TOXICANT PRETREATMENT PLANNING STUDY
TECHNICAL REPORT C1:

PRESENCE, DISTRIBUTION AND FATE OF TOXICANTS
IN PUGET SOUND AND LAKE WASHINGTON

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Since PCB levels are high in Elliott Bay and the aroclor ratio is distinct, it may be possible to identify the influence of Elliott Bay on the concentration of PCBs in seafood samples. PCB levels were found to be relatively low (less than 0.08 ppm fresh wt.) in crab tissue from the five collection sites, but the sample from Elliott Bay had noticeably higher concentrations and is the only sample with a higher level of aroclor 1260. Of the 5 resident chinook salmon analyzed, the one from Elliott Bay had the highest total PCB tissue level (1.8 ppm fresh wt.) and a higher level of aroclor 1260. The other four salmon were from near Richmond Beach and the fish with the highest PCB level (0.6 ppm fresh wt.) there also had a distinctively higher level of aroclor 1260. The fish with the lowest concentration (0.05 ppm fresh wt.) had far less aroclor 1260 than aroclor 1254.

Further work is needed to verify these kinds of trends. If substantiated, such ratio studies could be a useful tool in assessing sources of toxicants of concern.

7.3.3 Historic Trends

An important aspect in assessing the current and future quality of Puget Sound sediment is to determine how the present ambient concentrations compare with true background values before urbanization occurred in the region. In this study, the historic information was obtained by collecting a series of sediment cores from the deep central basin and analyzing sections of each core for metals and organic compounds. The deeper each section was in the core, the older the sediment it contained. The approximate time period that each section represents was determined by radioactive and stable lead dating techniques discussed in Appendix 4. By combining the information obtained from the box cores (50 cm deep), gravity cores (140 cm deep) and Kasten cores (250 cm deep), a comparison could be made between concentrations during the pre-industrial/urbanization period and current ambient levels in surface sediments. Locations where these three type cores were collected are shown in Figures 46 and 47.

Metal concentrations measured in the top and bottom segments of the nine box cores and three gravity cores analyzed in this study are listed in Table 26. The box cores showed only poorly defined, if any, differences between surface and buried sediments. The fairly uniform concentration in these 40-50 cm deep box cores is readily apparent from the sediment profiles shown in Figures 48 through 50. Only Mn showed a consistent change in trace metal concentration with depth in the sediment column; this change primarily reflects geochemical behavior (oxidation/reduction conditions) rather than temporal changes in input rates. The longer gravity cores, on the other hand, (Figures 51 through 53) show core profiles that consistently have lower sediment concentrations at depth, which demonstrates that enrichment has occurred for these metals above their pre-industrial baseline values. Another important trend apparent from the gravity core profiles is that, with the exception of Ag (at all sites sampled) and Hg (at the deep zone off urban Seattle), it appears that the levels of metals in sediments have decreased in recent years.

An accurate age determination was not possible with box cores so Kasten cores were used to establish the age of sediment layers and to verify trends. A complete listing of the data for the Kasten cores collected in this study is included in Appendix 4. The metals data have been summarized into the four sediment profiles shown in Figure 54. This figure depicts the chronology of Pb, Cu, Ag and Hg concentrations for each decade in terms of the Fractional Maximum Enrichment (FME). This quantity is defined as:

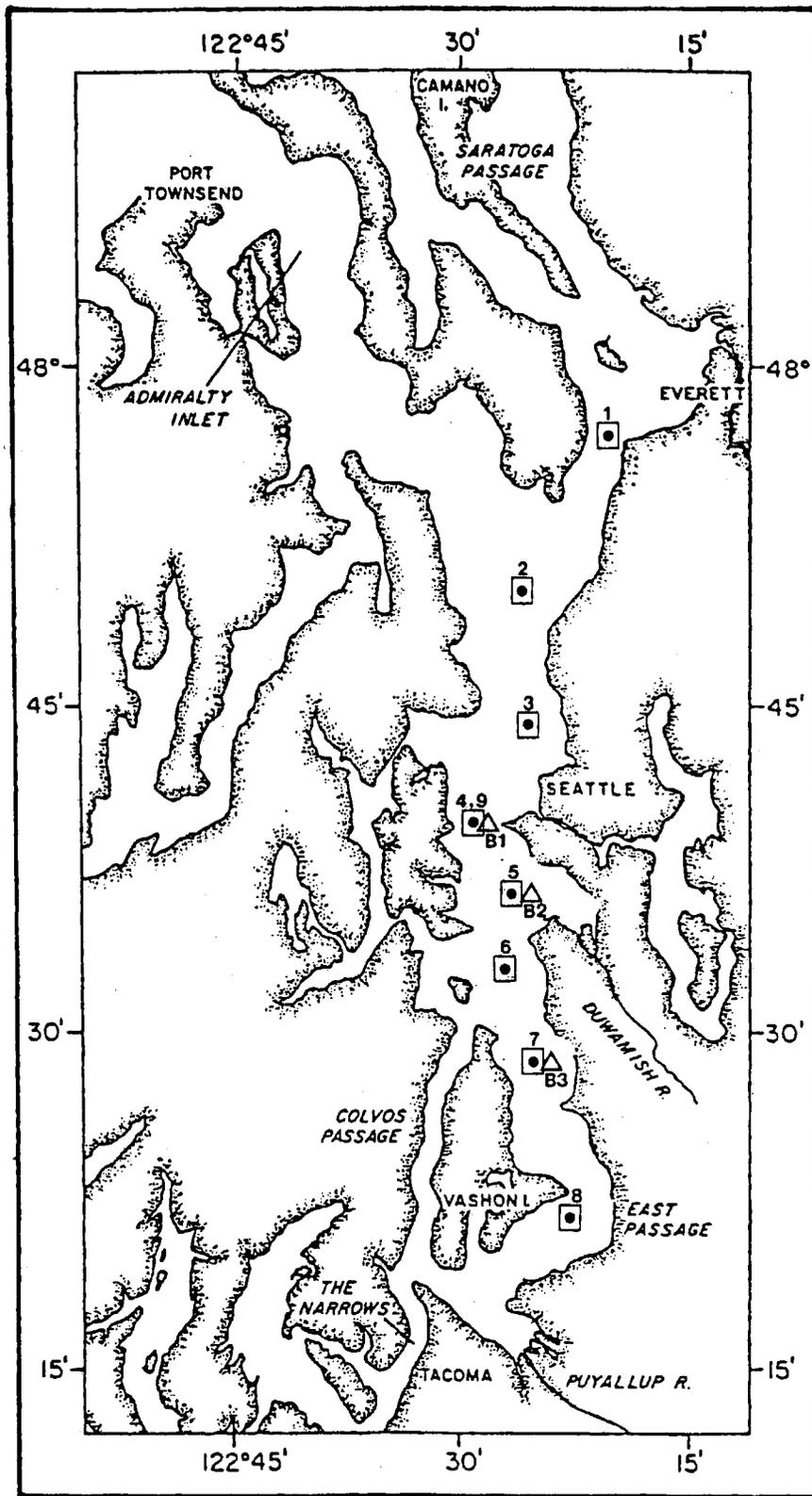$$\text{FME} = \frac{C_{X,T} - C_{X,B}}{C_{X,MAX} - C_{X,B}}$$

Figure 46. LOCATION OF BOX CORE AND GRAVITY CORE STATIONS IN
THE MAIN BASIN OF PUGET SOUND.
  ▣ - Box cores; 9 collected in 1980; 9 is a duplicate
       of 4.
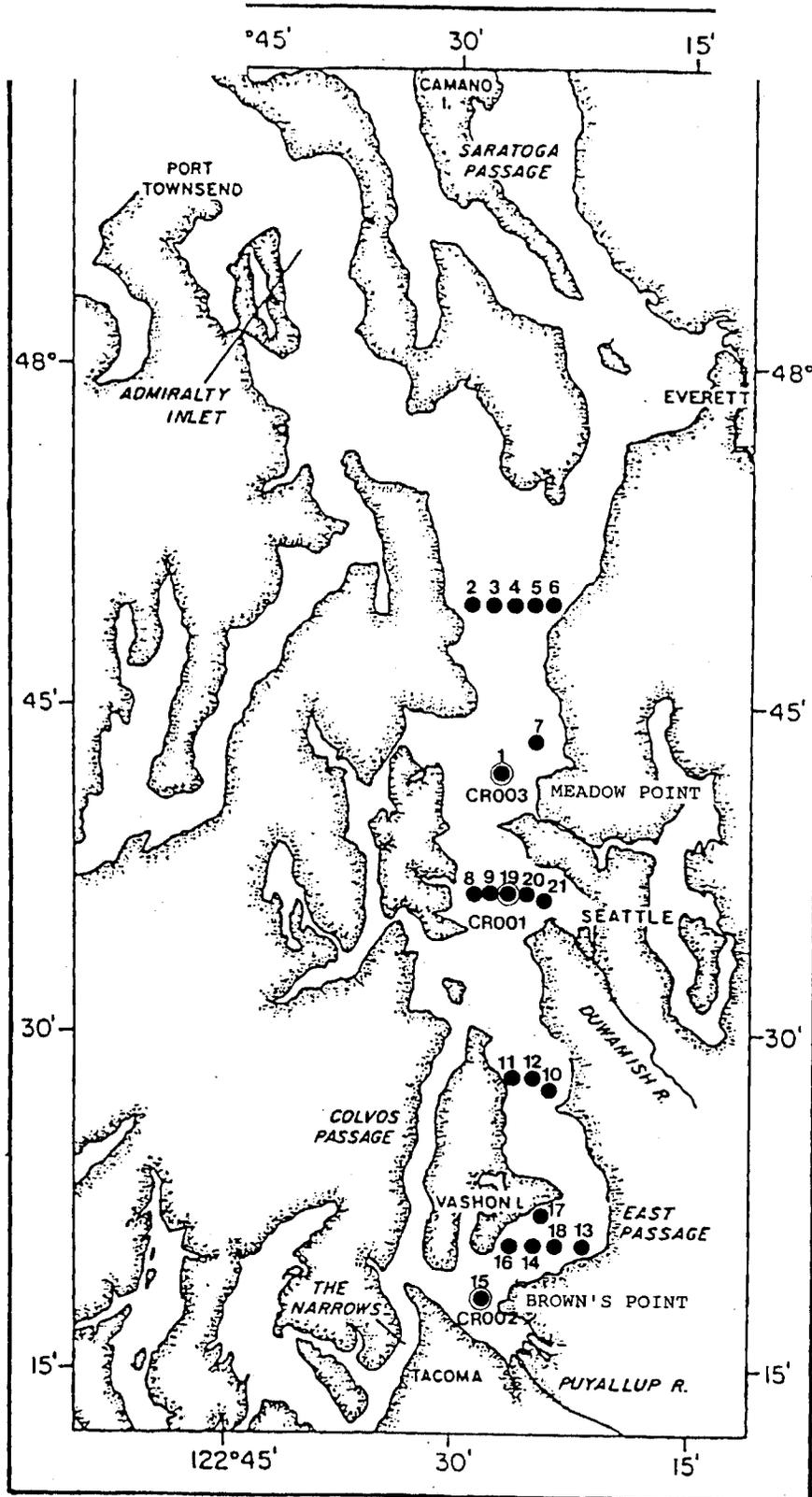 Δ - Gravity cores; 3 collected in 1981.

Figure 47.   LOCATIONS OF KASTEN CORE STATIONS IN THE MAIN BASIN
   OF PUGET SOUND.   (20 cores were collected in 1982; 1 box
   core was also collected at station 17.   Cores 1, 15, and 19
   were analyzed for trace organic priority pollutants at
   9 depth increments from 0 cm to 140 cm.

where $C_{X,T}$ is the concentration of X at a given age T, $C_{X,MAX}$ is the maximum concentration observed throughout the length of the core, and $C_{X,B}$ is the background concentration (pre-industrial period). Estimated dates were calculated based on depth in the sediment column as given in Appendix 4. The concentration values for each time increment is the mean of between 6 and 51 values taken from 14 cores. The error bars represent plus and minus one standard deviation from the mean FME for the decade. The FME quantity was recently develoepd by Crecelius and Bloom (unpublished results) to eliminate the effect of sediment texture variations on concentration when cores of different sediment types are compared.

The apparent decrease in Pb and Cu concentrations in recent years suggests a reduction in anthropogenic inputs of these metals and may reflect the results of regulatory control. For example, the decrease in Pb may be due to the elimination of Pb in gasoline mixtures and the reduction of emissions from the ASARCO smelter. The decrease of Hg levels may be attributable to the regulation of direct industrial dischargers in Puget Sound such as the Georgia Pacific chloro-alkali plant in Bellingham, or to a reduction in Hg mining in the Cascades. While Ag and Cu showed no decrease in surface sediment concentrations they do show a leveling off. It is also interesting to note that for Ag, sediment enrichment appears to have lagged behind the other elements by about 30-50 years even though the rate of increase (over the range of 30 to 70 percent of maximum enrichment) appears similar to the other metals.

The decrease of FME for Pb and Hg in the cores was tested for statistical significance using a t-test. The mean sediment concentrations of Pb and Hg in the 1970s are significantly lower (99% confidence level) than in sediments deposited in the 1950s. Possible reasons for this decrease (other than decreased input of these metals to the Sound) are dilution of the metals in the surface sediments by salt from pore water or change in bulk chemistry of the sediments. The FME profiles for several cores were calculated for salt-free sediment. This decreased the FME by not more than 3% since the salt content of the cores changes little with depth. For example, in core 12 the salt content of 0-5 cm sediment was 8% of dry sediment while at 95-100 cm it was 5%.

The bulk chemistry of the sediments does not change significantly with depth as indicated by the concentrations of Al and Fe in the gravity cores (see Appendix 4). Obtaining a ratio of a trace metal to a crustal metal will not change the relative enrichment.

To obtain some perspective on whether historic trends are consistent for different sediment types, profiles were developed to show the trends for Pb in three kinds of sediment: muddy sand, sandy mud, and mud. These profiles, shown in Figure 55, are based on the estimated load of Pb being deposited each decade in the bottom areas of Puget Sound that correspond to each sediment type. The specific loading values were estimated by calculating the average Pb concentration in each sediment type over a ten year period and then multiplying by the appropriate amount of bottom area as outlined in Appendix 4. The profiles for mud and sandy mud both show similar trends and due to their magnitude dominate the trend for total loading. These profiles all resemble the pattern seen in the Fractional Maximum Enrichment and indicate that the loading of Pb has decreased from the 1960s to the 1970s. Only the coarser-textured sediments of muddy sand show a different trend in the loading profile. This sediment type appears to reflect a much slower, more consistent increase in loading. While muddy sand represents the largest bottom area in the deep basin, the Pb concentrations are much lower in this sediment type as reflected by the low loading values. Due to these lower values and slow rate of change each decade, it will be more difficult to distinguish a decrease in the Pb loading in muddy sand.

Lead — Total Load Approach

| Area: | | 16.1x10⁷m² | 10.3x10⁷m² | 11.3x10⁷m² | | 37.7x10⁷m² |

Area: $16.1 \times 10^7 m^2$ $10.3 \times 10^7 m^2$ $11.3 \times 10^7 m^2$ $37.7 \times 10^7 m^2$

\# data points: 6   4   10

Type: Muddy Sand   Sandy Mud   Mud   Total Load

Legend:
- —— Muddy Sand
- ▪▪▪ Sandy Mud
- ▪▪▪▪▪▪ Mud
- •••• Accumulated Total

(Vertical axis: Age in decades corresponding to depth in core, 1850–1980; Horizontal axis: Metric tons Pb per decade, 0–700)
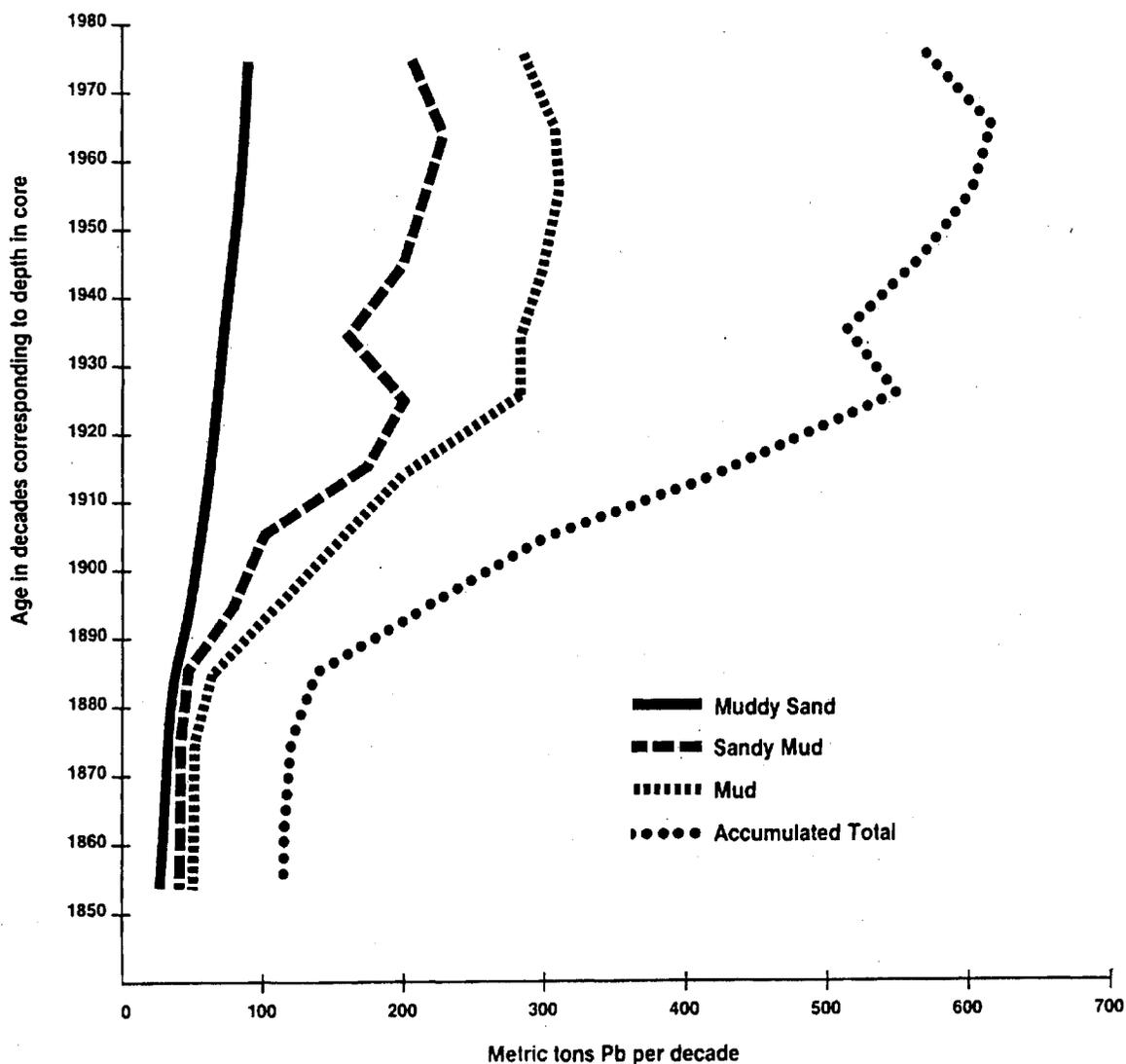
Figure 55. PROFILES OF LEAD LOADING IN 3 DIFFERENT TYPES OF
PUGET SOUND DEEP BASIN SEDIMENT: MUDDY SAND, SANDY MUD,
AND MUD (as defined by Roberts, 1979). Data for the
different sediments were derived from 6, 4, and 10 Kasten
cores, respectively, and represent bottom areas of 16.1 x
$10^7 m^2$, 10.3 x $10^7 m^2$ and 11.3 x $10^7 m^2$.

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The historic trends for the input of organic priority pollutants were evaluated at three Kasten core locations:

BPS-1 (CR003, off Meadow Point),
BPS-19 (CR001, off Elliott Bay), and
BPS-15 (CR002, off Brown's Point).

Nine sections of each core were analyzed and the raw data are presented in Appendix 4. Profiles of the dominant organic classes are presented in Figures 56-58. It is important to note that due to the limited number of samples and variability between cores, the apparent trends are only suggested by the data, but cannot be demonstrated statistically.

The concentrations of PAHs and CPAHs were substantially enriched above background levels in the late 1800s and early 1900s, prior to the appearance of significant concentrations of the other organic groups. The appearance of PAHs and CPAHs is coincident with increased industrialization and the increased use of fossil fuels. Increases in the concentrations of PAHs and CPAHs during approximately the same period were previously noted in Washington by Prahl and Carpenter (1979) and off Rhode Island by Hites et al. (1977). Since 1950, there has been a general decrease in concentrations of PAHs and CPAHs in Puget Sound sediments. There was some evidence of a recent rise in concentrations though this increase was evident only in one core sample off Meadow Point (Figure 56) and may not be representative of the Puget Sound area as a whole.

The phthalate esters were first commercially produced in the 1930s, but were found in all Puget Sound sediments at all depths examined, including those depths corresponding to the 1800s (Figure 57). A similar observation was made in Chesapeake Bay (Peterson and Freeman, 1982) and attributed to biological reworking of the sediments and the resultant mixing with upper sediment layers. The same mechanism probably accounts for the occurrence of phthalates in deeply buried sediments of Puget Sound. The first enrichment of phthalate concentrations above background levels was evident in Puget Sound sediments of the 1940s and 1950s. This increase was attributable solely to the one sample off Meadow Point which exhibited an enrichment in the 1940s three times that of any other period. This evidence suggests that a localized source of phthalates was present at that time but is no longer contributing significant quantities of phthalates to the Puget Sound system. The most dramatic enrichment of phthalates occurred between 1963 and 1974, evident only in the sample off Brown's Point. The rapid return of the Brown's Point sediment phthalate concentrations to levels comparable to the other sites by 1979 suggests either a high input of short duration or an analytical error or contamination in the measurement of phthalates at the depth interval corresponding to the mid 1970s. Ignoring the anomalous values, there appears to be a small but noticable increase in the input of phthalates into Puget Sound over the past decade (Figure 57).

Temporal trends of PCBs and DDTs were similar (Figure 58), with both groups exhibiting rapid increases in concentration in the 1930s and 1940s. Peak concentrations were obtained in sediments corresponding to 1950 and 1960 for PCBs and DDTs, respectively. Both groups showed slight decreases in their concentrations in sediments deposited since those times. It is noteworthy that enrichment of DDTs in surface sediments was greater than in buried sediments corresponding to the mid 1970s, the time its use was banned in the Northwest (Peakall, 1976). This can be attributed either to bioturbation of the upper sediment layers or continued input of DDTs into Puget Sound by terrestial runoff.
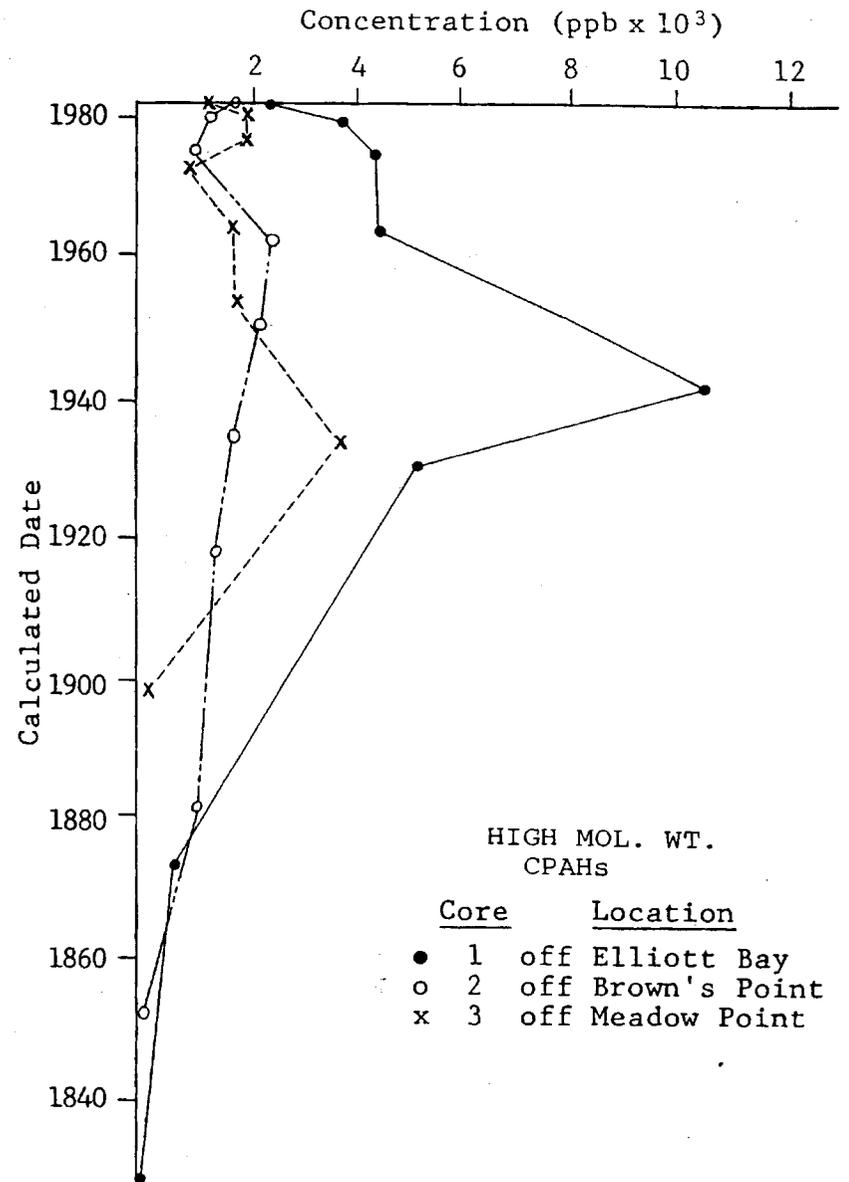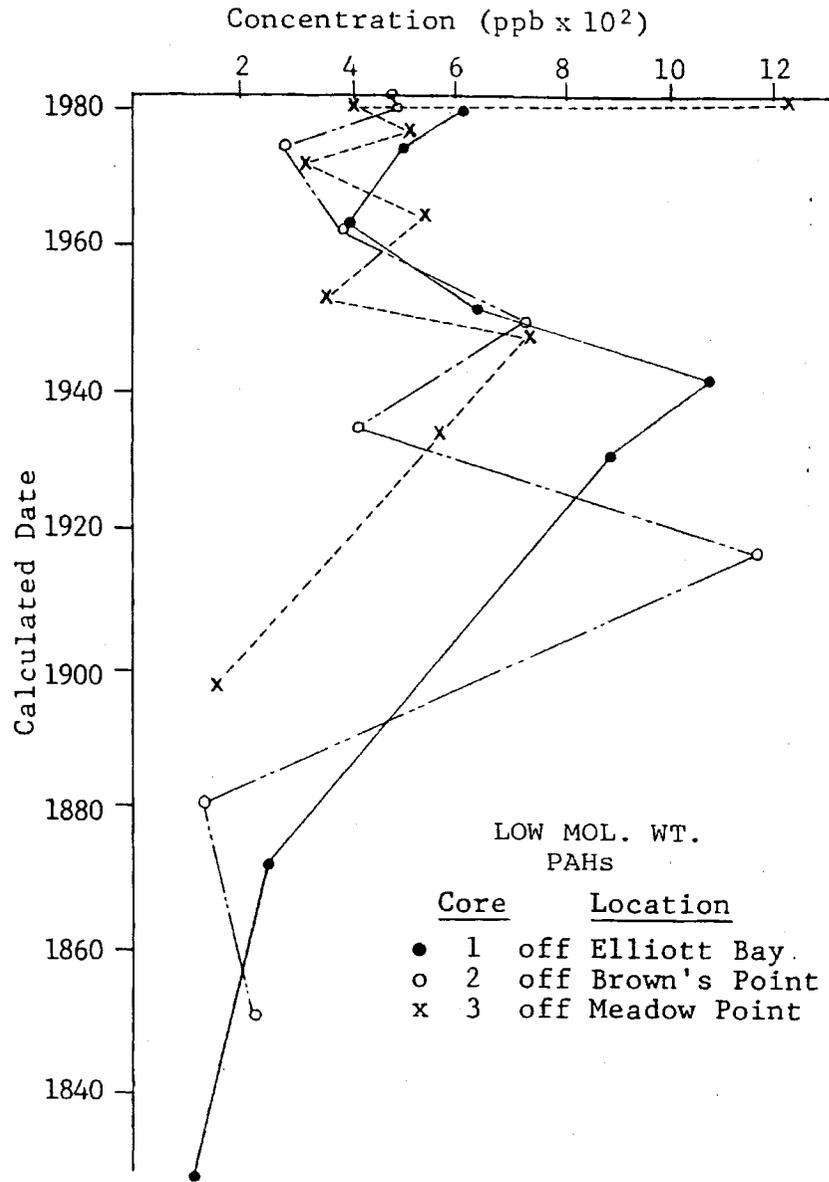
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Figure 56. SEDIMENT PAH CONCENTRATION PROFILES FROM THREE
CENTRAL BASIN KASTEN CORES. SEDIMENT DEPTH EXPRESSED IN
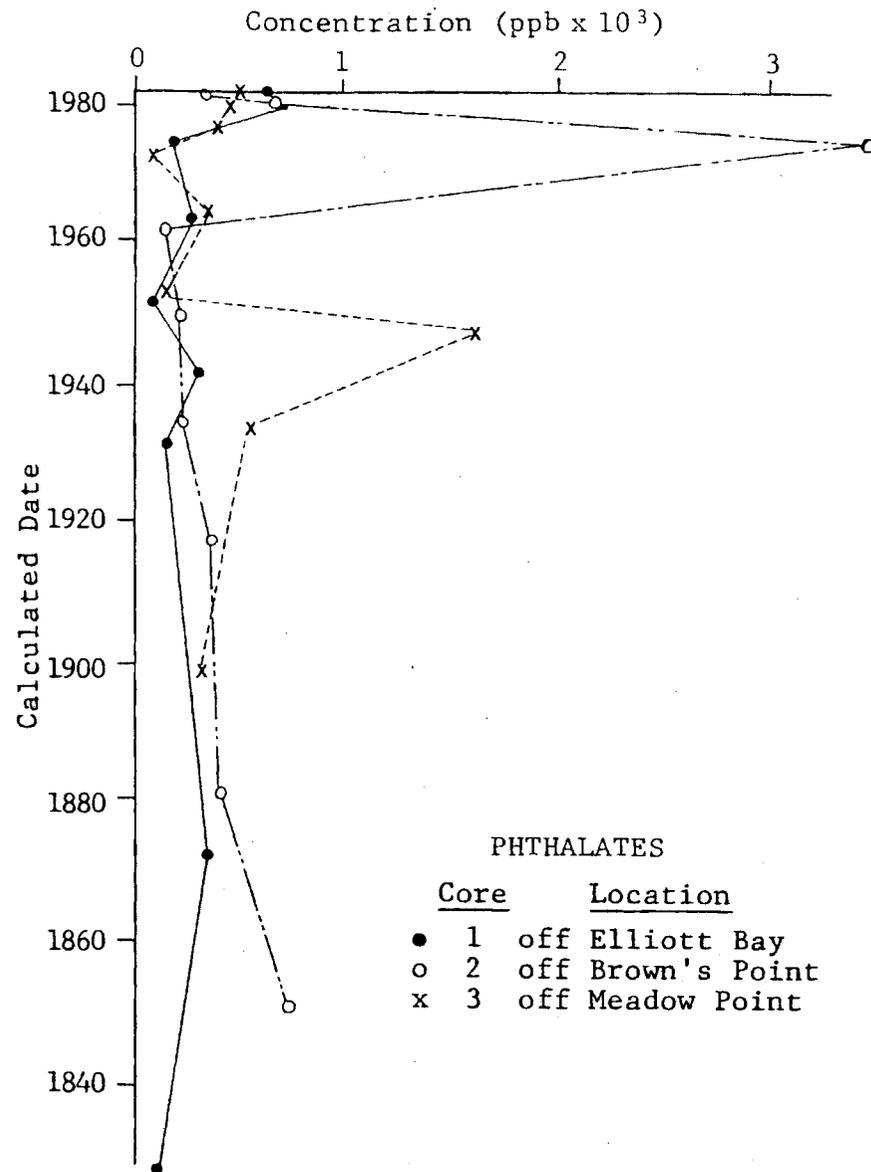TERMS OF CALCULATED DATE OF DEPOSITION.

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Figure 57. SEDIMENT PHTHALATE CONCENTRATION PROFILES FROM
THREE CENTRAL BASIN KASTEN CORES. SEDIMENT DEPTH EXPRESSED
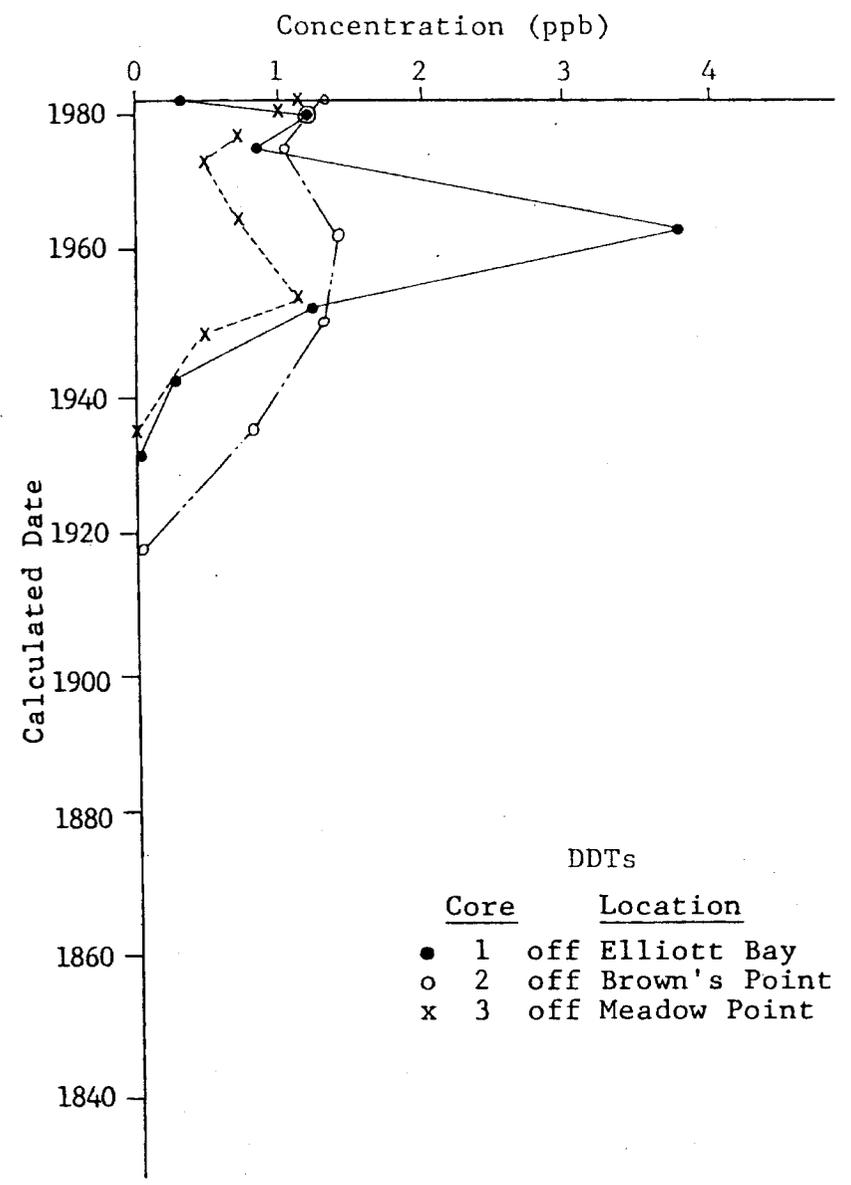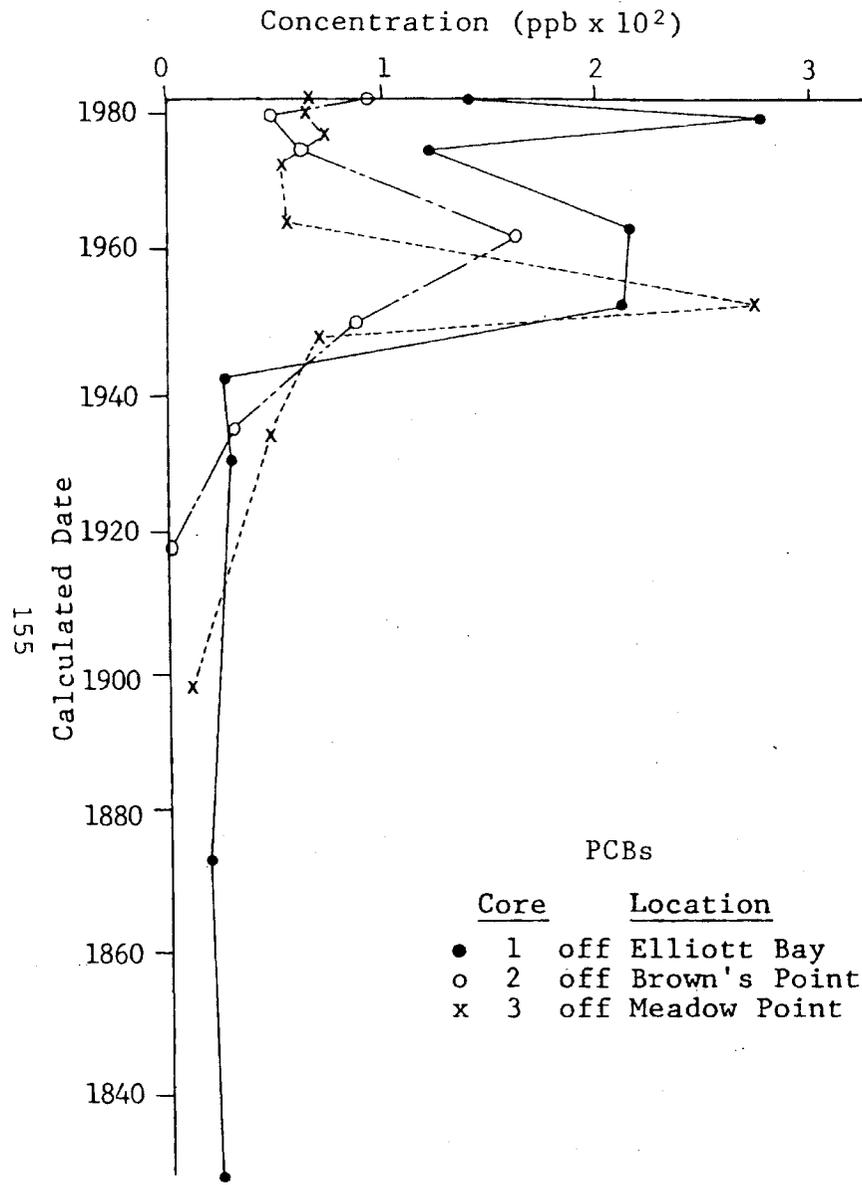IN CALCULATED DATE OF DEPOSITION.

Figure 58.   SEDIMENT TOTAL PCB AND TOTAL DDTs CONCENTRATION
PROFILES FROM THREE CENTRAL BASIN KASTEN CORES.   SEDIMENT
DEPTH EXPRESSED IN CALCULATED DATE OF DEPOSITION.

uncertainties and the lack of specific data, the computed trace metal input values are only rough estimates.

The input of synthetic organic compounds via shoreline erosion was assumed to be negligible since it is expected that erodable soils are not contaminated with these type of chemicals. To date, there is no data to either substantiate or negate this assumption.

1.1.3 Atmospheric Inputs

The atmospheric contributions to toxicant input in the main basin of Puget Sound has been estimated for total fallout and precipitation based on estimated contaminant concentrations and the fluxes computed in Appendix 1.

Total Fallout

Trace Metals - The atmospheric input rate of trace metal priority pollutants into the main basin of Puget Sound was calculated from measured concentrations in airborne particulates on land and estimated total deposition velocities values (TDV). The TDV were taken from literature values of TDV for specific metals. Due to the paucity of regional data two assumptions were made in obtaining these estimates: (1) the concentrations of airborne metals over the main basin can be presented from measurements obtained at the PSAPCA land based air monitoring sites and (2) the literature TDV values computed for other locations are applicable to this region.

The concentrations of metals (except Hg) in airborne dust were determined on air filters collected at the University of Washington along the Lake Washington Ship Canal, west of the Montlake Bridge. The particulate Hg data are from other urban areas and may not be representative of Seattle values.

The concentrations of metals in air over the main basin were estimated using published Pb and TSP data (PSAPCA, 1979). It was assumed that the ratios of the concentration of other metals to the Pb concentrations would be the same as the ones measured in Seattle. The PSAPCA reported quarterly averages for Pb and TSP for seven sites in 1978 and five sites in 1979. Similar data for 1979 were also available at three additional sites in Seattle, Sequim, and Quillayute (Crecelius, unpublished results). This information was combined and a linear regression equation was developed to establish the relationship between Pb and TSP, as Pb = 0.017 (TSP) - 0.136, with a correlation coefficient of 0.86.

This relationship was then used to calculate the airborne Pb concentrations for the major subregions of Puget Sound. The results

are shown in Tables 2.12 and 2.13. The Hg value from the literature is for only particulate Hg in urban air and may not represent Seattle air (National Research Council, 1978).

Intercomparison between the Seattle air chemistry data, and that of the PSAPCA was possible only for the PSAPCA Montlake Boulevard site on Portage Bay (across the ship canal from the University of Washington) where concurrent measurements were obtained during 1979. The PSAPCA value reported was 910 ng/m^3 as compared to 740 ng/m^3 estimated in this study. Considering the variability in the data and a site separation of approximately 0.5 km the two values agree well. The TDV literature values selected for the metals considered here were: 0.3 cm/sec for Pb, Zn, As, Cd, Ag, Hg, and 1.0 cm/sec for Fe and Cu (McMahon and Denison, 1979; Young and Silker, 1980; Crecelius, 1981) and from field measurements collected at Sequim, located approximately 30 km west of the Northern Basin (Crecelius, unpublished data).

The Pb flux (Table 2.13) was computed as the product of the airborne concentration and the TDV value of 0.3 cm/sec. The computed values range from 11 to 45 mg/m^2/year. This range is in agreement with values of 28 mg/m^2/year measured at La Jolla, California, 11mg/m^2/year at Ensenada, Mexico (Hodge et al., 1978) and 27mg/m^2/year at coastal sites around the North Sea (Cambray et al., 1969).

The atmospheric input rate for the metals into each of the major subregions were then computed by multiplying the flux values by the corresponding surface area of each subregion. These values are summarized in Table 2.14. To test the reliability of these estimates a comparison of the calculated atmospheric deposition rate in the Northern Basin for Fe, Cu, and As was made with the rates measured at Sequim, Washington.

The rates measured at Sequim were 4.7, 42, 0.90, and 0.46 mg/ m^2/year for Pb, Fe, Cu, and As respectively (Crecelius, 1981). The rates calculated for the Northern Basin are Pb 11.3, Fe 60, Cu 1.9, and As 0.71 mg/m^2/year. The rates measured for these four elements at Sequim average approximately 50% of those calculated for the Northern Basin. Two explanations are possible to account for this discrepancy: (1) Sequim air contains lower metal concentrations than calculated for the Northern Basin and therefore has a lower deposition rate, and (2) the mathematical correlation between Pb concentration in air and TSP does not hold for rural air. Both of these explanations are reasonable and are justified below.

The annual average airborne Pb concentratin at Sequim is 0.057 μg/m^3 (Crecelius, unpublished data) compared with 0.120 μg/m^3 calculated for the Northern Basin. Furthermore, the TSP at Sequim was 15.4 μg/m^3, the same as the Northern Basin, but the Sequim air Pb concentration was 0.057 μg/m^3 as compared to the

Table 2.12. AIR CHEMISTRY DATA MEASURED ON FILTERS
COLLECTED AT THE UNIVERSITY OF WASHINGTON DURING 1979

	Metal Concentration in Dust (μg/g)	Concentration[b] in Air (ng/m^3)
Pb	18,500	740
Zn	1,825	73
As	1,150	46
Cd	23	0.93
Ag	6.8	0.27
Fe	29,530	1,180
Cu	950	38
Hg[a]	5.5	0.22[c]

[a]Particulate Hg only

[b]Source: Crecelius, unpublished data.

[c]Source: National Research Council, 1978.

Table 2.13. ATMOSPHERIC FLUX OF Pb IN PUGET SOUND

Region	TSP[a] Concentration (μg/m^3)	Pb[b] Concentration in Air (μg/m^3)	Deposition[c] Rate (mg/m^2/yr)	Area x10^6m	Load Kg yr[1]
S. Puget Sound	20	0.21	20	4.69	9380
S. Basin	36[d]	0.48[d]	45[d]	1.65	7425
S. Central Basin	35	0.46	43	.417	1793
Central Basin	35	0.46	43[d]	1.61	6923
N. Central Basin	24[d]	0.27[d]	25[d]	1.3	3250
N. Basin	15	0.12	11.3	2.8	3080

[a]From PSAPCA (1979) countour map for total suspended particulates, Figure 1.6 and Table 1.5 of Appendix 1.

[b]Values obtained from regression equation: Pb=0.017(TSP)-0.136.

[c]Computed from the concentration values in the previous column assuming a TVD of 0.3 cmsec^{-1}· (94,500 m/yr)

[d]Revised values

Table 2.14. ANNUAL TOTAL ATMOSPHERIC INPUT OF TRACE METALS INTO THE MAIN BASIN OF PUGET SOUND

Region	Revised [a] Pb	Pb	Zn	As	Cd	Ag	Fe	Cu	Hg
				Input Rate $(kgyr^{-1})$					
S. Puget Sound	9,380	9,380	918	591	11.8	3.5	49,700	1,570	2.83
S. Basin	7,740	7,425	d						
S. Central Basin	6,321	1,793							
Central Basin	6,930	6,923							
N. Central Basin	6,910	3,250							
N. Basin	3,146	3,080							
Total Basin	40,427 (60,000) [b]		3,952 (50,000)	2,547 (77,000)	50.9	14.87	213,842	5,773 (20,000)	11.02
Revised Total Basin [c]		31,850	3,113	2,006	40.0	11.7	168,465	4,548	8.7

[a] Revisions based on Table 2.13

[b] Values in parentheses are the estimates presented in Dexter et al., 1981.

[c] Ratio of revised Pb total to initial Pb total used to adjust values for other metals.

[d] Missing values can be calculated based on the ratio of Pb to each metal for S. Puget Sound.

Table 2.15. ATMOSPHERIC INPUT RATES OF COMBUSTION POLYNUCLEAR AROMATIC HYDROCARBONS (CPNA) IN THE MAIN BASIN OF PUGET SOUND

Region	Input Rate $(kgyr^{-1})$						Total Combustion PAH
	Fluoranthene	Pyrene	Chrysene	Benzo-Fluouanthenes	Indeno-(c,d)Pyrene	Benzo-(g,h,i)Perylene	
S. Puget Sound	3.9	4.6	9.2	39	7.7	13	77.4
S. Basin	2.2	3.7	7.6	32	6.3	10.7	63.5
S. Central Basin	2.6	3.1	6.2	26	5.2	8.8	51.9
Central Basin	2.9	3.4	6.9	29	5.8	9.8	57.8
N. Central Basin	2.9	3.4	6.7	29	5.6	9.6	57.2
N. Basin	1.4	1.6	3.1	14	2.6	4.5	27.2
Total Basin	16.9	19.8	39.7	169	33.2	56.4	335.0

value of 0.120 µg/m^3 calculated using the Pb/TSP regression for urban air, as computed earlier. This difference in Pb concentration is not unexpected because the enrichment of Pb is much higher near urban areas than for rural area. Iron, which is not as enriched in urban air relative to rural air, shows good agreement between Sequim and the Northern Basin.

The total fallout rates estimates obtained for trace metals in this study were compared with historical information. The atmospheric input of five metals was reported for Seattle by Spyridakis and Barnes (1978) as 6 µg/m^{2}year for As, 19 µg/m^2/year for Cu, 490 µg/m^2/year for Fe, 68 µg/m^2/year for Pb and 330 µg/m^2/year for Zn. The corresponding rates determined in this study are 2.7, 7.2, 227, 43, and 4.2 µg/m^2/year for As, Cu, Fe, Pb, and Zn, respectively. The lower values reported here result from using the air over the Sound which has a lower suspended load compared to air over Seattle.

The estimated inputs of Pb, Zn, Cu and As reported by Dexter et al. (1981) are generally higher than the values computed in this work. There are several reasons for this discrepancy: (1) a different value was chosen for the areas of the basin, (2) over estimation of rate, and (3) changes in emission rates. Hood Canal, Whidbey Basin, Elliott Bay, Commencement Bay, and inlets west of Bainbridge Island were not included in this study as areas receiving atmospheric input. Dexter et al. (1981) included these areas which may explain why the Pb and Cu inputs are larger than those in this report. The large discrepancy between the As and Zn inputs are not as easily explained. Specifically, the As input used by Dexter et al. (1981) was taken from Crecelius et al. (1975). The flux of 77,000 kg/year[1] was an upper limit based on total deposition measured in Seattle during 1971. Since 1971 the As emission rate of the smelter has decreased significantly (approximately by a factor of 5; PSAPCA 1981). This may explain why our estimate of total As deposition is only 2547 kg/year based on air filters collected in Seattle during 1979. There is no apparent explanation for the discrepancy in the Zn flux which was estimated to be substantially less (more than a factor of 10) than that of Dexter et al. (1981). From Table 2.12 it can be seen that the concentration of Zn is Seattle air was a factor of 10 lower than that of Pb which resulted in a commensurate decrease in the flux value as compared to Pb. All in all, we believe that our values are the most recently updated figures, are reasonable and hence should supersede any previously published numbers.

Within the context and the goal of this study it should also be pointed out that except for Pb the atmospheric input term for all other metals is a minor contribution to the total metal input as compared to other sources and therefore the accuracy of the data is inconsequential to the results of the overall mass balance estimates.

Synthetic Organics - The atmospheric input rates for priority organic pollutants were estimated only for polynuclear aromatic hydrocarbons (PAH) since no data on other compounds were available at the time this study was performed. These calculations were based on the combustion PAH data reported by Prahl, 1982 and the assumption that the PAH input is proportional to the input of atmospheric lead. The PAH/Pb ratio in Seattle as compared to Sequim is virtually the same (0.0082 and 0.0089, respectively), therefore, the above assumption is justifiable.

The results of these computations are shown in Table 2.15. It should be pointed out that these values are consistent with the independent calculations of Prahl, 1982 who computed a flux value of 68 $\mu g/m^2$/year for the Olympic Peninsula. This figure is in good agreement with the value of 90 $\mu g/m^2$/year determined for the Northern Basin in this study.

Barrick (1982) estimated the direct atmospheric dry dust fall of four PAH compounds to Puget Sound to be 228 kgyr^{-1}. This flux included flouranthene, chrysene, benzo(e)pyrene, and indeno(c,d)-pyrene. There is relatively good agreement between Barrick's flux of 228 kgyr^{-1} for six CPNA in Table 2.15.

Precipitation

Although there is no field data available for the fraction of the total atmospheric deposition that is wet fallout, one can estimate the relative importance of the dry and wet deposition from measurements made at other geographic sites. Atmospheric eposition into southern Lake Michigan was estimated to be about equal for the wet and dry fractions (Gatz 1975); for the Lake Windermere region in England, dry deposition for seven metals ranged from 5% to 17% of total fallout (Peirson et al., 1973); for Bermuda dry fallout was 33% of total deposition (Duce et al., 1976) and for Greenland it was reported as 25% of the total (Davidson et al., 1981). If one assumes that wet fallout accounts for 75% of the total deposition and that the material deposits uniformly over time, then the concentration of pollutants in precipitation can be estimated.

Based on the above assumptions the mass loadings of the designated trace metals and CPAHs from precipitation were computed and summarized in Tables 2.16 and 2.17.

Dry Fallout

The computed dry fallout contribution to the total atmospheric mass loadings were estimated as 25% of the total. These values are summarized in Tables 2.18 and 2.19.

Table 2.16. MASS LOADING OF TRACE METAL INTO THE MAIN BASIN OF PUGET SOUND FROM PRECIPITATION[a]

Region	Input Rate (kgyr^{-1})[b]					
	Pb	Zn	As	Ag	Cu	Hg
S. Puget Sound	7035(22.26)	689(2.18)	443(1.40)	2.63(0.83)	1178(3.72)	2.12(0.67)
S. Basin	5569	c				
S. Central Basin	1345					
Central Basin	5192					
N. Central Basin	2438					
N. Basin	2310					

[a]Values in parentheses are in 10^{-2} gsec^{-1} except for Ag and Hg which are 10^{-4} gsec^{-1}.

[b]Precipitation values estimated at 75% of total atmospheric input listed in table 2.14.

[c]Missing values for other metals must be determined by first calculating the total atmospheric input values as listed in Table 2.14.

Table 2.17. MASS LOADINGS OF SYNTHETIC ORGANIC COMPOUNDS INTO THE MAIN BASIN OF PUGET SOUND FROM PRECIPITATION

Region	Input Rate (kgyr^{-1})						Total Combustion PAH
	Fluoranthene	Pyrene	Chrysene	Benzo-Fluoranthenes	Indeno-(c,d)Pyrene	Benzo-(g,h,i)Perylene	
S. Puget Sound	2.9	3.5	6.9	29	5.8	9.8	57.9
S. Basin	2.4	2.8	5.7	25	4.7	8.0	48.6
S. Central Basin	2.0	2.3	4.7	20	3.9	6.6	39.5
Central Basin	2.1	2.6	5.3	21	4.5	2.6	43.1
N. Central Basin	2.2	2.6	5.1	21.8	4.2	7.2	43.1
N. Basin	1.1	1.1	2.4	11	1.9	3.4	20.9
Total Basin	12.7	14.9	30.1	127.8	25.0	42.6	253.1

Table 2.18. MASS LOADINGS OF TRACE METALS INTO
THE MAIN BASIN OF PUGET SOUND FROM DRY FALLOUT[a]

| Region | Input Rate $(kg\,yr^{-1})$[b] | | | | | |
	Pb	Zn	As	Ag	Cu	Hg
S. Puget Sound	2345(7.42)	229(0.72)	148(0.47)	0.88(0.28)	392(1.24)	0.71(0.22)
S. Basin	1856	c				
S. Central Basin	448					
Central Basin	1731					
N. Central Basin	812					
N. Basin	770					

[a]Values in parentheses are $10^{-2}\,g\,sec^{-1}$, except for Ag and Hg which are $10^{-4}\,g\,sec^{-1}$.

[b]Dry fallout values estimated at 25% of atmospheric input listed in Table 2.14.

[c]Missing values for other metals must be determined by first calculating the total atmospheric input values as listed in Table 2.14.

Table 2.19. MASS LOADINGS OF SYNTHETIC ORGANIC COMPOUNDS INTO
THE MAIN BASIN OF PUGET SOUND FROM DRY FALLOUT

| Region | Input Rate $(kg\,yr^{-1})$ | | | | | | Total Combustion PAH |
	Fluoranthene	Pyrene	Chrysene	Benzo-Fluoranthenes	Indeno-(c,d) Pyrene	Benzo-(g,h,i) Perylene	
S. Puget Sound	1.0	1.1	2.3	10	1.9	3.2	19.5
S. Basin	0.8	0.9	1.9	7	1.6	2.7	14.9
S. Central Basin	0.6	0.8	1.5	6	1.3	2.2	12.4
Central Basin	0.8	0.8	1.6	8	1.3	2.2	14.7
N. Central Basin	0.7	0.8	1.6	7.2	1.4	2.4	14.1
N. Basin	0.3	0.5	0.7	3	0.7	1.1	6.3
Total Basin	4.2	4.9	9.6	41.2	8.2	13.8	81.9