08	3.3e-08	7.4e-08	5.1e-08	3.3e-08	1.2e-07	1.4e-07	2.1e-08	2.6e-08	2.3e-08	3.7e-08	2.3e-08	3.4e-08	2.9e-08	5.6e-08	6.1e-08	6.1e-08	8.6e-07
silage	1.0e-08	1.1e-08	2.5e-08	1.7e-08	1.1e-08	4.1e-08	4.9e-08	6.7e-09	8.8e-09	7.5e-09	1.2e-08	7.8e-09	1.2e-08	9.7e-09	1.9e-08	2.1e-08	2.1e-08	2.9e-07
Animal concentrations, µg/g																		
beef	1.3e-07	1.3e-07	1.4e-07	1.1e-07	5.5e-08	4.1e-08	7.0e-08	1.8e-08	7.7e-08	1.5e-08	8.1e-08	4.4e-08	6.5e-08	4.8e-08	2.2e-08	5.8e-08	1.2e-08	1.1e-06
dairy	5.7e-08	5.7e-08	6.4e-08	4.9e-08	2.5e-08	1.9e-08	3.2e-08	8.0e-09	3.4e-08	6.9e-09	3.6e-08	2.0e-08	2.9e-08	2.2e-08	9.9e-09	2.6e-08	5.3e-09	5.0e-07
poultry	3.5e-07	2.8e-07	3.5e-07	1.6e-07	2.4e-07	2.2e-07	5.6e-08	7.8e-08	2.5e-07	5.1e-07	2.3e-07	1.6e-07	1.8e-07	7.8e-08	7.4e-08	7.2e-08	2.4e-08	3.3e-06
eggs	3.1e-07	2.5e-07	5.2e-07	3.0e-07	4.4e-07	7.6e-07	8.0e-07	6.8e-08	2.6e-07	5.8e-07	3.5e-07	2.5e-07	2.7e-07	1.1e-07	2.3e-07	1.8e-07	1.1e-07	5.8e-06
SCH - fisher & farmer																		
Soil concentration, mg/g	3.6e-10	9.8e-11	2.3e-10	1.8e-10	1.4e-10	1.4e-10	1.4e-09	8.3e-11	2.4e-10	5.5e-11	3.1e-10	9.5e-11	2.0e-10	3.9e-10	4.7e-10	4.2e-09	4.0e-10	9.0e-09
Water concentration, mg/L	5.6e-11	1.5e-11	1.6e-11	1.5e-11	1.2e-11	9.3e-12	1.1e-10	1.5e-11	2.7e-11	7.1e-12	2.7e-11	8.2e-12	1.7e-11	2.5e-11	3.3e-11	2.9e-10	2.6e-11	7.1e-10
Whole fish concentration, mg/kg	5.6e-07	1.6e-07	1.7e-07	1.3e-07	1.0e-07	1.3e-08	2.5e-09	1.3e-07	3.8e-07	8.7e-08	2.2e-07	6.9e-08	1.4e-07	2.8e-07	4.3e-08	3.8e-07	7.2e-10	2.9e-06
Plant concentration, µg/g																		
grains	2.0e-10	5.5e-11	2.8e-11	2.1e-11	3.2e-11	9.7e-12	2.2e-10	5.4e-11	9.2e-11	2.5e-11	7.2e-11	2.2e-11	4.7e-11	9.0e-11	5.0e-11	4.4e-10	1.3e-11	1.5e-09
legumes	1.4e-09	3.8e-10	8.0e-10	6.1e-10	5.0e-10	4.7e-10	4.9e-09	3.3e-10	8.9e-10	2.1e-10	1.1e-09	3.4e-10	7.2e-10	1.4e-09	1.6e-09	1.4e-08	1.3e-09	3.1e-08

Table G-3. (Continued)

	PCDDs							PCDFs										
Scenario	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total
potatoes	1.3e-09	3.6e-10	4.6e-10	1.1e-09	3.7e-10	2.2e-10	3.1e-09	3.2e-10	7.5e-10	1.8e-10	8.1e-10	2.5e-10	5.3e-10	1.0e-11	8.8e-10	7.8e-09	4.6e-10	1.9e-08
root vegetables	1.3e-09	3.6e-10	4.6e-10	1.1e-09	3.7e-10	2.2e-10	3.1e-09	3.2e-10	7.5e-10	1.8e-10	8.1e-10	2.5e-10	5.3e-10	1.0e-11	8.8e-10	7.8e-09	4.6e-10	1.9e-08
fruits	7.4e-09	2.0e-09	4.7e-09	3.6e-09	2.9e-09	2.8e-09	2.9e-08	1.7e-09	4.9e-09	1.1e-09	6.4e-09	2.0e-09	4.1e-09	7.9e-09	9.7e-09	8.5e-08	8.1e-09	1.8e-07
fruiting vegetables	7.4e-09	2.0e-09	4.7e-09	3.6e-09	2.9e-09	2.8e-09	2.9e-08	1.7e-09	4.9e-09	1.1e-09	6.4e-09	2.0e-09	4.1e-09	7.9e-09	9.7e-09	8.5e-08	8.1e-09	1.8e-07
leafy vegetables	1.4e-08	3.9e-09	9.0e-09	6.9e-09	5.5e-09	5.4e-09	5.5e-08	3.3e-09	9.3e-09	2.2e-09	1.2e-08	3.7e-09	7.8e-09	1.5e-08	1.8e-08	1.6e-07	1.6e-08	3.5e-07
sforage	2.5e-08	7.0e-09	1.6e-08	1.2e-08	9.5e-09	9.2e-09	9.5e-08	6.0e-09	1.7e-08	3.9e-09	2.1e-08	6.5e-09	1.4e-08	2.6e-08	3.2e-08	2.8e-07	2.6e-08	6.1e-07
silage	8.3e-09	2.3e-09	5.3e-09	4.0e-09	3.2e-09	3.2e-09	3.2e-08	1.9e-09	5.5e-09	1.3e-09	7.1e-09	2.2e-09	4.6e-09	8.9e-09	1.1e-08	9.6e-08	9.1e-09	2.1e-07
Animal concentrations, µg/g																		
beef	1.0e-07	2.7e-08	3.0e-08	2.5e-08	1.6e-08	3.2e-09	4.7e-08	5.2e-09	4.8e-08	2.6e-09	4.7e-08	1.2e-08	2.6e-08	4.4e-08	1.2e-08	2.6e-07	5.0e-09	7.1e-07
dairy	4.5e-08	1.2e-08	1.3e-08	1.1e-08	7.0e-09	1.4e-09	2.1e-08	2.3e-09	2.1e-08	1.2e-09	2.1e-08	5.5e-09	1.2e-08	2.0e-08	5.6e-09	1.2e-07	2.3e-09	3.2e-07
poultry	2.8e-07	5.9e-08	7.3e-08	3.7e-08	6.9e-08	1.7e-08	3.7e-08	2.3e-08	1.5e-07	8.7e-08	1.3e-07	4.4e-08	7.2e-08	7.1e-08	4.2e-08	3.3e-07	1.0e-08	1.5e-06
eggs	2.5e-07	5.2e-08	1.1e-07	7.0e-08	1.3e-07	5.8e-08	5.3e-07	2.0e-08	1.6e-07	9.9e-08	2.0e-07	6.9e-08	1.1e-07	1.0e-07	1.3e-07	8.1e-07	4.9e-08	2.9e-06
								SOH - fisher & farmer										
Soil concentration, mg/g	2.5e-11	2.6e-11	6.0e-11	4.2e-11	2.7e-11	9.9e-11	1.2e-10	1.6e-11	2.1e-11	1.8e-11	3.0e-11	1.9e-11	2.8e-11	2.3e-11	4.6e-11	5.0e-11	5.1e-11	6.9e-10
Water concentration, mg/L	3.9e-12	4.0e-12	4.3e-12	3.6e-12	2.3e-12	6.7e-12	8.7e-12	2.9e-12	2.4e-12	2.3e-12	2.5e-12	1.6e-12	2.4e-12	1.5e-12	3.2e-12	3.5e-12	3.3e-12	5.9e-11
Whole Fish concentration, mg/kg	3.9e-08	4.1e-08	4.4e-08	3.0e-08	1.9e-08	8.9e-09	2.1e-10	2.5e-08	3.4e-08	2.8e-08	2.1e-08	1.4e-08	2.0e-08	1.7e-08	4.2e-09	4.5e-09	9.2e-11	3.5e-07
Plant concentration, µg/g																		
grains	1.4e-11	1.4e-11	7.4e-12	5.0e-12	6.3e-12	7.0e-12	1.8e-11	1.0e-11	8.1e-12	8.2e-12	6.9e-12	4.4e-12	6.5e-12	5.4e-12	5.4e-12	5.6e-12	1.6e-12	1.3e-10
legumes	9.7e-11	1.0e-10	2.1e-10	1.4e-10	9.6e-11	3.4e-10	4.1e-10	6.3e-11	7.8e-11	6.7e-11	1.1e-10	6.7e-11	9.9e-11	8.3e-11	1.6e-10	1.7e-10	1.7e-10	2.5e-09
potatoes	9.0e-11	9.3e-11	1.2e-10	2.6e-10	7.0e-11	1.6e-10	2.6e-10	6.1e-11	6.6e-11	6.0e-11	7.7e-11	4.9e-11	7.3e-11	6.1e-13	8.6e-11	9.3e-11	5.8e-11	1.7e-09
root vegetables	9.0e-11	9.3e-11	1.2e-10	2.6e-10	7.0e-11	1.6e-10	2.6e-10	6.1e-11	6.6e-11	6.0e-11	7.7e-11	4.9e-11	7.3e-11	6.1e-13	8.6e-11	9.3e-11	5.8e-11	1.7e-09
fruits	5.2e-10	5.3e-10	1.2e-09	8.5e-10	5.5e-10	2.0e-09	2.4e-09	3.3e-10	4.3e-10	3.7e-10	6.1e-10	3.9e-10	5.7e-10	4.8e-10	9.5e-10	1.0e-09	1.0e-09	1.4e-08
fruiting vegetables	5.2e-10	5.3e-10	1.2e-09	8.5e-10	5.5e-10	2.0e-09	2.4e-09	3.3e-10	4.3e-10	3.7e-10	6.1e-10	3.9e-10	5.7e-10	4.8e-10	9.5e-10	1.0e-09	1.0e-09	1.4e-08

Table G-3. (Continued)

	PCDDs							PCDFs										
Scenario	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total
leafy vegetables	9.8e-10	1.0e-09	2.4e-09	1.6e-09	1.0e-09	3.9e-09	4.5e-09	6.3e-10	8.2e-10	7.0e-10	1.2e-09	7.3e-10	1.1e-09	9.1e-10	1.8e-09	2.0e-09	2.0e-09	2.7e-08
forage	1.8e-09	1.8e-09	4.0e-09	2.8e-09	1.8e-09	6.6e-09	7.8e-09	1.1e-09	1.5e-09	1.2e-09	2.0e-09	1.3e-09	1.9e-09	1.6e-09	3.1e-09	3.4e-09	3.4e-09	4.7e-08
silage	5.8e-10	6.0e-10	1.4e-09	9.5e-10	6.2e-10	2.3e-09	2.7e-09	3.7e-10	4.9e-10	4.1e-10	6.8e-10	4.3e-10	6.4e-10	5.3e-10	1.1e-09	1.1e-09	1.2e-09	1.6e-08
Animal concentrations, µg/g																		
beef	7.1e-09	7.0e-09	7.8e-09	5.9e-09	3.0e-09	2.3e-09	3.9e-09	9.9e-10	4.2e-09	8.5e-10	4.4e-09	2.4e-09	3.6e-09	2.7e-09	1.2e-09	3.2e-09	6.4e-10	6.1e-08
dairy	3.1e-09	3.1e-09	3.5e-09	2.7e-09	1.3e-09	1.0e-09	1.7e-09	4.4e-10	1.9e-09	3.8e-10	2.0e-09	1.1e-09	1.6e-09	1.2e-09	5.5e-10	1.4e-09	2.9e-10	2.7e-08
poultry	1.9e-08	1.5e-08	1.9e-08	8.8e-09	1.3e-08	1.2e-08	3.1e-09	4.3e-09	1.4e-08	2.8e-08	1.2e-08	8.7e-09	1.0e-08	4.3e-09	4.1e-09	4.0e-09	1.3e-09	1.8e-07
eggs	1.7e-08	1.4e-08	2.9e-08	1.6e-08	2.4e-08	4.2e-08	4.4e-08	3.8e-09	1.4e-08	3.2e-08	1.9e-08	1.4e-08	1.5e-08	6.1e-09	1.3e-08	9.7e-09	6.2e-09	3.2e-07
LCC - Resident & Child																		
Soil concentration, mg/g	9.0e-13	2.3e-11	5.3e-11	4.0e-11	3.2e-11	3.2e-11	3.2e-10	2.2e-11	5.7e-11	1.3e-11	7.1e-11	2.2e-11	4.7e-11	9.0e-11	1.1e-10	9.5e-10	9.1e-11	2.0e-09
Water concentration, mg/L	1.4e-13	3.6e-12	3.8e-12	3.4e-12	2.7e-12	2.1e-12	2.4e-11	3.8e-12	6.4e-12	1.7e-12	6.1e-12	1.9e-12	4.0e-12	5.8e-12	7.6e-12	6.7e-11	6.0e-12	1.5e-10
Plant concentration, µg/g																		
grains	3.5e-12	1.5e-10	1.8e-10	4.3e-11	6.8e-11	6.0e-11	5.1e-11	7.9e-11	1.8e-10	5.7e-11	8.0e-11	2.5e-11	8.7e-11	1.1e-10	3.3e-09	1.5e-08	2.9e-12	1.9e-08
legumes	3.6e-12	9.6e-11	1.9e-10	1.5e-10	1.2e-10	1.1e-10	1.2e-09	8.9e-11	2.2e-10	5.3e-11	2.6e-10	8.2e-11	1.7e-10	3.3e-10	4.5e-10	3.7e-09	3.2e-10	7.5e-09
potatoes	3.3e-12	8.5e-11	1.1e-10	2.5e-10	8.4e-11	5.1e-11	7.1e-10	8.3e-11	1.8e-10	4.5e-11	1.9e-10	5.8e-11	1.2e-10	2.4e-12	2.0e-10	1.8e-09	1.1e-10	4.0e-09
root vegetables	3.3e-12	8.5e-11	1.1e-10	2.5e-10	8.4e-11	5.1e-11	7.1e-10	8.3e-11	1.8e-10	4.5e-11	1.9e-10	5.8e-11	1.2e-10	2.4e-12	2.0e-10	1.8e-09	1.1e-10	4.0e-09
fruits	1.9e-11	5.1e-10	1.1e-09	8.6e-10	6.8e-10	6.7e-10	6.8e-09	4.6e-10	1.2e-09	2.9e-10	1.5e-09	4.7e-10	9.9e-10	1.9e-09	2.4e-09	2.1e-08	1.9e-09	4.2e-08
fruiting vegetables	1.9e-11	5.1e-10	1.1e-09	8.6e-10	6.8e-10	6.7e-10	6.8e-09	4.6e-10	1.2e-09	2.9e-10	1.5e-09	4.7e-10	9.9e-10	1.9e-09	2.4e-09	2.1e-08	1.9e-09	4.2e-08
leafy vegetables	3.6e-11	9.5e-10	2.2e-09	1.6e-09	1.3e-09	1.3e-09	1.3e-08	8.7e-10	2.3e-09	5.4e-10	2.9e-09	8.9e-10	1.9e-09	3.6e-09	4.5e-09	3.9e-08	3.7e-09	8.1e-08
LOC - Resident & Child																		
Soil concentration, mg/g	1.6e-11	1.6e-11	3.6e-11	2.5e-11	1.6e-11	5.8e-11	6.9e-11	1.1e-11	1.3e-11	1.1e-11	1.8e-11	1.1e-11	1.7e-11	1.4e-11	2.7e-11	3.0e-11	3.0e-11	4.2e-10
Water concentration, mg/L	2.5e-12	2.4e-12	2.6e-12	2.1e-12	1.4e-12	3.9e-12	5.1e-12	1.9e-12	1.5e-12	1.4e-12	1.5e-12	9.5e-13	1.4e-12	9.0e-13	1.9e-12	2.1e-12	2.0e-12	3.5e-11

Table G-3. (Continued)

	PCDDs							PCDFs										
Scenario	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total
Plant concentration, µg/g																		
grains	1.0e-10	1.5e-10	1.8e-10	3.8e-11	4.9e-11	1.6e-10	1.1e-11	6.6e-11	6.1e-11	7.6e-11	2.8e-11	1.7e-11	4.5e-11	2.5e-11	1.2e-09	6.7e-10	9.5e-13	2.9e-09
legumes	6.8e-11	6.9e-11	1.4e-10	9.4e-11	6.3e-11	2.2e-10	2.6e-10	4.6e-11	5.3e-11	4.7e-11	6.9e-11	4.4e-11	6.6e-11	5.5e-11	1.3e-10	1.3e-10	1.1e-10	1.7e-09
potatoes	5.9e-11	5.7e-11	7.1e-11	1.5e-10	4.2e-11	9.3e-11	1.5e-10	4.1e-11	4.1e-11	3.8e-11	4.6e-11	2.9e-11	4.4e-11	3.6e-13	5.1e-11	5.5e-11	3.4e-11	1.0e-09
root vegetables	5.9e-11	5.7e-11	7.1e-11	1.5e-10	4.2e-11	9.3e-11	1.5e-10	4.1e-11	4.1e-11	3.8e-11	4.6e-11	2.9e-11	4.4e-11	3.6e-13	5.1e-11	5.5e-11	3.4e-11	1.0e-09
fruits	3.6e-10	3.6e-10	8.1e-10	5.6e-10	3.6e-10	1.3e-09	1.6e-09	2.4e-10	2.9e-10	2.5e-10	4.0e-10	2.5e-10	3.8e-10	3.1e-10	6.4e-10	6.8e-10	6.7e-10	9.4e-09
fruiting vegetables	3.6e-10	3.6e-10	8.1e-10	5.6e-10	3.6e-10	1.3e-09	1.6e-09	2.4e-10	2.9e-10	2.5e-10	4.0e-10	2.5e-10	3.8e-10	3.1e-10	6.4e-10	6.8e-10	6.7e-10	9.4e-09
leafy vegetables	6.8e-10	6.8e-10	1.5e-09	1.1e-09	6.9e-10	2.5e-09	3.0e-09	4.4e-10	5.5e-10	4.8e-10	7.6e-10	4.8e-10	7.2e-10	6.0e-10	1.2e-09	1.3e-09	1.3e-09	1.8e-08
SCC - Resident & Child																		
Soil concentration, mg/g	1.5e-11	4.3e-12	1.0e-11	7.9e-12	6.3e-12	6.2e-12	6.3e-11	3.5e-12	1.0e-11	2.4e-12	1.4e-11	4.3e-12	9.0e-12	1.7e-11	2.1e-11	1.9e-10	1.8e-11	4.0e-10
Water concentration, mg/L	2.3e-12	6.6e-13	7.4e-13	6.8e-13	5.3e-13	4.2e-13	4.8e-12	6.2e-13	1.2e-12	3.1e-13	1.2e-12	3.7e-13	7.6e-13	1.1e-12	1.5e-12	1.3e-11	1.2e-12	3.1e-11
Plant concentration, µg/g																		
grains	2.8e-10	1.2e-10	1.6e-10	3.6e-11	5.7e-11	5.3e-11	1.0e-11	6.1e-11	1.4e-10	4.7e-11	6.1e-11	1.9e-11	7.2e-11	8.9e-11	3.0e-09	1.3e-08	5.7e-13	1.8e-08
legumes	7.1e-11	2.2e-11	4.6e-11	3.3e-11	2.7e-11	2.6e-11	2.6e-10	1.6e-11	4.7e-11	1.1e-11	5.9e-11	1.8e-11	3.9e-11	7.4e-11	1.5e-10	1.0e-09	7.2e-11	2.0e-09
potatoes	5.6e-11	1.6e-11	2.1e-11	5.0e-11	1.6e-11	9.9e-12	1.4e-10	1.4e-11	3.3e-11	8.1e-12	3.6e-11	1.1e-11	2.4e-11	4.5e-13	3.9e-11	3.5e-10	2.1e-11	8.4e-10
root vegetables	5.6e-11	1.6e-11	2.1e-11	5.0e-11	1.6e-11	9.9e-12	1.4e-10	1.4e-11	3.3e-11	8.1e-12	3.6e-11	1.1e-11	2.4e-11	4.5e-13	3.9e-11	3.5e-10	2.1e-11	8.4e-10
fruits	3.6e-10	1.1e-10	2.5e-10	1.9e-10	1.5e-10	1.5e-10	1.5e-09	8.1e-11	2.5e-10	5.7e-11	3.4e-10	1.0e-10	2.2e-10	4.2e-10	5.8e-10	4.8e-09	4.4e-10	1.0e-08
fruiting vegetables	3.6e-10	1.1e-10	2.5e-10	1.9e-10	1.5e-10	1.5e-10	1.5e-09	8.1e-11	2.5e-10	5.7e-11	3.4e-10	1.0e-10	2.2e-10	4.2e-10	5.8e-10	4.8e-09	4.4e-10	1.0e-08
leafy vegetables	6.8e-10	2.0e-10	4.8e-10	3.7e-10	2.9e-10	2.9e-10	3.0e-09	1.5e-10	4.8e-10	1.1e-10	6.5e-10	2.0e-10	4.1e-10	8.0e-10	1.0e-09	9.0e-09	8.3e-10	1.9e-08
SOC - Resident & Child																		
Soil concentration, mg/g	1.6e-12	1.9e-12	4.7e-12	3.3e-12	2.1e-12	7.8e-12	9.2e-12	9.1e-13	1.5e-12	1.2e-12	2.3e-12	1.5e-12	2.1e-12	1.8e-12	3.6e-12	4.0e-12	4.0e-12	5.4e-11

Table G-3. (Continued)

	PCDDs							PCDFs										
Scenario	2378	12378	123478	123789	123678	1234678	Octa	2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total
Water concentration, mg/L	2.4e-13	2.9e-13	3.4e-13	2.8e-13	1.8e-13	5.3e-13	6.9e-13	1.6e-13	1.7e-13	1.6e-13	2.0e-13	1.3e-13	1.8e-13	1.2e-13	2.5e-13	2.8e-13	2.6e-13	4.5e-12
Plant concentration, µg/g																		
grains	4.5e-11	6.8e-11	8.5e-11	1.7e-11	2.2e-11	7.5e-11	1.5e-12	2.9e-11	2.7e-11	3.4e-11	1.2e-11	7.3e-12	2.0e-11	1.1e-11	5.9e-10	3.1e-10	1.3e-13	1.4e-09
legumes	8.8e-12	1.1e-11	2.4e-11	1.6e-11	1.1e-11	3.8e-11	4.5e-11	5.0e-12	8.0e-12	6.7e-12	1.1e-11	7.3e-12	1.1e-11	9.0e-12	2.9e-11	2.5e-11	1.9e-11	2.9e-10
potatoes	5.8e-12	6.8e-12	9.3e-12	2.1e-11	5.5e-12	1.2e-11	2.0e-11	3.5e-12	4.8e-12	4.1e-12	6.1e-12	3.8e-12	5.6e-12	4.7e-14	6.8e-12	7.4e-12	4.6e-12	1.3e-10
root vegetables	5.8e-12	6.8e-12	9.3e-12	2.1e-11	5.5e-12	1.2e-11	2.0e-11	3.5e-12	4.8e-12	4.1e-12	6.1e-12	3.8e-12	5.6e-12	4.7e-14	6.8e-12	7.4e-12	4.6e-12	1.3e-10
fruits	4.4e-11	5.4e-11	1.3e-10	9.4e-11	6.1e-11	2.2e-10	2.6e-10	2.5e-11	4.3e-11	3.4e-11	6.6e-11	4.2e-11	6.1e-11	5.2e-11	1.2e-10	1.2e-10	1.1e-10	1.5e-09
fruiting vegetables	4.4e-11	5.4e-11	1.3e-10	9.4e-11	6.1e-11	2.2e-10	2.6e-10	2.5e-11	4.3e-11	3.4e-11	6.6e-11	4.2e-11	6.1e-11	5.2e-11	1.2e-10	1.2e-10	1.1e-10	1.5e-09
leafy vegetables	8.3e-11	1.0e-10	2.6e-10	1.8e-10	1.2e-10	4.3e-10	5.0e-10	4.6e-11	8.1e-11	6.5e-11	1.3e-10	8.0e-11	1.2e-10	9.8e-11	2.1e-10	2.2e-10	2.2e-10	2.9e-09

Appendix G-4

Dioxin Congener Specific Lifetime Average Daily Dose (LADD) for Each Scenario and Pathway

Table G-4. Dioxin Congener Specific Lifetime Average Daily Dose (LADD) for Each Scenario and Pathway

Lifetime Average Daily Doses (LADDs)																	
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
LCH-fisher																	
Soil dermal	2.70e-16	7.43e-15	1.75e-14	1.34e-14	1.06e-14	1.05e-14	1.07e-13	6.28e-15	1.80e-14	4.13e-15	2.34e-14	7.24e-15	1.52e-14	2.94e-14	3.59e-14	3.17e-13	3.03e-14
Soil ingestion	1.44e-14	3.96e-13	9.35e-13	7.14e-13	5.64e-13	5.59e-13	5.71e-12	3.35e-13	9.62e-13	2.20e-13	1.25e-12	3.86e-13	8.11e-13	1.57e-12	1.91e-12	1.69e-11	1.62e-12
Veget. ingestion	5.76e-14	1.59e-12	3.20e-12	2.54e-12	2.02e-12	1.87e-12	1.98e-11	1.38e-12	3.64e-12	8.54e-13	4.47e-12	1.38e-12	2.90e-12	5.49e-12	6.50e-12	5.74e-11	5.32e-12
Water ingest	4.48e-14	1.24e-12	1.33e-12	1.22e-12	9.65e-13	7.52e-13	8.57e-12	1.21e-12	2.18e-12	5.73e-13	2.14e-12	6.61e-13	1.39e-12	2.04e-12	2.67e-12	2.36e-11	2.11e-12
Fish ingestion	1.37e-11	3.77e-10	4.06e-10	3.09e-10	2.44e-10	3.04e-11	6.19e-12	3.17e-10	9.28e-10	2.11e-10	5.40e-10	1.67e-10	3.51e-10	6.83e-10	1.04e-10	9.17e-10	1.76e-12
Direct inhalation	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
LOH-fisher																	
Soil dermal	5.15e-15	5.34e-15	1.26e-14	8.67e-15	5.60e-15	2.06e-14	2.43e-14	3.28e-15	4.36e-15	3.69e-15	6.17e-15	3.91e-15	5.78e-15	4.84e-15	9.66e-15	1.05e-14	1.06e-14
Soil ingestion	2.75e-13	2.85e-13	6.71e-13	4.62e-13	2.99e-13	1.10e-12	1.29e-12	1.75e-13	2.32e-13	1.97e-13	3.29e-13	2.08e-13	3.08e-13	2.58e-13	5.15e-13	5.59e-13	5.64e-13
Veget. ingestion	1.10e-12	1.14e-12	2.30e-12	1.64e-12	1.07e-12	3.68e-12	4.50e-12	7.21e-13	8.78e-13	7.62e-13	1.18e-12	7.46e-13	1.10e-12	9.02e-13	1.75e-12	1.90e-12	1.86e-12
Water ingest	8.57e-13	8.88e-13	9.56e-13	7.91e-13	5.11e-13	1.48e-12	1.95e-12	6.34e-13	5.27e-13	5.12e-13	5.63e-13	3.57e-13	5.28e-13	3.34e-13	7.18e-13	7.80e-13	7.36e-13
Fish ingestion	2.62e-10	2.71e-10	2.91e-10	2.00e-10	1.29e-10	5.97e-11	1.40e-12	1.65e-10	2.24e-10	1.89e-10	1.42e-10	9.01e-11	1.33e-10	1.12e-10	2.80e-11	3.04e-11	6.14e-13
Direct inhalation	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
SCH-fisher																	
Soil dermal	4.08e-15	1.12e-15	2.65e-15	2.02e-15	1.60e-15	1.59e-15	1.62e-14	9.50e-16	2.72e-15	6.28e-16	3.55e-15	1.10e-15	2.30e-15	4.43e-15	5.43e-15	4.80e-14	4.57e-15
Soil ingestion	2.18e-13	6.00e-14	1.41e-13	1.08e-13	8.53e-14	8.46e-14	8.62e-13	5.07e-14	1.45e-13	3.35e-14	1.89e-13	5.84e-14	1.23e-13	2.36e-13	2.90e-13	2.56e-12	2.44e-13
Veget. ingestion	8.72e-13	2.40e-13	4.84e-13	3.84e-13	3.05e-13	2.83e-13	3.00e-12	2.09e-13	5.50e-13	1.30e-13	6.78e-13	2.09e-13	4.39e-13	8.27e-13	9.84e-13	8.70e-12	8.02e-13

Table G-4. (Continued)

Lifetime Average Daily Doses (LADDs)																
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
Water ingestion	6.79e-13	1.87e-13	2.01e-13	1.85e-13	1.46e-13	1.14e-13	1.30e-12	1.84e-13	3.30e-13	8.70e-14	3.24e-13	1.00e-13	2.10e-13	3.06e-13	4.04e-13	3.57e-12	3.18e-13
Fish ingestion	2.07e-10	5.71e-11	6.14e-11	4.66e-11	3.69e-11	4.60e-12	9.35e-13	4.79e-11	1.40e-10	3.21e-11	8.19e-11	2.53e-11	5.30e-11	1.03e-10	1.57e-11	1.39e-10	2.65e-13
Direct inhalation	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
SOH-fisher																	
Soil dermal	2.84e-16	2.93e-16	6.92e-16	4.77e-16	3.08e-16	1.13e-15	1.33e-15	1.82e-16	2.40e-16	2.03e-16	3.39e-16	2.15e-16	3.19e-16	2.66e-16	5.32e-16	5.75e-16	5.81e-16
Soil ingestion	1.51e-14	1.56e-14	3.69e-14	2.55e-14	1.64e-14	6.05e-14	7.11e-14	9.69e-15	1.28e-14	1.08e-14	1.81e-14	1.15e-14	1.70e-14	1.42e-14	2.84e-14	3.07e-14	3.10e-14
Veget. ingestion	6.06e-14	6.27e-14	1.26e-13	9.06e-14	5.87e-14	2.03e-13	2.47e-13	4.00e-14	4.85e-14	4.20e-14	6.46e-14	4.11e-14	6.08e-14	4.96e-14	9.68e-14	1.04e-13	1.02e-13
Water ingest	4.72e-14	4.88e-14	5.26e-14	4.36e-14	2.81e-14	8.12e-14	1.07e-13	3.51e-14	2.91e-14	2.82e-14	3.09e-14	1.96e-14	2.91e-14	1.84e-14	3.96e-14	4.28e-14	4.04e-14
Fish ingestion	1.44e-11	1.49e-11	1.60e-11	1.10e-11	7.09e-12	3.29e-12	7.71e-14	9.16e-12	1.24e-11	1.04e-11	7.81e-12	4.96e-12	7.35e-12	6.17e-12	1.54e-12	1.67e-12	3.37e-14
Direct inhalation	2.38e-19	2.01e-19	4.20e-19	2.81e-19	1.84e-19	6.68e-19	7.76e-19	1.72e-19	1.69e-19	1.55e-19	2.05e-19	1.30e-19	1.99e-19	1.62e-19	3.18e-19	3.39e-19	3.38e-19
LCH-farmer																	
Soil dermal	1.50e-15	4.13e-14	9.73e-14	7.44e-14	5.87e-14	5.82e-14	5.94e-13	3.49e-14	1.00e-13	2.29e-14	1.30e-13	4.02e-14	8.44e-14	1.64e-13	1.99e-13	1.76e-12	1.68e-13
Soil ingestion	1.44e-14	3.96e-13	9.35e-13	7.14e-13	5.64e-13	5.59e-13	5.71e-12	3.35e-13	9.62e-13	2.20e-13	1.25e-12	3.86e-13	8.11e-13	1.57e-12	1.91e-12	1.69e-11	1.62e-12
Veget. ingestion	1.77e-13	4.88e-12	1.05e-11	8.41e-12	6.49e-12	6.17e-12	6.45e-11	4.20e-12	1.14e-11	2.66e-12	1.44e-11	4.45e-12	9.33e-12	1.75e-11	2.13e-11	1.88e-10	1.76e-11
Animal ingestion	3.12e-12	7.56e-11	9.71e-11	6.97e-11	7.44e-11	2.29e-11	2.03e-10	2.02e-11	1.65e-10	5.21e-11	1.63e-10	4.95e-11	8.99e-11	1.21e-10	6.10e-11	7.26e-10	2.22e-11
Water ingest	4.48e-14	1.24e-12	1.33e-12	1.22e-12	9.65e-13	7.52e-13	8.57e-12	1.21e-12	2.18e-12	5.73e-13	2.14e-12	6.61e-13	1.39e-12	2.04e-12	2.67e-12	2.36e-11	2.11e-12

Table G-4. (Continued)

Lifetime Average Daily Doses (LADDs)																	
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
Direct inhalation	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
LOH-farmer																	
Soil dermal	2.86e-14	2.97e-14	6.99e-14	4.81e-14	3.11e-14	1.14e-13	1.35e-13	1.82e-14	2.42e-14	2.05e-14	3.43e-14	2.17e-14	3.21e-14	2.69e-14	5.36e-14	5.82e-14	5.88e-14
Soil ingestion	2.75e-13	2.85e-13	6.71e-13	4.62e-13	2.99e-13	1.10e-12	1.29e-12	1.75e-13	2.32e-13	1.97e-13	3.29e-13	2.08e-13	3.08e-13	2.58e-13	5.15e-13	5.59e-13	5.64e-13
Veget. ingestion	3.38e-12	3.51e-12	7.51e-12	5.44e-12	3.44e-12	1.21e-11	1.46e-11	2.19e-12	2.76e-12	2.38e-12	3.79e-12	2.40e-12	3.55e-12	2.88e-12	5.74e-12	6.23e-12	6.15e-12
Animal ingestion	5.97e-11	5.44e-11	6.97e-11	4.51e-11	3.94e-11	4.49e-11	4.61e-11	1.05e-11	4.00e-11	4.65e-11	4.29e-11	2.67e-11	3.42e-11	1.98e-11	1.64e-11	2.40e-11	7.75e-12
Water ingest	8.57e-13	8.88e-13	9.56e-13	7.91e-13	5.11e-13	1.48e-12	1.95e-12	6.34e-13	5.27e-13	5.12e-13	5.63e-13	3.57e-13	5.28e-13	3.34e-13	7.18e-13	7.80e-13	7.36e-13
Direct inhalation	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
SCH-farmer																	
Soil dermal	2.27e-14	6.24e-15	1.47e-14	1.12e-14	8.88e-15	8.81e-15	8.98e-14	5.28e-15	1.51e-14	3.48e-15	1.97e-14	6.08e-15	1.28e-14	2.46e-14	3.02e-14	2.67e-13	2.54e-14
Soil ingestion	2.18e-13	6.00e-14	1.41e-13	1.08e-13	8.53e-14	8.46e-14	8.62e-13	5.07e-14	1.45e-13	3.35e-14	1.89e-13	5.84e-14	1.23e-13	2.36e-13	2.90e-13	2.56e-12	2.44e-13
Veget. ingestion	2.68e-12	7.38e-13	1.58e-12	1.27e-12	9.81e-13	9.34e-13	9.74e-12	6.35e-13	1.73e-12	4.04e-13	2.18e-12	6.72e-13	1.41e-12	2.64e-12	3.23e-12	2.85e-11	2.66e-12
Animal ingestion	4.73e-11	1.14e-11	1.47e-11	1.05e-11	1.13e-11	3.46e-12	3.07e-11	3.05e-12	2.50e-11	7.91e-12	2.47e-11	7.48e-12	1.36e-11	1.81e-11	9.24e-12	1.10e-10	3.35e-12
Water ingest	6.79e-13	1.87e-13	2.01e-13	1.85e-13	1.46e-13	1.14e-13	1.30e-12	1.84e-13	3.30e-13	8.70e-14	3.24e-13	1.00e-13	2.10e-13	3.06e-13	4.04e-13	3.57e-12	3.18e-13
Direct inhalation	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
SOH-farmer																	
Soil dermal	1.58e-15	1.63e-15	3.84e-15	2.65e-15	1.71e-15	6.30e-15	7.40e-15	1.01e-15	1.33e-15	1.13e-15	1.88e-15	1.20e-15	1.77e-15	1.48e-15	2.96e-15	3.20e-15	3.22e-15
Soil ingestion	1.51e-14	1.56e-14	3.69e-14	2.55e-14	1.64e-14	6.05e-14	7.11e-14	9.69e-15	1.28e-14	1.08e-14	1.81e-14	1.15e-14	1.70e-14	1.42e-14	2.84e-14	3.07e-14	3.10e-14

Table G-4. (Continued)

Lifetime Average Daily Doses (LADDs)																	
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
Veget. ingestion	1.86e-13	1.93e-13	4.13e-13	3.00e-13	1.89e-13	6.67e-13	8.03e-13	1.21e-13	1.52e-13	1.31e-13	2.08e-13	1.32e-13	1.96e-13	1.58e-13	3.17e-13	3.42e-13	3.37e-13
Animal ingestion	3.29e-12	2.98e-12	3.83e-12	2.49e-12	2.16e-12	2.47e-12	2.53e-12	5.83e-13	2.20e-12	2.56e-12	2.36e-12	1.47e-12	1.88e-12	1.09e-12	9.06e-13	1.32e-12	4.25e-13
Water ingest	4.72e-14	4.88e-14	5.26e-14	4.36e-14	2.81e-14	8.12e-14	1.07e-13	3.51e-14	2.91e-14	2.82e-14	3.09e-14	1.96e-14	2.91e-14	1.84e-14	3.96e-14	4.28e-14	4.04e-14
Direct inhalation	2.38e-19	2.01e-19	4.20e-19	2.81e-19	1.84e-19	6.68e-19	7.76e-19	1.72e-19	1.69e-19	1.55e-19	2.05e-19	1.30e-19	1.99e-19	1.62e-19	3.18e-19	3.39e-19	3.38e-19
LCC-resident																	
Soil dermal	1.03e-17	2.69e-16	6.12e-16	4.64e-16	3.67e-16	3.63e-16	3.69e-15	2.48e-16	6.57e-16	1.54e-16	8.17e-16	2.53e-16	5.35e-16	1.03e-15	1.25e-15	1.10e-14	1.05e-15
Soil ingestion	5.50e-16	1.43e-14	3.27e-14	2.47e-14	1.96e-14	1.94e-14	1.97e-13	1.32e-14	3.50e-14	8.22e-15	4.36e-14	1.35e-14	2.85e-14	5.49e-14	6.65e-14	5.85e-13	5.58e-14
Veget. ingestion	1.38e-15	4.51e-14	7.66e-14	4.33e-14	3.97e-14	3.75e-14	2.96e-13	3.26e-14	8.02e-14	2.13e-14	7.70e-14	2.38e-14	5.58e-14	9.83e-14	6.20e-13	3.16e-12	8.06e-14
Water ingest	1.67e-15	4.36e-14	4.64e-14	4.21e-14	3.33e-14	2.60e-14	2.95e-13	4.64e-14	7.85e-14	2.10e-14	7.41e-14	2.29e-14	4.85e-14	7.11e-14	9.26e-14	8.14e-13	7.28e-14
Direct inhalation	1.42e-17	3.05e-16	5.99e-16	4.34e-16	3.50e-16	3.40e-16	3.40e-15	3.76e-16	7.69e-16	1.96e-16	7.93e-16	2.45e-16	5.41e-16	1.01e-15	1.19e-15	1.03e-14	9.63e-16
LOC-resident																	
Soil dermal	1.85e-16	1.81e-16	4.12e-16	2.82e-16	1.83e-16	6.70e-16	7.87e-16	1.23e-16	1.49e-16	1.29e-16	2.02e-16	1.28e-16	1.91e-16	1.58e-16	3.15e-16	3.41e-16	3.43e-16
Soil ingestion	9.89e-15	9.64e-15	2.20e-14	1.50e-14	9.74e-15	3.57e-14	4.20e-14	6.56e-15	7.92e-15	6.88e-15	1.08e-14	6.82e-15	1.02e-14	8.45e-15	1.68e-14	1.82e-14	1.83e-14
Veget. ingestion	3.18e-14	3.95e-14	6.30e-14	2.95e-14	2.30e-14	8.06e-14	6.71e-14	2.08e-14	2.21e-14	2.28e-14	2.12e-14	1.34e-14	2.30e-14	1.70e-14	2.23e-13	1.34e-13	2.81e-14
Water ingest	3.01e-14	2.93e-14	3.13e-14	2.55e-14	1.66e-14	4.79e-14	6.28e-14	2.31e-14	1.77e-14	1.76e-14	1.83e-14	1.16e-14	1.73e-14	1.09e-14	2.34e-14	2.53e-14	2.39e-14
Direct inhalation	4.39e-16	3.16e-16	5.77e-16	3.70e-16	2.46e-16	8.80e-16	1.01e-15	3.41e-16	2.71e-16	2.68e-16	2.79e-16	1.77e-16	2.80e-16	2.22e-16	4.24e-16	4.48e-16	4.39e-16
SCC-resident																	
Soil dermal	1.76e-16	4.97e-17	1.19e-16	9.10e-17	7.19e-17	7.14e-17	7.29e-16	4.04e-17	1.20e-16	2.74e-17	1.59e-16	4.92e-17	1.03e-16	1.99e-16	2.44e-16	2.16e-15	2.06e-16

Table G-4. (Continued)

Lifetime Average Daily Doses (LADDs)																	
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
Soil ingestion	9.40e-15	2.65e-15	6.34e-15	4.85e-15	3.83e-15	3.81e-15	3.89e-14	2.16e-15	6.40e-15	1.46e-15	8.50e-15	2.62e-15	5.49e-15	1.06e-14	1.30e-14	1.15e-13	1.10e-14
Veget. ingestion	5.99e-14	2.42e-14	3.64e-14	1.39e-14	1.55e-14	1.47e-14	6.66e-14	1.32e-14	3.35e-14	9.92e-15	2.40e-14	7.39e-15	2.05e-14	3.17e-14	5.00e-13	2.30e-12	1.82e-14
Water ingest	2.87e-14	8.07e-15	9.02e-15	8.26e-15	6.52e-15	5.11e-15	5.82e-14	7.61e-15	1.44e-14	3.74e-15	1.45e-14	4.46e-15	9.34e-15	1.37e-14	1.81e-14	1.60e-13	1.43e-14
Direct inhalation	1.30e-15	2.78e-16	5.45e-16	3.94e-16	3.18e-16	3.09e-16	3.08e-15	3.42e-16	6.99e-16	1.79e-16	7.22e-16	2.23e-16	4.92e-16	9.11e-16	1.08e-15	9.35e-15	8.72e-16
SOC-resident																	
Soil dermal	1.82e-17	2.15e-17	5.40e-17	3.78e-17	2.42e-17	8.97e-17	1.06e-16	1.05e-17	1.74e-17	1.40e-17	2.65e-17	1.68e-17	2.46e-17	2.07e-17	4.19e-17	4.55e-17	4.62e-17
Soil ingestion	9.72e-16	1.15e-15	2.88e-15	2.01e-15	1.29e-15	4.78e-15	5.65e-15	5.60e-16	9.26e-16	7.44e-16	1.41e-15	8.98e-16	1.31e-15	1.11e-15	2.23e-15	2.43e-15	2.46e-15
Veget. ingestion	8.96e-15	1.31e-14	1.91e-14	6.60e-15	5.97e-15	2.11e-14	1.13e-14	5.57e-15	6.08e-15	6.80e-15	4.60e-15	2.92e-15	5.72e-15	3.82e-15	9.71e-14	5.45e-14	4.76e-15
Water ingest	2.98e-15	3.49e-15	4.10e-15	3.43e-15	2.20e-15	6.42e-15	8.47e-15	1.99e-15	2.08e-15	1.91e-15	2.40e-15	1.53e-15	2.23e-15	1.43e-15	3.11e-15	3.38e-15	3.21e-15
Direct inhalation	2.05e-16	1.46e-16	2.65e-16	1.70e-16	1.13e-16	4.04e-16	4.61e-16	1.60e-16	1.26e-16	1.25e-16	1.28e-16	8.13e-17	1.29e-16	1.02e-16	1.95e-16	2.05e-16	2.01e-16
LCC-child																	
Soil dermal	2.69e-17	7.02e-16	1.60e-15	1.21e-15	9.60e-16	9.49e-16	9.65e-15	6.48e-16	1.72e-15	4.03e-16	2.14e-15	6.60e-16	1.40e-15	2.69e-15	3.26e-15	2.87e-14	2.73e-15
Soil ingestion	9.06e-16	2.36e-14	5.38e-14	4.07e-14	3.23e-14	3.19e-14	3.24e-13	2.18e-14	5.77e-14	1.35e-14	7.18e-14	2.22e-14	4.70e-14	9.04e-14	1.10e-13	9.63e-13	9.19e-14
Veget. ingestion	4.17e-16	1.45e-14	2.26e-14	1.12e-14	1.10e-14	1.03e-14	6.97e-14	9.76e-15	2.34e-14	6.49e-15	2.00e-14	6.17e-15	1.53e-14	2.58e-14	2.33e-13	1.12e-12	1.86e-14
Water ingest	6.90e-16	1.80e-14	1.92e-14	1.74e-14	1.37e-14	1.07e-14	1.22e-13	1.92e-14	3.24e-14	8.66e-15	3.06e-14	9.46e-15	2.00e-14	2.93e-14	3.82e-14	3.36e-13	3.00e-14
Direct inhalation	9.37e-18	2.01e-16	3.95e-16	2.86e-16	2.31e-16	2.24e-16	2.24e-15	2.48e-16	5.07e-16	1.29e-16	5.22e-16	1.61e-16	3.57e-16	6.63e-16	7.83e-16	6.77e-15	6.34e-16
LOC-child																	
Soil dermal	4.85e-16	4.72e-16	1.08e-15	7.36e-16	4.77e-16	1.75e-15	2.06e-15	3.22e-16	3.88e-16	3.37e-16	5.28e-16	3.34e-16	4.98e-16	4.14e-16	8.23e-16	8.91e-16	8.96e-16
Soil ingestion	1.63e-14	1.59e-14	3.62e-14	2.47e-14	1.60e-14	5.88e-14	6.91e-14	1.08e-14	1.30e-14	1.13e-14	1.77e-14	1.12e-14	1.67e-14	1.39e-14	2.77e-14	2.99e-14	3.01e-14

Table G-4. (Continued)

Lifetime Average Daily Doses (LADDs)																	
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa
Veget. ingestion	1.02e-14	1.33e-14	1.95e-14	7.84e-15	6.64e-15	2.29e-14	1.58e-14	6.64e-15	6.78e-15	7.35e-15	5.65e-15	3.58e-15	6.54e-15	4.60e-15	8.51e-14	4.92e-14	6.48e-15
Water ingest	1.24e-14	1.21e-14	1.29e-14	1.05e-14	6.84e-15	1.98e-14	2.59e-14	9.52e-15	7.32e-15	7.25e-15	7.56e-15	4.78e-15	7.14e-15	4.52e-15	9.65e-15	1.04e-14	9.84e-15
Direct inhalation	2.90e-16	2.08e-16	3.80e-16	2.44e-16	1.62e-16	5.80e-16	6.64e-16	2.25e-16	1.79e-16	1.76e-16	1.84e-16	1.16e-16	1.85e-16	1.46e-16	2.80e-16	2.95e-16	2.89e-16
SCC-child																	
Soil dermal	4.60e-16	1.30e-16	3.11e-16	2.38e-16	1.88e-16	1.87e-16	1.90e-15	1.06e-16	3.14e-16	7.16e-17	4.17e-16	1.28e-16	2.69e-16	5.19e-16	6.38e-16	5.64e-15	5.38e-16
Soil ligest.	1.55e-14	4.37e-15	1.04e-14	8.00e-15	6.31e-15	6.27e-15	6.40e-14	3.55e-15	1.05e-14	2.41e-15	1.40e-14	4.32e-15	9.04e-15	1.75e-14	2.14e-14	1.90e-13	1.81e-14
Veget. ingestion	2.15e-14	8.97e-15	1.28e-14	4.18e-15	5.13e-15	4.84e-15	1.56e-14	4.72e-15	1.17e-14	3.58e-15	7.20e-15	2.22e-15	6.68e-15	9.71e-15	1.97e-13	8.90e-13	4.19e-15
Water ingest	1.18e-14	3.33e-15	3.72e-15	3.41e-15	2.69e-15	2.11e-15	2.40e-14	3.14e-15	5.93e-15	1.54e-15	5.97e-15	1.84e-15	3.85e-15	5.67e-15	7.48e-15	6.62e-14	5.92e-15
Direct inhalation	8.54e-16	1.83e-16	3.59e-16	2.60e-16	2.10e-16	2.04e-16	2.03e-15	2.26e-16	4.61e-16	1.18e-16	4.76e-16	1.47e-16	3.24e-16	6.00e-16	7.12e-16	6.16e-15	5.75e-16
SOC-child																	
Soil dermal	4.76e-17	5.62e-17	1.41e-16	9.87e-17	6.32e-17	2.34e-16	2.77e-16	2.74e-17	4.54e-17	3.65e-17	6.92e-17	4.40e-17	6.42e-17	5.42e-17	1.09e-16	1.19e-16	1.21e-16
Soil ingestion	1.60e-15	1.89e-15	4.74e-15	3.32e-15	2.13e-15	7.88e-15	9.31e-15	9.22e-16	1.53e-15	1.23e-15	2.33e-15	1.48e-15	2.16e-15	1.82e-15	3.68e-15	4.00e-15	4.06e-15
Veget. ingestion	3.29e-15	4.86e-15	6.70e-15	1.98e-15	1.96e-15	6.89e-15	2.63e-15	2.06e-15	2.13e-15	2.49e-15	1.37e-15	8.73e-16	1.86e-15	1.17e-15	3.83e-14	2.11e-14	1.09e-15
Water ingest	1.23e-15	1.44e-15	1.69e-15	1.41e-15	9.06e-16	2.65e-15	3.49e-15	8.20e-16	8.58e-16	7.88e-16	9.92e-16	6.30e-16	9.21e-16	5.92e-16	1.28e-15	1.40e-15	1.33e-15
Direct inhalation	1.35e-16	9.61e-17	1.75e-16	1.12e-16	7.43e-17	2.66e-16	3.04e-16	1.05e-16	8.31e-17	8.20e-17	8.43e-17	5.35e-17	8.52e-17	6.71e-17	1.29e-16	1.35e-16	1.32e-16

Appendix G-5

Dioxin Congener Specific Cancer Risk Results for Each Scenario and Pathway

Table G-5. Dioxin Congener Specific Risk Results for Each Scenario and Pathway

Cancer Risks																		
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total Equivalent
LCH-fisher																		
Soil dermal	4e-11	6e-10	3e-10	2e-10	2e-10	2e-11	2e-11	1e-10	1e-09	3e-11	4e-10	1e-10	2e-10	5e-10	6e-11	5e-10	5e-12	5e-09
Soil ingestion	2e-09	3e-08	1e-08	1e-08	9e-09	9e-10	9e-10	5e-09	8e-08	2e-09	2e-08	6e-09	1e-08	2e-08	3e-09	3e-08	3e-10	2e-07
Vegetable ingestion	9e-09	1e-07	5e-08	4e-08	3e-08	3e-09	3e-09	2e-08	3e-07	7e-09	7e-08	2e-08	5e-08	9e-08	1e-08	9e-08	8e-10	9e-07
Water ingestion	7e-09	1e-07	2e-08	2e-08	2e-08	1e-09	1e-09	2e-08	2e-07	4e-09	3e-08	1e-08	2e-08	3e-08	4e-09	4e-08	3e-10	5e-07
Fish ingestion	2e-06	3e-05	6e-06	5e-06	4e-06	5e-08	1e-09	5e-06	7e-05	2e-06	8e-06	3e-06	5e-06	1e-05	2e-07	1e-06	3e-10	2e-04
Direct inhalation	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00
LOH-fisher																		
Soil dermal	8e-10	4e-10	2e-10	1e-10	9e-11	3e-11	4e-12	5e-11	3e-10	3e-11	1e-10	6e-11	9e-11	8e-11	2e-11	2e-11	2e-12	2e-09
Soil ingestion	4e-08	2e-08	1e-08	7e-09	5e-09	2e-09	2e-10	3e-09	2e-08	2e-09	5e-09	3e-09	5e-09	4e-09	8e-10	9e-10	9e-11	1e-07
Vegetable ingestion	2e-07	9e-08	4e-08	3e-08	2e-08	6e-09	7e-10	1e-08	7e-08	6e-09	2e-08	1e-08	2e-08	1e-08	3e-09	3e-09	3e-10	5e-07
Water ingestion	1e-07	7e-08	1e-08	1e-08	8e-09	2e-09	3e-10	1e-08	4e-08	4e-09	9e-09	6e-09	8e-09	5e-09	1e-09	1e-09	1e-10	3e-07
Fish ingestion	4e-05	2e-05	5e-06	3e-06	2e-06	9e-08	2e-10	3e-06	2e-05	1e-06	2e-06	1e-06	2e-06	2e-06	4e-08	5e-08	1e-10	1e-04
Direct inhalation	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00
SCH-fisher																		
Soil dermal	6e-10	9e-11	4e-11	3e-11	2e-11	2e-12	3e-12	1e-11	2e-10	5e-12	6e-11	2e-11	4e-11	7e-11	8e-12	7e-11	7e-13	1e-09
Soil ingestion	3e-08	5e-09	2e-09	2e-09	1e-09	1e-10	1e-10	8e-10	1e-08	3e-10	3e-09	9e-10	2e-09	4e-09	5e-10	4e-09	4e-11	7e-08
Vegetable ingestion	1e-07	2e-08	8e-09	6e-09	5e-09	4e-10	5e-10	3e-09	4e-08	1e-09	1e-08	3e-09	7e-09	1e-08	2e-09	1e-08	1e-10	3e-07
Water ingestion	1e-07	1e-08	3e-09	3e-09	2e-09	2e-10	2e-10	3e-09	3e-08	7e-10	5e-09	2e-09	3e-09	5e-09	6e-10	6e-09	5e-11	2e-07
Fish ingestion	3e-05	4e-06	1e-06	7e-07	6e-07	7e-09	1e-10	7e-07	1e-05	3e-07	1e-06	4e-07	8e-07	2e-06	2e-08	2e-07	4e-11	6e-05
Direct inhalation	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00
SOH-fisher																		
Soil dermal	4e-11	2e-11	1e-11	7e-12	5e-12	2e-12	2e-13	3e-12	2e-11	2e-12	5e-12	3e-12	5e-12	4e-12	8e-13	9e-13	9e-14	1e-10
Soil ingestion	2e-09	1e-09	6e-10	4e-10	3e-10	9e-11	1e-11	2e-10	1e-09	8e-11	3e-10	2e-10	3e-10	2e-10	4e-11	5e-11	5e-12	7e-09
Vegetable ingestion	9e-09	5e-09	2e-09	1e-09	9e-10	3e-10	4e-11	6e-10	4e-09	3e-10	1e-09	6e-10	9e-10	8e-10	2e-10	2e-10	2e-11	3e-08

Table G-5. (Continued)

Cancer Risks																		
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total Equivalent
Water ingestion	7e-09	4e-09	8e-10	7e-10	4e-10	1e-10	2e-11	5e-10	2e-09	2e-10	5e-10	3e-10	5e-10	3e-10	6e-11	7e-11	6e-12	2e-08
Fish ingestion	2e-06	1e-06	2e-07	2e-07	1e-07	5e-09	1e-11	1e-07	1e-06	8e-08	1e-07	8e-08	1e-07	1e-07	2e-09	3e-09	5e-12	6e-06
Direct inhalation	4e-14	2e-14	7e-15	4e-15	3e-15	1e-15	1e-16	3e-15	1e-14	1e-15	3e-15	2e-15	3e-15	3e-15	5e-16	5e-16	5e-17	1e-13
LCH-farmer																		
Soil dermal	2e-10	3e-09	2e-09	1e-09	9e-10	9e-11	9e-11	5e-10	8e-09	2e-10	2e-09	6e-10	1e-09	3e-09	3e-10	3e-09	3e-11	3e-08
Soil ingestion	2e-09	3e-08	1e-08	1e-08	9e-09	9e-10	9e-10	5e-09	8e-08	2e-09	2e-08	6e-09	1e-08	2e-08	3e-09	3e-08	3e-10	2e-07
Vegetable ingestion	3e-08	4e-07	2e-07	1e-07	1e-07	1e-08	1e-08	7e-08	9e-07	2e-08	2e-07	7e-08	1e-07	3e-07	3e-08	3e-07	3e-09	3e-06
Animal ingestion	5e-07	6e-06	2e-06	1e-06	1e-06	4e-08	3e-08	3e-07	1e-05	4e-07	3e-06	8e-07	1e-06	2e-06	1e-07	1e-06	3e-09	3e-05
Water ingestion	7e-09	1e-07	2e-08	2e-08	2e-08	1e-09	1e-09	2e-08	2e-07	4e-09	3e-08	1e-08	2e-08	3e-08	4e-09	4e-08	3e-10	5e-07
Direct inhalation	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00
LOH-farmer																		
Soil dermal	4e-09	2e-09	1e-09	8e-10	5e-10	2e-10	2e-11	3e-10	2e-09	2e-10	5e-10	3e-10	5e-10	4e-10	8e-11	9e-11	9e-12	1e-08
Soil ingestion	4e-08	2e-08	1e-08	7e-09	5e-09	2e-09	2e-10	3e-09	2e-08	2e-09	5e-09	3e-09	5e-09	4e-09	8e-10	9e-10	9e-11	1e-07
Vegetable ingestion	5e-07	3e-07	1e-07	8e-08	5e-08	2e-08	2e-09	3e-08	2e-07	2e-08	6e-08	4e-08	6e-08	4e-08	9e-09	1e-08	1e-09	2e-06
Animal ingestion	9e-06	4e-06	1e-06	7e-07	6e-07	7e-08	7e-09	2e-07	3e-06	4e-07	7e-07	4e-07	5e-07	3e-07	3e-08	4e-08	1e-09	2e-05
Water ingestion	1e-07	7e-08	1e-08	1e-08	8e-09	2e-09	3e-10	1e-08	4e-08	4e-09	9e-09	6e-09	8e-09	5e-09	1e-09	1e-09	1e-10	3e-07
Direct inhalation	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00
SCH-farmer																		
Soil dermal	4e-09	5e-10	2e-10	2e-10	1e-10	1e-11	1e-11	8e-11	1e-09	3e-11	3e-10	9e-11	2e-10	4e-10	5e-11	4e-10	4e-12	7e-09
Soil ingestion	3e-08	5e-09	2e-09	2e-09	1e-09	1e-10	1e-10	8e-10	1e-08	3e-10	3e-09	9e-10	2e-09	4e-09	5e-10	4e-09	4e-11	7e-08
Vegetable ingestion	4e-07	6e-08	2e-08	2e-08	2e-08	1e-09	2e-09	1e-08	1e-07	3e-09	3e-08	1e-08	2e-08	4e-08	5e-09	4e-08	4e-10	8e-07
Animal ingestion	7e-06	9e-07	2e-07	2e-07	2e-07	5e-09	5e-09	5e-08	2e-06	6e-08	4e-07	1e-07	2e-07	3e-07	1e-08	2e-07	5e-10	1e-05
Water ingestion	1e-07	1e-08	3e-09	3e-09	2e-09	2e-10	2e-10	3e-09	3e-08	7e-10	5e-09	2e-09	3e-09	5e-09	6e-10	6e-09	5e-11	2e-07

Table G-5. (Continued)

Cancer Risks																		
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total Equivalent
Direct inhalation	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00	0e+00
SOH-farmer																		
Soil dermal	2e-10	1e-10	6e-11	4e-11	3e-11	1e-11	1e-12	2e-11	1e-10	9e-12	3e-11	2e-11	3e-11	2e-11	5e-12	5e-12	5e-13	7e-10
Soil ingestion	2e-09	1e-09	6e-10	4e-10	3e-10	9e-11	1e-11	2e-10	1e-09	8e-11	3e-10	2e-10	3e-10	2e-10	4e-11	5e-11	5e-12	7e-09
Vegetable ingestion	3e-08	2e-08	6e-09	5e-09	3e-09	1e-09	1e-10	2e-09	1e-08	1e-09	3e-09	2e-09	3e-09	2e-09	5e-10	5e-10	5e-11	9e-08
Animal ingestion	5e-07	2e-07	6e-08	4e-08	3e-08	4e-09	4e-10	9e-09	2e-07	2e-08	4e-08	2e-08	3e-08	2e-08	1e-09	2e-09	7e-11	1e-06
Water ingestion	7e-09	4e-09	8e-10	7e-10	4e-10	1e-10	2e-11	5e-10	2e-09	2e-10	5e-10	3e-10	5e-10	3e-10	6e-11	7e-11	6e-12	2e-08
Direct inhalation	4e-14	2e-14	7e-15	4e-15	3e-15	1e-15	1e-16	3e-15	1e-14	1e-15	3e-15	2e-15	3e-15	3e-15	5e-16	5e-16	5e-17	1e-13
LCC-resident																		
Soil dermal	2e-12	2e-11	1e-11	7e-12	6e-12	6e-13	6e-13	4e-12	5e-11	1e-12	1e-11	4e-12	8e-12	2e-11	2e-12	2e-11	2e-13	2e-10
Soil ingestion	9e-11	1e-09	5e-10	4e-10	3e-10	3e-11	3e-11	2e-10	3e-09	6e-11	7e-10	2e-10	4e-10	9e-10	1e-10	9e-10	9e-12	9e-09
Vegetable ingestion	2e-10	4e-09	1e-09	7e-10	6e-10	6e-11	5e-11	5e-10	6e-09	2e-10	1e-09	4e-10	9e-10	2e-09	1e-09	5e-09	1e-11	2e-08
Water ingestion	3e-10	3e-09	7e-10	7e-10	5e-10	4e-11	5e-11	7e-10	6e-09	2e-10	1e-09	4e-10	8e-10	1e-09	1e-10	1e-09	1e-11	2e-08
Direct inhalation	2e-12	2e-11	9e-12	7e-12	5e-12	5e-13	5e-13	6e-12	6e-11	2e-12	1e-11	4e-12	8e-12	2e-11	2e-12	2e-11	2e-13	2e-10
LOC-resident																		
Soil dermal	3e-11	1e-11	6e-12	4e-12	3e-12	1e-12	1e-13	2e-12	1e-11	1e-12	3e-12	2e-12	3e-12	2e-12	5e-13	5e-13	5e-14	8e-11
Soil ingestion	2e-09	8e-10	3e-10	2e-10	2e-10	6e-11	7e-12	1e-10	6e-10	5e-11	2e-10	1e-10	2e-10	1e-10	3e-11	3e-11	3e-12	4e-09
Vegetable ingestion	5e-09	3e-09	1e-09	5e-10	4e-10	1e-10	1e-11	3e-10	2e-09	2e-10	3e-10	2e-10	4e-10	3e-10	3e-10	2e-10	4e-12	1e-08
Water ingestion	5e-09	2e-09	5e-10	4e-10	3e-10	7e-11	1e-11	4e-10	1e-09	1e-10	3e-10	2e-10	3e-10	2e-10	4e-11	4e-11	4e-12	1e-08
Direct inhalation	7e-11	2e-11	9e-12	6e-12	4e-12	1e-12	2e-13	5e-12	2e-11	2e-12	4e-12	3e-12	4e-12	3e-12	7e-13	7e-13	7e-14	2e-10
SCC-resident																		
Soil dermal	3e-11	4e-12	2e-12	1e-12	1e-12	1e-13	1e-13	6e-13	9e-12	2e-13	2e-12	8e-13	2e-12	3e-12	4e-13	3e-12	3e-14	6e-11
Soil ingestion	1e-09	2e-10	1e-10	8e-11	6e-11	6e-12	6e-12	3e-11	5e-10	1e-11	1e-10	4e-11	9e-11	2e-10	2e-11	2e-10	2e-12	3e-09
Vegetable ingestion	9e-09	2e-09	6e-10	2e-10	2e-10	2e-11	1e-11	2e-10	3e-09	8e-11	4e-10	1e-10	3e-10	5e-10	8e-10	4e-09	3e-12	2e-08

Table G-5. (Continued)

Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total Equivalent
Cancer Risks																		
Water ingestion	4e-09	6e-10	1e-10	1e-10	1e-10	8e-12	9e-12	1e-10	1e-09	3e-11	2e-10	7e-11	1e-10	2e-10	3e-11	3e-10	2e-12	8e-09
Direct inhalation	2e-10	2e-11	8e-12	6e-12	5e-12	5e-13	5e-13	5e-12	5e-11	1e-12	1e-11	3e-12	8e-12	1e-11	2e-12	1e-11	1e-13	4e-10
SOC-resident																		
Soil dermal	3e-12	2e-12	8e-13	6e-13	4e-13	1e-13	2e-14	2e-13	1e-12	1e-13	4e-13	3e-13	4e-13	3e-13	7e-14	7e-14	7e-15	1e-11
Soil ingestion	2e-10	9e-11	4e-11	3e-11	2e-11	7e-12	9e-13	9e-12	7e-11	6e-12	2e-11	1e-11	2e-11	2e-11	3e-12	4e-12	4e-13	5e-10
Vegetable ingestion	1e-09	1e-09	3e-10	1e-10	9e-11	3e-11	2e-12	9e-11	5e-10	5e-11	7e-11	5e-11	9e-11	6e-11	2e-10	8e-11	7e-13	4e-09
Water ingestion	5e-10	3e-10	6e-11	5e-11	3e-11	1e-11	1e-12	3e-11	2e-10	1e-11	4e-11	2e-11	3e-11	2e-11	5e-12	5e-12	5e-13	1e-09
Direct inhalation	3e-11	1e-11	4e-12	3e-12	2e-12	6e-13	7e-14	2e-12	1e-11	1e-12	2e-12	1e-12	2e-12	2e-12	3e-13	3e-13	3e-14	7e-11
LCC-child																		
Soil dermal	4e-12	5e-11	2e-11	2e-11	1e-11	1e-12	2e-12	1e-11	1e-10	3e-12	3e-11	1e-11	2e-11	4e-11	5e-12	4e-11	4e-13	4e-10
Soil ingestion	1e-10	2e-09	8e-10	6e-10	5e-10	5e-11	5e-11	3e-10	5e-09	1e-10	1e-09	3e-10	7e-10	1e-09	2e-10	2e-09	1e-11	1e-08
Vegetable ingestion	7e-11	1e-09	4e-10	2e-10	2e-10	2e-11	1e-11	2e-10	2e-09	5e-11	3e-10	1e-10	2e-10	4e-10	4e-10	2e-09	3e-12	7e-09
Water ingestion	1e-10	1e-09	3e-10	3e-10	2e-10	2e-11	2e-11	3e-10	3e-09	7e-11	5e-10	1e-10	3e-10	5e-10	6e-11	5e-10	5e-12	7e-09
Direct inhalation	1e-12	2e-11	6e-12	4e-12	4e-12	3e-13	3e-13	4e-12	4e-11	1e-12	8e-12	3e-12	6e-12	1e-11	1e-12	1e-11	1e-13	1e-10
LOC-child																		
Soil dermal	8e-11	4e-11	2e-11	1e-11	7e-12	3e-12	3e-13	5e-12	3e-11	3e-12	8e-12	5e-12	8e-12	6e-12	1e-12	1e-12	1e-13	2e-10
Soil ingestion	3e-09	1e-09	6e-10	4e-10	3e-10	9e-11	1e-11	2e-10	1e-09	9e-11	3e-10	2e-10	3e-10	2e-10	4e-11	5e-11	5e-12	7e-09
Vegetable ingestion	2e-09	1e-09	3e-10	1e-10	1e-10	4e-11	2e-12	1e-10	5e-10	6e-11	9e-11	6e-11	1e-10	7e-11	1e-10	8e-11	1e-12	4e-09
Water ingestion	2e-09	9e-10	2e-10	2e-10	1e-10	3e-11	4e-12	1e-10	6e-10	6e-11	1e-10	7e-11	1e-10	7e-11	2e-11	2e-11	2e-12	5e-09
Direct inhalation	5e-11	2e-11	6e-12	4e-12	3e-12	9e-13	1e-13	4e-12	1e-11	1e-12	3e-12	2e-12	3e-12	2e-12	4e-13	5e-13	5e-14	1e-10
SCC-child																		
Soil dermal	7e-11	1e-11	5e-12	4e-12	3e-12	3e-13	3e-13	2e-12	2e-11	6e-13	6e-12	2e-12	4e-12	8e-12	1e-12	9e-12	8e-14	2e-10
Soil ingestion	2e-09	3e-10	2e-10	1e-10	1e-10	1e-11	1e-11	6e-11	8e-10	2e-11	2e-10	7e-11	1e-10	3e-10	3e-11	3e-10	3e-12	5e-09
Vegetable ingestion	3e-09	7e-10	2e-10	7e-11	8e-11	8e-12	2e-12	7e-11	9e-10	3e-11	1e-10	3e-11	1e-10	2e-10	3e-10	1e-09	7e-13	8e-09

Table G-5. (Continued)

Cancer Risks																		
Congener	PCDDs 2378	12378	123478	123789	123678	1234678	Octa	PCDFs 2378	23478	12378	123478	123678	123789	234678	1234678	1234789	Octa	Total Equivalent
Water ingestion	2e-09	3e-10	6e-11	5e-11	4e-11	3e-12	4e-12	5e-11	5e-10	1e-11	9e-11	3e-11	6e-11	9e-11	1e-11	1e-10	9e-13	3e-09
Direct inhalation	1e-10	1e-11	6e-12	4e-12	3e-12	3e-13	3e-13	4e-12	4e-11	9e-13	7e-12	2e-12	5e-12	9e-12	1e-12	1e-11	9e-14	2e-10
SOC-child																		
Soil dermal	7e-12	4e-12	2e-12	2e-12	1e-12	4e-13	4e-14	4e-13	4e-12	3e-13	1e-12	7e-13	1e-12	8e-13	2e-13	2e-13	2e-14	3e-11
Soil ingestion	2e-10	1e-10	7e-11	5e-11	3e-11	1e-11	1e-12	1e-11	1e-10	1e-11	4e-11	2e-11	3e-11	3e-11	6e-12	6e-12	6e-13	8e-10
Vegetable ingestion	5e-10	4e-10	1e-10	3e-11	3e-11	1e-11	4e-13	3e-11	2e-10	2e-11	2e-11	1e-11	3e-11	2e-11	6e-11	3e-11	2e-13	1e-09
Water ingestion	2e-10	1e-10	3e-11	2e-11	1e-11	4e-12	5e-13	1e-11	7e-11	6e-12	2e-11	1e-11	1e-11	9e-12	2e-12	2e-12	2e-13	5e-10
Direct inhalation	2e-12	7e-12	3e-12	2e-12	1e-12	4e-13	5e-14	2e-12	6e-12	6e-13	1e-12	8e-13	1e-12	1e-12	2e-13	2e-13	2e-14	5e-11

Appendix G-6

Comparison of Dioxin Local-Scale ISCST3 and Long-Range RELMAP Modeling Exposures and Risks

Table G-6. Comparison of Local-Scale ISCST3 and Long-Range RELMAP Modeling Exposures and Risks for Dioxin

	Total TEQ exposures (LADD in mg/kg-d)	Total TEQ Risk	RELMAP % of Total (mean data)
LCH-fisher			
Soil Dermal	2.9e-14	5e-09	0.14%
Soil Ingestion	1.6e-12	2e-07	0.14%
Vegetation Ingestion	6.0e-12	9e-07	4.92%
Water Ingestion	3.2e-12	5e-07	0.22%
Fish Ingestion	9.9e-10	2e-04	0.22%
Direct Inhalation	1.8e-15	3e-10	100.00%
LOH-fisher			
Soil Dermal	1.6e-14	2e-09	0.26%
Soil Ingestion	8.4e-13	1e-07	0.26%
Vegetation Ingestion	3.5e-12	5e-07	8.50%
Water Ingestion	2.1e-12	3e-07	0.33%
Fish Ingestion	6.5e-10	1e-04	0.33%
Direct Inhalation	1.8e-15	3e-10	100.00%
SCH-fisher			
Soil Dermal	8.5e-15	1e-09	0.48%
Soil Ingestion	4.5e-13	7e-08	0.48%
Vegetation Ingestion	2.0e-12	3e-07	14.64%
Water Ingestion	1.2e-12	2e-07	0.61%
Fish Ingestion	3.6e-10	6e-05	0.60%
Direct Inhalation	1.8e-15	3e-10	100.00%
SOH-fisher			
Soil Dermal	9.1e-16	1e-10	4.48%
Soil Ingestion	4.8e-14	8e-09	4.48%
Vegetation Ingestion	4.7e-13	7e-08	62.78%
Water Ingestion	1.2e-13	2e-08	5.75%
Fish Ingestion	3.8e-11	6e-06	5.68%
Direct Inhalation	1.8e-15	3e-10	99.97%
LCH-farmer			
Soil Dermal	1.6e-13	3e-08	0.14%
Soil Ingestion	1.6e-12	2e-07	0.14%
Vegetation Ingestion	1.9e-11	3e-06	2.48%
Animal Product Ingestion	2.0e-10	3e-05	0.85%
Water Ingestion	3.2e-12	5e-07	0.22%
Direct Inhalation	1.8e-15	3e-10	100.00%

Table G-6. (Continued)

	Total TEQ exposures (LADD in mg/kg-d)	**Total TEQ Risk**	**RELMAP % of Total (mean data)**
LOH-farmer			
Soil Dermal	8.7e-14	1e-08	0.26%
Soil Ingestion	8.4e-13	1e-07	0.26%
Vegetation Ingestion	1.0e-11	2e-06	4.42%
Animal Product Ingestion	1.4e-10	2e-05	1.24%
Water Ingestion	2.1e-12	3e-07	0.33%
Direct Inhalation	1.8e-15	3e-10	100.00%
SCH-farmer			
Soil Dermal	4.7e-14	7e-09	0.48%
Soil Ingestion	4.5e-13	7e-08	0.48%
Vegetation Ingestion	5.9e-12	9e-07	7.89%
Animal Product Ingestion	7.9e-11	1e-05	2.20%
Water Ingestion	1.2e-12	2e-07	0.61%
Direct Inhalation	1.8e-15	3e-10	100.00%
SOH-farmer			
Soil Dermal	5.0e-15	8e-10	4.48%
Soil Ingestion	4.8e-14	8e-09	4.48%
Vegetation Ingestion	1.0e-12	2e-07	45.66%
Animal Product Ingestion	9.4e-12	1e-06	18.54%
Water Ingestion	1.2e-13	2e-08	5.75%
Direct Inhalation	1.8e-15	3e-10	99.97%
LCC-resident			
Soil Dermal	1.1e-15	2e-10	3.74%
Soil Ingestion	5.8e-14	9e-09	3.74%
Vegetation Ingestion	2.4e-13	4e-08	37.37%
Water Ingestion	1.2e-13	2e-08	5.76%
Direct Inhalation	2.9e-15	5e-10	61.32%
LOC-resident			
Soil Dermal	5.8e-16	9e-11	7.01%
Soil Ingestion	3.1e-14	5e-09	7.01%
Vegetation Ingestion	1.8e-13	3e-08	49.78%
Water Ingestion	7.8e-14	1e-08	8.80%
Direct Inhalation	2.8e-15	4e-10	63.60%
SCC-resident			
Soil Dermal	4.1e-16	6e-11	9.85%

Table G-6. (Continued)

	Total TEQ exposures (LADD in mg/kg-d)	Total TEQ Risk	RELMAP % of Total (mean data)
Soil Ingestion	2.2e-14	3e-09	9.85%
Vegetation Ingestion	2.2e-13	3e-08	39.83%
Water Ingestion	5.6e-14	9e-09	12.19%
Direct Inhalation	4.1e-15	6e-10	43.52%
SOC-resident			
Soil Dermal	1.0e-16	2e-11	39.65%
Soil Ingestion	5.5e-15	9e-10	39.65%
Vegetation Ingestion	1.1e-13	2e-08	77.27%
Water Ingestion	1.5e-14	2e-09	46.34%
Direct Inhalation	2.2e-15	3e-10	79.02%
LCC-child			
Soil Dermal	2.8e-15	4e-10	3.74%
Soil Ingestion	9.5e-14	1e-08	3.74%
Vegetation Ingestion	8.1e-14	1e-08	43.34%
Water Ingestion	4.9e-14	8e-09	5.76%
Direct Inhalation	1.9e-15	3e-10	61.32%
LOC-child			
Soil Dermal	1.5e-15	2e-10	7.01%
Soil Ingestion	5.1e-14	8e-09	7.01%
Vegetation Ingestion	6.3e-14	1e-08	55.25%
Water Ingestion	3.2e-14	5e-09	8.80%
Direct Inhalation	1.8e-15	3e-10	63.60%
SCC-child			
Soil Dermal	1.1e-15	2e-10	9.85%
Soil Ingestion	3.6e-14	6e-09	9.85%
Vegetation Ingestion	8.3e-14	1e-08	42.00%
Water Ingestion	2.3e-14	4e-09	12.19%
Direct Inhalation	2.7e-15	4e-10	43.52%
SOC-child			
Soil Dermal	2.7e-16	4e-11	39.65%
Soil Ingestion	9.0e-15	1e-09	39.65%
Vegetation Ingestion	4.4e-14	7e-09	78.82%
Water Ingestion	6.1e-15	1e-09	46.34%
Direct Inhalation	1.5e-15	2e-10	79.02%

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Appendix H - Literature Review of the Potential Impacts of Hydrogen Chloride and Hydrogen Fluoride

H.1 OVERVIEW

The information presented in this appendix was collected to expand the EPA's knowledge of the potential impacts of HCl and HF emissions from utilities. The EPA is updating its current state of knowledge of health impacts (including dose/response relationships); atmospheric chemistry (e.g., half-life, impacts on the acid rain phenomenon); potential human exposure through pathways other than direct inhalation; and possible ecological harm. The EPA’s goal is to understand the potential impacts from HCl and HF emissions to any and all health and environmental areas. This appendix is not intended to provide a detailed, comprehensive treatise on the above subject area; rather, it is designed to provide general technical information that will identify possible problem areas that may call for additional, more detailed research.

Published evidence for potential impacts of HCl and HF was evaluated from a wide variety of sources. Overall, there is extensive information available on the toxicology of these two pollutants; however, literature pertaining specifically to HF and HCl atmospheric chemistry is relatively scarce, especially that pertaining to fine particulate matter and acid rain. Literature on HCl and HF from sources outside the United States and pertaining to emissions sources other than utilities has also been evaluated.

This appendix is organized so that the findings for HCl are presented first, followed by the findings for HF. Within each section, evidence from the literature for transport and transformation through atmospheric, terrestrial, and aquatic processes is presented first, followed by evidence for impacts on human health; vegetation; and wild, domestic, and aquatic animals.

H.2 FINDINGS FOR HYDROGEN CHLORIDE

H.2.1 HCl Emissions and Formation

The information on nationwide utility HCl emissions was obtained from Table 3-3 in Chapter 3 of this report (1990 estimate). Emissions are reported to be 148,000 tons/year for coal and natural gas boilers combined. Utility emissions are the most significant anthropogenic source of atmospheric HCl. Other important sources are industrial coal combustion, and solid waste combustion.

Atmospheric HCl is emitted by both natural and anthropogenic sources. For instance, anthropogenic sources contributing to measured concentrations of HCl in an urban area of Switzerland were found to include automobiles, heating units, and a garbage incinerator.[1] Wegner, et al. cite coal combustion and waste incineration as the main anthropogenic HCl sources.[2] Puxbaum, et al. cite coal combustion as the primary source of HCl in central Europe.[3] Other sources include biomass burning and the photolysis of Cl-atom precursors such as HCl and Cl_2, followed by hydrocarbon reactions.[4] Natural sources of HCl emissions include volcanic activity, marine plants/microorganisms, land plant combustion-generated methyl chloride, and sea-salt reactions.[5]

Graedel, et al. estimated the global acid-equivalent fluxes and reported on observations for the period 1977-1990.[6] They predicted HCl emissions growth to the year 2100. The dominant global source of atmospheric HCl is believed to be marine production by direct volatilization from deliquescent sea-salt aerosol that has been acidified by the incorporation of HNO_3 and/or H_2SO_4. Total global emissions of HCl are estimated at 55 Tg Cl per year, in a year with average volcanic activity. Acid-equivalent fluxes are calculated to be 2.0 Teq H^+ per year for SO_x (83 percent anthropogenic), 2.2 Teq H^+ per year for NO_x (57 percent anthropogenic), and 1.6 Teq H^+ per year for HCl. However, because most of the HCl is thought to be generated by acid-displacement reactions involving anthropogenically derived precursors, much of this HCl does not correspond to a net production of atmospheric acidity. Thus, the net influence of this acidity is already accounted for in the sum of SO_x and NO_x. Because SO_x and NO_x emissions as a proxy for HCl emissions decreased from 1975 to 1995 in more developed countries, the 1 to 11 percent per year increase in HCl concentrations observed during the 1977-1990 period is believed to be from enhanced volatilization of sea salt. Interestingly, Graedel, et al. predict that HCl emissions will grow from an estimated 55 Tg chlorine per year in 1990 to 158 Tg chlorine per year in 2100, and that acid-equivalent emissions will more than double in this period due to development.

HCl emissions are believed to be the third largest source of anthropogenic atmospheric acidity. In the United Kingdom, HCl emissions sources are coal-fired boilers, waste incineration, chlorinated hydrocarbons, automobile exhaust, glass-making, fuel oil combustion, steel pickling acid, and regeneration.[5] Coal burning, however, is responsible for 93 percent of total HCl emissions in the United Kingdom, with waste incineration emitting another 6 percent. All other sources combined emit the remaining 1 percent of HCl. HCl contributes only 4 percent of the United Kingdom's potential atmospheric acidity, while 71 percent and 25 percent are attributed to SO_2 and NO_x, respectively. When evaluated at the scale of Western Europe, the contribution by HCl to atmospheric potential acidity is estimated at 2 percent.

The past, current, and future quality and availability of emissions inventories for acid-related compounds was evaluated by Graedel, et al.[6] Information available on atmospheric fluxes of HCl was determined to be of poor quality. Because of the poor availability and quality of global emissions inventories of atmospheric acid-related compounds determined by a survey in 1992, the Global Emissions Inventory Activity (GEIA) was introduced under the auspices of the International Global Atmospheric Chemistry Project. GEIA was developed with the goal of establishing a framework for the development and evaluation of global emissions inventories, along with the generation and publication of inventories for use by the global science and policy communities. In the future, GEIA inventories are expected to be a significant aid to the characterization of global emissions of atmospheric species.

Information on ambient concentrations of HCl was relatively scarce, particularly for the United States. This review includes information on both urban and rural concentrations, as available.

Kelly, et al. reviewed concentrations and transformations of Hazardous Air Pollutants (HAPs). They noted that, as of August 1994, eight HCl monitoring stations had been operated in the United States and over 74 samples had been measured,[7] ranging from none detected to 4 μg/m^3.

Wegner, et al. found tropospheric HCl concentrations to vary substantially as a function of location and time, and to be strongly correlated with CH_2O concentrations ® = 0.93, 0.90, and 0.95 for polar, midlatitude maritime, and midlatitude continental, respectively).[2] Average HCl concentrations measured in a tropospheric column were 1.15 x 10^{15} molecules per square centimeter. The concentrations of HCl found in the three regions were as follows: continental maritime < midlatitude maritime < polar maritime. The reason posited for the highest HCl concentrations being observed in polar maritime air is high reaction rates of non-methane hydrocarbons with Cl atoms, yielding HCl as a reaction product. There was no correlation between HF and HCl concentrations.

High intermittent concentrations of HCl observed in an urban environment in Switzerland (up to 3.2 μg/m^3) were believed to originate largely from an incinerator located approximately 3 km away from the monitor.[1] Hutchinson, et al., in a paper on HCl-induced stone degradation, estimate typical atmospheric HCl concentrations ranges in North America of 0.1-1.4 μg/m^3 in rural areas and 0.2-3 μg/m^3 in urban areas.[8] Puxbaum, et al. measured HCl and several other chemical species at a rural site in northeast Austria.[3] HCl concentrations exhibited substantial variation between winter and summer, with elevated values of 0.7 μg/m^3 found in winter. Annual average HCl concentrations were 0.3 μg/m^3.

The ambient concentrations of HCl will be of interest as part of the effort to achieve compliance with the new particulate matter standards. If a significant portion of ambient HCl is in the fine fraction, it could conceivably contribute to $PM_{2.5}$ exceedances. The remainder of this section addresses the topics of HCl formation and formation by-products.

HCl can be formed several ways in the atmosphere. The burning of coal can yield HCl as a combustion product, with the quantity of HCl emitted in this manner being a function of coal composition, method of combustion, and air pollution control methods.[9] A study conducted by the U.S. Bureau of Mines indicated that the majority of chlorine contained in coal volatilizes to form HCl.[10] Additional tests found that only small amounts of chlorine remain in the combustion ash.

Other studies reveal the processes of chlorine and HCl formation in anthropogenic and natural systems.

Nonanthropogenic HCl is emitted by volcanoes, or it can be formed from deliquescent sea salt in the marine environment by the following process:[4]

$$HNO_{3(g)} + NaCl_{(s)} \rightarrow HCl_{(g)} + NaNO_{3(s)} \qquad (1)$$

$$H_2SO_{4(s)} + 2NaCl_{(s)} \rightarrow 2HCl_{(g)} + Na_2SO_{4(s)} \qquad (2)$$

Thus, sulfuric acid and nitric acid in the atmosphere can react with sea salt spray to generate hydrogen chloride. HCl in marine environments can also be formed by indirect pathways, which generate various chlorinated species such as $ClNO_2$ or Cl_2. Once this happens, photolysis of the chlorinated species will produce chlorine radicals, which react as follows to produce HCl:

$$Cl\bullet + RH \rightarrow HCl + R\bullet \qquad (3)$$

Rupert and Sigg investigated the interaction between fogwater and aerosols, which can create or destroy HCl.[1] The following reaction was found to account for the transition of chlorine between the aerosol and gaseous phase:

$$NH_4Cl_{aerosol} \rightleftarrows NH_{3(g)} + HCl_{(g)} \qquad (4)$$

In their study of fogwater and aerosols in an urban area of Switzerland, Ruprecht and Sigg found empirical values of NH_4Cl to be lower than theoretical values. One possible explanation is that insufficient amounts of NH_4Cl were available to sustain equilibrium concentrations. HCl was observed to dissolve in fogwater, causing high aqueous concentrations, which were subsequently released in the gas phase upon fog dissipation. It is thought that concentrations of gaseous NH_3 must have been too low to neutralize the HCl present, hindering the formation of NH_4Cl. A significant portion of atmospheric Cl^- is present as fine aerosol (<2.4 μm) under fog-free conditions. Fogwater acts as an ephemeral sink and possible reaction environment for HCl and other soluble gases.

Another source of atmospheric HCl is anthropogenic chlorocarbons, which can react principally with OH radicals to produce HCl, with a reaction rate of approximately 0.5 percent per day.[5] However, due to this slow reaction rate, which permits widespread dispersion of the HCl produced, it is believed that chlorocarbon-produced HCl emissions contribute a negligible amount to atmospheric acidity.

In their review of atmospheric transformations of HAPs, Spicer, et al. describe the identification of HCl reaction products (i.e., chloride salts) as qualitative in most cases, because few studies reported mass balance information.[11]

Based on thermodynamic equilibrium calculations performed in the U.S. Bureau of Mines study previously discussed, coal combustion can generate small amounts of other gaseous chlorine compounds, including Cl_2, HOCl, and NOCl.[10] Analysis during laboratory simulation did not, however, detect the presence of these compounds; the emissions were virtually all HCl.

In summary, HCl can be formed during coal combustion, waste incineration, the reaction and purification steps in the propylene oxide manufacturing process, hydrocarbon chlorination and dehydrochlorination, and combustion of the chlorinated hydrocarbons found in some gasolines. Nonanthropogenic HCl is formed from deliquescent sea salt in the marine environment or emitted by volcanoes. HCl can also be created or destroyed through the interaction between fogwater and aerosols. Reactions generating HCl can produce the following by-products in the atmosphere: $NaNO_3$, Na_2SO_4, hydrocarbon radicals, and NH_3.

H.2.2 HCl Atmospheric Processes

In a review of the stability and persistence of atmospheric HCl, two references were found that dealt directly with the atmospheric lifetime of HCl. No information was reviewed on the stability and persistence of HCl by-products.

In their review paper on atmospheric transformation of HAPs, Spicer, et al. estimate the atmospheric lifetime of HCl to be between 1 and 5 days.[11] Lifetime estimates are defined as the time required for a given HAP's concentration to decrease to 1/e (37 percent) of its original value, via atmospheric reaction or removal. Of the 178 chemicals on which lifetime estimates were obtained, 83 had lifetimes of less than one day, 25 had lifetimes in the 1-5 day category, and 57 had lifetimes greater than 5 days. Thirteen of the chemicals had conflicting estimates of lifetimes. Lifetime estimates were described as relative rather than absolute estimates of HAP transformation lifetimes, because of the varying information sources and calculation methods. Wegner, et al. report a typical HCl tropospheric lifetime of 1 to 2 days under conditions allowing photochemistry.[2]

Based on the lifetime information found in this research, HCl does not appear to be very persistent in the atmosphere. However, it is possible that chemical lifetimes of 1 to 5 days may result in utility emissions of HCl reaching acid rain-, or $PM_{2.5}$-sensitive receptors. Future research could address this question.

Several sources were found on the atmospheric chemistry and removal of HCl. Removal rate is an important factor when considering HCl's ability to be transported.

HCl is a highly reactive gas which is rapidly removed from the atmosphere by most surfaces, particularly those that are moist.[5,8] HCl's dry deposition rate is thought to be controlled by atmospheric turbulence, rather than surface conditions. In general, HCl gas will be removed from the atmosphere much faster than SO_2 or NO_2 and will be deposited in close proximity to the emissions source. Because HCl is highly soluble, washout is an efficient removal mechanism. Wet deposition is also likely to deposit HCl close to the source. A study by Patrinos, et al. found nearly all HCl to be wet-deposited within 15 km from the coal-burning power plant source.[12]

HCl has been found to be responsible for significantly enhancing the acidity of cloud water. In the absence of clouds or rain, HCl stack emissions are likely to be gaseous upon deposition. Chloride ions can catalyze the oxidation of sulphite to sulphate in concentrations commonly found in stack plumes. This can cause SO_2 suspended in chloride-containing water droplets to be oxidized more rapidly than if in pure water. If this is the case, SO_2 deposition close to the source will be decreased, but rain and cloudwater acidity will be increased, subject to further transport. However, because the solubility of SO_2 is pH dependent, if HCl lowers the pH below approximately 3, chloride ion catalysis may be reduced. HCl’s solubility is the reason it is more efficiently removed by rain than either SO_2 or NO_x. Thus the acidity in rainwater near a coal-fired power plant may be predominately the result of HCl, rather than SO_2 or NO_x.

More evidence of chloride ion catalyzation of SO_2 was found in a study by Clarke and Radojevic. This study was designed to provide more applicable kinetic data concerning the oxidation of SO_2 in fresh and salt water.[13] Clarke and Radojevic found that various chloride salts significantly increased the rate of SO_2 oxidation. The effects increased with increasing salt concentrations. Other nonchloride salts were also studied and did not significantly affect the reaction rate. The levels at which the reaction rate was affected, however, were greater than typical concentrations of chloride in cloud and rainwater. Consequently, the main significance for HCl atmospheric chemistry lies in reactions of marine or coastal aerosols and in HCl-enriched plumes at high humidity. In these environments, rates of oxidation of SO_2 in droplets will be elevated above the rates of pure water by up to four orders of magnitude.

Tropospheric HCl can be generated by reactions between sea-salt aerosol and atmospheric acids, volcanic eruptions, and the oxidation of nonmethane hydrocarbons (NMHC). In these reactions, Cl• is produced by photolysis of species such as Cl_2, HOCl, BrCl, and $ClNO_2$, which are all volatilized from sea-salt aerosol. In the troposphere, HCl is primarily removed via wet and dry deposition, while hydroxyl radical (OH•) and ocean hydrolysis reactions are minor sinks. The hydroxyl radical reaction proceeds as follows:

$$HCl + HO\bullet \rightarrow H_2O + Cl\bullet \qquad (5)$$

HCl can impact the atmospheric chemistry of other species, including other HAPs. For example, Selgneur, et al. identified the reaction between HCl and elemental mercury, Hg(0), as one of the relevant gas-phase reactions involving Hg(0) in the atmosphere.[14]

$$Hg(0)_{(g)} + HCl_{(g)} \rightarrow \text{products} \qquad (6)$$

Reaction rate parameter = 1.0×10^{-19} cm^3 $molecule^{-1}$ s^{-1}

In their theoretical study, Selgneur, Wrobel, and Constantinou noted that specific products of this reaction have yet to be identified. The authors found that aqueous-phase simulations of the Hg(0)/Hg(II) concentration ratio, based on an atmospheric chemical kinetic mechanism they developed, were quite sensitive to HCl concentrations. When HCl is not present, the Hg(0)/Hg(II) ratio calculated by the model ranged between 10 and 1,000,000. When HCl is present, this ratio is believed to be on the order of 10. Liquid water content, pH, and SO_2 concentration were all found to have a large and complex impact on the aqueous atmospheric chemistry of Hg(0). HCl may thus affect the toxicity of mercury emissions from utilities.

In summary, HCl is a highly reactive gas that is removed from the atmosphere via wet and dry deposition. In general, because of its high solubility, HCl will be removed from the atmosphere much faster than SO_2 or NO_2 and will be deposited in close proximity to the emissions source. It is possible that HCl may affect the toxicity of mercury emissions from utilities.

H.2.3 HCl Atmospheric Transport

The remainder of this section discusses the propensity of HCl to be transported in the atmosphere, which is an essential factor when evaluating the impact of utility HCl emissions. The partitioning of one chemical, such as HCl, in the atmosphere can influence what compounds other HAP species become.

HCl dissolved in clouds that are not precipitating can be transported long distances, and thus may impact both acid rain and $PM_{2.5}$.[5] One way in which HCl may move long distances to sensitive receptors is via the process of chloride ion catalyzation of SO_2 to SO_4, as described in the previous section. HCl is highly soluble, rapidly dissolving in clouds or rain, and has been found to significantly acidify cloud waters. A study conducted by March estimated that HCl that had mixed with power station plumes contributed 57 percent of the acidity measured in clouds.[15]

In a study of HCl at a rural site in northeast Austria, Puxbaum, et al. found that elevated HCl concentrations were coming from air parcels originating from the north and east.[3] The authors believe the source of this HCl is coal combustion. HCl’s winter transport was believed to been enhanced by the shallower boundary layer and smoother surface as a result of snow cover. The authors examined 48-hour back trajectories. If they are correct in their assertion that the higher HCl concentrations originating in the northern and eastern directions are from coal combustion, this would lend credence to the possibility of HCl transport, at least for up to a 48-hour period.

HCl was found to affect the gas/liquid partitioning of Hg(0).[14] For instance, when HCl is present, nearly all Hg(II) is present as $HgCl_2$. However, when HCl is absent, most Hg(II) is present in the form of $Hg(SO_3)_2^{2-}$ with a small portion present as $HgSO_3$. When liquid water content, pH, and total SO_2 concentration are kept constant at 0.1 g/m^3 and 4 and 10 ppb, respectively, moving from 0 μg/m^3 HCl to 1 μg/m^3 HCl was found to raise the Hg(II) gas/liquid equilibrium ratio by up to two orders of magnitude (from 0.0035 to 0.36).

According to the reports discussed in this section, conditions do exist under which HCl can alter the spatial and temporal deposition of acidic species. HCl concentrations were found to be among the factors impacting the atmospheric chemistry of mercury.

H.2.4 HCl Terrestrial Processes

A chemical's ability to accumulate in food chains and cause long-term harm is linked to its stability and persistence. Information on terrestrial HCl chemistry was found on three topics: fog events, damage to limestone, and the mobile anion hypothesis.

Fog events, by altering the oxidation rate of SO_2 to SO_4 and by producing strong acidity, may have an impact on acid deposition. Concentrations of gaseous HCl are limited by the equilibrium with solid aerosol phase NH_4Cl (see reaction #4).[1] During fog events, high concentrations of sulfate may be found in small aerosols as a result of high-pH, aqueous-phase SO_2 oxidation. The presence of HCl can lower the pH to the point that SO_2 oxidation is delayed, possibly altering the spatial deposition of acid species. Acid aerosol deposition may thus impact vegetation and soils in regions that experience fog.

Hutchinson, et al. demonstrated damage to limestone by gaseous HCl.[8] Humidity, degree of surface wetness, and temperature were all shown to affect the intake of acids by limestone. HCl is deposited on the stone by dry deposition which occurs in two stages. First, the pollutant is transferred to the surface of the stone, then, it is either absorbed or adsorbed by the stone. Relative humidity increases the absorption of the pollutant, as does surface wetness due to the solubility of many acids such as HCl. It was found that the reaction of HCl and limestone is very rapid, occurs very close to the source, and is more prevalent on moist surfaces. The calcium chloride produced in the HCl/limestone reaction was easily removed by runoff. Degradation gave the limestone the appearance of acid rain damage, which was attributed to the action of runoff.

As water flows through soils, it dissolves equivalent amounts of anions and cations. Typically, these are bicarbonate and organic anions balanced by base cations, hydrogen, and aluminum. The mobile anion hypothesis, as described in the NAPAP State of the Science Report 10, proposes that cation leaching in soils is controlled by the availability of mobile anions.[16] Acidic deposition may increase the concentration of mobile strong-acid anions, thus hastening base cation leaching in soils with medium to high base saturation, or leaching of acid cations (H^+, Al^{n+}) into surface waters from low base saturation soils. The extent or magnitude of soil acidification depends on the relative abundance of hydrogen, aluminum, and base cations, as influenced by vegetation uptake and cycling and weathering rates of soil minerals. Soils and soil water can be very acidic under natural conditions without resulting in acidic surface waters. What controls the concentrations of surface water acidity and base cations is the limited mobility of anions from the soils to surface waters. Many organic anions are retained in lower horizons of some soils or oxidized by soil microorganisms. Nitrate and sulfate are commonly retained through several biogeochemical mechanisms. Studies have confirmed the important influence of anion production and mobility on nutrient leaching following harvesting, fertilization, wastewater application and atmospheric sulfuric acid inputs.[16]

Deposition of sea salts on acid soils has been demonstrated to result in naturally acidic surface waters, which has been explained through exchange of base cations for hydrogen and aluminum in the soils and subsequent leaching balanced by mobile chloride anions. The proposed mechanism for this effect has been discounted as a cause of chronic acidification because it should result in a long-term buildup of base cations or an alkalization of the soil-vegetation system. Such an increase in the base cation content of soils has not been observed, and since such a buildup would eliminate the exchangeable hydrogen and aluminum, and hence acidic runoff, the mechanism seems to violate logic.[16]

Vegetation takes up and retains sea-salt-deposited base cations, producing strong mineral acid (HCl) and acidifying the soil. As would be the case for bicarbonate or organic anions, the presence of a mobile anion (Cl^-) not balanced by base cations could allow transport of hydrogen and aluminum from soils into surface waters if water did not flow over weatherable materials. Erosion would also prevent the buildup of base cations, maintaining soil acidity. There is no evidence to indicate that vegetation grows faster, or that erosion of forested systems is more rapid, in coastal areas that receive sea-salt spray than in inland systems. Whether sea-salts acidify systems would depend on whether vegetation uptake and removal of base cations outpace salt spray and weathering. In near-coastal systems with low weathering rates, or on poorly drained soils where roots and water do not contact weatherable materials, salt spray may be the primary source of base cations for vegetation uptake. In such systems chloride would leach hydrogen and aluminum from acid soils, and chronic acidification by this mechanism could occur. No near-coastal whole-ecosystem study exists to evaluate rates of base cation supply and cycling on a site-specific or regional basis.[16] The sea-salt mechanism may be an unlikely cause of regional acidification of inland systems; however, it has implications for the potential contribution to inland surface water acidification from atmospheric chloride deposition resulting from utility emissions.

In summary, by altering SO_2 oxidation, HCl can possibly impact the spatial and temporal deposition of acidic species. HCl is also capable of causing damage to limestone structures. Terrestrial HCl transport has the potential to affect numerous features of ecosystems, including soils, plants, and animals, both directly and indirectly. The mobile anion hypothesis is a possible explanation for how utility HCl emissions might impact terrestrial processes.

H.2.5 HCl Aquatic Processes

No information specific to the aquatic stability or persistence of HCl was found during this review. The aquatic chemistry of HCl is important to the toxic burden, if any, it presents to aquatic organisms. Two reports containing information on aquatic chemistry were identified for this review.

Stewart, et al. report that the chemistry of chlorinated compounds in natural waters is complex.[17] The pH, water temperature, nitrogenous compounds, and types and amounts of organic matter present all affect the persistence and toxicity of interim species of ubiquitous chlorine-containing compounds. The oxidation of chlorine-containing compounds such as OCl^- ultimately releases chloride ion.

Skeffington lists a potential equilibrium constant (pK_a) of -3 for the following reaction of hydrochloric acid and water:[18]

$$HCl + H_2O \rightarrow H_3O^+ + Cl^- \qquad (7)$$

A pK_a value in this range means that the above reaction goes to completion and HCl, as a strong acid, fully ionizes to hydronium and chloride in water.

Part of the understanding we have of acid rain is based on what we know about the behavior of chloride in aqueous environments. This review includes a study on acid rain and a study investigating whether chloride in waterbodies can be correlated with various anthropogenic activities.

Peters examined the factors controlling chloride anion (Cl^-) in two New York watersheds by examining precipitation, throughfall, soil water, groundwater, and surface water.[19] By combining previous research on each of these water types, he was able to determine that Cl^- cycling is more complex than the generally held view that there is rapid transport of atmospheric Cl^- deposition. In the Adirondack study system, it was shown that an additional Cl^- source in the watershed is the weathering of Cl^- from hornblende in surface minerals. Once Cl^- is available in the watershed, the author hypothesizes that the biotic system controls cycling; however, there was no direct quantification of this hypothesis, only an estimated comparison. Peters also found that annual Cl^- throughfall flux is two to five times that of precipitation, and that the watershed containing more hornblende in the surface minerals had a three-fold greater net flux of Cl^-.

Specifically, important findings include the observations that atmospheric deposition is not the only primary source of Cl^-, and that the biotic cycling of Cl^- plays a significant role in transportation/ retention in the system. A better understanding of Cl^- cycling would be beneficial for estimating the impacts of acid deposition on a watershed.

In a subsequent study of the interference of dissolved organic carbon (DOC) on colorimetric Cl^- measurements, Norton, et al. challenge the Cl^- cycling findings of several studies, including that of Peters, above.[20] The positive DOC interference was found to bias colorimetric Cl^- measurements, leading to overestimates of Cl^- concentrations. Norton, et al. present recalculated Cl^- concentrations from the Peters study, using their own empirical relationship between DOC and Cl^-. These authors conclude that the bias in data published in the literature may have lead to spurious conclusions about Cl^- budgets in forests and watersheds, and that the issue of dry deposition and recycling of Cl^- is far from settled.

Zahn and Grimm developed a qualitative spatial analysis of land use (agricultural, urban, forest, and groundwater protection areas) as compared to nitrate and chloride concentrations in underlying groundwater.[21] Within the study area in southern Germany, it was found that groundwater concentrations of nitrate ranged from 2 mg/L to >80 mg/L, with an average of 26.2 mg/L. Similarly, the concentration of chloride in the groundwater samples ranged from 1 to 86 mg/L and had an average value of 16.6 mg/L. There was no observed correlation between depth of sample and concentration. Expected values of nitrate and chloride, when considering regional geology, would be expected in the range of 10 and 5 mg/L, respectively. In an attempt to correlate the observed increase with anthropogenic sources, each observed increase was correlated with an anthropogenic factor if possible (i.e., agricultural runoff, roads, industries, waste dumps, etc.). When the results of these correlations were analyzed, it was found that elevated chloride levels were most closely associated with settlements, industries, waste dumps, and roadways, although the correlation between land use and groundwater concentrations was not apparent in all localities. It was determined that the only substantial correlation was in nitrate and chloride concentrations linked to waste dumps. The authors state that the anthropogenic impacts are greater for chloride than for nitrate.

Zahn and Grimm note that one of the largest anthropogenic sources of nitrate and chloride in soils (and hence groundwater) in Germany is atmospheric deposition. Depending on land use, nitrate deposition ranges from 1,500 to 2,050 $kg/km^2/yr$, and chloride deposition varies from 670 to 1,170 $kg/km^2/yr$. Although it is qualitative, this work indicates that there is a spatial correlation between land use practices and the nitrate and chloride levels observed in groundwater. A more quantitative analysis would be necessary to validate these results. The extensive development of quantitative spatial analysis with global imaging systems will make the correlation of groundwater pollution and land use or atmospheric deposition a potential screening instrument for groundwater contamination by anthropogenic activities, such as utility HCl emissions in future research.

In summary, chloride cycling in watersheds was found to be more complex than researchers previously thought. The traditional view has been that atmospherically deposited chloride is rapidly transported. This research indicates that a better understanding of chloride cycling is necessary. A qualitative spatial analysis found that elevated concentrations of chloride in groundwater were most strongly correlated with waste dumps.

H.2.6 HCl Human Health Impacts

Inhalation of HCl can cause injury to the respiratory tract. Risk from exposure to high concentrations of HCl is most likely to arise in occupational settings or through an industrial or transportation accident, while exposure to relatively low HCl concentrations occurs over a wide area subject to emissions from anthropogenic sources, including utility, industrial, and waste combustion, as well as natural sources.[22]

Chapter 6 of this report uses the RfC_i for HCl of 20 $\mu g/m^3$ to estimate the chronic noncancer hazard quotient (HQ), and lists the California Air Pollution Control Officers Association (CAPCOA) acute Reference Exposure Level (REL) as 3,000 $\mu g/m^3$ for HCl. Additional information on HCl acute toxicity was obtained from a Lab Chemical Safety Summary compiled by the National Research Council and EPA Region III.[23] The National Research Council reported a Threshold Limit Value (TLV) of 5 ppm and a Permissible Exposure Level (PEL) of 5 ppm for HCl. Toxicity values for chlorine range from 0.5 ppm to 1.0 ppm. Table H-1 contains the National Research Council’s reported toxicity values. EPA Region III’s Superfund Risk Based Concentration (RBC) for chlorine is 370 $\mu g/m^3$.

The results of a WHO review on hydrogen chloride exposure to humans yielded evidence of local irritation to the upper respiratory tract.[22] Mucous membranes were found to be especially susceptible. Exposure at higher concentrations caused conjunctival irritation and superficial corneal damage. Hydrogen chloride can cause transitory epidermal inflammation if it comes in contact with damp clothing or skin. Long-term exposure may induce erosion of the inciso-labial surfaces of the teeth. Even though one study did show possible induction of a tumor growth using HCL, the authors conclude that there are no mutagenic, carcinogenic, or teratogenic effects related to HCl. The WHO task force could not, due to the dearth of data, determine a not-to-be-exceeded ambient HCl concentration.

In summary, the National Research Council’s Lab Chemical Safety Summary contains pertinent toxicity data on HCl. Evidence of local irritation to the upper respiratory tract by HCl was found and long-term exposure may cause tooth erosion. The WHO concluded in a review

Table H-1. National Research Council Acute Toxicity Values for Chlorine

Chemical	**TLV-TWA**	**PEL**
Hydrogen chloride	5 ppm	5 ppm
Chlorine	0.5 ppm	1.0 ppm

PEL = Permissible Exposure Limit
TLV = Threshold Limit Value
TWA = Time Weighted Average

on HCl that there are no mutagenic, carcinogenic, or teratogenic effects related to HCl.

H.2.7 HCl Vegetation Impacts

The behavior of vegetation with regard to uptake, absorption, translocation, distribution, and accumulation of chloride might all have important effects on survival or growth. One study directly pertaining to chloride's effects on vegetation was found.

In a Texas study designed to ascertain the impact of marine salt on vegetation, McWilliams and Sealy found a strong correlation between atmospheric chloride levels and leaf chloride levels in Spanish moss (*Tillandsia usneoides*).[24] Most nonanthropogenic chloride is attributable to marine sources. *T. usneoides*, an epiphytic plant, is ideal as a subject for studies of atmospheric contaminants on plants. This study suggests that atmospheric and leaf chloride levels are closely correlated, at least in certain situations.

In their discussion of past findings on chloride salt vegetation toxicity, McWilliams and Sealy note that the "wind form" of vegetation along the North Carolina coast was found to be the result of lethal salt effects rather than wind.[24] Atmospheric chloride can enter plants via aerial organs and concentrate in leaf tissues, leading to foliar damage.

H.2.8 HCl Terrestrial Animal Impacts (Wild and Domestic)

HCl may reach plants and soil via wet and dry deposition. Once present it is available for intake by wild and domestic animals. Tukey et al. report that very little chloride leaches from leaves.[25] Therefore, animals foraging on HCl-enriched foliage will ingest additional chloride.

Research on the effects of HCl on animals was reviewed by WHO.[22] Symptoms include eye and nasal irritation, but the mucous membranes and respiratory tract are the primary targets. Edema is characteristic of the initial symptom of hydrogen chloride toxicity, proceeding to additional inflammation, degeneration, and necrosis when in contact with tissue. The authors of the various studies noted that chlorine does not appear to be teratogenic, mutagenic, carcinogenic, or cocarcinogenic in animals, and that the carcinogenic potential of hydrogen chloride could not be assessed due to a lack of adequate studies.

H.2.9 HCl Aquatic Animal Impacts

This section addresses the impacts of HCl on aquatic animals. Sources of information on the ingestion, distribution, and accumulation of HCl in aquatic animals, along with its toxicity, were reviewed. Two sources on aquatic HCl toxicity were found and are described below. No information pertaining to ingestion, distribution, or accumulation was found for this review.

Stewart, et al. report that some chlorinated compounds, such as $OC1^-$, found in natural waters are quite toxic to freshwater biota, while others, such as Cl^-, are not.[17] Chlorine is widely used in water treatment as a disinfectant for drinking water and wastewater prior to discharge; Manning, Wilson, and Chapman found it to be toxic to aquatic life.[26] Many treatment plants use ammonia and chlorine together for the treatment process in a ratio of 3:1 ammonia to chlorine. This study obtained data on Australian organisms to support development of appropriate treatment plant controls. It included an evaluation of the toxicity of a mixture of chlorine and ammonia to assist in evaluating the potential impacts of drinking water on ecosystems. The water flea *(Ceriodaphnia dubia*) was used as the freshwater organism, and the eastern king prawn (*Penaeus plebejus*) was used as the saltwater organism.

There is a distinct difference between the American and Australian *C. dubias*; the Australian water flea is not as sensitive to chlorine as the North American variety. Based on results of 24-hour LC_{50} obtained in these tests, the environmental concern level for chlorine in marine situations is 0.0018 mg/L and the environmental concern level for chlorine and ammonia in a freshwater situation is 0.003 mg/L. These figures are in line with the standards determined in the northern hemisphere. The chlorine/ ammonia mixture is not seen in salt water, so a test was not performed on the salt water organism. Environmental concern levels for chlorine and ammonia alone were not reported for fresh water.

H.3 FINDINGS FOR HYDROGEN FLUORIDE

H.3.1 HF Emissions and Formation

Hydrogen fluoride (HF) can exist as either a colorless, corrosive liquid or a gas at room temperature and is used in numerous production processes, including those of high-octane gasoline, aluminum, plastics, electrical components, fluorescent light bulbs, and refrigerants.[27]

The information on nationwide utility HF emissions was obtained from Table 3-3 in Chapter 3 of this report. The 1990 HF emissions are reported as 19,500 tons/year. Utility emissions are the most significant anthropogenic source of atmospheric HF. Other important sources in addition to utilities are phosphate manufacturing and aluminum manufacturing.

Anthropogenic sources are responsible for a considerable amount of atmospheric fluoride.[27] Anthropogenic emissions of HF originate from coal combustion and the aluminum, phosphate, and steel-making industries. For example, fluoride in the atmosphere of the Colorado Plateau region of the western United States is believed to originate primarily from power plant emissions; this is a result of the high fluoride content of western coals and the lack of aluminum electrolysis plants in the region.[28] Relatively small amounts are emitted from the production of HF itself (2,400 metric tons per year).

Volcanoes are the primary natural sources of HF. Total annual volcanic emissions of fluoride-containing compounds are estimated to be between 1 and 7.3 million metric tons per year. Ocean spray, fires, and dust from soil and rock weathering also contribute fluoride to the atmosphere; however, annual emissions from these sources are believed to be negligible.

Atmospheric concentrations of fluoride in remote rural areas are reported to be approximately 0.1 $\mu g/m^3$, which is at the limit of detection.[29] Urban fluoride concentrations ranging from 0.1 $\mu g/m^3$ to 1.0 $\mu g/m^3$ have been observed in British industrial cities.[29] However, HF accidentally released from chemical plants may produce concentrations on the order of 50 $\mu g/m^3$ for several hours. As of August 1994, only one HF monitoring station had been operated in the United States, with 20 samples measured.[7] Measured concentrations of HF ranged between 1 $\mu g/m^3$ and 8 $\mu g/m^3$.

As in the case of HCl, ambient concentrations of HF will be of interest as part of the effort to achieve compliance with the new particulate matter standards. If a significant portion of ambient HF is in the fine fraction, it could contribute to $PM_{2.5}$ exceedances.

Information was found concerning the relative proportions of gaseous to particulate fluoride emissions, fluoride particulate composition and diameters, HF formation from volcanic emissions, general formation material, and HF reactants. Hance, et al. report that up to 40 percent of atmospheric emissions of industrial fluoride are gaseous, with the remaining 60 percent emitted as particulate matter.[27] Slooff, et al. estimate that about 25 percent of atmospheric fluoride is emitted as particulate, with the remaining 75 percent in the gaseous state (mostly HF).[30]

Gaseous HF undergoes hydrolysis before dispersion throughout the atmosphere.[27] Fluoride particulates vary from distinct minerals to alumina with HF adsorbed to its surface, and particle diameters range from <0.1 μm to approximately 10 μm.[29] Davison reports that fugitive fluoridated alumina particle sizes near a dry-scrubbed aluminum smelter can range from <1 μm to about 5 μm in diameter.[29]

Volcanic emissions are not usually predominately HF, but rather other fluoride-containing compounds which react in the atmosphere to form HF. These compounds include boron trifluoride, carbonyl fluoride, phosphorus pentafluoride, silicon tetrafluoride, sulfur tetrafluoride, and phosphorus trifluoride, all of which produce HF via hydrolysis.

Both the conditions under which HF is released and the atmospheric conditions can effect the behavior of HF upon release.[31] Superheated HF that is released under pressure will form a cloud of HF vapor and aerosol, which will react with water vapor in the air. If present in a concentration greater than 40 percent, both anhydrous and aqueous HF will react with atmospheric moisture to produce white fumes, which have a pungent odor and are extremely irritating if they are inhaled or if they come in contact with an organism. HF molecules form variable length chains up to $(HF)_8$ at ambient temperatures through hydrogen bonding. HF can also be present as single molecules at higher temperatures. HF's hydrogen bonding leads to the formation of vapor clouds that can be either neutral or positively buoyant, depending on atmospheric conditions.

Based on thermodynamic data, Slooff, et al. suggest that HF reacts with other gaseous acidic species, including HNO_3, H_2SO_4 and HCl, present in the atmosphere.[30] Therefore, it is possible that HF may have some indirect, limited impact on acid rain formation, and may possibly have an indirect, limited impact on visibility and fine-PM issues. As is the case with HCl and chloride salt formation, Spicer, et al. found that HF reaction products also include primarily fluoride salts. The authors describe the identification of reaction products as qualitative in most cases, because few studies reported mass balance information.[11]

H.3.2 HF Atmospheric Processes

Stability and removal mechanisms of atmospheric HF determine its persistence. Spicer, et al. describe HF as moderately persistent in the atmosphere, with an estimated lifetime of between 1 and 5 days, which places it in the 46 percent of HAPs with lifetimes in this range.[11] Slooff, et al. estimate the half-life of gaseous HF to be 0.54 days, while fluoride aerosol has a half-life of 2.1 days.[30]

Spicer, et al. also found deposition, both wet and dry, to be the primary atmospheric removal process for HF; thus, they do not expect HF to undergo significant chemical transformation in the atmosphere.[11] Hance, et al. found precipitation to be the major global route of atmospheric HF removal.[27] Slooff, et al. also cite wet and dry deposition as the primary routes of HF removal from the atmosphere.[30]

Little information was available concerning atmospheric chemistry or degradation of HF. The material that was reviewed indicates that HF does not biodegrade, regardless of whether it is released to air, water, or land.[31]

In summary, HF is described as moderately persistent in the atmosphere, with an estimated lifetime of approximately 1 to 5 days. Wet and dry deposition are the primary routes of HF removal from the atmosphere.

H.3.3 HF Atmospheric Transport

Fluoride emissions from utilities are transported on a regional scale. In fact, fluoride (F_{total}:SO_x ratio) was used as an atmospheric tracer in one study reviewed because this ratio, which is relatively constant for the coal-fired utilities examined, provides a characteristic utility emissions fingerprint.

In a source-apportionment study of regional air pollution transport to the Canyonlands National Park, Eatough, et al. found evidence that coal-fired utility emissions, among other sources, contribute to measured concentrations of atmospheric fluoride at distances of up to 500 km.[32] Slooff, et al. report that atmospheric fluoride, especially aerosol fluoride, can be dispersed over great distances by wind or atmospheric turbulence.[30] Fine particles will also be deposited over a greater area than large particles. Leece, et al. found that elevated levels (18-21 ppm) of fluorides emitted from a power station at Liddell, New South Wales, could be detected in grape leaves at distances of up to 37 kilometers.[33] Background foliar fluoride concentrations were measured at less than 1 ppm.

The measurement of fluoride transport and deposition has been reported to have problems of accuracy due to methods used. Davison cites analytical problems and the lack of a suitable isotope as adversely affecting the scope and accuracy of measurements of fluoride deposition and transport.[29] The lack of an ideal analysis technique for biological materials has resulted in the absence of an absolute laboratory standard. This has commonly resulted in a sampling error of ±10 percent for repeated analysis of the same sample. The short half-life of the ^{18}F isotope (1.8 hours) has precluded its use for measuring fluoride deposition and movement through trophic levels. Thus, fundamental problems with fluoride present limits to the study of deposition, pathways, and accuracy of measurements.

To summarize, evidence was found that coal-fired utility emissions, among other sources, contribute to measured concentrations of atmospheric fluoride at distances of up to 500 km. The measurement of fluoride transport and deposition has been reported to have problems of accuracy due to methods.

H.3.4 HF Terrestrial Processes

The stability and persistence of a chemical is linked to its ability to accumulate in food chains. Information concerning HF volatilization and the ability of soluble fluoride to alter physical and chemical soil properties is reviewed in this section.

Davison’s synopsis of research shows that fluoride is lost from the various surfaces on which it is deposited and leaves ecosystems at a rapid rate.[29] Some possible mechanisms have been investigated with varying success. The pathway of volatilization to the air was presumed to account for most of the lost fluoride. However, research on this pathway yielded contradictory results. The hypothesis was proposed that although fluoride ions are not significantly volatile, a portion of fluoride forms complexes with hydrogen ions to form HF, which is volatile at a pH of less than 5. Davison concludes that the volatilization pathway as a route of fluoride export from ecosystems needs further investigation.

Arocena, et al. investigated the effects of soluble fluoride in the process water and leachate of phosphogypsum formed by the production of phosphate fertilizer from phosphate rock.[34] Analyses of both calcarious and non-calcarious soil clay components showed a mineral composition of smectite, kaolinite, mica, and trace amounts of chlorite. The phosphogypsum process water was shown to dissolve much of the fine clay fraction, as well as the smectite of the coarse clay fraction. The observed changes in the clay fraction can significantly alter physical and chemical properties of the soil, such as hydraulic conductivity, absorption, and ion exchange capacities. Although the HF described in this article originated in the production of fertilizer, the effects of HF in acid rain could be similar. The research in this study is significant, because the dissolution of clay by fluoride ion could be an important impact on ecosystem soils, as well as the performance of landfill clay liners.

Fluoride in non-saline soils is associated with the clay-sized fraction because it reacts with aluminum compounds and various minerals.[29] Soil can be both a sink and source of fluoride. In undisturbed soil profiles, the fluoride gradient usually increases in concentration with depth as a result of leaching. However, because soils can fix large amounts of fluoride, the rate of leaching is usually slow. Soil fluoride fixation also results in retention in the upper soil horizons. Fluoride is not usually available or labile in soils. Soil acts as both a sink and resistance, slowing and controlling fluoride transport to plant and aquatic systems. Ash deposition after fire events, saline (low calcium, high pH) soil conditions, and formation of aluminum-fluoride complexes as a result of aluminum smelter fluoride particulates are possible exceptions that can facilitate rapid water-soluble fluoride uptake and transport.

If sufficient calcium is present in soil or water, it will form an insoluble solid with fluoride ion, which removes it as an immediate environmental hazard. The natural buffering capacity of soils or water, or dilution can potentially reduce acidity added by the presence of HF.

Information on the ability of disturbances to cause HF mobility, and HF’s ability to mobilize soil aluminum were reviewed. HF does not biodegrade, regardless of whether it is released to air, water, or land.[31] Disturbance events can impact fluoride’s movement through an ecosystem. For instance, Murray found that fire increases the water-soluble fluoride fraction in standing soil by a factor of two, but only for a period of a few months.[35]

Bond, et al. conducted a study on the effects of sulfate and fluoride on soil pH, soil aluminum (Al) concentrations, and Al transport.[36] The experiments were designed to test the hypothesis that soil fluoride increases Al mobility in soils and raises soil pH, and were performed in soil columns. The results were compared to the findings of a similar set of batch studies that found that fluoride increased Al soil transport. Overall, the authors determined that under the more realistic soil:solution ratios experienced in soil columns, fluoride did not increase Al mobility. In fact, both sulfate and fluoride were found to slightly retard aluminum's mobility through soils. This experiment suggests that fluoride's hypothesized ability to raise soil pH by mobilizing Al was spurious, and that fluoride is not a preferred agent for alleviating soil acidity.

To summarize, fluoride is believed to be lost from the various surfaces on which it is deposited, and leaves ecosystems at a rapid rate. Soluble fluoride-containing process water and leachate of phosphogypsum were shown to dissolve much of the fine clay fraction, as well as the smectite of the coarse clay fraction of soils. HF does not biodegrade in air, on land, or in water. The natural buffering capacity of soils, water, or dilution can potentially reduce acidity added by the presence of HF. Fire was found to increase the water-soluble fluoride fraction in standing soil, and sulfate and fluoride were found to slightly retard aluminum's mobility through soils.

H.3.5 HF Aquatic Processes

Fluoride is known to be a major component of seawater, with a residence time of approximately one million years.[30] In fresh water with pH greater than 5, fluoride is mainly present as fluoride ion, based on stability diagrams derived from thermodynamic calculations.

In fresh water, fluorapatite, $Ca(PO_4)_3F$ or $Ca_{10}(PO_4)_6F_2$, is only somewhat soluble.[30] Slooff, et al. provide the following equilibrium reactions:

$$Ca(PO_4)_3F + 6H^+ \rightleftarrows 5Ca^{2+} + 3H_2PO_4^- + F^- \qquad \log K = -1 \tag{8}$$

and:

$$Ca_{10}(PO_4)_6F_2 \rightleftarrows 10Ca^{2+} + 6PO_4^{3-} + 2F^- \qquad \log K = -119 \tag{9}$$

which show that the solubility product is very low.

Elrashidi and Lindsay found that in waters with pH greater than 5.6, the fluoro-hydroxy-apatite system is dominant. The following equation describes the fluoro-hydroxy-apatite system:[37]

$$Ca_5(PO_4)_3F + H_2O \rightleftharpoons Ca_5(PO_4)_3OH + F^- + H^+ \quad (10)$$
$$\log K = -15.14$$

Dutch water quality officials regularly measure aquatic concentrations of fluoride that are three orders of magnitude greater than what is expected, based on the above calculations. Accordingly, there must be a factor increasing fluoride concentrations in fresh water beyond those accounted for in the theoretical calculations of thermodynamic equilibrium. Researchers are not able to balance the fluoride budget in the Netherlands. The addition of atmospheric HF to the equilibrium in reaction (10) would be likely to generate more fluorapatite.

Fluorapatite discharged from phosphate ore-processing plants or precipitating in water may accumulate in water body sediments.[30] Slooff, et al. note the need for greater understanding of the distribution of labile and insoluble fluoride to help resolve the uncertainty about partitioning of fluoride in waterbodies and sediments.

H.3.6 HF Human Health Impacts

Table 6-10 of this report lists the California Air Pollution Control Officers Association (CAPCOA) acute Reference Exposure Level (REL) for HF as 580 $\mu g/m^3$. Additional literature reviewed on human health impacts centered on ingestion and inhalation. Information on HF acute toxicity was obtained from National Research Council Lab Chemical Safety Sheets, which provided a TLV-TWA of 3 ppm (2.6 milligrams per cubic meter), and a PEL of 3 ppm for HF.[23] Table H-2 contains toxicity values for several fluoride species.

Whitford reviewed fluoride ingestion findings in a number of studies.[38] The difficulty in estimating the lethal dose or potentially toxic dose of fluoride in humans, particularly in children, can partly be attributed to the problem of estimating the actual dose after a poisoning event has taken place. Human studies of the relationship between treatment with fluoride and bone strength have revealed a conflicting picture. If there is either a beneficial or a detrimental effect, Whitford claims it is subtle, and convincing evidence will require large, carefully controlled, prospective research studies.

According to Davison, water sources in many cases account for about half of human fluoride intake in non-fluoridated water environments.[29] Daily human fluoride intake varies as a result of age, fluoride water content and consumption rate, and diet. Whether ingested fluoride is available for uptake will depend on its chemical state, which also affects retention time and pathway through the body. The chemical form of fluoride ingested will affect the pathway of elimination from the body and also the chemical species eliminated.

A considerable amount of information was available on the topic of HF’s toxicity to humans. This section discusses the wide range of potential toxic effects of HF, as well as some possible benefits.

Table H-2. National Research Council Acute Toxicity Values for Fluorides

Chemical	TLV-TWA	PEL
Hydrogen fluoride	3 ppm	3 ppm
Fluorine	1 ppm	0.1 ppm
Fluorides	2.5 mg/m3	2.5 mg/m3

PEL = Permissible Exposure Limit
TLV = Threshold Limit Value
TWA = Time Weighted Average

Fluoride, the ionic form of fluorine, is a potent inhibitor of many enzymes.[38] Unlike iodide it does not accumulate in the thyroid. The rate of elimination of fluoride from the kidneys is many times greater than it is for other halogens. It stimulates new bone formation and can inhibit, or even reverse the formation of dental caries. Fluoride has other beneficial effects. It may reduce the incidence and severity of osteoporosis, and increased consumption of fluoride following the introduction of water fluoridation in the United States may be related to the sharp decline in the death rate due to heart disease.

Fluoride is a hazardous substance when taken acutely in large doses, and numerous claims exist of harm arising from the chronic ingestion of low doses.[38] The effects range from dental fluorosis, gastric disturbances, and reductions in urinary concentrating ability, to skeletal fluorosis and death. Based on an earlier estimate by Hodge and Smith, the certainly lethal dose (CLD) of sodium fluoride is estimated to be between 70 and 140 mg/kg body weight for an adult.[39] Based on a report by Dukes, a dose of approximately 4 mg/kg may be fatal for a young child.[40] Insoluble forms, such as calcium fluoride and cryolite (Na_3AlF_6), are less toxic than sodium fluoride, as they are less well absorbed. Monofluorophosphate (MFP) is approximately half as acutely toxic as sodium fluoride.

Whitford summarizes the findings pertaining to humans of a recent USPHS report on fluoride (one of three recent critical reviews) as follows: “(1) there is no detectable risk of cancer in humans associated with the consumption of optimally fluoridated water; (2) there is no indication that organ systems are affected by chronic, low-level fluoride exposure (although more research on human reproduction, for which there is a paucity of data, was recommended); (3) fluoride exposure is not associated with birth defects, including Down’s syndrome; (4) genotoxicity studies, which are highly dependent on the methods and models used, have yielded contradictory results so that any possible effect of fluoride in humans and laboratory animals remains unresolved; (5) the prevalence of dental fluorosis in the USA is higher now than in the 1940s but there is disagreement about whether this condition is a toxic effect; (6) crippling skeletal fluorosis has not been and is not a public health problem in the USA; (7) the beneficial effect of high fluoride regimens in reducing osteoporosis has not been demonstrated; and (8) further epidemiological studies are required to determine whether or not an association exists between various levels of fluoride in the drinking water and bone fractures."[38]

According to the WHO, excessive fluoride exposure may affect human health, but it has been difficult to substantiate cases of human fluorosis brought about solely by exposure to atmospheric fluorine, even in old, poorly controlled industrial environments.[41]

In summary, fluoride is a potent inhibitor of many enzymes, stimulates new bone formation, can reverse dental caries, and may reduce the incidence and severity of osteoporosis. Fluoridation of water in the United States may be related to the sharp decline in the death rate due to heart disease. Adverse effects of fluoride on human health include dental fluorosis, gastric disturbances, reductions in urinary concentrating ability, skeletal fluorosis and even death. Whitford's recent review of fluoride's toxicity provides a comprehensive synopsis of the subject.

H.3.7 HF Vegetation Impacts

This section covers available literature pertaining to fluoride uptake and accumulation by vegetation, as well as injury to vegetation by fluoride. Studies reviewed pertain to Brazilian rainforest trees, grape leaves, lichens, and pine trees.

Klumpp, et al. investigated fluoride accumulation in three tree species in a field study in the Atlantic Rainforest, near Cubatao, Brazil, where a large fertilizer industry is a source of fluoride emissions.[42] All were pioneer tree species known to be ubiquitous in the study area. The vegetation and soils within a 60 km^2 radius of Cubatao have experienced severe damage as a result of 30 years of high pollution, resulting in a high frequency of landslides and the replacement of primary vegetation with secondary vegetation types over a vast area. The study assessed the spatial and temporal distribution of fluoride's effects on the natural vegetation in the region (passive monitoring), as well as fluoride's effects on exposed *Tibouchina* seedlings being tested as a cumulative indicator species (active monitoring). Foliar fluoride concentrations were measured at four sites with varying pollutants and concentrations.

Inherent differences in the resistance of some tropical tree species to fluoride may be related to their capacity to accumulate aluminum. Although some plant species are tolerant of elevated fluoride levels, the storage of large amounts of fluoride in plant tissues may present a risk to the ecosystem. For instance, the movement of fluoride from plants to soils may affect nutrient turnover. Litter decomposition impacts may include fluoride accumulation in soil detritivores, increased humus layer, and reduced microbial activities, all of which have been observed in the vicinity of aluminum smelters. Foliar accumulation of fluoride may also impact plant-insect relationships, which may have ramifications through the food chain, affecting insects, vertebrate herbivores, or rodents.

A study of fluoride by Slooff, et al. conducted under the auspices of the Dutch National Institute of Public Health and Environmental Protection found that plant uptake of fluoride is limited to the smaller, water-soluble and labile fractions.[30] The major pathway of fluoride to plants is atmospheric deposition both to plant surfaces and intake through the stomata.

Leece, et al. found that in foliage, gaseous HF dissolves within the leaf and is transported to the leaf margins, where it accumulates.[33] If the ionic concentrations are high enough, stable salts may form, thereby rendering both cations and anions physiologically inactive. Reactions are as follows:

$$HF_{(g)} \rightarrow H^{+} + F^{-} \qquad (11)$$

$$Ca^{2+} + 2F^{-} \rightarrow CaF_2 \qquad (12)$$

The authors found evidence of fluoride transport of up to 37 kilometers. Although elevated foliar fluoride concentrations attributable to power plant emissions were found, symptoms of fluoride toxicity were not. Grape leaves were found to accumulate up to 40 ppm fluoride without developing toxicity symptoms in drought conditions. Because foliar damage caused by environmental conditions and physiological stresses (e.g., moisture stress, potassium deficiency, and chloride toxicity) appears similar to fluoride contamination, the authors urge chemical analysis of foliar tissue.

In an effort to obtain data on fluoride emissions from volcanos over time, Davies and Notcutt sampled lichen species on and around Mt. Etna, on the island of Sicily.[43] Lichen fluoride concentrations were measured in a preliminary survey in 1985 (77 sites) and then again in 1987 (56 sites). The fluoride accumulation pattern was affected by the volcano, with concentrations highest downwind of the plume and tapering off with increasing distance from the volcano, as mediated by topography and prevailing winds. This paper indicates that lichens, because of their ability to accumulate fluoride from both industrial and natural emissions sources, can be a useful tool for measuring utility and/or industrial fluoride emissions. The higher fluoride concentrations (>100 ppm) found at five sites are similar to those found at certain types of industrial sources. Noting studies that found elevated fluoride levels in vegetation and small mammal bones near an aluminum smelter, the authors suggest that gaseous fluorides emitted from both industrial and natural sources may have effects higher up the food chain.

Amundson, Belsky, and Dickie investigated foliar fluoride concentrations, foliar decomposition, and upper soil horizon fluoride accumulation in a coastal pine plantation in both close and distant proximity to a new aluminum smelter located near Charleston, South Carolina.[44] Pine stands within 0.8 km of the smelter had foliar fluoride concentrations significantly higher than one stand 1.8 km from the source. Fluoride emitted from the newly operating smelter was not observed to alter needlefall amount, temporal patterns, or rates of needle decomposition over a six-month period, although it did accumulate in needles. Over a seven-year period, soluble fluoride concentrations in the upper 10 cm of soils increased significantly, but only at sites nearest to the emissions source. However, these findings on two species of *Pinus* may not apply to other arboreal species or herbaceous plant species. It is possible that as needles decompose in the absence of fire for longer than 6 months, the concentration of fluoride may eventually increase to levels that inhibit decomposition. Questions remain about whether deposited fluoride is adsorbed to the needle surface or absorbed inside the needle, and whether increased soluble fluoride in the soil contributes to elevated levels in the needles.

Atmospheric fluoride is capable of injuring certain plant species at lower concentrations than any other air pollutant studied.[45] For instance, a review conducted by Davison noted that ripening peach fruit tissues can be visibly injured by fluoride concentrations of 0.3 $\mu g/m^3$ for 12 hours, 5 days a week.[29] However, most plant species are considered resistant to fluoride and can tolerate concentrations of up to 30 $\mu g/m^3$.

In their study conducted on and around Sicily's Mt. Etna, Davies and Notcutt found no morphological damage to lichen species, although lichen samples from five sites contained fluoride concentrations in excess of 100 ppm.[43]

In the Brazilian Atlantic Rainforest study, Klumpp, et al. found one of the three native plant species to exhibit marginal foliar necroses and malformations at the most polluted site.[42] Crown heights of all three species were lower at the polluted sites when compared with the reference area. Fluoride was found to be the most important pollutant contributing to vegetation damage in one section of this tropical rainforest, although ambient concentrations of other pollutants, including ozone, NO_x, ammonia, and SO_x, are likely to be affecting the vegetation as well. The study notes that the question of whether Al-F complexes are phytotoxic has not been clarified to date.

In summary, although some plant species are tolerant of elevated fluoride levels, the storage of large amounts of fluoride in plant tissues may present a risk to ecosystems. Inherent differences in the resistance of some tropical tree species to fluoride may be related to their capacity to accumulate aluminum. Plant uptake of fluoride was found to be limited to the smaller, water-soluble and labile fractions. The major pathway of fluoride to plants is atmospheric deposition. Although elevated concentrations of fluoride were found in lichens, grape leaves, and pine trees, foliar damage was not. In one section of a Brazilian rainforest, however, fluoride was found to damage vegetation.

H.3.8 HF Terrestrial Animal Impacts (Wild and Domestic)

The literature reviewed for this section pertained to small mammal and raptor species. Information concerning fluoride's accumulation in food chains, distribution, toxicity, and soil contamination as a route of fluoride transfer was also reviewed.

Fluoride can be ingested by small mammals as they consume vegetation. However, few of the numerous studies on fluoride's effects on vegetation attempt to quantify cycling from vegetation to soil and animals. Boulton, Cooke, and Johnson conducted a study on the toxicity of fluoride on short-tailed field voles and found that 19 to 43 percent of daily uptake of fluoride was retained.[46] Fluoride concentrations were increased in femur and incisors, but significantly decreased in molars. Incisors also had visible lesions. The difference in results between study locations is attributed to the chemical speciation of fluoride in the plant material and the variable levels of other dietary components, including Ca^{2+} and Al^{3+}. The authors of this study suggest that for absorbed fluoride, kidney excretion is likely a minimal removal pathway as compared to deposition in bones and teeth.

In another study, Boulton, Cooke, and Johnson compare the toxic effects of inorganic fluoride on four species of small mammal: laboratory white mice (*Mus musculus* L.), wood mice (*Apodemus sylvaticus* L.), short-tailed field voles (*Microtus agrestis* L.), and bank voles (*Clethrionomys glareolus* L). The primary difference between the species was the fluoride uptake rate, which was heavily influenced by the administration in the drinking water.[47] The order of uptake among species was *M. Agrestis > C. Glareolus > A. Sylvaticus > M. Musculus.* The observation of intake helps explain premature mortalities observed in *M. Agrestis* and *C. Glareolus.* Differential absorption and retention were observed in metabolic cage studies of *M. Agrestis* and *M. Musculus*. Interspecific variation related to intake of fluoride was observed in assays for fluoride concentration in femurs, molars, and incisors. Severe dental lesions were observed in animals surviving the highest dose at the termination of the experiment. The authors conclude that the interspecies effects in this experiment were the consequence of different intake levels. This report may be of interest because it examines several wildlife species, the use of which opens the possibility for the development of biomarkers. The observation that effects were primarily due to intake rates may also help identify susceptible species that have increased rates of consumption of fluoride-contaminated media.

Seel, Thomson, and Bryant measured bone fluoride in four predatory bird species in the British Isles: sparrow-hawk (*Accipiter* *nisus*), kestrel (*Falco tinnunculus*), barn owl (*Tyto alba*), and tawny owl (*Strix aluco*).[48] Fluoride loads increased with age in all four species. The data for all species were combined for comparison with a known fluoride gradient on Anglesey, North Wales, and an estimate of the regional occurrence of fluoride in predatory birds in the British Isles. Lower-than- average fluoride loads occurred in most of the study area, but higher-than-average loads were reported in more industrial regions of England. Although females weighed more in all species, fluoride bone concentrations were found to be greater in males of all species examined. Seel, Thomson, and Bryant report that fluoride does tend to bioaccumulate in birds, and therefore might have secondary or tertiary food chain effects.

Conflicting information was found concerning fluoride accumulation in food chains. Slooff, et al. report that exposure to high concentrations of fluoride causes terrestrial organisms to accumulate more fluoride than when exposed to background levels.[30] Both invertebrate and vertebrate predators accumulate higher levels of fluoride than herbivores. Thus, although the data are ambiguous, predator concentrations of fluoride are higher, indicating moderate biomagnification. However, Davison reports that although data are limited, it is generally assumed that fluoride concentrations do not increase with trophic level.[29]

Of the three main routes of fluoride transport to animals— inhalation, ingestion, and deposition to outer surfaces— inhalation is not usually considered an important route.[29] This is because the amount of fluoride taken up and retained by the lungs is small both in magnitude and in comparison to the other two routes. Although Davison reports that no analyses exist which allow the calculation of deposition rates on skin, wings, shells, etc., these routes may be important, especially because some particulate fluorides are known to be toxic and have been used as insecticides. The preening of bird coats, pollen collection from bee bodies, and coat-licking by cattle are all examples of how fluoride deposition can be taken up by animals. Variations in fluoride concentrations within plant parts can result in animal species with differing feeding niches ingesting different amounts of fluoride. For instance, animals feeding on nectar (e.g., adult butterfly), phloem (e.g., aphid), or whole leaf tissues (e.g., caterpillar) of the same plant are likely to ingest varying amounts of fluoride. Invertebrates have several attributes which render them more suitable for fluoride pathway analysis and transfer rate analysis than vertebrates, such as smaller feeding areas and smaller variation in longevity between trophic level.

Contamination of foliage with soil that has been treated with phosphate fertilizer or exposed to substantial airborne deposition of fluoride may constitute an important route of fluoride transfer to large herbivores.[29] Davison reported that, as of 1987, virtually nothing was known about fluoride transport to invertebrates or detritus chain organisms. However, one study on this topic was found during this review and is discussed below in the section on toxicity of fluoride.

As with human studies, animal studies of the relationship between treatment with fluoride and bone strength have revealed a conflicting picture. Whitford concludes that when considering the extensive literature on the subject of fluoride and bone fractures, it appears that fluoride has little or no effect on bone strength.[38]

A large amount of literature is dedicated to the subject of fluoride toxicity to animals. Studies reviewed pertain to mice, voles, isopods, rats, and rabbits. Several studies have been conducted on the effects of fluoride, both alone and in conjunction with other compounds/pollutants, on small mammals. Lead accumulation, absorption, and assimilation can be affected by the presence of various other dietary components, including fluoride. For instance, in a study conducted by Cooke, Andrews, and Johnson, laboratory wood mice (*Apodemus sylvaticus*) were fed solutions containing soluble salts of lead, zinc, cadmium, and fluoride in their drinking water.[49] Fluoride concentrations showed their typical accumulation in femur tissue. The highest mean Pb concentrations in liver, kidney, and femur were found in the treatment comprised of all four elements. High levels of Pb and fluoride were found to thwart Zn’s antagonistic effect on the accumulation of Cd in kidneys of wood mice. The authors note that although other studies have found Pb to have a small but significant antagonistic effect on bone fluoride accumulation, this study did not.

In the high Zn treatment, greater femur fluoride levels were observed but not believed to be of any biological significance. The treatment combining Pb, Zn, Cd, and fluoride reflected the dietary intake of wild animals living in the grasslands established on fluorspar wastes and yielded the highest Pb concentrations in all three tissue types, the highest kidney and femur Cd levels, and the highest kidney fluoride levels. The observed reversal of Zn antagonism on Cd kidney levels in the treatment group exposed to all four elements is important because of Cd’s renal toxicity. The authors urge caution in extrapolating these lab results to the field because the availability of soluble forms of these elements in field situations is unknown. This research indicates the complexity of the interactions among trace elements in small mammals.

Boulton, Cooke, and Johnson conducted an investigation into the effects of fluoride toxicity on small mammals by examining populations of two species of vole (*Microtus agrestis* L. and *Clethrionomys glareolus* L.) at three different types of contaminated sites.[50] Vegetation was collected and analyzed at each site as a means of estimating fluoride dose to resident voles. Fluoride uptake by new vegetation was relatively low, even when hydrochloric acid was used to mimic extraction by vole stomach acids. This suggests that because fluoride solubility is moderate, so is assimilation by both small mammals. Study results suggest that the order of assimilation of dietary fluoride by location was: reference site < tailings dam < smelter < chemical works. Fluoride concentrations in the teeth and bones of both vole species were significantly higher at each of the three contaminated sites than at the reference site. The chemical plant and aluminum smelter both experienced a combination of hydrogen fluoride gas and fluoride-laden dust deposition. Both of these sites had considerably higher fluoride concentrations than the tailings dam.

Fluorides taken up by leaf stomata as hydrogen fluoride gas and/or from deposition on leaf surfaces clearly contributed to higher water- and acid-soluble fluoride in vegetation at the atmospherically contaminated sites. Both species of vole exhibited severe incisor and molar dental lesions and had significantly higher tissue fluoride concentrations at the chemical plant and aluminum smelter sites; the damage at the mine tailings dam was less pronounced. The variation in effects at different types of sites was believed to be the result of differences in fluoride speciation and its impacts on fluoride availability for bioassimilation. This study indicates that small mammals can experience severe dental lesions at sites atmospherically contaminated by fluoride and can be effective indicators of environmental contamination.

Another possible impact that fluoride can have on terrestrial animals is reduction of the vitality of the decomposer community and its ability to carry out the decomposition function. Van Wensem and Adema investigated fluoride’s impact on soil fauna-mediated decomposition and survival and growth of isopods (as represented by *Porcellio scaber*) and found neither to be adversely affected by soil fluoride concentrations of up to 170 micro mol per gram.[51] A previous study by Buse found fluoride to accumulate in invertebrates.[52] However, in this study fluoride did have a significant adverse effect on the extractable ammonium, nitrate, and phosphate concentrations of a terrestrial micro-ecosystem and it was concluded that fluoride is toxic to microbial processes at concentrations found at moderately polluted sites. During weeks one through four of the study, the respiration rate and total carbon dioxide production were significantly increased, but only at the highest fluoride concentration. The order of sensitivity of mineralization processes to fluoride was found to be phosphate < ammonium < nitrate. The no–observed–effects concentrations for net mineralization of ammonium, nitrate, and phosphorus were 17, 5.3, and 53 micro mol fluoride per gram dry weight of litter, respectively. Another study by Wang and Bian found silkworms to be relatively sensitive to atmospheric deposition of fluoride.[53] High accumulation of substances in isopods without concomitant toxicity, such as the authors found in this study, has also been observed with heavy metals.

The combined effect of more than one pollutant on nutrient bioavailability is another consideration when evaluating fluoride’s impact on terrestrial animals. For example, Clerklewski and Ridlington’s examination of the influence of dietary soluble (lead acetate) and insoluble (lead carbonate) lead on fluoride bioavailability in rats found depressed weight gain at both high and low fluoride doses, and significant reductions in femur and second molar fluoride at the higher fluoride dose.[54] Fluoride failed to influence the increased lead concentration in the plasma, femur, liver, and kidney, as well as the increased excretion of delta-aminolevulinic acid. The overall conclusion is that dietary lead may reduce fluoride bioavailability, but fluoride does not influence lead utilization. Cerklewski and Ridlington were unable to determine if the reduced fluoride bioavailability was due to decreased absorption or retention. It is valuable to note this combined effect of dietary fluoride and lead, as environmental pollutants are commonly found in mixtures.

Susheela and Bhatnagar investigated fluoride’s toxicity to rabbit teeth.[55] The authors examined the effects of long-term fluoride toxicity on rabbit teeth morphology and inorganic chemical (fluoride, calcium, phosphorous) composition. Researchers found a significant increase in fluoride and a significant decrease in calcium levels in both the 18- and 23-month dose groups; however, the calcium/phosphate ratio changed significantly only in the 23-month group. The scanning electron microscope analysis showed hypoplastic, rough, uneven, pitted, and cracked enamel covered with granular deposits. Overall, the study shows that chronic fluoride ingestion causes changes in the structural and biochemical constituents of rabbit tooth enamel. This research is valuable because it considers the effects of chronic exposure to a wildlife species since the exposure of wildlife to environmental fluoride pollution may be chronic in nature.

Fluoride can sometimes ameliorate the toxicity of other compounds. For instance, Pleasants, et al. explored the effects of several vitamins and fluoride on cadmium (Cd^{2+}) toxicity in rats.[56] They found that fluoride reduced the weight depression associated with cadmium toxicity and lowered the relative weights of rat testes. The combined dose of fluoride and D hormone was found to significantly increase the dry and mineral femur weights of cadmium-exposed rats, which may be of interest in the treatment of osteoporosis. A group of rats treated with fluoride in conjunction with vitamins A and 1,25-dihydroxyvitamin D_3 appeared to have experience reduced Cd^{2+} hematoxicity. In another study, Yu, et al. examined whether high dietary fluoride inhibits selenite toxicity in rats.[57] The significant effect shown in this experiment was the use of fluoride to prevent Se liver pathology. However, other considerations such as Se concentrations in plasma and kidney, depressed weight gain, and the enzymatic activity of glutathione peroxidase and xanthine oxidase were unaffected by fluoride. The authors suggest the most likely mechanism for fluoride protection from Se liver pathology is the formation of an insoluble Ca and F complex, which prevents excess intracellular Ca (produced as an effect of Se toxicity) from activating phospholipases and proteases. The possibility of fluoride use in the prevention of Se toxicity may be of interest to EPA; however, the results of this paper are not viewed as substantial enough to make the claim that fluoride dosage is beneficial for this purpose. Further research would be necessary to determine with certainty whether fluoride would be useful for this purpose.

Zeiger, et al. conducted an extensive review of fluoride genetic toxicity publications, presenting many articles both supporting and denying genetic toxicity.[58] The review covers microbial, *in vitro* mammalian, and *in vivo* mammalian test systems. The review of *in vitro* tests includes assays for mutation, chromosome aberrations, sister chromatid exchanges, DNA damage/repair, and cell transformation for both animal and human cell lines. The *in vivo* tests reviewed were concerned with the somatic and germ cells of various rodents. The article states that, based on the conflicting evidence from many of the assays listed above, the mechanisms of fluoride toxicity are purely speculative. In an attempt to summarize the conflicting evidence, the authors find that the fluoride ion is not mutagenic in standard bacterial systems but can cause some of the lesions examined *in vitro*. Furthermore, the issue of *in vivo* genetic toxicity should be considered unresolved. This review leaves the question of the genetic toxicity of fluoride open for interpretation. The authors provide many published assays containing conflicting evidence in similar or identical test systems, which means that interpretation must be based on weight of evidence considerations.

Davison reports that the first detectable signs of fluorosis occur in animals consuming herbage exposed to airborne fluoride concentrations in excess of approximately 0.3 $\mu g/m^3$ to 0.5 $\mu g/m^3$ for a continuous period of several months. If higher concentrations are present, more severe fluorosis symptoms will occur more rapidly.[29]

Whitford’s summary of a recent USPHS report on fluoride concludes that data from animal studies have not established an association between fluoride exposure, even extremely high and life-long exposure, and cancer.[38]

To summarize, for absorbed fluoride, kidney excretion is likely a minimal removal pathway as compared to deposition in bones and teeth. Differential absorption and retention observed in metabolic cage studies of mice and voles was attributed to different intake levels. Bone fluoride concentrations were found to be greater in male than in female raptors. Conflicting information was found concerning whether fluoride accumulates in food chains. Variations in fluoride concentrations within a given species of plant can result in animal species with differing feeding niches ingesting different amounts of fluoride. Contamination of foliage with soil may constitute an important route of fluoride transfer to large herbivores in situations where soil has been treated with phosphate fertilizer or exposed to substantial airborne deposition of fluoride. It appears that fluoride has little or no effect on bone strength. Voles living near a mine tailings dam, an aluminum smelter, and a chemical plant were found to exhibit severe incisor and molar dental lesions and significantly higher tissue fluoride concentrations at the chemical plant and aluminum smelter sites, than at the mine tailings dam. High soil fluoride concentrations were not found to adversely affect soil fauna-mediated decomposition or the survival and growth of isopods, but were found to be toxic to microbial processes at concentrations found at moderately polluted sites. Limited evidence was found that dietary lead may reduce fluoride bioavailability, but fluoride does not influence lead utilization. Fluoride can sometimes ameliorate the toxicity of other compounds. Limited evidence also exists showing that high dietary fluoride inhibits selenite toxicity in rats. The subject of fluoride's genetic toxicity remains controversial. Animal consumption of foliage exposed to 0.3 $\mu g/m^3$ to 0.5 $\mu g/m^3$ concentrations of fluoride for a continuous period of several months may cause fluorosis. Finally, data from animal studies have not established an association between fluoride exposure, even extremely high and life-long exposure, and cancer.

H.3.9 HF Aquatic Animal Impacts

According to Slooff, et al., bioconcentration factors of less than 10 have been calculated for crustaceans and fish exposed to fluoride concentrations of up to 50 mg/L.[30] Retained fluoride is usually stored in skeletal structures (e.g., bones and shells), with the lowest fluoride levels observed in muscle tissues. Based on limited data, it was concluded that biomagnification is negligible to very slight.

Fluoride aquatic toxicity literature pertaining to trout, benthic macroinvertebrates, minnows, water fleas, and diatoms was reviewed for this study. Literature on both hard and soft water environments, as well as both acute and chronic toxicity, was investigated as available.

In the first of two studies by Camargo and Tarazona examining the effects of fluoride ion (F^-) on aquatic species, two species of trout exposed to NaF in soft water were found to have 120-, 144-, 168-, and 192-hour LC_{50} values of 92.4, 85.1, 73.4 and 64.1 ppm F^- (rainbow trout) and 135.6, 118.5, 105.1, and 97.5 ppm F^- (brown trout), respectively.[59] Subject fish in fluoride-containing aquaria exhibited hypoexciteability, darkened backs, and a decrease in respiration before death. Trout mortality increased as fluoride concentrations and exposure times increased. Rainbow trout fingerlings were found to be significantly more sensitive to F^- than brown trout fingerlings of similar age and weight. The higher resistance of brown trout to fluoride may be due to their greater physiological ability to inhibit the toxic action of fluoride on groups of enzymes within cells or through removing or immobilizing fluoride ions more effectively. Both trout species appear to be more resistant to fluoride than freshwater benthic macroinvertebrates. The authors suggest the possibility that fluoride ions form stable complexes with calcium in the blood and bones of fish, but not in freshwater insect larvae, as one explanation for the lesser sensitivity of fish to fluoride. Freshwater fish may be more resistant to high fluoride concentrations in hard water than in soft water. It is possible that the reservoir of calcium that surrounds fish in hard water may compensate for the loss of calcium, thereby delaying fluoride’s toxic effects.

Fluoride concentrations in sea water typically range from 1.2 to 1.4 mg/L, while most fresh waters contain less than 0.2 mg/L of fluoride. Freshwater fluoride concentrations are believed to be rising due to industrial pollution. In a second study of the acute toxicity of fluoride to aquatic species, Camargo and Tarazona found that five species of freshwater benthic macroinvertebrates (*Chimarra marginata* Linnaeus, *Hydropsyche lobata* MacLachlan, *Hydropsyche bulbifera* MacLachlan, *Hydropsyche exocellata* Dufour, and *Hydropsyche pellucidula)* were found to be more sensitive to fluoride than the aquatic species previously investigated in other studies.[60] These five freshwater benthic macroinvertebrates, all native to the rivers of the Iberian Peninsula, were found to have 96-hour LC_{50} values of 44.90, 48.20, 26.30, 26.50, and 38.50 mg/L F^-, respectively. Macroinvertebrate mortality was observed to increase as fluoride concentrations increased. The sensitivity of the five species investigated indicates their possible usefulness as aquatic fluoride bioindicators in soft water environments. The more sensitive species investigated might, therefore, be useful in setting fluoride water quality criteria in fresh water.

Khan, et al. used a solution containing 1.5 percent hydrofluozirconic acid and 1.0 percent of ammonium bifluoride to determine the chronic toxicity of a fluoride mixture to the fathead minnow (*Pimephales promelas*) and water flea (*Ceriodaphnia dubia*).[61] Three toxicity tests were conducted, two with the solution at a pH of 2.5 and a third with pH adjusted to 7.5. The endpoints used were survival, reproduction, and/or growth. After a 7-day exposure period, fathead fry had an LC_{50} of 2.26 percent, a survival NOEC of 1.25 percent, and a growth NOEC of 0.625 percent. After an 8-day exposure period, water fleas had an LC_{50} of 3.11 percent, a survival NOEC of 2.5 percent, and a reproduction NOEC of 0.625 percent. When the solution’s pH was adjusted to 7.5, fathead minnows had a 7-day LC_{50}, a survival NOEC, and a growth NOEC all of <6.25 percent. The article does not state why these particular solution components were chosen. Although the adjustment of pH from 2.5 to 7.5 effectively ruled out acidity as the causal agent in the observed mortality, the presence of zirconium ions in the solution precluded a definitive assessment of whether the remaining mortality effect was due to fluoride or zirconium ions.

Joy and Balakrishnan studied the effects of fluoride ion on cultures of diatoms *Nitzschia palea* (freshwater) and *Amphora coffeaeformis* (brackishwater) in a laboratory setting.[62] The number of cells of both organisms and the amount of chlorophyll *a* and *c* were increased, with greater concentrations of fluoride. Statistical analysis showed that fluoride concentrations above 10 mg/L resulted in significant stimulation of growth in terms of cell number and chlorophyll in all tested concentrations. The results indicate that these diatoms can tolerate and are stimulated to grow by high fluoride concentrations; however, the ecological significance of this would be varied because of fluoride's propensity to transfer easily to animal populations. Therefore, the impact of fluoride pollution in waterbodies is still to be viewed with caution.

Slooff, et al. report that field studies of the toxicity of fluoride-containing effluent to aquatic animals resulted in effects on both abundance and diversity of estuarine/marine organisms at relatively low fluoride levels.[30] However, it was unclear to what extent other effluent parameters such as pH and other substances were affecting the organisms.

To summarize, biomagnification in aquatic animals is reported to be negligible to very slight. Two trout species were demonstrated to be more resistant to fluoride than freshwater benthic macroinvertebrates. Diatoms appear to be tolerant of, and stimulated to grow by, high fluoride concentrations, with uncertain ecological significance. Finally, limited evidence exists for fluoride-containing effluent effects on both abundance and diversity of estuarine/marine organisms at relatively low fluoride levels.

H.4 FUTURE RESEARCH

The following topics, discovered in the process of this research, were found either not to be covered in the available literature, or to be unresolved to some degree. Further research on these topics can be expected to yield information important to gaining a better understanding of the impacts of HCl and HF emissions from utilities.

1. ***What are HCl's impacts on the spatial and temporal deposition of SO_x, a species implicated in both acid rain and $PM_{2.5}$?***

   SO_2 and HCl interact in the atmosphere. For instance, chloride ions can catalyze the oxidation of sulfite to sulfate when present in concentrations typical of plumes containing HCl. The net effect of this change is that less SO_2 will be dry-deposited close to the source, but more H_2SO_4 will be generated in cloud and rain water, subject to further transport. HCl might, therefore, alter the spatial and temporal deposition of SO_2, which may be an important way in which utility HCl emissions indirectly affect acid rain and $PM_{2.5}$.

2. ***Do HCl and/or HF have an impact on the speciation and deposition of other HAPs?***

   Atmospheric HCl can affect the atmospheric chemistry of other HAPs, such as mercury. In a theoretical study based on their chemical kinetic model, Selgneur, Wrobel, and Constantinou found that HCl concentration, among other factors (e.g., atmospheric liquid water content, pH, and SO_2 concentration), had a strong effect on both the Hg(0)/Hg(II) ratio and the gas/liquid partitioning of Hg(0).[14]

Although the effect of this change may be considered positive—shifting more mercury to a less toxic form—this might not always be the case with mercury or with other HAPs emitted by utilities.

3. ***What percentage of ambient HCl and HF particulate is in the fine fraction?***

EPA has promulgated a primary 24-hour $PM_{2.5}$ standard of 65 $\mu g/m^3$ and a primary annual $PM_{2.5}$ standard of 15 $\mu g/m^3$. As part of the effort to meet these standards, it will be important to know what the contribution of HCl and HF are to ambient $PM_{2.5}$ concentrations.

The literature evaluated for this paper contained several references to ambient HCl and HF concentrations, both rural and urban. Urban HCl concentrations were found to reach as high as 4 $\mu g/m^3$. Information on rural HF concentrations was more scarce, with a reported high end concentration of 1 $\mu g/m^3$. However, accidental industrial releases of HF can produce short-term concentrations of up to 50 $\mu g/m^3$. Davison reports fluoride particulates range in diameter from <0.1 μm to approximately 10 μm. If a significant portion of ambient HCl and HF are in the fine fraction, they could conceivably contribute to $PM_{2.5}$ exceedances.

4. ***Are HCl and HF capable of being transported large distances? How far are they typically transported and in what regions, if any, do they reach sensitive receptors?***

Several references to HCl and HF transport were found. A key question is whether HCl/HF-enriched air parcels reach acid rain or $PM_{2.5}$-sensitive receptors and if so, how long this process takes. The literature reports atmospheric HCl lifetimes of between 1 and 5 days and HF lifetimes of 0.54 to 5 days. Future research might be directed at clarifying the extent of atmospheric transport of these two species.

5. ***What are the interactive effects of fluoride and other pollutants?***

Interactions between trace elements in small mammals can be complex. For example, Cooke, Andrews, and Johnson observed that high concentrations of one or more pollutants can suppress another pollutant’s antagonism of a third pollutant’s toxicity, such as in the case of fluoride, Pb, Zn, and Cd with wood mice.[49] And although the evidence is far from conclusive, fluoride has been shown in at least one case to prevent Se toxicity.

The combined effect of more than one pollutant on nutrient bioavailability is another consideration when evaluating fluoride’s impact on terrestrial animals. Cerklewski and

Ridlington found that dietary Pb may reduce fluoride bioavailability, but that fluoride does not influence Pb utilization.[54] Further examination of HF and HCl emissions may be warranted to elucidate subtle ecosystem effects.

6. ***Are ambient concentrations of fluoride toxic to microbial processes?***

Van Wensem and Adema found that fluoride had a significant adverse effect on the extractable ammonium, nitrate, and phosphate concentrations of a terrestrial micro-ecosystem, and concluded that fluoride is toxic for microbial processes at concentrations found at moderately polluted sites.[51] If other research corroborates these findings, some elements of the detritus cycle may be inhibited at such sites.

7. ***Are Al/F complexes phytotoxic?***

The most recent information found (1996) states that the question of whether Al/F complexes are phytotoxic has not been clarified. Bond, et al. found that fluoride forms strong complexes with Al in soil.[36] Klumpp, et al. found strong indications that the resistance of some tropical tree species may be related to their capacity to accumulate Al.[42]

8. ***Does fluoride accumulate with trophic level?***

Conflicting information was found concerning whether fluoride accumulates in food chains. Slooff, et al. report (1989) that exposure to high concentrations of fluoride causes terrestrial organisms to accumulate more fluoride than when exposed to background levels, with both invertebrate and vertebrate predators accumulating higher levels of fluoride than herbivores.[30] Davison, however, reports (1987) that although data are limited, it is generally assumed that fluoride concentrations do not increase with trophic level.[29] Resolving the question of biomagnification is essential to evaluating the ecological impacts of HF and HCl.

9. ***Information on the following HCl topics was not available or not found.***

Information pertaining to several topics concerning HCl was scarce. The following topics, in addition to those listed above, would be good candidates for future research:

1. Terrestrial stability and persistence of HCl and by-products;
2. terrestrial HCl transport, mobility, and partitioning;
3. aquatic stability and persistence of HCl and by-products;

4. terrestrial animal HCl ingestion, inhalation, dermal absorption, distribution, accumulation; and

5. aquatic animal HCl ingestion, distribution, and accumulation.

H.5 GLOSSARY

Biomagnification -	The tendency of a substance to accumulate through a food chain. Occurs when substances stored in the tissues of a large number of organisms at lower trophic levels are taken in and stored by a predator at a higher trophic level.
Chlorosis -	A symptom of disease or disorder in plants, which involves a reduction in or loss of the normal green coloration. Affected plant(s) will be pale green or even yellow. Chlorosis is caused by conditions that prevent the formation of chlorophyll.
Fluorosis -	A disease that results from the ingestion of fluorine in amounts that substantially exceed bodily requirements. In excess, fluorine leads to the thickening of bones, sometimes to the extent that joints stiffen and the increased weight of the skeleton makes it difficult for the head to be raised. Teeth may also be softened and stained.
Hornblende -	A mineral, $CaNa(Mg,Fe)_4(Al,Fe,Ti)_3Si_6O_{22}(OH,F)_2$, commonly green to black in color, formed late in the cooling of igneous rock.
Interspecific -	Arising or occurring between species.
Solubility product -	The product of the molar concentrations of the constituent ions, each raised to the power of its stoichiometric coefficient in the equilibrium equation.

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Appendix I - Mercury Control Technologies

This Appendix provides information extracted from the Mercury Study Report to Congress, Volume VIII: An Evaluation of Mercury Control Technologies and Costs. EPA-452/R-97-010. December 1997 and is provided for the readers information. Appendix I contains text elements provided by the Department of Energy regarding resent research on mercury controls for electric utility steam generating units.

I.1 INTRODUCTION

I.1.1 Coal Cleaning

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

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

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

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

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

I.1.2 Flue Gas Treatment Technologies

Most metals have sufficiently low vapor pressures at typical air pollution control device operating temperatures that condensation onto particulate matter is possible. Mercury, on the other hand, has a high vapor pressure at typical control device operating temperatures, and collection by particulate matter control devices is highly variable. Factors that enhance mercury control are low temperature in the control device system (less than 150°C [300 to 400°F]), the presence of an effective mercury sorbent, and a method to collect the sorbent. In general, high levels of carbon in the fly ash enhance mercury sorption onto particulate matter which is subsequently removed by the particulate matter control device. Additionally, the presence of hydrogen chloride (HCl) in the flue gas stream can result in the formation of mercuric chloride ($HgCl_2$), which is readily adsorbed onto carbon-containing particulate matter. Conversely, sulfur dioxide (SO_2) in flue gas can act as a reducing agent to convert oxidized mercury to elemental mercury, which is more difficult to collect.

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

- Wet scrubbing;
- Treated activated carbon adsorption; and
- Activated carbon injection.

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

- Factors that enhance mercury control are low temperature in the control device system (less than 150°C [300 to 400°F]), the presence of an effective mercury sorbent and a method to collect the sorbent. In general, high levels of carbon in the fly ash enhance mercury sorption onto particulate matter which is subsequently removed by the particulate matter control device. Additionally, the presence of HCl in the flue gas stream can result in the formation of $HgCl_2$, which is readily adsorbed onto carbon-containing particulate matter, so it can be efficiently scrubbed by a wet FGD system. Conversely, sulfur dioxide (SO_2) in flue gas can act as a reducing agent to convert oxidized mercury to elemental mercury, which is more difficult to collect.

- Control technologies designed for control of pollutants other than mercury (e.g., acid gases and particulate matter) vary in their mercury-removal capability, but in general achieve reductions no greater than 50 percent (except for high removal efficiencies for $HgCl_2$ by wet scrubbers).

- Selenium filters are a demonstrated technology in Sweden for control of mercury emissions from lead smelters. Carbon filter beds have been used successfully in Germany for mercury control on utility boilers and MWCs. These technologies have not been demonstrated in the U.S. for any of these source types.

- Injection of activated carbon into the flue gas of MWCs and MWIs can achieve mercury reductions of at least 85 percent. The addition of activated carbon to the flue gas of these source types would not have a significant impact on the amount of particulate matter requiring disposal.

- No full-scale demonstrations of mercury controls have been conducted in the U.S. for utility boilers. Based on limited pilot-scale testing, activated carbon injection provides variable control of mercury for utility boilers (e.g., the same technology might capture 20 percent of the mercury at one plant and 80 percent at another). The most important factors affecting mercury control on utility boilers include the flue gas volume, the flue gas temperature and chloride content, the mercury concentration, and the chemical form of mercury being emitted.

- The chemical species of mercury emitted from utility boilers vary significantly from one plant to another. Removal effectiveness depends on the species of mercury present. To date, no single control technology has been identified that removes all forms of mercury.

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

I.2 MERCURY CONTROLS

This section provides information on mercury controls that provide opportunities for significant further reductions of mercury emissions. Two major types of control techniques are described:

- Coal cleaning; and
- Flue gas treatment technologies.

I.2.1 Coal Cleaning

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

Conventional coal cleaning methods are based on the principle that coal is lighter than the pyritic sulfur, rock, clay, or other ash-producing impurities that are mixed or embedded in it. Mechanical devices using pulsating water or air currents can physically stratify and remove impurities. Centrifugal force is sometimes combined with water and air currents to aid in further separation of coal from impurities. Another method, dense media washing, uses heavy liquid solutions usually consisting of magnetite (finely ground particles of iron oxide) to separate coal from impurities.

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

I.2.1.1 Advanced Coal Cleaning. Advanced coal cleaning methods such as selective agglomeration and column froth flotation have the potential to increase the amount of mercury removed by conventional cleaning alone. In one bench-scale study, five types of raw coal were washed by conventional cleaning methods followed by column froth flotation or selective agglomeration. Conventional cleaning and column froth flotation reduced mercury concentrations from the raw coals by 40 to greater than 57 percent, with an average of 55 percent (Smit, 1996). Column froth flotation reduced mercury concentrations remaining in the washed coals by 1 to greater than 51 percent, with an average of 26 percent (Smit, 1996). Conventional cleaning and selective agglomeration reduced mercury concentrations from the raw coals by greater than 63 percent to 82 percent, with an average of 68 percent (Smit, 1996). Selective agglomeration reduced mercury concentrations remaining in the washed coals by greater than 8 percent to 38 percent, with an average of 16 percent (Smit, 1996).

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

Bench-scale testing is also being carried out by DOE to investigate the use of naturally occurring microbes to reduce the mercury (and other trace elements) from coal.

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

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

I.2.1.2 Commercial Status. As mentioned above, approximately 77 percent of the Eastern and Midwestern bituminous coal is cleaned to meet customer specifications for heating value, ash content, and sulfur content. While most of this coal is cleaned by conventional cleaning methods, advanced cleaning methods, such as column froth flotation, are starting to emerge. Microcel™ is a type of column froth flotation available through ICF Kaiser Engineers and Control International. The company is the exclusive licensee for the technology in the coal fields east of the Mississippi River and has sold units for commercial operation in Virginia, West Virginia, and Kentucky, as well as in Australia under sub-license to Bulk Materials Coal Handling Ltd. Ken-Flote™ is another type of column froth flotation cell.

I.2.2 Fuel Switching

Fuel switching refers to switching from one fuel to another (e.g., high-sulfur coal to low-sulfur coal, or coal to natural gas) to achieve required emission reductions in a more flexible or cost-effective way. For example, coal-fired utilities might switch to natural gas during the high ozone season in the Northeast, or to achieve reductions in greenhouse gases. This would also lower their mercury emissions. In addition, installing pollution control equipment may not be cost-effective for sources that are not affected by Title IV regulations, which are generally smaller than affected utilities. Given the economic benefits of the opt-in program, fuel switching can be more cost-effective for such smaller sources.

I.2.3 Flue Gas Treatment for Utility Boilers

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

I.2.3.1 Activated Carbon Injection for Utility Boilers. The effectiveness of activated carbon injection in controlling mercury emissions from MWCs has been demonstrated (U.S. EPA, 1989a; U.S. EPA, 1989b). The application of activated carbon injection to utility flue gas, however, cannot be directly scaled from the application at MWCs due to differences in the amount and composition of flue gas at utility plants and MWCs. At utility plants, small concentrations of mercury are contained in a large volume of flue gas, and large amounts of activated carbon are needed to provide adequate contact between the carbon particles and mercury. The differences in flue gas characteristics at MWCs and utility plants must be carefully examined before considering any technology transfer assumptions.

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

As shown in Table I-1, typical MWC flue gas is hotter than utility flue gas after leaving an air preheater. The air preheater cools the utility flue gas by transferring heat to the incoming combustion air. Moreover, the mercury concentration of the two gas streams differs significantly. Mercury concentrations in MWC flue gas streams may be up to several orders of magnitude greater than those seen in utility flue gas streams. Likewise, the chloride content of MWC flue gas may be from 1.4 to 400 times greater than the content seen in utility flue gas. Finally, with regard to the volume of flue gas, a utility boiler may have flow rates up to 30 times that of an MWC.

Because of differences in the amount and composition of flue gas at utility plants and MWCs, pilot-scale studies of activated carbon

Table I-1. Comparison of Typical Uncontrolled Flue Gas Parameters at Utilities and MWCs[a,b]

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

[a] Standard conditions are 0°C and 1 atmosphere.
[b] Moisture content in the MWC flue gas was assumed to be 13.2 percent.
[c] Radian Corporation, 1993a, UNDEERC, 1996, CONSOL INC, 1997.
[d] Heath, 1994.
[e] Radian Corporation, 1994.
[f] Radian Corporation, 1993b.
[g] Brown and Felsvang.
[h] Nebel and White, 1991.
[i] It is not known if oil-fired utility boilers release less mercury overall than coal-fired boilers because the mercury release during oil refining is essentially unstudied.
[j] Min = minute

injection were conducted on utility flue gas where the nominal concentration of mercury is one part per billion and may have a wide range of distribution between the different forms of mercury. Preliminary results from a limited number of pilot-scale tests on utility flue gas are summarized in Figure I-1 and presented in greater detail in section I.2.3.2. These data indicate that the effectiveness of activated carbon injection varies with several factors. The mercury removal efficiency for fabric filter and activated carbon systems ranged from a low of 14 to 47 percent with a median of 29 percent (107-121°C, low carbon injection) to a high of 95 to 99 percent with a median of 98 percent (88-107°C, high carbon injection). When activated carbon injection was used ahead of a spray dryer absorber, mercury removal efficiency ranged from 50 to 99 percent with a median of 60 percent when a fabric filter was used for particulate control, and from 75 to 91 percent with a median of 86 percent when an ESP was used for particulate control.

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

I.2.3.1.1 Utility Flue Gas Factors Affecting Mercury Removal by Activated Carbon Injection. The level of mercury control achieved in utility flue gas depends on the temperatures upstream and within the existing APCDs, residence time (e.g., extent of contact between the carbon and flue gas mercury) upstream and within the APCDs, volume of flue gas, flue gas vapor and particulate phase constituents (i.e., chlorine as HCl, nitrogen oxides, sulfur oxides, metal oxides on the

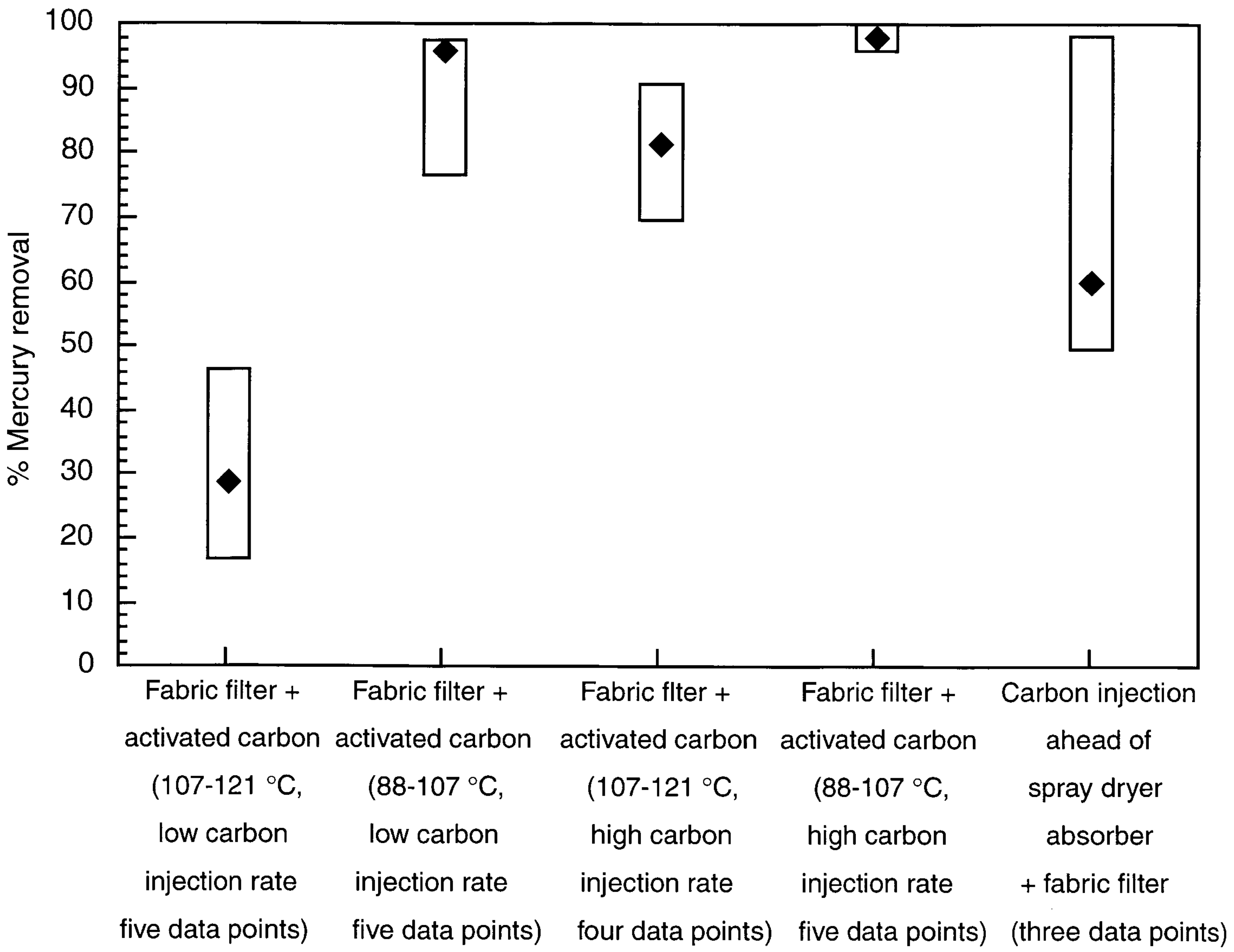


Figure I-1. Comparison of Mercury Removal Efficiencies with Activated Carbon Injection

surfaces of particulate matter, fly ash composition, percent carbon in fly ash, etc.), their interactions with the various types of carbon(s)/sorbent(s), and the mercury concentration and chemical species being formed.

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

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

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

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

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

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

As indicated, the mercury removals of the native fly ash at these conditions are not typical of the past and more recent field characterizations and pilot-scale mercury technology investigations.

This utility site is proposed to be further characterized in mid-1998 with the more precise Ontario Hydro mercury speciation method. In addition, in-flight capture of mercury will be investigated upstream of the baghouse along with the baghouse removals (DOE/FETC et al, Phase II, 1997). Currently, laboratory tests are being conducted on the fly ash under simulated flue gas conditions to provide some insight into the factors influencing high elemental mercury capture at nominal flue gas temperatures of 149°C (300°F) (U.S. DOE/FETC R&D, 1997).

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

The pilot-scale activated carbon injection studies indicated that more mercury was removed and less carbon was needed at lower flue gas temperatures; in other words, the ideal use of activated carbon may be at lower flue gas temperatures. It may not be possible, however, to lower the flue gas temperature at a given utility plant because utility plants typically operate with a stack gas temperature between 121 and 177°C (250 to 351°F) upstream of any particulate control device to avoid acid condensation and, consequently, equipment corrosion. The stack gas temperature may be lowered below 96°C (205°F) and acid condensation may be avoided provided low-sulfur coals (less than about 1 weight percent sulfur) are burned, but it may depend on whether the coal is a subbituminous or a bituminous coal (McKenna and Turner, 1989; ABB et al, 1996 and 1997; PSCO/ADA Technologies, Inc. et al, 1996 and 1997; Sjostrum et al, 1997; Haythornthwaite 1997; Radian et al, 1997; Carey et al, 1996 and 1997; Hargrove et al, 1997; Waugh et al, 1997). If a utility burns low-sulfur coal and uses an ESP for particulate control, however, the flue gas will probably require conditioning to reduce the high resistivity of the fly ash because high resistivity makes the fly ash difficult to collect with an ESP, but again, it is dependent on coal type.

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

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

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

<u>Volume</u>. At utility plants, mercury control techniques must adequately treat the entire volume of gas in order to remove relatively small concentrations of mercury (0.2 to 21 µg/dscm, at 7 percent O_2). High mass carbon-to-mercury ratios will be required due to a nominal one ppb of mercury being in different forms and being in the high flue gas volumes with competing vapor phase compounds at many orders of magnitude higher. Currently, mercury mass transfer limitations are encountered regardless of the type of coal, operating conditions, and APCD.

<u>Mercury Speciation and Type of Activated Carbon</u>. With a few exceptions, the total mercury concentration in coal is relatively constant across the United States (20 ppb to 120 ppb). However, when the different coals are fired in a combustor there are substantial variations in the concentrations of elemental versus ionic mercury. The percentage of Hg^0 is from near zero percent to >70 percent. The speciation then is very dependent on coal type. The chemical species of mercury formed during the combustion process and post-combustion conditions vary significantly from one plant to another. While combustion conditions vary, the subsequent fly ash, carbon in the ash, and vapor phase constituents may play a major role in the percentage of the chemical species of mercury formed. Understanding the rate controlling mechanisms (i.e. transport, equilibrium, and kinetics) will aid in predicting the species formed and eventually will aid in

optimizing existing APCDs for mercury removal. Kinetics may play more of a role on the form of mercury than anticipated. Depending on the type of coal utilized, effective removal may be dependent on the species of mercury present in the flue gas (Senior et al, June and November, 1997; PSI et al, 1997). For example, the ionic mercury form (i.e., Hg^{++}) is water soluble and is less volatile than elemental mercury (i.e., Hg^0). Thus, reducing the temperature of the flue gas and wet scrubbing of the flue gas may result in increased ionic mercury removal.

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

As shown in Table I-2, the distribution of ionic mercury, most likely $HgCl_2$ in coal-fired utility flue gas, ranged from 12 to 99 percent of the total mercury content and averaged 79 percent; the distribution of elemental mercury in coal-fired utility flue gas ranged from 0.8 to 87.5 percent of the total mercury content and averaged 21 percent. Analysis of two samples of flue gas taken from oil-fired boilers, however, suggests that mercury in oil-fired boiler flue gas is predominantly in the elemental form (see Table I-2). The variability in the speciation of vapor-phase mercury in coal-fired flue gas may explain the variation in mercury removal that is seen with existing control devices (DeVito et al, 1993).

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

In the pilot-scale work conducted at EPRI’s ECTC by Radian International and the pilot-scale work conducted by the UNDEERC for both EPRI and DOE, it was proven that U.S. EPA Method 29 does not properly speciate mercury under certain conditions (Hargrove et al, 1995; Laudal et al, 1996; Stouffer et al, 1996; Khosah et al, 1996;

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

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

(continued)

Table I-2. (continued)

µg/Nm^{3}[a]			Total[b] (µg/Nm^{3})	Percent		Reference
Ionic	Other Hg Species[c]	Elemental		Total Ionic[d]	Elemental	
2.99	---	1.51	4.5	66.44	33.56	g
2.83	---	0.88	3.71	76.28	23.72	g
3.33	---	1.26	4.59	72.55	27.45	g
3.83	---	1.52	5.35	71.59	28.41	g
2.2	---	1.57	3.77	58.36	41.64	g
	---	---	---	12.5	87.5	h
7.37	0.0021	1.85	9.22	79.92	20.07	i
5.01h	---	1.26	6.28	79.90	20.10	j
5.72	1.39	<0.1	7.10	98.61	1.39	j
4.49	0.40	0.89	5.78	84.60	15.40	j
2.93	0.07	0.16	3.15	94.94	5.06	j
2.79	0.32	1.02	4.13	75.30	24.70	j
5.71	1.37	2.99	10.22	70.31	29.69	j
7.01	1.02	0.11	8.13	98.65	1.35	j
4.11	0.62	2.32	7.05	67.09	32.91	j
1.14	0.30	3.12	4.56	31.58	68.42	j
COAL-FIRED BOILER DATA RANGES						
1.1 - 10.8	0.002 - 1.4	0.06 - 3.1	3.1 - 11	12 - 99	1.8 - 88	--

(continued)

Table I-2. (continued)

µg/Nm^3[a]			Total[b] (µg/Nm^3)	Percent		Reference
Ionic	Other Hg Species[c]	Elemental		Total Ionic[d]	Elemental	
OIL-FIRED BOILERS						
0.03	0.01	0.15	0.19	21.05	78.95	j
<0.02	0.01	<0.02	---	---	---	j
0.02	0.01	0.09	0.12	25.00	75.00	
OIL-FIRED BOILER DATA RANGES						
0.02 - 0.03	---	0.02 - 0.15	0.12 - 0.19	23	77	---

[a] µg/Nm3 = micrograms per normal cubic meter (at 0°C).

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

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

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

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

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

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

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

[i] Dismukes et al, 1993.

[j] Bloom et al, 1993.

and Laudal et al, December, 1997). Further studies at UNDEERC indicated this finding is related to high SO_2 concentrations with the method overestimating the ionic mercury up to 50 percent. Therefore, tests were conducted to evaluate a number of alternative sampling methods. Mercury speciation sampling methods that have been tested include the following:

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

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

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

The variability in the distribution of vapor-phase mercury species in coal-fired flue gas may depend upon the chloride concentration in coal. Using the analytical techniques developed by Bloom et al, (1993), it has been observed that higher concentrations of ionic mercury are obtained in utility flue gas when the combusted coal has a high chloride concentration (0.1 to 0.3 weight percent)

(Felsvang et al, 1993; Noblett et al, 1993), but more data are needed to verify this association. The distribution of mercury species in coal-fired flue gas also appears to vary with the type of coal (e.g., bituminous, subbituminous, or lignite) (Chang, 1994; Boyce, 1994; Laudal et al, 1996 and 1997; Redinger et al, 1996 and 1997; and DeVito et al, 1997).

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

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

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

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

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

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

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

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

Recently, researchers at the University of Pittsburgh developed a series of sulfur-impregnated carbons that exhibited high elemental mercury uptake efficiency at 140°C (284°F) when compared to commercially available activated carbons. Dynamic adsorption capacity of these carbons as high as 4,000 µg Hg/g was measured using a fixed-bed absorber with an empty bed contact time of 0.011 second and influent mercury concentration of 55 $\mu g/m^3$. This capacity is almost three orders of magnitude greater than the capacity of virgin activated carbon and an order of magnitude greater than the capacity of commercially available impregnated activated carbon. The comparisons were conducted at identical operating conditions using nitrogen as a carrier gas.

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

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

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

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

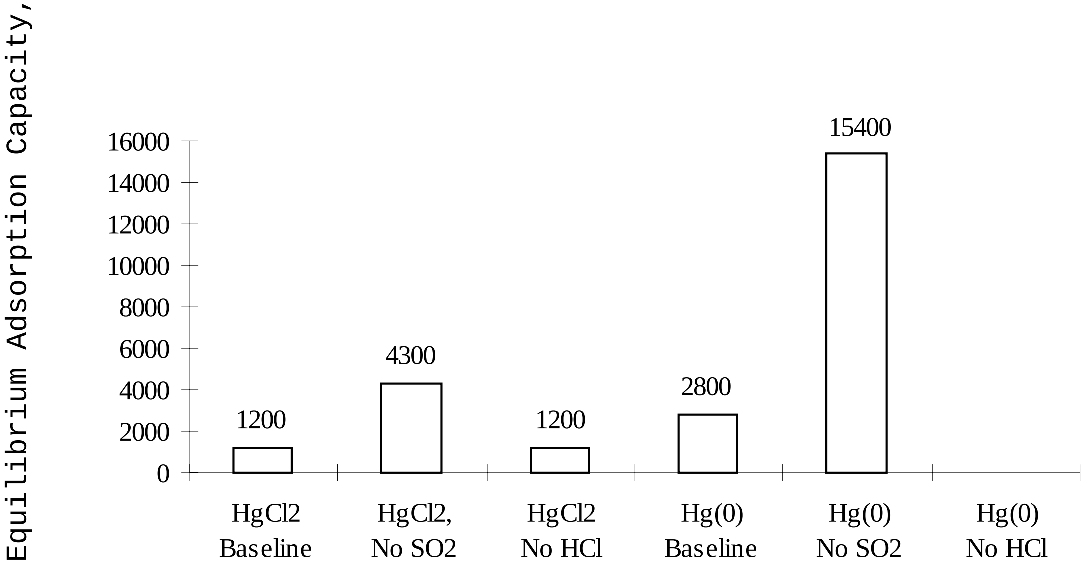
Studies of a pilot-scale wet FGD system treating coal-fired flue gas indicate that more than 90 percent of the ionic mercury was removed while hardly any of the elemental mercury was removed (Noblett et al, 1993; Redinger et al, 1996 and 1997; Carey et al, June and July 1996; Evans et al, 1996; and Hargrove et al, 1995 and 1997). Similarly, studies at a pilot-scale SDA/ESP system treating coal-fired flue gas suggest that 95 percent of the ionic mercury and essentially none of the elemental mercury were removed (Felsvang et al, 1993). The effectiveness of activated carbon injection in recovering different forms of mercury is still being studied. Preliminary results are available from the studies described in Section 2.3.1.2, Current Research on Activated Carbon Injection for Utilities.

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

A recent bench-scale study investigated the effects of SO_2 and HCl on the adsorption of elemental mercury and mercuric chloride ($HgCl_2$) by a lignite-based activated carbon (Carey et al, 1997). Equilibrium adsorption capacities were determined for fixed beds of the carbon at 275°F and three flue gas compositions: one containing 1,600 ppm SO_2 and 50 ppm HCl (the baseline composition); a second containing no SO_2 and 50 ppm HCl; and a third containing 1,600 ppm SO_2 and no HCl. (All three compositions of flue gas had the same concentration of elemental mercury, mercuric chloride, CO_2, water, and O_2).

Figure I-2 illustrates the effect of SO_2 and HCl on the equilibrium adsorption capacity of the lignite-based activated carbon for elemental mercury and mercuric chloride. Removing SO_2 from the flue gas increased the equilibrium adsorption capacities for both kinds of mercury (compared to the baseline capacities). The increase was particularly notable for the adsorption of elemental mercury. For example, after removing SO_2 from the flue gas, the equilibrium adsorption capacity for elemental mercury increased by a factor of about 5.5 compared to 3.5 for mercuric chloride.

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



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

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

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

I.2.3.1.2 Current Research on Activated Carbon Injection for Utilities. Previously, research was conducted on activated carbon injection at a facility with a pilot-scale SDA/ESP system in Denmark (Felsvang et al, 1993); at a facility with both a pilot- and full-scale SDA/FF system by Joy/Niro and Northern States Power (Felsvang et al, 1993); at a pilot-scale coal combustor and FF by Miller et al (1994 and 1995); and at a pilot-scale pulse-jet FF system at a utility power plant by EPRI (Chang et al, 1993). These results are presented in detail in section I.2.3.2. Preliminary results are available from the first three studies as described below.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

I.2.3.2 Test Data on the Effectiveness of Activated Carbon Injection for Utility Boilers.

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

Table I-3 presents pilot-scale test data on the mercury removal efficiency of AC injection when used ahead of FFs. Such a configuration, with no prior PM control, has a median mercury removal efficiency that varies with temperature and AC injection rate. With a low AC injection rate (<1,000 wt C/wt inlet Hg) and an average flue gas temperature between 107°C (225°F) and 121°C (250°F), a median mercury removal efficiency of 29 percent was found, with a range from 14 percent to 47 percent removal. With a low AC injection rate (same as above) and an average flue gas temperature between 88°C and 107°C, a median mercury removal efficiency of 97 percent was found, with a range from 76 percent to 99 percent removal. A high AC injection rate (>1,000 wt C/wt inlet Hg) and an average flue gas temperature between 107°C (225°F) and 121°C (250°F) produced a median mercury removal efficiency of 82 percent, with a range from 69 percent to 91 percent removal. A high AC injection rate (same as above) and an average flue gas temperature between 88°C (190°F) and 107°C (225°F) produced a median mercury removal efficiency of 98 percent, with a range from 95 percent to 99 percent removal (Chang et al., 1993).

Table I-4 presents test data for AC injection when used before SDA systems. Tested SDA/ESP systems with AC injection had a median mercury removal efficiency of 85.9 percent, with a range from 74.5 percent to 90.9 percent removal (Felsvang, 1993). Pilot-scale testing of a SDA/FF system with AC injection had a median mercury removal

Table I-3. Activated Carbon Injection Before Fabric Filter Data[a]

Unit	Control Device	Hg removal %
Low temperature + low carbon injection rate (< 1000 wt C/wt Hg) based on inlet Hg		
Test #4, Run #1	AC + FF (88°C (190°F) and 216 wt C/wt inlet Hg; inlet Hg concentration of 5.35 μg/dscm)	97
Test #4, Run #2	AC + FF (88°C (190°F) and 126 wt C/wt inlet Hg; inlet Hg concentration of 8.19 μg/dscm)	99
Test #4, Run #3	AC + FF (91°C (196°F) and 123 wt C/wt inlet Hg; inlet Hg concentration of 8.62 μg/dscm)	97
Test #6, Run #3	AC + FF (102°C (216°F) and 727 wt C/wt inlet Hg; inlet Hg concentration of 1.94 μg/dscm)	76
High temperature + low carbon injection rate (< 1000 wt C/wt Hg) based on inlet Hg		
Test #5, Run #1	AC + FF (107°C (225°F) and 362 wt C/wt inlet Hg; inlet Hg concentration of 5.53 μg/dscm)	14
Test #5, Run #2	AC + FF (110°C (230°F) and 373 wt C/wt inlet Hg; inlet Hg concentration of 4.45 μg/dscm)	28
Test #5, Run #3	AC + FF (116°C (241°F) and 457 wt C/wt inlet Hg; inlet Hg concentration of 3.47 μg/dscm)	47
Test #6, Run #1	AC + FF (121°C (250°F) and 286 wt C/wt inlet Hg; inlet Hg concentration of 5.04 μg/dscm)	29
Test #6, Run #2	AC + FF (118°C (244°F) and 367 wt C/wt inlet Hg; inlet Hg concentration of 4.22 μg/dscm)	35
Low temperature + high carbon injection rate (> 1000 wt C/wt Hg) based on inlet Hg		
Test #2, Run #1	AC + FF (91°C (196°F) and 2843 wt C/wt inlet Hg; inlet Hg concentration not measured but assumed to be 7.00 μg/dscm)	95
Test #2, Run #2	AC + FF (96°C (205°F) and 3132 wt C/wt inlet Hg; inlet Hg concentration not measured but assumed to be 7.00 μg/dscm)	98
Test #2, Run #3	AC + FF (93°C (200°F) and 3121 wt C/wt inlet Hg; inlet Hg concentration not measured but assumed to be 7.00 μg/dscm)	98
Test #3, Run #2	AC + FF (93°C (200°F) and 4361 wt C/wt inlet Hg; inlet Hg concentration of 6.23 μg/dscm)	99
Test #3, Run #3	AC + FF (96°C (205°F) and 3850 wt C/wt inlet Hg; inlet Hg concentration of 6.91 μg/dscm)	99

(continued)

Table I-3. (Continued)

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

[a] Source: Chang et al., 1993

Table I-4. Activated Carbon Injection Before Spray Dryer Absorption Data[a]

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

[a] Source: Felsvang, 1993

efficiency of 60 percent, with a range from 50 percent to 99 percent removal (Felsvang, 1993).

I.2.3.3 Flue Gas Desulfurization (FGD) Scrubbers. Wet FGD systems are currently installed on about 25 percent of the coal-fired utility generating capacity in the United States (Redinger et al, 1997). Although their primary function is to remove SO_2 emissions from boiler flue gas, wet FGD systems can also be effective in removing mercury emissions from boiler flue gas. The mercury removal efficiencies of wet FGD systems can vary widely depending on the mercury species in the incoming flue gas, the design and operation of the wet FGD system, and reactions of mercury species in the scrubbing solution.

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

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

The coals fired during the separate tests were both N. Appalachian coals (Pittsburgh Seam/Blacksville and a blend of Ohio No.

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

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

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

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

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

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

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

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

Previous field studies conducted by EPRI and DOE did indicate higher levels of elemental mercury (Hg^0) at the outlet of wet FGD scrubbers relative to the inlet. In addition, the removals indicated higher than 95 percent of the reported oxidized mercury at the inlet. These measurements were reported from separate U.S. EPA Draft Method 29 (M29) samples and in combination with the MESA Method samples. Two questions were raised: “Was the U.S. EPA M29 capable of accurately measuring the oxidized form of mercury?” or “Was the oxidized form of mercury being captured in the wet FGD scrubber solutions being released as an “alternate” form not capable of being collected in the appropriate impinger solutions?”

Innovative pilot-scale studies were conducted by Radian International at the EPRI ECTC to address these two questions. Extensive flue gas and intra-train mercury spiking tests were conducted to investigate the acidified peroxide solutions of M29 (solutions for collecting the oxidized form of mercury). The first series of tests had Hg^0 and $HgCl_2$ injected separately into the flue gas stream at the inlet of the wet FGD. Results indicated 96 percent of the $HgCl_2$ (naturally occurring and spiked) was collected across the wet FGD and the increase in Hg^0 across the FGD was from 0.66 to 0.96 $\mu g/m^3$. The results for the Hg^0 spiking indicated 37 percent of spike was measured in the acidified peroxide solutions and the total Hg removal was only 29 percent. These results indicated the injected $HgCl_2$ was being effectively collected in the scrubber solutions and not being reduced and subsequently re-emitted as Hg^0. In addition, M29 was not effective in speciating the mercury at the inlet of this wet FGD system when Hg^0 was spiked.

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

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

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

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

I.2.3.4 Spray Dryer FGD Systems. In 1990, spray dryer FGD systems were installed on approximately one percent of coal-fired units in the United States (UDI, 1992). The primary function of spray dryer FGD systems is to remove SO_2 emissions from boiler flue gas; however, they can also be effective in removing mercury emissions from boiler flue gas.

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

I.2.4 Research and Emerging Technologies for Controlling Mercury Emissions from Utilities

Considerable research continues to develop efficient and cost-effective technologies for mercury emission reductions from utility plants. This section describes ongoing research and summarizes the results to date. Much of the research is being sponsored by three organizations: U.S. EPA, DOE and EPRI. Table I-5 lists the areas of research currently being funded by these groups.

Eleven Phase I mercury control projects have been completed as part of DOE’s Advanced Emissions Control Technology “MegaPRDA Program.” These Phase I efforts began in October 1995 and encompassed two years of laboratory and bench scale testing and evaluation of several approaches for controlling the emission of mercury from coal-fired utility boilers. The approaches included those listed in Table I-5. DOE has selected six Phase II proposals (two to three year efforts) to further investigate and develop fine particulate and

Table I-5. Current Mercury Control Research for Utility Boilers

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

mercury control technologies and concepts. Given the relative low maturity level of these technologies, commercial deployment is still at least several years away, and will be strongly dependent on the results of the Phase II efforts.

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

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

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

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

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

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

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

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

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

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

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

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

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

I.2.4.1 Sorbent Technology. Research continues on developing potential technologies for mercury emission reduction from utility plants. Although sorbent injection with activated carbon has been shown to be a promising technology, even greater mercury removal may be possible with impregnated activated carbons, sodium sulfide, and other types of sorbents. The application of an activated carbon circulating fluidized bed (CFB) also shows promise in removing mercury.

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

Sulfur-impregnated carbons can potentially be enhanced for mercury sorption by the impregnation of the carbon(s) with sulfur at elevated temperatures of 400 - 600°C (752 - 1112°F). This has promoted a more uniform distribution of short linear chains of sulfur allotropes (S_2 and S_6) on the carbon surface as opposed to having predominately S_8 rings condensed in the macropore region of commercially available sulfur impregnated carbons. In addition, the sulfur impregnated carbons prepared at elevated temperatures have exhibited significantly better thermal stability since no sulfur loss was observed even after exposure at 400°C (752°F). The sulfur impregnated carbons exhibited high elemental mercury uptake efficiency at 140°C (284°F) when compared to commercially available activated carbons. Dynamic adsorption capacity of these carbons were measure as high as 4,000 µg Hg/g C. This capacity is almost three orders of magnitude greater then the capacity of virgin activated carbon and an order of magnitude greater than the capacity of commercially available impregnated activated carbon (Vidic et al, 1996; Korpiel et al, 1997; and Liu et al, 1997).

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

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

Chloride-impregnated activated carbon injection has only been tested on MWCs in Europe. The chloride reacts with mercury to form $HgCl_2$ on the carbon, and the carbon is removed by a particulate control device. Experiments have shown that impregnating activated carbon with chloride salts increases adsorptive capacity of the activated carbon by a factor of 300 (Teller and Quimby, 1991).

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Another potential process for the reduction of mercury emissions is the use of activated carbon in a CFB (Clarke and Sloss, 1992). In a CFB, the activated carbon is continuously fed to the reactor where it is mixed with the flue gas at a relatively high velocity, separated in the subsequent FF and recycled to the reactor. A small part of the used activated carbon is withdrawn from the process and replaced by fresh material (Riley, 1991). The main advantages to CFB's over fixed carbon beds are the increased flue gas-to-carbon contact area and the smaller overall pressure drop. This system has been used in Germany for MWC operation.

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

I.2.4.2 Improving the Mercury Capture Efficiency of Existing Pollution Control Technology. Research on improving the mercury capture efficiency of existing pollution control technology can be categorized as an investigation of either mercury removal with wet FGD systems or particulate control technology for capturing mercury.

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

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

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

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

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

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

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

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

I.2.4.3 New Technology for Controlling Mercury. A new technology for controlling all forms of mercury from coal-fired electric utility units has been investigated at the laboratory- and bench-scales on simulated and on actual flue from coal combustion. ADA Technologies has been developing a technology utilizing a regenerable sorbent allowing for the recovery of liquid elemental mercury from the flue gas and appropriately called the Mercu-RE process. The process takes mercury from flue gases and produces liquid, elemental mercury with no secondary wastes. Noble metals are used to adsorb mercury at typical flue gas temperatures. The mercury is then thermally desorbed.

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

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

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

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

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

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

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

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