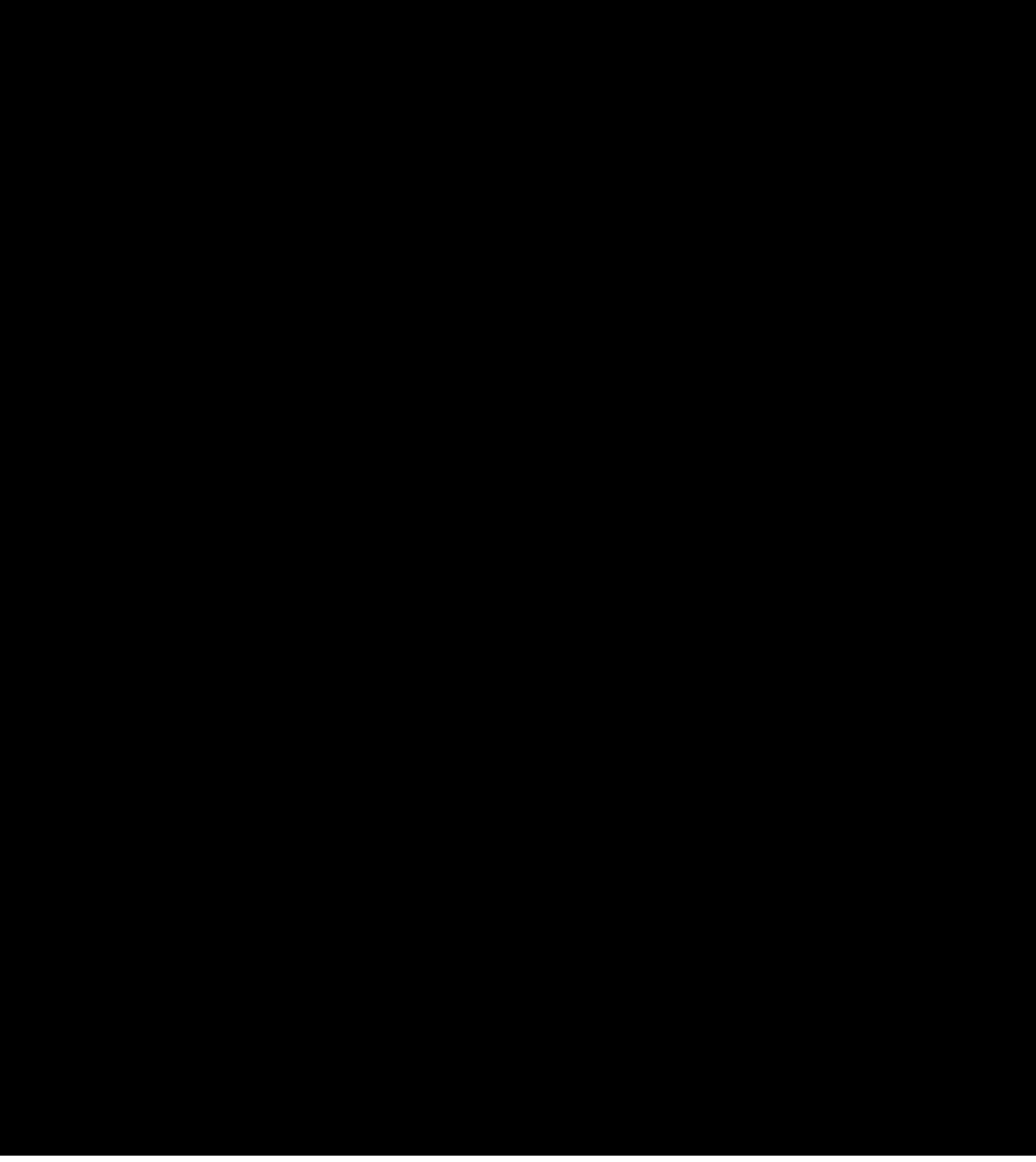




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7. ENVIRONMENTAL CONCENTRATIONS AND POTENTIAL EXPOSURES

7.1 AMBIENT AIR EXPOSURES

Several studies on concentrations of lead in the ambient air have been undertaken. These studies were generally intended to survey the levels and distributions of lead in the general air environment and around sources. They usually were not designed in conjunction with epidemiological studies of the concurrent effects of lead on man or other organisms. Yet that is the context in which these studies must now be interpreted to shed the most light possible on the concentrations likely to be encountered in various environmental settings.

Measurements taken with high-volume samplers, dustfall buckets, and particle size fractionators are included in these studies; however, with the exception of the NASN data from 1970 through 1974, quality control and interlaboratory comparability are unspecified. The effectiveness of some filter media in collecting very small lead-containing particles has been questioned; this subject is discussed in Chapter 4. The studies show that:

1. Lead typically occurs in urban airborne suspended particles 0.5 μm or less in mass median equivalent diameter at annual average concentrations ranging from <0.1 to $5 \mu\text{g}/\text{m}^3$, with an overall average of 1 to $2 \mu\text{g}/\text{m}^3$.
2. Urban concentrations of lead have declined somewhat since 1970.
3. Suspended particles in rural air samples contain lead at concentrations ranging from <0.01 to $1.4 \mu\text{g}/\text{m}^3$, with an overall average of about $0.2 \mu\text{g}/\text{m}^3$.

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4. Monthly average concentrations of lead in urban settleable particles range from 3 to 12 mg/m²-mo.
5. Indoor concentrations of lead are quite variable, but are generally one- to two-thirds the concentrations of adjacent outdoor levels.

A discussion of the NASN measurements follows. Summaries of additional studies can be found in Appendix C.

7.1.1 Airborne Lead Concentrations

There are two principal data bases containing extensive information on ambient air concentrations of lead in the U.S. Data from EPA's National Air Sampling Network (NASN), the most comprehensive nationwide data on long term trends, are discussed first. The second data base contains information contributed to EPA's National Aerometric Data Bank by State and local agencies, whose stations are sited to monitor compliance with, or progress toward compliance with the current ambient air standard for lead (1.5 µg/m³ averaged over a calendar quarter) promulgated in 1978.

EPA Nationwide Sampling Network. Tables 7-1 and 7-2, categorize respectively the urban and nonurban NASN sites with valid annual averages (4 valid quarters) into several annual average concentration ranges (Akland, 1976; Shearer et al.; 1972, U.S. Environmental Protection Agency, 1978, 1979; Quarterly Averages...from the National Filter Analysis Network, 1982). Nearly all of the urban sites reported annual averages below 2.0 µg/m³ and the majority of the nonurban sites reported annual averages below 0.2 µg/m³. Although the decreasing number of stations in service in recent years could account for some of the shift in averages toward lower concentrations, trends at individual urban stations, discussed below, confirm the indicated general trend.

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TABLE 7-1. NUMBER OF NASN URBAN STATIONS WHOSE DATA
FALL WITHIN SELECTED ANNUAL AVERAGE LEAD CONCENTRATION INTERVALS, 1966-1980
(AKLAND, 1976; SHEARER et al., 1972; U.S. EPA 1978, 1979;
Annual averages...from NFAN, 1982)

Year	Concentration interval, $\mu\text{g}/\text{m}^3$					Total
	<0.5	0.5-0.9	1.0-1.9	2.0-3.9	≥ 4.0	
1966:						
No. stations	9	40	40	6	—	95
Percent	9	42	42	6	—	100
1967:						
No. stations	4	37	63	9	—	113
Percent	3	32	55	7	—	100
1968:						
No. stations	14	67	54	10	1	146
Percent	9	45	36	6	1	100
1969:						
No. stations	5	46	103	23	1	178
Percent	2	25	57	12	1	100
1970:						
No. stations	9	54	80	15	1	159
Percent	5	33	50	9	1	100
1971:						
No. stations	—	23	64	21	1	109
Percent	—	21	58	19	1	100
1972:						
No. stations	16	67	84	12	1	180
Percent	9	37	47	7	0	100
1973:						
No. stations	20	76	36	4	1	137
Percent	15	55	26	3	1	100
1974:						
No. stations	19	69	38	4	0	130
Percent	15	53	29	3	0	100
1975:						
No. stations	28	94	38	7	1	168
Percent	17	56	22	4	1	100

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TABLE 7-1 (continued).

Year	Concentration interval, $\mu\text{g}/\text{m}^3$					Total
	<0.5	0.5-0.9	1.0-1.9	2.0-3.9	4.0-5.3	
1976:						
No. stations	23	99	36	4	0	162
Percent	14	62	22	2	0	100
1977:						
No. stations	21	73	35	5	0	134
Percent	16	54	26	4	0	100
1978:						
No. stations	21	42	8	1	0	72
Percent	29	58	11	2	0	100
1979:						
No. stations	23	15	4	1	0	43
Percent	54	35	9	2	0	100
1980:						
No. stations	51	6	0	0	0	57
Percent	89	11	0	0	0	100

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TABLE 7-2. NUMBER OF NASN NONURBAN STATIONS WHOSE DATA FALL WITHIN
 SELECTED ANNUAL AVERAGE LEAD CONCENTRATION INTERVALS, 1966-1980
 (AKLAND, 1976; SHEARER et al., 1972; U.S. EPA 1978, 1979;
 Annual averages...from NFAN, 1982)

Year	Concentration interval, $\mu\text{g}/\text{m}^3$				Total
	<0.03	0.03-0.096	0.10-0.19	0.20-0.45	
1966:					
No. stations	—	10	6	3	19
Percent	—	52	32	16	100
1967:					
No. stations	1	7	10	2	20
Percent	5	35	50	10	100
1968:					
No. stations	1	15	4	—	20
Percent	5	75	20	—	100
1969:					
No. stations	—	11	9	1	21
Percent	—	52	43	5	100
1970-1971:					
No. stations	—	—	7	3	10
Percent	—	—	70	30	100
1972:					
No. stations	10	4	9	11	34
Percent	29	12	26	33	100
1973:					
No. stations	9	7	6	1	23
Percent	39	31	26	4	100
1974:					
No. stations	3	5	6	2	16
Percent	19	31	38	12	100
1975:					
No. stations	0	0	1	4	5
Percent	0	0	20	80	100
1976:					
No. stations	0	0	3	3	6
Percent	0	0	50	50	100

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TABLE 7-2 (continued).

Year	Concentration interval, $\mu\text{g}/\text{m}^3$				Total
	<0.03	0.03-0.096	0.10-0.19	0.20-0.45	
1977:					
No. stations	5	8	7	1	21
Percent	24	38	33	5	100
1978:					
No. stations	1	3	1	0	5
Percent	20	60	20	0	100
1979:					
No. stations	1	1	1	1	4
Percent	25	25	25	25	100
1980:					
No. stations	1	2	0	0	3
Percent	33	67	0	0	100

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Tables 7-3 and 7-4 provide cumulative frequency distributions of all quarterly lead concentrations for urban and nonurban NASN stations, respectively. Samples collected by the NASN from 1970 through 1976 were combined for analysis into quarterly composites. Since 1977, the 24-hour samples have been analyzed individually. These data have been arithmetically averaged for comparison with the quarterly composite data. (Note, also, that the EPA data base has been renamed the National Filter Analysis Network, or NFAN.)

Each of the summary percentiles and means for urban stations (Table 7-3) has decreased over the period from 1970 to 1980; the 1980 levels are in the range of one-third to one-fourth of the values in 1970. The data from non-urban locations (Table 7-4) represent far fewer sites than the urban data and many concentrations are below the measurement method's detection limit, therefore, summary statistics are more susceptible to the presence or absence of individual sites from year to year, and if more than half the samples contain less than detectable amounts of lead, the means are not reported. The upper percentiles are fairly stable, however, and while the composite nonurban concentrations are approximately one-seventh of the urban concentrations, they exhibit a relative decrease over the 1979-1980 period similar to that seen in the data from the urban sites.

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TABLE 7-3. CUMULATIVE FREQUENCY DISTRIBUTIONS OF QUARTERLY LEAD MEASUREMENTS
AT URBAN STATIONS BY YEAR, 1970 THROUGH 1980
(AKLAND, 1976; U.S. EPA, 1978, 1979; Quarterly averages...from NAFN, 1982)
($\mu\text{g}/\text{m}^3$)

Year	No.	Min.	Percentile							Arithmetic			Geometric	
			10	30	50	70	90	95	99	Max.	Mean	Std. dev.	Mean	Std. dev.
1970	797	LD	0.47	0.75	1.05	1.37	2.01	2.59	4.14	5.83	1.19	0.80	0.99	1.80
1971	717	LD	0.42	0.71	1.01	1.42	2.21	2.86	4.38	6.31	1.23	0.87	1.00	1.89
1972	708	LD	0.46	0.72	0.97	1.25	1.93	2.57	3.69	6.88	1.13	0.78	0.93	1.87
1973	559	LD	0.35	0.58	0.77	1.05	1.62	2.08	3.03	5.83	0.92	0.64	0.76	1.87
1974	594	0.08	0.36	0.57	0.75	1.00	1.61	1.97	3.16	4.09	0.89	0.57	0.75	1.80
1975	695	LD	0.37	0.58	0.78	0.96	1.54	2.02	3.15	4.94	0.89	0.59	0.74	1.82
1976	670	LD	0.37	0.58	0.74	0.96	1.41	1.72	3.07	4.54	0.85	0.55	0.72	1.80
1977	533	0.063	0.37	0.57	0.75	0.95	1.67	2.13	3.29	3.96	0.91	0.80	0.68	1.79
1978	282	0.07	0.27	0.43	0.57	0.74	1.19	1.49	2.40	3.85	0.68	0.64	0.50	1.87
1979	167	0.10	0.22	0.33	0.43	0.63	1.09	1.33	2.44	3.59	0.56	0.58	0.39	1.89
1980	220	0.03	0.14	0.21	0.30	0.38	0.55	0.66	0.84	1.06	0.32	0.27	0.24	1.88

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TABLE 7-4. CUMULATIVE FREQUENCY DISTRIBUTIONS OF QUARTERLY LEAD MEASUREMENTS
AT NONURBAN STATIONS BY YEAR, 1970 THROUGH 1980
(AKLAND, 1976; U.S. EPA 1978, 1979; Quarterly averages...from NAFN, 1982)
($\mu\text{g}/\text{m}^3$)

Year	No.	Min.	Percentile							Arithmetic		Geometric	
			10	30	50	70	90	95	99	Max.	Mean	Std. dev.	Mean
1970	124	LD	LD	LD	LD	LD	LD	0.267	0.383	0.628	1.471	--	--
1971	85	LD	LD	LD	LD	LD	LD	0.127	0.204	0.783	1.134	--	--
1972	137	LD	LD	LD	LD	LD	LD	0.107	0.166	0.294	0.392	0.950	1.048
1973	100	LD	LD	LD	LD	LD	LD	0.132	0.233	0.392	0.698	0.939	--
1974	79	LD	LD	LD	LD	LD	LD	0.053	0.087	0.141	0.221	0.317	0.496
1975	98	LD	LD	LD	LD	LD	LD	0.144	0.255	0.311	0.431	0.649	--
1976	98	LD	LD	LD	LD	LD	LD	0.150	0.240	0.285	0.336	0.483	--
1977	84	0.006	0.01	0.04	0.08	0.11	0.18	0.20	0.25	0.40	0.09	0.10	0.07
1978	20	0.002	0.007	0.04	0.06	0.09	0.24	0.33	0.33	0.33	0.08	0.10	0.07
1979	16	LD	0.02	0.02	0.10	0.14	0.21	0.27	0.32	0.11	0.11	0.13	0.11
1980	12	LD	0.01	0.005	0.03	0.05	0.11	0.13	0.13	0.13	0.04	0.06	0.05

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Long-term trends and seasonal variations in airborne lead levels at urban sites can be seen in Figure 7-1. The 10th, 50th, and 90th percentile concentrations are graphed, using quarterly composite and quarterly average data from an original group of 92 urban stations (1965-1974) updated with data for 1975 through 1980. Note that maximum lead concentrations typically occur in the winter, while minima occur in the summer. In contrast, automotive emissions of lead would be expected to be greater in the summer for two reasons: (1) gasoline usage is higher in the summer, and (2) lead content is raised in summer gasolines to replace some of the more volatile high-octane components that cannot be used in summertime gasolines.

Figure 7-1 also clearly portrays the significant decrease in airborne lead levels over the past decade. This trend is attributed to the decreasing lead content of regular and premium gasoline, and to the increasing usage of unleaded gasoline. The close parallel between these two parameters is discussed in detail in Chapter 5. (See Figure 5-4 and Table 5-6.)

The decrease in lead concentrations, particularly in 1979 and 1980, were not caused by the disappearance from the network of sites with characteristically high concentrations; the quarterly values for sites in six cities representing the east coast, the central, and the western sections of the country (Table 7-5) indicate that the decrease is a real and pervasive phenomenon.

State and Local Agency Data. Table 7-6 lists stations operated by State and local agencies where one or more quarterly averages exceeded the current standard of $1.5 \mu\text{g}/\text{m}^3$ in 1979 and/or 1980. A portion of each agency's compliance monitoring network consists of monitors sited in areas expected to yield high concentrations associated with identifiable sources. In the case of lead, these locations are most likely to be near stationary point sources

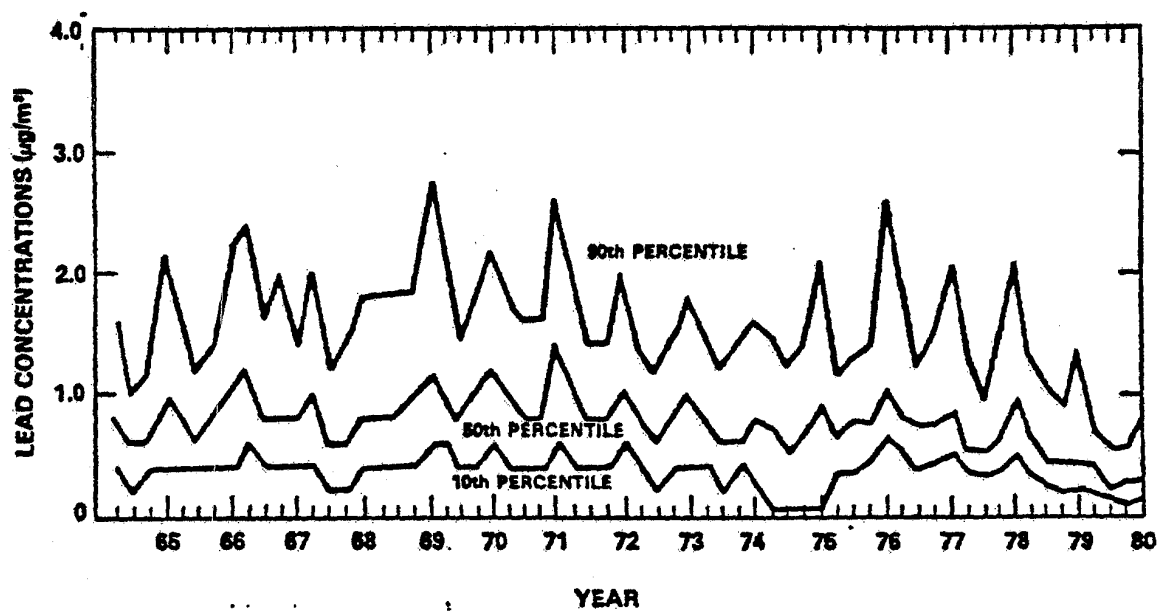


Figure 7-1. Seasonal patterns and trends in quarterly average urban lead concentrations.

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TABLE 7-5. TRENDS IN QUARTERLY LEAD VALUES FOR SELECTED URBAN SITES, 1975-1980

	Q	1975	1976	1977	1978	1979	1980
Tucson, Arizona	1	0.78	0.68	0.87	0.74	0.52	0.35
	2	0.49	0.63	0.33	0.42	0.39	0.19
	3	0.42	0.38	0.30	0.37	0.22	0.21
	4	1.27	0.94	0.91	--	0.22	0.34
Des Moines Iowa	1	0.71	0.44	--	0.67	0.68	0.37
	2	0.66	1.18	--	--	0.56	0.34
	3	0.86	--	--	0.72	--	0.34
	4	1.18	1.04	--	0.79	0.38	0.28
Worcester, Massachusetts	1	<0.12	--	0.58	0.42	0.57	0.56
	2	<0.12	0.75	0.71	0.54	0.34	0.37
	3	0.78	0.78	0.94	0.76	0.45	0.50
	4	0.89	0.60	1.03	1.30	0.76	0.45
Reno, Nevada	1	1.48	--	--	0.74	0.54	0.53
	2	0.50	--	0.40	0.37	0.32	0.19
	3	0.60	0.96	0.50	0.45	0.23	0.27
	4	2.15	3.00	1.98	2.05	0.61	0.79
Newark, New Jersey	1	0.91	1.09	--	0.67	1.17	0.54
	2	<0.12	1.37	0.99	0.54	0.63	0.27
	3	1.18	1.08	1.11	1.06	0.54	0.36
	4	1.25	--	1.16	1.72	0.79	0.42
Akron, Ohio	1	0.61	0.37	0.59	0.36	0.42	0.29
	2	0.50	1.02	0.59	0.62	0.37	0.24
	3	0.78	0.77	0.63	0.55	0.46	0.38
	4	0.79	0.77	0.44	0.58	0.29	0.26

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Table 7-6. Locations Reporting Quarterly Lead Concentrations Greater than 1.5 $\mu\text{g}/\text{m}^3$
(The Current Ambient Air Standard) in 1979 and 1980.* (OAQPS, 1982)

STATE	CITY OR COUNTY	YEAR	MAX. QTR. $\mu\text{g}/\text{m}^3$	NO. OF QUARTERS >1.5 $\mu\text{g}/\text{m}^3$
Alabama	Troy (Pike)	1979	2.78	2
Arizona	Phoenix	1979	2.59	2
	"	1979	1.55	1
Colorado	Adams County	1979	1.77	1
	Arvada	1979	1.60	1
	Denver	1979	1.70	1
	"	1979	3.47	1
	"	1980	1.53	1
	"	1979	2.13	1
	"	1979	1.57	1
	"	1979	1.67	1
	"	1979	1.67	1
	Englewood	1979	1.80	1
	Grand Junction	1979	1.53	1
	"	1979	1.57	1
Connecticut	New Haven	1979	1.57	1
DC	D.C.	1979	1.89	3
	"	1979	1.90	1
Florida	Tampa	1979	1.60	1
Idaho	Kellog	1979	9.02	4
	"	1980	6.88	2
	"	1979	8.25	4
	"	1980	8.72	4
	Shoshone Co.	1979	2.27	1
	"	1979	4.57	4
	"	1980	3.33	3
	"	1979	4.11	1
	"	1980	2.15	2
	"	1979	13.54	4
	"	1980	13.67	4
	"	1979	10.81	4
	"	1980	7.18	3
Illinois	Elgin	1980	1.95	1
	Granite City	1979	3.00	1
	"	1980	2.97	1
Indiana	E. Chicago	1979	2.19	1
	"	1979	1.67	1
Kentucky	Jefferson Co.	1980	1.78	1
	Louisville	1980	2.41	1
	"	1980	1.76	1
	"	1980	1.58	1
	"	1980	2.52	1
	Jefferson Co.	1980	2.31	1
	"	1980	1.83	1
Louisiana	Baton Rouge	1979	1.57	1
Maryland	Cheverly	1979	1.51	1

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Table 7-6. Locations Reporting Quarterly Lead Concentrations Greater than $1.5 \mu\text{g}/\text{m}^3$
 (The Current Ambient Air Standard) in 1979 and 1980.* (OAQPS, 1982)
 (continued)

STATE	CITY OR COUNTY	YEAR	MAX. QTR. $\mu\text{g}/\text{m}^3$	NO. OF QUARTERS $>1.5 \mu\text{g}/\text{m}^3$
Massachusetts	Springfield	1979	1.68	1
Minnesota	Minneapolis	1979	2.44	1
	"	1979	2.10	1
	"	1980	2.41	3
	Richfield	1979	1.95	3
	St. Louis PK	1979	2.87	2
	"	1980	3.04	3
	St. Paul	1980	1.82	2
Montana	Lewis & Clark	1979	4.19	4
	"	1980	2.75	4
Pennsylvania	Lauderdale	1979	3.30	3
	"	1980	1.86	2
	Philadelphia	1979	2.71	4
Puerto Rico	Guayanilla	1979	1.60	1
	San Juan	1979	3.59	4
Rhode Island	Providence	1979	1.92	4
Texas	Dallas	1979	1.59	1
	El Paso	1979	1.90	1
	"	1979	2.60	2
	"	1979	1.91	2
	"	1979	1.84	2
	"	1979	2.12	1
	"	1979	2.15	1
	"	1979	2.47	1
	"	1979	1.97	1
	"	1989	1.94	1

*As of March 3, 1982

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such as smelters or refineries, and near routes of high traffic density. Both situations are represented in Table 7-6, e.g., the Idaho data reflect predominantly stationary source emissions, the Washington, D.C. data reflect predominantly vehicular emissions.

Table 7-7 summarizes the maximum quarter lead values for those stations reporting 4 valid quarters in 1979, 1989, and 1981, grouped according to principal exposure orientation or influence--population, stationary source, or background. The stationary source-oriented sites clearly dominate the concentrations over $2.0 \mu\text{g}/\text{m}^3$; however, new siting guidelines, discussed below, will probably effect some increase in the upper end of the distribution of values from population-oriented sites by adding sites closer to traffic emissions.

TABLE 7-7. SUMMARY OF MAXIMUM QUARTERLY LEAD AVERAGE BY SELECTED CONCENTRATIONS ACCORDING TO SITE-TYPE, 1979-81.*

Site-Type	CONCENTRATION RANGES ($\mu\text{g}/\text{m}^3$)					Total No. of Site-Years
	$\leq .5$	$>.5$ ≤ 1.0	>1.0 ≤ 1.5	>1.5 ≤ 2.0	>2.0	
Population	300	173	46	7	5	531
Stationary Source	50	12	10	2	21	104
Background	21	0	0	0	0	21
Total (site-years)	380	185	56	9	26	656
Percent of Sites in Concentration Range	58	28	9	1	4	

*Any site-year which had all four quarters valid was included in the summary.

Source:

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New guidelines for siting ambient air lead monitors went into effect in July, 1982 (Federal Register, 1981). "Microscale" sites, placed between 5 and 15 meters from thoroughfares and 2 to 7 meters above the ground, are prescribed, but until now few monitors have been located that close to heavily travelled roadways. Many of these microscale sites might be expected to show higher lead concentrations than those historically measured at urban sites. One study (Pedco, 1981) does give limited insight into the relationship between a microscale location and locations further from a roadway. The data in Table 7-8 summarize TSP and particulate lead concentrations in samples collected in Cincinnati, Ohio, on 21 consecutive days in April and May, 1980, adjacent to a 58,500 vehicles-per-day expressway connector. Simple interpolation indicates that a microscale monitor as close as 5 meters from the roadway and 2 meters above the ground would record concentrations some 20 percent higher than those at a "middle scale" site 21.4 meters from the roadway. On the other hand, these PEDCO data also indicate that although lead concentrations very close to the roadway (2.8m setback) are quite dependent on the height of the sampler, the averages at the three selected heights converge rapidly with increasing distance from the roadway. In fact, the average lead for the one monitor (6.3m height, 7.1m setback) that satisfies the microscale site definition proves to be not significantly different from the averages for its two companions at 7.1m, or from the averages for any of the three monitors at the 21.4m setback.

Other urban locations around the country with their own characteristic wind flow patterns and complex settings, such as multiple roadways, may produce situations where the microscale site does not record the highest concentrations. Collectively, however, the addition of these microscale sites to the nation's networks can be expected to shift the distribution of reported

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TABLE 7-8. SUMMARY OF LEAD CONCENTRATIONS NEAR A ROADWAY* IN CINCINNATI, OHIO (Pedco, 1981)

a: Total Suspended Particulate Concentrations**, $\mu\text{g}/\text{m}^3$

<u>Elevation</u>	<u>Distance from Roadway</u>		
	<u>2.8 m</u>	<u>7.1 m</u>	<u>21.4 m</u>
10.5 m:	95	104	106
6.3 m:	103	119†	109
1.1 m:	133	126	112

b: Lead Concentrations**, $\mu\text{g}/\text{m}^3$

<u>Elevation</u>	<u>Distance from Roadway</u>		
	<u>2.8 m</u>	<u>7.1 m</u>	<u>21.4 m</u>
10.5 m:	0.81	0.93	0.90
6.3 m:	0.96	1.07	0.97
1.1 m:	1.33	1.16	1.01

*58,500 vehicles per day.

**Averages of 21 samples between 4/17/80 and 5/7/80.

†This site would qualify as a "microscale" site.

quarterly averages toward higher values. This shift will result from the change in composition of the networks and is a separate phenomenon from downward trend at long established sites described above, reflecting the decrease in lead additives used in gasoline.

7.1.2 Airborne Particle Size Distribution

In 1970, a cascade impactor network was established by EPA in six cities (Cincinnati, Chicago, Denver, Philadelphia, St. Louis, and Washington, D.C.) to collect particulates of different size ranges for subsequent analysis for lead and other metals (Lee et al., 1972). The samples, collected once every 2

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weeks for a full year, were analyzed for size distribution. Samples from each city were also composited quarterly and analyzed for lead by optical emission spectroscopy.

The average annual total lead concentration as determined in this study ranged from a high quarter of $3.2 \mu\text{g}/\text{m}^3$ in Chicago to a low quarter of $1.3 \mu\text{g}/\text{m}^3$ in Washington, D.C. The average mass median diameter for lead particles ranged from $0.69 \mu\text{m}$ in St. Louis to $0.42 \mu\text{m}$ in Washington, D.C. Fifty-nine to 74 percent of the lead was associated with particles smaller than $1 \mu\text{m}$ in diameter. Quarterly and annual particle size distribution data are presented in Table 7-9.

Table 7-10 demonstrates the fraction of inhaled airborne lead deposited in each compartment of the human respiratory system based on the size distributions of Figure 7-2 and the deposition curves of Figures 7-3 through 7-5.

7.1.3 Vertical Gradients of Lead in the Atmosphere

Very few studies have been conducted to determine the variation of lead concentration with height above the ground. Of those that were found, all contain the premise, either explicitly or implicitly, that the concentrations being studied derive principally from automotive emissions. Of the studies found in the literature, none were adequately designed to establish the lead concentration versus height relationship. Such a study would require simultaneous measurement (preferably continuous) at given height intervals over a long period of time (a minimum of one year). Even then, the results would be valid only for a location having the same characteristics and experiencing the same atmospheric conditions. A single set of concentration values obtained over short and variable time intervals not sufficient to draw conclusions regarding general exposure conditions.

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TABLE 7-9. INFORMATION ASSOCIATED WITH THE AIRBORNE LEAD SIZE DISTRIBUTIONS OF FIGURE 7-2.

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C _T ng/m ³	Approx. MMD μm
1	Lee et al., 1972	Jan. - Dec. 1970 Average of 4 quarterly composited samples, representing a total of 21 sampling periods of 24 hours each	Chicago, Illinois	Modified Andersen impactor with backup filter	3200	0.68
2	Lee et al., 1972	Mar. - Dec. 1970 Same averaging as Graph 1, total of 18 sampling periods	Cincinnati, Ohio	Same as Graph 1	1800	0.48
3	Lee et al., 1972	Jan. - Dec. 1970 Same averaging as Graph 1, total of 21 sampling periods	Denver, Colorado	Same as Graph 1	1800	0.50
4	Lee et al., 1972	Mar. - Dec. 1970 Same averaging as Graph 1, total of 20 sampling periods	Philadelphia, Pennsylvania	Same as Graph 1	1500	0.47
5	Lee et al., 1972	Jan. - Dec. 1970 Same averaging as Graph 1, total of 22 sampling periods	St. Louis, Missouri	Same as Graph 1	1800	0.69
6	Lee et al., 1972	Jan. - Dec. 1970 Same averaging as Graph 1, total of 23 sampling periods	Washington, D.C.	Same as Graph 1	1300	0.42

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TABLE 7-9 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	CT ³ ng/m ³	Approx. MMD μm
7	Lee et al., 1968	September 1966 Average of 14 runs, 24 hours each	Cincinnati, Ohio	Andersen impactor with backup filter, 1.2m above the ground	2800	0.29
8	Lee et al., 1968	February 1967 Average of 3 runs 4 days each	Fairfax, Ohio, suburb of Cincinnati	Same as Graph 7	690	0.42
9	Peden, 1976	Summer 1975 Average of 4 runs, average 8 days each	Alton, Illinois, industrial area near St. Louis	Andersen impactor, no backup filter	240	2.1
10	Peden, 1976	Summer 1972 Average of 3 runs, average 10 days each	Centreville, Illinois, downwind of a zinc smelter	Andersen impactor with backup filter	620	0.41
11	Peden, 1976	Summer 1973 Average of 2 runs, average 5 days each	collinsville, Illinois industrial area near St. Louis	Same as Graph 10	670	0.24
12	Peden, 1976	Summer 1973 Average of 2 runs, average 6 days each	KMOX radio trans- mitter, Illinois, industrial area near St. Louis	Same as Graph 10	600	0.31
13	Peden, 1976	Summer 1972 Average of 9 runs, average 9 days each	Pere Marquette State Park, Illinois, upwind of St. Louis	Same as Graph 10	150	0.51
14	Peden, 1976	Summer 1975 Average of 4 runs, average 8 days each	Wood River, Illinois, industrial area near St. Louis	Same as Graph 9	270	1.8
15	Cholak et al., 1968	April 1968 average of several runs, 3 days each	3 sites: 10,400 and 3300m from Interstate 75, Cincinnati, Ohio	Andersen impactor with backup filter	7800* 1700 1100	0.32

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TABLE 7-9 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C _T ³ ng/m ³	Approx. MMD μ m
16	McDonald and Duncan, 1979	June 1975 One run of 15 days	Glasgow, Scotland	Casella impactor with backup filter, 30m above the ground	530	0.51
17	Dorn et al., 1976	Winter, spring, summer 1972 Average of 3 runs, 27 days each	Southeast Missouri, 800m from a lead smelter	Andersen impactor, no backup filter, 1.7m above the ground	1000	3.8
18	Dorn et al., 1976	Winter, spring, summer 1972 Average of 3 runs, 14 days each	Southeast Missouri, 75 km from the lead smelter of Graph 17	Same as Graph 17	110	2.4
19	Daines et al., 1970	1968 Average of continuous 1-week runs over an 8-month period	3 sites: 9, 76, and 530m from U.S. Route 1, New Brunswick, New Jersey	Cascade impactor with backup filter	4500* 2200 1500	0.35
20	Martens et al., 1973	July 1971 One run of 4 days	9 sites throughout San Francisco area	Andersen impactor with backup filter	840	0.49
21	Lundgren, 1970	November 1968 Average of 10 runs, 16 hours each	Riverside, California	Lundgren impactor	590	0.50
22	Huntzicker et al., 1975	May 1973 One run of 8 hours	Shoulder of Pasadena Freeway near downtown Los Angeles, California	Andersen impactor with backup filter, 2m above the ground	14,000	0.32
23	Huntzicker et al., 1975	February 1974 One run of 6 days	Pasadena, California	Andersen impactor with backup filter, on roof of 4 story building	3500	0.72

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TABLE 7-9 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C _T ng/m ³	Approx. MMD μ m
24	Davidson, 1977	May and July 1975 Average of 2 runs, 61 hours each	Pasadena, California	Modified Andersen impactor with backup filter on roof of 4 story building	1200	0.97
25	Davidson et al., 1980	October 1979 One run of 120 hours	Clingman's Dome Great Smokies National Park, elev. 2024m	2 Modified Andersen impactors with backup filters, 1.2m above the ground	14	1.0
26	Davidson et al., 1981a	July-Sep. 1979 Average of 2 runs, 90 hours each	Pittsburgh, Pennsylvania	Modified Andersen impactor with backup filter, 4m above the ground	600	0.56
27	Davidson et al., 1981b	December 1979 One run of 52 hours	Nepal Himalayas elev. 3962m	Modified Andersen impactor with backup filter, 1.2m above the ground	1.4	0.54
28	Goold and Davidson, 1982	June 1980 One run of 72 hours	Export, Pennsylvania rural site 40 km east of Pittsburgh	2 Modified Andersen impactors with backup filters, 1.2m above the ground	111	1.2
29	Goold and Davidson, 1982	July 1980 One run of 34 hours	Packwood, Washington rural site in Gifford Pinchot National Forest	Modified Andersen impactor with backup filter, 1.5m above the ground	16	0.40
30	Goold and Davidson, 1982	July-Aug. 1980 One run of 92 hours	Hurricane Ridge Olympic National Park elev. 1600m	Same as Graph 29	2.4	0.87
31	Duce et al., 1976	May-June 1975 One run of 112 hours	Southeast coast of Bermuda	Sierra high-volume impactor with backup filter, 20m above the ground	8.5	0.57

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TABLE 7-9 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C _T ng/m ³	Approx. MMD μm
32	Duce et al., 1976	July 1975 One run of 79 hours	Same as Graph 31	Same as Graph 31	4.1	0.43
33	Harrison et al., 1971	April 1968 Average of 21 runs, 2 hours each	Ann Arbor, Michigan	Modified Anderson impactor with backup filter, 20m above the ground	1800	0.16
34	Gillette and Winchester, 1972	Oct. 1968 Average of 15 runs, 24 hours each	Ann Arbor, Michigan	Andersen impactor with backup filter	950	0.28
35	Gillette and Winchester, 1972	NA	Chicago, Illinois	Same as Graph 34	NA	0.39
36	Gillette and Winchester, 1972	NA	Lincoln, Nebraska	Same as Graph 34	NA	0.42
37	Johansson et al., 1976	June-July 1973 Average of 15 runs, average 50 hr each	2 sites in Tallahassee, Florida	Deltron Battelle-type impactor, no backup filter, on building roofs	240	0.62
38	Cawse et al., 1974	July-Dec. 1973	Chilton, England	Andersen impactor with backup filter, 1.5m above the ground	160	0.57
39	Pattenden et al., 1974	May-Aug. 1973 Average of 4 runs, 1 month each	Trebanos, England	Same as Graph 38	230	0.74
40	Bernstein and Rahn, 1979	Aug. 1976 Average of 4 runs, 1 week each	New York City	Cyclone sampling system with backup filter, on roof of 15 story building	1200	0.64

*Airborne concentrations for filters run at the same sites as the impactor, but during different time periods. Impactor concentrations not available.

Na = not available.

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TABLE 7-10. FRACTION OF INHALED AIRBORNE LEAD DEPOSITED
IN EACH COMPARTMENT OF THE HUMAN RESPIRATORY SYSTEM*

	Range	Arithmetic Average and Standard Deviation
Pulmonary Min.	0.08 - 0.16	0.11 ± 0.016
Max.	0.21 - 0.35	0.26 ± 0.031
Tracheobronchial	0.05 - 0.40	0.16 ± 0.075
Extrathoracic	0.03 - 0.23	0.09 ± 0.048
Total (based on max. pulmonary deposition)	0.30 - 0.94	0.51 ± 0.13

*Based on the size distributions of Figure 7-2 and the deposition curves of Figures 7-3 through 7-5.

Edwards (1975) has reported measurements made in downtown Ft. Collins, Colorado. Measurements were made in a street canyon formed by two and three story buildings (average height 9 m). With a 2.3 m/sec d from the northeast (street running north-south), lead concentrations along the east side of the street canyon ranged from 11.3 $\mu\text{g}/\text{m}^3$ at street level to 4.0 $\mu\text{g}/\text{m}^3$ at roof level. On the west side of the street, concentrations ranged from 0.9 $\mu\text{g}/\text{m}^3$ at street level to 1.3 $\mu\text{g}/\text{m}^3$ at roof level. Values for two additional sampling points above the rooftops on each side of the street were 0.4 $\mu\text{g}/\text{m}^3$ (east side) and 0.9 $\mu\text{g}/\text{m}^3$ (west side). Lead concentrations 2 to 5 blocks away ranged from 0.1 to 0.3 $\mu\text{g}/\text{m}^3$. These data reflect the wide variability that can be expected in urban traffic environments. Under moderate cross-wind conditions, concentrations within the canyon were strikingly anisotropic, and street-level concentrations along the upwind building faces were substantially

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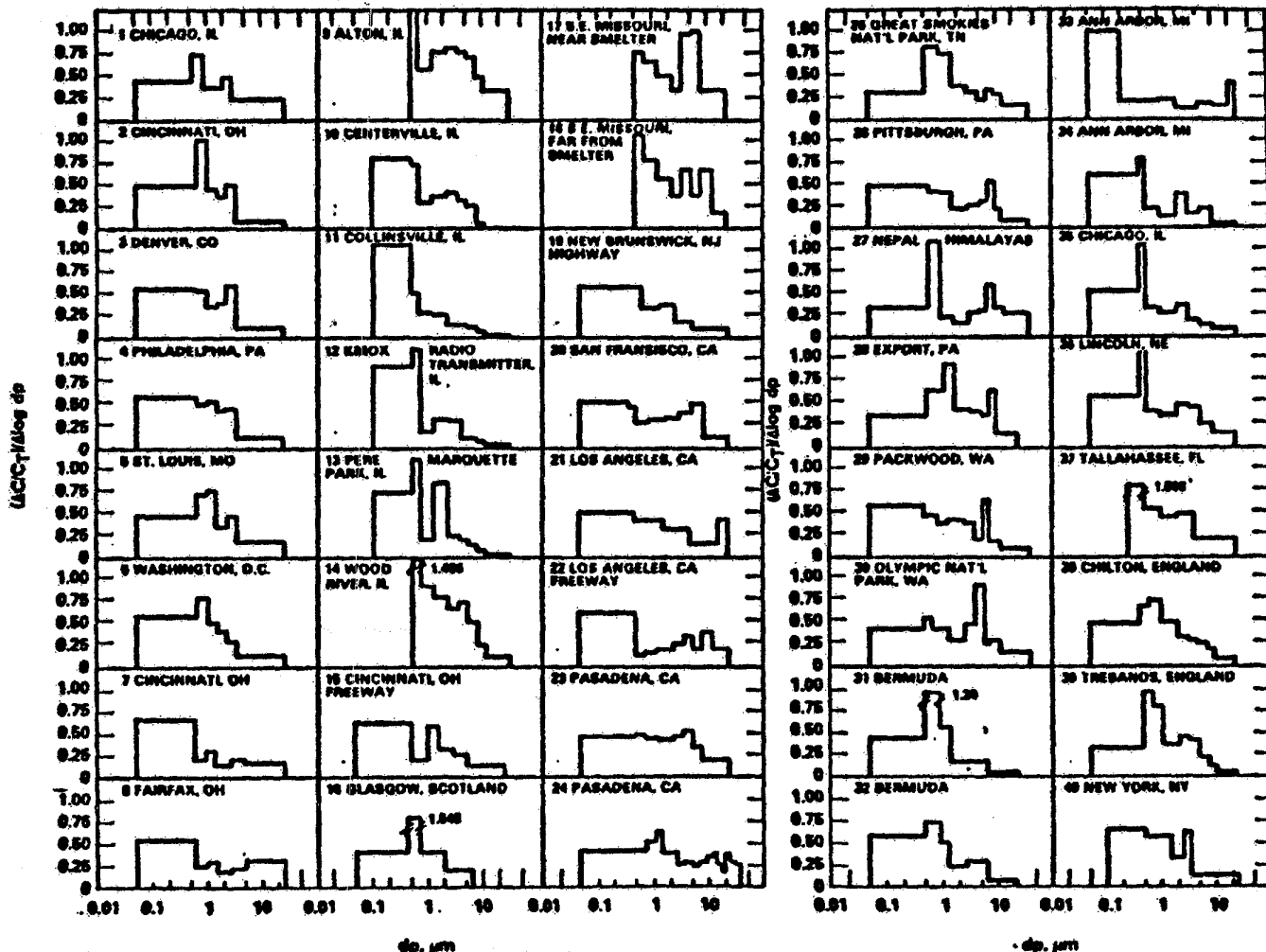


Figure 7-2. Normalized concentration versus particle size for 40 sites.

Figure 7-2. Airborne mass size distributions for lead taken from the literature. ΔC represents the airborne lead concentration in each size range, C_T is the total airborne lead concentration in all size ranges, and d_p is the aerodynamic particle diameter. A density of 6 g/cm³ for lead-containing particles has been used to convert aerodynamic to physical diameter when applying the lower end of the lung deposition curves of Figures 7-3 through 7-5.

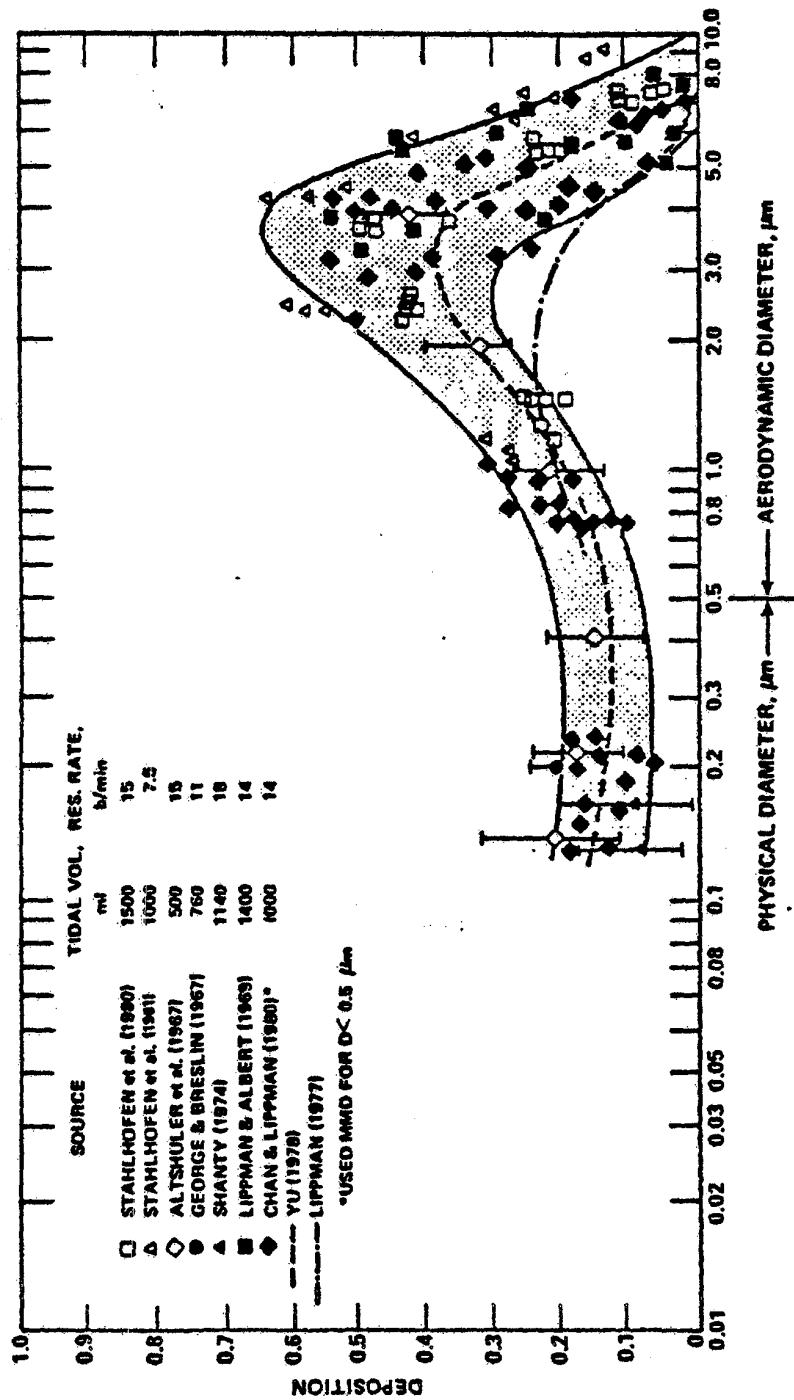


Figure 7-3. Deposition of monodisperse aerosols in the pulmonary region for mouth breathing as a function of aerodynamic diameter, except below $0.5 \mu m$ where deposition is plotted vs. physical diameter. The eye-fit band envelope deposition data cited by the different investigators. The dashed line is the theoretical deposition model of Yu (1978) and the broken line is the estimate of pulmonary deposition for nose breathing derived by Lippmann (1977).

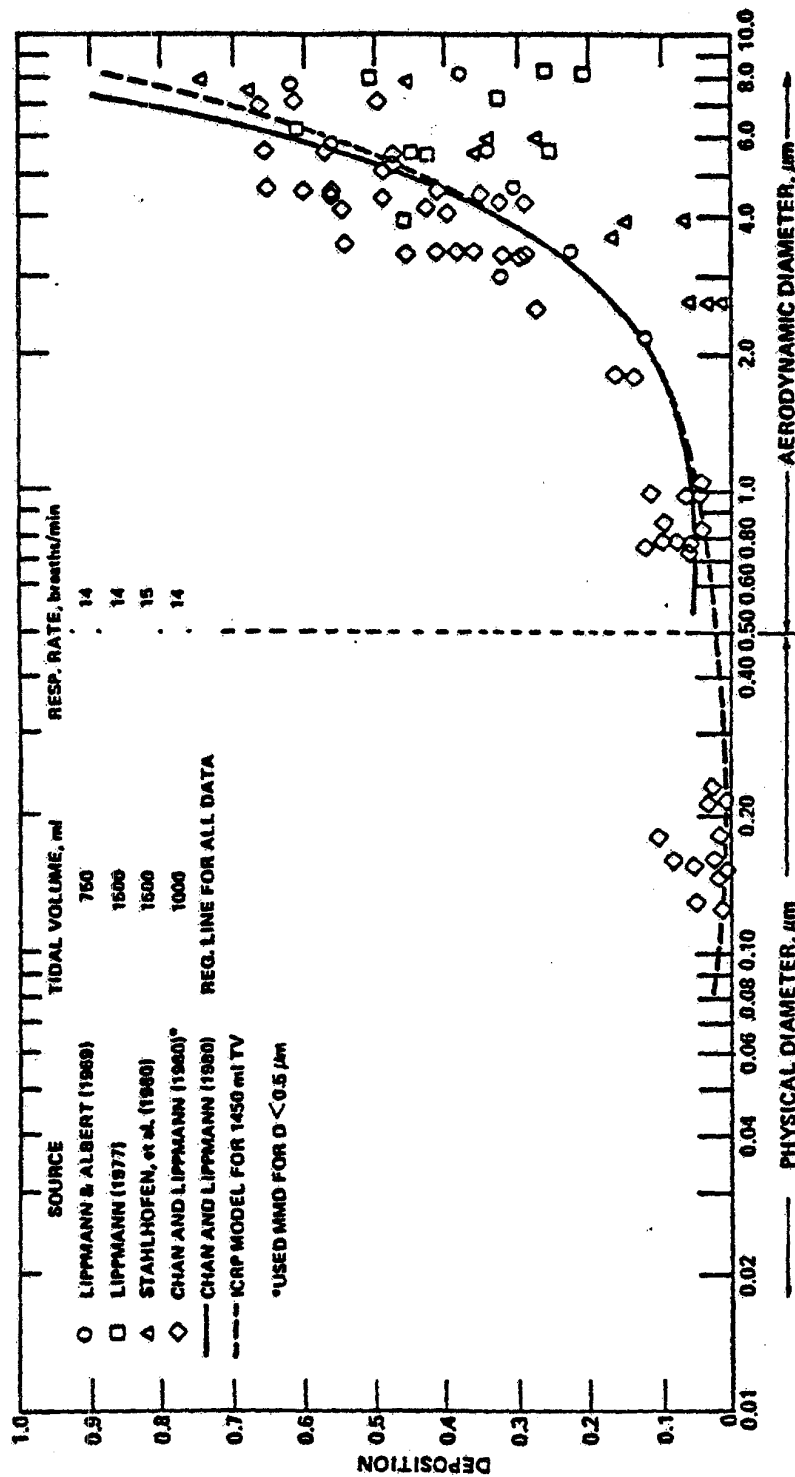


Figure 7-4. Deposition of monodisperse aerosols in the tracheobronchial region for mouth breathing in percent of the aerosols entering the trachea as a function of aerodynamic diameter except below $0.5 \mu m$ where deposition is plotted vs. physical diameter as cited by different investigators. Dashed line is ICRP model for 1450 ml tidal volume. The solid line is the overall regression derived by Chan and Lippmann (1980).

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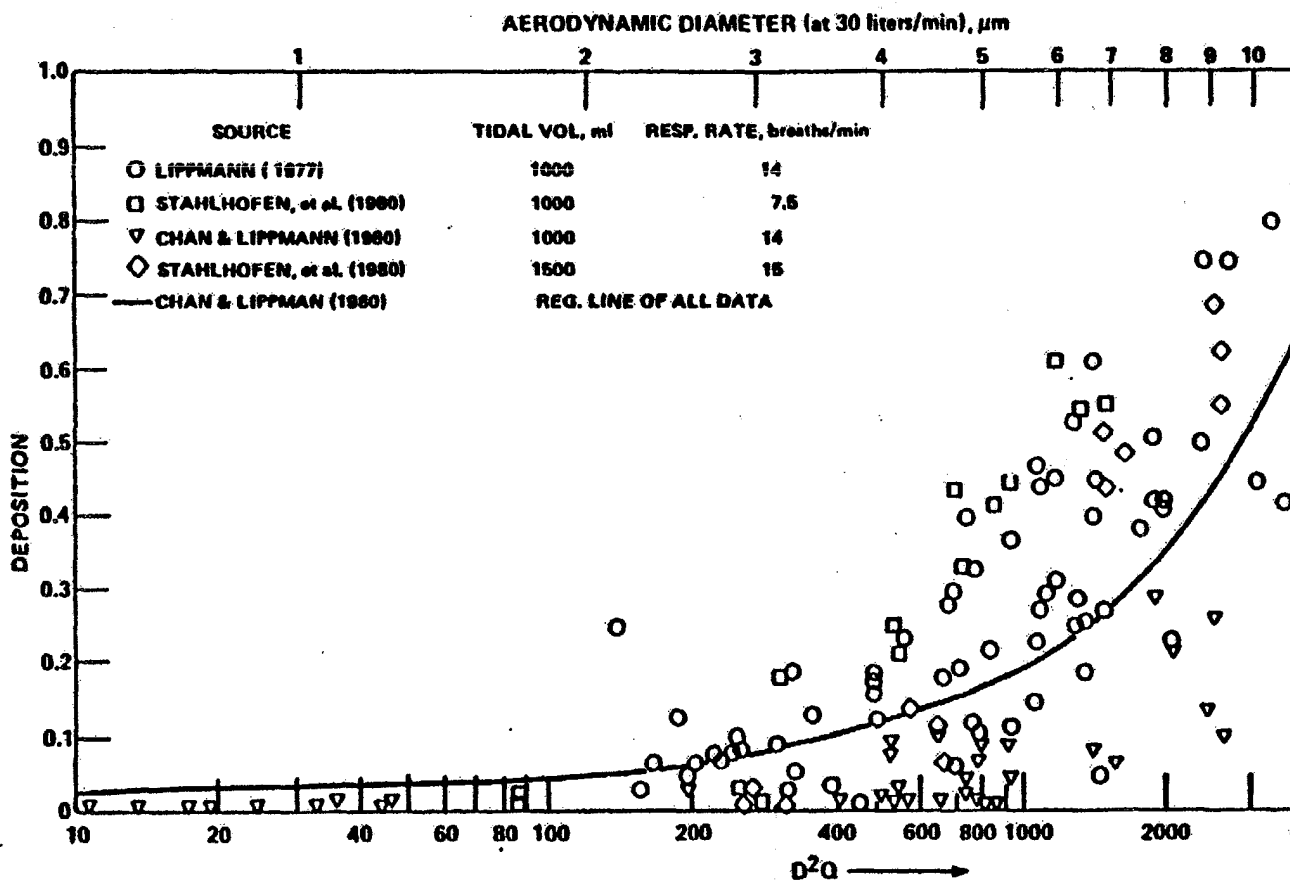


Figure 7-5. Deposition of monodisperse aerosols in extrathoracic region for moderate breathing as a function of D^2Q , where Q is the average inspiratory flow rate in liters/min. The data are the individual observations as cited by the various investigators. The solid line is the overall regression derived by Chan and Lippmann (1980).

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higher than those along the downwind faces. With different wind regimes, building configurations, and stability conditions, the distribution concentrations would also be different.

Barltrop and Strehlow (1976) conducted an air sampling program at a proposed nursery site under an elevated motorway. The height of the motorway was 9.3 m. Air samplers were operated at five to seven sites during the period from Monday to Friday, 8 a.m. to 6 p.m., for one year. The maximum individual value observed was $18 \mu\text{g}/\text{m}^3$. The 12 month mean ranged from $1.51 \mu\text{g}/\text{m}^3$ to $1.35 \mu\text{g}/\text{m}^3$, with standard deviations of 0.91 and 0.66, respectively. The authors reported that the airborne concentrations were independent of height from ground level up to 7 m.

Pedco-Environmental (1977) measured lead concentrations at heights of 5 and 20 feet at sites in Kansas City, Missouri and Cincinnati, Ohio. The sampling sites in Kansas City were described as unsheltered, unbiased by local pollution influences, and not immediately surrounded by large buildings. The Cincinnati study area was located in a primarily residential area with one commercial street. Samplers were rated for 24-hour periods from 8 a.m. to 8 a.m., but a few 12-hour samples were collected from 8 a.m. to 8 p.m. Data were obtained in Kansas City on 35 days and in Cincinnati on 33 days. The range and average values reported are shown in Table 7-11. In all cases except two, the measured concentrations were greater at 5 feet than at 20 feet. Note that the difference between the east side and west side of the street was approximately the same as the difference between 5 feet and 20 feet in height.

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TABLE 7-11. AIRBORNE LEAD CONCENTRATIONS AT 5- AND 20-FT ELEVATIONS ABOVE STREET LEVEL (PEDCO ENVIRONMENTAL, 1977)
($\mu\text{g}/\text{m}^3$)

Location	Range	Averages					
		East side of street			West side of street		
		20 ft.	5 ft.	Diff.	20 ft.	5 ft.	Diff.
Kansas City	0.8 - 4.0	1.7	2.0	0.3	1.5	1.7	0.2
Cincinnati	0.1 - 4.6	0.9	1.4	0.5	0.6	0.8	0.2

^aSide of street.

^bHeight above street level.

Ter Haar (1979) measured airborne lead at several heights above the ground, using samplers positioned 6 m from a heavily traveled road in Detroit. A total of nine 8-hour daytime samples were collected. The overall average airborne lead concentrations at heights of 0.3, 0.9, 1.5, and 3.0 m were 4.2, 4.8, 4.7, and $4.6 \mu\text{g}/\text{m}^3$, respectively, indicating a uniform concentration over this range of heights at the measurement site. It should be noted that at any one height, the concentration varied by as much as a factor of 10 from one day to the next; the importance of simultaneous sampling when attempting to measure gradients is clearly demonstrated.

Sinn (1980) investigated airborne lead concentrations at heights 3 and 20 m above a road in Frankfurt, Germany. Measurements conducted in December 1975, December 1976, and January 1978 gave monthly mean values of 3.18, 1.04, and $0.66 \mu\text{g}/\text{m}^3$, respectively, at 3 m. the corresponding values at 20 m were

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0.59, 8, and 0.31 $\mu\text{g}/\text{m}^3$, showing a substantial reduction at this height. The decrease in concentration over the 2-year period was attributed to a decrease in the permissible lead content of gasoline from 0.4 to 0.15 g/liter beginning in January 1976.

These data reflect the strong influence of the geometry of the boundary layer, wind, and atmospheric stability conditions on the vertical gradient of lead resulting from automobile emissions. The variability of concentration with height is further complicated by elevated emissions (i.e., from stacks). Concentrations measured from sampling stations on the roofs of buildings several stories high may not reflect actual human exposure conditions, but neither would a single sampling station located at ground level in a building complex. The height variation in concentration resulting from vertical diffusion of automobile emissions is likely to be small compared to temporal and spatial variations resulting from surface geometry, wind, and atmospheric conditions.

7.2 MOBILE SOURCE EXPOSURES

Several major studies have been undertaken to determine the lead levels in the air and in settled dust near busy highways that are far from any stationary lead source. Among the most intensive of these studies is the Los Angeles Catalyst Study of 1974-1977, undertaken by EPA to measure the impact of the catalytic converter on air quality near a major traffic lead source (U.S. Environmental Protection Agency, 1979).

Table 7-12 summarizes the average 4-hour and 24-hour airborne lead concentrations during the summer months (May-October). Data for sites C and D, located on the downwind side of the freeway, are shown. Concentrations measured at upwind sites A and B are substantially smaller.

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TABLE 7-12. FOUR-HOUR AND 24-HOUR AIRBORNE LEAD CONCENTRATIONS AT TWO ROADSIDE SITES IN THE LOS ANGELES CATALYST STUDY, 1974-1977

Site	Year(s)	4-Hr Pb Concentrations (3 p.m. to 7 p.m.)		
		Mean	Standard Deviation	Number of Observations
C	1975	8.16	2.78	174
	1976	6.77	2.25	166
	1977	11.41	2.14	92
D	1974	5.64	1.29	151
	1975	4.90	1.00	176
	1976	4.40	0.96	166
	1977	6.18	1.18	20

Site	Year(s)	24-Hr Pb Concentrations		
		Mean	Standard Deviation	Number of Observations
C	1974	8.19	1.83	143
	1975	7.98	1.48	173
	1976	7.00	1.60	166
	1977	7.43	1.19	74
D	1974	5.00	1.02	46
	1975	4.19	1.01	57
	1976	3.75	0.73	57
	1977	4.16	0.70	63

Source: Ledolter et al. (1979)

The data show a progressive decrease in airborne lead through 1974-1976, followed by a substantial increase in 1977. The decrease is attributed to the growing fraction of catalytic converter-equipped vehicles, which use unleaded gasoline. The increase in 1977 is due to the opening of a new northbound lane; this resulted in an increase in traffic speed on the lanes closest to sites C and D. Hirschler et al. (1957) have shown that lead emissions increase with traffic speed.

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Ledolter et al. (1979) have used 1976-1977 data from the Los Angeles Catalyst Study to develop a simple model for the 4-hour afternoon lead level at site C. The final equation has the form:

$$Pb_t = a_t + k_1(\alpha C_t^N S_t^N + (1-\alpha)C_t^S S_t^S) \exp(-b(W_{pt}-w)^2)$$

where Pb_t = observed 4-hour afternoon lead concentration at site C for day t

a_t = error term for day t

k_1, c, b, w = empirical constants based on least squares regression analysis of the available data k_1 can assume different values for 1976 and 1977.

C_t^N or S = total afternoon northbound or southbound traffic count for day t.

S_t^N or S = average afternoon northbound or southbound traffic speed for day t

W_{pt} = average afternoon windspeed component perpendicular to the road for day t

Values of α , b , and w are 0.75 ± 0.10 , 0.011 ± 0.0035 , and 2.19 ± 0.56 , respectively. The large value of α suggests that the northbound lanes have a much larger contribution to the airborne lead level at site C than the southbound lanes (which are slightly farther away). Rapid settling of large lead-containing particles, as well as dilution, are most likely responsible.

In a study conducted in London, England, Harrison et al. (1975) measured ambient air levels of organic lead at urban sites. Concentrations of organic lead ranging from 0.04 to $0.11 \mu\text{g}/\text{m}^3$ were found on streets of varying widths and traffic flow. These values were 0.3-2.65 percent of the total airborne lead. The level of organic lead inside a busy tunnel was $0.02 \mu\text{g}/\text{m}^3$, or 0.1

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percent of the total airborne lead. Not surprisingly, organic lead concentrations measured at a busy service station ranged from 0.21 to 0.59 $\mu\text{g}/\text{m}^3$, or 3.9-9.7 percent of total airborne lead at that site. At a less busy service station, the concentration of organic lead was 0.07 $\mu\text{g}/\text{m}^3$, or 4.2 percent of the total airborne lead.

Jansen et al. (1978) studied the influence of automotive lead emissions on child lead exposure in Morton Grove, Illinois, a Chicago suburb. Annual average airborne lead levels between November 1974 and October 1975 ranged from 0.73 to 3.04 $\mu\text{g}/\text{m}^3$, with the greatest values measured near a heavily traveled road. Soil lead levels were generally greater within 200 feet of the road, compared with samples collected at greater distances. Similarly, children aged 1-12 years living within 200 feet of the road had greater blood lead levels than children living farther away. The greatest blood lead levels were found in the youngest children, aged 1-3 years, who had lived all of their lives close to the road. The authors concluded that child blood lead levels may have been influenced by ingestion of deposited automobile-emitted lead as a result of hand-to-mouth activities.

Chamberlain et al. (1978) summarized the results of several studies conducted at Harwell Laboratories, England over the past decade, the overall objective being to assess human exposure to lead emitted from mobile sources. Many of the measurements were conducted near a highway carrying 90,000 vehicles per day in London. Size distributions measured with an inertial impactor, a diffusion battery, thermal precipitator, and a cascade centripeter showed that most of the mass of automobile-emitted particles was submicron; nearly all of the lead mass was also submicron. Airborne lead concentration measurements and atmospheric dispersion calculations suggested that roadside lead levels

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were about $2 \mu\text{g}/\text{m}^3$ for each 1000 vehicles/hour. For a traffic flow of 1000 vehicles/hour, an airborne lead concentration exceeding $1 \mu\text{g}/\text{m}^3$ extended from the curb to 15 m on either side of the road; the zone increased to 100 m on either side for 4000 vehicles/hour. Deposition data for trays of washed grass set beside the highway were used to calculate that about 10 percent of the lead emitted during steady cruise operations on a level highway was deposited within 100 m of the road. However, accounting for the lead content of soil and vegetation near the highway since its opening in 1965, the authors concluded that 40 percent of the lead emitted during 1965-1976 had deposited within 100 m. Even greater fractions of deposited lead could be expected in the vicinity of stop-and-go traffic, rapid accelerations, or uphill grades. Additional data from this study, covering intake and absorption of automobile-emitted lead the human body, are discussed in Chapter 12.

Harrison (1979) measured the lead content of street dust in Lancaster, England. Results are shown in Table 7-13. Note that the lead content of the dust is greatest in areas of high traffic density. In contrast, Day et al. (1975) measured relatively uniform lead concentrations in dust throughout the city of Manchester, England. Harrison (1979) hypothesized that because Manchester is heavily industrialized, stationary source emissions may be more important than vehicular emissions in determining lead levels there; the influence of mobile sources may be more pronounced in a nonindustrial city such as Lancaster.

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TABLE 7-13. LEAD CONCENTRATIONS IN STREET DUST IN LANCASTER, ENGLAND

Site	No. of samples	Range of concentrations	Mean	Standard deviation
Car parks	4	39,700-51,900	46,300	5,900
	16	950-15,000	4,560	3,700
Garage forecourts	2	44,100-48,900	46,500	--
	7	1,370-4,480	2,310	1,150
Town centre streets	13	840-4,530	2,130	960
Main roads	19	740-4,880	1,890	1,030
Residential areas	7	620-1,240	850	230
Rural roads	4	410-870	570	210

Source: Harrison (1979)

Duggan and Williams (1977) found a similar trend in the lead content of street dust as a function of traffic density. Seventy-nine street dust samples collected in several residential areas of London yielded an average of 1460 ppm lead for main roads and 900 ppm for side roads. An additional seven samples collected from a footpath in a rural area near London contained an average of 35 ppm lead.

Data obtained in a number of other studies on lead in dust near roadways are summarized in Tables 7-14 and 7-15. These data demonstrate that abnormally high concentrations of lead are found in air and dust near major roadways, and that people who live or work in such areas (e.g., traffic policemen, service station and garage attendants) are exposed to high lead concentrations.

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TABLE 7-14. LEAD DUST ON AND NEAR HEAVILY TRAVELED ROADWAYS

Sampling site	Concentration, µg Pb/g
Washington, D.C.:	
Busy intersection	12820
Many sites	(4000-8000) (Fritsch and Prival, 1972)
Chicago:	
Near expressway	6600
Philadelphia:	
Near expressway	(3000-8000) (Kennedy, 1973)
Brooklyn:	
Near expressway	(900-4900) (Lombardo, 1973)
New York City:	
Near expressway	2000 (Pinkerton et al., 1973)
Detroit:	
Street dust	(966-1213) (Ter Haar and Aronow, 1974)
Philadelphia:	
Gutter (low pressure)	1507 (270-2626) (Shapiro et al., 1973)
Gutter (high pressure)	3262 (280-8201) (Shapiro et al., 1973)
Miscellaneous U.S. Cities:	
Highways and tunnels	(10000-20000) (Buckley et al., 1973)
Netherlands:	
Heavily traveled roads	5000 (Rameau, 1973)

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TABLE 7-15. LEAD CONTENT IN OR ON ROADSIDE SOIL AND GRASS AS A FUNCTION OF DISTANCE FROM TRAFFIC AND GRASS DEPTH IN PROFILE^a

Site and distance from road	Lead content, µg/g dry weight			
	Grass	0-5 cm Soil depth	5-10 cm Soil depth	10-15 cm Soil depth
West of U.S. 1, near Plant Industry Station, Beltsville, MD:				
8	68.2	522	460	416
16	47.5	378	260	104
32	26.3	164	108	69
West of southbound lanes, Washington-Baltimore Parkway, Bladensburg, MD:				
8	51.3	540	300	98
16	30.0	202	105	60
32	18.5	140	60	38
West of Interstate 29, Platte City, MO:				
8	21.3	242	112	95
16	12.5	140	104	66
32	7.5	61	55	60
North of Seymour Road Cincinnati, OH:				
8	31.3	150	29	11
16	26.0	101	14	8.2
32	7.6	55	10	6.1

^aAdapted from Lagerwerff and Specht (cited in National Academy of Sciences, 1972).

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For comparison, Table 7-16 summarizes lead in dusts from nominally residential urban areas. These concentrations have a wider range and are in general smaller than those in traffic-oriented dust samples.

It is of interest to determine the fraction of lead in street dust which is soluble in hydrochloric acid at the concentration existing in the stomach. This allows calculation of the amount of lead readily available for absorption into the bloodstream. Day et al. (1979) analyzed samples from Manchester, England and Christchurch, New Zealand for lead using hydrochloric acid at concentrations of 10^{-5} M to 1.0 M (pH 5 to 0). The maximum extractable lead was determined using boiling 2 M nitric acid. Results are presented in Figure 7-6, and show that the solubility is a strong function of pH; most of the lead soluble at a pH of 1, characteristic of the stomach acid concentration. Similarly, Harrison (1979) found that 48-77 percent of the lead in Lancaster street dust is soluble at stomach acid concentration, while Duggan and Williams (1977) report an average of 60 percent soluble lead in London street dust. A large fraction of the lead in street dust is apparently available for absorption into the bloodstream.

Overall, these data suggest that environmental lead levels in the vicinity of heavily traveled roads are elevated above background levels, and that these levels may result in significant increases in lead exposure. Such exposure may be due to direct inhalation of airborne lead, ingestion of food or water containing deposited lead-containing aerosol, or direct ingestion of leaded dust. Accounting for all major exposure pathways, Stephens (1981) estimates that 54 percent of the total lead absorption of a two-year-old child without pica living in an area of high traffic density is due to vehicular-emitted lead. The contribution of lead emitted from mobile rces relative to total lead uptake is discussed further in Chapter 12.

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TABLE 7-16. LEAD DUST IN RESIDENTIAL AREAS

Sampling site	Concentration, $\mu\text{g Pb/g}$
Philadelphia:	
Classroom	2000
Playground	3000
Window frames	1750 (Shapiro et. al., 1973)
Boston and New York:	
House dust	(1000-2000) (Needleman and Scanlon, 1973)
Brattleboro, Vt.:	
In home	(500-900) (Darrow and Schroeder, 1973)
Birmingham, England:	
In home	5000 (Lombardo, 1973)
New York City:	
Middle class	(608-742) (Pinkerton et. al., 1973)
Residential	
Philadelphia:	
Urban industrial	3855 (929-15680) (Needleman et. al., 1974)
Residential	614 (293-1030) (Needleman et. al., 1974)
Suburban	830 (277-1517) (Needleman et al., 1974)
Derbyshire, England:	
Low soil lead area	518 (130-3000) (Barltrop et al., 1975)
High soil lead area	4881 (1050-28000) (Barltrop et al., 1975)

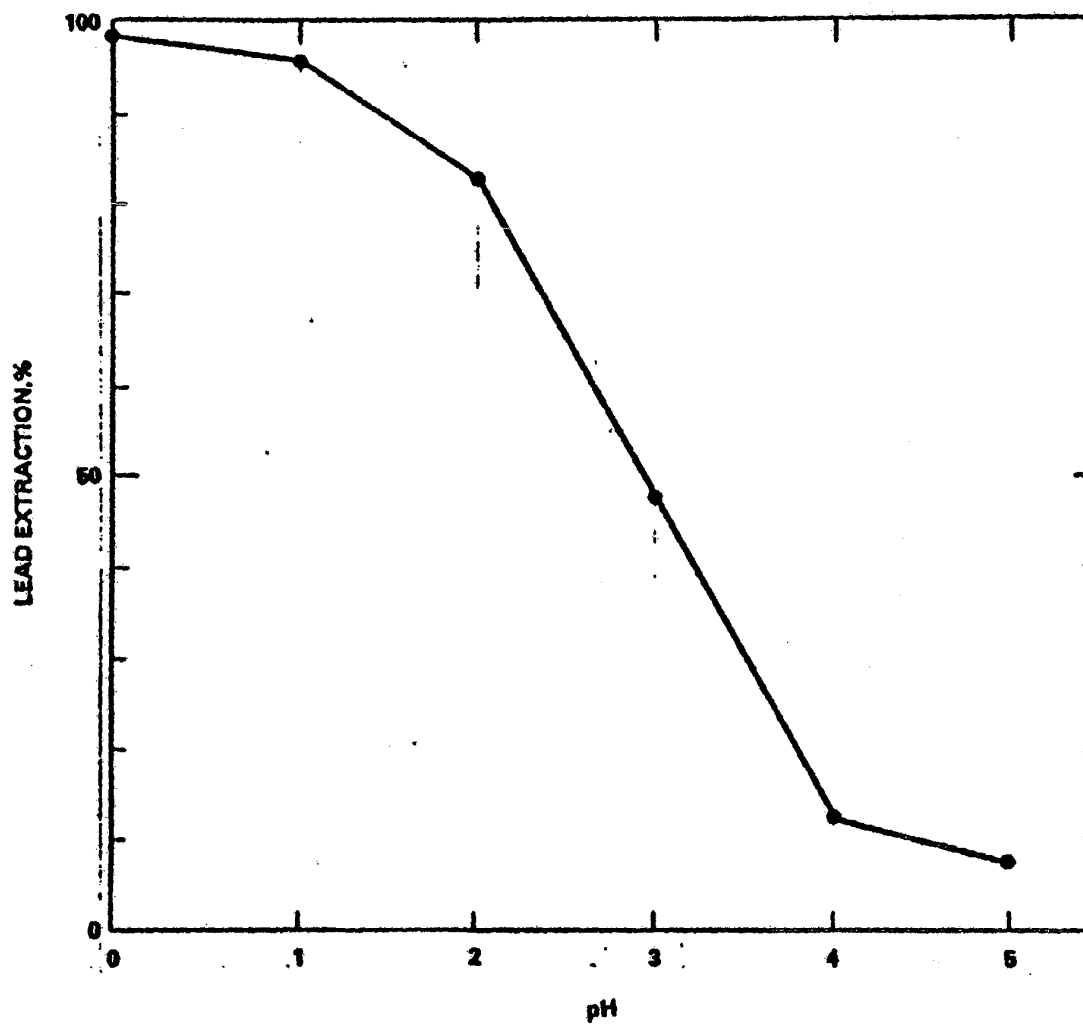


Figure 7-8. Solubility of lead in street dust as a function of pH, using hydrochloric acid at 20° C.

Source: Day et al. (1979).

7.3 POINT SOURCE EXPOSURES

Several studies have been undertaken to investigate lead levels in the vicinity of various point sources of lead emission such as smelters or battery plants. By far the most complete and informative studies are those carried out by Yankel et al. (1977) and by Landrigan et al. (1976) in the neighborhood of a smelter in Silver Valley, Idaho. Consequently, the data from these studies will be described in some detail. Other studies carried out in the United States, Canada, and Europe are summarized in Appendix C. Their findings are in substantive agreement with those of the Idaho study.

Yankel et al. (1977) defined five study areas arranged concentrically around the smelter and two control areas. Area I consisted of homes within 1 mile of the smelter; area II, 1 to 2-1/2 miles from the smelter; area III, 2-1/2 to 6 miles; area IV, 6 to 15 miles; and area V, 15 to 20 miles. Environmental samples, including surface soil, house dust, paint, grass, and garden vegetables were collected at the homes in each area, as were blood samples from the resident children aged 1 to 9 years. The mean lead levels found in the ambient air, soil, and house dust all decreased with increasing distance from the smelter. As mentioned in Chapter 12, the blood lead levels of the resident children followed a similar pattern.

Ambient air lead levels were measured by high-volume samplers stationed throughout Silver Valley. A highly significant relationship between distance from the smelter and ambient lead concentration was found, and relationship was used to estimate the ambient air lead level for any location in the study area. The mean annual ambient air lead levels near the smelter for two different years (1974, 1975) are shown graphically in Figure 7-7. Similar results

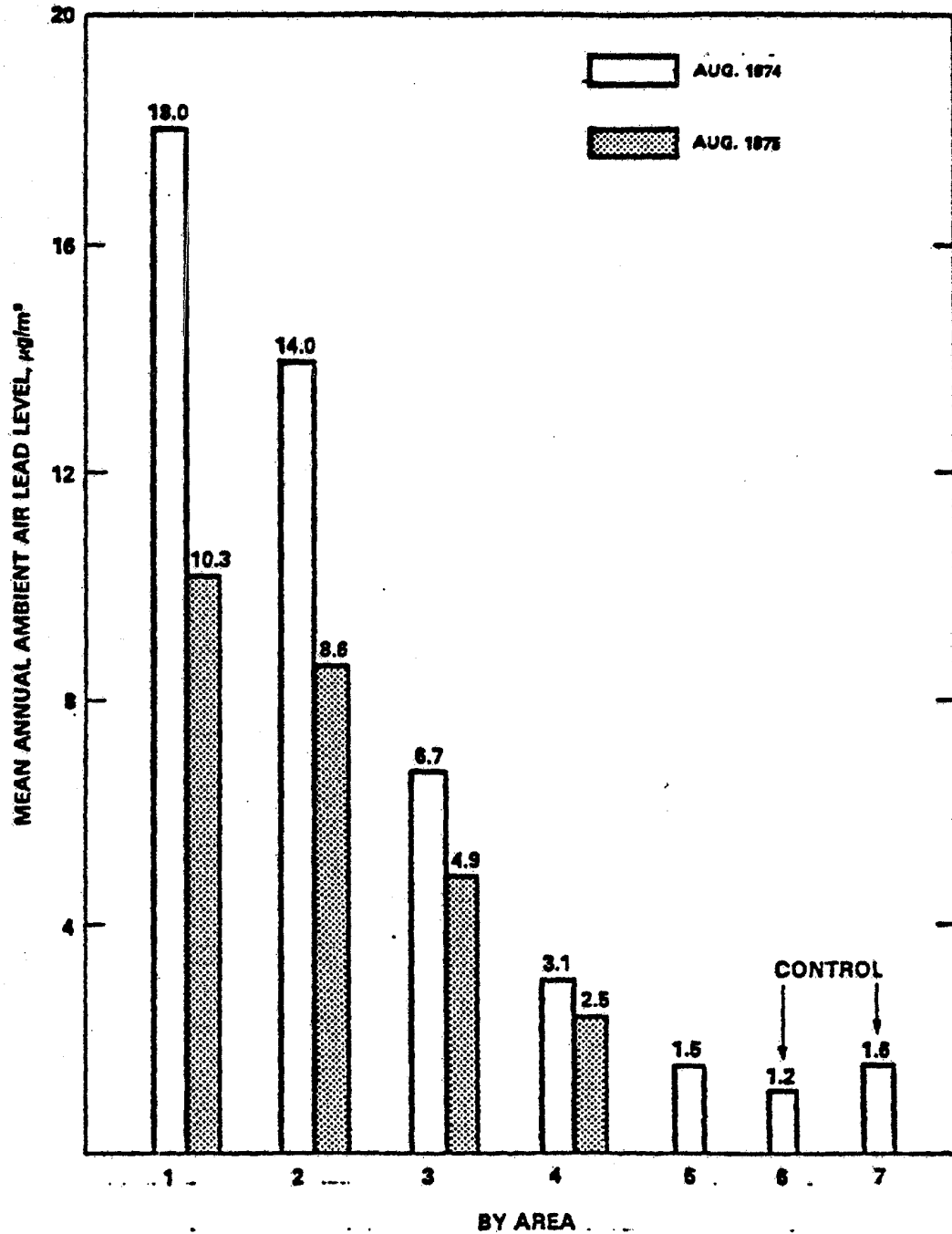


Figure 7-7. Annual ambient air lead concentration near a smelter, by area, before the August 1974 and August 1975 surveys. Area 1 is within 1 mile of smelter; Area 2 is 1 to 1½ miles from smelter; Area 3, 2½ to 6 miles; Area 4, 6 to 15 miles; and Area 5, 15 to 20 miles.
Source: Yankel et al. (1977).

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were obtained for the lead content of soil and house dust. As noted in Chapter 12, the childrens' blood lead levels correlated quite closely with ambient air lead levels, although this result should not be interpreted as suggesting direct inhalation of lead is the principal exposure mechanism. One result of this study was that some specific emergency measures were taken in 1974 (including covering contaminated soil with clean soil and reducing smelter emissions), and these measures brought about a decrease in blood lead levels that were determined a year later. The details of the blood lead levels and their significance are presented in Chapter 12. The conclusion to be drawn from this study (and from the similar studies referred to above) is that people who live in the vicinity of a major industrial source of lead (e.g., a smelter) are exposed to abnormally high lead concentrations.

7.4 DIETARY EXPOSURES

7.4.1 Food

*Gross
Gi* The route by which most people receive the largest portion of their daily lead intake is through foods, with estimates of the dietary lead intake for Americans ranging from 100 to 500 $\mu\text{g}/\text{day}$ (Schroeder and Tipton, 1968; Tepper, 1971; Mahaffey, 1978). Gross (1981) analyzed results of the extensive lead mass balance experiments of Kehoe (1961), which were conducted from 1937 to 1972; according to these data, total dietary lead intake decreased from approximately 300 $\mu\text{g}/\text{day}$ in 1937 to 100 $\mu\text{g}/\text{day}$ in 1970, although there is considerable variability in the data. Only a fraction of this lead is absorbed, as discussed in Chapter 10.

The sources of the lead content of unprocessed vegetable foods have been noted earlier (Section 6.4.3). Studies of the lead associated with crops (near highways) have shown that both lead taken up from soil and aerosol lead delivered by deposition are found with the edible portions of common vegetable

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crops. However, there is enormous variability in the amount of lead associated with such crops and in the relative amounts of lead in the plants versus on the plants. This depends upon several factors, the most prominent of which the plant species, the traffic density, the meteorological conditions, and the local soil conditions (Welch and Dick, 1975; Rabinowitz, 1974; Arvik, 1973; Dedolph et al., 1970; Motto et al., 1970; Schuck and Locke, 1970; Ter Haar, 1970). The variability induced differences in the above factors, coupled with the fact that many studies have neglected differentiation between lead on plants versus lead in plants, makes it difficult to generalize. Data of Schuck and Locke (1970) suggest that in some cases (e.g., tomatoes and oranges) much of the surface lead is readily removed by washing. But as noted in Section 6.4.3, this is not universally true; in some cases, much more vigorous washing procedures are necessary.

In view of the wide variability of soil conditions (pH, organic matter, cation exchange capacity, phosphorus content, etc.), of meteorology (especially wind conditions and rainfall), and of the effects of species diversity on the routes of lead accumulation, only crude general correlations between air lead levels and food crop levels are possible. Furthermore, the lead associated with plants may be derived from natural sources, from automotive sources, and from other sources such as manufacturing or combustion. One study in Southern California reported that 60 to 70 percent of the lead associated with oat tops was directly attributable to automobile (aerosol) emissions, but did not distinguish between lead in the edible portion (grain) and lead on the hulls or chaff (Rabinowitz, 1974). This same study reported that lettuce grown in the Salinas Valley had 3 to 25 ppm lead (dry weight) associated with it, whereas the soil content was only 10 ppm. The lead content in the lettuce was

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reported to be 0.15 to 1.5 ppm on a fresh weight basis. The limited data accumulated were used to deduce that the excess lead was delivered to the lettuce by atmospheric transport of automobile-emitted material, and that removal of lead from automobile exhaust would reduce the lead content of the lettuce by as much as 80 percent (Rabinowitz, 1974). Other studies have similarly reported the importance of deposited airborne lead in influencing lead levels in leafy vegetables (Motto et al., 1970; Schuck and Locke, 1970).

In contrast, food grains may be somewhat less influenced by airborne lead. Ter Haar (1970) found that inedible portions of several plants (bean leaves, corn husks, soybean husks, and chaff from oats, wheat, and rice) had two to three times the lead concentration when grown near a busy highway compared with similar plants grown in a greenhouse supplied with filtered air. The edible portions of these and other plants showed little or no difference in lead content between those grown in ambient air and those grown in the filtered air. Dedolph et al. (1970) found that while ryegrass and radish leaves grown near a busy highway contained deposited airborne lead, the edible portion of the radish was unaffected by variations in either soil lead or air lead. It is noteworthy that the filtered air used in these two studies contained an average lead concentration of 90 ng/m^3 , with limited additional tests run at 30 ng/m^3 ; the ambient air levels were on the order of $1\text{-}2 \text{ } \mu\text{g/m}^3$. Results of the studies might have been different had the lead levels in the filtered air been closer to natural baseline values (less than 1 ng/m^3).

An FDA survey shows that grains contain approximately 20 percent as much lead as leafy vegetables (Kolbye et al., 1974). It cannot be concluded, however, that 80 percent of the lead in all leafy vegetables derives directly

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from air because the difference must also reflect species-dependent differences in uptake from soil.

An overall analysis of the data available supports the contention that plants grown near busy highways consistently have more lead in them and on them than those in other areas. This difference is typically very hard to detect at distances greater than about 100 to 200 m from the highway, due to dilution of the emitted lead as well as settling of the largest lead-containing particles. The available data are not sufficient to permit the quantitative estimate of the contribution of automotive lead to foodstuffs on a national or even a regional scale.

The concentrations of lead in various food items are highly variable, and as much variation is found within specific food items as between different food categories. Schroeder and Balassa (1961), in a study of American foods, have found maximum concentrations of 1.5 mg/kg (ppm) for condiments, 2.5 mg/kg for fish and other seafood, 3.7 mg/kg for meats and eggs, 1.4 mg/kg for grains, and 1.3 mg/kg for vegetables. All of these values refer to unprocessed foods. A British report (United Kingdom Ministry of Agriculture, Fisheries, and Food, 1972) on lead in foods describes similar maximum values for meat and eggs, grain products (flour and bread), and vegetables, but concentrations up to 14 mg/kg in condiments, and up to 18 mg/kg in shellfish.

The amount of lead taken in with food varies from person to person. It depends upon (a) the types of food in the diet, (b) the total amount of food eaten, (c) the history of the food during growth, (d) its opportunity to acquire intrinsic lead (absorbed from soil or water), and (e) the manner in which the food is prepared. As an example of the last category, it has been

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shown that vegetables prepared in water containing lead can absorb a considerable fraction of this lead during boiling. Moore et al. (1979) estimate that water containing 200 μg lead/liter can contribute up to 396 μg lead/day through consumption of vegetables. Little et al. (1981) report ranges of 18-46 μg /day and 72-182 μg /day for vegetables boiled in water containing 50 and 200 μg lead/liter, respectively. The latter study also reports wide variations in lead absorption from the human GI tract for different lead compounds associated with vegetables. Using the United Kingdom Total Diet Study (Buss and Lidsay, 1978) to estimate food consumption, Smart et al. (1981) estimate a lead intake as small as 23 μg /day for vegetables prepared in soft water containing 55 μg lead/liter; hard water containing 520 μg lead/liter provides a vegetable lead intake as great as 130 μg /day. These authors estimate a total dietary lead intake of 87-442 μg /day.

On a per-weight basis, the dietary intake of children has been shown to be two to three times that of adults. This additional dietary intake is especially significant when the lead added to food by processing and to water by plumbing (vide infra) is considered. The Glasgow Duplicate Diet Study (United Kingdom Department of the Environment, 1982) reports that children approximately 13 weeks old living in lead-plumbed houses consume 6 to 480 μg lead/day. Water lead levels in the 131 homes studied ranged from less than 50 to over 500 μg /liter. Those children and mothers living in the homes containing high water lead levels generally had greater total lead consumption and higher blood lead levels, according to the study. Breast-fed infants were exposed to much less lead than bottle-fed infants. Because the project was designed to investigate child and mother blood lead levels over a wide range

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of water lead concentrations, the individuals studied do not represent a typical cross-section of the population. However, results of the study suggest that infants living in lead-plumbed homes may have exposure to considerable amounts of lead. This conclusion was also demonstrated by Sherlock et al. (1982) in a duplicate diet study in Ayr, Scotland.

In a survey of heavy metals in foods, the U.S. Food and Drug Administration (1975) found relatively high lead concentrations in metal-canned foods. In the adult food category, canned foods averaged 0.376 ppm lead, and non-canned foods averaged 0.156 ppm lead. In the baby food category, canned foods (juices) averaged 0.329 ppm lead, and foods in jars averaged 0.090 ppm. The report concluded that from the age of about 1 year on, canned foods comprise 11-12 percent of a person's diet, but they contribute about 30 percent of the average dietary lead intake. In a more recent survey, Beloian (1982) estimates that canned foods contribute 51 percent, 30 percent, and 33 percent of the total dietary lead intake for children aged 0-5 months, 6-23 months, and 2-5 years, respectively. In a comparison made the United Kingdom (Tolan and Elton, 1973), lead concentrations in canned foods were found to vary widely with the precise nature of the food, but they averaged about ten times greater than those in fresh foods.

The soldered seam of tin cans is evidently the major source of the additional lead in canned foods, and increasing lead concentrations in samples of a can's contents taken progressively nearer the seam have been found (Michell and Aldous, 1974). Similarly, there is a correlation between increasing lead concentrations in canned products and the increasing ratio of the cans' seam length to volume. Of 256 metal-canned foods examined, 37 percent contained 200 µg lead/liter or more; 12 percent contained 400 µg lead/liter or more.

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These levels are markedly above the potable water standard of 50 µg lead/liter (0.05 mg/kg) established by the U.S. Public Health Service. However, recent data show that lead levels in canned foods have decreased somewhat through the 1970's (Schaffner, 1981).

Canned pet foods have been found to contain from 0.9 to 7.0 ppm lead (900 to 7000 µg/liter), and 18 products averaged 2.7 ppm (Hankin et al., 1975). Apart from the possible toxic effects on pets, products pose a hazard to people who may include them in their own diet.

The contribution of the canning process to overall lead levels in albacore tuna has been reported by Settle and Patterson (1980). Using rigorous clean laboratory procedures, these investigators analyzed lead in fresh tuna, as well as in processed tuna packaged in soldered and unsoldered cans. The data, presented in Table 7-14, show that lead concentrations in canned tuna are elevated above levels in fresh tuna by a factor of 4000. This factor becomes 40,000 if one compares the canned tuna levels with estimated concentrations assumed to exist prior to the widespread use of lead. Nearly all of the increase results from leaching of the lead from the soldered seam of the can; tuna from an unsoldered can is elevated by a factor of only 20 compared with tuna fresh from the sea. Note that when the tuna is dried and pulverized, as in the NBS reference material, lead levels are seen to increase by a factor of 400 over fresh sea tuna. Table 7-17 also shows the results of analyses conducted by the National Marine Fisheries Service. The lead concentrations reported by NMFS in fresh tuna are considerably greater than the Caltech laboratory data. Settle and Patterson (1980) explain this difference as due to poor contamination control by NMFS during sample handling and analysis; they also state that most other lead analysis laboratories have similar problems which preclude obtaining

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TABLE 7-17. LEAD CONCENTRATIONS IN VARIOUS SAMPLES AS MEASURED BY THE CALTECH AND NMFS LABORATORIES

Sample	Lead Concentration*
Analysis by Caltech laboratory	
Surface seawater, prehistoric (estimated)	0.005
Surface seawater, modern	0.005
Albacore muscle, prehistoric (estimated)	0.03
Albacore muscle, fresh (dissected in Caltech laboratory)	0.3
Albacore muscle from die-punched unsoldered can	7
Albacore muscle, NBS reference material	120
Albacore muscle from lead-soldered can	1,400
Entire albacore	6
Entire anchovy from albacore stomach	21
Part of anchovy from lead-soldered can	4,200
Analysis by NMFS laboratory	
Albacore muscle, fresh (dissected by NMFS Laboratory)	400
Albacore muscle, fresh (dissected in Caltech Laboratory)	20
Albacore muscle from lead-soldered can	700

*All values are expressed as ppb wet weight.

Source: Settle and Patterson (1980).

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accurate lead data at levels commonly encountered in food and in the environment. Such contamination problems are discussed further in the final section of this section.

The lead content in milk is of special interest because it is a major component in the diets of infants and young children. Brandt and Bentz (1971) report that levels in fresh milk are typically less than 5 $\mu\text{g/liter}$. The survey by the U.S. Food and Drug Administration (1975) found lead concentrations in whole milk ranging from 10 to 70 $\mu\text{g/liter}$, and averaging about 20 $\mu\text{g/liter}$. In a recent study by Ziegler et al. (1978), seven samples of baby formula and three samples of whole cow's milk were analyzed in duplicate. Mean concentrations of lead were 18 $\mu\text{g/kg}$ (range 15 to 20) in formula and 10 $\mu\text{g/kg}$ in milk (1 $\mu\text{g/kg}$ is approximately 1 $\mu\text{g/liter}$). Lead concentrations in infant fruit juices ranged from 23 to 327 $\mu\text{g/kg}$; in five varieties of strained fruits, it ranged from 13 to 131 $\mu\text{g/kg}$; and in seven varieties of strained vegetables, lead concentrations were 14 to 73 $\mu\text{g/kg}$. Tolan and Elton (1973) reported lead levels of 30 $\mu\text{g/liter}$ in fresh milk in Great Britain and 50 $\mu\text{g/liter}$ in canned (evaporated) milk. Michell and Aldous (1974) reported a comparable average for fresh whole milk purchased in New York State--40 $\mu\text{g/liter}$. But their results for evaporated milk averaged 202 $\mu\text{g/liter}$ and ranged as high as 820 $\mu\text{g/liter}$.

Hankin et al. (1974) suggest an additional food-related source of potential lead exposure, again predominantly affecting children. The colored portions of wrappers from bakery confections, candies, gums, and frozen confections have lead concentrations ranging from 8 to 10,100 ppm. The higher concentrations are attributed to lead-containing inks. No related illnesses

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were identified, nor was contamination of the food implied; but the eating of foods from such wrappers and the licking or chewing of the wrappers were postulated as one more avenue for an additional increment of lead exposure.

The presence of high-lead concentrations in illicit whiskey moonshine), which is still popular in some parts of the United States despite the repeal of prohibition, causes lead poisoning in adults. The apparent source of the lead is the soldered joints in the distilling apparatus.

Another potential source of dietary lead poisoning is the use of inadequately glazed earthenware vessels for food storage and cooking. An impressive example of this danger involved the severe poisoning of a physician's family in Idaho which resulted from drinking orange juice that had been stored in an earthenware pitcher (Block, 1969). Similar cases, sometimes including fatalities, have involved other relatively acidic beverages such as fruit juices and soft drinks, and have been documented by other workers (Klein et al., 1970; Harris and Elsen, 1967). Because of these incidents, the U.S. Food and Drug Administration (1979) has established a maximum permissible concentration of 7 ppm lead in solution after leaching with 4 percent acetic acid in the earthenware vessel for 24 hours.

Inadequately glazed pottery manufactured in other countries continues to pose a significant health hazard. For example, Spielholtz and Kaplan (1980) report 24-hour acetic acid-leached lead concentrations as great as 4400 ppm in Mexican pottery. The leached lead decreases with exposure time, and after several days appears to asymptotically approach a value which may be as great as 600 ppm. These investigators have also measured excessive lead concentrations leached into acidic foods cooked for two hours in the same pottery. Similarly, Acra et al. (1981) report that 85 percent of 275 earthenware vessels

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produced in primitive Lebanese potteries had lead levels above the 7 ppm USFDA limit. However, only 9 percent of 75 vessels produced in a modern Beirut pottery exceeded the limit. Cubbon et al. (1981) have examined properly glazed ceramic plates in the United Kingdom, and have found a decrease in leached lead with exposure time down to very low levels. The authors state that earthenware satisfying the 7 ppm limit will contribute about 3 µg/day to the dietary intake of the average consumer.

Reports on lead in European wines (Olsen et al., 1981; Boudene et al., 1975; Zurlo and Graffini, 1973) show concentrations averaging 100-200 µg/liter and ranging as high as 299 µg/liter. Measurements of lead in domestic wines have not been undertaken; if the European data are indicative, domestic wines could contain lead concentrations comparable to processed foods previously discussed. Lead levels in beer are generally smaller than those in wine: Thalacker (1980) reports a maximum concentration of 80 µg/liter several brands of German beer.

Lead is also present in tobacco. WHO (1977) estimates a lead content of 2.5-12.2 µg per cigarette; roughly two to six percent of this lead may be inhaled by the smoker. The Committee on Lead in the Human Environment, National Research Council (1980) has used these data to conclude that a typical urban resident who smokes 30 cigarettes per day may inhale roughly equal amounts of lead from smoking and from breathing urban air.

7.4.2 Water

The U.S. Public Health Service standards specify that lead levels in drinking water should not exceed 50 µg/liter. The average adult drinks about 1 liter of water per day. The presence of detectable amounts of lead in untreated public water supplies was shown by Durum (1971) to be widespread, but only a few samples contained amounts above the 50 µg/liter standard. Durfor and

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Becker (1964) analyzed untreated and treated water for the largest United States cities, and almost all pairs of samples showed a substantial decrease in lead that was ascribable to treatment provided. A maximum lead concentration of 62 µg/liter was detected in finished water from one of several wells used in Salt Lake City to supplement their surface water supply. Some 95 percent of the water supplies sampled, however, had lead concentrations below 10 µg/liter in the treated water before entering the distribution system. Eight of the water supplies distributed water with a pH of less than 7, which could be corrosive to the distribution piping; most of these were in the Northwest. A chemical analysis of 592 interstate carrier water supplies in 1975 showed that only 0.3 percent exceeded the 50 µg/liter standard (U.S. Environmental Protection Agency, 1975). These samples were collected after treatment but before distribution, and they represent both suspended and dissolved lead. Interstate carrier water supplies serve planes, trains, buses, and vessels in interstate commerce, and they include almost all of the largest United States water supplies.

The presence of lead in drinking water may result from contamination of the water source or from the use of lead materials in the water distribution system. Although lead is a relatively minor constituent the earth's crust, it is widely distributed in low concentrations in sedimentary rock and soils (as discussed in Chapter 3), and naturally occurring deposits may be an important source of contamination in isolated instances. Industrial waste may also contribute to the lead content of water sources, but this appears to be a local and not a widespread problem. The extensive use of lead compounds as gasoline additives has greatly increased the availability of lead for solution in ground and surface waters. For example, in a study in east-central Illinois

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(Rolfe and Haney, 1975), the urban portion of an 86-square-mile watershed (constituting 14 percent of the area) contributed about 75 percent of the lead in drainage waters. The principal source of this lead was identified as automotive emissions. Detailed data reported for 1 month (June 1972) showed that drainage waters from this urban portion contained an average total lead concentration of 69.5 µg/liter, including 6.3 µg/liter of soluble lead. The rural portion yielded an average lead concentration of 7.4 µg/liter in drainage water, including 2.1 µg/liter of soluble lead.

Hem and Durum (1973) discussed the solubility of those species of lead that may be present in drinking water and suggested that the solution lead from environmental sources may be an important contribution in certain areas, depending upon the chemical composition of the runoff water. Above pH 8.0, the solubility of lead is below 10 µg/liter, regardless of the alkalinity of the water. In waters near pH 6.5 with alkalinity, however, the solubility of lead could approach or exceed 100 µg/liter.

Lazrus et al. (1970) determined the lead content of precipitation at 32 points in the United States for a period of 6 months in 1966 and 1967. They reported an average lead concentration of 34 µg/liter after filtering the samples. Samples of rainfall at Menlo Park, California during 1971 showed a wide range of lead concentrations, from a few µg/liter to more than 100 µg/liter (Hem and Durum, 1973). These authors hypothesized that higher lead concentrations should occur in runoff water and impounded raw water supplies in the Northeast, certain urban areas of the South, and along the Pacific Coast because of low pH and alkalinity in waters. However, in much of rest of the United States, lead fallout rates and chemical composition of the runoff (pH > 8, alkalinity > 100 µg/liter) would minimize the problem. Information to test their hypothesis is limited at present. Of the few surveys of surface waters

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that have been conducted, most were not done after periods of heavy rainfall, and most have measured dissolved rather than total lead.

Durum (1971) measured lead at 700 lake and river sites in the United States. These measurements were primarily single samples, taken at times of relatively low stream flows in October and November 1970. Detectable concentrations of dissolved lead (>1 $\mu\text{g/liter}$) were found in 63 percent of the samples, but only three samples contained more than 50 $\mu\text{g/liter}$. A large proportion of the samples for the northeastern and southeastern states contained lead above the detection limit, and quite a few of the samples showed levels above 10 $\mu\text{g/liter}$. This regional distribution of lead in stream water is in accord with the idea that water composition in the eastern states is more commonly favorable for solution of lead. A substantial number of samples from southern California were high in lead, and these influenced the data from the southwestern states.

Kopp and Kroner (1967) presented data on dissolved lead in rivers and lakes of the United States. The data were gathered over a 5-year period (1962 to 1967) and represent more than 1500 samples. A detectable concentration of dissolved lead was found in 305, or 19.3 percent, the samples; the observed values ranged from 2 to 140 $\mu\text{g/liter}$. The highest concentration was detected in the Ohio River at Evansville, Indiana. Twenty-seven of their samples exceeded 50 $\mu\text{g/liter}$. Observed mean observations of >30 $\mu\text{g/liter}$ dissolved lead were found in the following river basins: Ohio, Lake Erie, Upper Mississippi, Missouri, Lower Mississippi, and Colorado.

The major source of lead contamination in drinking water is the water supply system itself. Water that is corrosive can leach considerable amounts of lead from lead plumbing and lead compounds used to join pipe. Several widely adopted codes, such as the ASA-A40 Code, Uniform Plumbing Code, and

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BOCA Code, allow the use of lead pipe and list lead an acceptable soldering material for joining pipes that convey water. Lead pipe is currently in use in many parts of the United States for water service lines and interior plumbing, particularly in older urban areas. In a community water supply survey of 969 water systems conducted in nine geographically distributed areas of the United States in 1969 and 1970, it was found that 1.4 percent of all tap water samples exceeded the 50 µg/liter standard (McCabe et al., 1970). The maximum concentration found was 640 µg/liter total standard. The occurrence of samples exceeding the standard was more prevalent in waters with a relatively low pH and low specific conductance. It was estimated that 2 percent of the survey population of 18.2 million was exposed to high lead levels at the tap. Considerable research related to lead in home tap water has been conducted in the United Kingdom, where lead plumbing is prevalent. Moore (1977) reported lead concentrations in Glasgow drinking water to be considerably greater than 50 µg/liter in many homes using lead pipes, due to leaching caused by the soft water. The lead levels decreased if the water was allowed to run for several minutes. Highest lead concentrations were found in the early morning after the water had been standing in the pipes all night (Figure 7-8). This study also measured blood lead concentrations in residents of western Scotland, and found a significant correlation between blood lead and water lead. It was concluded that the use of lead plumbing in soft water areas may pose a more severe hazard than atmospheric lead.

In another study by Moore et al. (1977), blood lead levels in mentally retarded children were found to be greater than in control children. A significant correlation was also found between blood lead and water lead levels in the childrens' homes. Water lead concentrations as great as 1850 µg/liter were measured in these homes.

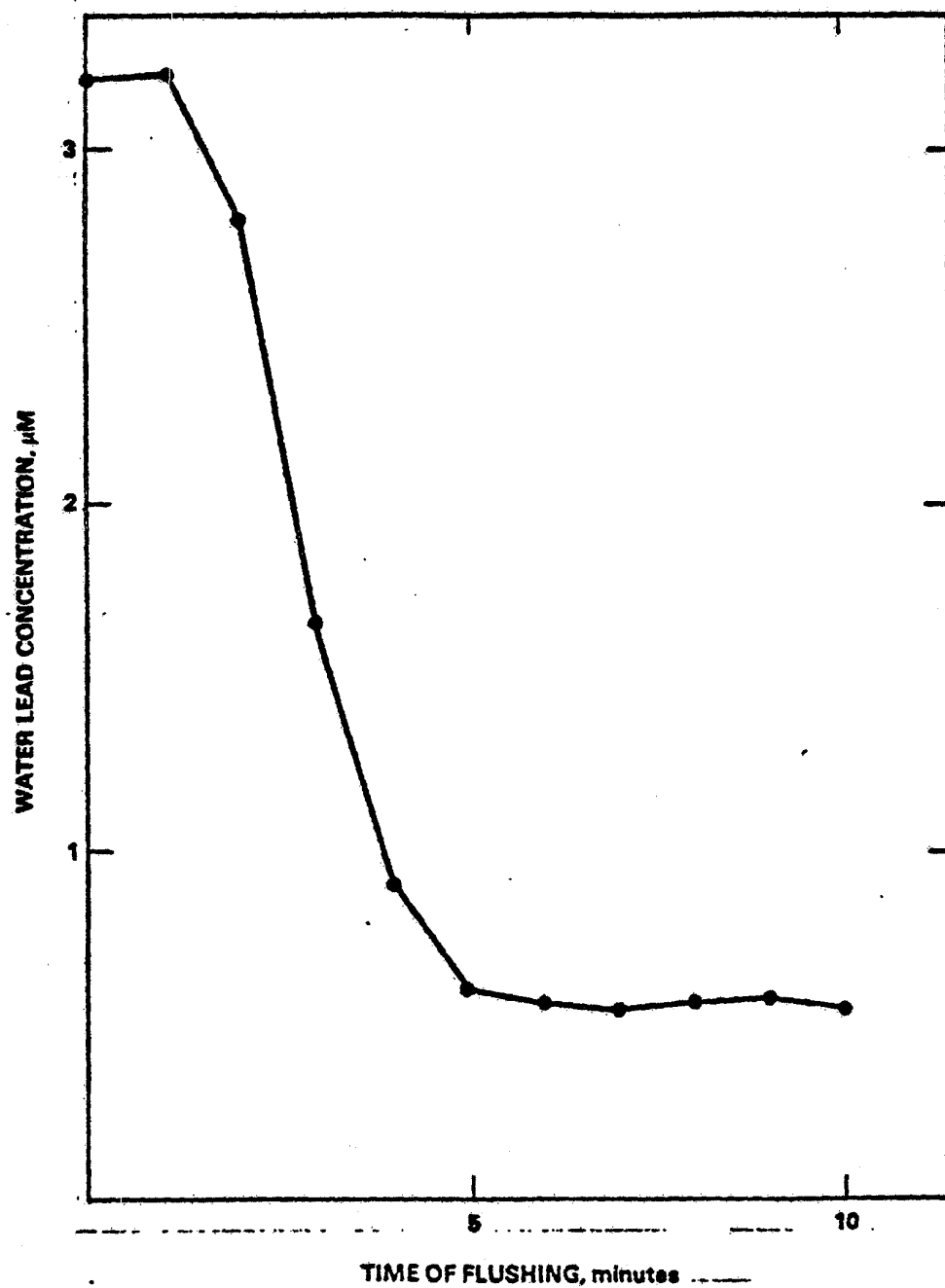


Figure 7-8. Change in drinking water lead concentration in a house with lead plumbing for the first use of water in the morning. Flushing rate was 10 liters/minute. Source: Moore (1977).

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In April 1978, the pH of the water supply to 920,000 Glasgow residents was increased from 6.3 to 7.8 using lime-dosing. Prior to this treatment, over 50 percent of the random daytime water samples collected in the city had lead levels above 100 µg/liter. This figure was reduced to 20 percent following treatment (Moore et al., 1981). In August 1980, the pH of the water supply was increased to 9, resulting in an estimated further reduction to 5 percent of the water samples in excess of 100 µg/liter. The reduction in water lead content caused a significant decrease in blood lead levels of mothers in the postnatal ward of a Glasgow hospital, demonstrating the positive effect of the lime-dosing treatment.

A series of studies has been conducted in northern Wales involving lead levels in home tap water. In one study, mean lead concentrations first draw water samples from 14 dwellings with lead plumbing was 370 µg/liter. This level dropped to 85 µg/liter after the water was allowed to run for ten minutes (Thomas and Elwood, 1978). In a related set of studies (Badawy, 1978; Thomas et al., 1979; Thomas, 1980) water lead levels were measured in 60 lead-plumbed houses and in 75 copper-plumbed houses: first flush samples averaged 1075 µg lead/liter in the former, but only 4 µg/liter in the latter. Blood lead concentrations in mothers and their children living in the lead-plumbed homes were significantly greater than concentrations in residents of the houses which used copper plumbing. Within the homes containing lead plumbing, lead levels were greatest in those individuals who regularly consumed first draw water.

A survey of water lead concentrations in over 2000 households throughout Great Britain has been conducted by Pocock (1980). The effects of water stagnation time in the pipes, pH, alkalinity, position/extent of lead plumbing,

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age of dwelling, and number of occupants were investigated. Results were in agreement with previous studies, showing that soft, acidic water in homes with considerable lead plumbing has the greatest lead concentration. On the average, the daytime concentration of lead was about 57 percent of the first draw concentration.

Bailey and Russell (1981) have developed a model for population exposure to lead in home drinking water. The model incorporates data for lead concentration as a function of stagnation time in the pipes, as well as probability distributions for times of water use throughout the day. Population surveys conducted as part of the United Kingdom Regional Heart Survey provided these water use distributions. The final probability density function for water lead levels is given in Figure 7-9. The effect of instructing all individuals to refrain from drinking first draw water is evident; median, mean, and 95th percentile lead levels decrease measurably.

Other recent studies have been conducted in Canada and Belgium. Lead levels in water boiled in electric kettles were measured in 574 households in Ottawa (Wigle and Charlebois, 1978). Concentrations greater than 50 µg/liter were observed in 42.5 percent of the households, and excessive lead levels were associated with kettles more than five years old. Blood lead concentrations were not significantly correlated with lead levels in water boiled in electric kettles. However, age, sex, and smoking habits were correlated with blood lead. The authors concluded that lead exposure from such kettles does not pose a significant health hazard to adults, but may present a hazard to infants.

Hubermont et al. (1978) examined the influence of water lead concentration on the transplacental transfer of lead in 70 pregnant women in rural Belgium. Water lead levels ranged from 0.2 to 43.4 µg/liter in 41 households (group A), and from 61.5 to 1228.5 µg/liter in an additional 29 households (group B).
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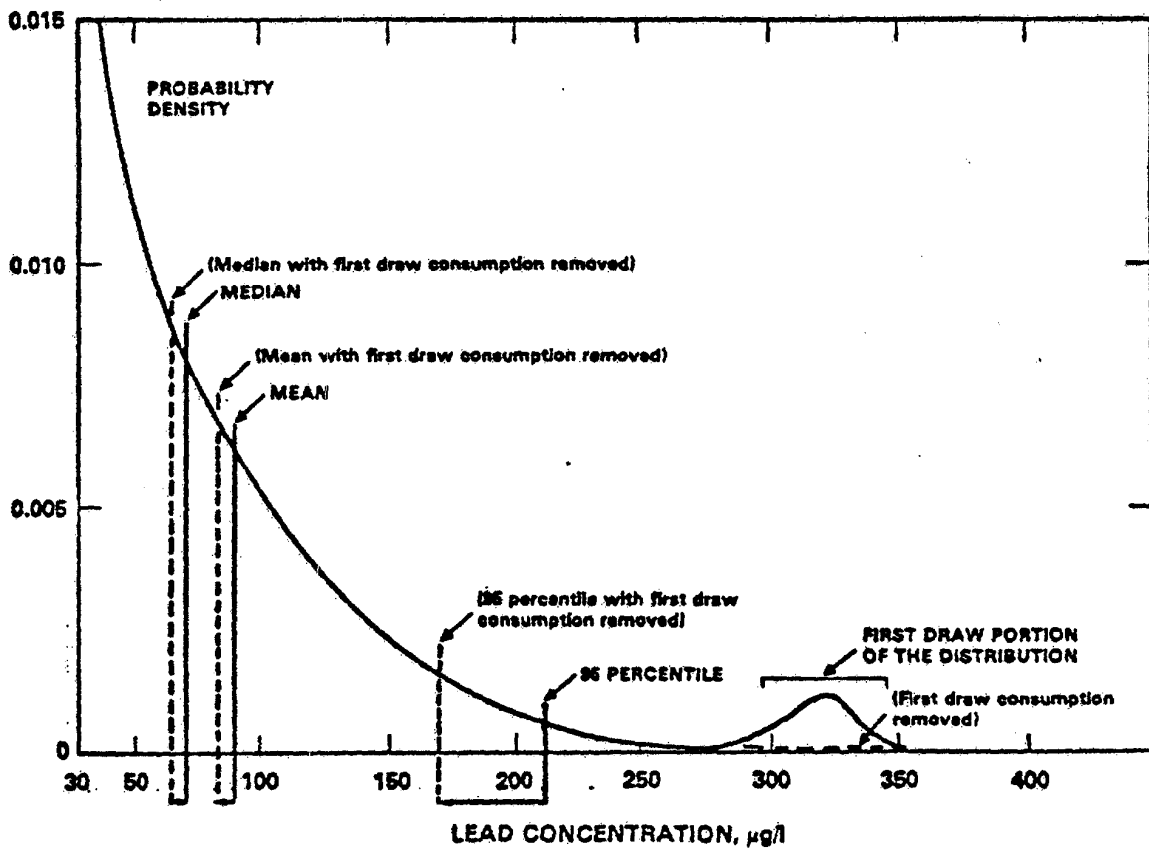


Figure 7-9. Theoretical distribution of drinking water lead concentrations based on the model of Bailey and Russell (1981). Shown on the graph are the reductions in median, mean, and 95 percentile lead levels if first draw water consumption is removed.

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Blood lead levels in the mothers, umbilical cords, and placentas were significantly greater in group B than in group A.

Sartor and Rondia (1980) studied the relation between water lead level and blood lead concentration in two Belgian urban areas: Liege (population 426,777, serviced by hard water) and Verviers (population approximately 50,000, serviced by soft water). The subjects included males over a wide range of ages who were not occupationally exposed. A total of 390 Liege residents and 320 Verviers residents were sampled. Water lead levels in Verviers were considerably greater than those in Liege, with some concentrations as great as 1500 µg/liter. The blood lead levels in Verviers residents were considerably greater than levels in Liege residents, presumably due to the higher water lead concentrations. Blood lead levels in Liege increased with age until approximately 25 years, then achieved a constant value. In Verviers, blood lead levels continued to increase past age 60. The authors concluded that the excessive lead exposure in Verviers lengthened the exposure time required to reach equilibrium between blood lead and environmental lead levels; equilibrium was not achieved in a normal lifetime.

Numerous studies regarding the lead content of drinking water, and the relationship to blood lead levels, were conducted in Europe in the early and mid 1970's. A thorough literature review of these studies has been conducted by Berlin et al. (1977). On the basis of these studies, it was concluded that lead in drinking water at even modest concentrations may have a significant effect on blood lead level, both from direct ingestion and from cooking with the water.

EXPOSURE FROM MAN-MADE MATERIALS

At least two manmade materials in widespread use are known to contain lead, namely paint and plastics.

In 1974, the Consumer Product Safety Commission collected several household paint samples and analyzed them for lead content (Committee on Toxicology, National Research Council, 1976). Analysis of 489 samples showed that 8 percent of the oil-based paints and 1 percent of the water-based paints contained greater than 0.5 percent lead (5000 µg lead/g paint, based on dried solids), which was the statutory limit at the time of the study. The current statutory limit for Federal construction is 0.06 percent. Several studies have shown that children living in homes containing accessible leaded paint may have a greater tendency to develop elevated blood lead levels. For example, Guinee (1972) found loose leaded paint in 76 percent of the houses of lead-poisoned children, compared with 38 percent of the houses of non-poisoned children (controls). Gilbert et al. (1979) reported loose leaded paint in 100 percent of the houses of children with elevated blood lead levels, compared with 50 percent for controls. A significant correlation was found between child fecal lead content and the presence of leaded paint, and between blood lead content and the presence of leaded paint (Hammond et al., 1980). The greatest amounts of leaded paint are typically found in the kitchens, bathrooms, and bedrooms (Tyler, 1970; Laurer et al., 1973; Gilbert et al., 1979). However, Stark et al. (1982) found a better correlation between exterior paint lead content and blood lead than between interior paint lead and blood lead.

*Focused on
Lead Paint
Hazard*

Some investigators have shown that flaking paint can cause elevated lead concentrations in nearby soil. For example, Hardy et al. (1971) measured soil lead levels of 2000 ppm next to a barn in rural Massachusetts. A steady

*Discuss w/ of Ill
study here*

decrease in lead level with increasing distance from the barn was shown, reaching 60 ppm at fifty feet from the barn. Ter Haar (1974) reported elevated soil lead levels in Detroit near eighteen old wood frame houses painted with lead-based paint. The average soil lead level within two feet of a house was just over 2000 ppm; the average concentration at ten feet was slightly more than 400 ppm. The same author reported smaller soil lead elevations in the vicinity of eighteen brick veneer houses in Detroit. Of course, vehicular-emitted lead as well as paint lead may be deposited near houses and other structures which impede airflow. This aspect is discussed further in Appendix C.

*Ter Haar
discusses
this*

Mouthing of non-food items is prevalent in most youngsters, and hence ingestion of leaded paint is a major problem for small children in general. An especially severe health problem is posed for children with pica, who may habitually ingest 1 to 3 g (or more) of paint per week (Committee on Toxicology, National Research Council, 1976).

Plastics contain a number of heavy metals that are constituents of organometallic stabilizers added during the manufacturing process. The most commonly used lead-containing stabilizer is dibasic lead stearate, in amounts ranging from 0.5 to 2.0 parts per 100 parts of resin (Piver, 1977). The stabilizer is normally used in rigid PVC ducts. Diffusion, or leaching by solvents, is estimated to be quite slow--on the order of 10^{-10} to 10^{-12} cm²/sec at room temperature--but no definitive information is available.

Incineration of lead-containing plastics may become an increasingly significant source of localized lead pollution. It has been estimated that in the year 2000, for example, there could be approximately 2.54×10^9 kg of PVC plastic waste to be disposed of annually, of which about 0.59×10^9 would

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probably be incinerated (Vaughn et al., 1975). Assuming that lead will be emitted from the uncontrolled incineration of PVC's at the rate of 0.2 g lead/kg waste (a figure applied to all solid waste, according to the U. S. Environmental Protection Agency, 1976), about 1.2×10^5 kg lead could be released each year. This would be an increase of more than fourteenfold over the estimate for 1975. Since the greater part of the lead in these incinerated plastic wastes will remain in the ash, electrostatic precipitators can substantially decrease the emitted fraction (to an estimated 0.03 g/kg). This process only aggravates the difficulties of residual solid waste disposal with its attendant problems of fugitive dust and the potential contamination of soil, surface waters, and ground waters through leaching from landfill operations.

Lead is present in other products which may constitute sources of lead exposure when used or disposed of. Lead may be found in newsprint, craft and hobby materials, toothpaste tubes, cosmetic products, candle wicks, pewter and silver hollowware, painted utensils, and decals on glassware. For example, colored newsprint may contain up to 2800 ppm lead (Hankin et al., 1973), while colored gift wrapping may contain up to 14,300 ppm lead (Bertagnolli and Katz, 1979). Some cosmetic hair-darkening preparations contain lead acetate, although the absorption of this lead through the skin is small (Moore et al., 1980). Lead in the paint on handles of kitchen utensils has been found by Hankin et al. (1976) to contain as much as 97,000 ppm lead (9.7 percent); more than half of the samples exceeded the allowable limit for painted toys, which is 0.06 percent.

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7.6 TOTAL EXPOSURE FROM VARIOUS SOURCES

On the basis of the published studies summarized in this chapter, it is apparent that the total exposure to lead for the general population depends upon the contributions from several pathways. The amount of airborne lead inhaled, the amount of lead ingested from food, water, dust, and manmade materials, and the absorption of this lead into the body are important factors. Drill et al. (1979) used typical lead concentrations in various media, with estimates of human intake and absorption factors, to determine the contribution to total lead absorbed in urban children from each of several exposure pathways. The technique was applied by the Committee on Lead in the Human Environment, National Research Council (1980) to estimate lead absorption for four subsets of the general population: children with pica and an accessible source of lead-based paint, children without pica, urban adults who smoke cigarettes, and rural nonsmoking adults. The results of these calculations are shown in Tables 7-15a and 7-15b. Note that ingestion of leaded paint is by far the most important exposure pathway for children in the first category; ingestion of food dominates for the other three populations. It is important to recognize that a certain fraction of the lead in food may result from lead-containing particles which have settled from the atmosphere, hence airborne lead may have a significant effect on total lead exposure. Of course, the values in Tables 7-15a and 7-15b are only rough estimates. Environmental lead concentrations, amounts consumed, and absorption factors can vary markedly from individual to individual. For example, note that the dietary contributions in the Table are somewhat smaller than the estimates given in Section 7.4.1. because the emissions of lead from motor vehicles have decreased over the past several years, it is likely that the exposures in some categories of Tables 7-15a and 7-15b may be greater than typical current exposures.

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TABLE 7-15a. TWO HYPOTHETICAL ESTIMATES OF CONTRIBUTIONS OF SPECIFIC ROUTES OF EXPOSURE TO THE TOTAL ABSORPTION OF LEAD BY POPULATIONS OF URBAN CHILDREN

Route of Exposure	Concentration of lead in environment	Amount inhaled or ingested per day	Absorption factor	Amount of lead absorbed (µg/day)	Percent of total lead absorbed
Case I: Children with pica and an accessible source of lead-based paint					
Air	0.75 µg/m ³	10m ³	0.4	3	0.5
Water	10 µg/l	1.4l	0.5	7	1.3
Food	0.1 µg/g	1,000 g	0.5	50	9.1
Soil/dust	500 µg/g ^b	1 g	0.03	150	27.3
Paint	10,000 µg/g	0.2 g	0.17	340	61.8
Total				550	100
Case II: Children without pica and without accessible paint					
Air	0.75 µg/m ³	10 m ³	0.4	3	4
Water	10 µg/l	1.4 l	0.5	7	9
Food	0.1 µg/g	1,000 g	0.5	50	67
Soil/dust	500 µg/g	0.1 g	0.3	15	20
Paint	--	--	--	--	--
Total				75	100

Source: Committee on Lead in the Human Environment, National Academy of Sciences (1980).

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TABLE 7-15b. TWO HYPOTHETICAL EXAMPLES OF ESTIMATES OF CONTRIBUTIONS OF SPECIFIC SOURCES OF EXPOSURE TO TOTAL ABSORPTION OF LEAD FOR SUBSETS OF THE GENERAL ADULT POPULATION

Route of Exposure	Concentration of lead in environment	Amount inhaled or ingested per day	Absorption factor	Amount of lead absorbed (µg/day)	Percent of total lead absorbed
Case I: Urban residents who smoke cigarettes					
Air	0.75 µg/m ³	20 m ³	0.4	6	13.3
Water	10 µg/l	2l	0.1	2	4.4
Food	0.15 µg/g	2,000 g	0.1	30	66.7
Soil/dust	500 µg/g	0.02 g	0.1	1	2.2
Tobacco	0.5 µg/cig	30 cigs	0.4	6	13.3
Total				45	100
Case II: Rural residents who do not smoke					
Air	0.083 µg/m ³	20 m ³	0.4	0.7	2
Water	10 µg/l	2l	0.1	2	6
Food	0.15 µg/g	2,000 g	0.1	30	91.7
Soil/dust	50 µg/g	0.02 g	0.1	0.1	0.3
Total				32.8	100

Source: Committee on Lead in the Human Environment, National Academy of Sciences (1980).

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It is apparent that lead exposures of children can be considerably greater than those of adults. The second national Health and Nutrition Examination Survey (HANES II) shows that blood lead levels generally decline during pre-school years. This may be due to decreased mouthing of non-food objects and decreased hand-to-mouth activity, as well as physiological changes. The latter include a decrease in basal metabolism rate with age (resulting in smaller amounts of air, food, and water consumed per unit body weight) and reduced gastrointestinal tract absorption (Mahaffey et al., 1979). The fact that children are exposed to greater amounts of lead than adults, combined with the greater sensitivity of children to the effects of lead (Needleman and Landrigan, 1981), suggest that child exposures via all major pathways merit considerable attention.

7.7 OCCUPATIONAL EXPOSURES

The highest and most prolonged exposures to lead are found among workers in the lead smelting, refining, and manufacturing industries (World Health Organization, 1977). In the work areas, the major route of lead exposure is by inhalation and ingestion of both lead-bearing dusts and fumes. Airborne dusts settle out from the air onto food, water, the workers' clothing, and other objects, and may be subsequently transferred to the mouth. Therefore, good housekeeping and good ventilation have a major impact on exposure. It has been found that exposure levels might be quite high in one factory and low in another solely because of differences in ventilation, or differences in housekeeping practices and worker education.

7.7.1 Exposures in Lead Mining, Smelting, and Refining

The greatest potential for high-level exposure exists in the process of lead smelting and refining (World Health Organization, 1977). The most hazardous operations are those in which molten lead and lead alloys are brought to

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high temperatures, resulting in the vaporization of lead. This is because condensed lead vapor or fume has, to a substantial degree, a small (respirable) particle size range. Although the total air lead concentration may be greater in the vicinity of ore-proportioning bins than it is in the vicinity of a blast furnace in a primary smelter, the amount of particle mass in the respirable size range may be much greater near the furnace.

7.7.1.1 Primary Lead Smelters--A measure of the potential lead exposure in primary smelters was obtained in a study of three typical installations in Utah (World Health Organization, 1977). Air lead concentrations near all major operations, as determined using personal monitors worn by workers, were found to vary from about 100 to more than 4000 $\mu\text{g}/\text{m}^3$. Obviously, the hazard to these workers would be extremely serious if it were not for the fact that the use of respirators is mandatory in these particular smelters. Maximum airborne lead concentrations of about 300 $\mu\text{g}/\text{m}^3$ were measured in a primary lead-zinc smelter in the United Kingdom (King et al., 1979). These authors found poor relations between airborne lead and blood lead in the smelter workers, and concluded that a program designed to protect these workers should focus on monitoring of biological parameters rather than environmental levels.

7.7.1.2 Secondary Lead Smelters--The operation of secondary smelters is similar to that of primary smelters except that no ore-processing is involved, since secondary smelters depend upon the local supply of lead scrap in the form of discarded electric storage batteries, cable casings, pipes, and other materials for their supply of lead. Consequently, the exposure hazard to workers in secondary smelters is probably similar to that found in primary smelters.

Spivey et al. (1979) studied a secondary smelter in southern California which recovers lead mainly from automotive storage batteries. Airborne lead

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concentrations of 10 to 4840 $\mu\text{g}/\text{m}^3$ were measured. The project also involved measurement of biological parameters as well as a survey of symptoms commonly associated with lead exposure; a poor correlation was found between indices of lead absorption and symptom reporting. The authors suggested that such factors as educational level, knowledge of possible symptoms, and biological susceptibility may be as important as lead absorption in influencing symptom reporting. In a second article covering this same study, Brown et al. (1980) reported that smokers working at a smelter had greater blood lead levels than non-smokers. Furthermore, smokers who brought their cigarettes into the workplace had greater blood lead levels than those who left their cigarettes elsewhere. It was concluded that direct environmental contamination of the cigarettes by lead-containing dust may be a major exposure pathway for these individuals.

Winegar et al. (1977) examined environmental concentrations as well as biological indicators and symptom reporting in workers in a secondary lead smelter near St. Paul, Minnesota. The smelter recovers approximately 9000 metric tons of lead per year from automotive batteries. The lead concentrations in cuff dust from trousers worn by two workers were 60,000 and 600,000 ppm. The amount of lead contained in pieces of cloth 1 in² cut from the bottoms of trousers worn by six workers ranged from 700 to 19,000 μg , with a median of 2640 μg . In all cases, the trousers were worn under coveralls. Dust samples from 25 households of smelter workers ranged from 120 to 26,000 ppm, with a median of 2400 ppm. No significant correlations were found between dust lead concentrations and biological indicators, or between symptom reporting and biological indicators. However, there was an increased frequency of certain objective physical signs, possibly due to lead toxicity, with increased

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blood lead level. The authors also concluded that the high dust lead levels in the workers' homes are most likely due to lead originating in the smelter.

Secondary lead smelters in Memphis, Tennessee and Salt Lake City, Utah were studied by Baker et al. (1979). The former plant extracted lead principally from automotive batteries, producing 11,500 metric tons of lead in the eleven months preceding the measurements. The latter plant used scrap to recover 258 metric tons of lead in the six months preceding the measurements. Airborne concentrations of lead in Tennessee exceeded $200 \mu\text{g}/\text{m}^3$ in some instances, with personal air sampler data ranging from $120 \mu\text{g}/\text{m}^3$ for a battery wrecker to $350 \mu\text{g}/\text{m}^3$ for two yard workers. At the Utah plant, airborne lead levels in the office, lunchroom, and furnace room (furnace not operating) were 60, 90, and $100 \mu\text{g}/\text{m}^3$, respectively. When charging the furnace, the last value increased to $2650 \mu\text{g}/\text{m}^3$. Personal samplers yielded concentrations of $17 \mu\text{g}/\text{m}^3$ for an office worker, $700 \mu\text{g}/\text{m}^3$ for two welders, and $2660 \mu\text{g}/\text{m}^3$ for two furnace workers. Some workers in both plants showed clinical manifestations of lead poisoning; a significant correlation was found between blood lead levels and symptom reporting.

High levels of atmospheric lead are also found in foundries in which molten lead is alloyed with other metals. Berg and Zenz (1967) found in one such operation that average concentrations of lead in various work areas were 280 to $600 \mu\text{g}/\text{m}^3$. These levels were subsequently reduced to 30 to $40 \mu\text{g}/\text{m}^3$ with the installation of forced ventilation systems to exhaust the work area atmosphere to the outside.

7.7.1.3 Mining and Grinding of Lead Ores--Exposures for workers involved in lead mining depend to some extent upon the solubility of the lead from the ores. The lead sulfide (PbS) composing galena is insoluble, and absorption through the lung may be slight. In the stomach, however, some of the lead

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sulfide may be converted to slightly soluble lead chloride, which may then be absorbed in moderate amounts.

Roy (1977) studied exposures during mining and grinding of lead sulfide at a mill in the Missouri lead belt. Primary smelting operations were 2.5 miles from the mill, hence the influence of the smelter was believed to be negligible. Personal samplers worn by mill employees gave results shown in Table 7-16. Note that the total airborne lead levels are much greater than the concentrations of respirable lead, indicating a predominance of coarse material. Solubility tests conducted as part of this study showed that only 0.77-1.39 percent (mean 0.94 percent) of the lead sulfide dissolved in 0.1 N hydrochloric acid, representative of the human stomach. The author also reported generally poor correlation between overall air lead and blood lead levels. It was concluded that because of the large particle sizes and low solubility, a health standard based on total airborne lead concentration would not be appropriate for lead sulfide workers.

7.7.2 Exposures in Welding and Cutting of Metals Containing Lead

When metals that contain lead or are protected with a lead-containing coating are heated in the process of welding or cutting, copious quantities of lead in the respirable size range may be emitted. Under conditions of poor ventilation, electric arc welding of zinc silicate-coated steel (containing 29 mg lead/in² of coating) produced breathing-zone concentrations of lead reaching 15,000 µg/m³, far in excess of 450 µg/m³, the current occupational short-term exposure limit (STEL) in the United States (Pegues, 1960). Under good ventilation conditions, a concentration of 140 µg/m³ was measured (Tabershaw et al., 1943).

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TABLE 7-16. AIRBORNE LEAD CONCENTRATIONS BASED ON PERSONAL SAMPLERS, WORN BY EMPLOYEES AT A LEAD MINING AND GRINDING OPERATION IN THE MISSOURI LEAD BELT

(mg/m³)

Occupation	N	.	High	Low	Mean
Mill operator	6	T	0.30	0.05	0.18
	6	R	0.06	0.01	0.03
Flotation operator	4	T	0.75	0.10	0.32
	4	R	0.04	0.03	0.04
Filter operator	4	T	2.45	0.38	1.33
	4	R	0.24	0.05	0.11
Crusher operator	4	T	0.59	0.02	0.19
	4	R	0.01	0.01	0.01
Sample finisher	2	T	10.00	7.07	8.53
	2	R	0.19	0.16	0.17
Crusher utility	1	T	--	--	0.07
	1	R	--	--	0.07
Shift boss	5	T	0.56	0.11	0.29
	6	R	0.08	0.01	0.05
Equipment operator	1	T	--	--	0.43
	2	R	0.14	0.03	0.08

N denotes number of air samples.

• T - total lead on air ??

R - Respirable lead on air

Source: Roy (1977).

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In a study of salvage workers using oxy-acetylene cutting torches on lead-painted structural steel under conditions of good ventilation, breathing-zone concentrations of lead averaged $1200 \mu\text{g}/\text{m}^3$ and ranged as high as $2400 \mu\text{g}/\text{m}^3$ (Rieke, 1969). Lead poisoning in workers dismantling a painted bridge has been reported by Graben et al. (1978). Fishbein et al. (1978) discuss the exposure of workers dismantling an elevated subway line in New York City, where the lead content of the paint is as great as 40 percent. The authors report that one cubic millimeter of air can contain 0.05 g lead at the source of emission. Similarly, Grandjean and Kon (1981) report elevated lead exposures of welders and other employees in a Baltimore, Maryland shipyard.

7.7.3 Exposures in the Electric Storage Battery Industry

At all stages in battery manufacture except for final assembly and finishing, workers are exposed to high air lead concentrations, particularly lead oxide dust. For example, Boscolo et al. (1978) report air lead concentrations of $16\text{--}100 \mu\text{g}/\text{m}^3$ in a battery factory in Italy, while values up to $13\text{--}15 \mu\text{g}/\text{m}^3$ have been measured by Richter et al. (1979) in an Israeli battery factory. Excessive concentrations, as great as $5400 \mu\text{g}/\text{m}^3$, have been quoted by World Health Organization (1977). The hazard in plate casting, which is a molten-metal operation, is from the spillage of dross, resulting in dusty floors. During oxide mixing, which is probably the most hazardous occupation, ventilation is needed when the mix is loaded with the lead oxide powder, and frequently cleanup is necessary to prevent the accumulation of dust. In the pasting of the plates, whether by hand or by machine, the danger is again from dust which accumulates as the paste dries. The forming and stacking processes are also dusty, and ventilation is needed there. The data cited are sufficiently alarming to suggest that respirators must be worn in most of these operations.

7.7.4 Exposures in the Printing Industry

In a printing establishment, the exposure to lead is probably in direct proportion to the dispersion of lead oxide dust, secondary to the remelt operation. Brandt and Reichenbach (1943) have reported on a study in which melting pots were located in a variety of places where used type was discarded. The pots were maintained at temperatures ranging from 268° to 446°C. The highest air lead concentration recorded was 570 $\mu\text{g}/\text{m}^3$. In 1960, Tsuchiya and Harashima (1965) found airborne lead levels of 30-360 $\mu\text{g}/\text{m}^3$ in several print shops in Japan. More recently, Parikh et al. (1979) reported concentrations approximately 10-30 $\mu\text{g}/\text{m}^3$ in five type foundries in India; these investigators also reported much greater levels, up to 140 $\mu\text{g}/\text{m}^3$, in seven battery reconditioning plants. Greene et al. (1979) discuss the increased risk of cancer to employees in the Government Printing Office. Excess deaths from myeloma were reported for workers who spent considerable amounts of time in the composing room, where lead is the principal contaminant.

7.7.5 Exposures in Alkyl Lead Manufacture

Workers involved in the manufacture of both tetraethyl lead and tetramethyl lead, two alkyl lead compounds, are exposed to both inorganic and alkyl lead. Some exposure also occurs at the petroleum refineries where the two compounds are blended into gasoline, but no exposure data are available on these blenders.

The major potential hazard in the manufacture of tetraethyl lead and tetramethyl lead is from skin absorption, but this is guarded against the use of protective clothing. Lynch et al. (1970) found a correlation between an index of organic plus inorganic lead concentrations in a plant and the rate of lead excretion in the urine of workers. Significant concentrations of organic lead in the urine were found in workers involved with tetramethyl lead and

those involved with tetraethyl lead; lead levels in the tetramethyl lead workers were slightly higher because the reaction between the organic reagent and lead alloy takes place at a somewhat higher temperature and pressure than that employed in tetraethyl lead production.

Cope et al. (1979) used personal air samplers to assess exposures of five alkyl lead workers exposed primarily to tetraethyl lead. Blood and urine levels were measured over a six-week period. Alkyl lead levels ranged from 1.3 to 1248.7 $\mu\text{g}/\text{m}^3$, while inorganic lead varied from 1.3 to 62.6 $\mu\text{g}/\text{m}^3$. There was no significant correlation between airborne lead (either alkyl or inorganic) and blood or urine levels. The authors concluded that biological monitoring, rather than airborne lead monitoring, is a more reliable indicator of potential exposure problems.

7.7.6 Exposures in Other Occupations

In both the rubber products industry and the plastics industry there are potentially high exposures to lead. The potential hazard of the use of lead stearate as a stabilizer in the manufacture of polyvinyl chloride was noted in the 1971 Annual Report of the British Chief Inspector of Factories (1972). The inspector stated that the number of reported cases of lead poisoning in the plastics industry was second only to that in the lead smelting industry. Scarlato et al. (1969) and Maljkovic (1971) have reported on other individual cases of exposure. The source of this problem is the dust that is generated when the lead stearate is milled and mixed with the polyvinyl chloride and the plasticizer.

Sakurai et al. (1974), in a study of bioindicators of lead exposure, found ambient air concentrations averaging 58 $\mu\text{g}/\text{m}^3$ in the lead covering department of a rubber hose manufacturing plant. Unfortunately, no ambient air measurements were taken for other departments or the control group.

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The manufacture of cans with leaded seams may expose workers to elevated environmental lead levels. Bishop (1980) reports airborne lead concentrations of 25-800 $\mu\text{g}/\text{m}^3$ in several can manufacturing plants in the United Kingdom. Between 23 percent and 54 percent of the airborne lead was associated with respirable particles, based on cyclone sampler data.

Firing ranges may be characterized by high airborne lead concentrations, hence instructors who spend considerable amounts of time in such areas may be exposed to lead. For example, Smith (1976) reports airborne lead concentrations of 30-160 $\mu\text{g}/\text{m}^3$ at a firing range in the United Kingdom. Anderson et al. (1977) discuss plumbism in a 17-year-old male employee of a New York City firing range, where airborne lead concentrations as great as 20,899 $\mu\text{g}/\text{m}^3$ were measured during sweeping operations. Another report from the same research group presents time-weighted average exposures of instructors of 45-900 $\mu\text{g}/\text{m}^3$ in three New York City firing ranges (Fishbein et al., 1979).

Removal of leaded paint from walls and other surfaces in old houses may pose a health hazard. Feldman (1978) reports an airborne lead concentration of 510 $\mu\text{g}/\text{m}^3$, after 22 minutes of sanding an outdoor post coated with paint containing 2.5 mg lead/ cm^2 . After only five minutes of sanding an indoor window sill containing 0.8-0.9 mg lead/ cm^2 , the air contained 550 $\mu\text{g}/\text{m}^3$.

Garage mechanics may be exposed to excessive lead concentrations. Clausen and Rastogi (1977) report airborne lead levels of 0.2-35.5 $\mu\text{g}/\text{m}^3$ in ten garages in Denmark; the greatest concentration was measured in a paint workshop. Used motor oils were found to contain 1500-3500 ppm lead, while one brand of gear oil, unused, contained 9280 ppm lead. The authors state that absorption through damaged skin could be an important exposure pathway.

7.7.8 Historical Changes

Numerous published studies have reported increases in environmental levels and human uptake of lead since the advent of lead mining and smelting several thousand years ago. Perhaps the most impressive data are those of Murozumi et al. (1969), showing a 300-fold increase in the lead content of Greenland snow over the past 3000 years. More recent data from this same research group confirm the earlier data, and suggest a ten-fold increase in the lead content of Antarctic snow over the past century (Ng and Patterson, 1981). The authors state that these increases are due primarily to anthropogenic lead emissions, particularly from the combustion of leaded gasoline.

Based on analysis of teeth and bones of Peruvians buried 1600 years ago, Ericson et al. (1979) estimate that the skeletons of present-day American and British adults contain roughly 500 times the amount of lead which would occur naturally, in the absence of widespread anthropogenic lead emissions. Grandjean et al. (1979) and Shapiro et al. (1980) report lead levels in teeth and bones of contemporary populations to be elevated 100-fold over levels in ancient Nubians buried before 750 A.D. On the other hand, Barry and Connolly (1981) report excessive lead concentrations in buried medieval English skeletons; the lead is attributed mainly to absorption from the surrounding soil.

In other studies, modest increases have been reported. Ruhling and Tyler (1968) have measured a fourfold increase in the lead content of Swedish moss samples between 1890 and present. Fosse and Wesenberg (1981) report that lead concentrations in the deciduous teeth of Norwegian children buried at least 180 years ago were roughly 1/3 to 1/4 of the lead content of modern deciduous teeth. The smaller increases could reflect the fact that lead production was already widespread during the earliest dates involved in these studies.

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It is important to note that any of the research reported in this section could have been influenced by artifact contamination during sample collection, handling, and analysis. Patterson and Settle (1976) discuss the extreme difficulty of measuring lead concentrations at low levels; the results of several retrospective studies have been compromised by contamination problems. For example, Jaworowski et al. (1981) measured anomalously large lead concentrations in pre-industrial glacial ice in several remote areas, and used these data to conclude that anthropogenic lead emissions have had a negligible influence on the global cycle of lead. However, the samples were collected with using methods which may have introduced contamination, and were melted in the field with a portable heater which may have resulted in further contamination. Jaworowski's findings, which are in opposition to conclusions of the afore-mentioned studies, are thus likely to be invalid.

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7.8 REFERENCES

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