CHAPTER 7
ENVIRONMENTAL CENCENTRATIONS AND POTENTIAL HUMAN EXPOSURE
TABLE OF CONTENTS

needs to be more quantitative, precise, & documented to database for decision

N33758

LIST OF TABLES

List of Tables (continued)

LIST OF FIGURES

9 Figures / 87 pages

1 Figure / 10 pages

7.1 INTRODUCTION

Lead-bearing aerosol particles emitted from the sources described in Chapter 5, arrive at their point of deposition by mechanisms described in Chapter 6. Even though the distance traveled may be a few meters or several thousand kilometers, the important point is that until the particle deposits on a surface, it will have no effect on plants, animals, or humans. If this surface is alveolar lung tissue, the lead may be incorporated directly. For other surfaces, the lead must be ingested (animals) or absorbed (plants) before any effect is possible. Therefore, it is necessary to understand the ambient concentrations of lead in air normally breathed by humans, and in other environmental components such as soil, dust, and vegetation likely to be directly or indirectly ingested by adults and children.

This chapter is divided into two parts. In the first part, the range of ambient concentrations normally encountered by humans in typical working, living, and playing environments are emphasized. Air concentrations represent the bulk of this discussion because the most information is available on this topic. Lead in soil, dust, and vegetation are one step closer to ingestion but data are limited since there are no organized sampling networks for these media.

In the second part of the chapter, ambient concentrations are discussed in the context of potential human exposure. It is at this point that the effects of atmospheric lead are combined with lead from other sources, such as paint, industrial milling processes, and plasticizers to produce a composite picture of human exposure based on diet, the living and recreational environment, and a few sensitive occupational situations.

7.2 AMBIENT CONCENTRATIONS

Quantifying human exposure to lead requires an understanding of ambient lead levels in environmental media. Of particular importance are lead concentrations in air, soil, dust, and vegetation. The sections which follow discuss environmental lead concentrations in each of these media in the context of how these concentration increase or decrease human exposure.

7.2.1 Ambient Air

Ambient airborne lead concentrations may influence human exposure through direct inhalation of lead-containing particles, and through ingestion of lead which has been deposited from the air onto surfaces. Although a plethora of data on airborne lead is now available, our understanding of human exposure is far from complete: most ambient measurements were not taken in conjunction

with epidemiological studies of the 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.

Three categories of studies are included in this section. First, data on total airborne lead concentrations are summarized. Included in this category are data from government sampling networks as well as the results of independent investigators. The second category involves airborne lead size distribution measurements. Finally, vertical gradients of lead in the atmosphere, particularly near roadways, are discussed.

7.2.1.1 Total Airborne Lead Concentrations--A thorough understanding of human exposure to airborne lead requires detailed knowledge of spatial and temporal variations in ambient concentrations. The wide range of concentrations is apparent from Table 7-1, which summarizes data obtained from numerous independent measurements. Concentrations vary from less than 0.001 $\mu g/m^3$ in remote areas to over 10 $\mu g/m^3$ near sources. Many of the remote areas are far from human habitation and therefore do not reflect human exposure. However, a few of the regions characterized by small lead concentrations are populated by individuals with primitive lifestyles; these data provide baseline airborne lead data to which modern American lead exposures can be compared. Examples include some of the data from South America and the data for Nepal.

Comparing urban, rural, and remote airborne lead concentrations in Table 7-1 suggests that human exposure to lead has increased over time, as the use of lead in inhabited areas has increased. This is consistent with published results of retrospective human exposure studies. For example, Ericson et al. (1979) have analyzed the teeth and bones of Peruvians buried 1600 years ago. Based on their data, they 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.

It is important to note that the remote area concentrations reported in Table 7-1 do not necessarily reflect natural, preindustrial lead. Murozumi et

TABLE 7-1. ATMOSPHERIC LEAD IN URBAN, RURAL, AND REMOTE AREAS OF THE WORLD. Adapted from Nriagu (1978).

Time averaging period

Location	Sampling Period	Lead conc. ($\mu g/m^3$)	Reference
Urban			
Berlin	1966-67	3.8	Blokker, 1972
Vienna	1970	2.9	Hartl and Resch, 1973
Zurich	1970	3.8	Högger, 1973
Liege	1972	2.7	Rondia and DeGraeve, 1973
Turin	1974	5.0	Magi et al., 1975
Rome	1972-73	4.5	Colacino and Lavagnini, 1974
Paris	1964	4.6	Blokker, 1972
Miami	1974	1.3	HASL, 1975
New York	1974	1.1	HASL, 1975
Boston	1972	4.5	O'Brien et al., 1975
St. Louis	1973	1.1	Tanner et al., 1974
Cincinnati	1970	1.8	Lee et al., 1972
Chicago	1970	1.1	Lee et al., 1972
Salt Lake City	1974	0.89	HASL, 1975
Denver	1970	0.8	Lee et al., 1972
Los Angeles	1968-69	3.6	Chow, 1973
Rio de Janeiro	1972-73	0.8	Branquinho and Robinson, 1976
Ottowa	1975	1.3	NAPS, 1975
Toronto	1975	1.3	NAPS, 1975
Montreal	1975	2.0	NAPS, 1975
Rural			
United Kingdom	1972	0.13	Cawse, 1974
New York Bight	1974	0.13	Duce et al., 1975
Framingham, MA	1972	0.9	O'Brien et al., 1975
Chadron, NE	1973-74	0.045	Struempler, 1975
Remote			
White Mtn., CA	1969-70	.008	Chow et al., 1972
Antarctica	1971	.0004	Duce, 1972
South Pole	1970	.00063	Zoller et al., 1974
Thule, Greenland	1965	.0005	Murozumi et al., 1969
Thule, Greenland	1978-79	.008	Heidam, 1971
Prins Christian-sund, Greenland	1978-79	.018	Heidam, 1981
Dye 3, Greenland	1979	.00015	Davidson et al., 1981c
High Sierra, CA	1976-77	.021	Elias and Davidson, 1980
Olympic Nat. Park, WA	1980	.0022	Davidson et al., 1982
Enewetak, Pacific Ocean	1979	.00017	Settle and Patterson, 1982
Kumjung, Nepal	1979	.00086	Davidson et al., 1981b
Bermuda	1973-75	.0041	Duce et al., 1976
Spitsbergen	1973-74	.0058	Larssen, 1977

al. (1969) and Ng and Patterson (1981) have measured a 200-fold increase in the lead content of Greenland snow over the past 3000 years. The authors state that this lead originates in populated mid latitude regions, and is transported over thousands of kilometers through the atmosphere to the Arctic. It is apparent that most of the concentrations in Table 7-1, including values for remote areas, may be influenced by anthropogenic lead emissions.

All of the data in Table 7-1 refer to the ambient atmosphere. Because people spend much of their time indoors, these data cannot be used to obtain an accurate indication of exposure to airborne lead. Table 7-2 summarizes the results of several indoor/outdoor airborne lead studies. In nearly all cases, the indoor concentration is substantially smaller than the corresponding value outdoors; the only indoor/outdoor ratio exceeding unity is for a high-rise apartment building, where air taken in near street level is rapidly distributed through the building air circulation system. Some of the studies of Table 7-2 include published data showing smaller indoor/outdoor ratios during the winter, when windows and doors are tightly closed. Overall, it has been suggested that indoor/outdoor ratios of 0.6 to 0.8 are typical for airborne lead in non-air conditioned houses. Ratios in air conditioned houses are expected to be closer to 0.3 to 0.5 (Yocum, 1982).

The avaliable data imply that virtually all airborne lead found indoors is associated with material transported in from the outside. Because of the complexity of factors affecting infiltration of air into buildings, however, it is difficult to accurately predict indoor lead concentrations based on outdoor levels.

Detailed knowledge of indoor and outdoor airborne lead concentrations may still be insufficient to assess human exposure to airborne lead. The study of Tosteson et al. (1981) in Table 7-2 included measurement of airborne lead concentrations using personal exposure monitors, carried by individuals going about their day-to-day activities. In contrast to the lead concentrations of 0.092 and 0.12 $\mu g/m^3$ given in the table, the average personal exposure was 0.16 $\mu g/m^3$. The authors suggest the inadequacy of using fixed monitors at either indoor or outdoor locations to assess exposure.

A problem with the studies referenced in Tables 7-1 and 7-2 is that quality control and interlaboratory comparability are generally unspecified. In contrast, there are two principal airborne lead data bases which include

Include AZAR et al ?

TABLE 7-2. COMPARISON OF INDOOR AND OUTDOOR AIRBORNE LEAD CONCENTRATIONS

Type of Building	Airborne Lead Concentration µg/m³ Indoor	Outdoor	Indoor/Outdoor Ratio	Location	Reference
Library	1.12	2.44	0.46	Hartford, CT	Yocum et al. (1971)
City Hall	1.31	1.87	0.70	"	"
Office Building 1	0.73	1.44	0.51	"	"
Office Building 2	0.55	1.09	0.51	"	"
House 1	1.37	2.48	0.55	"	"
House 2	0.94	1.34	0.70	"	"
Apartment Building 1					
Second Floor	1.46	2.67	0.55	New York, NY	General Electric
Roof	1.50	1.38	1.09	"	Company (1972)
Apartment Building 2					
Third Floor	--	1.21	--	"	"
Eleventh Floor	1.68	--	--	"	"
Eighteenth Floor	1.86	--	--	"	"
Roof	--	1.42	--	"	"
New Air Conditioned Apartment	0.12-0.40	0.13-0.50	0.82	New York, NY	Halpern (1978)
Older Non-Air Conditioned Apartment	0.14-0.51	0.17-0.64	0.87	"	"
Air Conditioned Public Building	0.15-0.79	0.33-1.18	0.63	"	"
Non-Air Conditioned Storeroom in Public Building	0.45-0.98	0.38-1.05	0.81	"	"
Houses	--	--	0.53	Pittsburgh, PA	Cohen and Cohen (1980)
University Buildings	--	--	0.28	"	"
Public Schools	--	--	0.28	"	"
Store	--	--	0.31	"	"
Commercial Office	--	--	0.27	"	"

Table 7-2 (continued)

Type of Building	Airborne Lead Concentration µg/m³ Indoor	Outdoor	Indoor/Outdoor Ratio	Location	Reference
Houses	0.092	0.12	0.74	Topeka, KS	Tosteson et al. (1981)
Houses with gas stoves	--	--	0.65	Boston, MA	Moschandreas et al. (1981)
Houses with electric stoves	--	--	0.68	"	"
Office Buildings	--	--	0.42	"	"
House 1					
Before energy conservation retrofit	0.039	0.070	0.56	Medford, OR	Berk et al. (1981)
After energy conservation retrofit	0.037	0.084	0.44	Medford, OR	Berk et al. (1981)
House 2					
Before energy conservation retrofit	0.035	0.045	0.78b	"	"
After energy conservation retrofit	0.038	0.112	0.34	"	"

measurements subjected to quality assurance procedures. The first is EPA's National Air Sampling Network (NASN), providing comprehensive nationwide data on long-term trends. 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 airborne standard for lead (1.5 $\mu g/m^3$ averaged over a calendar quarter) promulgated in 1978.

EPA Nationwide Sampling Network. Tables 7-3 and 7-4, 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 $\mu g/m^3$ and the majority of the nonurban sites reported annual averages below 0.2 $\mu g/m^3$. 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.

Tables 7-5 and 7-6 provide cumulative frequency distributions of all quarterly lead concentrations for urban and nonurban NASN stations, respectively (1st quarter = Jan-Mar, etc). 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 7-5) 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 nonurban locations (Table 7-6) 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.

TABLE 7-3. 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, $\mu g/m^3$					
	<0.5	0.5-0.9	1.0-1.9	2.0-3.9	≥4.0	Total
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

TABLE 7-3 (continued).

Year	Concentration interval, $\mu g/m^3$					
	<0.5	0.5-0.9	1.0-1.9	2.0-3.9	4.0-5.3	Total
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

TABLE 7-4. 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, $\mu g/m^3$				
	<0.03	0.03-0.096	0.10-0.19	0.20-0.45	Total
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

TABLE 7-4 (continued).

Year	Concentration interval, $\mu g/m^3$				
	<0.03	0.03-0.096	0.10-0.19	0.20-0.45	Total
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

TABLE 7-5. 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)
($\mu g/m^3$)

Year	No.	Min.	Percentile							Max.	Arithmetic		Geometric	
			10	30	50	70	90	95	99		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

add 1981-82 data

TABLE 7-6. 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)
($\mu g/m^3$)

Year	No.	Min.	Percentile							Max.	Arithmetic		
			10	30	50	70	90	95	99		Mean	Std. dev.	M
1970	124	LD	LD	LD	LD	LD	0.267	0.383	0.628	1.471	--	--	
1971	85	LD	LD	LD	LD	LD	0.127	0.204	0.783	1.134	--	--	
1972	137	LD	LD	LD	0.107	0.166	0.294	0.392	0.950	1.048	0.139	0.169	
1973	100	LD	LD	LD	LD	0.132	0.233	0.392	0.698	0.939	--	--	
1974	79	LD	LD	0.053	0.087	0.141	0.221	0.317	0.496	0.534	0.111	0.111	
1975	98	LD	LD	LD	LD	0.144	0.255	0.311	0.431	0.649	--	--	
1976	98	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	
1978	20	0.002	0.007	0.04	0.06	0.09	0.24	0.33	0.33	0.33	0.08	0.10	
1979	16	LD	0.02	0.02	0.10	0.14	0.21	0.27	0.32	0.11	0.11	0.13	
1980	12	LD	0.01	0.005	0.03	0.05	0.11	0.13	0.13	0.13	0.04	0.06	

add 1981-82 data

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. The controlling influence, apparently, is the seasonal pattern of lower meteorological ventilation capacity in winter, higher capacity in summer.

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 7-7) indicate that the decrease is a real and pervasive phenomenon.

State and Local Agency Data. Table 7-8 lists stations operated by state and local agencies where one or more quarterly averages exceeded the current standard of 1.5 $\mu g/m^3$ 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 such as smelters or refineries, and near routes of high traffic density. Both situations are represented in Table 7-8, e.g., the Idaho data reflect predominantly stationary source emissions, the Washington, D.C. data reflect predominantly vehicular emissions. Table 7-9 shows the maximum quarter lead concentrations for Standard Metropolitan Statistical Areas greater than 1 million.

Table 7-10 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

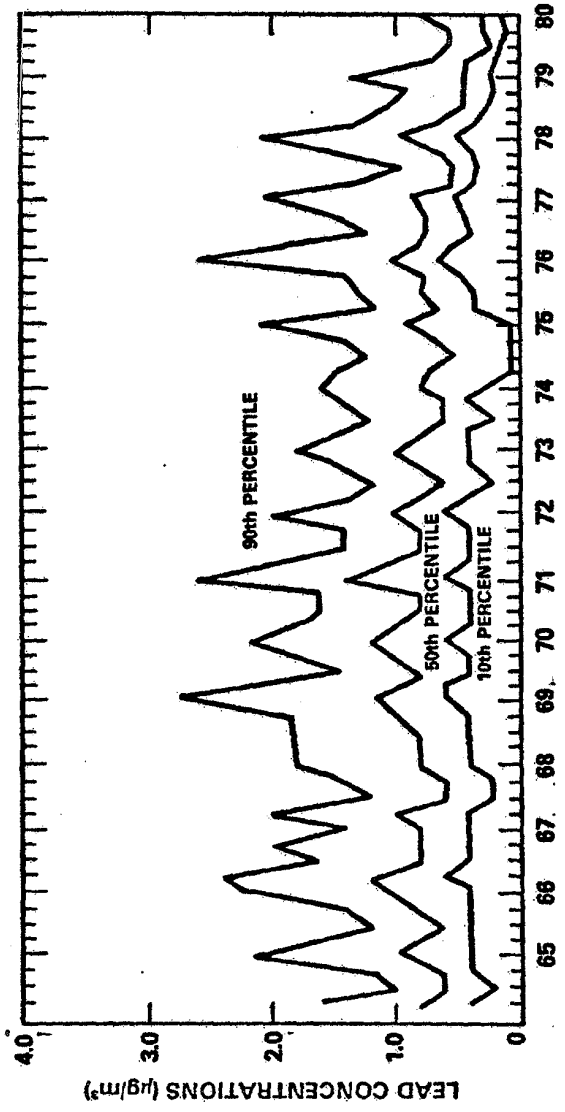

Figure 7-1. Seasonal patterns and trends in quarterly average urban lead concentrations.

TABLE 7-7. 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
Worchester, 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

Table 7-8. Locations Reporting Valid Quarterly Lead Concentrations Greater than 1.0 μg/m^3 (The Current Ambient Air Standard) in 1979 and 1980.* (OAQPS, 1982)

STATE	CITY OR COUNTY	YEAR	MAX. QTR. μg/m^3	NO. OF QUARTERS >1.0 μg/m^3
Alabama	Troy (003)	1979	2.78	2
	"	1980	1.13	2
Arizona	Glendale (001)	1979	1.06	1
	Phoenix (002A)	1979	1.54	1
	"	1980	1.29	2
	Phoenix (002G)	1979	2.59	2
	"	1980	1.49	2
	Phoenix (004)	1979	1.48	2
	Phoenix (013)	1979	1.55	2
	"	1980	1.06	1
	Scottsdale (003)	1979	1.41	2
	"	1980	1.13	1
	Tucson (009)	1979	1.18	1
	Nogales (004)	1980	1.10	1
California	Los Angeles (001)	1979	1.51	1
	Anaheim (001)	1979	1.11	1
Colorado	Adams Co. (001)	1979	1.77	2
	Arapahoe Co. (001)	1979	1.10	1
	Arvada (003)	1979	1.60	1
	Brighton (001)	1979	1.17	1
	Colorado Springs (004)	1979	1.37	1
	Denver (001)	1979	1.70	2
	Denver (002)	1979	3.47	4
	"	1980	1.53	2
	Denver (003)	1979	2.13	3
	"	1980	1.03	1
	Denver (009)	1979	1.57	1
	"	1980	1.23	2
	Denver (010)	1979	1.67	2
	Denver (012)	1979	1.67	2
	"	1980	1.10	1
	Englewood (001)	1979	1.80	1
	Garfield Co. (001)	1979	1.20	1
	Grand Junction (010)	1979	1.53	2
	"	1980	1.27	1
	Longmont (001)	1979	1.07	2
	Pueblo (001)	1979	1.03	1
	Pueblo (003)	1979	1.03	1
	Routt Co. (003)	1979	1.33	1
Connecticut	New Haven (123)	1979	1.57	3
	Waterbury (123)	1979	1.41	3
Delaware	Wilmington (002)	1979	1.21	2

Table 7-8 (continued)

STATE	CITY OR COUNTY	YEAR	MAX. QTR. $\mu g/m^3$	NO. OF QUARTERS $>1.0\ \mu g/m^3$
Dist. of Col.	Washington (005)	1979	1.49	1
	Washington (007)	1979	1.89	4
	Washington (008)	1979	1.90	1
	Washington (011)	1979	1.44	2
	Washington (015)	1979	1.06	2
	Washington (017)	1979	1.45	1
Florida	Dade Co. (020)	1979	1.16	1
	Miami (016)	1979	1.46	3
	"	1980	1.10	2
	Perrine (002)	1979	1.01	1
	Hillsborough Co. (082)	1979	1.31	2
	"	1980	1.09	1
	Tampa (043)	1979	1.60	3
	"	1980	1.07	1
Idaho	Boise City (003)	1980	1.01	1
	Kellogg (004)	1979	9.02	4
	"	1980	6.88	2
	Kellogg (006)	1979	8.25	4
	"	1980	8.72	4
	Shoshone Co. (015)	1979	1.21	2
	Shoshone Co. (016)	1979	2.27	1
	"	1980	1.02	1
	Shoshone Co. (017)	1979	4.57	4
	"	1980	3.33	3
	Shoshone Co. (020)	1979	4.11	2
	"	1980	2.15	2
	Shoshone Co. (021)	1979	13.54	4
	"	1980	13.67	4
	Shoshone Co. (027)	1979	10.81	4
	"	1980	7.18	3
Illinois	Chicago (022)	1980	1.02	1
	Chicago (030)	1980	1.06	1
	Chicago (005)	1979	1.05	1
	Chicago (036)	1979	1.02	1
	Chicago (037)	1979	1.14	1
	Cicero (001)	1979	1.00	1
	Elgin (004)	1980	1.95	1
	Granite City (007)	1979	1.04	1
	Granite City (009)	1979	1.15	4
	Granite City (010)	1979	3.17	4
	"	1980	2.97	3
	"	1981	7.27	4
	Granite City (011)	1979	1.33	4
	"	1980	1.43	1
	"	1981	1.13	1

Table 7-8 (continued)

STATE	CITY OR COUNTY	YEAR	MAX. QTR. $\mu g/m^3$	NO. OF QUARTERS >1.0 $\mu g/m^3$
Indiana	Jeffersonville (001)	1979	1.38	3
	East Chicago (001)	1979	2.19	2
	East Chicago (003)	1979	1.42	2
	East Chicago (004)	1979	1.67	1
	East Chicago (006)	1979	1.34	2
	"	1980	1.04	1
	Hammond (004)	1979	1.18	2
	Hammond (006)	1979	1.46	1
	Indianapolis (030)	1979	1.16	1
Iowa	Des Moines (051)	1979	1.30	1
Kentucky	Buechel (001)	1980	1.41	1
	Covington (001)	1979	1.12	2
	Covington (008)	1979	1.16	1
	Greenup Co. (003)	1979	1.42	1
	Jefferson Co. (029)	1979	1.05	1
	"	1980	1.78	1
	Louisville (004)	1979	1.01	1
	"	1980	2.41	1
	Louisville (009)	1980	1.75	1
	Louisville (019)	1980	1.59	1
	Louisville (020)	1980	2.52	1
	Louisville (021)	1979	1.29	1
	"	1980	1.42	1
	Louisville (028)	1979	1.06	1
	Newport (002)	1979	1.06	1
	Okolona (001)	1979	1.51	1
	"	1980	2.31	2
	Paducha (004)	1979	1.41	1
	Paducha (020)	1979	1.22	1
	St. Matthews (004)	1979	1.20	1
	"	1980	1.83	1
	Shively (002)	1979	1.56	1
Louisiana	Baton Rouge (002)	1979	1.57	1
Maine	Portland (009)	1979	1.02	2
Maryland	Anne Arundel Co. (001)	1979	1.27	1
	Anne Arundel Co. (003)	1979	1.45	2
	Baltimore (001)	1979	1.06	2
	Baltimore (006)	1979	1.09	1
	Baltimore (008)	1979	1.24	1
	Baltimore (009)	1979	1.08	1
	Baltimore (018)	1979	1.12	2
	Cheverly (004)	1979	1.51	4
	Essex (001)	1979	1.15	2
	Hyattsville	1979	1.18	2

Table 7-8 (continued)

STATE	CITY OR COUNTY	YEAR	MAX. QTR. $\mu g/m^3$	NO. OF QUARTERS >1.0 $\mu g/m^3$
Massachusetts	Springfield (002)	1979	1.68	1
	"	1980	1.04	1
	Boston (012)	1979	1.01	1
Minnesota	Minneapolis (027)	1979	2.44	1
	Minneapolis (055)	1980	2.41	3
	Richfield (004)	1979	1.95	4
	"	1980	1.18	2
	St. Louis Park (007)	1979	2.87	2
	"	1980	3.04	4
	St. Paul (031)	1979	1.04	1
	St. Paul (038)	1979	1.36	1
	"	1980	1.82	3
Montana	Lewis & Clark Co. (002)	1979	4.19	4
	"	1980	2.75	4
	Lewis & Clark Co. (008)	1980	1.19	1
Nebraska	Omaha (034)	1979	1.08	1
Nevada	Las Vegas (001)	1979	1.15	1
New Jersey	Newark (001)	1979	1.17	1
	Perth Amboy (001)	1979	1.08	1
	Paterson (005)	1979	1.42	1
	Elizabeth (002)	1979	1.16	1
New York	Yonkers (001)	1979	1.08	1
Ohio	Cincinnati (001)	1979	1.15	1
Pennsylvania	Laureldale (717)	1979	3.30	4
	"	1980	1.86	2
	Reading (712)	1979	1.11	1
	E. Conemaugh (804)	1979	1.28	3
	Throop (019)	1979	1.13	1
	Lancaster City (315)	1979	1.18	1
	New Castle (015)	1979	1.01	1
	Montgomery Co. (103)	1979	1.23	1
	Pottstown (101)	1979	1.16	2
	Philadelphia (026)	1979	1.21	3
	Philadelphia (028)	1979	2.71	4
	"	1980	1.26	3
	Philadelphia (031)	1979	1.29	2
	Philadelphia (038)	1979	1.06	1
Puerto Rico	Guaynabo Co. (001)	1979	1.60	2
	"	1980	1.06	1
	Ponce (002)	1979	1.08	1
	San Juan Co. (003)	1979	3.59	4
Rhode Island	E. Providence (008)	1979	1.10	2
	Providence (007)	1979	1.92	4
	"	1980	1.16	2
	Providence (015)	1979	1.34	1
South Carolina	Greenville (001)	1979	1.38	2

Table 7-8 (continued)

STATE	CITY OR COUNTY	YEAR	MAX. QTR. $\mu g/m^3$	NO. OF QUARTERS >1.0 $\mu g/m^3$
Tennessee	Nashville-Davidson (006)	1979	1.05	1
Texas	San Antonio (034)	1979	1.23	1
	Dallas (018)	1979	1.59	1
	Dallas (029)	1979	1.07	1
	Dallas (035)	1979	1.12	3
	Dallas (046)	1979	1.22	1
	Dallas (049)	1979	1.01	1
	Dallas (050)	1979	1.13	2
	El Paso (002A)	1980	1.90	1
	"	1980	2.12	1
	El Paso (002F)	1979	1.90	1
	El Paso (002G)	1979	2.60	4
	El Paso (018)	1979	1.91	2
	El Paso (021)	1979	1.02	1
	El Paso (022)	1979	1.84	2
	El Paso (023)	1979	2.12	2
	El Paso (027)	1979	2.15	2
	"	1980	1.74	2
	El Paso (028)	1980	1.16	1
	El Paso (030)	1979	1.02	1
	El Paso (031)	1979	2.47	1
	El Paso (033)	1979	1.97	1
	Houston (001)	1979	1.35	2
	Houston (002)	1979	1.39	2
	Houston (037)	1979	1.26	1
	Houston (049)	1979	1.13	3
	Ft. Worth (003)	1979	1.14	2
Washington	Seattle (057)	1979	1.36	1
	Tacoma (004)	1979	1.06	1
West Virginia	Charleston (001)	1979	1.09	1

*As of November 1982

Need more Graphs

TABLE 7-9. MAXIMUM QUARTER LEAD CONCENTRATIONS REPORTED FOR SMSA's ≥ 1 MILLION POPULATION, 1979-1981

Anaheim-Santa Ana-Garden Grove, CA	1979	1.11
	1980	0.47
	1981	0.93
Atlanta, GA	1980	0.51
	1981	0.39
Baltimore, MD	1979	3.41
	1980	1.11
	1981	0.61
Boston, MA	1979	1.01
	1980	0.57
Buffalo, NY	1979	0.47
	1980	0.41
	1981	0.38
Chicago, IL	1979	1.15
	1980	1.95
	1981	0.89
Cincinnati, OH-KY-IN	1979	1.16
	1980	0.85
	1981	0.37
Cleveland, OH	1979	0.38
	1980	0.34
Columbus, OH	1979	0.43
	1980	0.35
	1981	0.34
Dallas-Fort Worth, TX	1979	1.59
	1980	0.67
Denver-Boulder, CO	1979	3.47
	1980	1.53
	1981	0.97
Fort Lauderdale-Hollywood, FL	1979	0.33
	1980	0.36
	1981	0.23
Houston, TX	1979	1.39
	1980	0.64
Indianapolis, IN	1979	1.16
	1980	0.63
	1981	0.42
Kansas City, MO-KS	1979	0.82
	1980	0.38
	1981	0.19
Los Angeles-Long Beach, CA	1979	1.51
	1980	0.68
	1981	1.43
Miami, FL	1979	1.46
	1980	1.10
	1981	0.75

Table 7-9 (continued)

Milwaukee, WI	1979	0.72
	1980	0.49
	1981	0.31
Minneapolis-St. Paul, MN-WI	1979	2.87
	1980	3.04
	1981	3.11
Newark, NJ	1979	1.17
	1980	0.53
New Orleans, LA	1979	0.70
	1980	0.35
	1981	0.25
New York, NY-NJ	1979	1.08
	1980	0.47
Philadelphia, PA-NJ	1979	2.71
	1980	1.26
	1981	1.30
Phoenix, AZ	1979	2.59
	1980	1.49
	1981	1.39
Pittsburgh, PA	1979	0.82
	1980	0.44
	1981	0.41
Portland, OR-WA	1979	0.60
	1980	0.41
	1981	0.29
Riverside-San Bernardino-Ontario, CA	1979	0.91
	1980	0.54
	1981	0.57
Sacramento, CA	1979	0.69
	1980	0.18
	1981	0.62
San Antonio, TX	1979	1.23
	1980	0.79
San Diego, CA	1979	0.91
	1980	0.83
	1981	0.60
San Francisco-Oakland, CA	1979	0.42
	1980	0.13
	1981	0.28
San Jose, CA	1979	0.92
	1980	0.86
	1981	0.61
San Juan, PR	1979	3.59
	1980	1.06
	1981	1.02
Seattle-Everett, WA	1979	1.36
	1980	0.86
	1981	0.52

Table 7-9 (continued)

St. Louis, MO-IL	1979	3.17
	1980	2.97
	1981	7.27
Tampa-St. Petersburg, FL	1979	1.60
	1980	1.09
	1981	0.68
Washington, DC-MD-VA	1979	1.90
	1980	0.69
	1981	0.48

Source: OAQPS (1982)

TABLE 7-10. SUMMARY OF MAXIMUM QUARTERLY LEAD AVERAGE BY SELECTED CONCENTRATIONS ACCORDING TO SITE-TYPE, 1979-81.*

CONCENTRATION RANGES
($\mu g/m^3$)

Site-Type	≤.5	>.5 ≤1.0	>1.0 ≤1.5	>1.5 ≤2.0	>2.0	Total No. of Site-Years
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:

background. The stationary source-oriented sites clearly dominate the concentrations over 2.0 $\mu g/m^3$; 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.

New guidelines for siting ambient air lead monitors went into effect in July, 1981 (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 7-11 summarize TSP and particulate lead concentrations in samples collected in Cincinnati, Ohio, on 21 consecutive days in April and May, 1980, adjacent to a

TABLE 7-11. SUMMARY OF LEAD CONCENTRATIONS NEAR A ROADWAY* IN CINCINNATI, OHIO (Pedco, 1981)

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

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

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

	Distance from Roadway		
Elevation	2.8 m	7.1 m	21.4 m
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.

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 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.

7.2.1.2 Airborne Particle Size Distributions--The effects of airborne lead on human health and welfare depend upon the sizes of the lead-containing particles. As discussed in Chapter 6, large particles are removed relatively quickly from the atmosphere by dry and wet deposition processes. Such particles are also unlikely to reach the lower human repiratory system when inhaled, because of entrapment by the cilia and walls of the upper respiratory tract. Particles with diameter smaller than a few micrometers tend to remain airborne for long time periods, and penetrate into the lower lung.

Figure 7-2 summarizes airborne lead size data from the literature. Minimum and maximum aerodynamic particle diameters of 0.05 μm and 25 μm, respectively, have been assumed unless otherwise specified in the original reference. Note that most of the airborne lead mass is associated with small particles. There is also a distinct peak in the upper end of many of the distributions. Two separate categories of sources are responsible: the small particles result from nucleation of vapor phase lead emissions (predominantly automotive), while the larger particles represent primary aerosol emitted from combustion or from mechanical processes (such as soil erosion, abrasion of metal products, resuspension of automobile tailpipe deposits, and flaking paint).

Table 7-13 presents information associated with each of the distributions of Figure 7-2. The first six distributions were obtained by an EPA cascade impactor network established in several cities during the calendar year 1970 (Lee et al., 1972). These distributions represent the most extensive size distribution data base available. However, the impactors were operated at excessive air flow rates most likely resulting in particle bounceoff, biasing the data toward smaller particles (Dzubay et al., 1976). Many of the later distributions, although obtained by independent investigators with unknown quality control, were collected using techniques which minimize particle bounceoff and hence may be more reliable.

It is important to note that a few of the distributions were obtained without backup filters to capture the smallest particles. These distributions

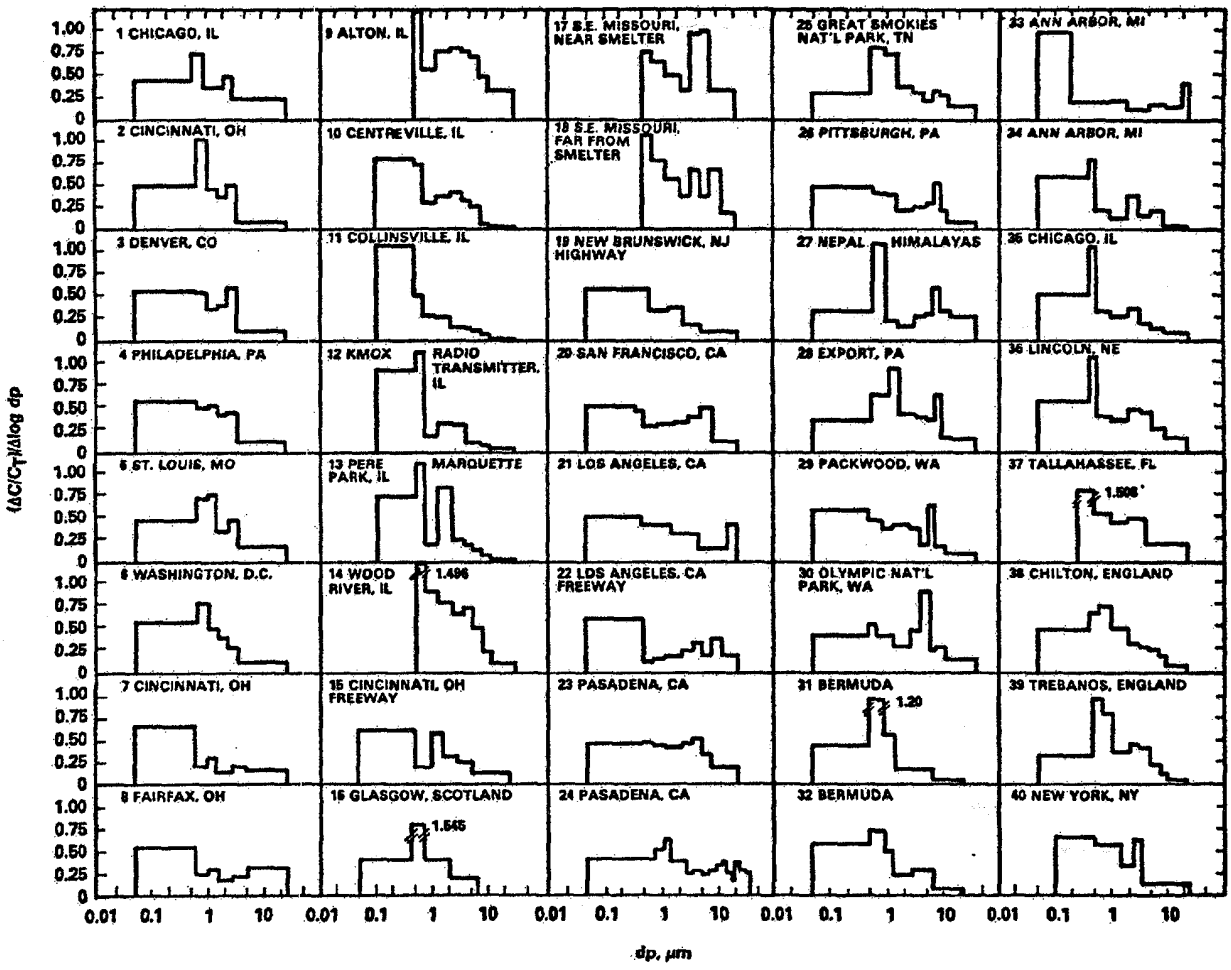

Figure 7-2. Airborne mass size distributions for lead taken from the literature. ΔC represents the airborne lead concentration in each size range, C_T is the total airborne lead concentration in all size ranges, and d_p is the aerodynamic particle diameter. A density of 6 g/cm³ for lead-containing particles 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.

TABLE 7-12. 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 $\mu g/m^3$	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	3.2	0.68
2	Lee et al., 1972	Mar. - Dec. 1970 Same averaging as Graph 1, total of 18 sampling periods	Cincinnati, Ohio	Modified Andersen impactor with backup filter	1.8	0.48
3	Lee et al., 1972	Jan. - Dec. 1970 Same averaging as Graph 1, total of 21 sampling periods	Denver, Colorado	Modified Andersen impactor with backup filter	1.8	0.50
4	Lee et al., 1972	Mar. - Dec. 1970 Same averaging as Graph 1, total of 20 sampling periods	Philadelphia, Pennsylvania	Modified Andersen impactor with backup filter	1.6	0.47
5	Lee et al., 1972	Jan. - Dec. 1970 Same averaging as Graph 1, total of 22 sampling periods	St. Louis, Missouri	Modified Andersen impactor with backup filter	1.8	0.69
6	Lee et al., 1972	Jan. - Dec. 1970 Same averaging as Graph 1, total of 23 sampling periods	Washington, D.C.	Modified Andersen impactor with backup filter	1.3	0.42

TABLE 7-12 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	CT $\mu g/m^3$	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	2.8	0.29
8	Lee et al., 1968	February 1967 Average of 3 runs 4 days each	Fairfax, Ohio, suburb of Cincinnati	Andersen impactor with backup filter, 1.2m above the ground	0.69	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	0.24	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	0.62	0.41
11	Peden, 1976	Summer 1973 Average of 2 runs, average 5 days each	Collinsville, Illinois industrial area near St. Louis	Andersen impactor with backup filter	0.67	0.24
12	Peden, 1976	Summer 1973 Average of 2 runs, average 6 days each	KMOX radio transmitter, Illinois, industrial area near St. Louis	Andersen impactor with backup filter	0.60	0.31
13	Peden, 1976	Summer 1972 Average of 9 runs, average 9 days each	Pere Marquette State Park, Illinois, upwind of St. Louis	Andersen impactor with backup filter	0.15	0.51
14	Peden, 1976	Summer 1975 Average of 4 runs, average 8 days each	Wood River, Illinois, industrial area near St. Louis	Andersen impactor, no backup filter	0.27	1.8

TABLE 7-12 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C_T $\mu g/m^3$	Approx. MMD μm
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	7.8* 1.7 1.1	0.32
16	McDonald and Duncan, 1979	June 1975 One run of 15 days	Glasgow, Scotland	Casella impactor with backup filter, 30m above the ground	0.53	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	1.0	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	Andersen impactor, no backup filter, 1.7m above the ground	0.11	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	4.5 2.2 1.5	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	0.84	0.49
21	Lundgren, 1970	November 1968 Average of 10 runs, 16 hours each	Riverside, California	Lundgren impactor	0.59	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	14.0	0.32

TABLE 7-12 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C_T $\mu g/m^3$	Approx. MMD μm
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	3.5	0.72
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	1.2	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	0.014	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	0.60	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	0.0014	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	0.111	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	0.016	0.40

TABLE 7-12 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C_T $\mu g/m^3$	Approx. MMD μm
30	Goold and Davidson, 1982	July-Aug. 1980 One run of 92 hours	Hurricane Ridge Olympic National Park elev. 1600m	Modified Andersen impactor with backup filter, 1.5m above the ground	0.0024	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	0.0085	0.57
32	Duce et al., 1976	July 1975 One run of 79 hours	Southeast coast of Bermuda	Sierra high-volume impactor with backup filter, 20m above the ground	0.0041	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	1.8	0.16
34	Gillette and Winchester, 1972	Oct. 1968 Average of 15 runs, 24 hours each	Ann Arbor, Michigan	Andersen impactor with backup filter	0.82	0.28
35	Gillette and Winchester, 1972	May-Sep. 1968 Average of 10 runs, 8 hours each	Chicago, Illinois	Andersen impactor with backup filter	1.9	0.39
36	Gillette and Winchester, 1972	Oct. 1968 Average of 3 runs, 24 hours each	Lincoln, Nebraska	Andersen impactor with backup filter	0.14	0.42
37	Johansson et al., 1976	June-July 1973 Average of 15 runs, average 50 hr each	2 sites in Tallahassee, Florida	Delron Battelle-type impactor, no backup filter, on building roofs	0.24	0.62

TABLE 7-12 (continued).

Graph No	Reference	Dates of Sampling	Location of Sampling	Type of Sampler	C_T µg/m^3	Approx. MMD µm
38	Cawse et al., 1974	July-Dec. 1973	Chilton, England	Andersen impactor with backup filter, 1.5m above the ground	0.16	0.57
39	Pattenden et al., 1974	May-Aug. 1973 Average of 4 runs, 1 month each	Trebanos, England	Andersen impactor with backup filter, 1.5m above the ground	0.23	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	1.2	0.64

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

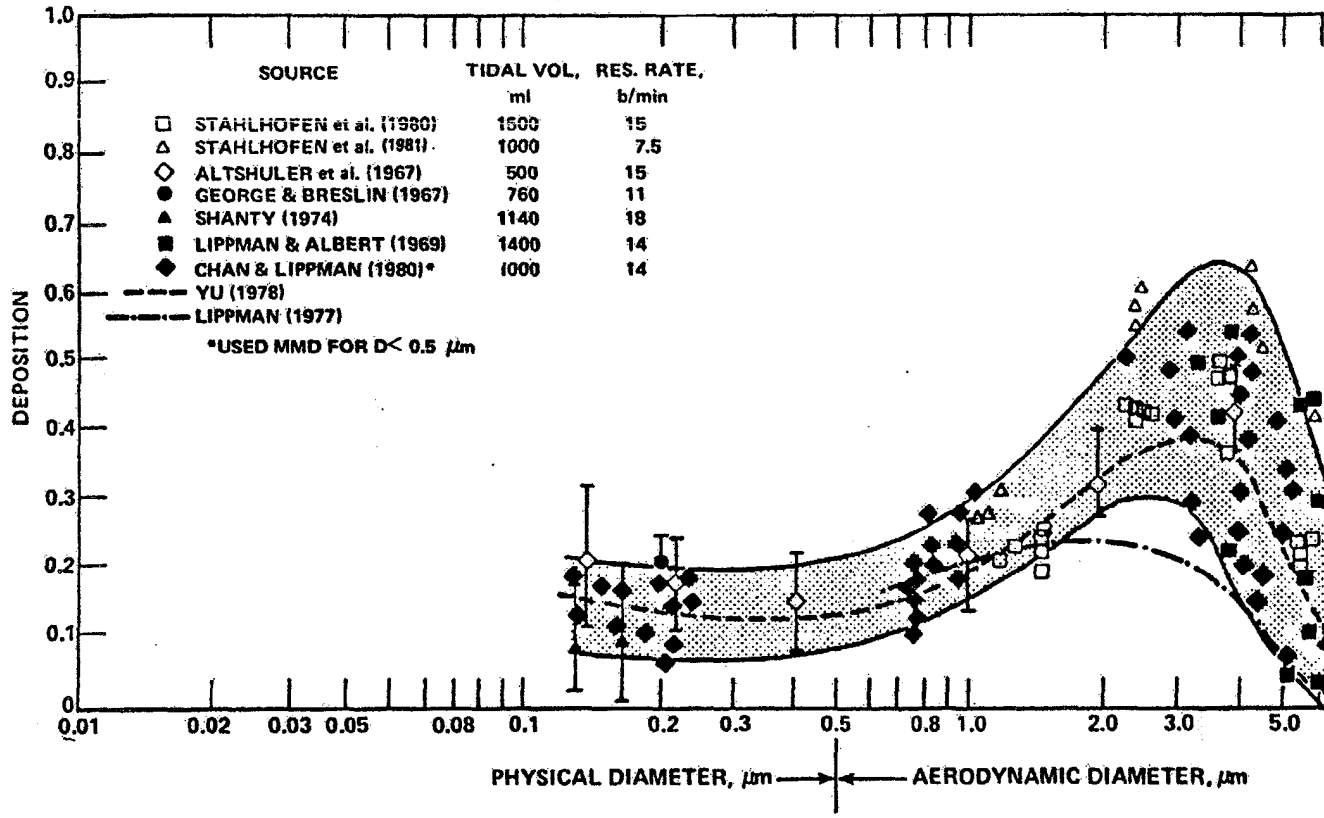

Figure 7-3. Deposition of monodisperse aerosols in the pulmonary region for mouth breathing as a function of aerodynamic diameter, except below 0.5 μm where deposition is plotted vs. physical diameter. The eye-fit band envelops deposition data cited by the different investigators. The dashed line is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

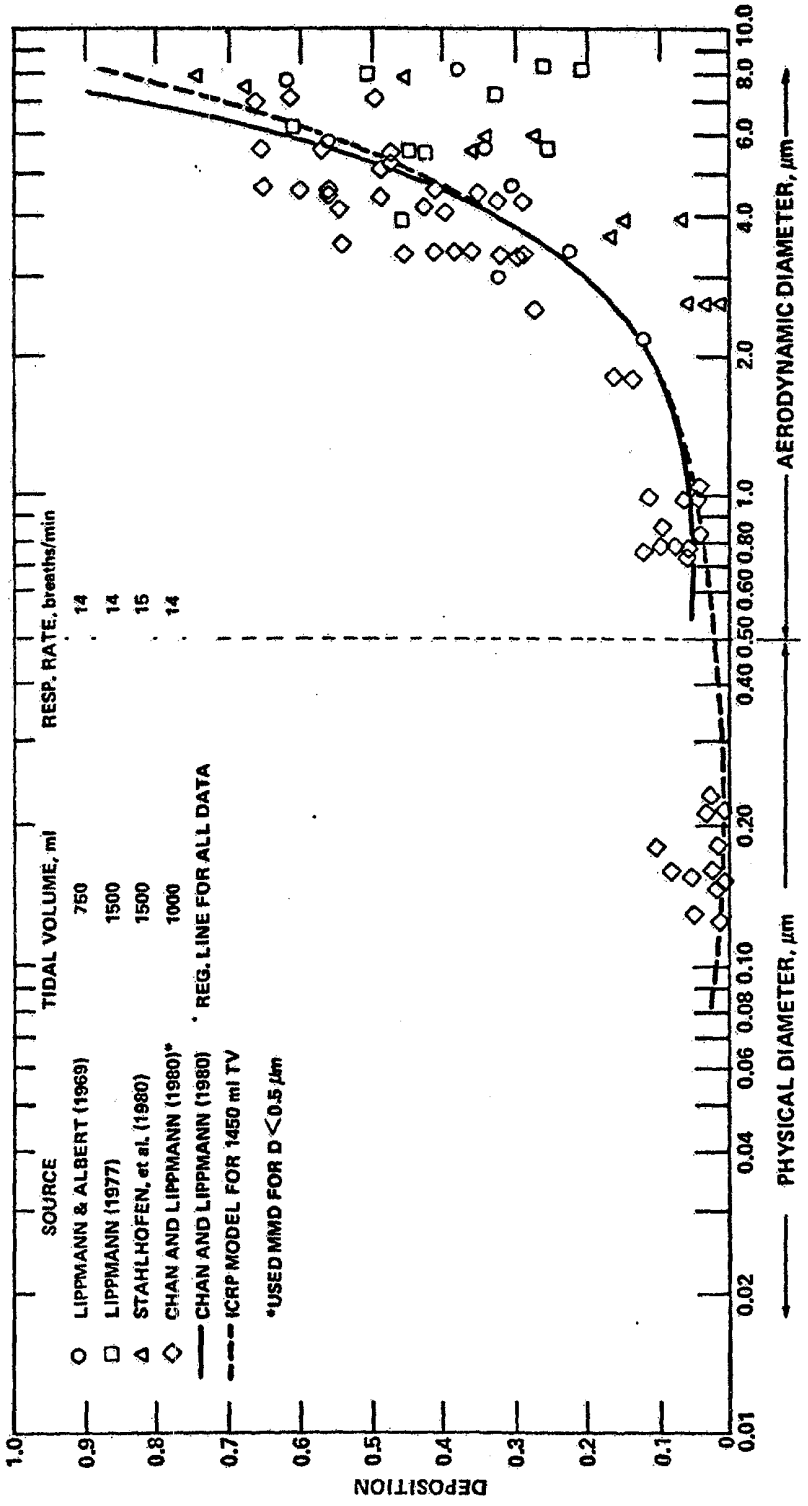

Figure 7-4. Deposition of monodisperse aerosols in the tracheobronchial region for mouth breathing in percent of the aerosols entering the trachea as a function of aerodynamic diameter except below 0.5 μm where deposition is plotted vs. physical diameter as cited by different investigators. Dashed line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

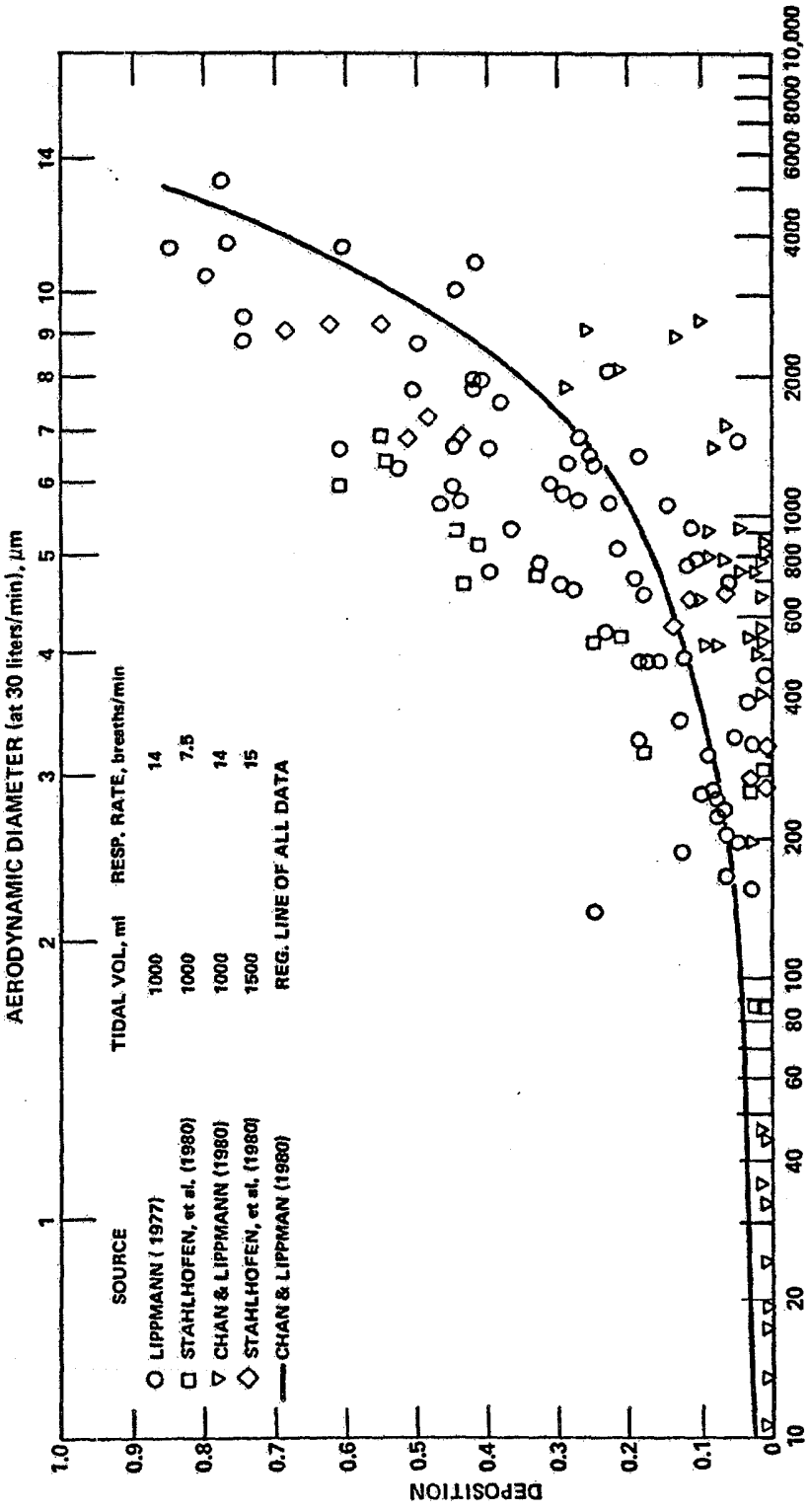

Figure 7-5. Deposition of monodisperse aerosols in extrathoracic region for mouth breathing as a function of D^2Q, where Q is the average inspiratory flow rate in liters/min. The data are the individual observations as cited by the various investigators. The solid line is the overall regression derived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

are likely to be inaccurate, since an appreciable fraction of the airborne lead mass was probably not sampled.

The distributions of Figure 7-2 have been used with published lung deposition data to estimate the fraction of inhaled airborne lead deposited in the human respiratory system. Figures 7-3 through 7-5, showing lung deposition as a function of particle size, have been taken from the EPA document Air Quality Criteria for Particulate Matter and Sulfur Oxides. Results of this calculation are shown in Table 7-13. Note that a significant fraction of lead can reach the pulmonary region where transfer to the bloodstream occurs. The observed deposition fractions are consistent with the results of lead isotope studies, showing appreciable transfer of inhaled lead into the blood (Chapter 10).

7.2.1.3 Vertical Gradients of Lead in the Atmosphere--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 is not sufficient to draw conclusions regarding general exposure conditions.

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 wind from the northeast (street running north-south), lead concentrations along the east side of the street canyon ranged from 11.3 $\mu g/m^3$ at street level to 4.0 $\mu g/m^3$ at roof level. On the west side of the street, concentrations ranged from 0.9 $\mu g/m^3$ at street level to 1.3 $\mu g/m^3$ at roof level. Values for two additional sampling points above the rooftops on each side of the street were 0.4 (east side) and 0.9 $\mu g/m^3$ (west side). Lead concentrations 2 to 5 blocks away ranged from 0.1 to 0.3 $\mu g/m^3$. These data reflect the wide variability that can be expected in urban traffic environments. Under moderate cross-wind conditions, concentrations within the canyon were stringly anisotropic, and street-level concentrations along the upwind building faces were substantially higher than those along the

TABLE 7-13. 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.

downwind faces. With different wind regimes, building configurations, and stability conditions, the distribution of 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 meters. Air samplers were operated at five to seven sites during the period from Monday to Friday, 8 AM to 6 PM, for one year. The maximum individual value observed was 18 $\mu g/m^3$. The 12 month mean ranged from 1.51 $\mu g/m^3$ to 1.35 $\mu g/m^3$, 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 meters.

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 operated for 24-hour periods from 8 AM to 8 AM, but a few 12-hour samples were collected from 8 AM to 8 PM. 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 7-14. 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.

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 $\mu g/m^3$, 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 of 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 $\mu g/m^3$, respectively, at 3 m. The corresponding values at 20 m were 0.59, 0.38, and 0.31 $\mu g/m^3$, 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.

TABLE 7-14. AIRBORNE LEAD CONCENTRATIONS AT 5- AND 20-FT ELEVATIONS ABOVE STREET LEVEL (PEDCO ENVIRONMENTAL, 1977)
($\mu g/m^3$)

		Averages					
		East side of street			West side of street		
Location	Range	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

[a]Side of street.
[b]Height above street level.

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. Our understanding of the complex factors affecting the vertical distribution of airborne lead is extremely limited.

7.2.2 Soil

Lead concentrations in soil at a depth of 20 cm range from less than 10 to greater than 70 µg/g (Shacklett et al., 1971). The arithmetic mean of 20 and geometric mean of 16 µg/g reflects the fact that most of the 863 samples were below 30 µg/g at this depth. The study was designed to determine the elemental composition of soil materials derived from the earth's crust, not the atomosphere. The range of values probably represent natural levels of lead in soil, although there may have been some contamination with anthropogenic lead during collection and handling. McKeague and Wolyneta (1980) found the same arithmetic wear (20 µg/g) for 53 uncultivated Canadian soils. The range was 5 to 50 µg/g and there was no differences with depth between the A, B and C horizons in the soil profile.

Studies discussed in Chapter 6 have determined that atmospheric lead is retained in the upper two centimeters of undisturbed soil, especially soils with at least 5% organic matter and a pH of 5 or above. There has been no general survey of this upper 2 cm of the soil surface in the United States, but several studies of soil lead near roadsides and smelters and a few studies of soil lead near old houses with leadbased paint can provide the backgound information for determining potential human exposures to lead from soil.

Because lead is immobilized by the organic component of soil (Chapter 6), the concentration of anthropogenic leac in the upper 2 cm is determined by the flux of atmospheric lead to the soil surface. Near roadsides and smelters, this flux is largely by dry deposition and the rate depends on particle size and concentration. These factors vary with traffic density and average vehicle speed. (Chapter 6). In general, deposition flux drops off abruptly with

increasing distance from the roadway. This effect is demonstrated in studies which show surfacesoil lead decreases exponentially up to 25 m from the edge of the road. The original work of Quarles et al (1974) showed decreases in soil lead from 550 µg/g within 25 m along side a highway with 12,500 vehicles/day in Virginia. Their findings were confirmed by Wheeler and Rolfe (1979), who observed an exponential decrease linearly correlated with traffic volume. Agrawal et al (1981) found similar correlations between traffic density and roadside proximity in Baroda City, as did Garcia-Miragaya et al (1980) in Venzuela and Wong and Tam (1978) in Hong Kong. The extensive study of Little and Wiffen (1978) is discussed in Chapter 6; these authors found additional relationships between particle size and roadside proximity and decreases with depth in the soild profile. The general conclusion from these studies is that roadside soils may contain atmospheric lead from 30 to 2000 µg/g in excess of natural levels within 25 meters of the roadbed, all in the upper layer of the soil profile.

Near primary and secondary smelters, lead in soil decreases exponentially within a 5 to 10 km zone ground the smelter complex. Soil lead contamination varies with the smelter emission rate, length of time the smelter has been in operation, prevailing windspeed and direction, regional climatic conditions and local topography (Roberts, 1976).

Little and Martin (1972) observed decreases from 125 to 10 µg/g in a 6 km zone around a smelting complex in Great Britain, and all of the excess lead in the upper 6 cm of the soil profile. Roberts (1976) reported soil lead between 15,000 and 20,000 near a smelter in Toronto. Kerin (1976) found 5,000 to 9,000 µg/g adjacent to a Yugoslavian smelter; the contamination zone was 7 km in radius. Ragaini et al. (1977) observed 7900 µg/g near a smelter in Kellogg, Idaho; they also observed a 100-fold decrease with depth doax to 20 cm in the soil profile. Palmer and Lucera (1980) observed soil lead in excess of 60,000 µg/g near two smelters in Missouri, decreasing to 10 µg/g at 10 km.

Urban soils may be contaminated from a variety of atmospheric and non-atmospheric sources. The major sources of soil lead seem to be paint chips from older houses and deposition from nearby highways. Lead in soil adjacent to a house decreased with distance; this may be due to paint chips or to dust of atmospheric origin washing from the rooftop (Wheeler and Rolfe, 1979). Davies (1978) and Davies et al. (1979) have described the effect of soil lead contaminations in urban gardens in London. Soils ranged from 10 to 2600 µg/g; lead in radishes was 50-fold higher in the contaminated soil. This relation-

ship between soil and plant lead is discussed further in Section 7.2.4 and in Chapter 8.

Andresen et al (1980) reported lead in the litter layer of 51 forest soils in the northeastern United States. They found values from 20 to 700 µg/g, which can be compared only qualitatively to the soil lead concentration cited above. This study clearly shows that the major pathway of lead to the soil is by the decomposition of plant material containing high concentrations of atmospheric lead.

Finally, a definitive study which describes the source of soil lead was reported by Gulson et al (1981) for soils in the vicinity of Adelaide, South Australia. In an urban to rural transect, stable lead isotopes were measured in the top 10 cm of soils over a 50 km distance. By their isotopic compositions, three sources of lead were identified: natural, non-automotive industrial lead from Australia, and tetraethyl lead manufactured in the United States. The results indicated most of the soil surface lead originated from leaded gasoline. Similar studies have not been conducted in the United States.

7.2.3. Dusts

Dusts are solid aerosol particles which are usually produced by the disintegration of larger particles (Friedlander, 1977). They should be distinguished from soil, which is complex mixture of organic and inorganic substances of varied origin. Exposed surface soil may become airborne during wind erosion, at which time it becomes dust. Other sources of dust may be automotive exhausts, industrial stack emissions and fugitive dust from manufacturing processes. Dusts of importance to potential human exposure are roadway dusts which may wash or settle into adjacent play areas, manufacturing dusts which may be carried home on the clothing of workers, and household dusts which accummulate on windowsills and other flat surfaces likely to contact the hands and mouths of small children.

Nriagu (1978) reviewed several studies of lead in street dust, including a report by Kaye and Reznikoff (1947) that lead in New York City street dust was 1200 µg/g in 1924 before the use of lead additives in gasoline. The source of lead was probably flue dust from burning coal. Warren et al. (1971) reported lead in street dust of 20,000 µg/g in a heavily traffiked area. In the review by Nriagu (1978), street dust lead concentrations ranged from 300 to 18,000 µg/g in several cities in the United States. In Hong Kong, lead in street dust ranged from 960 to 7400 µg/g with no direct relationship to traffic volume (Ho, 1979). In other reports from Hong Kong, Lau and Wong (1982) found

values from 130 µg/g at 20 vehicles/day to 3900 µg/g at 37,000 vehicles/day. Fourteen sites in this study showed close correlation with traffic density.

In the United Kingdom, lead in urban and rural street dusts was determined to be 970 and 85 µg/g, respectively, by Day et al. (1975). A later report by this group (Day et al., 1979) discusses the persistency of lead dusts in rainwashed areas of Great Britain and New Zealand and the potential health hazard due to ingestion by children. They concluded that, whereas the acidity of rain was insufficient to dissolve and transport lead particles, the potential health hazard lies with the ingestion of these particles during the normal play activities of children residing near these areas. A child playing at a playground near a roadside might consume 20 to 200 µg Pb while eating a single piece of candy with unwashed hands.

Lead in manufacturing dusts varies greatly with the type of industry. There are two types, stack emissions which are dispersed locally or regionally, and fugitive dusts which are usually confined at or near the working environment. Occupational exposure to manufacturing dusts are discussed in Section 7.3.6. Of interest here is the contribution of industrial dusts on workers' clothing to the household environment.

Dust is a normal component of the home environment. It accumulates on all upfacing surfaces, especially furniture, rugs and windowsills. For reasons of hygiene and respiratory health, most homemakers devote much time and money toward removing this dust from the household. Because there are at least two circumstances where these measures are inadequate, it is important to consider the possible concentration of lead in these dusts in order to determine potential exposure to young children. First, some households do not practice regular dust removal, and secondly, in some households, of workers exposed occupationally to lead dusts, the worker may carry dust home.

In Omaha, Nebraska, Angle and McIntire (1979) found lead in household dust ranged from 18 to 5600 µg/g. In Lancaster, England, a region of low industrial lead emissions, Harrison (1979) found household dust ranged from 510 to 970 with a mean of 720 µg/g. They observed soil particles (10200 µm), carpet and clothing fibers, animal and human hairs, food particles, and an occasional chip of paint. The previous Air Quality Criteria for Lead document (U.S. Environmental Protection Agency, 1977) summarized earlier reports of lead in household dust showing residential surburban areas range from 280 to 1500 µg Pb/g, urban residential from 600 to 2000 µg/g, urban industrial from 900 to 16,000. In El Paso, Texas, household dust ranged from 2800 to 100,000 µg/g within 2 km of a smelter (Landrigan et al. 1975).

7.2.4 Vegetation

Lead in vegetation may be atmospheric on the surface or a combination of atmospheric and soil in the internal tissues. The mechanisms for the dry deposition of aerosol particles directly to vegetation surfaces are discussed in Chapter 6 and the mechanisms for the transfer of lead from soil to roots are discussed in Chapter 8. Of interest here are lead concentrations in roadside and urban vegetation which may affect potential human exposures.

As with soils, lead on vegetation surfaces decreases exponentially with distance away from roadsides and smelters (Cannon and Bowles, 1962; see also Chapter 8). This lead is persistent. It is neither washed off by rain nor taken up through the leaf surface. For many years, plant surfaces have been used as as indcators of lead pollution (Garty and Fuchs, 1982; Pilegaard, 1978; Ratcliffe, 1975; Ruhling and Tyler, 1969; Tanaka and Ichikuni, 1982) and in some cases for the exploration of lead (Brooks et al., 1979). These data all show that lead on the surface of leaves and park is proportional to traffic density and distance from the highway, which are the primary determinants of air lead concentrations and particle size, distrubutions. Other factors such as surface roughness, wind direction and speed are discussed in Chapter 6. The data also show that lead in internal plant tissues is directly related to lead in soil.

Lead in forage was found to exceed 950 µg/g within 25 m of roadsides by Graham and Kalman (1974), with 25,000 or more vehicles per day. At lesser volumes, 200 µg/g were found. These authors review other reports of 20 to 660 µg/g with the same same relationship to traffic volume and distance from the road. A more recent study of Crump and Barlow (1982) showed the accumulation of lead on forage is directly related to the deposition rate, which varied seasonally according to traffic density. The deposition rate was measured using the moss bag technique. Rain was not effective in removing lead from the surface.

The effect of lead on edible crops was measured by Ter Haar (1970), who showed that edible plant parts not exposed to air (potatoes, cron, carrots, etc.) are not affected by atmospheric concentrations of lead. Leafy vegetables are. In edible parts, such as corn husks, wheat and oat chaff and soybean hulls were also contaminated. These results were confirmed by McLean and Shields (1977), who found most of the lead on leaves and husks. The general conclusion from these studies is that lead in food crops varies according to exposure to the atmosphere and in proportion to the effort taken to separate

husks, chaff and hulls from edible parts during processing for human or animal consumption.

Near Smelters, Merry et al., (1981) found a pattern different from roadside studies cited above. They observed that wheat crops contained lead in proportion to the amount of soil lead, not surface contamination. A similar effect was reported by Harris (1981).

7.3 POTENTIAL HUMAN EXPOSURES

The preceding section discussed ambient concentrations of lead in the environment, focusing on levels in the air, soil, and vegetation. The available data showed that lead levels in environmental media vary greatly, because of natural variations as well as anthropogenic activities. In this section, environmental lead concentrations are examined from the perspective of potential exposures. Six categories of sources of exposure are considered: emissions from mobile sources, emissions from stationary sources, dietary lead (food and water), lead in manmade materials, lead in soil and dust, and occupational exposures.

7.3.1 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 (USEPA, 1979).

Table 7-15 summarizes the average 4-hour and 24-hour airborne lead concentrations during the summer months (May-October). Note that concentrations measured at sites upwind of the freeway are considerably smaller than levels at downwind locations.

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. Hirschler et al. (1957) have shown that lead emissions increase with traffic speed.

Ledolter et al. (1979) have used 1976-1977 data from the Los Angeles catalyst study to develop a simple model for roadside airborne lead concentrations. The model shows that concentrations are functions of traffic density, vehicle speed, atmospheric stability, windspeed, and wind direction. The influence of

TABLE 7-15. FOUR-HOUR AND 24-HOUR AIRBORNE LEAD CONCENTRATIONS AT TWO ROADSIDE SITES IN THE LOS ANGELES CATALYST STUDY, 1974-1977

		4-Hr Pb Concentrations (3 p.m. to 7 p.m.)		
Site	Year(s)	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

		24-Hr Pb Concentrations		
Site	Year(s)	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 road on ambient airborne concentrations can be observed only for a short downwind distance, due to rapid settling of large lead-containing particles as well as dilution.

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 $\mu g/m^3$ were found on streets of varying widths and traffic flow. These values were 0.3 to 2.65 percent of the total airborne lead. The level of organic lead inside a busy tunnel was 0.02 $\mu g/m^3$ or 0.1 percent of the total airborne lead. Not surprisingly, organic lead concentrations measured at a busy service station ranged from 0.21 to 0.59 $\mu g/m^3$, or 3.9 to

9.7 percent of total airborne lead at that site. At a less busy service station, the concentration of organic lead was 0.07 $\mu g/m^3$, 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 $\mu g/m^3$, 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 to 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 to 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, a 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 were about 2 $\mu g/m^3$ for each 1000 vehicles/hour. For a traffic flow of 1000 vehicles/hour, an airborne lead concentration exceeding 1 $\mu g/m^3$ 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 by 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 7-16. 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.

TABLE 7-16. 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 μg Pb/g for main roads and 900 μg Pb/g for side roads. An additional seven samples collected from a footpath in a rural area near London contained an average of 35 μg Pb/g.

Data obtained in a number of other studies on lead in dust near roadways are summarized in Tables 7-17 and 7-18. These data demonstrate that abnormally high concentrations of lead are found in the air and dust near major roadways, and that people who live or work in such areas (e.g., traffic policemen, service station and garage attendents) are exposed to high lead concentrations.

TABLE 7-17. 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)

TABLE 7-18. LEAD CONTENT IN OR ON ROADSIDE SOILS 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

[a]Adapted from Lagerwerff and Specht (cited in National Academy of Sciences, 1972).

For comparison, Table 7-19 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 is 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 which is ingested 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 airorne lead, ingestion of food or water containing deposited lead-containing aerosol, or direct ingestion of leaded dust.

7.3.2 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 descibed 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,

TABLE 7-19. LEAD DUST IN RESIDENTIAL AREAS

Sampling site	Concentration, μg 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)

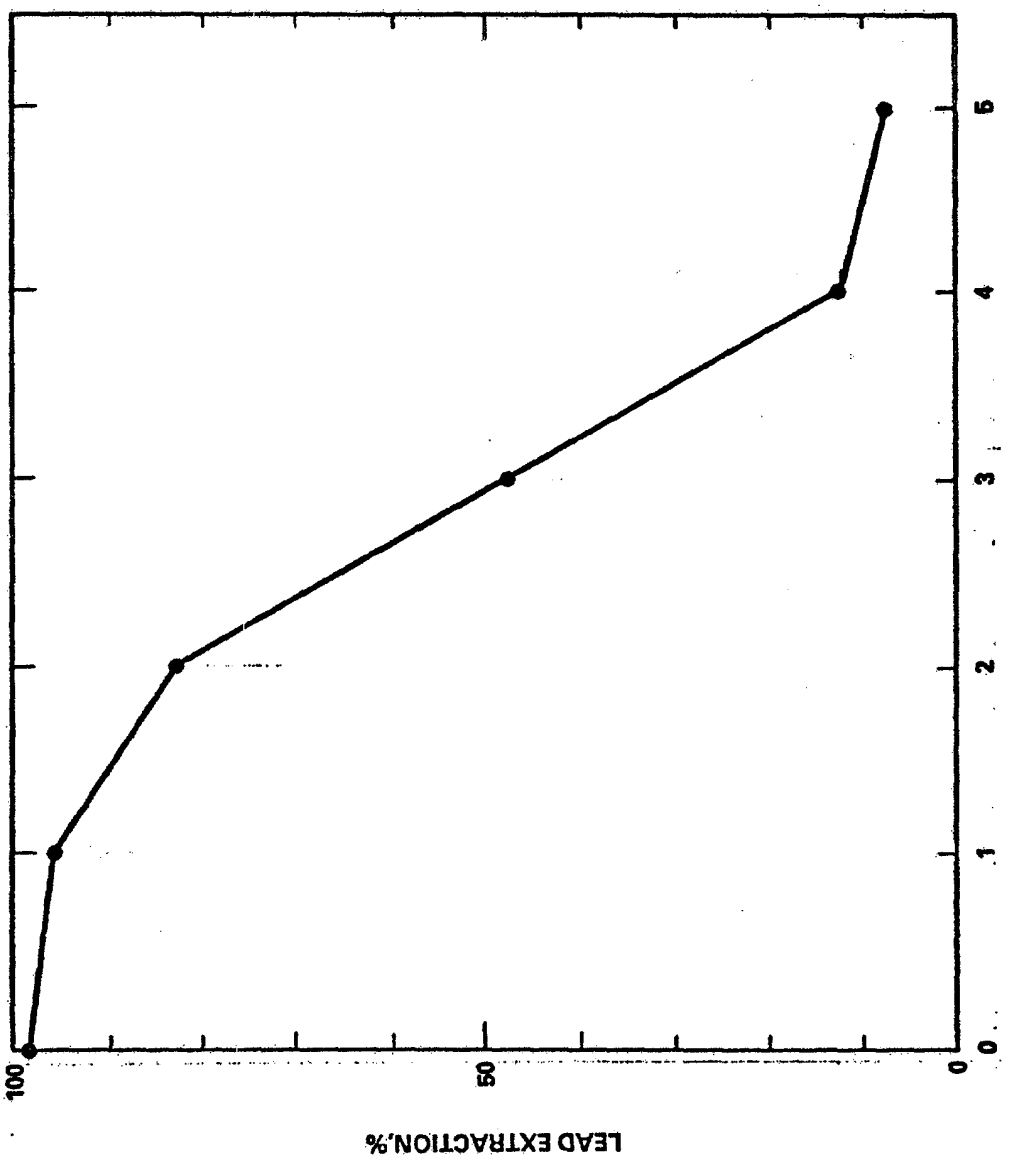

Figure 7-6. Solubility of lead in street dust as a function of pH, using hydrochloric acid at 20° C.
Source: Day et al. (1979).

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 lead for any location in the study area. The mean annual ambient air levels near the smelter for two different years (1974, 1975) are shown graphically in Figure 7-7. Similar results 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 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.

7.3.3 Dietary Exposures

7.3.3.1 Food--The route by which most people receive the largest portion of their daily lead intake is through foods. Several studies have reported average dietary lead intakes in the range 100 to 500 µg/day for adults, with individual diets covering a much greater range (Schroeder and Tipton, 1968; Tepper, 1971; Mahaffey, 1978; Nutrition Foundation Expert Advisory Committee, 1982). 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 µg/day in 1937 to 100 µg/day in 1970, although there is considerable variability in the data. Only a fraction of this lead is absorbed, as discussed in Chapter 10.

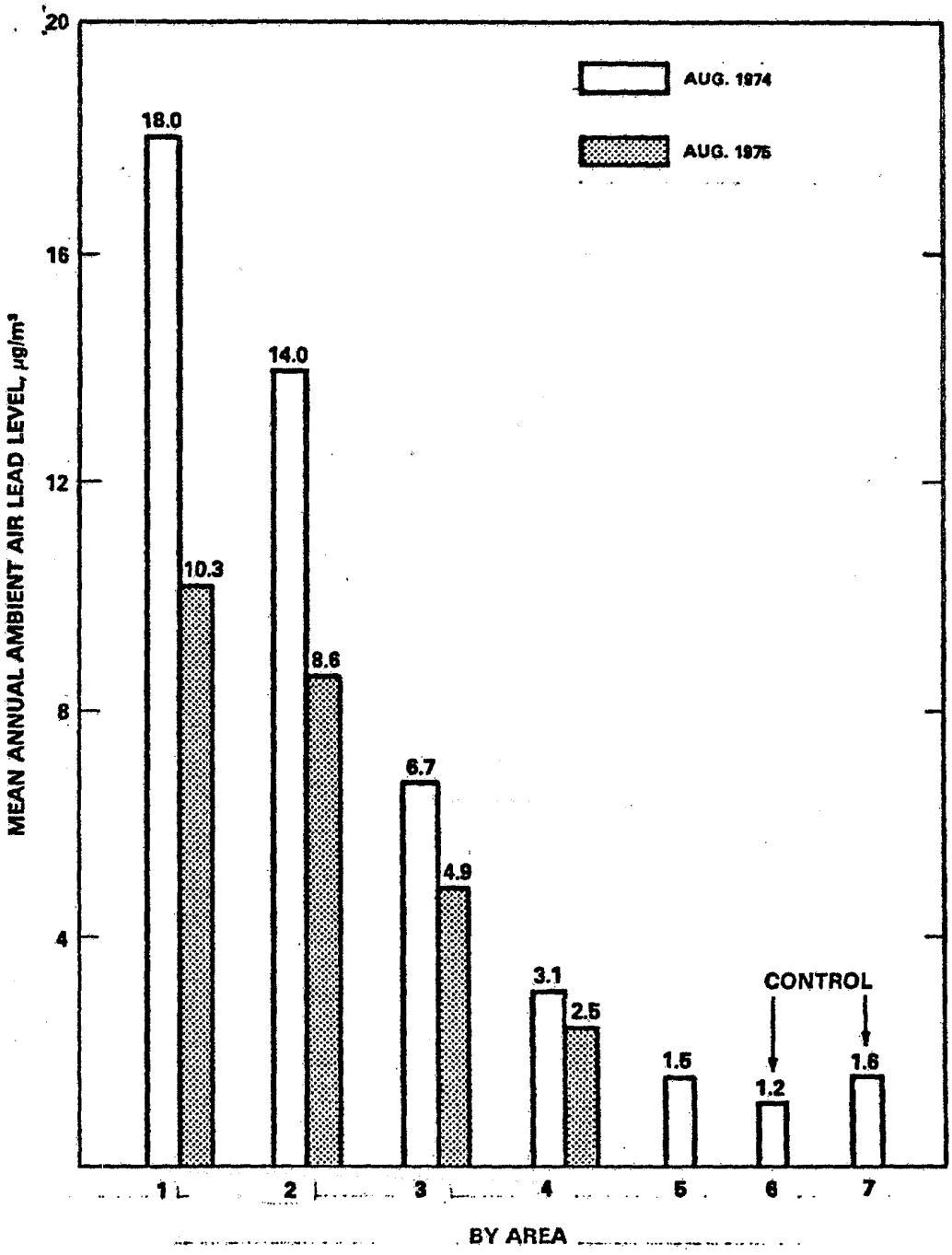

Figure 7-7. Annual ambient air lead concentration near a smelter, by area, before the August 1974 and August 1975 surveys. Area 1 is within 1 mile of smelter; Area 2 is 1 to 1-½ miles from smelter; Area 3, 2½ to 6 miles; Area 4, 6 to 15 miles; and Area 5, 15 to 20 miles.
Source: Yankel et al. (1977).

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 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 are 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 by 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 µg/g lead (dry weight) associated with it, whereas the soil content was only 10 µg/g. The lead content in the lettuce was reported to be 0.15 to 1.5 µg/g 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.

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 and deposition of the emitted lead. 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 conducted during 19??, have found maximum concentrations of 1.5 µg/g for condiments, 2.5 µg/g for fish and other seafood, 3.7 µg/g for meats and eggs, 1.4 µg/g for grains, and 1.3 µg/g for vegetables. All of these values refer to unprocessed foods. A British report (United Kingdom Ministry of Agriculture, Fisheries, and Food, 1982) in a study conducted in 1979-80 on lead in foods describes similar maximum values for meat and eggs, grain products (flour and bread), and vegetables, but concentrations up to 14 µg/g in condiments, and up to 18 µg/g in shellfish. More recently, the Nutrition Foundation Expert Advisory Committee (1982) has used data from a 1975 Canadian survey to estimate lead levels in several categories of infant food. Strained meats, vegetables, and fruits typically contained 0.03 to 0.05 µg/g, while cereals contained 0.09 µg/g. Ziegler et al. (1978) reported in a study in 1974-75, a wide range of 0.013 to 0.327 µg/g for lead in infant fruit juices, fruits, and vegetables.

The lead content of milk is of special interest because it is a major component of the diets of infants and young children. Brandt and Bentz (1971)

report that levels in fresh milk are typically less than than 5 µg/liter. The survey by USFDA (1975) found lead concentrations in whole milk ranging from 10 to 70 µg/liter, and averaging about 20 µg/liter. In a recent survey 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 µg/liter (range 15 to 20 µg/liter) in formula and 10 µg/liter in milk. Tolan and Elton (1973) reported lead levels of 30 µg/liter in fresh milk in Great Britain and 50 µg/liter in canned (evaporated) milk. Michell and Aldous (1974) reported a comparable average of 40 g/liter for fresh whole milk purchased in New York State, but their results for evaporated milk averaged 202 µg/liter and ranged as high as 820 µg/liter.

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 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 Pb/liter can contribute up to 396 µg Pb/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 Pb/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 Pb/liter; hard water containing 520 µg Pb/liter provides a vegetable lead intake as great as 130 µg/day. These authors estimate a total dietary lead intake of 87 to 440 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 Pb/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 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. For typical North American lead exposures, generally in the absense of lead plumbing, the Nutrition Foundation Expert Advisory Committee (1982) used 1975 Canadian food data to estimate lead intakes of 61-73 µg/day for children aged 1-4 years, and intakes of 77-132 µg/day for older children and adults.

In a survey of heavy metals in foods, the USFDA (1975) found relatively high lead concentrations in metal-canned foods. In the adult food category, canned foods averaged 0.376 µg Pb/g, and non-canned foods averaged 0.156 µg Pb/g. In the baby food category, canned foods (juices) averaged 0.329 µg Pb/g, and foods in jars averaged 0.090 µg/g. The report concluded that from the age of about 1 year on, canned foods comprise 11 to 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 in 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 Pb/liter or more; 12 percent contained 400 µg Pb/liter or more. These levels are markedly above the potable water standard of 50 µg Pb/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 μg Pb/g (900 to 7000 μg/liter), and 18 products averaged 2.7 μg/g (Hankin et al., 1975). Apart from the possible toxic effects on pets, the products pose a hazard to people who may include themq 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 7-20, 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 7-20 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 accurate lead data at levels commonly encountered in food and in the environment.

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 10,000 μg/g. The higher concentrations are attributed to lead-containing inks. No related illnesses 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, occasionally causes lead poisoning in adults. The apparent source of the lead is the soldered joints in the distilling apparatus.

TABLE 7-20. 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).

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 USFDA (1979) has established a maximum permissible concentration of 7 µg Pb/g 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 µg/g in Mexican pottery. The leached lead decreases with exposure time, and after several days appears to asymptotically oach a value which may be as great as 600 µg/g. 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 produced in primitive Lebanese potteries had lead levels above the 7 µg/g 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 µg/g 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 300 µg/liter. Measurements of lead in domestic wines have not been undertaken; if the European data ndicative, 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 in 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.

7.3.3.2 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 Becker (1964) analyzed untreated and treated water for the largest U.S. 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% 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 interstate carrier water supplies in 1975 showed that only 0.3 percent exceeded the 50 µg/liter standard (USEPA, 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 U.S. 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 of 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 (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 solubilty of those species of lead that may be present in drinking water and suggested that the solution of 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 low 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 the 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 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 µg/liter) were found in 63 percent of the samples, but only three samples contained more than 50 µg/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 µg/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, of the samples; the observed values ranged from 2 to 140 µg/liter. The highest concentration was detected in the Ohio River at Evansville, Indiana. Twenty-seven of their samples exceeded 50 µg/liter. Observed mean observations of >30 µg/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 BOCA Code, allow the use of lead pipe and list lead as 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 lead. 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, 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.

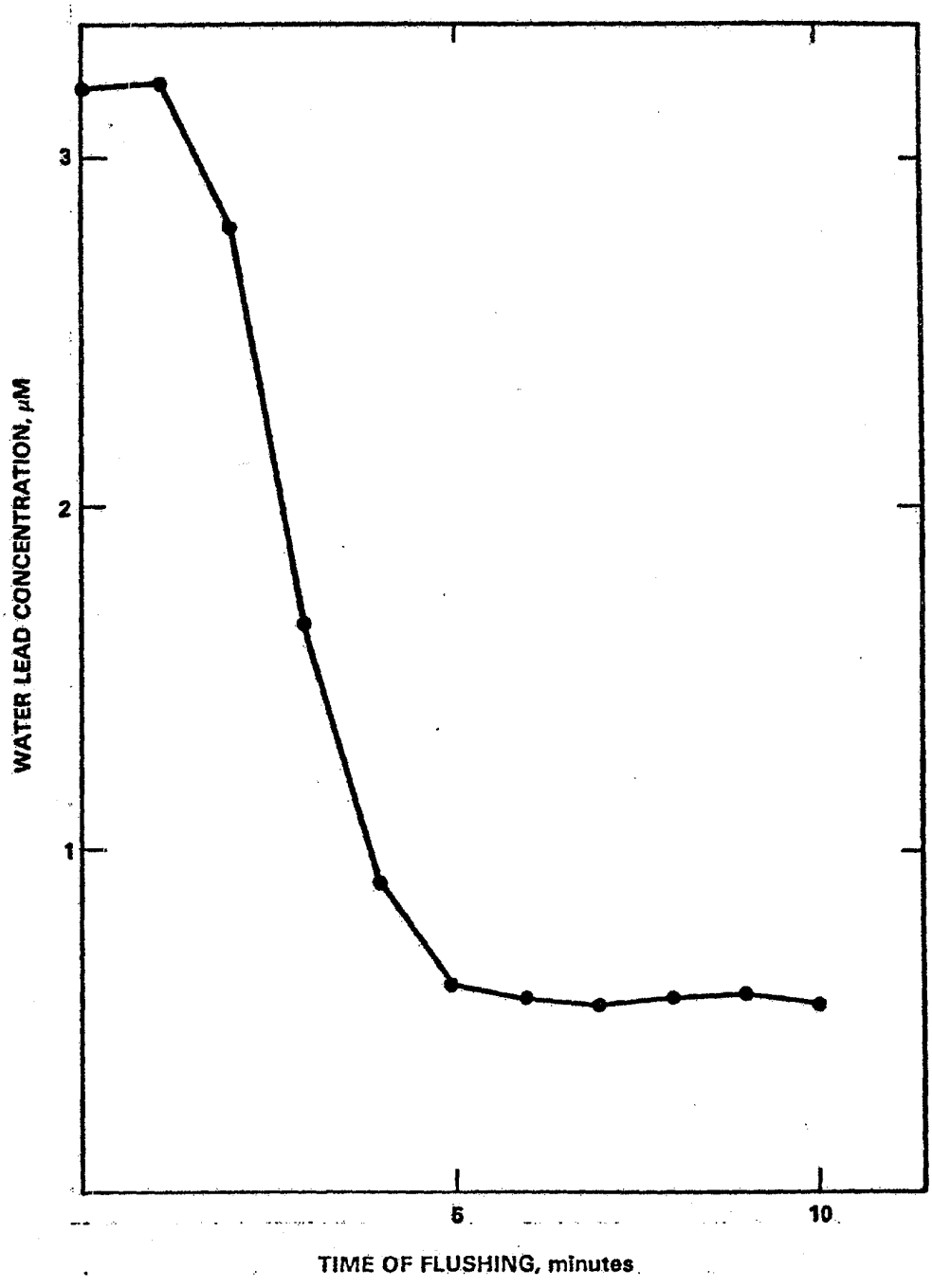

Figure 7-8. Change in drinking water lead concentration in a house with lead plumbing for the first use of water in the morning. Flushing rate was 10 liters/minute.

Source: Moore (1977).

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.

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 in 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/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, 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, 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). 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

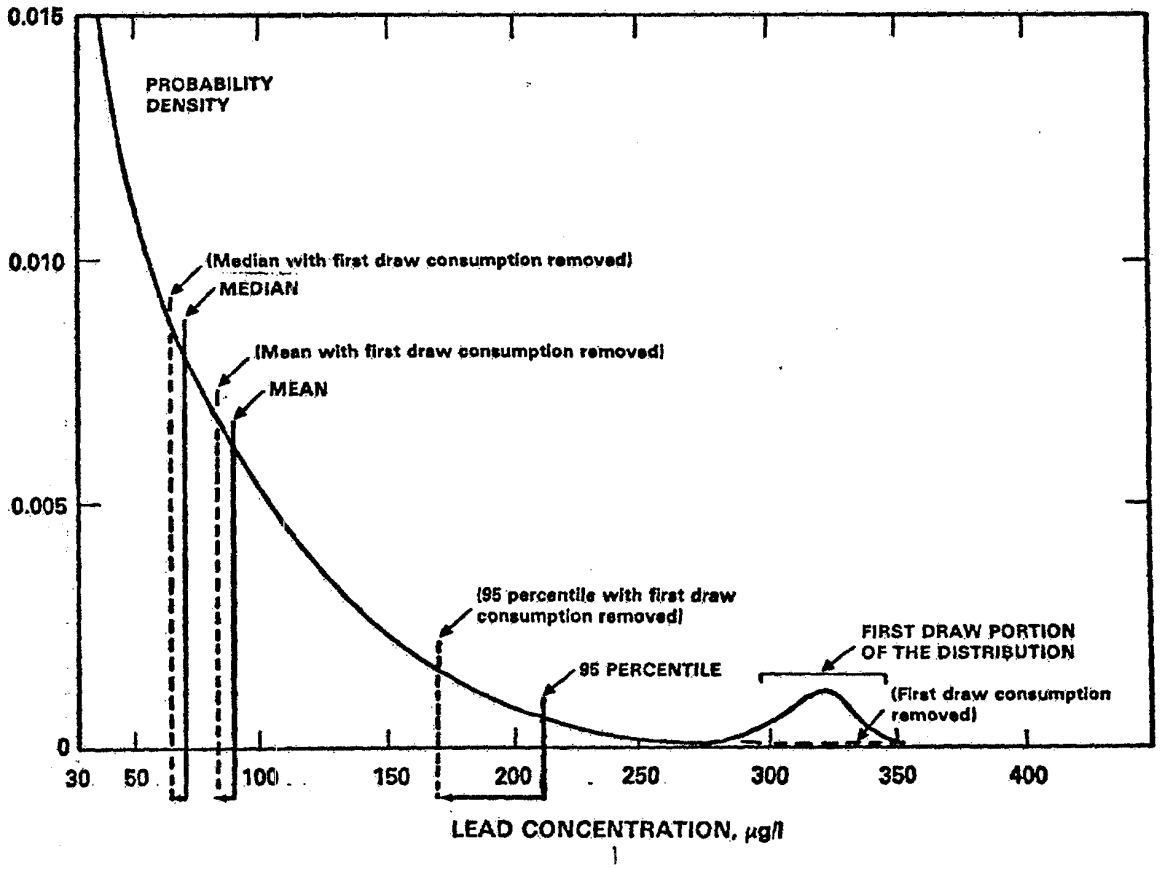

Figure 7-9. Theoretical distribution of drinking water lead concentrations based on the model of Bailey and Russell (1981). Shown on the graph are the reductions in median, mean, and 95 percentile lead levels if first draw water consumption is removed.

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.

7.3.4 Exposures Resulting from Manmade 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 Pb/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 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.

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 µg/g next to a barn in rural Massachusetts. A steady decrease in lead level with increasing distance from the barn was shown, reaching 60 µg/g 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 µg/g; the average concentration at ten feet was slightly more than 400 µg/g. The same author reported smaller soil lead elevations in the vicinity of eighteen brick veneer houses in Detroit. Soil lead levels near painted barns located in rural areas were similar to urban soil lead concentrations near painted houses, suggesting the importance of leaded paint at both urban and rural locations.

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 products. Diffusion, or leaching by solvents, is estimated to be quite slow--on the order of 10^{-10} to 10^{-12} cm^2/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 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 μg Pb/g (Hankin et al., 1973), while colored gift wrapping may contain up to 14,300 μg Pb/g (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 μg Pb/g (9.7 percent); more than half of the samples exceeded the allowable limit for painted toys, which is 0.06 percent.

7.3.5 Soil and Dust Exposures

In each of the previous subsections on potential exposures, the role of lead in soil and dust may have been important. The discussion of mobile sources illustrated that the lead content of road dust and of surface soil near roads increases as traffic density increases. Household dust and surface soil in the vicinity of lead-emitting stationary sources may also have elevated lead content. Dietary intake can be influenced by deposited lead-containing dust, particularly if the diet includes crops grown near sources of lead. Regarding manmade materials, flaking paint can increase the lead content of housedust and surface soil near painted surfaces.

It is clear that even in the absence of anthropogenic lead emissions, there would be some human exposure to lead through inadvertant ingestion of natural lead in soil. In the case of small children, deliberate ingestion of lead-containing soil is possible. The extent to which current human exposure to lead exceeds that expected from natural sources is discussed in the final section of this chapter.

7.3.6 Occupational Exposures

The highest and most prolonged exposures to lead are found among workers in the lead smelting, refining, and manufacturing industries (WHO, 1977). In the work areas, the major route of lead exposure is by inhalation and ingestion of lead-bearing dusts and fumes. Airborne dusts settle out of 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.

7.3.6.1 Exposures in Lead Mining, Smelting, and Refining--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 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 7-21. 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 to 1.4 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.

The greatest potential for high-level exposure exists in the process of lead smelting and refining (WHO, 1977). The most hazardous operations are those in which molten lead and lead alloys are brought to 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 smelter, the amount of particle mass in the respirable size range may be much greater near the furnace.

A measure of the potential lead exposure in smelters was obtained in a study of three typical installations in Utah (WHO, 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 $\mu g/m^3$. Obviously,

TABLE 7-21. AIRBORNE LEAD CONCENTRATIONS BASED ON PERSONAL SAMPLERS, WORN BY EMPLOYEES AT A LEAD MINING AND GRINDING OPERATION IN THE MISSOURI LEAD BELT

(mg/m^3)

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).

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 $\mu g/m^3$ were measured in a primary lead-zinc smelter in the United Kingdom (King et al., 1979). These authors found poor correlations 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.

Spivey et al. (1979) studied a secondary smelter in southern California which recovers lead mainly from automotive storage batteries. Airborne lead concentrations of 10 to 4800 g/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 nonsmokers. 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 µg/g. The amount of lead contained in pieces of cloth 1 in^2 cut from the bottoms of trousers worn by the 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 µg/g, with a median of 2400 µg/g. 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 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 $\mu g/m^3$ in some instances, with personal air sampler data ranging from 120 $\mu g/m^3$ for a battery wrecker to 350 $\mu g/m^3$ 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 $\mu g/m^3$, respectively. When charging the furnace, the last value increased to 2650 $\mu g/m^3$. Personal samplers yielded concentrations of 17 $\mu g/m^3$ for an office worker, 700 $\mu g/m^3$ for two welders, and 2660 $\mu g/m^3$ 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 $\mu g/m^3$. These levels were subsequently reduced to 30 to 40 $\mu g/m^3$ with the installation of forced ventilation systems to exhaust the work area atmosphere to the outside.

7.3.6.2 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 quantites 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 Pb/in^2 of coating) produced breathing-zone concentrations of lead reaching 15,000 $\mu g/m^3$, far in excess of 450 $\mu g/m^3$, the current occupational short-term exposure limit (STEL) in the United States (Pegues, 1960). Under good ventilation conditions, a concentration of 140 $\mu g/m^3$ was measured (Tabershaw et al., 1943).

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 $\mu g/m^3$ and ranged as high as 2400 $\mu g/m^3$ (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.

7.3.6.3 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 16-100 $\mu g/m^3$ in a battery factory in Italy, while values up to 1315 $\mu g/m^3$ have been measured by Richter et al. (1979) in an Israeli battery factory. Excessive concentrations, as great as 5400 $\mu g/m^3$, have been quoted by WHO (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, either 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.

7.3.6.4 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 a 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 $\mu g/m^3$. In 1960, Tsuchiya and Harashima (1965) found airborne lead levels of 30-360 $\mu g/m^3$ in several print shops in Japan. More recently, Parikh et al. (1979) reported concentrations approximately 10-30 $\mu g/m^3$ in five type foundries in India; these investigators also reported much greater levels, up to 140 $\mu g/m^3$, 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.

7.3.6.5 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 sure 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 by the use of protective clothing. Linch 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 1249 $\mu g/m^3$, while inorganic lead varied from 1.3 to 62.6 $\mu g/m^3$. 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.

7.3.6.6 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. An encapsulated stabilizer which greatly reduces the occupational hazard is reported by Fischbein et al. (1982).

Sakurai et al. (1974), in a study of bioindicators of lead exposure, found ambient air concentrations averaging 58 $\mu g/m^3$ 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.

The manufacture of cans with leaded seams may expose workers to elevated environmental lead levels. Bishop (1980) reports airborne lead concentrations of 25-800 $\mu g/m^3$ 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 $\mu g/m^3$ 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 1,000 $\mu g/m^3$ were measured during sweeping operations. Another report from the same research group presents time-weighted average exposures of instructors of 45-900 $\mu g/m^3$ 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 $\mu g/m^3$, after 22 minutes of sanding an outdoor post coated with paint containing 2.5 mg Pb/cm^2. After only five minutes of sanding an indoor window sill containing 0.8-0.9 mg Pb/cm^2, the air contained 550 $\mu g/m^3$. Homeowners who attempt to remove leaded paint themselves may constitute a relatively large group at risk of excessive lead exposure.

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

Other occupations involving risk of lead exposure include stained glass manufacturing and repair, arts and crafts, and soldering and splicing.

7.4 SUMMARY

This chapter has reviewed the available information on lead concentrations in various environmental media which may result in lead exposure in populations. The first half of the chapter discusses ambient lead levels in air, soil, and vegetation. These data have generally not been obtained in conjunction with human exposure or epidemiologic information, and serve only to provide estimates of environmental lead levels. The second half of the chapter presents information

Needs to be better documented & quantified

on several categories of sources of potential lead exposure in humans. Included are mobile source emissions, stationary source emissions, dietary intake, lead in manmade materials, lead in soil and dust, and occupational exposures. Each section of the chapter will now be briefly summarized.

Ambient airborne lead concentrations have shown no marked trend from 1965 to 1977. Over the past five years, however, distinct decreases have occurred. These decreases reflect the smaller lead emissions from mobile sources in recent years. Airborne size distribution data indicate that most of the airborne lead mass is found in submicron particles. Application of respiratory deposition curves to these distributions suggests that a significant fraction of airborne lead mass can reach the lower lungs. Our understanding of vertical gradients of airborne lead is poor: data in the literature are inconsistent with respect to changes in airborne lead concentration with height. Mobile source studies have shown elevated lead concentrations in air, soil, and vegetation near heavily traveled roads. The amounts of lead in these environmental media are functions of distance from the road, traffic density, vehicle speed, atmospheric stability, windspeed, and wind direction. Persons spending considerable time in the vicinity of heavily traveled roads may experience increased lead exposure compared to the general population.

Point source studies have shown that environmental lead concentrations increase as one approaches a smelter, battery plant, or other lead emission source. Persons living in the vicinity of such a source may experience significant lead exposure. Families having at least one individual employed at or near this type of source may receive additional exposure, due primarily to lead-bearing dust carried home air, skin, and clothing.

Most people receive the largest portion of their lead intake through foods. Unprocessed foods such as fresh fruits and vegetables receive lead by atmospheric deposition as well as uptake from soil; crops grown near heavily traveled roads generally have greater lead levels than those grown at greater distances from traffic. For many crops, the edible internal portions of the plant (e.g. kernals of corn and wheat) have considerably less lead than the outer more exposed parts such as stems, leaves, and husks. Processed foods have greater lead concentrations than unprocessed foods, due to lead inadvertantly added during processing. Foods packaged in soldered cans have much greater lead levels than foods packaged in other types of containers. Lead in inadequately glazed pottery may also be leached into foods, occasionally causing high exposures.

Significant amounts of lead in drinking water can result from contamination at the water source and from the use of lead plumbing in the water distribution system. Atmospheric deposition has been shown to increase lead levels in rivers, reservoirs, and other sources of drinking water; in many areas, however, lead pipes pose a more serious problem. Soft, acidic water in homes with lead plumbing may have excessive lead concentrations. Besides direct consumption of the water, exposure may occur when vegetables and other foods are cooked in water containing lead.

A number of manmade materials are known to contain lead, the most important being paint and plastics. Lead-based paints, although no longer used, are a major problem in older homes. Small children who ingest paint flakes can receive excessive lead exposure. Incineration of plastics may emit large amounts of lead into the atmosphere. Because of the increasing use of plastics, this source is likely to become more important. Other manmade materials containing lead include colored dyes, cosmetic products, candle wicks, and products made of pewter and silver.

All of the categories of potential lead exposure discussed above may influence or be influenced by dust and soil. For example, lead in street dust is derived primarily from vehicular emissions, while leaded house dust may originate from nearby stationary or mobile sources. Food and water may include lead adsorbed from soil as well as deposited atmospheric material. Flaking lead-based paint has been shown to increase soil lead levels. Natural concentrations of lead in soil average approximately 15 μg/g; this natural lead, in addition to authropogenic lead emissions, influences human exposure.

The final section of the chapter concerns occupational exposure to lead. The greatest exposures are found in the lead smelting and refining industries. Excessive airborne lead concentrations and dust lead levels are occasionally found in primary and secondary smelters; smaller exposures are associated with mining and processing of the lead ores. Welding and cutting of metal surfaces coated with lead-based paint may also result in excessive exposure. Other occupations with potentially high exposures to lead include the manufacture of lead storage batteries, printing equipment, alkyl lead, rubber products, plastics, and cans; individuals removing lead paint from walls and those who work in indoor firing ranges may also be exposed to lead.

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 Table 7-22. 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 Table 7-22 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 7.3.3.) 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 Table 7-22 may be greater than typical current exposures.

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.

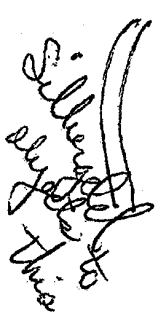

TABLE 7-22a. TWO HYPOTHETICAL ESTIMATES OF CONTRIBUTIONS OF SPECIFIC ROUTES OF EXPOSURE TO THE TOTAL ABSORPTION OF LEAD BY POPULATIONS OF URBAN CHILDREN

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	1.41	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).

TABLE 7-22b. 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).

For perspective, the environmental concentrations in Table 7-22 can be compared with estimated levels during natural, pre-industrial conditions. Such information is shown in Table 7-22. Note the wide concentration ranges, attributed to actual variation within each category as well as overall uncertainties in the estimates.

It is apparent from Tables 7-22 and 7-23 that the greatest increase in human exposure to lead since pre-industrial times has occurred in the food category. Unfortunately, the food concentrations given in Table 7-22 vary over three orders of magnitude; obtaining a quantitative estimate of the increase relative to natural exposures is difficult. Nevertheless, the natural concentration estimates suggest that modern lead exposures are considerably greater than pre-industrial exposures, a conclusion in agreement with the retrospective studies discussed earlier in the chapter.

TABLE 7-23. COMPARISON OF ESTIMATED NATURAL LEVELS OF LEAD IN THE ENVIRONMENT WITH TYPICAL PRESENT-DAY LEVELS

Medium	Estimated Natural Lead Concentrations	Typical Present-Day Lead Concentrations[a]	Approximate Ration, Present-Day/Natural
Air			
Rural/remote	0.01-0.1 ng/m^3	0.1-100 ng/m^3	10-1,000
Inhabited	0.1-1.0 ng/m^3	0.1-10 µg/m^3	100-10,000
Soil			
Rural/remote	5-25 µg/g	5-50 µg/m^3	1-2
Inhabited	5-25 µg/g	10-5,000 µg/g	2-200
Water			
Fresh	0.005-10 µg/l	0.005-10 µg/l	1
Ocean	0.001 µg/l	0.005-0.015 µg/l	10
Foods	0.0001-0.1 µg/g	0.01-10 µg/g	100

[a]See Cases I and II for basis for estimates of typical present-day levels.

need measures of central tendency

7.8 REFERENCES

Acra, A.; Dajani, R.; Raffoul, Z.; Karahagopian, Y. (1981) Lead-glazed pottery: A potential health hazard in the Middle East. Lancet 1: 433-434.

Akland, G. G. (1976) Air Quality Data for Metals, 1970 through 1974, from the National Air Surveillance Networks. U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, N.C. Publication No. EPA 600/4-76-041. 154 p.

Agrawal, Y. K.; Patel, M. P.; Merh, S. S. (1981) Lead in soils and plants: its relationship to traffic volume and proximity to highway (Lalbag, Baroda City). Int. J. Environ. Stud. 16: 222-224.

Altshuler, B.; Palmas, E. D,; Nelson, N. (1967) Regional Aerosol Deposition in the Human Respiratory Tract. In: Inhaled Particles and Vapours. C. N. Davies, ed., Pergamon Press, p. 323.

Anderson, K. E.; Fishbein, A.; Kestenbaum, D.; Sassa, S.; Alvares, A. P.; Kappas, A. (1977) Plumbism from airborne lead in a firing range. Am. J. Med. 63: 306-312.

Andresen, A. M.; Johnson, H. A.; Siccama, T. G. (1980) Levels of lead, copper and zinc in the forest floor in the northeastern United States. J. Environ. Qual. 9: 293-296.

Angle, C. R.; McIntire, M. S. (1979) Environmental lead and children: the Omaha study. J. Toxicol. Environ. Health 5: 855-870.

Arvik, J. H. (1973) Factors Affecting Uptake of Lead by Plants. Ph.D. Dissertation, Colorado State Univ., Fort Collins, Colo.

Badawy, A. A. B. (1978) Blood lead levels in mothers and their children. Lancet 1: 1363-1364.

Bailey, R. J.; Russell, P. F. (1981) Predicting drinking water lead levels. Environ. Technol. Let. 2: 57-66.

Baker, E. L., Jr.; Landrigan, P. J.; Barbour, A. G.; Cox, D. H.; Folland, D. S.; Ligo, R. N.; Throckmorton, J. (1979) Occupational lead poisoning in the United States: Clinical and biochemical findings related to blood lead levels. Br. J. Ind. Med. 36: 314-322.

Baltrop, D.; Thornton, I.; Strehlow, C. D.; Webb, J. S. (1975) Absorption of lead from dust and soil. Postgrad. Med. J. 51: 801-804.

Barltrop, D.; Strelow, C. D. (1976) Westway Nursery Testing Project. Report to the Greater London Council. August.

Barry, P. S. I.; Connolly, R. (1981) Lead concentrations in mediaeval bones. Int. Arch. Occup. Environ. Health 48: 173-177.

Beloian, A. (1982) Use of a food consumption model to estimate human contaminant intake. Environ. Monit. Assess. (in press).

Berg, B. A.; Zenz, C. (1967) Environmental and clinical control of lead exposure in a nonferrous foundry. J. Amer. Ind. Hyg. Assoc. 28(2): 175-178.

Berk, J. V.; Young, R. A.; Brown, S. R.; Hollowell, C.D. (1981) Impact of energy-conserving retrofits on indoor air quality in residential housing. 74th annual meeting, Air Pollut. Control Assoc., Philadelphia, PA, June 21-26, 1981.

Berlin, A.; Amavis, R.; Langevin, M. (1977) Research on Lead in Drinking Water in Europe. Commission of the European Economic Communities.

Bernstein, D. M.; Rahn, K. A. ( ) New York summer aerosol study: trace elements as a function of particle size. Ann. N.Y. Acad. Sci.

Bertagnolli, J. F.; Katz, S. A. (1979) Colored gift wrapping papers as a potential source of toxic metals. Int. J. Environ. Anal. Chem. 6: 321-326.

Bishop, J. R. (1980) Atmospheric lead and the related blood-levels of workers in high-speed can-making. Am. Ind. Hyg. Assoc. J. 41: 61-63.

Block, J. L. (1969) The accident that saved five lives. Good Housekeeping, November.

Boscolo, P.; Porcelli, G.; Cecchetti, G. et al. (1978) Urinary kallikrein activity of workers exposed to lead. Br. J. Ind. Med. 35: 226-229.

Boudene, C.; Arsac, F.; Meininger, J. (1975) Study of air and population levels in France. Arch. Hig. Rada. Tekisol. 26(Suppl.): 179-189.

Brandt, A. D.; Reichenbach, G. S. (1943) Lead exposures at the government printing office. J. Ind. Hyg. Toxicol. 25(10): 445-450.

Brandt, M.; Bentz, J. M. (1971) A cooperative study in the determination of lead in milk. Microchemical J. 16: 113-120.

Brooks, R. R.; Trow, J. M.; Bolviken, B. (1979) Biogeochemical anomalies in Fennoscandia: a study of copper, lead and nickel levels in Melandrium dioicum and Viscaria alphia. J. Geochem. Explor. 11: 73-87.

Brown, C. P.; Spivey, G. H.; Valentine, J. L.; Browdy, B. L. (1980) Cigarette smoking and lead levels in occupationally exposed lead workers. J. Toxicol. Environ. Health 6: 877-833.

Buss, D. H.; Lindsay, D.G. (1978) Reorganization of the UK Total Diet Study for monitoring minor constituents of food. Food Cosmet. Toxicol. 16: 597-600.

Cannon, H. L.; Bowles, J. M. (1962) Contamination of vegetation by tetraethyl lead. Science 137: 765-766.

Cawse, P. A. (1974) A Survey of Atmospheric Trace Elements in the U.K. (1972-73). Report AERE-R7669, Harwell Laboratories, United Kingdom, October.

Chamberlain, A. C.; Heard, M. J.; Little, P. L.; Newton, D.; Wells, A. C.; Wiffen, R. D. (1978) Investigations into Lead from Motor Vehicles, Report AERE-R9198, Harwell Laboratories, United Kingdom, November 1978.

Chan, T. L.; Lippmann, M. (1980) Experimental methods and empirical modeling of the regional deposition of inhaled particles in humans. Am. Ind. Hyg. Assoc. J. 41: 399-409.

Cholak, J.; Schafer, L.; Yeager, D. (1968) The air transport of lead compounds present in automobile exhaust gases. Am. Ind. Hyg. Assoc. J. 29: 562-568.

Clausen, J.; Rastogi, S. C. (1977) Heavy metal pollution among autoworkers. I. Lead. Br. J. Ind. Med. 34: 208-215.

Cohen, A. F.; Cohen, B. L. (1980) Protection from being indoors against inhalation of suspended particulate matter of outdoor origin. Atmos. Environ. 14: 183-184.

Committee on Lead in the Human Environment, National Research Council, National Academy of Sciences. (1980) Lead in the Human Environment. National Academy Press, Washington, DC.

Committee on Toxicology, National Research Council. (1976) Recommendations for the Prevention of Lead Poisoning in Children. Consumer Product Safety Commission, Washington, DC. p. 45, 51.

Cope, R. F.; Pancamo, B. P.; Rinehart, W. E.; Ter Haar, G. L. (1979) Personnel Monitoring for tetraalkyl lead in the workplace. Am Ind. Hyg. Assoc. J. 40: 372-379.

Crump, D. R.; Barlow, P. J. (19820 Factors controlling the lead content of a pasture grass. Environ. Pollut. (Series B) 3: 181-192.

Cubbon, R. C. P.; Roberts, W.; Marshall, K. The extraction of lead from ceramic tableware by foodstuffs. Trans. Br. Ceram. Soc. 80: (no page number).

Daines, R. H.; Motto, H.; Chilko, D. M. (1970) Atmospheric lead: relationship traffic volume and proximity to highways. Environ. Sci. Tech. 4: 318-323.

Darrow, D. K.; Schroeder, H. A.; (1973) Childhood exposure to environ-mental lead. Presented at American Chemical Society National Meeting, Chicago, Ill., August 29, 1973.

Davidson, C. I.; Goold, W. D.; Wiersma, G. A. (1982) Dry deposition of trace elements in Olympic National Park. Proceedings, Fourth International Confeence on Precipitation Scavenging, Dry Deposition, and Resuspension, Santa Monica, California, November 29-December 3, 1982.

Davidson, C. I.; Chu, L.; Grimm, T. C.; Nasta, M. A.; Qamoos, M. P. (1981) Wet and dry deposition of trace elements onto the Greenland ice sheet. Atmos. Environ. 15: 1429-1437.

Davidson, C. I. (1977) The deposition of trace metal-containing particles in the Los Angeles area. Powder Technol. 18: 117-126.

Davidson, C. I.; Goold, W. D.; Nasta, M. A.; Reilly, M. T. (1981a) Airborne size distributions of trace elements in an industrialized section of Pittsburgh. Air Poll. Control Assoc., 74th Annual Meeting, Philadelphia, PA.

Davidson, C. I.; Grimm, T. C.; Nasta, M. A. (1981b) Airborne lead and other elements derived from local fires in the Himalayas. Science 214: 1344-1346.

Davidson, C. I.; Nasta, M. A.; Reilly, M. T.; Suuberg, E. M. (1981a) Dry Deposition of Trace Elements in Great Smoky Mountains National Park. Part I. Airborne Concentrations and Size Distributions. Carnegie-Mellon University report, October.

Davies, B. E. (1978) Plant-available lead and other metals in British garden soils. Sci. Total Environ. 9: 243-262.

Davies, B. E.; Conway, D.; Holt, S. (1979) Lead pollution of London soils: a potential restriction on their use for growing vegetables. J. Atric. Sci. Camb. 93: 749-752.

Day, J. P.; Fergusson, J. E.; Chee, T. M. (1979) Solubility and potential toxicity of lead in urban street dust. Bull. Environ. Contam. Toxicol. 23: 497-502.

Day, J. P.; Hart, M.; Robinson, M. S. (1975) Lead in urban street dust. Nature 253: 343-345.

Dedolph, R.; Ter Haar, G.; Holtzman, R.; Lucas, H., Jr. (1970) Sources of lead in perennial ryegrass and radishes. Environ. Sci. Technol., 4(3): 217-223.

Dorn, C. R., Pierce, J. O.; Phillips, P. E.; Chase, G. R. (1976) Airborne Pb, Cd, Zn, and Cu concentration by particle size near a Pb smelter. Atmos. Environ. 10: 443-446.

Drill, S.; Konz, J.; Mahar, H.; Morse, M. (1979) The Environmental Lead Problem: An Assessment of Lead in Drinking Water from a Multi-Media Perspective. EPA-570/9-79-003 PB-296 556/4ST May.

Duce, R. A.; Ray, B. J.; Hoffman, G. L.; Walsh, P. R. (1976) Trace metal concentration as a function of particle size in marine aerosols from Bermuda. Geophys. Res. Lett. 3: 339-342.

Duggan, M. J.; Williams, S. (1977) Lead-in-dust in city streets. Sci. Total Environ. 7: 91-97.

Durfor, C. N.; Becker, E. (1964) Public Water Supplies of the 100 Largest Cities in the United States, 1962. U.S. Dept. of Interior, Geologic Survey, Washington, DC USGS Water Supply Paper No. 1812. 364 p.

Durum, W. H. (1971) Reconnaissance of Selected Minor Elements in Surface Waters of the U.S. U.S. Dept. of Interior, Geologic Survey, Washington, D.C. USGS Circular No. 643.

Dzubay, T. G.; Hines, L. E.; Stevens, R. K. (1976) Particle bounce errors in cascade impactors. Atmos. Environ. 10: 229-234.

Edwards, H. W. (1975) Environmental contamination by automotive lead. In: Inter. Symp. Proc. on Recent Advances in the Assessment of the Health Effects of Environmental Pollution, Vol. III. Paris, June 1974. Luxembourg, Commission of the European Communities. p. 1277-1286.

Elias, R. W.; Davidson, C. (1980) Mechanisms of trace element disposition from the free atmosphere to surfaces in a remote High Sierra canyon. Atmos. Environ. 14: 1427-1432.

Ericson, J. E., Shirahata, H.; Patterson, C. C. (1979) Skeletal concentrations of lead in ancient Peruvians. N. Engl. J. Med. 300: 946-951.

Faoro, R. B.; McMullen, T. B. (1977) National Trends in Trace Metals in Ambient Air, 1965-1974. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, Research Triangle Park, NC Pub. No. EPA-450/1-77-003. 41 p.

Federal Register (1981 September 3). 46: 44159-44172

Feldman, R. G. (1978) Urban lead mining: Lead intoxication among deleaders. N. Engl. Med. 298: 1143-1145.

Fischbein, A.; Thornton, J. C.; Berube, L.; Villa, F.; Selikoff, I. J. (1982) Lead exposure reduction in workers using stabilizers in PVC manufacturing: the effect of a new encapsulated stablilizer. Am. Ind. Hyg. Assoc. J. 43: 653-655.

Fischbein, A.; Daum, S. M.; Davidow, B.; Slavin, G.; Alvares, A. P.; Sassa, S.; Anderson, K. E.; Kappas, A.; Eisinger, J.; Blumberg, W. E.; Winicow, E. H.; Selikoff, I. J. (1978) Lead hazard among ironworkers. Dismantling lead-painted elevated subway line in New York City. N.Y. State J. Med. 78: 1250-1259.

Fischbein, A.; Rice, C.; Sarkozi, L; Kon, S. H.; Petrocci, M.; Selikoff, I. J. (1979) Exposure to lead in firing ranges. J. Am. Med. Assoc. 241: 1141-1144.

Fosse, G.; Wesenberg, G. B. (1981) Lead, cadmium, zinc and copper in deciduous teeth of Norwegian children in the pre-industrial age. Int. J. Environ. Stud. 16: 163-170.

Fritsch, A.; Prival, M. (1972) Response to the United States Environmental Protection Agency's notice for additional health effects information concerning the use of leaded gasoline. Center for Science in the Public Interest. Washington, DC. 11 p.

Garcia-Miragaya, J.; Castro, S.; Paolini, J. (1981) Lead and zinc levels and chemical fractionation in road-side soils of Caracas, Venezuela. Water Air Soil Pollut. 15: 285-297.

Garty, J.; Fuchs, C.; (1982) Heavy metals in the lichen Ramalina duriaei transplanted in biomonitoring stations. Water Air Soil Pollut. 17: 175-183.

General Electric Company (1972) Indoor-outdoot carbon monoxide pollution study. Final report EPA-RA-73-020, Contract CPA 70-77, Office of Research and Monitoring, USEPA, December 1972.

George, A. C.; Breslin, A. J. (1967) Deposition of natural radon daughters in human subjects. Health Phys. 13: 375.

Gilbert, C.; Tuthill, R. W.; Calabrese, E. J.; Peters, H. A. (1979) A comparison of lead hazards in the housing environment of lead poisoned children versus non poisoned controls. J. Environ. Sci. Health 14: 145-168.

Gillette, D. A.; Winchester, J. W. (1972) A study of aging of lead aerosols-I. Atmos. Environ. 6: 443-450.

Goold, W. D.; Davidson, C. I. (1982) Sources and Sinks of Airborne Trace Elements in Olympic National Park Biosphere Reserve, Carnegie-Mellon University report, August.

Gordon, G. E.; Moyers, J. L.; Rahn, K. A.; Gatz, D. F.; Dzubay, T. G.; Zoller, W. H.; Corrin, M. H. (1982) Atmospheric trace elements: cycles and measurements. Draft Report of the NSF Atmospheric Chemistry Workshop, NCAR, Boulder, Colorado, 1978. Manuscript in preparation.

Graben, N.; Wilhelms, W.; Kloppel, H. A.; Tiepermann, R. V.; Doss, M. (1978) Acute heavy lead poisoning in welders. J. Clin. Chem. Clin. Biochem. 16: 63.

Graham, D. L.; Kalman, M. (1974) Lead in forage grass from a suburban area in northern California. Environ. Pollut. 7: 209-215.

Grandjean, P.; Kon, S. H. (1981) Lead exposure of welders and bystanders in a ship repair yard. Am. J. Ind. Med. 2: 65-70.

Grandjean, P.; Nielsen, O. V.; Shapiro, I. M. (1979) Lead retention in ancient Nubian and contemporary populations. J. Environ. Pathol. Toxicol. 2: 781-787.

Greene, M. H.; Hoover, R. N.; Eck, R. L.; Fraumeni, J. F., Jr. (1979) Cancer mortality among printing plant workers. Environ. Res. 20: 66-73.

Gross, S. B. (1981) Human organ and inhalation exposures to lead: summary of Kehoe balance experiments. J. Toxicology and Environ. Health 8: 333-377.

Guinee, V. F. (1972) Lead poisoning. Am. J. Med. 52: 283-288.

Gulson, B. L.; Tiller, K. G.; Mizon, K. J.; Merry, R. M. (1981) Use of lead isotopes in soils to identify the source of lead contamination near Adelaide, Sout Australia. Environ. Sci. Technol. 15: 691-696.

H. M. Chief Insp. of Factories. H. M. Great Britain Dept. of Employment. Annual Report, 1971. London, Her Majesty's Stationery Office. 1972. p. 60, 95.

Halpern, M. (1978) Indoor/outdoor air pollution exposure continuity relationships. J. Air Pollut. Control Assoc. 28: 689-691.

Hammond, P. B.; Clark, C. S.; Gartside, P. S.; Berger, O.; Walker, A.; Michael, L. W. (1980) Fecal lead excretion in young children as related to sources of lead in their environments. Ind. Arch. Occup. Environ. Health 46: 191-202.

Hankin, L.; Heichel, G.; Botsford, D. (1973) Lead poisoning from colored printing inks. Clin. Pediatrics 12: 664-668.

Hankin, L.; Heichel, G. H.; Botsford, R. A. (1975) Lead content of pet foods. Bull. Environ. Contamin. and Toxicol. 13(5): 630-632.

Hankin, L.; Heichel, G. H.; Botsford, R. A. (1974) Lead on wrappers of specialty foods as a potential hazard for children. Clinical Pediatrics. 13(12): 1064-1065.

Hankin, L.; Heichel, G. H.; Botsford, R. A. (1976) Lead on painted handles of kitchen utensils. Clin. Pediat. 15(7): 635-636.

Hardy, M. L.; Chamberlin, R. I.; Boylen, G. W.; Howell, M. C. (1971) Lead as an environmental poison. Clinical Pharmacology and Therapeutics 12: 982-1002.

Harris, M. R. (1981) Distribution of lead in winter wheat plants grown in the vicinity of a lead smelter. Environ. Technol. Lett. 2: 243-252.

Harris, R. W.; Elsen, W. R. (1967) Ceramic glaze as a source of lead poisoning. J. Am. Med. Assoc. 202(6): 208-210.

Harrison, P. R.; Matson, W. R.; Winchester, J. W. (1971) Time variations of lead, copper and cadmium concentrations in aerosols in Ann Arbor, Michigan. Atmos. Environ. 5: 613-619.

Harrison, R. M. (1979) Toxic metals in street and household dusts. Sci. Total Environ. 11: 89-97.

Harrison, R. M.; Perry, R.; Slater, D. H. (1974) The contribution of organic lead compounds to total lead levels in urban atmospheres. In: Proc. Inter. Symp. on Recent Advances in the Assessment of the Health Effects of Environmental Pollution, Vol. III. Paris, June 24-28, 1974. Luxembourg, Commission of the European Communities. p. 1783-1788.

Heidam, N. (1982) Data published by Gordon et al.

Hem, J. D.; Durum, W. H. (1973) Solubility and occurrence of lead in surface water. J. Amer. Water Works Assn. 65(8): 562-568.

Ho, Y. G. (1979) Lead contamination in street dust in Hong Kong. Bull. Environ. Contam. Toxicol. 21: 639-642.

Hubermont, G.; Buchet, J. P.; Roels, H.; Lauwerys, R. (1978) Placental transfer of lead, mercury and cadmium in women living in a rural area. Importance of drinking water in lead exposure. Int. Arch. Occup. Environ. Health. 41: 117-124.

Huntzicker, J. J.; Friedlander, S. K.; Davidson, C. I. (1975) Material balance for automobile-emitted lead in Los Angeles Basin. Environ. Sci. Tech. 9: 448-457.

Jansen, S. J.; Carnow, B. W.; Namekata, T. (1978) Morton Grove Lead Study: An Investigation of the Contribution of Airborne Lead from Automobile Exhaust to Blood Lead levels in Suburban Children. PB-280 717/OST April.

Jaworowski, Z. (1968) Stable lead in fossil ice and bones. Nature. 217: 152-153, January 13, 1968.

Jaworowski, Z.; Bysiek, M.; Kownacka, L. (1981) Flow of metals into the global atmosphere. Geochim. Cosmochim. Acta 45: 2185-2199.

Johansson, T. B.; Van Grieken, R. E.; Winchester, J. W. (1976) Elemental abundance variation with particle size in north Florida aerosols, J. Geophys. Res. 81: 1039-1046.

Kehoe, R. A. (1961) The Harben Lectures. The metabolism of lead in health and disease. J. Roy. Inst. Publ. Health Hyg. 24: 81-97.

Kennedy, G. L., Jr. (1973) Blood and Tissue Lead Content Study Following Exposure of Male Albino Rats to Lead-Containing Expressway Dirt, Neighborhood Street Dirt, Lead Acetate, or Lead Oxide. Industrial Bio-Test Laboratories, Northbrook, Illinois. Research Report IBT No. E1733C. 16 p.

Kerin, Z. (1975) Relationship between lead content in the soil and in the plants contaminated by industrial emissions of lead aerosols. In: Hutchinson, T. C., ed. International conference on heavy metals in the environment. Symposium Proceedings Vol. II, Part 2. Toronto, Ontario, Canada, October 27-31, 1975. p. 487-502.

King, E.; Conchie, A.; Hiett, D.; Milligan, B. (1979) Industrial lead absorption. Ann. Occup. Hyg. 22: 213-239.

Klein, M.; Namer, R.; Harpur, E.; Corbin, R. (1970) Earthenware containers as a source of fatal lead poisoning. New England J. Med. 283(13): 669-672.

Kolbye, A. C.; Mahaffey, K. R.; Fiorino, J. A.; Corneliussen, P. C.; Jelinek, C. F. (1974) Food exposure to lead. Environ. Health Persp. 7: 65-74.

Kopp, J. F.; Kroner, R. C. (1967) Trace Metals in Waters of the United States; A Five-Year Summary of Trace Metals in Rivers and Lakes of the United States (Oct. 1, 1962 - Sept. 30, 1967). U.S. Dept. of Interior, Federal Water Pollution Control Administration, Cincinnati, Ohio. 218 p.

Landrigan, P. J.; Baker, E. L.; Feldman, R. G.; Cox, D. H.; Eden, K. V.; Orenstein, W. A.; Mather, J. A.; Yankel, A. J.; von Lindern, I. H. (1976) Increased lead absorption with anemia and slowed nerve conduction in children near a lead smelter. J. Pediatrics. 89(6): 904-910. Dec. 1976.

Landrigan, P. J.; Gehlbach, S. H.; Rosenblum, B. F.; Shoults, J. M.; Candelaria, R. M.; Barthel, W. F.; Liddle, J. A.; Smrek, A. L.; Staehling, N. W.; Sanders, J. D. F. (1975) Epidemic lead absorption near an ore smelter: The role of particulate lead. New Eng. J. Med. 292(3): 123-129.

Larssen, S. (1972) Norweigian Institute for Air Research. Data published by Gordon et al.

Lau, W. M.; Wong, H. M. (1982) An ecological survey of lead contents in road-side dusts and soils in Hong Kong. Environ. Res. 28: 39-54.

Laurer, G. R.; Albert, R. E.; Kneip, T. J.; Pasternack, B.; Strehlow, C.; Nelson, N.; Kent, F. S. (1973) The distribution of lead paint in New York City tenement buildings. Am. J. Public Health 63: 163-168.

Lazrus, A. L.; Lorange, E.; Lodge, J. P., Jr. (1970) Lead and other metal ions in U.S. precipitation. Environ. Sci. Tech. 4(1): 55-58.

Ledolter, J.; Tiao, G. C.; Graves, S. B.; Hsieh, J.; Hudak, G. B. (1979) Statistical Analysis of the Los Angeles Catalyst Study Data. EPA-600/4-79-070.

Lee, R. E.; Patterson, R. K.; Wagman, J. (1968) Particle-size distribution of metal components in urban air. Environ. Sci. Technol. 2: 288-290.

Lee, R. E., Jr.; Goranson, S. S.; Enrione, R. E.; Morgan, G. B. (1972) National air surveillance cascade impactor network. II: Size distribution measurements of trace metal components. Environ. Sci. Technol. 6: 1025-1030.

Lee, R. E.; Goranson, S. S.; Enrione, R. E.; Morgan, G. B. (1972) The NASN cascade impactor network, Part II. Size distribution measurements of trace-metal components. Environ. Sci. Tech. 6(12): 1025-1030.

Linch, A. L.; Wiest, E. G.; Carter, M. D. (1970) Evaluation of tetraalkyl lead exposure by personal monitor surveys. J. Amer. Ind. Hyg. Assoc. 31(2): 170-179.

Lippmann, M. (1977) Regional Deposition of Particles in the Human Respiratory Tract. In: Handbook of Physiology, Section 9: Reactions to Environmental Agents. D. H. K. Lee, H. L. Falk, and S. D. Murphy, eds., The American Physiological Society, Bethesda, MD. pp. 213-232.

Lippmann, M.; Albert, R. (1969) The effect of particle size on the regional deposition of inhaled aerosols in the human respiratory tract. Am. Ind. Hyg. Assoc. J. 30: 257.

Little, P.; Fleming, R. G.; Heard, M. J. (1981) Uptake of lead by vegetable foodstuffs during cooking. Sci. Total Environ. 17: 111-131.

Little, P.; Wiffen, R. D. (1978) Emission and deposition of lead from motor exhausts--II: airborne concentration particle size and deposition of lead near motorways. Atmos. Environ. 12: 1331-1341.

Little, P.; Martin, M. H. (1972) A survey of zinc, lead and cadmium in soil and natural vegetation around a smelting complex. Environ. Pollut. 3: 241-254.

Lombardo, L. V. (1973) The Public Interest Campaign, Washington, D.C. Personal Communication to Mr. Wm. D. Ruckelshaus, Administrator, United States Environmental Protection Agency. March 9, 1973.

Lundgren, D. A. (1970) Atmospheric aerosol composition and concentration as function of particle size and time. J. Air Poll. Control Assoc. 20: 603-607.

Mahaffey, K. R. (1978) Environmental exposure to lead, in the Biogeochemistry of Lead in the Environment, J. O. Nriagu, ed., Elsevier-North Holland Biomedical Press, Amsterdam.

Mahaffey, K. R.; Annest, J. L.; Barbano, H. E.; Murphy, R. S. (1979) Preliminary analysis of blood lead concentrations for children and adults: HANES II, 1976-1978. Proceedings Univ. Missouri 13th Annual Conf. pp. 37-51.

Maljkovic, J. (1971) A case of occupational poisoning with lead carbonate and stearate. Sigurnost u Pogonu. 13: 123-124.

Martens, C. S., Wesolowski, J. J.; Kaifer, R.; John, W. (1973) Lead and bromine particle size distributions in the San Francisco Bay area. Atmos. Environ. 7: 905-914.

McCabe, L. J. et al. (1970) Survey of community water supply systems. J. Amer. Water Works Assn. 62(11): 670.

McDonald, C.; Duncan, H. J. (1979) Particle size distributions of metals in the atmosphere of Glasgow. Atmos. Environ. 13: 977-980.

McKeague, J. A.; Wolynetz, M. S. (1980) Background levels of minor elements in some Canadian soils. Geoderma 24: 299-307.

McLean, R. O.; Shields, B. (1977) A study of factors causing changes in the lead levels of crops growing beside roadways. Environ. Pollut. 14: 267-273.

Merry, R. H.; Tiller, K. G.; De Vries, M. P. C.; Cartwright, B. (1981) Contamination of wheat crops around a lead-zinc smelter. Environ. Pollut. 2: 37-48.

Michell, D. G.; Aldous, K. M. (1974) Lead content of foodstuffs. Env. Hlth. Pers. 7: 59-64. May 1974.

Ministry of Agriculture, Fisheries and Food. (1972) Survey of Lead in Food. (Second Report). Appendix IV. Her Majesty's Stationery Office, London.

Moore, M. R. (1977) Lead in drinking water in soft water areas--health hazards. Sci. Total Environ. 7: 109-115.

Moore, M. R.; Goldberg, A.; Fyfe, W. M.; Richards, W. N. (1981) Maternal lead levels after alterations to water supply. Lancet 1: 203-204.

Moore, M. R.; Hughes, M. A.; Goldberg, D. J. (1979) Lead absorption in man from dietary sources. The effect of cooking upon lead concentrations of certain foods and vegetables. Int. Arch. Occup. Environ. Health. 44: 81-90.

Moore, M. R.; Meredith, P. A.; Goldberg, A. (1977) A retrospective analysis of blood-lead in mentally retarded children. Lancet 1: 717-719.

Moore, M. R.; Meredith, P. A.; Watson, W. S.; Sumner, D. J.; Taylor, M. K.; Goldberg, A. (1980) The percutaneous absorption of lead-203 in humans from cosmetic preparations containing lead acetate, as assessed by whole-body counting and other techniques. Food Cosmet. Toxicol. 18: 399-405.

Moschandreas, D. J.; Zabransky, J.; Pelton, D. J. (1981) Comparison of indoor and outdoor air quality, EPRI report EA-1733, Research project 1309, Palo Alto, CA, March 1981.

Motto, H. L.; Daines, R. H.; Chilko, D. H.; Motto, C. K. (1970) Lead in soils and plants: Its relationship to traffic volume and proximity to highways. Environ. Sci. Technol., 4(3): 231-238.

Murozumi, M.; Chow, T. J.; Patterson, C. C. (1969) Chemical concentrations of pollutant lead aerosols, terrestrial dusts, and sea salts in Greenland and Antarctic snow strata. Geochim. Cosmochim. Acta. (London). 33: 1247-1294.

National Academy of Sciences. (1972) Lead: Airborne Lead in Perspective. Washington, DC. p. 29.

Needleman, H. L.; Landrigan, P. J. (1981) The health effects of low level exposure to lead. Ann. Rev. Pub. Health 2: 277-2988.

Needleman, H. L.; Scanlon, J. (1973) Getting the lead out. New Eng. J. Med. 288(9): 466-467.

Needleman, H. L.; Davidson, I.; Sewell, E. M.; Shapiro, I.M. (1974) Subclinical lead exposure in Philadelphia school children: Identification by dentine lead analysis. New Eng. J. Med. 290(5): 245-248.

Ng, A.; Patterson, C. C. (1981) Natural concentrations of lead in ancient Arctic and Antarctic ice. Geochim. Cosmochim. Acta 45: 2109-2121.

Nutrition Foundation, Inc. (1982) Assessment of the safety f lead and lead salts in food: a report of the Nutrition Foundation's Expert Advisory Committee. Available from: The Nutrition Foundation, Washington, D. C.

Olsen, N. B.; Hollnagel, H.; Grandjean, P. (1981) Indicators of lead exposure in an adult Danish suburban population. Danish Med. Bull. 28: 168-176.

Palmer, K. T.; Kucera, C. L. (1980) Lead contamination of sycamore an soil from lead mining and smelting operations in eastern Missouri. J. Environ. Qual. 9: 106-111.

Parikh, D. J.; Pandya, C. B.; Ghodasara, N. B.; Ramanathan, N. L. (1979) Exposure of workers to inorganic lead in some small and medium industries. Indian Med. Res. 70: 116-124.

Pattenden, N. J. (1974) Atmospheric Concentrations and Deposition Rates of some Trace Elements Measured in the Swansea/Neath/Port Talbot Area. Report AERE-R7729, Harwell Laboratories, United Kingdom, March.

Patterson, C. C.; Settle, D. M. (1976) The reduction of order of magnitude errors in lead analyses of biological materials and natural waters by evaluating and controlling the extent and sources of industrial contamination introduced during sample collecting, handling, and analysis. Accuracy in Trace Analysis: Sampling, Sample Handling, Analysis. Natl. Bur. Stand. Spec. Publ. 422: 321-355.

PEDCO Environmental, Inc. (1981) Field study to determine spatial variability of lead from roadways. EPA Contract No. 68-02-3013. Available from:

PedCo-Environmental, Inc. (1977) Lead Analysis for Kansas City and Cincinnati. Report to U.S. Environmental Protection Agency, Research Triangle Park, N.C. PN3264-E. March 1977.

Peden, M. E. (1976) Flameless atomic absorption determinations of cadmium, lead, and manganese in particle size fractionated aerosols. Proceedings of the 8th IMR Symposium. Natl. Bur. Stand. Spec. Publ. 464: 367-377.

Pegues, W. L. (1960) Lead fume from welding on galvanized and zinc-silicate coated steels. J. Amer. Ind. Hyg. Assoc. 21(3): 252-255.

Pilegaard, K. (1978) Airborne metals and SO_2 monitored by epiphytic lichens in an industrial area. Environ. Pollut. 17: 81-92.

Pinkerton, C.; Creason, J. P.; Hammer, D. I.; Colucci, A. V. (1973) Multimedia Indices of Environmental Trace Metal Exposure in Humans. In: Proc. 2nd Inter. Symp. on Trace-Element Metabolism in Animals, Vol. 2. Madison, Wisconsin. p. 465-469.

Piver, W. T. (1977) Office of Health Hazard Assessment, NIEHS, Personal communication to H. L. Falk, Assoc. Dir. for Health Hazard Assessment, NIEHS, Research Triangle Park, NC.

Pocock, S. J. (1980) Factors influencing household water lead: A British national survey. Arch. Environ. Health 35: 45-51.

Quarles, H. D., III; Hanawalt, R. B.; Odum, W. E. (1974) Lead in small mammals, plants and soil at varying distances from a highway. J. Appl. Ecol. 11: 937-949.

Quarterly averages of lead from NAFN as of June 1982. From: NAFN, National Air Filter Network (data base). Research Triangle Park, NC; U.S. Environmental Protection Agency. Environmental Monitoring Laboratory. Printout. Available from: U.S. Environmental Protection Agency, Environmental Criteria and Assessment Office, Research Triangle Park, NC.

Rabinowitz, M. B. (1974) Lead Contamination of the Biosphere by Human Activity. Ph.D. Dissertation, University of California at Los Angeles.

Ragaini, R. C.; Ralston, H. R.; Roberts, N. (1977) Environmental trace metal contamination in Kellogg, Idaho, near a lead smelting complex. Environ. Sci. Technol. 11: 773-781.

Rameau, J. T. (1973) Lead as an Environmental Pollutant. In: Proc. Inter. Symp. on Environmental Health Aspects of Lead. Amsterdam, October 1972. Luxembourg, Commission of the European Communities. p. 189-200.

Ratcliffe, J. M. (1975) An evaluation of the use of biological indicators in an atmospheric lead survey. Atmos. Environ. 9: 623-629.

Richter, E. D.; Yaffe, Y.; Gruener, N. (1979) Air and blood lead levels in a battery factory. Environ. Res. 20: 87-98.

Rieke, F. E. (1969) Lead intoxication in shipbuilding and shipscrapping, 1941-1968. Arch. Env. Health. 19: 521-539.

Roberts, T. M. (1975) A review of some biological effects of lead emissions from primary and secondary smelters. In: Hutchinson, T. C., ed., International conference on heavy metals in the environment. Symposium Proceedings, Vol. II, Part 2. Toronto, Ontario, Canada, October 27-31, 1975. p. 503-531.

Rolfe, G. L.; Haney, A. (1975) An Ecosystem Analysis of Environmental Contamination by Lead. Inst. for Env. Studies, U. of Illinois, at Urbana-Champaign, Ill. Res. Rept. No. 1. p. 22-34.

Roy, B. R. (1977) Effects of particle sizes and solubilities of lead sulphide dust on mill workers. Am. Ind. Hyg. Assoc. J. 38: 327-332.

Ruhling, A.; Tyler, G. (1979) Ecology of heavy metals--a regional and historical study. Bot. Notis. 122: 248-259.

Ruhling, A.; Tyler, G. (1968) An ecological approach to the lead problem. Bot. Notis. (Stockholme). 121: 321-342.

Sakurai, H.; Sugita, M.; Tsuchiya, K. (1974) Biological response and subjective symptoms in low level lead exposure. Arch. Env. Hlth. 29: 157-163.

Sartor, F.; Rondia, D. (1980) Blood lead levels and age: A study in two male urban populations not occupationally exposed. Arch. Environ. Health 35: 110-116.

Scarlato, G.; Smirne, S.; Poloni, A. E. (1969) L'encefalopatia saturnina acta dell adulto. Acta Neurol. 24: 578-580.

Schaffner, R. M. (1981) Lead in canned foods. Food Technology 35: 60-64.

Schroeder, H. A.; Balassa. J. J. (1961) Abnormal trace metals in man: Lead. J. Chron. Dis. 14(4): 408-425.

Schroeder, H. A.; Tipton, I. H. (1968) The human body burden of lead. Arch. Environ. Health 17: 965-978.

Schuck, E. A.; Locke, J. K. (1970) Relationship of automotive lead particulates to certain consumer crops. Environ. Sci. Technol. 4(4): 324-330.

Settle, D. M.; Patterson, C. C. (1982) Magnitude and sources of precipitation fluxes of industrial and natural leads to the North Pacific at Enewetak. J. Geophys. Rev., 1982 (in press).

Settle, D. M.; Patterson, C. C. (1980) Lead in albacore: Guide to lead pollution in Americans. Science 207: 1167-1176.

Shanty, F. (1974) Deposition of Ultrafine Aerosols in the Respiratory Tract of Human Volunteers. Doctoral Dissertation. School of Hygiene and Public Health of the Johns Hopkins University, Baltimore, MD.

Shapiro, I. M.; Grandjean, P.; Nielsen, O. V. (1980) Lead levels in bones and teeth of children in ancient Nubia: Evidence of both minimal lead exposure and lead poisoning. In: Low Level Lead Exposure: The Clinical Implications of Current Research. H. L. Needleman, ed., Raven Press, New York. pp. 35-41.

Shapiro, I. M.; Needleman, H. L.; Dobkin, B.; Tuncay, O. C. (1973) Lead levels in denture and circumpulpal dentine of deciduous teeth of normal and lead exposed children. Clin. Chem. Acta. 46(2): 119-123.

Shearer, S. D.; Akland, G. G.; Fair, D. H.; McMullen, T. B.; Tabor, E. C. (1972) Concentrations of particulate lead in the ambient air of the United States. Paper presented at Public Hearing on Gasoline Lead Additives Regulations. Los Angeles, May 2-4.

Sherlock, J.; Smart, G.; Forbes, G. I.; Moore, M. R.; Patterson, W. J.; Richards, W. N.; Wilson, T. S. (1982) Assessment of lead intake and dose-response for a population in Ayr exposed to a plumbosolvent water supply. Human Toxicol. 1: 115-122.

Sinn, W. (1980) Uber den zusammenhang von liftbleikonzentration und bleigehalt des blutes von anwohnern und berufstatigen im kerngebiet einer grosstadt (Blutbleistudie Frankfurt). I. Versuchsanlage und differenzprufung. Int. Arch. Occup. Environ. Health 47: 93-118.

Smart, G. A.; Warrington, M.; Evans, W. H. (1981) Contribution of lead in water to dietary lead intakes. J. Sci. Food Agric. 32: 129-133.

Smith, D. L. (1976) Lead absorption in police small-arms instructors. J. Soc. Occup. Med. 26: 139-140.

Spielholtz, G. I.; Kaplan, F. S. (1980) The problem of lead in Mexican pottery. Talanta 27: 997-1000.

Spivey, G. H.; Brown, C. P.; Baloh, R. W.; Campion, D. S.; Valentine, J. L.; Massey, F. J., Jr.; Browdy, B. L.; Culver, B. D. (1979) Subclinical effects of chronic increased lead absorption--a prospective study. I. Study design and analysis of symptoms. J. Occup. Med. 21: 423-429.

Stahlhofen, W.; Gebhart, J.; Heyder, J. (1980) Experimental determination of the regional deposition of aerosol particles in the human respiratory tract. Amer. Ind. Hyg. Assoc. J. 41: 385-398a.

Stark, A. D.; Quah, R. F.; Meigs, J. W.; DeLouise, E. R. (1982) The relationship of environmental lead to blood-lead levels in children. Environ. Res. 27: 372-383.

Stephens, R. (1981) Human exposure to lead from motor vehicles emissions. Int. J. Environ. Stud. 17: 73-83.

Tabershaw, I. R.; Ruotolo, B. P. W.; Gleason, R. P. (1943) Plumbism resulting from oxyacetylene cutting of painted structural steel. J. Ind. Hyg. Toxicol. 25(5):189-191.

Tanaka, J.; Ichikuni, M. (1982) Monitoring of heavy metals in airborne particles by using bark samples of Japanese cedar collected from the metropolitan region of Japan. Atmos. Environ. 16: 2015-2018.

Tepper, L. B., cited in B. G. King. (1971) Maximal daily intake of lead without excessive body lead-burden in children. Am. J. Dis. Child. 122: 337.

Ter Haar, G. (1970) Air as a source of lead in edible crops. Environ. Sci. Technol., 4(3):226-229.

Ter Haar, G. (1974) Proceedings, International Symposium on Recent Advances in the Assessment of the Health Effects of Environmental Pollution. Paris, June 24-28.

Ter Haar, G. (1979) Sources of lead in children. In: International Conference Management and Control of Heavy Metals in the Environment, London, September 1979. Commission of the European Communities, and others, sponsors, CEP Consultants, Ltd., Edinburgh, United Kingdom. pp 70-76.

Ter Haar, G.; Aronow, R. (1974) New information and lead in dirt and dust as related to the childhood lead problem. Environ. Hlth. Persp. 7: 83-90.

Thalacker, V. R. (1980) Untersuchungen an hessischen bieren. VII. Mitteilung: Ermittlung des gehalts einiger spurenelemente in untergarigen vollbieren. Monatsschrift fur brauerei 33: 401-405.

Thomas, H. F. (1980) Domestic water usage and blood levels. Public Health London 94: 294-295.

Thomas, H. F.; Elwood, P. C. (1978) "First flush" water lead. Lancet 2: 109-110.

Thomas, H. F.; Elwood, P. C.; Welsby, E.; St. Leger, A. S. (1979) Relationship of blood lead in women and children to domestic water lead. Nature 282: 712-713.

Tolan, A.; Elton, G. A. H. (1973) Lead intake from food. In: Proc. Inter. Symp. on Environmental Health Aspects of Lead. Amsterdam, October 1972. Commission of the European Communities, Luxembourg. p. 77-84.

Tosteson, T. D.; Spengler, J. D.; Weker, R. A. (1981) Aluminum, iron, and lead concentrations of personal, indoor, and outdoor respirable particles. International Symposium on Indoor Air Pollution, Health, and Energy Conservation. University of Massachusetts, Amherst, MA, October 1981.

Tsuchiya, K.; Harashima, S. (1965) Lead exposure and the derivation of maximum allowable concentrations and threshold limit values. Brit. J. Ind. Med. 22(3): 181-186.

Tyler, R. L. (1970) Philadelphia combats "silent epidemic" in the ghetto, lead poisoning control. J. Environ. Health 33: 64-71.

U.S. Energy Research and Development Administration. Motor Gasolines. Bartlesville Energy Research Center. Bartlesville, Oklahoma.

U. S. Environmental Protection Agency (1973) Position on the Health Implications of Airborne lead. Office of Health and Ecological Effects. Washington, DC National Technical Information Service, Springfield, Va. Pub. No. NTIS PB 228 594/8.

U. S. Environmental Protection Agency. (1975) Chemical Analysis of Interstate Carrier Water Supply Systems. Research Triangle Park, NC Pub. No. EPA 430/9-75-005. 88 p.

U.S. Environmental Protection Agency (1978) Air Quality Data for Metals 1975 from the National Air Surveillance Networks. Office of Research and Development, Research Triangle Park, NC. Pub. No. EPA-600/4-78-059.

U.S. Environmental Protection Agency (1979) Air Quality Data for Metals 1976 from the National Air Surveillance Networks. Office of Research and Development, Research Triangle Park, NC. Pub. No. EPA 600/4-79-054.

U.S. Environmental Protection Agency. (1976) Control Techniques for Lead Air Emissions. (Draft final report.) EPA Contract No. 68-02-1375, Cincinnati, Ohio. 424 p.

U.S. Environmental Protection Agency. (1977) Second Annual Catalyst Research Program Report: Supplement II. Office of Research and Development, Research Triangle Park, NC. p. 359-466.

U.S. Environmental Protection Agency. (1979) Los Angeles Catalyst Study, EPA-600/4-79-033.

U.S. Environmental Protection Agency. (1981) Air Quality Criteria for Particulate Matter and Sulfur Oxides.

U.S. Food and Drug Administration. (1975) Compliance Program Evaluation, FY-1974, Heavy Metals in Foods Survey. Bureau of Foods. Washington, DC., 99 p.

U.S. Food and Drug Administration. (1979) Administrative Guidelines Manual, Transmittal No. 79-3, April 23.

United Kingdom Department of the Environment (1982) The Glasgow Duplicate Diet Study (1979/80). Pollution Report No. 11.

Vaughn, D. A.; Ifeadi, C.; Markle, R. A.; Krause, H. H. (1975) Environmental Assessment of Future Disposal Methods for Plastics in Municipal Solid Waste. U.S. Environmental Protection Agency, National Environmental Research Center. Cincinnati, Ohio. Pub. No. EPA-670/2-75-058. 86 p.

Welch, W. R.; Dick, D. L. (1975) Lead concentrations in tissues of roadside mice. Environ. Pollut. 8: 15-21.

Wheeler, G. L.; Rolfe, G. L. (1979) The relationship between daily traffic volume and the distribution of lead in roadside soil and vegetation. Environ. Pollut. 18: 265-274.

Wigle, D. T.; Charlebois, E. J. (1978) Electric kettles as a source of human lead exposure. Arch. Environ. Health 33: 72-78.

Winegar, D. A.; Levy, B. S.; Andrews, J. S., Jr.; Landrigan, P. J.; Scruton, W. H.; Karuse, M. J. (1977) Chronic occupational exposure to lead: An evaluation of the health of smelter workers. J. Occup. Med. 19: 603-606.

Wong, M. H.; Tam, F. Y. (1978) Lead contamination of soil and vegetables grown near motorways in Hong Kong. J. Environ. Sci. A13: 13-22.

World Health Organization. (1977) Environmental Health Criteria. Vol. 3., Lead. United Nations Environment Program Geneva, Switzerland. p. 59-65.

Yankel, A. J.; von Lindern, I. H.; Walter, S. P. (1977) The Silver Valley lead study: The relationship between childhood blood lead levels and environmental exposure. J. Air Poll. Contr. Assoc. 27(8): 763-767.

Yocum, J. E. (1982) Indoor-outdoor air quality relationships: a critical review. J. Air Pollut. Control Assoc. 32: 500-520.

Yocum, J. E.; Clink, W. L.; Cote, W. A. (1971) Indoor/outdoor air quality relationships. J. Air Pollut. Control Assoc. 21: 251.

Yu, C. P. (1978) A two compartment theory of aerosol deposition in human lung airways. Bull. Math. Biol. 40: 693-706.

Ziegler, E. E.; Edwards, B. B.; Jensen, R. L.; Mahaffey, K. R.; Fomon, S. D. (in press) Absorption and retention of lead by infants. Pediatric Research.

Ziegler, E. E.; Edwards, B. B.; Jensen, R. L.; Mahaffey, K. R.; Fomon, S. J. (1978) Absorption and retention of lead by infants. Pediatr. Res. 12: 29-34.

Zurlo, N.; Griffini, A. M. (1972) Lead contents in food and beverages consumed in Milan. In: Proc. Inter. Symp. on Environmental Health Aspects of Lead, Amsterdam, Oct. 1972. Luxembourg, Commission of the European Communities. p. 93-98.