

REVIEW OF METHODS TO REDUCE URBAN STORMWATER LOADS: TASK 3.4

FINAL REPORT

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SUMMARY

Mercury (Hg) and polychlorinated biphenyls (PCBs) are of current environmental concern in San Francisco Bay due to their lengthy persistence in the environment and their potential adverse effects on human health and wildlife. In response to human and wildlife health risks, San Francisco Bay is listed as a water body impaired due to Hg and PCBs under Section 303(d) of the federal Clean Water Act. The listings require that Total Maximum Daily Load (TMDL) reports be prepared that define the problems and include source assessments, numeric targets, a linkage analysis, load allocations, and an implementation plan. The San Francisco Bay Regional Water Quality Control Board (SFBRWQCB) has recently developed TMDLs for Hg and for PCBs. The implementation plans in these reports call for greater effort to track source areas of higher contamination in urban settings, the development of a source control program, the implementation of Best Management Practices (BMPs) capable of controlling or removing Hg and PCBs from urban areas and storm water, measurements of loads in stormwater, and demonstration that either loads or particle concentrations in urban stormwater are decreasing. However there are presently no specifications as to which urban BMPs can be applied to achieve these recommended load reductions and there remains limited information on the extent of contamination of urban soils and stormwater in the Bay Area, or the rest of California.

This report collates the state of knowledge on Hg and PCB sources in the Bay Area and develops a preliminary assessment of BMPs that might be applicable for Hg and PCBs loads reduction. The report includes an analysis of sources of Hg and PCB in the urban environment, including magnitude and distribution in relation to source control BMPs and the storm drain conveyance network, concentrations of Hg and PCBs found in a variety of urban media (e.g. soils, sediments, roof, runoff), processes by which Hg and PCB are transported and transformed within the urban drainage that affect the treatability of these constituents in BMPs and the placement of treatment control BMPs, the efficacy of source control programs to prevent these constituents from entering the storm drainage network, the efficacy of treatment devices in removing Hg and PCB from stormwater in the drainage network, the current level of implementation of BMPs including maintenance activities, monitoring that is needed to fill data gaps, and the limitations of our current knowledge. The report is organized into the following sections:

- Section 1: Provides a review of regulatory issues in relation to PCBs and Hg, an overview of Bay Area land use and population, describes the history of use of Hg and PCBs, comments on the linkage between these pollutants and other pollutants of concern, and provides a series of definition for some key terminology used in other sections.*
- Section 2: Develops the framework for and implements a mass balance analysis for Hg and PCBs in rivers, creeks, and stormwater conveyance systems in the Bay Area. The mass balances are developed for two periods, the highest use period (1950 -1990) and the recent period (1990 - 2005). The mass balance provides part of the basis for preliminary prioritization of best management practices (BMPs) (Section 5).*
- Section 3: Describes the concentrations and particle characteristics of Hg and PCBs in various urban media (soils, road surfaces, roof tops, catch basins, and storm drains) based on an extensive search of the local and international literature. This is used to build a series of hypotheses on what might be found in components of the Bay Area environment is provided to support decisions about BMPs (in particular, treatment control options).*
- Section 4: Discusses the transport of PCBs and Hg in urban stormwater. It includes discussions on particles size, organic carbon, and iron, and discusses density, coagulation,*

flocculation, and settling and ends with a conceptual model of particle size distribution within different urban components.

Section 5: Provides a review of BMP control options and argues their suitability for controlling PCBs and Hg. For example, the section includes a discussion on pollution prevention source control, soil remediation, street sweeping, street washing, storm drain maintenance, and channel de-silting. The section then finishes with a discussion on unit operations and treatment performance in relation to treatment control BMPs.

Section 6: Provides a short summary of the entire report, an analysis of data gaps, and recommendations for filling data gaps through further information collation or focused data collection.

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1. INTRODUCTION

1.1 SUMMARY OF POLLUTION AND TMDL ISSUES

Mercury (Hg) and polychlorinated biphenyls (PCBs) are of current environmental concern in San Francisco Bay due to their lengthy persistence in the environment and their potential adverse effects on human health and wildlife. Interpretations of data collected in San Francisco Bay since 1993 by the Regional Monitoring Program for Trace Substances (RMP) suggest that concentrations of Hg and PCBs in water, sediment, and fish tissue are of magnitudes that pose human and ecological health risk (SFEI, 2005a, SFEI, 2005b). A fish consumption advisory for Hg was first issued in 1994 and then updated in 1999 (OEHHA 1999). For PCBs similarly, a fish consumption advisory was issued in 1994 and has remained in place (OEHHA 1994). In addition there is growing evidence that Hg is the cause of hatching failures in some rare and endangered native species (Davis et al., 2003). In 1998 and 1999, SFEI partnered with the California Department of Health Services to conduct a San Francisco Bay fish consumption study to develop data on fishing, cooking and eating habits of recreational and subsistence fishers (SFEI, 1999; CDHS and SFEI, 2001). This report series helped to improve outreach and education programs and reduce public risk associated with catching and eating fish from the Bay. In addition, there is ongoing research on the relationships between Hg and PCB pollution and biological effects in wildlife (Schwarzbach and Adelsbach, 2002; Davis et al., 2003; Schwarzbach et al., 2006) that might also help to reduce exposure especially in relation to wetland restoration efforts.

In response to human and wildlife health risks, San Francisco Bay is listed as a water body impaired with Hg and PCBs under Section 303(d) of the federal Clean Water Act. The listings require that Total Maximum Daily Load (TMDL) reports be prepared that define the problems and include source assessments, numeric targets, a linkage analysis, load allocations, and an implementation plan. The Region 2 Regional Water Quality Control Board (RWQCB) has recently developed TMDLs for Hg (Looker and Johnson, 2004) and for PCBs (Hetzl, 2004). These reports use a mass balance approach to compare contaminant loads entering the Bay from each source and propose load allocations that need to be met over 20 years (Table 1-1 and Table 1-2). The urban runoff implementation actions recommended by the Hg TMDL to achieve these loads reductions are:

1. *Evaluate and report on the spatial extent, magnitude, and cause of contamination for locations where elevated mercury concentrations exist.*
2. *Develop and implement a mercury source control program.*
3. *Develop and implement a monitoring system to quantify either mercury loads or the loads reduced through treatment, source control, and other management efforts.*

The Guadalupe Watershed received special attention in the Hg TMDL (Looker and Johnson, 2004) because of the history of Hg mining at New Almaden:

1. *Quantify the annual average mercury load reduced by implementing:*
 - i) *Pollution prevention activities*
 - ii) *Source and treatment controls, and*

- iii) *If applicable, other efforts to reduce methylation or mercury-related risks to humans and wildlife consistent with the watershed-based strategy. The Water Board will recognize loads reduced resulting from activities implemented after 1996 (or earlier if actions taken are not reflected in the 2001 load estimate) to estimate load reductions.*
- 2. *Quantify the mercury load as a 5-year annual average mercury load using data on flow and water column mercury concentrations.*
- 3. *Quantitatively demonstrate that the mercury concentration of suspended sediment that best represents sediment discharged from the watershed to San Francisco Bay is below the suspended sediment target (0.2 mg/kg).*

The PCB TMDL for San Francisco Bay is on a slower timeline relative to the Hg TMDL. The staff report (Hetzl, 2004) however, contains urban runoff implementation recommendations similar in many ways to those of Hg:

- 1. *Demonstrate attainment of the sediment target (0.002 mg/kg [preliminary – not yet finalized]) in discharges*
- 2. *Demonstrate load reductions in discharges*
- 3. *Demonstrate loads removed by actions taken that might include:*
 - i) *Cleanup of hotspots on land, in storm drains, and in the vicinity of storm drain outfalls*
 - ii) *Capture, detention, and treatment of highly contaminated runoff*
 - iii) *Implementation of urban runoff management practices and controls that have PCBs removal benefit.*

Table 1-1. Current loads of Hg and future allocations (Modified from Table 7.1: Looker and Johnson, 2004).

Source	2003 Mercury Load (kg/yr)	Allocation (kg/yr)	Reduction (%)
Bed Erosion	460	220	53
Central Valley Watershed	440	330	24
Urban Stormwater Runoff	160	82	48
Guadalupe River Watershed (mining Legacy)	92	2	98
Atmospheric Deposition	27	27	0
Non-urban stormwater runoff	25	25	0
Wastewater (municipal and industrial)	20	20	0
Dredging and Disposal	Net loss	0	-

Table 1-2. Current loads of PCBs and future allocations (Modified from Table 27: Hetzel, 2004).

Source	Current PCB Loads (kg/yr)	Proposed PCB Loads (kg/yr)	Proposed Load Reductions (kg/yr)	Reductions (%)
Atmospheric	-7	-7	0	0
Delta	42	32	10	24
Waste Water Discharges	2.3	2.3	0	0
Urban Runoff	34	2	32	94
Dredged Material	12	1.4	11	92
In-Bay PCB "Hotspots"	NQ	NQ	NQ	-
Total	83	31	53	64

NQ – Not quantified.

The information contained in these tables along with the implementation recommendations demonstrates an emphasis on reduction of urban runoff loads. However there are presently no specifications on which urban Best Management Practices (BMPs) can be applied to achieve these recommended load reductions. In fact, in very polluted watersheds with no “clean” upland sediment supply from open space / agricultural land use or low density residential land use, it may be easier to demonstrate compliance by load avoided or load reduction and almost impossible to reach sediment targets. In less polluted watersheds, with only a few polluted hotspots and high upland “clean” sediment supply, it may be easier (more economical) to demonstrate compliance by reaching sediment targets. It should be recognized that loads avoided may not translate into a 1:1 load trend in a downstream storm drain; the trend will be, in part, a function of the hydrological connection of mass of PCBs or Hg removed. In addition, the success of an implementation plan for Hg and PCBs is somewhat dependant on multiple benefits from reducing the impacts of other pollutants and human influences on receiving water impairments.

Aims

This report will collate the state of knowledge on Hg and PCB sources in the Bay Area and develop a preliminary assessment of BMPs that might be applicable for Hg and PCBs loads reduction:

1. Sources of Hg and PCB in the urban environment, including magnitude and distribution in relation to source control BMPs and the storm drain conveyance network.
2. Concentrations of Hg and PCBs found in a variety of urban media (e.g. soils, sediments, roof, runoff)
3. Processes by which Hg and PCB are transported and transformed within the urban drainage that affect the treatability of these constituents in BMPs, and the placement of treatment control BMPs.
4. The efficacy of source control programs to prevent these constituents from entering the storm drainage network, and the efficacy of treatment devices in removing Hg and PCB from stormwater in the drainage network.
5. The current level of implementation of BMPs including maintenance activities.
6. Monitoring that is needed to fill data gaps and the limitations of our state of knowledge.

1.2 GENERAL HISTORY OF BAY AREA POPULATION AND LAND USE

Runoff, pollutant supply, distribution, and transport are all affected by intensity of land use, land and water management, and history and changes in loading over time. Climatic influences on water, sediment, Hg and PCB loads are occurring in concert with the changing influences of human population and development. The European history of the Bay Area essentially began with the discovery of gold in 1848 and the associated rapid influx of prospectors, farmers, and service people. In 1860, as the gold rush era began to draw to a close, the population of the Bay Area rose when displaced gold workers from the Sierra Nevada began to seek a new life (Figure 1-1). During the last 40 years of the

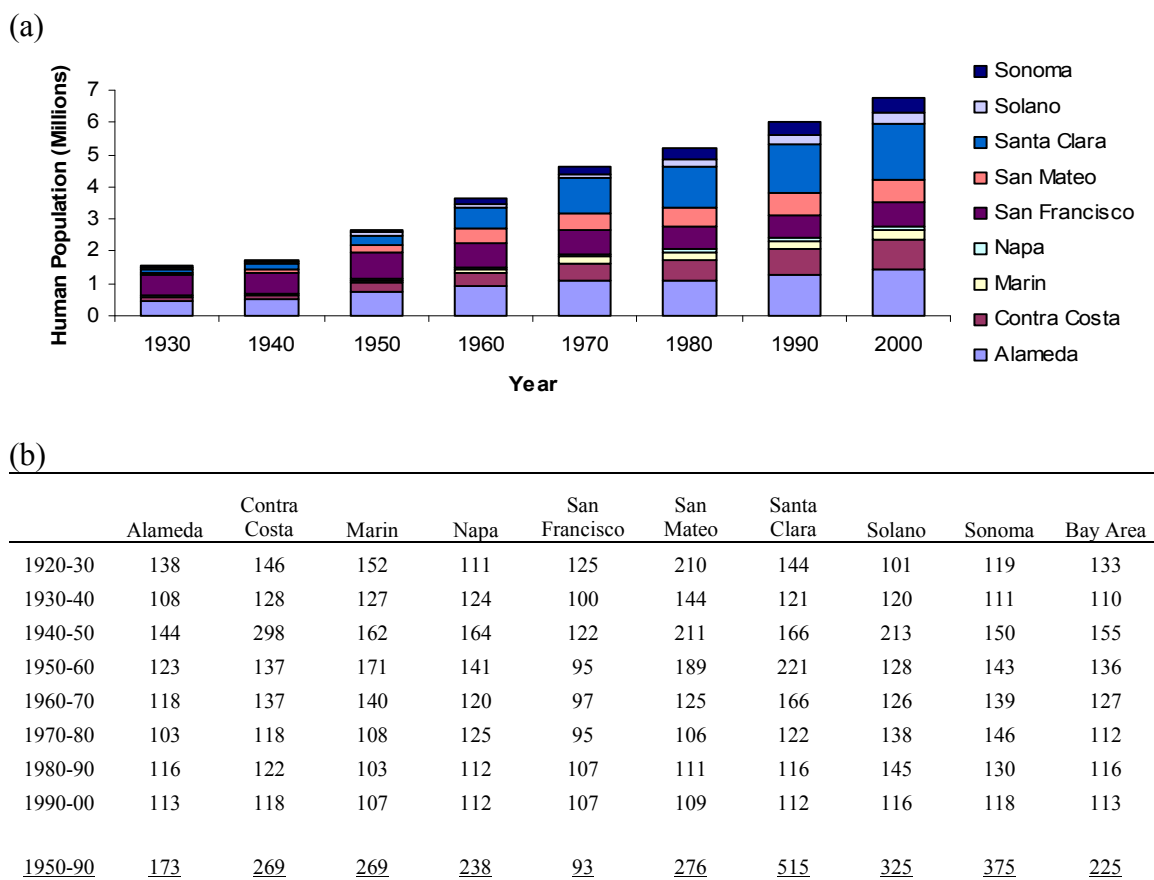


Figure 1-1. Population trends in the Bay Area from 1930 to present. (a) Total population by county; (b) population rate of change (%) by county and total Bay Area rate of change.

19th century, population and agriculture continued to expand and by 1900, the population of the Bay Area had reached 700,000. In the early 1900s industry was beginning to boom. The petroleum industry was established (for example, Standard Oil established its west coast refinery in Richmond) and rail transportation improved the transmission of goods and services throughout the Bay Area and connected San Francisco to the eastern United States. By 1915, the population of the nine counties had surpassed 1,000,000 and by 1945 2,000,000 people lived around the Bay (Figure 1-1a). During the period of peak Hg and PCB usage (see below), Santa Clara, Alameda, and Contra Costa became the most populous counties in the Bay Area accounting for 60% of total population. Population increased rapidly during the post war “baby boom” (Figure 1-1b) and flat areas around the Bay previously in agriculture were converted to suburban land use. The largest population increase occurred in Santa Clara, the most populous county (over 5x increase between 1950 and 1990).

1.3 GENERAL HISTORY OF Hg AND PCB PRODUCTION AND USE

Hg and PCBs are legacy pollutants – their peak production and use occurred decades ago and their new use has largely been banned. In this section, we discuss their general use and history in the United States and California as context for the Bay Area problem. It is this same data that is incorporated with a variety of other local data and manipulated to develop the mass balances presented in Chapter 2.

Hg has been mined and used in the U.S. since the early 1800s. The first Hg production peak occurred during and after the gold rush in California some 125 years ago (Figure 1-2). Hg production and use then gradually declined until the 1930s when a number of new industrial uses and products began to appear. A second peak in production and use occurred during World War II when Hg was used in the war effort. At the close of the war, many of the products that were developed or perfected during WWII were introduced into the consumer market, including portable radios and cameras that heavily rely on batteries. Many cameras from the 60s through 80s used Hg batteries and the general use of Hg in batteries continued until 1991 when new environmental laws banned its use (DTSC, 2002; Sznoppek, 2000) (Figure 1-3). Batteries were the largest use of Hg in the U.S. and likely in the Bay Area.

The second largest use of mercury in the U.S. was the chlor-alkali process (Sznoppek, 2000). The term chlor-alkali refers to the three chemicals [chlorine, and an alkali (sodium hydroxide or potassium hydroxide)] which are simultaneously produced as a result of the electrolysis of a saltwater. The mercury cell process takes place in an electrolytic cell, where liquid mercury acts as a cathode. It attracts sodium (or potassium) cations with which it forms an amalgam. Chlorine gas collects at the anode (graphite). When the amalgam is added to water, the sodium (or potassium) reacts with the water to form

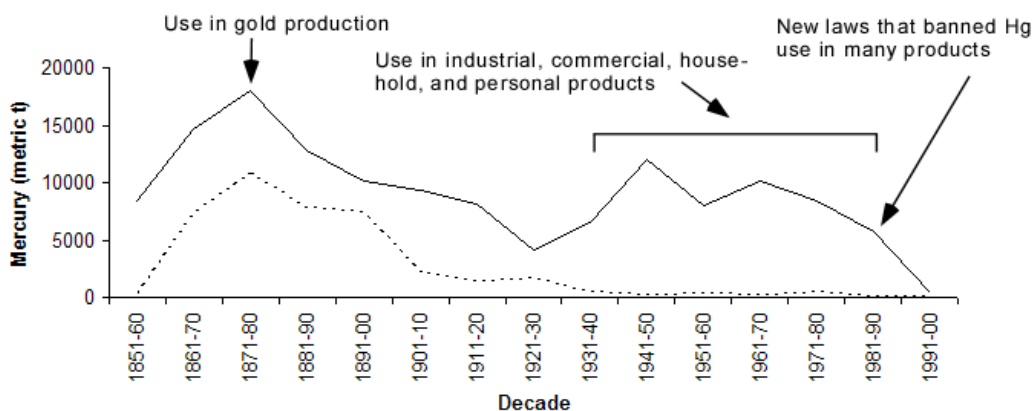


Figure 1-2. Hg production in the entire U.S. and New Almaden Mining District, San Jose, (Bay Area) CA.

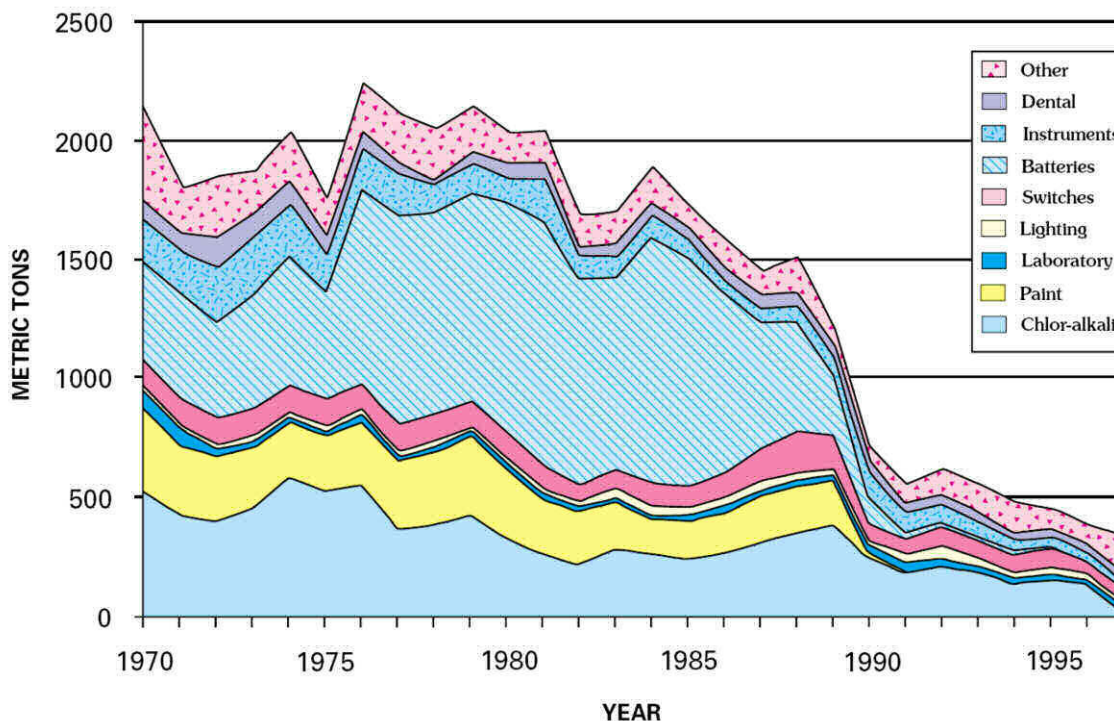


Figure 1-3. Mercury use (metric t) in U.S. over the past 30 years (Sznoppek, 2000).

sodium hydroxide and hydrogen, leaving the mercury, which can then be reused. Because mercury is highly volatile, mercury pollution occurs throughout the process, commonly leading to both the product (caustic soda) and the wastewater stream containing small amounts of mercury (GreenFacts, 2005). To our knowledge, there were no chlor-alkali plants in the Bay Area. However, there main have been small operations that may come to light in the next few years.

The other large single use of Hg in the U.S. was latex paint. This, similar to the use of Hg in batteries, was also outlawed in 1991. In addition primary mining production of Hg ceased in 1991. Recycling and secondary Hg production supports the remaining uses of mercury, including dental, switches, lighting and laboratory (Sznoppek, 2000). In general, there has been a large decline in the use of Hg in the major uses (batteries, chlor-alkali and paint) largely due to legislative pressure, however, Hg use in other products has not shown similar declines. Ongoing uses and improper disposal of out-of-use equipment and products represent a continuing source of mercury to the urban environment, although present use is approximately 5x less than the peak of the 1950-90 period.

PCBs were commercially produced and consumed in the U.S. from 1929 to 1977 mainly by Monsanto who acquired the process in 1935. Monsanto has a number of branch offices in California, but none are presently operating in the Bay Area. PCBs are not known to occur naturally, however they can occur as minor byproducts in a number of chemical industrial processes, during drinking water chlorination and from thermal

degradation (Erickson, 1992). The U.S. total production of PCBs by Monsanto has been reported to be approximately 640,000 t (de Voogt and Brinkman, 1989 cited in Breivik et al., 2002a). Production peaked in 1970 at approximately 30,000 t or about 6% of the total U.S. production (Figure 1-4). Overall approximately 57% of total production occurred between 1960 and 1974 and 73% of the U.S. production occurred between 1955 and 1977. There is some uncertainty on the total production given unreliable reporting of mass associated with a complex chemical mixture (there are 209 possible congeners), a number of common mixtures with different proportions of congeners (Aroclors), and a number of industrial processes that inadvertently produce PCBs as incidental byproducts. Overall, it appears that total production is proportional to total consumption in the USA and that population might be a reasonable surrogate to approximate the distribution of use (Breivik et al, 2002b). Population is used as a surrogate in Section 2 of this report.

PCB use can be classified into three primary source / use classes (Erickson, 1992): 1. Controllable closed systems where leakage is avoided by design during the lifespan of the equipment, 2. Uncontrollable closed systems, which are technically closed but where leakage usually occurs (also referred to as nominally closed), and 3. Dissipative Uses where the PCBs were in direct contact with the environment and there is no way of recovering them when the product reaches end-of-life (also referred to as open-ended applications). PCBs were primarily used in controllable closed systems, but there were also significant uses in uncontrollable closed systems and dissipative uses (Table 1-3). We have no specific data on the uses in the Bay Area, so for this report it will be assumed that Bay Area consumption patterns mimic national consumption patterns.

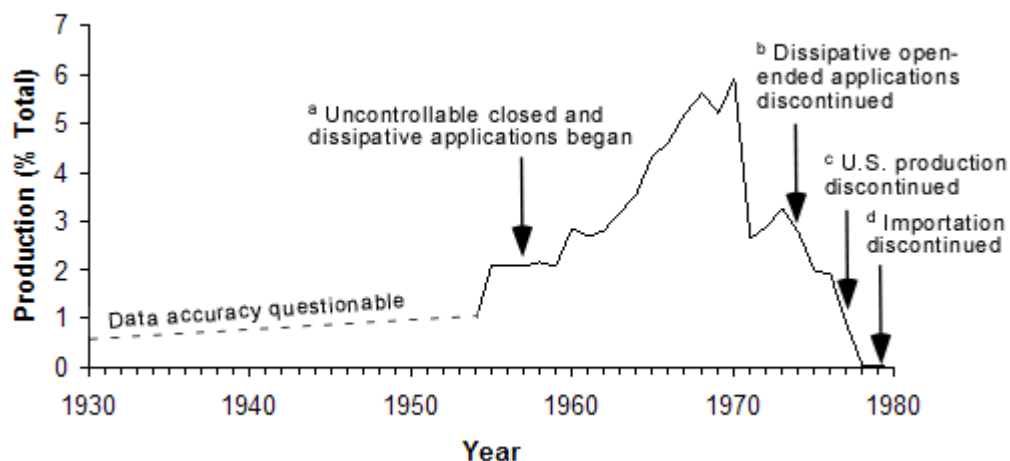


Figure 1-4. Annual PCB production in the U.S. from 1930 to 1977. References: a: USEPA, 1987 cited in EIP Associates, 1997; b: Hetzel, 2004; c: Erickson, 1992; d: Hetzel, 2004.

Table 1-3. The main uses of PCBs in the U.S. up to 1977 (Modified from Keeler et al., 1993).

Class	Uses	Total Production to 1977		
		(%)	10 ⁶ lbs	10 ⁶ kg
Controllable closed systems	Transformers, capacitors, fluorescent light ballasts	60	850	385
Uncontrollable closed systems (nominally closed)	Hydraulic fluids and lubricants	10	140	63
Dissipative (open-ended)	Plasticizers (additives in plastics that maintain softness and pliability)	25	350	159
Dissipative (open-ended)	Flame retardants, paints, inks, sealants, and carbonless copy paper	5	71	32
<u>Total</u>		<u>100</u>	<u>1,411</u>	<u>640</u>

1.4 OTHER POLLUTANTS OF CONCERN AND MULTIPLE BENEFITS

The success of an implementation plan for the reduction of Hg and PCBs loads to San Francisco Bay may be improved if there are multiple benefits for other lower priority and emerging pollutants and also benefits to flood control, minimizing hydro-modification, and trash reduction. Recently the Sources Pathways and Loadings Workgroup (SPLWG) of the Regional Monitoring Program for Trace Substances (RMP) compiled a prioritized list of pollutants based on 2005 concerns (Table 1-4). This table highlights the evolution of problem development and shows how the emphasis has changed over the past 5 years and how polybrominated diphenyl ethers (PBDEs), endocrine disruptors, and pyrethroids have been added to the priority list. The mass load of each priority pollutant entering San Francisco Bay is proportional to both its concentration in stormwater and the volume of stormwater.

Table 1-4. The evolution of priority pollutants in the San Francisco Bay Region in the context of the SPLWG.

2000			2005	
PCBs	Top		PCBs and Hg	Top
PAHs	High		PBDEs	High
			Endocrine	High
OPs	High		disruptors	
Hg	Medium		Pyrethroids	High
Se	Medium		Se	Medium
Cu	Medium		Cu	Medium
			DDT, chlordane,	Low
Ni	Medium		dieldrin	
			Ag, As, Cd, Cr, Ni,	Low
TBT	Medium		Pb, Zn	
Ag	Medium		Dioxins/Furans	Low
Cd	Medium		PAH hotspots	Low
Chlordane	Low		OPs	Low
DDT	Low			

It is well accepted that structural treatment control BMPs that include retention or detention in their treatment process remove pollutants most effectively (e.g., CASQA, 2003, Burton and Pitt, 2002). An interesting phenomenon is that as watershed imperviousness increases with the degree of urbanization, event mean concentrations (EMC) flow can remain reasonably constant, even though loads increase due to increased runoff (Cabezas et al., 2005). This may be truer for Hg than for PCBs because mercury is likely more evenly dispersed in the environment. Although there is little literature to support this, BASMAA data from the 1988-95 monitoring efforts showed that EMC of some trace metals were difficult to distinguish between commercial and residential uses, in spite of variability in imperviousness. In addition, for one location (Rheem Creek) that spanned a wide range of rainfall intensities, there were inverse relationships noted between rainfall intensity and SSC, total and dissolved Cu and Ni suggesting that hydrology can have a measurable influence on concentrations during individual storms. Addressing hydro-modification (volume reduction not just peak reduction) could be a factor in the long-term success of an implementation plan written in response to the Hg and PCB TMDLs. Urban BMPs and regulations seldom achieve no net volume increase. Runoff from increased imperviousness and other factors related to urbanization increase flows in stormwater conveyances which, depending on conditions in the conveyance, can disrupt the sediment supply and transport and lead to excessive erosion and/or sedimentation. Some Bay Area stormwater programs have permit requirements that address this issue, starting with the development and implementation of hydro-modification management plans. Trash is a problem in the Bay Area. Trash itself is a source of pollutants when it degrades in the urban and Bay environment but it is not known if it is a significant source of Hg and PCBs. However, trash obstructs the proper functioning of structural treatment control BMPs, reducing their effectiveness and increasing maintenance costs. Overall, the successful implementation of the Hg and PCB TMDLs may identify areas for coordination with measures to address other pollutants, hydro-modification, and trash.

1.5 DEFINITIONS USED THROUGHOUT THIS REPORT

- Study Area: Small tributaries and urban drainages that flow into San Francisco Bay with a total non-tidal area of 6,650 km² [i.e. excludes the Central Valley watershed (154,000 km²)]. The area of small tributaries that drain to the Bay is entirely within the Region 2 RWQCB boundary but excludes the watersheds that drain to Tomales Bay, coast side Marin, the combined sewer system of San Francisco, and coast side San Mateo.
- Pollution Prevention: Given the majority of Hg and PCBs in the environment are historic, the term pollution prevention, in the context of this report, is reserved for further bans on use and reuse. This is an issue particularly for Hg which is still used in, for example, dentistry, switches, lighting, and laboratory applications.
- Source Control BMPs: Source control BMPs are those that are applied on-site before the pollutant enters the hydrological cycle in a runoff or waste stream that

leave its primary source site where it had not yet been mobilized. Given the majority of Hg and PCBs in the environment are historic, the term source control includes activities such as:

- a) Recycling
 - b) Removal (e.g. remediation of on-land hotspots)
 - c) Onsite structural (e.g. check dams, mine capping)
 - d) Nonstructural (e.g. education, change in business practices)
 - e) Polluted rainfall interception
- Treatment Control BMPs: A treatment control BMP is one which treats runoff downstream or off-site from a source area. Treatment control BMPs treat runoff or waste streams that contain pollutants that have been mobilized and are part of the hydrological cycle. In this report we will recognize scale as an important factor:
 - a) Catchment scale (i.e. hotspots, individual property)
 - b) Sub-watershed scale
 - c) Watershed scale
 - d) Retrofits
 - e) Stream hydro-modification
- Maintenance Activities: Maintenance activities are those activities carried out by cities and counties that remove sediment and debris from areas where water actively flows and carries pollutants. Maintenance activities mostly treat runoff downstream or off-site from a source area but could also include activities at the scale of a known source area. Maintenance activities treat runoff or waste streams that contain pollutants that are temporarily at rest and part of the hydrological cycle and include:
 - a) Street sweeping
 - b) Inlet cleaning
 - c) Channel dredging
 - d) Catch basin cleaning
 - e) Pump station cleaning

1.6 REFERENCES

- Breivik, K., Sweetman, A., Pacyna, J. M., and Jones, K. C. (2002a). Towards a global historical emission inventory for selected PCB congeners - a mass balance approach. 1. Global production and consumption. *Science of the Total Environment*, 290, 181-198.
- Breivik, K., Sweetman, A., Pacyna, J. M., and Jones, K. C. (2002b). Towards a global historical emission inventory for selected PCB congeners - a mass balance approach. 2. Emissions. *Science of the Total Environment*, 290, 199-224.
- Burton, G.A. Jr and R. Pitt (2002), *Stormwater Effects Handbook: A Tool Box for Watershed Managers; Scientists and Engineers*, CRC Press, Inc., Boca Raton, FL.

- Cabezas, M., Jensen, P., and Merriam, J., 2005. Managing impacts of development on stormwater runoff: Benefits of low impact development approaches. Proceedings of the Eighth Annual Stormwater Research & Watershed Management Conference. Tampa, Florida. April 27-28 April, 2005. p40-43.
- CASQA 2003. Stormwater Best Management Practice Handbooks, California Stormwater Quality Association. <http://casqastore.com/customer/home.php?cat=4>
- CDHS and SFEI, 2001. Public Summary of the San Francisco Bay Seafood Consumption Study. Technical Report prepared by California Department of Health Services (CDHS) and San Francisco Estuary Institute (SFEI). 13pp.
- Davis, J., D. Yee, J. Collins, S. Schwarzbach, and S. Luoma, 2003. Potential for Increased Mercury Accumulation in the Estuary Food Web. San Francisco Estuary and Watershed Science, pp. 1 to 8.
- De Voogt, P., Brinkman, U.A.Th., 1989. Production, properties and usage of polychlorinated biphenyls. In Kimbrough, R.D., and Jensen, A.A. (Eds.) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Topics in Environmental Health. Elsevier. p 3-45.
- DTSC, 2002. Mercury report. Department of Toxic Substances Control, Hazardous Waste Management Program, State Regulatory Programs Division, Sacramento, August 2002. 125pp.
- EIP Associates, 1997. Polychlorinated biphenyls (PCBs) source identification. A report prepared for Palo Alto Regional Water Quality Control Plant, Palo Alto, CA. October 1997. 16pp + appendix.
- Erickson, M.D., 1992. Analytical chemistry of PCBs. CRC Press, Inc./ Lewis Publishers, Boca Raton, Florida. 508pp.
- GreenFacts, 2005. <http://www.greenfacts.org/glossary/abc/chlor-alkali-process-chlor-alkali-plant.htm> (cited July 2005).
- Hetzel, F. 2004. PCBs in San Francisco Bay: Total Maximum Daily Loads Report. San Francisco Bay Regional Water Quality Control Board. Oakland, CA. January 2004. 69pp. http://www.waterboards.ca.gov/sanfranciscobay/TMDL/SFBayPCBs/pcbs_tmdl_project_report010804.pdf (cited July 2005).
- Keeler G.J., Pacyna J.M., Bidleman T.F., Nriagu J.O., 1993. Identification of Sources Contributing to the Contamination of the Great Waters (Revised) EPA/453/R-94/087. Washington, DC:U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards.
- Looker, R., and Johnson, W., 2004. Mercury in San Francisco Bay Total Maximum Daily Load (TMDL): Proposed Basin Plan Amendment and Staff Report. California Regional Water Quality Control Board San Francisco Bay Region, September 2, 2004. 118pp + appendix (A, B). http://www.waterboards.ca.gov/sanfranciscobay/Agenda/09-15-04/September%2015,%202004%20Board%20Meeting_files/09-15-04-10_appendix_c.pdf (cited July 2005).
- OEHHA. 1994. Health advisory on catching and eating fish: interim sport fish advisory for San Francisco Bay. Office of Environmental Health Hazard Assessment. California Environmental Protection Agency. Sacramento, CA.

- OEHHA, 1999. California Sport Fish Consumption Advisories 1999. Office of Environmental Health Hazard Assessment. California Environmental Protection Agency. Sacramento, CA.
- Schwarzbach, S. and Adelsbach, T., 2002. Assessment of Ecological and Human Health Impacts of Mercury in the Bay-Delta Watershed: CALFED Bay-Delta Mercury Project Subtask 3B: Field assessment of avian mercury exposure in the Bay-Delta ecosystem. 30pp. <http://loer.tamug.tamu.edu/calfed/Report/Final/Bay-Delta%20Bird%20Hg%20final%20report.pdf> (cited July 2005).
- Schwarzbach, S., Albertson, J.D., and Thomas, C.M., 2006. Impacts of predation, flooding and contamination on the reproductive success of the California clapper rail (*rallus longirostris obsoletus*) in San Francisco Bay. *The Auk*, 123 (1), 45-60.
- SFEI, 2005a. San Francisco Regional Monitoring Program for Trace Substances (RMP). <http://www.sfei.org/rmp/index.html> (cited July 2005).
- SFEI, 2005b. The Pulse of the Estuary: Monitoring and managing water quality in San Francisco Estuary. SFEI contribution 78. San Francisco Estuary Institute, Oakland, CA.
- SFEI, 1999. San Francisco Bay seafood consumption study. Technical report of a study conducted by the Environmental Health Investigations Branch (EHIB) of the California Department of Health Services. San Francisco Estuary Institute (SFEI), Richmond, Ca. 85pp + appendix.
- Sznoppek, J.L. and Goonan, T.G., 2000, The materials flow of mercury in the economics of the United States and the world. Denver, CO. June 2000. 28pp.
- USEPA, 1987. Locating and estimating air emissions from sources of polychlorinated biphenyls (PCB). Office of Air Quality Planning and Standards. EPA-450/4-84-007n. May, 1987.

2. HG AND PCB SOURCES IN THE BAY AREA

2.1. INTRODUCTION

Mercury and PCB pollution in the urban environment is derived from a plethora of sources at a range of scales. This section summarizes what is known about sources of Hg and PCBs in the urban environment, describes the rationale for likely release potential (or a loading factor), and develops estimates of the release. This information is very relevant for considering the potential for increased pollution prevention and source control, especially for Hg given atmospheric deposition and its ongoing use in the urban environment. The information also has strong implications for treatment control and maintenance activities. The information is organized according to a conceptual model based on conservation of mass. Here we are only interested in developing a mass balance for the rivers, creeks, and stormwater conveyance systems. Terms that describe the other possible fates of these substances in the watershed environment such as permanent burial and volatilization (PCBs and Hg) and degradation and natural attenuation (PCBs) that are part of the overall fate of all of these substances are not included.

2.2. MASS BALANCE CONCEPTUAL MODELS

The nine main anthropogenic urban input categories for Hg (Sznoppek, 2000) are illustrated along with the atmospheric deposition. Only a small part of the total mass input actually passes into the surface hydrological cycle (represented on the conceptual model by dashed lines) (Figure 2-1). Equation 1 constrains the movement of Hg mass through the Bay Area watershed systems. There are three main source categories for PCBs: 1. Controllable closed systems, 2. Uncontrollable closed systems (nominally closed), and 3. Dissipative (open-ended) (Erickson, 1992). Keeler et al. (1993) broke the dissipative (open-ended class) into two smaller classes [3a: Plasticizers and 3b: Other Dissipative Uses (Flame retardants, paints, inks, sealants, and carbonless copy paper)]. In the same manner as Hg, the information on PCB sources and potential release is organized according a conceptual model based on mass balance (Figure 2-2). Equation 2 constrains the movement of PCB mass through the Bay Area watersheds. A term called storage that represents all removal of Hg or PCBs from the system (permanent burial in soils, capture of recycled Hg, and disposal of Hg to landfill or PCBs to landfill or incineration). These were factored onto calculations but are not explicitly included in the conceptual models since they were quantified as the residual of individual equations.

Equation 1: Other uses + Instruments + Dental + Batteries + Switches + Lighting + Laboratory + Paint + Chlor-alkali + Atmospheric Deposition – Storage – Street Sweeping – Inlet Cleaning – Maintenance Dredging = Stormwater Conveyance \pm Error

Equation 2: Controllable closed systems + Uncontrollable closed systems (nominally closed) + Dissipative (Plasticizers) + Other Dissipative Uses + Atmospheric Deposition – Storage – Street Sweeping – Inlet Cleaning – Maintenance Dredging = Stormwater Conveyance \pm Error

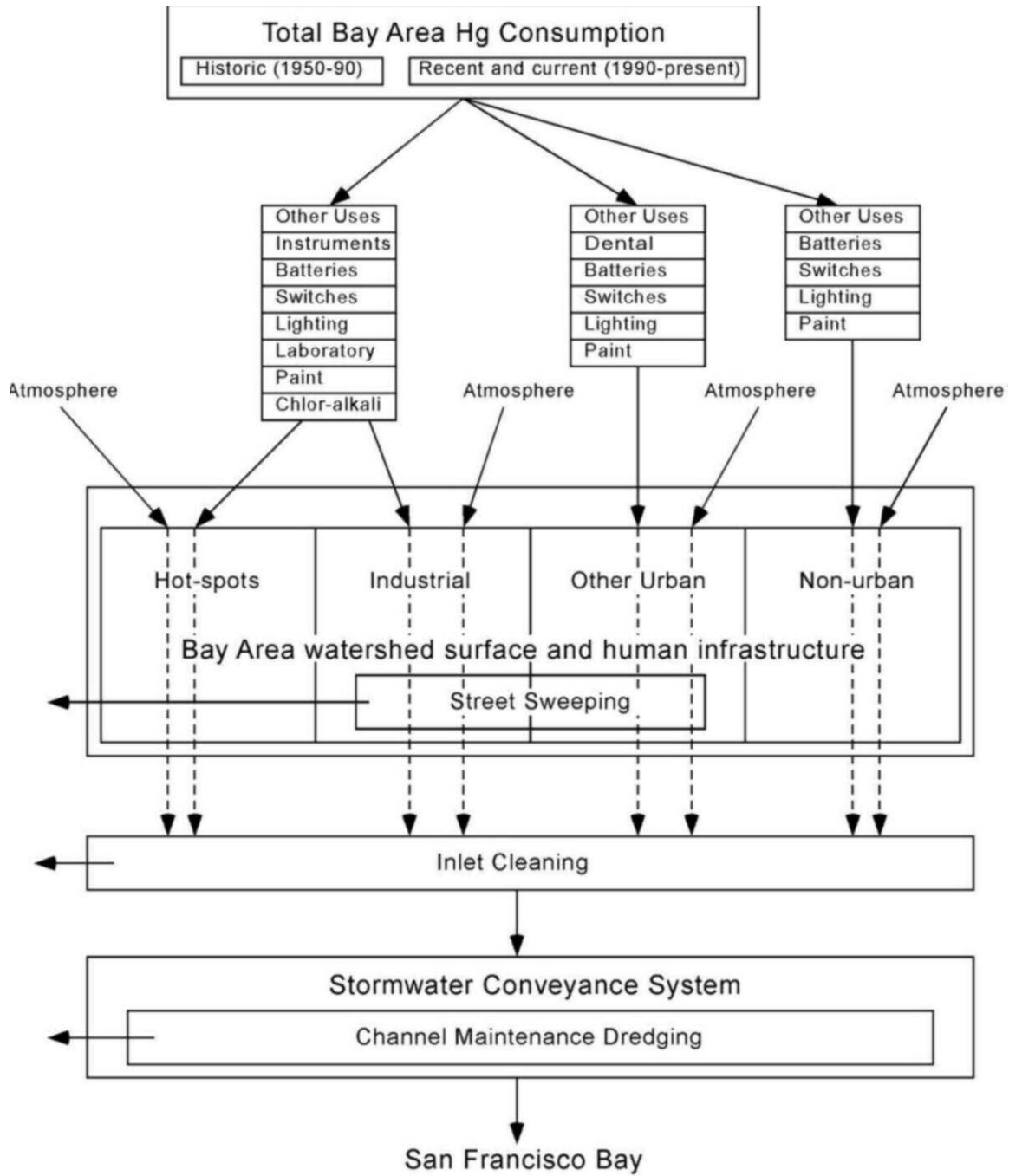


Figure 2-1. Conceptual model for Hg input into watershed areas of San Francisco Bay. Boxes represent storage in the system. Arrows show direction of mass flow. Dashed lines indicate partial mass transfer.

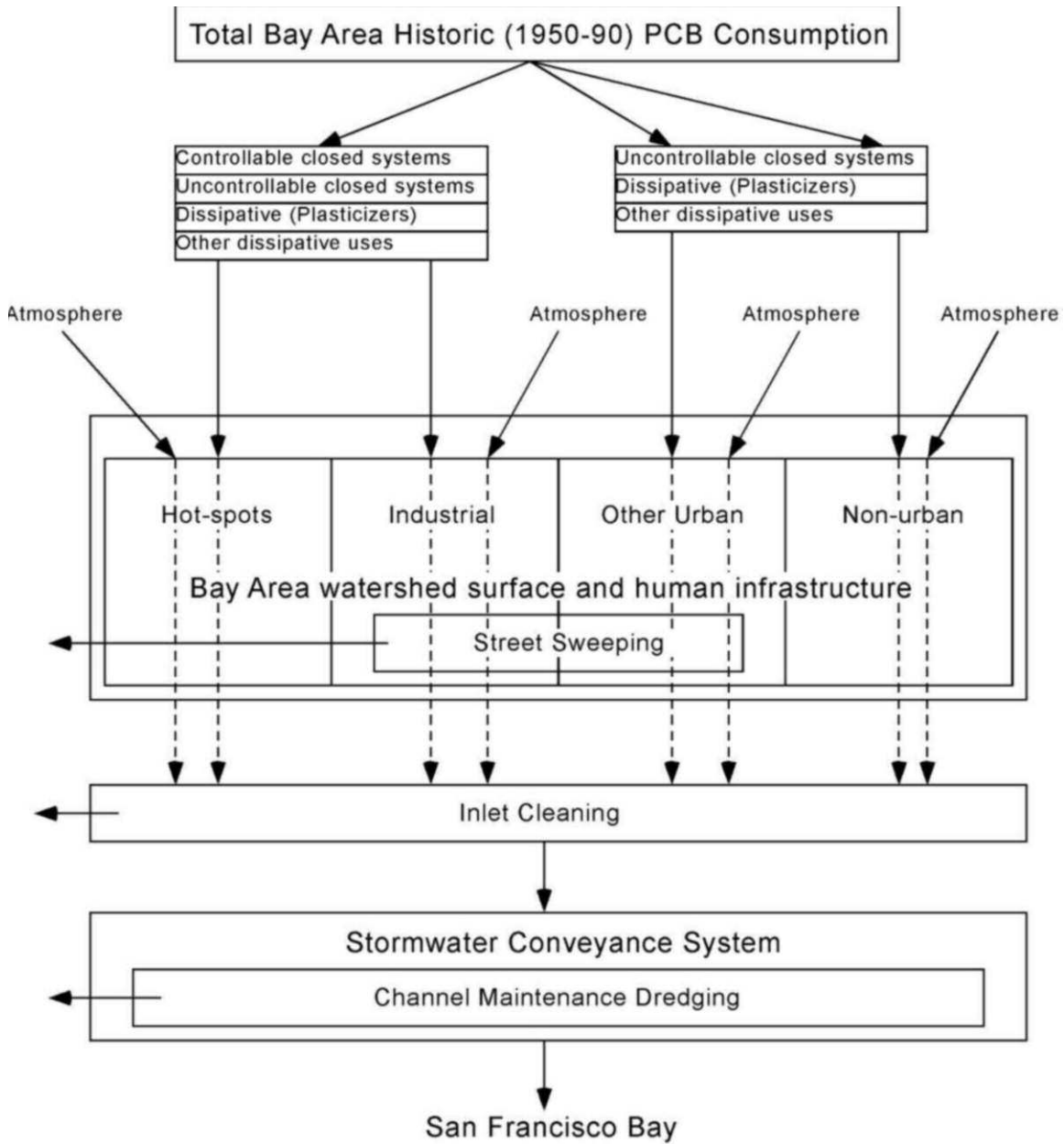


Figure 2-2. Conceptual model for PCB input into watershed areas of San Francisco Bay. Boxes represent storage in the system. Arrows show direction of mass flow. Dashed lines indicate partial mass transfer.

2.3. DETERMINING TRANSPORT AND FATE Hg AND PCBs IN THE URBAN ENVIRONMENT

2.3.1. Conceptual Model of Hg and PCB Transport and Fate

In order to understand how Hg and PCBs move from sources or use areas through the urban environment and to determine the portion that eventually finds its way into stormwater conveyances, we have developed another simple conceptual model (Figure 2-3) The model helps to answer the question: Can mass go via a given pathway and get to the stormwater conveyance system or is it blocked in some way before it gets there?

2.3.2. Land Use Areas

Another critical piece of information for determining flow of Hg and PCBs is knowledge of land use. As stated previously, the total area of small tributaries, including urban areas that drain to the Bay in the 9-county Bay Area is 6,650 km². According to the analysis completed by Davis, et al. (2000) using the 1995 ABAG land use statistics, the Bay Area is comprised of 374 km² of industrial land use or about 6% of the total small tributaries land area draining to the Bay (Table 2-1). It would be better to have land use statistics for the peak use period for Hg and PCBs (1950-90) for load estimation. However, where these are available they are likely to be less reliable before the widespread use of GIS.

2.3.3. Effort to Describe Sources of Hg and PCBs in California and Bay Area

There has been a number of efforts to-date that have collated information on Hg and PCB sources in California (Table 2-2). These can be used to develop hypotheses about what might be found in the Bay Area. In addition the information can sometimes be scaled down to make estimates of the likely magnitude of sources, processes, and loads in the Bay Area. There have also been a number of efforts to collate information on sources in the Bay Area (Table 2-3). These along with international literature have been used to develop an improved understanding of mass use and movement within the study area.

Table 2-1. Land use statistics used to estimate mass and populate the mass balance model for local tributary watersheds and urban areas draining to the Bay Area.

	Land Use				Total
	Industrial	Commercial	Residential	Open/Agriculture	
Area (km ²)	374	404	1,726	4,147	6,650
Area (%)	5.6	6.1	25.9	62.4	100

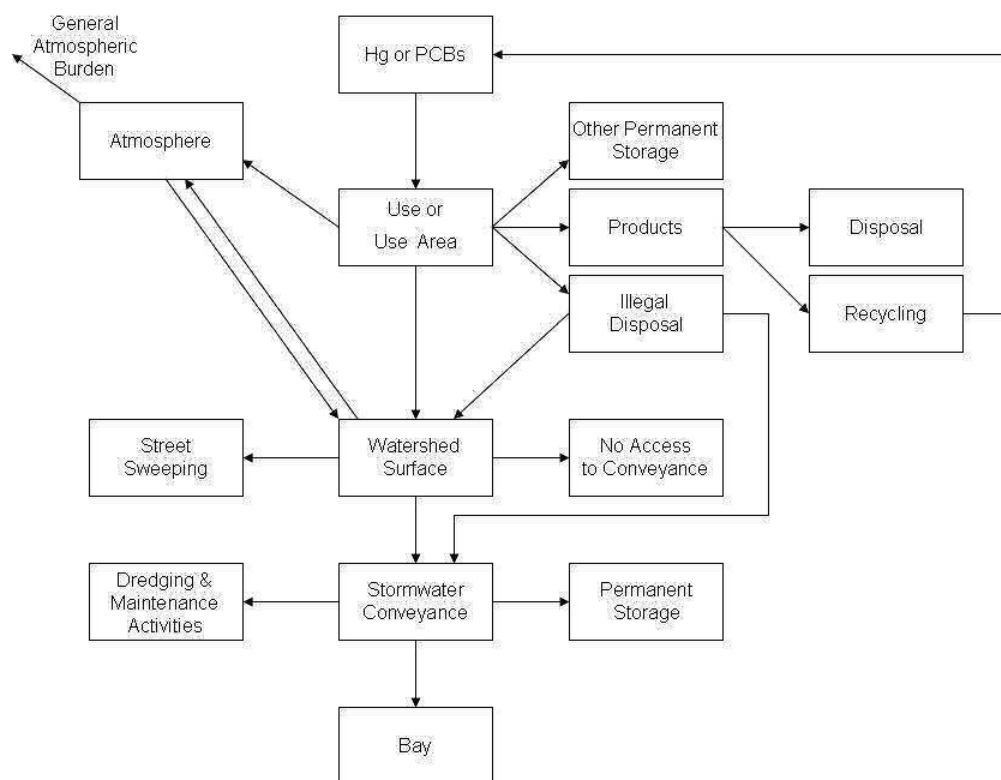


Figure 2-3. Conceptual model of flow of Hg and PCBs through the urban environment to San Francisco Bay.

Table 2-2. Information on sources of Hg and PCBs in the watersheds of California.

Reference	Description	Hg	PCBs	Other
Tsai and Hoenicke, 2001 Tsai et al., 2002	These discuss wet and dry deposition of mercury and PCBs in the Bay Area based on <1 year of data from three study sites (Moffett, Treasure Island and Martinez).	✓	✓	
DTSC, 2002	The report examines the problem of Hg pollution in California and the contribution of the disposal of mercury containing wastes not currently regulated as hazardous.	✓		
Steding, 2002	The paper examines mercury concentrations in rainfall and estimates annual loading based on mean concentrations and annual rainfall totals for the two California study locations	✓		
DTSC, 2003	This report discusses PCB use in light ballasts manufactured before 1978 and installed before 1980		✓	
USEPA, 2005a	This web site provides detailed information on superfund sites in California and the Bay Area. The information can be queried by pollutant, county, priority, and data can be mapped	✓	✓	✓
USEPA, 2005b	This web site provides access to the Toxics Release Inventory (TRI) which contains information on >650 toxic chemicals that are being used, manufactured, treated, transported, or released into the environment.	✓	✓	✓

Table 2-3. Information on sources of Hg and PCBs in the watersheds of the Bay Area.

Reference	Description	Hg	PCBs	Other
EIP, 1997	This report was written to help the City of Palo Alto to manage PCBs entering wastewater. It contains mass estimates of PCBs sources.		√	
SVTC, 2001	This report discusses Hg pollution associated with E-waste and mainly focuses on TVs and PCs.			
STOPPP, 2002	This memo inventories PCB use and release sites in San Mateo County.		√	
McKee et al., 2003	This report contained estimates of Hg supply to the Bay Area associated with vehicular fuel consumption.	√		
Salop and Akashah, 2004	This report reviews source control options for Alameda County including site cleanup, storm drain cleaning, and street sweeping	√	√	√
Dovzak and Sommers, 2004	Summarizes available information on potential sources of pollutants of concern to stormwater in Contra Costa County	√	√	√
DTSC, 2005	This fact sheet provides information on PCB pollution at the Hayward Air National Guard station.		√	
Kleinfelder, 2005	This report documents PCB pollution in the Ettie Street pump station watershed, Oakland, Ca.		√	
LWA, 2006	This report described sources of PCBs in the Bay Area focusing on building sealants and caulking.		√	

2.3.4. Total Potential Mass of Hg and PCBs used in the Bay Area

An estimate of mass of Hg and PCBs used in the Bay Area is useful for giving dimension to our system universe. The dimension of this universe constrains our estimates of mass associated with the likely largest sources and source areas. Comparisons to the findings from other studies in the world where much more rigorous analyses have been completed forces reconciliation if our system is dissimilar to these other systems.

The 1950-90 period was chosen for analysis because it provides the best understanding of the how the substances were distributed in the environment and also because the documentation of dispersal and magnitude of use is more detailed than for other periods. This choice seems defensible. PCB use prior to 1950 is estimated to be 22% of the total use (based on data of lower reliability) – or an average of about 1% of the total consumption per year. During the 1960s, use ranged from 3-6% of the total each year. Prior to the 1960s much of the production was mixtures that contained the lighter congeners which are less persistent. After the 1970s there was no further production but US laws allowed for a ramping down of use mostly during the 1980s. The period after 1990 was approached by considering the useful life span. For mercury, overall use closely followed the development of portable equipment and the use of batteries to power such equipment. The usage peak was much broader for Hg than for PCBs and the reduction in use much slower. But by 1990, the Hg use had been diminished by about 20% of the peak. Documentation of Hg use is best during the period 1970-1990. Use prior to that and the most recent and ongoing uses in modern personal devices is less well documented.

Our system universe for Hg and PCBs can be broadly defined by using population as a surrogate (Breivik et al., 2002a, 2002b) and assuming that production for both substances was equal to use (Hg: DTSC, 2002; PCBs: Breivik et al., 2002a, 2002b). Since peak production and use of both substances occurred between 1950 and 1990, we used the population census information for the U.S., California, and the Bay Area for this period in our scaling. The ratio of average Bay Area population to average U.S. population for this period is 0.022 (2.2%). The total U.S. consumption of Hg for this period (including Hg derived from import and domestic production) was 32,300 t (DTSC, 2002; Sznopce, 2000). Therefore the first order estimate of Bay Area Hg consumption is $2.2\% \times 32,300 = 711$ t. The total U.S. PCB production for the 1950-90 period was approximately 560,000 t. Again using population as a surrogate, the first order estimate of PCB consumption is $2.2\% \times 560,000 = 12,300$ t.

The import of Hg to the Bay Area is distinguishable from the rest of the U.S. in at least two ways. Firstly, there is natural mercury in the coast range serpentine geology that was mined from 1848 – 1975. The Guadalupe River in South San Francisco Bay is enriched with Hg derived from the historic New Almaden Mining District (McKee and Leatherbarrow, 2005). No adjustment is made for the mining input from Guadalupe because, for the purposes of this report, the mining component will be treated as a separate and unique watershed attribute. The second distinguishing feature of the Bay Area is that there are no known industrial chlor-alkali operations. The proportion of Hg used in chlor-alkali plants in the U.S. was approximately 25%, so therefore the local usage estimate is reduced to 530 t. Unlike PCBs, Hg is still being imported into the urban environment for various uses including dental, lighting, laboratories, batteries and “other uses” including electrical components in cell phones and computers. Based on scaling national usage figures for 1997 (Sznopce, 2000) and assuming no change, it is estimated that an additional 100 t of mercury has been imported into the Bay Area from 1990-2005 or about 7 t/y, down from an average of 13 t/y for the 1950-90 period.

2.3.5. Atmospheric deposition of Hg and PCBs

There are a number of efforts both previous and ongoing that have estimated atmospheric mercury fluxes in the Bay Area (Tsai, 2001; Steding, 2002; Yee, 2005). Tsai and Hoenicke (2001) measured wet and dry deposition of mercury in the Bay Area based on <1 year of data from three study sites (Moffett, Treasure Island and Martinez). The average dry deposition across all three sites was 18-21 $\mu\text{g}/\text{m}^2/\text{y}$ and the average wet deposition was 3.5-4.5 $\mu\text{g}/\text{m}^2/\text{y}$. These wet deposition estimates compare closely to the estimates generated from the National Atmospheric Deposition Program (NADP) which maintains a sampling site in San Jose, south San Francisco Bay (1.7-3.6 $\mu\text{g}/\text{m}^2/\text{y}$) (Yee, 2005) and the results of Steding and Flegal (2002) (4.4 $\mu\text{g}/\text{m}^2/\text{y}$). Using these estimates, the current average mass loading of Hg to the watershed areas of the Bay Area is approximately 130-170 kg/y (Table 2-4). In this case the best estimate we have of contemporary loading is the average of the low and high estimates (150 kg/y). We know of no long term estimate of Hg deposition in the Bay Area. In the absence of a better method, it is assumed that loadings over time have remained constant; the 1950-90

loading would then be 5.2-6.8 t. These estimates are very small compared to the Bay Area total usage estimates provided in the previous section but further populate our mass budget models (Hg Historical: Figure 2-4; Hg Current: Figure 2-5).

There was just one local effort to estimate PCB deposition from atmospheric sources (Tsai et al., 2002). Data collected by this study cover 6 months and were from a single location (Concord, Ca) in the northern part of the San Francisco Bay Area. Taking the average of the monthly estimates provided by Tsai et al. (2002), the average annual dry deposition is approximately $0.92 \text{ ng/m}^2/\text{d}$ or $0.34 \text{ } \mu\text{g/m}^2/\text{y}$ (monthly range $0.14\text{-}0.75 \text{ } \mu\text{g/m}^2/\text{y}$). This is considerably lower than the estimate for urban United Kingdom by Harrad, (1994) ($310 \text{ } \mu\text{g/m}^2/\text{y}$) which might have included both wet and dry deposition. A study completed in Paris found a dry deposition of $29 \text{ } \mu\text{g/m}^2/\text{y}$ and that dry deposition only accounted for 35% of the total deposition of PCBs (Granier and Chevreuil, 1997). A study in Switzerland found a dry deposition of $1.06 \text{ } \mu\text{g/m}^2/\text{y}$ (Rossi et al., 2004). Apparently, a ratio of 2:1 wet:dry is common but others have used a ratio of 10:1 (see references in Granier and Chevreuil, 1997) or even 12:1 (Rossi et al., 2004).

The current average mass loading of PCBs to the watershed areas of the Bay Area was estimated using a range in ratios from 2:1 to 10:1 wet:dry (Table 2-4). The current total deposition ranges between 3 and 55 kg. PCB concentrations in rainfall have been measured around the world and range between $1.3\text{-}35 \text{ ng/L}$ (Bremle and Larsson, 1997; Rossi et al., 2004). Using this range and assuming an average Bay Area rainfall of 500 mm ($\sim 20 \text{ in}$), the total load would be 4-116 kg/y (which a similar magnitude to the ratio method). Given that wet deposition appears to dominate in other systems, the lack of local wet deposition data for PCBs limits a confident understanding of the contribution of atmospheric PCB sources to urban stormwater. The best estimate ($\sim 9 \text{ kg}$) provided in Table 2-4 was derived assuming a 2:1 wet:dry ratio and using the average for the local dry deposition data. Similar to Hg, there is no long term estimate of PCB deposition in the Bay Area. Unlike Hg, where new use is not completely banned, new use of PCBs is banned and it seems likely that peak atmospheric loadings might have followed peak usage more closely than for Hg. However, in the absence of better information, it is assumed that loadings in the past were similar to the estimate for current loading; thus the 1950-90 loading would then be 360-2,200 kg. Given the higher usage in the past, perhaps the upper limit of this estimate might be closer to past reality. Again, these estimates are very small compared to the Bay Area usage estimates provided in the previous section but further populate our mass budget models (PCB Historical: Figure 2-6; PCB Current: Figure 2-7).

Not all mass entering a watershed finds its way to storm drain conveyances. However, compared to transfer of Hg and PCBs from, for example, batteries (Hg), dielectric fluids (PCBs) or paint (Hg and PCBs), Hg and PCBs in rainfall has a much less complex pathway to drainage systems and is more likely to occur in runoff. There have been several Hg budgets developed for watershed areas (Scherbatskoy, 1998; Grigal, 2002). Although there is plenty of complexity and ongoing discussion, in general forested watersheds appear to be less “leaky” than urban watersheds. For example, stream flow Hg in a Vermont forested catchment was only 5-8% of atmospheric deposition

(Scherbatskoy, 1998). In contrast, Mason (1998) found that runoff Hg from urban areas near Washington D.C. accounts for 38-103% of atmospheric deposition and they attributed the apparent difference to land use. In the absence of local studies, we will use the same methodology adopted by Tsai and Hoenicke (2001) where runoff coefficients (percentage of rainfall that forms stream flow on an annual basis for a defined watershed area) for land use categories are assumed to approximate loss factors. Davis et al. (2000) reviewed the literature of land use-related runoff coefficients and applied the following coefficients to Bay Area watersheds: Commercial/industrial (60-95%); Residential (20-50%); Agricultural/open (5-50%). These are used to make estimates of the amount of total atmospheric deposition of Hg and PCBs that enters storm water conveyances annually (Table 2-4). These mass transfers into stormwater are represented by dashed arrows on Figures 2-4, 2-5, and 2-7. These runoff coefficients will also be used to estimate movement of mass from other sources into storm water conveyances in later sections.

Table 2-4. Mass loading of Hg and PCBs associated with atmospheric deposition for local tributary watersheds and urban areas draining to the Bay Area.

		Industrial	Commercial	Residential	Open/ Agriculture	Total
Hg Dry Deposition (kg)	Low	7	7	31	75	120
	High	8	8	36	87	140
Hg Wet Deposition (kg)	Low	1	1	3	7	11
	High	2	2	8	18	29
<u>Hg Total Deposition (kg)</u>	Low	7	8	34	82	131
	High	9	10	44	105	169
	<u>Best</u>	<u>8</u>	<u>9</u>	<u>39</u>	<u>94</u>	<u>150</u>
<u>Hg loss to stormwater conveyance (kg)</u>	Low	4	5	7	4	20
	High	9	10	22	53	93
	<u>Best</u>	<u>8</u>	<u>8</u>	<u>14</u>	<u>19</u>	<u>48</u>
PCB Dry Deposition (kg)	Low	0.1	0.1	0.2	0.6	1
	High	0.3	0.3	1.3	3.1	5
PCB Wet Deposition (kg)	Low	0.1	0.1	0.5	1.2	2
	High	2.8	3.0	12.9	31.1	50
<u>PCB Total Deposition (kg)</u>	Low	0.2	0.2	0.7	1.7	2.8
	High	3.1	3.3	14	34	55
	<u>Best</u>	<u>0.5</u>	<u>0.5</u>	<u>2.3</u>	<u>5.5</u>	<u>8.9</u>
<u>PCB loss to stormwater conveyance (kg)</u>	Low	0.1	0.1	0.1	0.1	0.4
	High	2.9	3.2	7.1	17	30
	<u>Best</u>	<u>0.4</u>	<u>0.5</u>	<u>0.8</u>	<u>1.1</u>	<u>2.8</u>

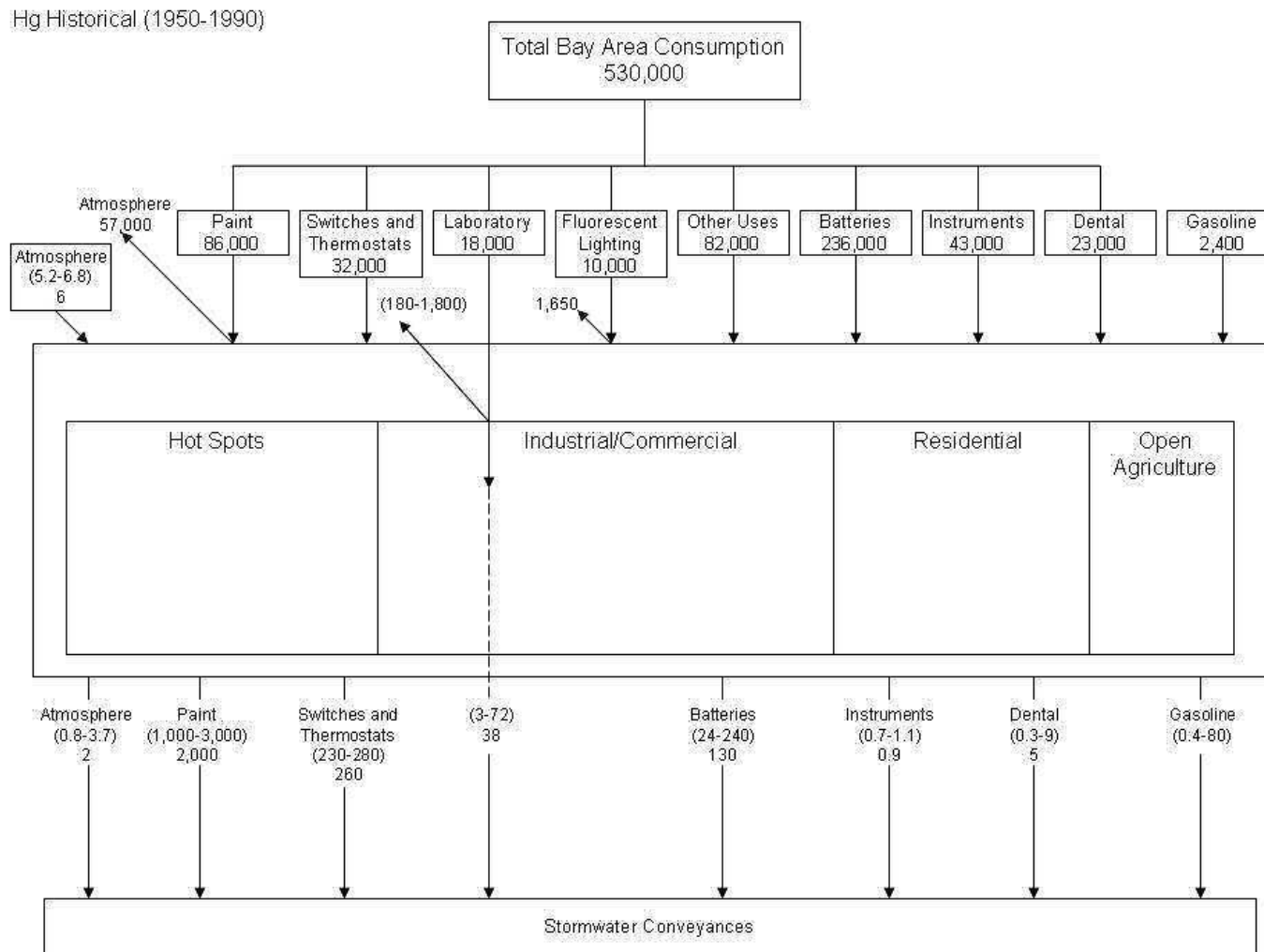


Figure 2-4. Estimates of total Hg mass movement in the Bay Area for the period 1950-90 (kg over 40 years). For estimation methods see relevant sections of this report.

Hg Today (2005)

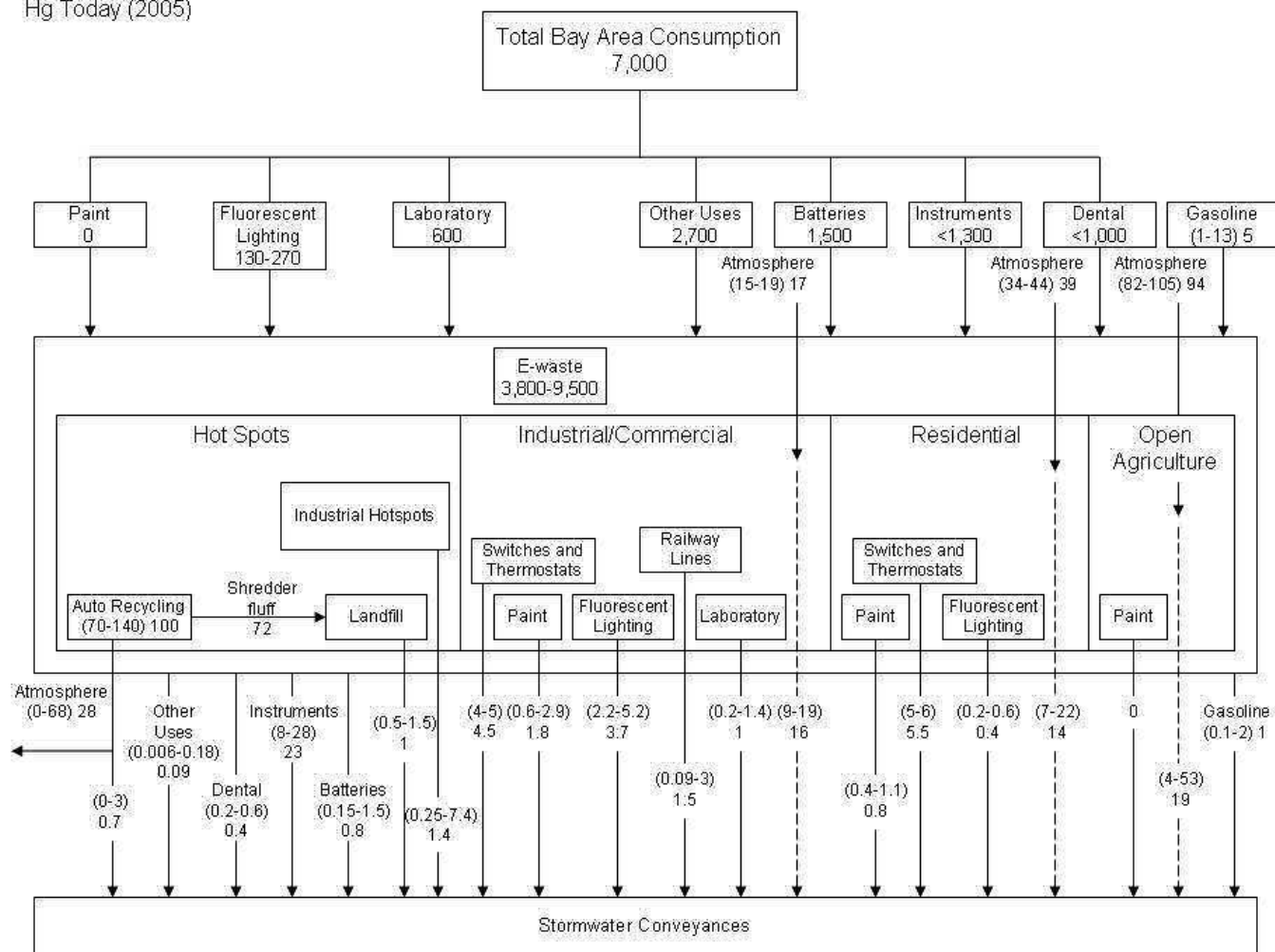


Figure 2-5. Estimates of Hg mass movement in the Bay Area for 2005 (kg/y). For estimation methods see relevant sections of this report.

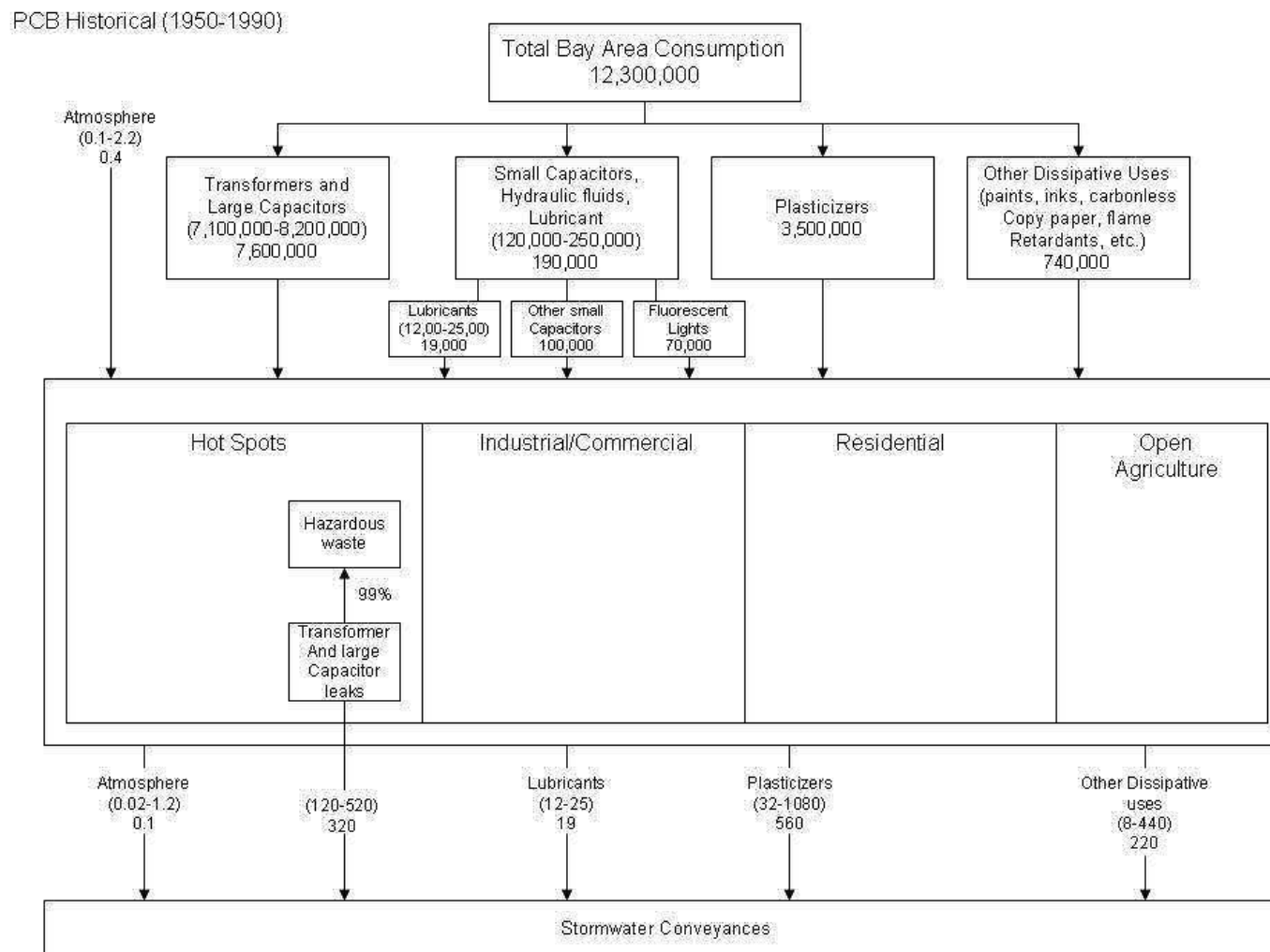


Figure 2-6. Estimates of the total PCB mass movement in the Bay Area for the period 1950-90 (kg over 40 years). For estimation methods see relevant sections of this report.

PCB Today (2005)

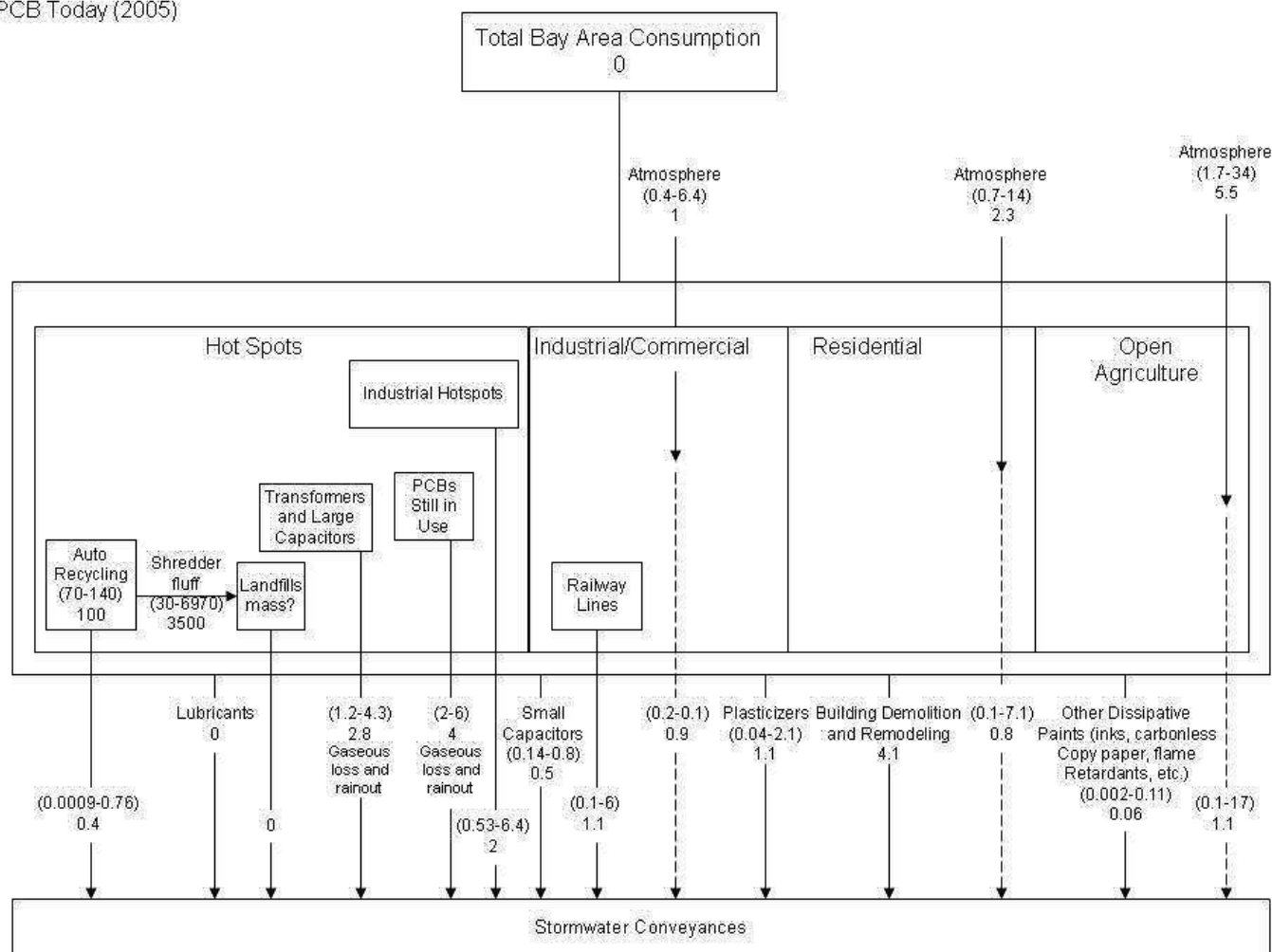


Figure 2-7. Estimates of PCB mass movement in the Bay Area for 2005 (kg/y). For estimation methods see relevant sections of this report.

2.3.6. Mass Associated With Hg Uses across the Urban Environment

2.3.6.1. Paint

Mercury was used as a mildewicide / fungicide in latex paints. Mercury use in paint was phased out in 1990 and 1991 (DTSC, 2002). Mercury was also used for pigment in antifouling paints. On average, Hg use in paint made up approximately 16% of total usage in California over the period 1970-90 (see DTSC, 2002, p 73). If it is assumed that this percentage use was characteristic of the period of peak use (1950-90) and that the Hg use in paint in the Bay Area is proportional to population, and scale use to population accordingly, total use of mercury in paint was approximately 86 t (Table 2-5) or 2.2 t/y.

Studies have shown that as much as 66% of mercury in paint is released to the atmosphere over the useful lifetime of the paint (see references in EPA, 1997). Thus, for the period 1950-90, approximately 57 t was released to the atmosphere from paint alone in the Bay Area. If we assume that emissions on days when it rains are immediately re-deposited on the local watershed surface, a portion of this atmospheric emission may become available for stormwater transport. We assume that on dry or lower rainfall days, emissions escape and mix becoming part of general background atmospheric deposition already accounted for in the atmospheric deposition section. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). Thus, about 1-3 t may have entered stormwater conveyances from gaseous releases from paint from 1950-90 or about 30-90 kg/y. From 1991, the addition of mercury to new paint was discontinued although there was likely an unknown but significant stock used up during the 90s. In 2000, it was estimated that about 0.45 t of paint was being discarded in California landfills; scaled to the population of the Bay Area this amounts to 0.08 t (80 kg) annually. If we assume that buildings are repainted every 20 years, much of the 2.2 t/y of Hg applied in paint for the last 5 years before the 1991 phase out could still be on exposed building surfaces. Using the same rationale described above for release and washout, about 1-4 kg/y might still be finding its way into stormwater from gaseous release from painted surfaces.

It is difficult to estimate the distribution of paint usage amongst land use classes but a first order estimate can be made by assuming a relationship with roof area. Residential land use areas are typified by 12-17% area in roofs and commercial/industrial land uses are typified by 20-26% area in roofs (Bannerman, 1993; Bannerman, 2003; Schueler, 1996). If we assume that buildings in commercial and industrial areas are 2-4 times taller, there would be 2-4 times more paint used per equivalent area. Thus based on land use estimates for the Bay Area (Table 2-1), between 60-73% of the Hg in paint would be have been applied to industrial/commercial areas. Today's loadings for release and washout from painted surfaces are therefore estimated at: industrial/commercial (0.6-2.9 kg; average 1.8 kg); urban (0.4-1.1 kg; average 0.8); open space/ agriculture (~0 kg).

Table 2-5. Estimates of dissipated uses of mercury in the Bay Area for the period 1950-90 and 1997 based upon estimates from national (EPA, 1997) and California inventories (DTSC, 2002) and scaled according to population trends.

Use	1970 (% usage)	1985 (% usage)	Average (1970-90) (%)	Mass (1950-90) (t)	1997 (% usage)	Mass (1997) (t)**
Other	24.4	6.4	15	82	42.9	2.7
Dental	5.0	3.6	4	23	11.7	1.0
Instruments	10.9	5.5	8	43	11.7	1.3
Batteries	24.8	64.3	45	236	0.0	0.0
Switches/thermostats	6.6	5.6	6	32	12.9	1.6
Lighting	1.7	2.2	2	10	7.4	0.3
Laboratory	5.0	1.8	3	18	13.5	0.6
Paint	21.8	10.6	16	86	0.0	0.0
	<u>100</u>	<u>100</u>	<u>100</u>	<u>530*</u>	100	<u>7.5</u>

* Estimate of total use of mercury in the Bay Area from 1950-90 (see section above on "Total Potential Mass of Hg and PCBs used in the Bay Area").

** Estimate based on scaling the mass in use in the U.S. to the population of the Bay Area.

2.3.6.2. Laboratory

Mercury is used in many reagents, slide preparations, electro-analyses, and sample preservatives. During the period of peak usage (1950-90), an estimated 18 metric t of Hg was used in the Bay Area. There is very little information available on losses of Hg from general laboratory use. The EPA report to congress (EPA 1997) promulgated an estimate of 4% based on a very old estimate. If we assume that 1-10% of this is released through building ventilation and fume-hood extraction, then between 180-1,800 kg or 4.5-45 kg/y would have been released to the atmosphere over 1950-90. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). If we also assume that laboratory releases do not occur on wet season public holidays or weekends, this number is reduced to 1.5-4%. Thus between 3-72 kg or 0.08-1.8 kg/y might have been released to the stormwater conveyance system. Unlike Hg use in paint and batteries, Hg use in laboratories has remained relatively constant over the past 30 years. Presently an estimated 0.6 t is used in the Bay Area annually. Although most of the mercury used in laboratories passes into the wastewater stream, a small amount probably passes into the atmosphere via fume-hoods and air-conditioning systems and this might locally deposit and be washed into local drainages during storms. Following the same logic as above, this would amount to

approximately 0.2-1.4 kg/y presently or an average of 1 kg in the industrial/commercial land use sector where laboratories are located.

2.3.6.3. Lighting

Because Hg conducts electricity well, it is used in a variety of electrical applications, such as electrical lights and switches. Mercury is a component in many lamps, including fluorescent, high-pressure sodium, mercury arc, metal halide, neon, and ultraviolet disinfectant lamps. Today, an average fluorescent lamp contains approximately 10 to 21 mg of mercury (DTSC, 2002) down from historic levels of 48 mg (U.S.EPA, 1997). However, mercury cannot be eliminated from fluorescent lamps, as it is essential to proper functioning. Historically (1950-90) approximately 10 t or an average of 250 kg/y of mercury has been used in the Bay Area for lighting (Table 2-5). Currently, approximately 528 million lamps are sold in the U.S. annually. Scaling this to the 2000 U.S. census figures (a ratio of 0.024 or 2.4%) this equates to 12.7 million lamps or about 130-270 kg – similar to the estimate (1.2 t for California provided for 2001 (DTSC, 2002) and scaled to the Bay Area population (190 kg). Unlike waste associated with batteries and paint, the percentage of waste in the solid waste stream associated with lighting is increasing (1989: 4%; 2000: 24%). Nationally the EPA has estimated that 6% of lamps will be broken during use or disposal and that overall 16.5% of the Hg used in lighting ends up as air emissions. This equates to about 1,650 kg or 40 kg/y for the period 1950-90 and about 20-40 kg/y currently. It is very difficult to estimate the proportion of this that directly enters the stormwater systems. One method is to assume that all Hg entering the atmosphere during rain days enters the stormwater conveyance system. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we assume that no transport or breakage occurs on wet season weekends or public holidays this estimate is reduced to 11-13% or 2-5 kg (average = 3.5 kg) (see paint section for rationale). This does not take into account illegal dumping and breakage much of which occurs near or in creeks or in dumpsters – a likely very important pathway for which we have no data. In the absence of any data, we assume that 80-90% of fluorescent lighting is used in industrial/commercial land uses. A number of present initiatives are addressing pollution from disposal of lamps. SB 1180 requires a retail purchaser of a fluorescent lamp to pay a fluorescent lamp recycling fee to the retail seller. AB 1699 (LS: 03) establishes the Mercury Recycling Enhancement Act of 2003. It prohibits any person from disposing of a fluorescent lamp in a solid waste facility.

2.3.6.4. Switches and Thermostats

Hg is used in tilt switches because it is liquid at room temperature and conducts electricity very well. Tilt switches are small tubes with electrical contacts at one end of the tube. As the tube tilts, the mercury collects at the lower end, providing a conductive path to complete the circuit. When the switch is tilted back, the circuit is broken. Mercury tilt switches have been used in light switches, thermostats, off-balance switches in

household appliances, trunk light switches in automobiles, thermocouples, among others. Typical masses used are: thermostats (3-6g), freezer light switches and washing machines switches (2g), silent switches (2.6g), and flame sensors in gas ranges (2.5g) (Wisconsin DNR, 1999). Based on scaling estimates for California (DTSC, 2002), it is estimated that a total of 32 t was used in the Bay Area over the 1950-90 period or 0.8 t/y. The use of Hg in household silent switches was discontinued in 1991 (Wisconsin DNR, 1999). Tilt switch use in thermostats was largely replaced by digital technologies in the 1990s, but even in 1997 it is estimated that 1.6 t was still deployed in older switching devices in the Bay Area (Table 2-5). Studies indicate that the lifetime of electrical switches is very long – the average unit life for mercury thermostats exceeds 20 years, with upgrading, remodeling or building demolition being the principal causes for removal from service. It is difficult to estimate how much Hg associated with these switches might find its way into the stormwater conveyance systems of the Bay Area. One might assume that the potential for breakage of the little glass tubes in tilt switches is lower than for fluorescent light tubes because of the differences in relative strength associated with dimensional properties. In addition, there is approximately 3 times more Hg entering the municipal solid waste stream from electric lighting than for switches and thermostats, despite a greater overall use for switches and thermostats (DTSC, 2002). In the absence of better information, we will use the same factor we used for fluorescent lighting (16.5% entering the atmosphere and 11-13% of this immediately re-depositing) to derive an estimate of the Hg entering stormwater conveyances, except that we assume that only half the rate of breakage and uncontrolled dispersal occurs. This equates to 0.23-0.28 t for the period 1950-90 or 3-4 kg/y and a current estimate of 9-11 kg (Average = 10 kg). It is also difficult to estimate how this is dispersed across the industrial, commercial and residential sectors. Unlike fluorescent lights which were largely used in industrial and commercial building rather than in residential applications, tilt switches were/are more universally used. A first order estimate can be made by assuming a relationship with roof area – the greater the roof area the more switches in a building. Residential land use areas are typified by 12-17% area in roofs and commercial/industrial land uses are typified by 20-26% area in roofs (Bannerman, 1993; Bannerman, 2003; Schueler, 1996). Using these roof areas, it is estimated that approximately 41-43% of the Hg associated with switches would be associated with industrial and commercial land uses in the Bay Area with the rest associated with residential urban land use. How does the future look? AB 1415 (LS: 05) will prohibit a person from selling, offering to sell, or distributing for promotional purposes in this state, a mercury switch or mercury relay, as defined. The Bill would exclude from this prohibition a switch or relay, as specified, that was in use prior to January 1, 2007

2.3.6.5. Batteries

Battery production was the single largest use of mercury up until its use was largely phased out in the early 90s. It is estimated that 236 t was imported into the Bay Area for use in batteries between 1950-90 (Table 2-5). This is equivalent to 6 t/y. Hg was used in dry-cell batteries as an active electrode, to protect the zinc cathode from oxidation and to prevent evolution of carbon dioxide gas in alkaline and carbon-zinc batteries (DTSC, 2002). The Hg battery invented by Samuel Ruben in the 1940s had the desirable property

of a stable voltage output down to the last 5% of its life. Soon after the war, the first button Hg cells were introduced and used in small devices such as hearing aids. Through to 1991, batteries contained an average of 0.5% Hg by weight. After 1991, industry changed its formulations so that, with the exception of button and coin cells, all batteries did not exceed 0.025% Hg by weight (U.S.EPA, 1997) or no more than 25 mg per battery (DTSC, 2002), which was at least 20 times less mercury. New button batteries contain approximately 25 mg of mercury. Button batteries manufactured before 1996 contain significantly more (DEQ, 2003). The U.S. demand for batteries reached about 10 billion in 2002 and is increasing by 6% per year, mainly due to the demand from small consumer appliances such as cell phones, digital cameras and MP3 players. By 2009 it is expected that such consumer applications will account for more than 70 percent of all primary battery sales. Using the maximum permissible mass of Hg in a modern battery (0.025% by weight), assuming an average weight of 50 g/battery and scaling the national sales figures to the Bay Area population, it is estimated that 1.5 t of mercury would have been imported into the Bay area in 2002 in the form of batteries for personal appliances.

Given the stability of batteries, it is almost impossible to estimate what percentage of Hg in batteries makes it into the stormwater conveyance system. Casual observations of areas near bus stops and other public transport points of entry/exit in the Bay area do indicate a level of discard on streets curbs, gutters and sidewalks (Figure 2-8). As a hypothesis we will assume that between 0.01-0.1% (1:10,000-1:1,000) of the batteries sold since 1950 were discarded in a manner that allowed battery Hg to enter the stormwater conveyance system. This equates to 0.6-6 kg/y for the 1950-90 period and 0.15-1.5 kg presently.



Figure 2-8. Example of a battery discarded on roads and pavements near areas of public transportation embarkation. The battery shown have been run over many times squashing and canister and allowing the contents to leak out.

2.3.6.6. Instruments

Mercury is used in many types of instruments such as barometers, hydrometers, manometers, pyrometers, sphygmomannometers, and thermometers. To a large extent these have been phased out and replaced by electronic or solid state technologies. However, prior to the 1990s, mercury containing instruments were common in the home, in agriculture, and in laboratories, hospitals, and veterinary hospitals. For example, studies in major milk producing states have found that about 10-20% of dairy farms were still using mercury filled manometers (containing approximately 340 g of Hg) to measure the vacuum on milking equipment. Similar types of measurement devices were used in gas works, laboratories, and hospitals. It is estimated that in the Bay Area, approximately 43 t or 1.1 t/y was used in instruments over the period 1950-90 (Table 2-5). In 1997, this figure had slightly increased to 1.3 t/y. Since then, there have been a number of California State Bills that have placed restrictions on the manufacture, sale and use of instruments containing Hg, have encouraged replacement, and introduced recycling programs. In a similar manner to fluorescent light tubes and switches and thermostats, it is assumed that the main pathway of Hg into the environment is presently illegal dumping, and handling during recycling. Unlike switches and thermostats, many Hg-containing instruments are fragile and not easily handled without the risk of breakage. Instruments are likely to be similar to or even more problematic than fluorescent lights in terms of breakage during handling and transport. In the absence of better methods, we will use the same factor we used for fluorescent lighting (16.5% entering the atmosphere overall and 11-13% of this immediately re-depositing during rain events and entering the storm drainage systems) to derive an estimate of the Hg entering stormwater conveyances. This equates to 18-28 kg/y for the historic period and currently.

2.3.6.7. Dental

Because mercury combines with other metals to form stable alloys which have desirable filling characteristics (curing and expansion), it has been widely used in dentistry for the past 150 years. Dental amalgam is a mercury alloy prepared by mixing an approximately equal part of elemental liquid mercury with an alloy powder composed of silver, tin, and copper. The use of mercury in dental amalgams is being seriously debated worldwide. Many governments have taken steps towards eliminating or limiting amalgam use. In California, Senate Bill 134 (Chapter 532, Statutes of 2002) requires a disclosure form signed by all patients regarding the comparative risks and efficacy of various types of dental restorative materials. AB 999 (LS: 03) established insurance requirements for alternatives to mercury amalgam fillings. AB 966 (LS: 05) requires the Department of Health Services to adopt regulations establishing standards regulating the discharge of mercury and other byproducts related to the use of amalgam in the process of providing dental and related services. The Bill will also preclude health insurance coverage or health care service plans entered into or amended on or after January 1, 2006, from denying insurance coverage for amalgam alternatives based upon the cosmetic aspects of the alternatives. Other than atmospheric losses during handling and disposal, there is little opportunity for Hg used in dentistry to get into the stormwater conveyance system. In the

absence of better data, it is assumed that the same kinds of losses might occur from dental facilities as occur from laboratories. Historically there was approximately 23 t Hg used for dental purposes in the Bay Area between 1950-90 (Table 2-5) or about 0.6 t/y. In 1997, the use estimate was about 1 t (DTSC, 2002). If we assume that 0.1-1% of this is released through building ventilation and fume-hood extraction then between 23-230 kg or 0.6-6 kg/y would have been released to the atmosphere over 1950-90. Note EPA (1997) suggested 2% in their report to congress but commented that it may have been an underestimate given some is transported to landfills and medical waste incinerators. Given, dental offices are now trapping and disposing of waste more carefully in response to the various California Bills and waste rules, <2% seems reasonable. It is interesting to note that for many wastewater treatment plants (for example the Hyperion Plant in Southern California, dental sources are still considered a major source (M. Strenstrom, personal communication 2006). There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). If we also assume that dental releases do not occur on wet season public holidays or weekends, this number is reduced to 1.5-4%. Thus for the historic period, this equates to about 0.3- 9 kg or 0.009-0.2 kg/y. The present estimate of Hg release to stormwater from dental practices is derived differently by scaling the estimated national air emission release (0.6 t) (EPA, 1997) to the year 2000 Bay Area population (2.4% of the nation) and assuming the same rainout process. This equates to 0.2-0.6 kg, similar to the historic estimate.

2.3.6.8. Gasoline and Dieseline Consumption

Mercury entering the Bay Area from the consumption of gasoline and dieseline in motor vehicles is not included in the over all estimates of Bay Area Hg consumption provided above (530 t). There have been estimates made of Hg supply to the watershed/airshed from mobile combustion emissions for the U.S. (EPA, 1997), California (DTSC, 2002), and the San Francisco Bay Area (McKee et al., 2003; Conaway, 2005). Nationally, the EPA reviewed studies on tail pipe emissions and questioned the reliability of data, so there are uncertainties in the estimates calculated here. We used an estimate of 1.3 µg/km-traveled from a study done in the early eighties. Estimates of annual total travel distance in the Bay Area are provided by the California Department of Transportation. Distances have been increasing by an average of about 15% every 5 years since 1980 when the first statistics were published (Caltrans, 2005). Using a relationship between population and distance traveled, it is estimated that 1,800 billion km were traveled in the Bay Area during the period 1950-90 or an average of 45,300 million km/y. This equates to 2,350 kg Hg or 59 kg/y. This estimate seems a little high, but given uncertainties caused by changes in fuel consumption and engine technologies it is difficult to develop better estimates. Estimates of Hg air emissions for on-road mobile for California in 2000 were 161 kg (356 lbs) (DTSC, 2002). Scaling these to the Bay Area sources using population amounts to approximately 30 kg/y; about half the estimate derived from national figures for 1950-90, but in remarkable agreement considering the uncertainties. Perhaps the most accurate estimate for the Bay Area is found in a recent study of Hg

concentrations in Bay Area gasoline and diesel (Conaway et al., 2005). They found Hg concentrations ranged from 0.08-1.4 ng/g. They used a mass density of gasoline of 0.7 g/cm³ and the recent annual Bay Area annual fuel consumption estimate (13 billion liters) to estimate an annual Hg emission of 1-13 kg (average 5 kg and increasing by about 1 kg each 5 years) (Conaway et al., 2005). For lack of better information, we will use the low end of the California estimate and the high end of Conaway et al. (2005) as the range (13-30 kg) of the historic period. Like other gaseous emissions it is hard to estimate the amount that actually enters the stormwater conveyance network but using the rain day thought experiment described several times above, a first order estimate is presented. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only on those days when there is enough rain to cause runoff from roadways (>2.54 mm or 0.1 in) for lower rainfall urban and industrial areas, this number is reduced to 33-44 days or 9-12% of the year (see paint section for rationale). Roadways are designed to efficiently shed water thus it is assumed that all Hg deposited on wet days has the potential to enter the stormwater conveyance. Thus, it is estimated that fuel consumption may have provided about 1-4 kg/y during the 1950-90 period and 0.1-2 kg/y (average = 1 kg/y) presently. In terms of dispersal, most of this Hg would be dispersed along arterial freeways (on the industrial and commercial Bay margin) and in other urban areas but the distribution of distances traveled in these areas is presently unknown. In terms of the amount that actually enters the stormwater conveyances, we expect a runoff coefficient of 95% which is within error bounds so the estimates above were not adjusted.

2.3.6.9. Other uses

Over the past decade, the primary use of mercury in the urban environment has switched from batteries and paint to “other uses” largely because of the phase out and new laws associated with batteries and paint. Historically other uses are estimated as 82 t for the period 1950-90 or 2 t/y and the most recent estimate is 2.7 t/y (DTSC, 2002; Sznoppek, 2000). Unfortunately, neither of these references contains an inventory of the types of uses in this category so it is impossible to determine the likely environmental fate. As a ‘place holder’ we will assume that 0.01-0.1% has the potential to enter the stormwater conveyance system or about 0.3-3 kg/y (average 1.5 kg/y) for the historic and current periods. Applying the rainfall/runoff thought experiment, this would equate to 0.006-0.18 kg/y (average = 0.09 kg/y) entering stormwater conveyances.

2.3.7. Mass Associated With PCB Uses across the Urban Environment

2.3.7.1. Controllable Closed Systems (Transformers and Large Capacitors)

Transformers and capacitors are “Controllable Closed Systems” (Erickson, 1992). Approximately 60% of all PCBs produced in the U.S. were used in closed applications

(Keeler et al., 1993). This compares similarly with Japan (66%) (UNEP, 1999), Germany (57%) (OSPAR, 2004), however, data from England appears to differ (18%) (OSPAR, 2004). In Germany about 1.4% of the closed system use was in small capacitors and light ballasts. If we assume a similar use distribution for the U.S., then the use in transformers and larger capacitors would be 58-59% of the total use.

Pacific Gas and Electric (PG&E) was probably the largest user of PCBs in the Bay Area (Figure 2-9). It is not known how many transformer and capacitor applications are inside buildings or in some other way isolated from potential leakage to the soil environment. PG&E has a rigorous inspection and clean up program in place, so it is unlikely that all lost mass associated with PG&E operations would make it into the adjacent soil environment. To a large extent, PG&E have removed a lot of transformer PCB mass from service through change-out programs. In the situation of mass loss to soils, there have been many site cleanups that have involved the removal of the polluted soil material. For example, their latest environment report (PG&E, 2004) lists activities for the past three years (2002-04) that suggest approximately 0.33-0.75 t/y of PCBs have been properly disposed of in California through soil removal. Scaling to the Bay Area, this would equate to 0.07-0.15 t/y. These cleanups, however, are typically completed to various levels depending on the end uses of the site and other adjacent land uses. For example, the EPA superfund record for Lorentz Barrel and Drum Co (EPA 1993) described precondition onsite concentrations of 0.23 - 380 ppm and offsite parameter concentration as high as 1.2 ppm. For this particular site, the EPA adopted the RWQCB recommended clean up level of 1 ppm. Clean up options including capping soils at concentrations between 1-50 ppm on site. Therefore, even after cleanup there is likely still PCB mass available for mobilization into the storm water system although clearly the mass available for transport is much reduced.

To estimate the total mass used in the Bay Area, we assumed that over the period of peak usage (1950-90) 58-59% of the total use of 12,300 t (see previous section) was used in the U.S. for transformers and large capacitors. Scaling to population, this equates to 7,100-7,300 t in the Bay Area. This compares well to another U.S. estimate of PCBs used in transformers alone (147,500 t) (see references in Harrad, 1994) which when scaled to the Bay Area equates to 3,250 t. There was also a U.S. estimate of PCB use in large capacitors of 225,000 t, which scaled to the Bay Area amounts to 4,950 t adding to a total of 8,200 t, in good agreement with the earlier estimate. Combining these estimates gives 7,100-8,200 t (average = 7,600 t). Inevitably, some amount of PCBs used in transformers and large capacitors is spilt or leaks into the environment (Harrad, 1994). According to references cited in OSPAR (2004), about 2% of transformers and 3% of large capacitors in the U.S. had leaks. In a U.S. study, about 0.05% was estimated to leak from transformers and 0.35% from large capacitors each year (see references in Harrad, 1994 and page B-2 of EIP and Associates, 1997). This equates to approximately 1.4-1.6 t/y of PCBs from transformers for the Bay Area and approximately 15-17 t/y for large capacitors. If we consider a 30 year service period this would amount to 40-50 t for transformers and 450-510 t for large capacitors.

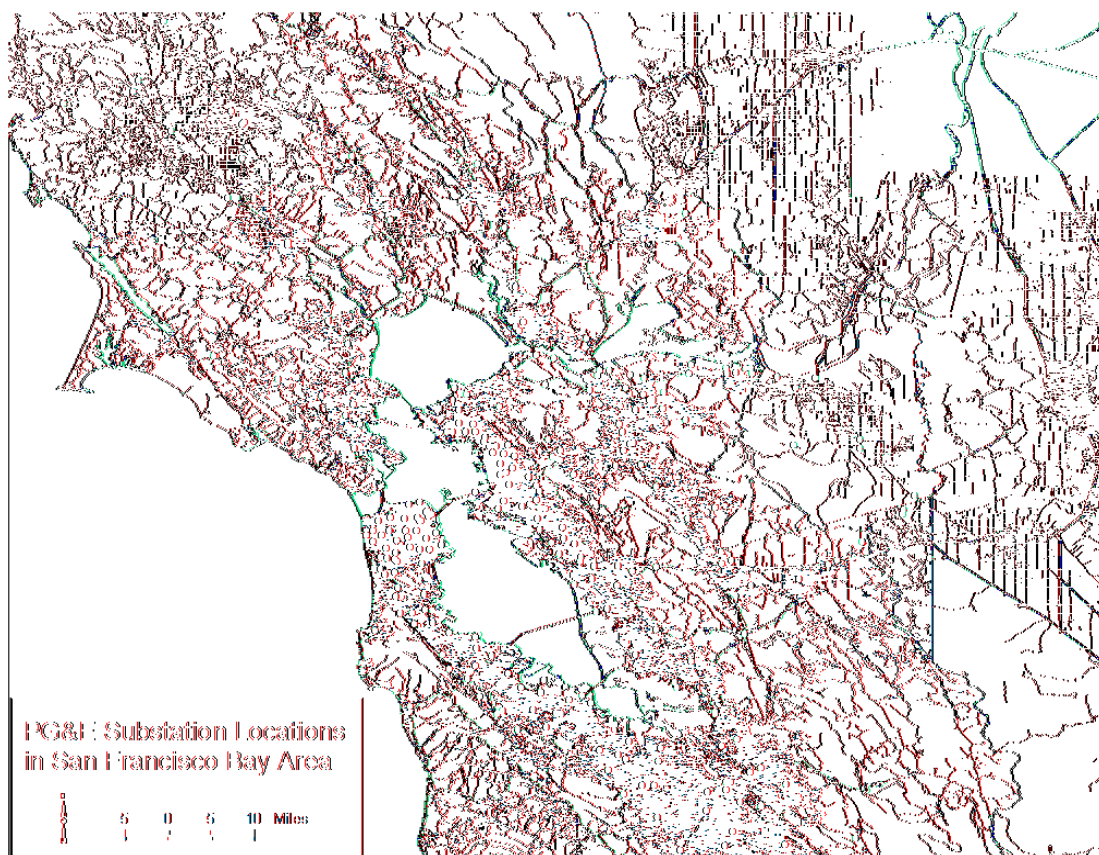


Figure 2-9. Pacific Gas and Electric (PG&E) properties in the Bay Area. Note, the data on the locations have not been fully quality checked.

In terms of the fate of these spills, we will assume that when spills occur, 99% of the materials are successfully cleaned up. We will also assume that only 1% of the remaining leaked material remains on the “erodable” soil surface. To estimate washoff to the stormwater system by this surface pathway, we assume that all mass leaked is outdoors and that runoff coefficients approximate the fraction that can enter the local storm drains. Thus we estimate a release of 0.02-0.09 kg/y (average = 0.06 kg/y) of PCBs from transformers and 0.16-0.88 kg/y (average = 0.52 kg/y) of PCBs from large capacitors. Keeler et al. (1993) estimated that about 0.3% of PCBs released during spills enters the atmosphere. Using this estimate, a total release would be about 4.2-4.8 kg/y (average = 4.6 kg/y) of PCBs from transformers and 45-51 kg/y (average = 48 kg/y) of PCBs from large capacitors. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). Thus we estimate an additional 0.08-0.29 kg/y for transformers and an additional 0.9-3 kg/y for large capacitors that might find it way to the stormwater conveyance from atmospheric wash out. Therefore the total PCB load entering

stormwater conveyances from controllable closed systems is estimated to be 1.2-4.3 kg/y (Average = 2.8 kg/y).

2.3.7.2. Uncontrollable Closed Systems (Small Capacitors, Hydraulic Fluids, and Lubricants)

Small capacitors containing < 3.36 kg of dielectric fluid (EPA definition), hydraulic and heat transfer fluids, and lubricants are part of the uncontrollable closed systems inventory (Erickson, 1992). Many devices that use hydraulic fluids and lubricants are designed to leak slowly. Small capacitors were used in applications such as electric motors where a boost of power is required at start up. Prior to 1978, all light ballasts (specific small capacitor designed to provide extra voltage to get a fluorescent light started) used PCBs as a dielectric fluid. In addition, the “potting” compound that surrounds the light ballast also contained PCBs. Unlike controllable closed systems (transformers and large capacitors), the use and disposal of small capacitors, hydraulic fluids and lubricants is much harder to track and regulate. The ultimate fate is usually disposal via the mixed general urban waste stream to landfills, illegal dumping, and minor hazardous waste disposal.

Again it is difficult to determine the total mass used in each of these applications in the Bay Area. Looking at the problem from the top down, if we assume use characteristics similar to Germany, small capacitors, including light ballast capacitors, would account for about 1-2% of total PCB usage in the U.S. Scaling this to the Bay Area equates to about 120-250 t over the period 1950-90. The total amount for each light tube is approximately 0.9 ounces (25 g) (DTSC, 2003). If we assume 125 million fluorescent light assemblies were sold in the US prior to 1980 and scale this number to the Bay Area, this would equate to up to 2.25 million light assemblies or approximately 70 t of PCBs – in reasonable agreement to our estimate based on 1-2% total PCB use. The mass associated with hydraulic fluids and lubricants is difficult to determine for the Bay Area. In Germany, 14.3% of the total PCB usage was in hydraulics. In 1970 (the year of peak production and use), U.S. usage was 56% in dielectric fluids, 30% in plasticizers (primarily carbonless copy paper), and 12% in hydraulic fluids and lubricants (see references in UNEP, 1999). Overall, it appears that about 10% of the total 1950-90 usage was hydraulic fluids and lubricants (Keeler et al., 1993). Scaling this to the Bay Area, amounts to 30 t.

Unlike large capacitors, small capacitors are less prone to leakage, except during waste disposal processes that involves shredding (Harrad, 1994). However, there is growing evidence that some leakage occurs from light ballast capacitors and associated potting compounds (DTSC, 2003). Annual emissions in Belgium amount to about 0.3% of total product use (OSPAR, 2004). Scaled to the Bay Area, this equates to about 370-780 kg would be released over the peak period of use or about 7-14 kg/yr. In Britain it has been estimated that about 0.05% of PCBs usage in small capacitors is released into the atmospheric environment annually during the waste recycling processes. If we assume a similar loss for the Bay Area, this would amount to 60-125 kg or about 1.1-2.3 kg/yr

since 1950 entering the atmosphere. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). Thus, our estimate for PCBs from small capacitor use entering the stormwater conveyances of the Bay Area would be about 0.14-0.8 kg/y in total and about 0.02-0.14 kg/y for waste recycling alone.

Leakage associated with applications that use hydraulic fluids and lubricants has not been reported. If 90% of the applications are housed and disconnected from the environment and 1% of fluid leaks on average from all applications, we would estimate a total of 12-25 kg to be released or about 0.5 kg/y for the period 1950-90. Assuming a useful service period of 5-10 years maximum in some applications and likely <1 year in other applications, it would seem likely that the current releases would be zero because all mass would have already been passed in to the urban and industrial waste cycle.

2.3.7.3. Dissipative Uses (Plasticizers)

Plasticizers were the primary dissipative use in the U.S. PCBs in plasticizers are in direct contact with the environment, and there is usually poor recovery and disposal at the end of useful life. PCBs were used extensively in polyvinyl chloride (PVC), neoprene, and other chlorinated rubbers (UNEP, 1999). They were also used in polyurethanes and polycarbonates, and plastic decorative articles (EIP Associates, 1997). Assuming usage in the Bay Area proportional to population, it is estimated that 3,500 t (25% of the total estimated PCB use for the Bay Area) would have been used in plastics and rubbers over the period of peak usage (1950-90). Given the magnitude of the application and such widespread (dissipative) use, plasticizers might have had a large impact on environmental PCBs burdens. Presently we have found no information regarding PCB release from plasticizers. Concentrations of PCBs in rubber coatings for metals can be 40% by weight, in PVC (5-8%), in Thiokol (rubber joint concrete seal) (1-40%), in caulk (15-20%), and in cambric tape (used in high voltage electrical cables) (6-11%) (EIP Associates, 1997). In a recent review of studies on caulking and sealants, LWA (2005) found concentrations ranging between 2,700-310,000 mg/kg. A Swedish study of 1,348 joint sealants found concentrations ranging between 0-55% by weight (Kohler et al., 2005). There is evidence that uses of these products in buildings constructed in the 1960s and 1970s are still leading to elevated concentrations in air (Benthe et al., 1992; Heinzow et al., 2004; Kohler et al., 2005). There is also evidence that concentrations in soils surrounding buildings are elevated where PCBs were used in construction materials (Priha et al., 2005). Their data shows mean concentrations in soils at the base of walls of 12 mg/kg diminishing to 0.5 mg/kg at a distance of 5 m. These data provide evidence of direct dispersal into likely hydrologically-active areas.

It is difficult to determine what mass of PCBs associated with plasticizers might have entered the stormwater conveyances of the Bay Area over the period 1950-90. Estimates of PCBs entering the atmosphere from Other Dissipative Uses (paints, concrete, and glue)

in Norway and Belgium amount to 0.05-1% of the total mass usage (OSPAR, 2004). If we assume similar characteristics for the Bay Area, approximately 1,750-35,000 kg or 40-900 kg/y might have reached the atmosphere. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). Thus, our estimate for PCB entry into stormwater conveyances is 0.8-54 kg/y (average = 27 kg/y) for the period 1950-90.

The estimation of the direct contemporary supply of PCBs from plasticizers to stormwater conveyances is even more difficult. If the usable life of plasticizers is up to 40 years (OSPAR, 2004), and the majority of the usage was in the 1960s and 70s, one would assume that 99.9% total product use should have been forwarded to waste treatment facilities. Thus we estimate a current release to stormwater via immediate atmospheric re-deposition of 0.035-2.1 kg/y (average 1.1 kg/y). The LWA review cited one study that described losses of PCBs to soil and water during demolition and used this to estimate a contemporary loading of 2.1 kg/y for demolition and an additional 2 kg/y for remodeling and building maintenance for the Bay Area (LWA, 2006).

2.3.7.4. Other Dissipative Uses (paints, inks, carbonless copy paper, flame retardants)

PCBs were used in 1000s of products (dissipative uses) other than plasticizers including paints, inks, carbonless copy paper, and flame retardants. Just like the plasticizers, PCBs in these uses are often in direct contact with the environment, and there is usually poor recovery and disposal at the end of useful life. Overall use in the U.S is estimated at about 5% or 32,000 t (Keeler, 1993). If this mass is scaled to the population of the Bay Area for the period 1950-90 (2.2% of the total U.S population), the consumptive PCB use for Other Dissipative Uses is estimated to be 704 t. It is difficult to determine what proportion might have entered the stormwater conveyances of the Bay Area over the period 1950-90. Estimates of PCBs entering the atmosphere from Other Dissipative Uses (paints, concrete, and glue) in Norway and Belgium 0.05-1% of the total mass usage (OSPAR, 2004). If we assume similar characteristics for the Bay Area, approximately 350-7,000 kg or 9-180 kg/y might have reached the atmosphere. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). Thus, our estimate for PCB entry into stormwater conveyances is 0.2-11 kg/y (average = 6 kg/y) for the period 1950-90. In a similar manner to plasticizers, the useable lifetime for the Other Dissipative Uses is <40 years and more likely <25 years. Therefore most (<99%) of the mass associated with these uses should have already passed into the urban waste stream and been buried in landfills. Thus the present day potential mass entering storm water conveyances is estimated at 0.002-0.11 kg/y. At a glance, this estimate seems low, however PCB

concentrations in these products are much lower than in calking – a fact that might help to explain the relatively low estimated contribution to stormwater.

2.3.8. Estimated Mass Associated With Hg and PCB Contaminated Areas

2.3.8.1. Regulatory Data Bases

A number of regulatory authorities in California keep data bases that record and describe, with varying levels of detail, Hg and PCBs or soils and water associated with spills, recycling yards, historic uses of these substances, and current emissions. These data bases were searched to try to discover many possible more contaminated locations in the Bay Area (Hg: Table 2-6; PCBs: Table 2-7). This inventory of potentially contaminated areas builds upon the previous efforts by STOPPP (2002) for San Mateo County, Salop et al. (2004) for Alameda County, Dovzak and Sommers (2004) for Contra Costa County, Kleinfelder (2005) for Ettie Street Pump Station catchment, and Hetzel (2004) for current PCB uses. It is interesting to note that there appears to be fewer potentially Hg contaminated areas than PCB areas. This is consistent with the next chapter of the report that describes the world literature on Hg and PCBs in soils, street dusts and other urban media.

2.3.8.2. Estimation of Mass Entering Stormwater Conveyances from the data base of potentially contaminated areas

The estimation of mass entering stormwater conveyances from potentially contaminated areas is made extremely difficult because in most cases data are lacking on concentrations, off site flows, or even areas of pollution (Table 2-6 and Table 2-7). Here we will use a very simple method that combines an estimate of area, an estimate of sediment export, and an estimate of soil concentrations. The median size of these areas in the Bay Area is 11 ha (0.11 km²) with a quartile range of 2.3-60 ha (0.023-0.60 km²).

The estimate of halo area is made using information found in a later section of the report “Hg and PCB Pollution in the Urban Environment”. In that section the world literature was reviewed for Hg and PCB dispersion around known use areas. It was found that area contaminated with Hg exhibit a halo effect up to 3 km down wind and more typically 1-2 km. So we will assume that every Hg contaminated area in the Bay Area has a width of 0.5 km in the cross wind direction and length of 1.5 km in the downwind direction (Best area = 0.75 km²; Range = 0.33-3 km²). Adding this to the estimated contaminated area above gives a best estimate of area of 0.86 km² and a range of 0.35-3.6 km². There are a total of 28 identified Hg contaminated areas in the Bay Area. We only know the area of 3 (together 9 km²). For the other 25 we will use the estimates of area above for a single contaminated area and halo making a best estimate of total area of 31 km² of polluted watershed surface (Range = 18-99 km²).

Table 2-6. Hg contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (mg/kg)
AB&I FOUNDRY	TRI (CAS # 007439976)		7825 SAN LEANDRO STREET, OAKLAND			
ADVANCED RADIATION CORP	TRI (CAS # 007439976)		2210 WALSH AVE, SANTA CLARA			
AERC.COM INCORPORATED	TRI (CAS # 007439976)		30677 HUNTWOOD AVENUE, HAYWARD			
AGILENT TECHNOLOGIES INCORPORATED	TRI (CAS # 007439976)		350 WEST TRIMBLE ROAD, SAN JOSE			
ALMADEN QUICKSILVER COMAPNY PARK SITE	SLIC	43S0740	ALMADEN RD, SAN JOSE		Mercury mine	
ALMADEN QUICKSILVER COUNTY PARK	DTSC CalSites TRI (CAS # 007439976)	43100001	ALMADEN ROAD & HICKS AVENUE, SAN JOSE			max. 1,700 ave. 39- 420
CHEVRON TEXACO CORPORATION			100 CHEVRON WAY, RICHMOND			
CLOROX COMPANY	DTSC CalSites	01280083	850 42ND AVENUE, OAKLAND		Mercury used in liquid bleach production	
DEL GRANDE PROPERTY **(TPHg)	SLIC	01S0461	4831 MILDRED DR, FREMONT			7-770,000 ppb
FEDERATED METALS CORPORATION	DTSC CalSites	38330005	1901 CESAR CHAVEZ, SAN FRANCISCO	0.13	Production of lead and brass ingot	
FREMONT CITY OF **(TPHg)	SLIC	01S0459	4488 NOROCCO CIRCLE, FREMONT			550-230,000 ppb
GAMBONINI MINE/ALMADEN MINE	SLIC	21S0021	VALENCIA ST, NOVATO		Former mercury mine	
JEFFERSON SMURFIT CORPORATION	TRI (CAS # 007439976)		2600 DE LA CRUZ BOULEVARD, SANTA CLARA			
SANTA CLARA MILL	TRI (CAS # 007439976)		1015 COMMERCIAL ST., SAN CARLOS			
KELLY MOORE PAINT COMPANY INCORPORATED	Other		30101 Industrial Parkway SW, Union City			
Lehigh Southwest Cement Company			WEST OF ENTERPRISE DRIVE, NEWARK		Disposal of waste	
LESLIE SALT/FMC MAGNESIA WASTE PILE	DTSC CalSites TRI (CAS # 007439976)	01280072				
LORAL CORPORATION SPACE SYSTEMS			3825 FABIAN WAY, PALO ALTO W END OF TENNESSEE STREET, MARE ISLAND, VALLEJO		Naval shipyard	
MARE ISLAND NAVAL SHIPYARD	DTSC CalSites	48970002	MOFFETT FIELD, Santa Clara, Santa Clara	890.69	Various closed and open uses	0.1 - 6.2 (Landfill soils)
MOFFETT FIELD	EPA Superfund TRI (CAS # 007439976)		399 JAVA DRIVE, SUNNYVALE			
PERKIN ELMER ILC	TRI (CAS # 007439976)		44370 CHRISTY ST, FREMONT			
PERKIN ELMER OPTOELECTRONICS	DTSC CalSites	41280138	200 VALLEY DR., BRISBANE		Mercury recycling facility	
QUICKSILVER PRODUCTS, INC						

Table 2-6 continued. Hg contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (mg/kg)
RHONE POULENC/(ZOECON) SANDOZ	EPA Superfund		1990 Bay Road, East Palo Alto, California CORDELLIA & GREEN VAL @ I-680,	5.34	Pesticide manufacture, leaking underground storage tanks	1900
SHELL TANKER SPILL **(TPHg)	SLIC	48S0015	CORDELIA			73ppm
SPACE SYSTEMS / LORAL - B12	TRI (CAS # 007439976)		1034/1036 E MEADOW CIRCLE B12, PALO ALTO			
UNITED STATES COAST GUARD	DTSC CalSites	01970014	ELEVENTH COAST GUARD DISTRICT, B. 50-6, ALAMEDA		Early buoys used batteries containing mercury	
US STEEL POSCO INDUSTRIES	TRI (CAS # 007439976)		900 LOVERIDGE ROAD, PITTSBURG			
VALERO BENICIA ASPHALT PLANT	TRI (CAS # 007439976)		3001 PARK ROAD, BENICIA			

Table 2-7. PCB contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (ug/kg)(ppb)
AGNEWS STATE HOSPITAL	DTSC CalSites	43800001	AVENUE A AND LICK ROAD, SANTA CLARA		Former waste incinerator	
ALAMEDA NAVAL AIR STATION	DTSC CalSites	01970005	ATLANTIC AVENUE, ALAMEDA (2,616 acres)	89.07	Hazardous waste disposal site on military base	
AMCHEM PRODUCTS, INC *	Salop list	01280014	37899 NILES BOULEVARD, FREMONT			
BLACK POINT ANTENNA FIELD	DTSC CalSites	21970013	STONETREE LANE, NOVATO		Transformers	
CHURCH AND FRUIT JUNKYARD	DTSC CalSites	10490090	CHURCH & FRUIT AVENUES, FRESNO	1.09	Abandoned auto salvage yard	
DELTA STAR	SLIC	41S0081	270 INDUSTRIAL RD, SAN CARLOS		Transformer manufacture	
EASTERN ELECTRIC APP REPAIR COMPANY	SLIC	43S0470	1138 N 5TH ST, SAN JOSE			2300
FASS METALS	DTSC CalSites	07330030	818 W. GERTRUDE AVENUE, RICHMOND	0.81	Transformers dismantled onsite, and oil was spilled	
FLEET AND INDUSTRIAL SUPPLY CENTER	DTSC CalSites	01970007	2155 MARINER SQUARE LOOP, ALAMEDA	59.51	Previously an airport, warehouse facility and scrap yard lot	

Table 2-7 continued. PCB contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (ug/kg)(ppb)
FLEET INDUSTRIAL SUPPLY CENTER, OAKLAND	Salop list	01420124	EASTERN SHORE OF THE SF BAY, OAKLAND			
GENERAL ELECTRIC - OAKLAND	DTSC CalSites	01360059	5441 EAST 14TH STREET, OAKLAND		Used as dielectric fluids in finished products	
GENERAL ELECTRIC CO VALLECITOS NUCLEAR CENTER	TRI (CAS # 001336363)		6705 VALLECITOS ROAD, PLEASANTON			
GENERAL ELECTRIC NUCLEAR ENERGY	TRI (CAS # 001336363)		175 CURTNER AVENUE MC 402, SAN JOSE			
H K PORTER CO INC	DTSC CalSites	41360068	1777 INDUSTRIAL WAY, SAN CARLOS	2.27	Manufacture, test and repair electrical transformers and substations	440,000
HABITAT FOR HUMANITY PROJECT *	Salop list	01750036	10900 EDES AVENUE, OAKLAND			
HAMILTON ARMY AIRFIELD - BRAC	DTSC CalSites	21970008	HIGHWAY 101; 3 MI N OF LUCAS VALLEY ROAD, NOVATO		Aircraft maintenance	
HAYWARD AIR	DTSC CalSites	01970009	1525 WEST WINSTON AVE, HAYWARD		Maintenance of aircraft, vehicles and aerospace ground equipment	
NATIONAL GUARD	DTSC CalSites	43320001	2885 LAFAYETTE ST, SANTA CLARA	10.53	Storage of transformers and conductors	
HITACHI DATA SYSTEMS	Salop list	01290019	8130 ENTERPRISE DRIVE, NEWARK		Oil spill in 1979	
HOLLAND OIL * HOMART DEVELOPMENT	DTSC CalSites	41330052	480 INDUSTRIAL WAY, SAN FRANCISCO			
HUNTERS POINT ANNEX	DTSC CalSites	38440001	HUNTERS PT, SAN FRANCISCO		Unlined oil storage ponds, illegal disposal of hazardous materials, transformer and capacitor storage	
HUNTERS POINT NAVAL SHIPYARD	EPA Superfund		HUNTERS POINT NAVAL SHIPYARD, San Francisco	199.60	PCB-bearing transformers, drums, and polluted soil	
IKEA (FORMER BARBARY COAST STEEL)	DTSC CalSites	01440005	4300 EASTSHORE HIGHWAY, EMERYVILLE	9.51	Storing and melting scrap iron	
K & D SALVAGE	DTSC CalSites	15500001	600 SOUTH UNION AVENUE, BAKERSFEILD		Waste from old transformers and old cars	
KAISER ALUMINUM CHEMICAL	SLIC	01S0027	SUNOL BLVD, ALAMEDA			
KAISER ALUMINUM AND CHEMICAL CORPORATION	TRI (CAS # 001336363)		6177 SUNOL BOULEVARD, PLEASANTON			

Table 2-7 continued. PCB contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (ug/kg)(ppb)
LAWRENCE LIVERMORE NATIONAL LABORATORY	DTSC CalSites	01730095	7000 EAST AVENUE, LIVERMORE		Electrical capacitors, military and non-military research	
Lehigh Southwest Cement Company	Other		30101 Industrial Parkway SW, Union City			
LENNAR MARE ISLAND IA3	DTSC CalSites	48330002	900 WALNUT AVENUE, QUARTERS, VALLEJO		Transformer installations	
LIQUID GOLD OIL CORP LORENTZ BARREL & DRUM CO.	DTSC CalSites	07290039	HOFFMAN BLVD & S 47TH ST, RICHMOND	8.91	Leaks from hazardous material storage tanks and drums	
LUBRICATION COMPANY OF AMERICA (LCA)	EPA Superfund		1515 S 10TH ST, San Jose, Santa Clara	2.02	Drums and polluted soils	230 - 380,000
MAJOR SALVAGE *	DTSC CalSites	19290153	12500 LANG STATION ROAD 1770 NEPTUNE DR, SAN		Primarily oil processing and recycling plant	
MOFFETT FIELD	Salop list	01330034	LEANDRO		Drum leaks	
MYERS DRUM - EMERYVILLE	EPA Superfund		MOFFETT FIELD, Santa Clara, Santa Clara	890.69	Various closed and open uses	92 - 12,000 (landfill soils)
MYERS DRUM - OAKLAND *	Salop list	01340110	4500 SHELLMOUND ST, EMERYVILLE		Drum recycling facility	100,000
NAROM DEVELOPMENT *	Salop list	01340111	6549 SAN PABLO AVENUE, OAKLAND		Drum recycling facility	
NORTH STATE ENVIRONMENTAL	Salop list	01S0301	85 WINTON AVE WEST, HAYWARD		"Chlorinated solvents from off site"	
OAKLAND GATEWAY DEVELOPMENT AREA	PADS		90 S. SPRUCE AVE, SAN FRANCISCO		Transporter	
OAKLAND NAVAL HOSPITAL *	DTSC CalSites	01970016	700 MURMANSK STREET, SUITE 3, OAKLAND	147.17	Former Army base	
PACIFIC BELL PARKS RESERVE	Salop list	01970003	8750 MOUNTAIN BOULEVARD, OAKLAND			
FORCES TRAINING AREA	SLIC	43S0476	1051 RICHARD AVENUE, SANTA CLARA		Electrical equipment storage	70
PARKSIDE COMMONS APARTMENTS *	DTSC CalSites	01970012	BLDG. 790, 5TH STREET, DUBLIN			
PG&E SPILL	Salop list	01S0454	900 143RD AVE, SAN LEANDRO		"Organic compounds, pesticides and metals"	
PG&E SPILL	BaySpillReports		146 2ND AVE, DALY CITY		Pole-mounted transformer struck by lightning	
PG&E SPILL	R2 PCB Spills		570 LAKEVIEW RD, REDWOOD CITY		Lightning damage to electrical transformer	

Table 2-7 continued. PCB contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (ug/kg)(ppb)
PG&E SPILL	PG&E		329 W. 26th ST, SAN MATEO		1 gallon 91 ppm PCB oil released when lid blew off of transformer in 1995.	
PG&E SPILL	PG&E		1110 LORYN LN, HALF MOON BAY		1 quart 394 ppm PCB oil leaked, 1995.	
PG&E SPILL	PG&E		323 W. HILLSDALE AVE, SAN MATEO		1 gallon 9 ppm PCB transformer oil released in 1995. Leak spread to house, fence and deck.	
PG&E SPILL	PG&E		2013 EASTON, BURLINGAME		10 gallons 77 ppm PCB transformer oil released due to equipment failure caused by tree branches in 1995.	
PG&E SPILL	PG&E		1125 CHERRY AVE, SAN MATEO		150 gallons of 13 ppm PCB oil from underground vault in 1996.	
PG&E SPILL	PG&E		1020 AND 1024 SPRINGFIELD, 1019 AND 1023 SYLVAN DR, SAN CARLOS		4 gallons 900 ppm PCB oil over rear of 4 houses and yards in 1996.	
PG&E SPILL	PG&E		333 NORTH AMPHLETT, 312 NORTH IDAHO, SAN MATEO		1 pint of 82 ppm PCB oil to 2 houses, 1996.	
PG&E SPILL	PG&E		978 LAKEVIEW WAY, REDWOOD CITY		2 gallons of 410 ppm PCB oil to street, landscaping and soil and drainage ditch in 1996.	
PG&E SPILL	PG&E		25TH AND DELAWARE STREETS, SAN MATEO		1 pint of 53 ppm PCB oil, 1996.	
PG&E SPILL	PG&E		BEATTY RD, 100 YDS E OF TUNNEL RD, BRISBANE		225 gallons of 196 ppm PCB oil from underground transformer in 1997.	
PG&E SPILL	PG&E		1125 CEDARWOOD, REDWOOD CITY		2 gallons of 82 ppm PCB oil from overhead transformer to fence and landscaping in 1997.	
PG&E SPILL	PG&E		90TH AND JUNIPERO SERRA BLVD, DALY CITY		2 quarts of 210 ppm PCB oil released from overhead to area including wall of large building in 1997.	
PG&E SPILL	PG&E		4 ELDER CT, MENLO PARK		3 gallons of 242 ppm PCB oil to soil/ landscaping in 1998.	
PG&E SPILL	PG&E		825 OAK GROVE RD, MENLO PARK		1 gallon of 54 ppm PCB oil to landscaping in 1998.	
PG&E SPILL (A-1 TRUCK AND EQUIPMENT RENTAL)	PG&E		1125 ARGUELLO ST, REDWOOD CITY		3 gallons of 623 ppm PCB oil to commercial parking lot in 1997.	
PG&E SPILL (CALIFORNIA BANK AND TRUST)	PG&E		300 BROADWAY, MILLBRAE		1 pint of 177 ppm PCB oil, 1996.	
PG&E SPILL (HUMAN SERVICES AGENCY)	PG&E		1487 HUNTINGTON DR, S SAN FRANCISCO		1 gallon 932 ppm PCB transformer oil released over long period of time to a splice box in 1995.	

Table 2-7 continued. PCB contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (ug/kg)(ppb)
PG&E SPILL (TRAVELODGE MILLBRAE)	PG&E		110 SOUTH EL CAMINO REAL, MILLBRAE		2 gallons of 16 ppm PCB oil to asphalt and possibly one vehicle in 1998.	
PG&E SPILL (WESTERN GRINDING SERVICES) PITTSBURGH-DES	PG&E		601 HARBOR, BELMONT 3500 BASSETT ST, SANTA CLARA	12.55	2 gallons 17ppm PCB oil released from a pole bolted transformer that caught fire and burned in 1995.	
MOINES STEEL PORT OF OAKLAND, BERTH 25 AND 26 *	DTSC CalSites	43340056			Leakage from underground gas tank	
PORT OF RICHMOND (SHIPYARD #3)	Salop list	01280092	2700 7TH STREET, OAKLAND 1312 CANAL BLVD,		Chemical blending, packaging and storage facility Waste piles, including abandoned drums, solvent containers, etc.	
PORT OF SAN FRANCISCO	DTSC CalSites	07370030	RICHMOND	21.46		
RAYCHEM/TYCO ELECTRONICS	DTSC CalSites	38440006	PIER 70, SAN FRANCISCO 308 CONSTITUTION DR, MENLO PARK		Leaking transformers	
RHONE POULENC/(ZOECON)	PADS				Generator	
SANDOZ ROMIC	EPA Superfund		1990 Bay Road, East Palo Alto, California	5.34	Pesticide manufacture, leaking underground storage tanks	
ENVIRONMENTAL TECHNOLOGY CORPORATION	PADS		2081 BAY ROAD, EAST PALO ALTO		Transporter	
SANTA CLARA COUNTY JAIL	DTSC CalSites	43920002	180 WEST HEDDING ST, SAN JOSE		Leaking transformers	
SF DYKE RECONSTRUCTION CITY	SLIC	41S0139	SF AIRPORT, SAN FRANCISCO TREASURE ISLAND, SAN		"Motor oil issues"	
SF NAVY TECHNICAL TRAINING CENTER	DTSC CalSites	38370044	FRANCISCO	4.05	Industrial landfill and bayfill areas	
SITE K (SEAWALL LOT 333)	DTSC CalSites	38750002	THE PRESIDIO, SAN FRANCISCO		Removal and disposal of transformers and oil switches	
SOUTHERN PACIFIC RIGHT-OF-WAY			WEST OF 4525 HOLLIS STREET, EMERYVILLE		Adjacent PG&E Materials Distribution Center	50,000 -
EMERYVILLE	DTSC CalSites	01400002			Previously waste oil recycler	1,400,000
SOUTHLAND OIL SPACE SYSTEMS / LORAL - B12	DTSC CalSites TRI (CAS # 007439976)	19290003	5619-5621 RANDOLPH STREET 1034/1036 E MEADOW CIRCLE B12, PALO ALTO	0.61		1,100,000

Table 2-7 continued. PCB contaminated areas in the Bay Area.

Name of Hot spot	Database	DB Ref. #	Location	Area (ha)	Main sources	Soils Before remediation (ug/kg)(ppb)
STANFORD LINEAR ACCELERATORY CENTER	PADS		2575 SAND HILL ROAD, MENLO PARK		Generator, and a small (1kg) transformer	
TRAVIS AFB	EPA Superfund		TRAVIS AFB, Solano	2034.41	Landfills, spills, and combustion of wastes	
TRIPLE A MACHINE SHOP	DTSC CalSites	38440002	HUNTERS POINT, SAN FRANCISCO		Illegal disposal of hazardous materials	
U S PIPE AND FOUNDRY COMPANY	TRI (CAS # 001336363)		1295 WHIPPLE ROAD, UNION CITY			
UNION PACIFIC OAKLAND COLISEUM SITE	DTSC CalSites	01400015	700 73RD AVENUE, OAKLAND		Previously operated as auto salvage yard	
US STEEL POSCO INDUSTRIES	TRI (CAS # 001336363)		900 LOVERIDGE ROAD, PITTSBURG			
USS-POSCO INDUSTRIES	Other		900 LOVERIDGE ROAD, PITTSBURG, CA	21.46	oil disposal	1,200,000
USX Corporation//Bay West Cove/Wetland Creations	Water Board		Oyster Point Blvd., San Francisco			
VALERO BENICIA	TRI (CAS # 007439976)		3001 PARK ROAD, BENICIA			
ASPHALT PLANT						
WESTINGHOUSE ELECTRIC (SUNNYVALE PLANT)	DTSC CalSites	43350001	401 EAST HENDY AVE, SUNNYVALE			
WESTINGHOUSE ELECTRIC CORP	EPA Superfund		SUNNYVALE, Santa Clara	30.36	Transformer manufacture, onsite use of Interteen (contains PCBs) as a weed killer, pollution along railroad spurs	10,700 - 20,000,000
WESTINGHOUSE ELECTRIC CORP	SLIC	01S0021	5899 PELADEAU, EMERYVILLE		Spillage of transformer oils	

For PCBs it was found that contaminated areas exhibit a halo effect of up to 600 m and more typically <300 m. So we will assume that every PCB contaminated area in the Bay Area has dimensions of 100 m in the cross wind direction and 300 m in the downwind direction (Best area = 0.03 km^2 ; Range = $0.013\text{-}0.12 \text{ km}^2$). Adding this to the above estimates of areas of contaminated areas gives a best estimate of area of 0.14 km^2 and a range of $0.036\text{-}0.72 \text{ km}^2$. There are a total of 89 identified PCB contaminated areas in the Bay Area. We only know the area of 20 which together add to an area of 35.5 km^2 . For the other 69 we will use the estimates of area above for a single contaminated area and halo making a best estimate of total area of 45 km^2 of polluted watershed surface (Range = $38\text{-}85 \text{ km}^2$).

The range of soil Hg concentrations found in industrial areas of cities (again see next section of this report “Hg and PCB Pollution in the Urban Environment”) is $0.35\text{-}230 \text{ mg/kg}$ (median = 0.86 mg/kg ; 25th percentile = 0.5 mg/kg ; 75th percentile = 15 mg/kg). The range of soil PCB concentrations found in industrial areas of cities (again see next section of this report “Hg and PCB Pollution in the Urban Environment”) is $0.18\text{-}510,000 \text{ mg/kg}$ (median = 11 mg/kg ; 25th percentile = 4.6 mg/kg ; 75th percentile = 590 mg/kg). In our calculations we will use the 25th percentile, the median, and the maximum concentrations found in the local study in the Ettie Street pump station watershed (31 mg/kg) (Kleinfelder, 2005).

Davis et al. (2000) used the simple model to estimate a total Bay Area suspended sediment load of $310,000 \text{ t/y}$ of which 9% was from industrial areas. Based on this, industrial areas on average were estimated to produce $75 \text{ t/km}^2/\text{y}$ or $3,000 \text{ t}$. Based on a quick review of urban literature, a typical soil loss associated with urban land use of $140\text{-}320 \text{ kg/ha/y}$ (see references in Pearce et al., 2005). For our estimates, we will use the lower bound of the literature and the estimate by Davis et al. (2000) as the upper bound. Using these estimates and the estimates of areas associated with Hg and PCB contaminated areas, it is estimated that between $250\text{-}7,400 \text{ t}$ of suspended sediment enter stormwater from Hg contaminated areas and between $530\text{-}6,400 \text{ t}$ of suspended sediment enter stormwater from PCB contaminated areas.

Combining these sediment load estimates with the Hg and PCB concentrations described above give a first order estimate of Hg and PCB loads (Table 2-8). There are a number of possible sources of error that we have tried to capture in these estimates. However, the major problem with our estimate is the determination of the actual number of contaminated areas. It is likely that the number might be 10 x greater or even 100 x greater but our estimation methods is fraught by the great potential for a low bias of 10s to 100s of times. In contrast, the estimates of sediment export and soil concentrations if estimated too high might only decrease by a factor of 2-5x.

Table 2-8. Estimates of Hg and PCB loads associated with contaminated areas in the Bay Area.

	Area (km ²)			SS (t/km ²)			SS (t)			Concentration (mg/kg)			Load (kg)		
	Low	High	Best	Low	High	Best	Low	High	Best	Low	High	Best	Low	High	Best
Hg	18	99	31	14	75	45	252	7,425	1,395	0.5	15	0.86	0.25	7.4	1.4
PCB	38	85	45	14	75	45	532	6,375	2,025	4.6	31	11	0.53	6.4	2.0

2.3.8.3. Other Large Industrial Uses

There are / were a number of industrial users in the Bay Area. In some cases PCBs are still in use (Table 2-9) (total mass = 196,836 kg). The largest current user (72%) is USS-POSCO, a steel manufacturer. We will assume leakage is similar to that from large transformers and capacitors. In a U.S. study, about 0.05% was estimated to leak from transformers each year (see references in Harrad, 1994 and page B-2 of EIP and Associates, 1997). This amounts to 98 kg that could reach the atmosphere each year. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for lower rainfall urban and industrial areas, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). Thus, our estimate for PCB entry into stormwater conveyances is 2-6 kg/y (average = 4 kg/y).

Table 2-9. Reported PCBs still in use in the Bay Area. (Table 4: Hetzel, 2004)

Company	City	Number of transformers	PCB mass (kg)
USS-POSCO Industries	Pittsburg	65	141,494
Quebecor Printing San Jose Inc.	San Jose	5	32,094
NASA	Moffett Field	17	7,052
Gaylord Container Corp.	Antioch	2	6,078
General Chemical	Pittsburg	3	4,800
Rhodia Inc.	Martinez	4	3,356
DOT Maritime Administration Suisun Bay Reserve Fleet	Benicia	3	1,048
Macaulay Foundry Inc.	Berkeley	1	913
Stanford Linear Accelerator	Menlo Park	1	1

2.3.8.4. Railway Lines

Mercury is likely associated with railway lines due to the use of coal then diesel for fueling locomotive engines and due also to oil and grease leakage from fuel tanks, bearings and other mechanical devices. PCBs were used historically for switching equipment and other electrical uses along rail lines as well as for dust suppression. In

addition there was probably a level of spillage of substances containing Hg and PCBs during the haulage of goods (Malawska and Wilkomirski, 2001). A quick literature review revealed concentrations of Hg associated with soils on a railway siding in Poland of 0.262 mg/kg (Malawska and Wilkomirski, 2001). In a study of pedestrian dust, Chutke et al., 1995 measured Hg concentrations on several railway bridges and a railway station ranging between 0.16-1.02 mg/kg. To give some perspective, concentrations of Hg found in low density urban or sub-urban soils with no industrial influence range between 0.15-0.44 mg/kg and show a median of 0.16 mg/kg (See section 3 and Section 6, Table 6-1 of this report). PCB concentrations associated with railway stations and bridges of railway lines were studied as part of an urban characterization survey in German cities (Yang, 1996). Concentrations ranged from 0.2-2 mg/kg and averaged 0.8 mg/kg. Locally Kleinfelder (2005) measured a PCB concentration of 0.61 mg/kg in soils near the Southern Pacific Railroad in the Ettie Street Pump Station watershed. Again to provide some perspective, we have no data on urban areas with no discernable industrial influence but agricultural areas with no industrial influence have concentrations ranging between 0.001-0.13 mg/kg (See section 3 and Section 6, Table 6-1 of this report).

Railway has been an important part of the transportation network of the Bay Area for over 100 years and is still a prominent feature in the modern landscape (Figure 2-10 left panel). Historically, there were many “spur” lines feeding off the main trunk lines (Figure 2-10 right panel). The area associated with railway lines in the Bay Area was estimated based on the length determined using GIS and a average width estimate of 20m based on a sample of railway widths measured on aerial photos of the east Bay. The total area of main trunk lines estimated in this manner was 34.5 km². The area associated with spur lines in the east Bay (Figure 2-10b) was estimated to be 2.9 km². Adding these two estimates and considering other industrial areas on the Bay margin, the total area associated with historical railway for the period 1950-90 is approximately 40 km² or 0.6% of the Bay Area watershed and 11% of the industrial land use.

Estimation of Hg and PCB loads associated with railways lines is difficult but is facilitated by estimates of sediment loads. Davis et al. (2000) used the simple model to estimate a total Bay Area suspended sediment load of 310,000 t/y of which 9% was from industrial areas. Based on this, industrial areas on average were estimated to produce 75 t/km²/y or 3,000 t from railway areas. Based on a quick review of urban literature, a typical soil loss associated with urban land use of 140-320 kg/ha/y (see references in Pearce et al., 2005). For our estimates, we will use the lower bound of the literature and the estimate by Davis et al. (2000) as the upper bound. Using this range, it is estimated that sediment load associated with railway lines will be 560-3,000 t/y (best estimate = 1,800 t/y). Combining these estimates with concentrations of Hg and PCBs associated with railway lines (Yang, 1996; Kaminski and Landsberger, 2000; Malawska and Wilkomirski, 2001; Kleinfelder, 2005), provided a first order estimate of 0.09-3.0 kg/y Hg (best = 1.5 kg/y Hg) and 0.1-6 kg PCBs (best = 1.1 kg/y PCBs). These estimates might be biased to high loads given that the concentration estimates from the literature tend to be for railway loading and switching areas.

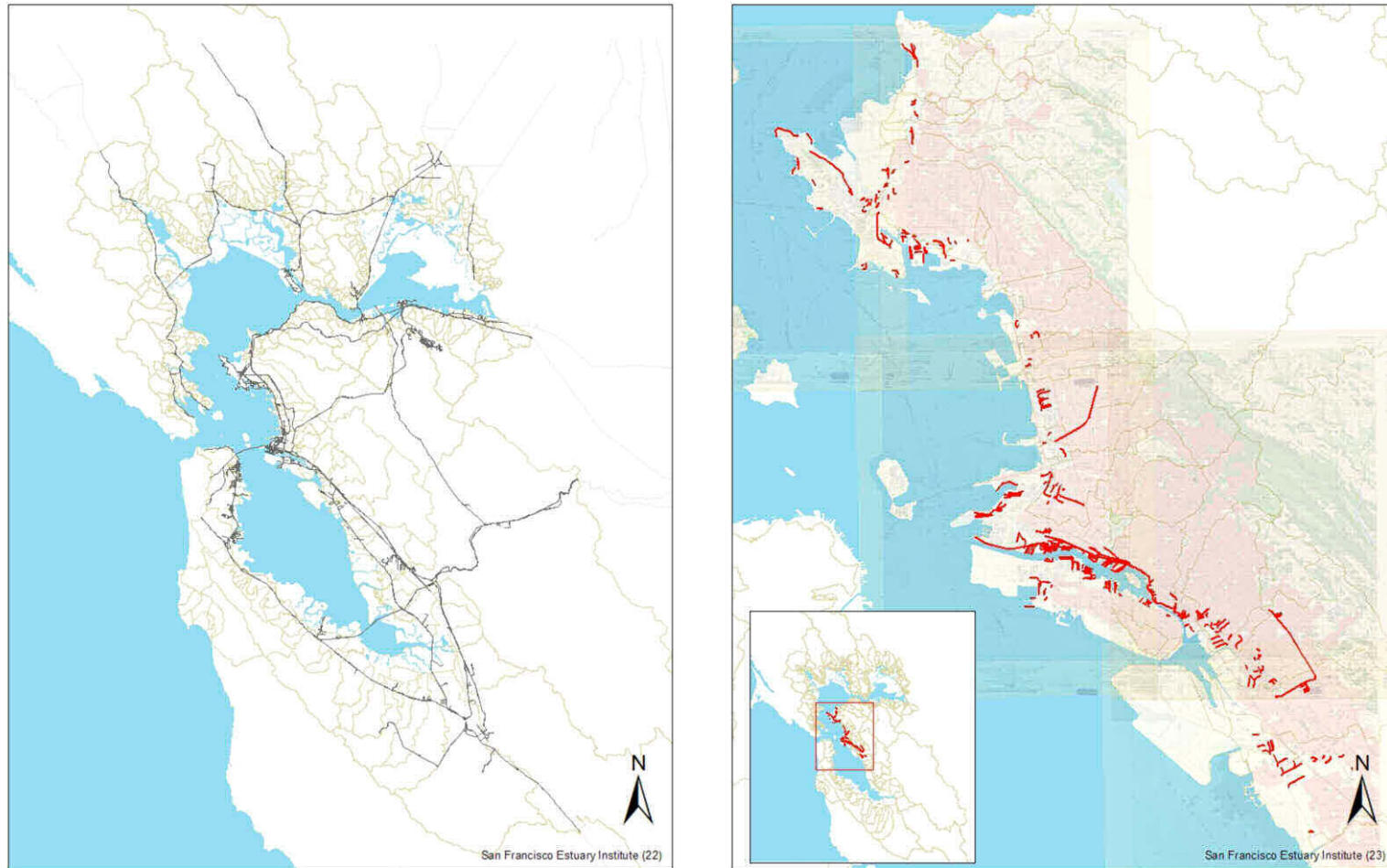


Figure 2-10. Locations of railway infrastructure in the Bay Area. Left panel - Main trunk lines derived from the USGS maps of the 1980s era. Each rail line was assumed to be buffered 10m on each side of the track (a total of ~20m) and dissolved in the GIS to eliminate double counting. The width of 20m was chosen after measuring widths on aerial photography; Right panel - 1950s railroad “spur” lines for the Richmond, Berkeley, Oakland and San Leandro areas of the East Bay. The 1959's railroad lines were digitized from 1959 topographic maps (7.5 minute series) of San Quentin, Richmond, Oakland-East, Oakland-West, and San Leandro. The 1959 railroad area calculation also used 10m as the buffer distance, and each buffer was then dissolved In the GIS to eliminate double counting of the areas.

2.3.8.5. Wastewater Treatment and Landfill Disposal

Off gassing from wastewater treatment facilities and landfills are a point source of Hg and PCBs to local watershed surfaces. In the case of Hg, this may be a measurable amount but at present there is no estimates available for PCBs and in Brittan it has been found to be negligible (Harrad et al., 1994). The best current estimates for losses of Hg from waste disposal are provided by DTSC (2002). Their 2000 estimate was 280 lbs (130 kg) for California. Scaling this to the population of the Bay Area amounts to 25 kg. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for the Bay margin where landfills are located, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). Thus our best estimate for waste disposal Hg entering stormwater conveyances is 0.5-1.5 kg/y (Average = 1 kg/y).

2.3.9. Other activities or products where both Hg and PCBs are present

2.3.9.1. Auto-recycling

Cars and other vehicles are a cocktail of parts and components that incorporate Hg and PCBs to fulfill a variety of functions. To a large extent, these individual parts such as switches (Hg) and capacitors (PCBs) have been inventoried above. This section on auto-recycling is included as a sink (stored mass) and transfer estimate that is part of the overall budget being careful to avoid double counting.

Mercury is used in vehicles in several ways, including: hood and trunk convenience light switches, anti-lock braking systems, high intensity discharge lamps, and entertainment and navigational systems (Arbitman and Gerel, 2003). About 89% of the mercury found in 1999 or older vehicle is associated with switches (DTSC, 2004). Each end-of-life vehicle (ELV) contains between 0.5 and 1 g of Hg (DTSC, 2002). This mercury can be released into the environment (mainly atmosphere) during recycling or transported as a pollutant in “auto-shredder waste” to non-hazardous landfills where it is used for capping (DTSC, 2002). Senate Bill 633 (Sher, 2001) requires mercury-containing switches that are voluntarily removed from motor vehicles to be managed in accordance with DTSC’s universal waste rule. DTSC and local agencies are required by the Bill to provide coordinated technical assistance to businesses in the “safe removal and proper disposal of mercury-containing light switches from motor vehicles.” The Bill also prohibits the sale of vehicles manufactured after January 1, 2005 which contain mercury switches. The Bill also mandates DTSC to coordinate and encourage replacement and recycling of mercury-containing motor vehicle light switches, therefore this mercury source will likely diminish with time. There are a number of licensed auto-recyclers in the Bay Area (Figure 2-11). There are also an unknown number of unlicensed dismantlers and private rebuilders/resellers that handle approximately 66% of the total number of vehicles.

Because of this unknown, we estimated the numbers of vehicles dismantled or recycled in the Bay Area from California statistics. Approximately 700,000 autos are recycled in California each year (DTSC, 2002). The population of California in 2000 was 33.87 and of the Bay Area was 6.78 million. If we assume that auto wrecking is proportional to population there would be 140,000 autos recycled in the Bay Area each year. The vehicles would contain a approximately 70-140 kg (average = 100 kg) of mercury.

PCBs can be present in automobiles in hydraulic fluids, plastics, voltage regulators, electric motors, switches, small capacitors, and light ballasts. At present we have no data on the total average mass PCBs in motor vehicles so it is difficult to assess the potential for mass to enter stormwater. However, PCB concentrations in untreated auto-shredder waste are between 0.59 – 129 mg/kg and treated shredder waste contains 2.6 - 45.1 mg/kg (DTSC, 2002b). In California, there is about 270,000 t of shredder waste generated each year from both autos and appliances. If we scale this to the population of the Bay Area, this equates to 30-6,970 kg of PCBs (Average = 3,500 kg) in untreated shredded waste and 140-2,440 kg of PCBs (Average = 1,300 kg) of treated shredder waste.

The fate of auto-Hg is difficult to determine. In a study conducted in Michigan on removal of switches containing Hg from vehicles manufactured between 1971-2003, no evidence was found of leakage prior to dismantling, no Hg leaked during removal, it took an average of 95 seconds for removal of a switch assembly from a vehicle and the removal of the Hg pellet from a switch assembly, and 1% of the switch pellets leaked after transport to the hazardous waste facility (MDEQ, 2002). Presently in California, there is little active removal of devices containing Hg from vehicles prior to crushing (DTSC, 2002). DTSC's Auto Shredder Initiative sampling and laboratory analyses showed that, in 2001, approximately 840 kg of mercury was found auto shredder waste (resulting from shredding automobiles and appliances), and that 360 kg originated from automobiles. Scaling this to the census 2000 Bay Area population results in approximately 72 kg/y transported to non-hazardous waste landfills.

Using the same deposition and runoff reasoning described above we can make estimate of the fate of the remaining 0-68 kg/y (best estimate = 28 kg/y) that is still unaccounted for and probably enters the atmosphere during the shredding processes. We need to allow for local deposition of this Hg and not count Hg contributed to regional deposition from this loss during shredding. We do this by considering working days and the likelihood of rain occurring. There are between 60-70 rain days in the Bay Area on average (McKee et al., 2003) or about 16-19% of the days in a year. If we consider deposition only occurs on those days with substantial rain (>12.7 mm or 0.5 in) for near-Bay industrial areas where auto-recyclers and shredders reside, this number is reduced to 9-23 days or 2-6% of the year (see paint section for rationale). If we also assume that releases do not occur on wet season public holidays or weekends, this number is reduced to 1.5-4%. Thus, approximately 0-3 kg (best estimate = 0.8 kg) of Hg has the potential to enter stormwater conveyances. Taking into account the runoff coefficient for industrial areas (60-95%) this equates to 0-3 kg/y (best estimate 0.7 kg/y).

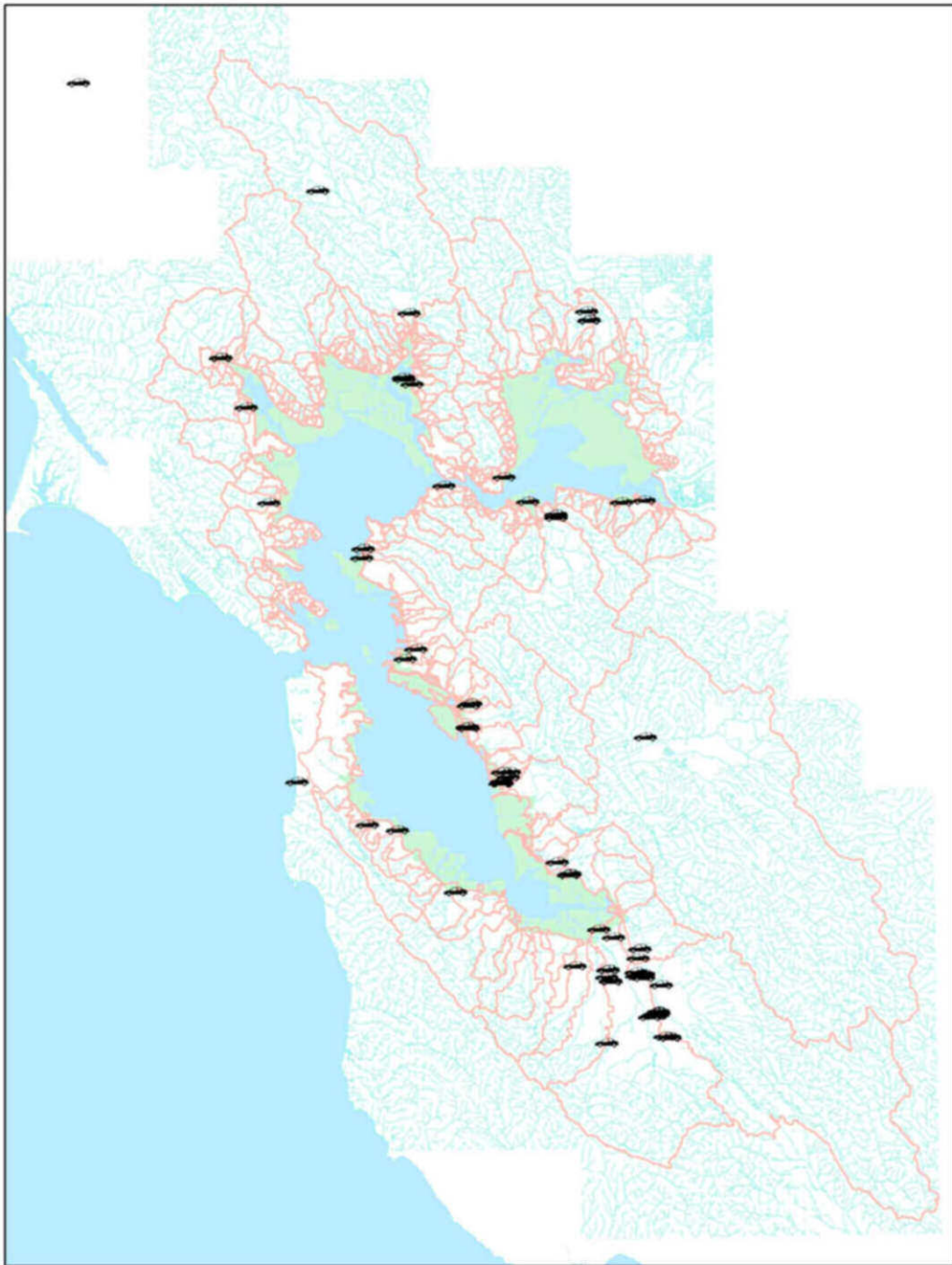


Figure 2-11. Auto-recyclers in the Bay Area (Source: Fred Hetzel, 2006).

An alternative approach to determining pollutant loads associated with auto-dismantling is to use estimates of sediment loads and combine these estimates with typical sediment concentrations of Hg and PCBs found in or adjacent to wrecking yards. There are a total of 58 licensed auto-wrecking yards in the Bay Area (Figure 2-11). We took a sample of 30 of these and measured areas from modern aerial photographs. These ranged in size from 0.4-28 ha with a median area of 2.9 ha (0.029 km²). Using the median and inter-quartile range it is estimated that the total area of licensed wrecking facilities in the Bay Area is 80-270 ha (Best estimate = 170 ha (1.7 km²)). Davis et al. (2000) used the simple model to estimate a total Bay Area suspended sediment load of 310,000 t/y of which 9% was from industrial areas. Based on this, industrial areas on average were estimated to produce 75 t/km²/y or 60-200 t/y from auto-dismantling yards. Based on a quick review of urban literature, typical soil loss associated with urban land use is 140-320 kg/ha/y (see references in Pearce et al., 2005). For our estimates, we will use the lower bound of the literature and the estimate by Davis et al. (2000) as the upper bound. Using this range, it is estimated that sediment load associated with auto-dismantling yards will be 11-200 t/y (best estimate = 77 t/y). We presently know of no measurements of Hg in soils associated with Auto-dismantlers but we could use the lower and upper bounds for industrial urban land use (0.35-230 mg/kg) (mean = 21 mg/kg) (see Chapter 3 or the summary in Chapter 6). Combining these with the estimates of soil loss we get 0.004-46 kg/y (best estimate = 1.6 kg/y). For Hg we have two estimates.

PCB concentration in sediment associated with auto-dismantlers has recently been measured in the Ettie Street Pump Station watershed (Kleinfelder, 2005). Concentrations were measured at Cole Brother Auto Wreckers (3.8 mg/kg) and Cypress Auto Salvage (0.08 mg/kg). Combining sediment load estimates with concentrations of PCBs associated with auto-dismantling (Kleinfelder, 2005), provided a first order estimate of 0.0009-0.76 kg PCBs (best = 0.4 kg/y PCBs).

2.3.9.2. E-waste

Electronic waste is a specific type of often hazardous waste specifically associated with computer equipment, TVs and other electronics. In a similar vein to auto-recycling, E-waste is by and large included in the inventories of other categories above; the information here provides an understanding of the sink (stored mass) and transfer associated with a specific type of urban waste. E-Waste contains Hg in batteries, switches, printed circuit boards, and flat panel screens and PCBs in capacitors, transformers, and plastic casings (fire retardants). Consumers have on average 2-3 out-of-use computers stored in their houses. It is estimated that 75% of all computers ever sold in the US remain stockpiled in private homes, and business storerooms. Of the 25% disposed of, only about 10% of computers are recycled, the rest being illegally disposed in landfills or dumped. E-waste also includes printers, drives, TVs, stereo equipment, LCD games, watches, cell phones, and a huge variety of other small electronic gadgets. It is difficult to estimate the amount of E-waste stored, so U.S estimates have a large uncertainty and range between 315-680 million computers and maybe double that number of TVs; scaled to the Bay Area this would be 8-16 million computers alone. On

average, a computer contains 0.6-0.7 g Hg, so the estimate for Hg stored in computer waste alone is 3.8-9.5 t. We are not aware of any estimates of mass of PCBs associated with E-waste except in electronic shredder residue in Japan, where it has been measured to be 1,200 ng/g (Sakai et al., 1998). Presently we have no estimate for PCB mass associated with electronic waste alone but we believe it could be a large PCB sink.

2.3.10. Estimated Hg and PCB Mass Supply Associated With Legacy Usage (1950-90)

2.3.10.1. Bed and Bank Erosion

It is well known that Bay Area creeks, rivers and channels with no engineered bed elevation controls are incising in response to changes to sediment and water supply associated with climatic and anthropogenic factors (see references in McKee et al., 2003). In the review by McKee et al. (2003), it was recognized that the degree of incision varied substantially from one watershed to another and ranged from 8-60% (Best = 20%) of the total sediment supply to channels. The total sediment supply to the Bay from local tributaries was estimated to be 560,000-1,000,000 t/y (Best = 780,000 t/y) (McKee et al., 2003). Therefore, our first order estimate for sediment supply from bed incision and bank erosion is 45,000-600,000 t/y (Best = 150,000 t/y) for the Bay Area.

Concentration of Hg and PCBs in bed sediments have been measured by BASMAA agencies (Gunther et al., 2001; KLI, 2001; KLI, 2002; Salop et al., 2002). The quartile ranges in concentrations for mixed land use range between 0.09-0.26 mg/kg Hg (median = 0.14 mg/kg Hg) and 0.0047-0.071 mg/kg PCBs (median = 0.019 mg/kg PCBs). One might argue that the concentrations in the bed below the surface are likely to be lower, however, the concentrations measured by these authors are, in fact, the result of the mixing of concentrations from multiple sources including bed and bank sources, and soils supplied from upland erosion sources. It is therefore not a true source, but an accounting of the PCB and Hg mass associated with a particular process – that of channel evolution. Therefore it is valid to combine these concentration estimates with the estimates of bed and bank erosion to derive an estimate of between 4.1-160 kg Hg (Best = 21 kg Hg) and 0.2-43 kg PCBs (Best = 2.9 kg PCBs) supplied to stormwater conveyances processes annually.

2.3.10.2. Watershed Surface Sediment Erosion

Some of the sediment in stormwater conveyances is derived from upland sediment erosion and erosion from urban areas (e.g. construction sites, vacant lots, unpaved foot paths and ride sides, and wear debris from road and building surfaces). Pollutant mass associated with these erosion sources is not accounted for in the previous sections; these only accounted loading and loss processes occurring “today”. Here we calculate the pollutant mass associated with 50+ years of legacy accumulation on soils surfaces and the

erosion of these soils, or in other words, erosion of soils with elevated background concentrations. In the review by McKee et al. (2003), the sediment supply in Bay Area watersheds from land slide surface erosion was found to range from 38-64% (Best = 50%) of the total sediment supply to channels. The total sediment supply to the Bay from local tributaries was estimated to be 560,000-1,000,000 t/y (Best = 780,000 t/y) (McKee et al., 2003). Therefore, our first order estimate for sediment supply from landslide erosion in uplands is 200,000-640,000 t/y (Best = 390,000 t/y) for the Bay Area. We cannot directly apply the urban sediment loss estimates from the review of urban literature (140-320 kg/ha/y) (see references in Pearce et al., 2005) because some of the bed and bank erosion and land slide sediment supply is associated with urban land use. Our estimate for urban surface sediment supply therefore is provided by the difference between total estimated sediment load to the Bay (780,000 t) and the sum of sediment supply from bed and bank erosion and landslide erosion. Thus our first order estimate is 240,000 t. To estimate the range of error around these first order estimates, we apply the same range adopted in the review by Pearce et al. (2005) ($\pm 40\%$).

Soil concentrations for Hg and PCBs in various types of land uses were reviewed by gathering data from the world literature (see later section of this report). Because we are interested in the general diffuse load of Hg and PCB associated with surface sediment erosion not that from hot spots, we will use the median and inter-quartile ranges of concentrations found in soils associated with urban landscapes from other parts of the world (ideally we would use local data but none exists). Median Hg concentration in open and agricultural areas is 0.053 mg/kg (inter-quartile range is 0.048-0.090 mg/kg). Median Hg concentration in urban areas with little or no industrial influence is 0.16 mg/kg (inter-quartile range is 0.15-0.37 mg/kg). In the case of PCBs, data from our review of world literature suggests the median concentration in open and agricultural areas is 0.020 mg/kg (inter-quartile range is 0.012-0.031 mg/kg). Median PCB concentration in urban areas with little or no industrial influence is 0.092 mg/kg (inter-quartile range is 0.010-0.16 mg/kg). The calculations for Hg and PCB mass entering stormwater conveyances associated with surface erosion is summarized (Table 2-10).

2.3.11. Estimated Mass Removal Associated With Street Sweeping and Inlet Maintenance

The previous sections have developed estimates of mass of Hg and PCBs associated with various inputs to the Bay Area watersheds and made estimates of the portion of mass that might become available for hydrological transport to stormwater conveyances. To determine what might actually enter the stormwater conveyances, the mass removed by stormwater agency maintenance efforts must be deducted.

Salop and Akashah (2004) calculated mass removed via street sweeping in Alameda County for Hg of 0.6-1.9 kg (Best = 1.2 kg Hg) and for PCBs of 0.3-2 kg (Best = 0.6 kg). These estimates are uncertain because they are based on sediment concentrations from inlets, catch basins and pump stations and not based on sweeping material. If we assume a similar rate of removal for the other Counties and scale activities according to

Table 2-10. First order estimates of Hg and PCB mass entering stormwater conveyances associated with surface erosion.

	Suspended sediment			Hg						PCBs					
	Low (t)	High (t)	Best (t)	Low (mg/kg)	High (mg/kg)	Best (mg/kg)	Low (kg)	High (kg)	Best (kg)	Low (mg/kg)	High (mg/kg)	Best (mg/kg)	Low (kg)	High (kg)	Best (kg)
Landslides	200,000	640,000	390,000	0.048	0.09	0.053	10	58	21	0.012	0.031	0.020	2	20	8
All urban areas	144,000	336,000	240,000	0.15	0.37	0.16	22	124	38	0.010	0.158	0.092	1	53	22
						<u>Total</u>	<u>31</u>	<u>182</u>	<u>59</u>			<u>Total</u>	<u>4</u>	<u>73</u>	<u>30</u>

population (Excluding San Francisco given the combined sewer system there), the estimate for street sweeping is 5 kg Hg (Range = 2.5-8 kg Hg). The estimate for PCBs is 2.5 kg (Range = 1.3-8.4 kg PCBs).

Alameda County has made estimates of mass removed via storm drain facility cleaning (inlets, catch basins, and pump stations) using concentrations observed in these types of facilities for Hg of 0.5-1.5 kg (Best = 1 kg) and for PCBs of 1.1-7.6 kg (Best = 2.6 kg) (Salop and Akashah, 2004). These authors as made estimates using concentrations observed in a larger data set that included creek concentration data but we suggest that these estimate are less valid because of the added uncertainty associated with differing hydraulics in creeks versus inlets, catch basins, and pump stations. Using the same first order method, our first order estimate for the Bay Area for storm drain facility cleaning is 4.2 kg Hg (Range = 2.1-6.3 kg) and 11 kg PCBs (Range 4.6-32 kg).

Salop and Akashah (2004) also made estimates for Alameda County for channel de-silting. Their estimates ranged from 9.2-39.7 (Best = 20.5) for Hg and 0.8-6 kg (Best = 2.3 kg) for PCBs. Population is not a suitable surrogate measure for estimating channel de-silting needs for a particular county – channel de-silting is likely more related to channel design and sediment supply. Santa Clara Valley Water District has kept records for 27 years on channel de-silting operations and these have been recently compiled by colleagues at SFEI (Grossinger and Pearce, 2005 unpublished data). The total average mass removed by SCVWD each year was 67,000 yd³. Using the same mass density adopted by Salop and Akashah (2004) (2,835 lbs/yd³ equal to 1,690 kg/m³) we estimate the SCVWD removed an average of 86,000 t/y over the past 27 years. If we use the same channel sediment concentrations adopted by Salop and Akashah (2004) for mixed land use (0.1-0.6 mg/kg Hg (Best = 0.3 mg/kg Hg) and 0.011-0.081 mg/kg PCBs (Best = 0.032 mg/kg PCBs), this amounts to a total mass removal of 9-52 kg/yr of Hg (Best = 26 kg Hg) and 1-7 kg/yr of PCBs (Best = 3 kg PCBs). Together, the watersheds of Alameda County and Santa Clara Valley Water District comprise 57% of the urbanized Bay margin (excluding San Francisco). If we assume that similar de-silting efforts are going on in other areas, our estimate (using a 1.8 multiplier) for channel de-silting for the Bay Area is 33-165 kg/yr of Hg (Best=84 kg Hg) and 3-23 kg/yr of PCBs (Best=10 kg PCBs).

2.4. SUMMARY

This report section has developed detailed information on the sources and loads of Hg and PCBs entering stormwater conveyances organized according to a conceptual model based on mass balance (or conservation of mass). It should be emphasized that the information represents an extrapolation of often disparate pieces of information that have been developed by a large number of scientists and engineers for different systems around the world with widely differing study objectives. It should be looked upon a framework for thinking and an inventory that needs to be criticized and further refined. It is also a starting point for management decisions on the potential for increased pollution prevention and source control, especially for Hg given its ongoing use in the urban environment and atmospheric deposition.

While we should keep in mind the lower and upper range of our estimates (Table 2-11; Figure 2-12), based on our best estimates, it appears that the largest sources of Hg to Bay Area stormwater conveyance channels are Watershed Surface Sediment Erosion > Atmospheric Deposition > Instruments > Bed and Bank Erosion > Switches and > Thermostats > Fluorescent Lighting > Paint > Railway Lines > Identified Industrial Contaminated areas (Table 2-11; Figure 2-13).

While we should keep in mind the lower and upper range of our estimates (Table 2-12; Figure 2-14), based on our best estimates, it appears that the largest sources of PCBs to Bay Area stormwater conveyance channels are Watershed Surface Sediment Erosion > Building Demolition and Remodeling > PCBs Still in Use > Bed and Bank Erosion > Transformers and Large Capacitors > Atmospheric Deposition > Identified Industrial Contaminated areas (Table 2-12; Figure 2-15).

The final check on the estimates of mass entering stormwater conveyances from watershed sources and uses is to compare these to estimates of removal and existing estimates of loads to the Bay. The Region 2 RWQCB estimated urban runoff loads entering the Bay for both Hg (Looker and Johnson, 2004) and for PCBs (Hetzel, 2004). For Hg the estimates for non-urban areas were 25 kg and urban areas were 160 kg. No estimates of range or error in these loads were given (Looker and Johnson, 2004). The PCB load estimate were developed by KLI (2002) who combined bed sediment concentrations collected by BASMAA agencies (Gunther et al., 2001; KLI, 2001; KLI, 2002; Salop et al., 2002) with sediment loads estimates provided by SFEI (Davis et al., 2000) to calculated mass loads. The best estimate for PCB mass load to the Bay from both urban and non-urban areas is currently 39 kg (Range = 9-100 kg). The PCB TMDL adopted 34 kg as the best estimate for urban runoff (Hetzel, 2004).

Assuming steady state (that is in channel pollutant storage is unchanging over a decade or more), the channel mass budget is constrained by the following equation:

$$\begin{aligned} &\text{Watershed Input} - \text{Street Sweeping Removal} - \text{Storm drain Facility Cleaning} - \text{Channel De-silting} \\ &= \\ &\text{Load to the Bay} \pm \text{Error} \end{aligned}$$

Using these load estimates to the Bay we can constrain a stormwater conveyance Hg and PCB budget (Table 2-13). Here we have just presented budgets based on the best estimates for Hg but it will be informative to develop these further to include ranges in the loads estimates. It is seen from the table that using our best estimates we have 77 kg (44%) of Hg not accounted for. This could be associated with any one of the terms or all of the terms, however, load from the watershed, channel de-silting and load to the Bay being the largest terms are most likely to contain the majority of the unaccounted mass. In contrast, the best estimate for PCBs agree surprisingly well (only 21%) helping to support the notion that estimates are better for PCBs than for Hg. Refining the estimates for Hg should be a primary goal for further work.

Table 2-11. Summary of mass input of Hg to Bay Area stormwater conveyances.

Source	Low		High		Best	
	(kg)	(%)	(kg)	(%)	(kg)	(%)
Watershed Surface Sediment Erosion	30	39.3	182	33.2	59	34
Atmospheric Deposition	20	26.2	93	17.0	48	27
Instruments	8	10.5	28	5.1	23	13
Bed and Bank Erosion	4.1	5.4	160	29.2	21	12
Switches and Thermostats	9	11.8	11	2.0	10	5.7
Fluorescent Lighting	2.4	3.1	5.8	1.1	4.1	2.3
Paint	1	1.3	4	0.7	2.6	1.5
Railway Lines	0.09	0.1	3	0.5	1.5	0.9
Identified Industrial Contaminated Areas	0.25	0.3	7.4	1.4	1.4	0.8
Landfill	0.5	0.7	1.5	0.3	1	0.6
Laboratory	0.2	0.3	1.4	0.3	1	0.6
Gasoline	0.1	0.1	2	0.4	1	0.6
Batteries	0.15	0.2	1.5	0.3	0.8	0.5
Auto-Recycling	0.4	0.5	46	8.4	0.7	0.4
Dental	0.2	0.3	0.6	0.1	0.4	0.2
Other Uses	0.006	0.01	0.18	0.03	0.09	0.1
	<u>76</u>	<u>100</u>	<u>547</u>	<u>100</u>	<u>176</u>	<u>100</u>

Table 2-12. Summary of mass input of PCBs to Bay Area stormwater conveyances.

Source	Low		High		Best	
	(kg)	(%)	(kg)	(%)	(kg)	(%)
Watershed Surface Sediment Erosion	4.0	31	73	41	30	58
Building Demolition and Remodeling	4.1	32	4.1	2.3	4.1	7.9
PCBs Still in Use	2.0	16	6.0	3.4	4.0	7.7
Bed and Bank Erosion	0.20	1.6	43	24	2.9	5.6
Transformers and Large Capacitors	1.2	9.4	4.3	2.4	2.8	5.4
Atmospheric Deposition	0.40	3.1	30	17	2.8	5.4
Identified Industrial Contaminated Areas	0.53	4.2	6.4	3.6	2.0	3.9
Plasticizers	0.035	0.28	2.1	1.2	1.1	2.1
Railway Lines	0.10	0.79	6.0	3.4	1.1	2.1
Small Capacitors	0.14	1.1	0.80	0.45	0.50	0.97
Auto-Recycling	0.0009	0.0071	0.76	0.43	0.40	0.77
Other Dissipative Uses	0.002	0.016	0.11	0.062	0.060	0.12
Lubricants	0	0	0	0	0	0
Landfills	0	0	0	0	0	0
	<u>13</u>	<u>100</u>	<u>177</u>	<u>100</u>	<u>52</u>	<u>100</u>

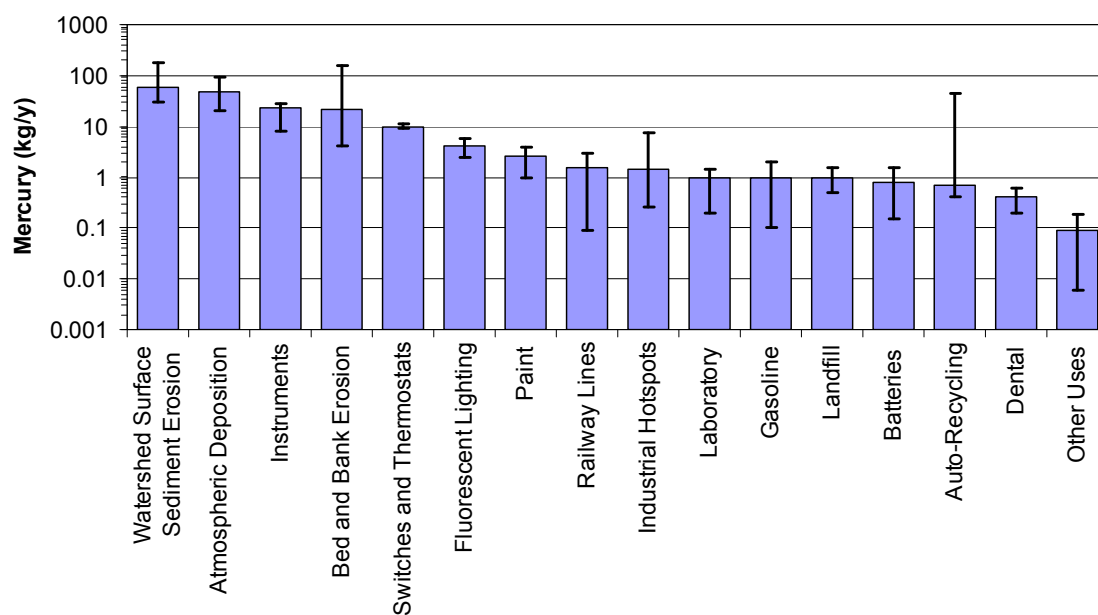


Figure 2-12. Comparison of mass input of Hg to Bay Area stormwater conveyances showing the range in the current estimates.

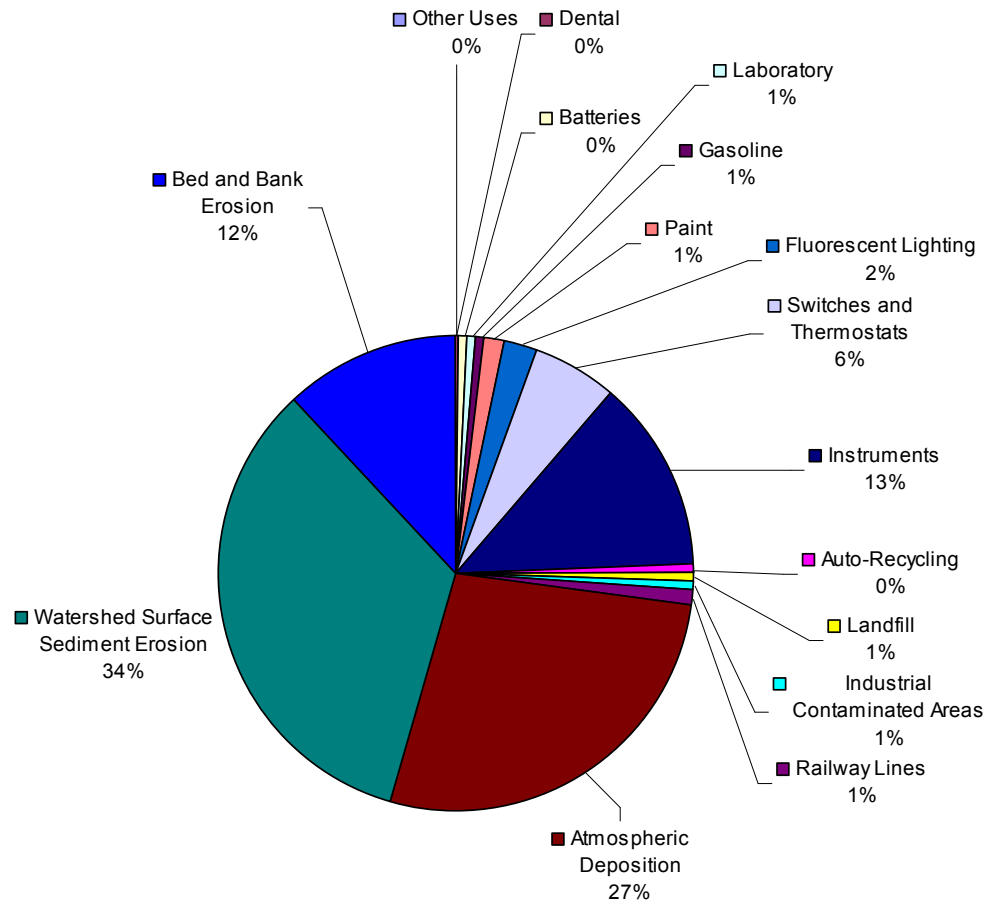


Figure 2-13. Comparison of mass input of Hg to Bay Area stormwater conveyances based on our best current estimates.

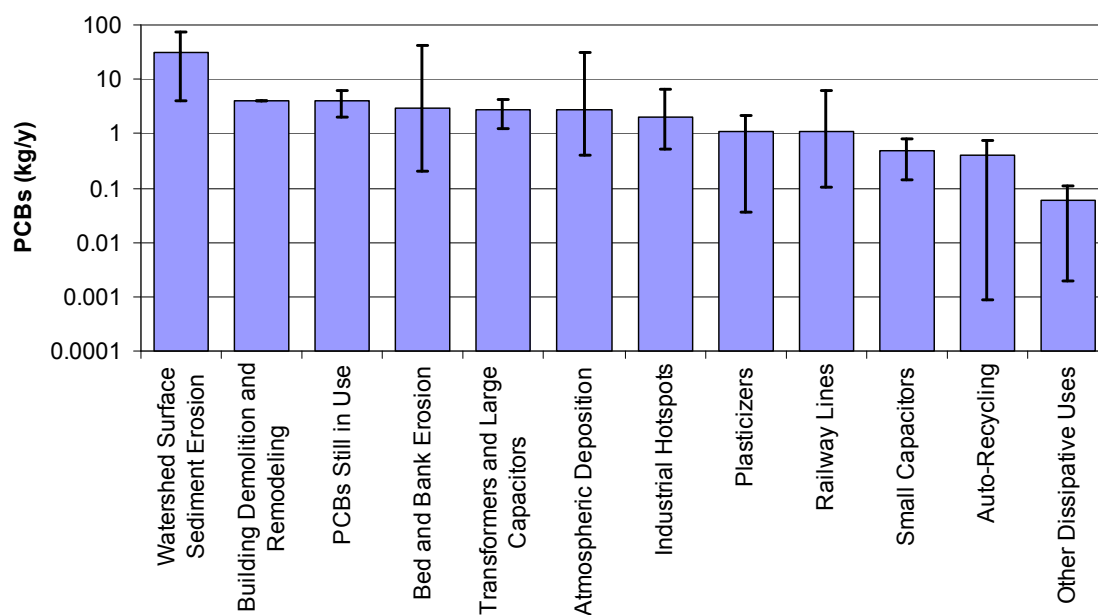


Figure 2-14. Comparison of mass input of PCBs to Bay Area stormwater conveyances showing the range in the current estimates.

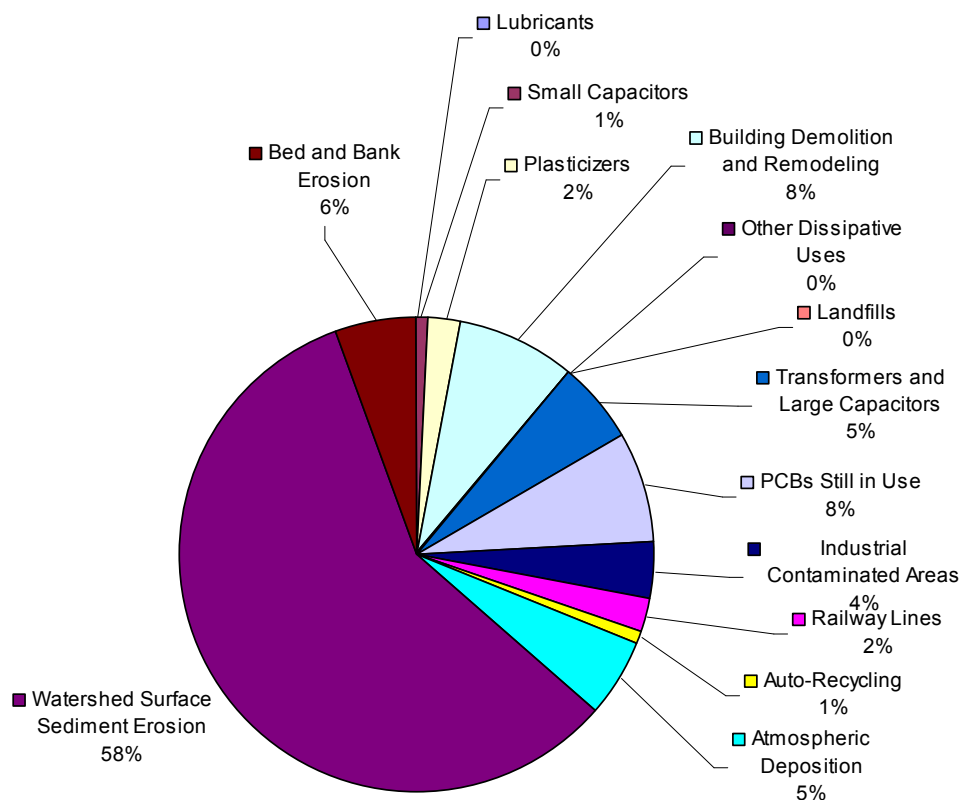


Figure 2-15. Comparison of mass input of PCBs to Bay Area stormwater conveyances based on our best current estimates.

Table 2-13. Stormwater conveyance Hg and PCB budgets based on estimates of inputs, removals and outputs.

	Hg	PCBs		
		Low Estimate	High Estimate	Best Estimate
Watershed input (kg)	176	13	177	52
Street Sweeping Removal (kg)	5	1	8	3
Storm drain Facility Cleaning (kg)	4	5	32	11
Channel De-silting (kg)	84	3	23	10
Load to the Bay (kg)	160	9	100	39
Error / Unaccounted (kg)	-77	-5	14	-11

2.5. REFERENCES

- Al-Haddad, A., Madany, I. M. and Abdullah, F. J. 1993. Levels of PCBs and PAHs in Bahrain soil. *Environment International*, 19 (3), 277-284
- Arbitman, N. and Gerel, M. 2003. Managing End-of-Life Vehicles to Minimize Environmental Harm: White paper on Sustainable Conservation's Auto Recycling Project. San Francisco, CA. December 2003. pp.1.
- Bannerman, R. T. Owens, D. W. Dodds, R. B. and Hornewer, N. J. 1993. Sources of Pollutants in Wisconsin Stormwater. pp.241.
- Bannerman, R., Fries, G. and Horwath, J. 2003. Source area and regional storm water treatment practices: options for achieving phase II retrofit requirements in Wisconsin. February 2003. pp.1.
- Benthe, C., Heinzow, B., Jessen, H., Mohr, S. and Rotard, W. 1992. Polychlorinated biphenyls. Indoor air contamination due to thiokol-rubber sealants in an office building. pp.1481.
- Breivik, K., Sweetman, A., Pacyna, J. M., and Jones, K. C. (2002a). Towards a global historical emission inventory for selected PCB congeners - a mass balance approach. 1. Global production and consumption. *Science of the Total Environment*, 290, 181-198.
- Breivik, K., Sweetman, A., Pacyna, J. M., and Jones, K. C. (2002b). Towards a global historical emission inventory for selected PCB congeners - a mass balance approach. 2. Emissions. *Science of the Total Environment*, 290, 199-224.
- Bremle, G. and Larsson, P. 1997. Long-term variations of PCB in the water of a river in relation to precipitation and internal sources. *Environmental Science and Ecology*. pp.3232.
- Caltrans, 2005 . Traffic data branch. <http://traffic-counts.dot.ca.gov/>
- Chutke, N. L., Ambulkar, M. N. and Garg, A. N. 1995. An environmental pollution study from multielemental analysis of pedestrian dust in Nagpur city, Central India. *The Science of the Total Environment*, 164, pp.185-194.

- Conaway, C. H., Mason, R. P., Steding, D. J. and Flegal, A. R. 2005. Estimate of mercury emission from gasoline and diesel fuel consumption, San Francisco Bay area, California. Atmospheric Environment. San Francisco, CA. pp.101.
- Davis, J.A., L. McKee, J. Leatherbarrow, and T. Daum. 2000. Contaminant Loads from Stormwater to Coastal Waters in the San Francisco Bay Region: Comparison to Other Pathways and Recommended Approach for Future Evaluation. San Francisco Estuary Institute, Richmond, CA.
- DEQ, 2003. Fact Sheet: Common products containing mercury. Oregon Department of Environmental Quality. March, 2003. 1pp.
<http://www.deq.state.or.us/wmc/factsheets/commonproductscontainingmercuryfactsheet.pdf>
- Dovzak, C., and Sommers, C., 2004. Pollutants of concern source assessment report. Report prepared by the Contra Costa County Clean Water Program in fulfillment of the NPDES Permit Provision C.8(c) for the San Francisco Bay Regional Water Quality Control Board, July 2004. 64pp.
- DTSC, 2002. Mercury report. Department of Toxic Substances Control, Hazardous Waste Management Program, State Regulatory Programs Division, Sacramento, August 2002. 125pp.
- DTSC, 2002b. Draft Report: California's automobile shredder waste initiative. Department of Toxic Substances Control, Hazardous Waste Management Program, Statewide Compliance Division, Sacramento, December 2002. 44pp.
- DTSC, 2003. PCB advisory for schools: How voluntary lighting retrofits can address hidden dangers. Department of Toxic Substances Control, Sacramento, June 2003. 17pp.
- DTSC, 2004. Self-training manual for removing mercury switches from vehicles: A guide for auto repair shops. Department of Toxic Substances Control, Sacramento, June, 2004. 36pp.
- EIP Associates, 1997. Polychlorinated biphenyls (PCBs) source identification. A report prepared for Palo Alto Regional Water Quality Control Plant, Palo Alto, CA. October 1997. 16pp + appendix.
- EPA, 1993 EPA superfund record of decision: Lorentz Barrel & Drum Co. EPA/ROD/R09-93/094.
- EPA, 1997a. Location and estimating air emissions from sources of mercury and mercury compounds. United States Office of Air Quality Planning And Standards. Environmental Protection Agency. EPA-454/R-97-012. December 1997.
- EPA, 1997b. Mercury Study Report to Congress Volume II: An Inventory of Anthropogenic Mercury Emissions in the United States. Office of Air Quality Planning & Standards and Office of Research and Development. United States Environmental Protection Agency. EPA-452/R-97-004. December 1997.
- Erickson, M.D., 1992. Analytical chemistry of PCBs. CRC Press, Inc./ Lewis Publishers, Boca Raton, Florida. 508pp.
- Granier, L. and Chevreuil, M. 1997. Behaviour and spatial and temporal variations of polychlorinated biphenyls and lindane in the urban atmosphere of the Paris area, France. Atmospheric Environment. pp.3787.
- Grigal, D.F. 2002. Inputs and outputs of mercury from terrestrial watersheds: a review. Environmental Reviews. pp.1.

- Gunther, A.J., Salop, P., Bell, D., Feng, A., Wiegel, J., and Wood, R., 2001. Initial characterization of PCB, mercury, and PAH contamination in drainages of western Alameda County. Report prepared by Applied Marine Sciences for the Alameda Countywide Clean Water Program. 43pp.
- Harrad, S.J., Sewart, A.P., Alcock, R., Boumphrey, R., Burnett, V., Durarte-Davidson, R., Halsall, C., Sanders, G., Waterhouse, K., Wild, S.R. and Jones, K.C. 1994. Polychlorinated biphenyls (PCBs) in the British environment: sinks, sources and temporal trends. *Environmental Pollution*. pp.131
- Heinzow, B. G. J., Mohr, S., Ostendorp, G., Kerst, M. and Korner, W. 2004. Dioxin-like PCB in indoor air contaminated with different sources. *Organohalogen Compounds*. pp.2470.
- Hetzel, F. 2004. PCBs in San Francisco Bay: Total Maximum Daily Loads Report. San Francisco Bay Regional Water Quality Control Board. Oakland, CA. January 2004. 69pp. http://www.waterboards.ca.gov/sanfranciscobay/TMDL/SFBayPCBs/pcbs_tmdl_project_report010804.pdf (cited July 2005).
- Irvine, K. N. and Loganathan, B. G.. 1998. Localized enrichment of PCB levels in street dust due to redistribution by wind. *Water Air and Soil Pollution*. pp.603.
- Kaminski, D., and Landsberger, S., 2000. Heavy metals in urban soils of East St. Louis, IL, Part I: Total concentration of heavy metals in soils. *Journal of the Air & Waste Management Association* vol. 50, n^o9, pp. 1667-1679.
- Keeler G.J., Pacyna J.M., Bidleman T.F., Nriagu J.O., 1993. Identification of Sources Contributing to the Contamination of the Great Waters (Revised) EPA/453/R-94/087. Washington, DC:U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards.
- Kleinfelder, Inc. 2005. Sediment sampling report etite street pump station wawtershed, Oakland, California. City of Oakland PWA – ESD. July 2005. pp.1-31.
- KLI, 2001. Joint stormwater agency project to study urban sources of mercury and PCBs. Report prepared by Kinnetic Laboratories, Inc. for Santa Clara Valley Urban Runoff Pollution Prevention Program, Contra Costa Clean Water Program, San Mateo Countywide Stormwater Pollution Prevention Program, Marin County Stormwater Pollution Prevention Program, Vallejo Flood Control and Sanitation District, Fairfield-Suisun Sewer District. 44pp + appendices.
- KLI, 2002. Joint stormwater agency project to study urban sources of mercury, PCBs, and organochlorine pesticides. Report prepared by Kinnetic Laboratories, Inc. for Santa Clara Valley Urban Runoff Pollution Prevention Program, Contra Costa Clean Water Program, San Mateo Countywide Stormwater Pollution Prevention Program, Marin County Stormwater Pollution Prevention Program, Vallejo Flood Control and Sanitation District, Fairfield-Suisun Sewer District. 71pp.
- Kohler, M., Tremp, J., Zennegg, M., Seiler, C., Minder-Kohler, S., Beck, M., Lienemann, P., Wegmann, L. and Schmid, P. 2005. Joint sealants: and overlooked diffuse sousce of Polychlorinated biphenyls in buildings. *Environmental Science and Technology*. pp.1967.
- LWA, 2006. PCB TMDL Implementation Plan Development. A report prepared for the Clean Estuary Partnership by Larry Walker Associates, TDC Environmental, LLC and Ann Blake. <http://www.bacwa.org/LinkClick.aspx?fileticket=2m8g5aRitKQ%3d&tabid=126&mid=572>.

- Malawska, M. and Wilkomirski, B. 2001. An analysis of soil and plant (*Taraxacum officinale*) contamination with heavy metals and polycyclic aromatic hydrocarbons (PAHs) in the area of the railway junction Ilawa Główna, Poland. *Water, Air, and Soil Pollution*, 137, pp.339-349.
- Mason, Robert P. and Sullivan, Kristin A. 1998. Mercury and methylmercury transport through an urban watershed. *Watershed*. pp.331.
- McKee, L., Leatherbarrow, J., Pearce, S., and Davis, J., 2003. A review of urban runoff processes in the Bay Area: Existing knowledge, conceptual models, and monitoring recommendations. A report prepared for the Sources, Pathways and Loading Workgroup of the Regional Monitoring Program for Trace Substances. SFEI Contribution 66. San Francisco Estuary Institute, Oakland, Ca.
- McKee, L., and Leatherbarrow, 2005. Concentrations and loads of mercury, PCBs, and OC pesticides in the lower Guadalupe River, San Jose, California: Water Years 2003 and 2004. A Technical Report of the Regional Watershed Program: SFEI Contribution 409. San Francisco Estuary Institute, Oakland, CA. 72pp.
- MDEQ, 2002. Michigan Mercury Switch Study. A report prepared by a stakeholder workgroup Work Group in Cooperation With Kalamazoo County Household Hazardous Waste Center: Dan Adsit - Ford, representing the Alliance; Ross Good – Daimler Chrysler, representing the Alliance; Marcia Horan – MDEQ; Steve Kratzer - MDEQ; Ken Schram - Schram Auto Parts; Bill Stough - Sustainable Research Group. Michigan Department of Environmental Quality (MDEQ). December 19th, 2002. 38pp.
<http://www.deq.state.mi.us/documents/deq-ess-p2-mercury-michiganswitchstudy.pdf>
- OSPAR, 2004. Polychlorinated biphenyls (PCBs). OSPAR Commission 2001 (2004 Update) Hazardous substances series. ISBN 0 946956 78 2.
- Pearce, S., McKee, L., and Shonkoff, S., 2005. Pinole Creek watershed sediment source assessment. A Technical Report of the Regional Watershed Program prepared for the Contra Costa Resources Conservation District (CC RCD): SFEI Contribution #316. San Francisco Estuary Institute, Oakland, CA. 102pp + appendix.
- PG&E, 2004. Second Annual Corporate Responsibility Report. Pacific Gas and Electric Corporation. http://www.pgecorp.com/corp_responsibility/reports/2004/
- Priha, E., Hellman, S. and Sorvari, J. 2005. PCB contamination from polysulphie sealants in resident areas-exposure and risk assessment. *Chemosphere*, 59, 537-543.
- Rossi, L., de Alencastro, L., Kupper, T. and Tarradellas, J. 2004. Urban stormwater contamination by polychlorinated biphenyls (PCBs) and its importance for urban water systems in Switzerland. *Science of the Total Environment*. pp.179.
- Sakai, S., Urano, S. and Takatsuki, H. 1998. Leaching behavior of persistent organic pollutants (POPs) in shredder residues. *Chemosphere*. pp.2047.
- Salop, P. and Akashah, M. 2004. A review of source control options for selected particulate-associated TMDL pollutants. Alameda Countywide Clean Water Program. March 2004. pp.1-76.
- Salop, P., Abu-Saba, K., Gunther, A., and Feng, A., 2002. 2000-01 Alameda County watershed sediment sampling program: Two-year summary and analysis. Report prepared for the Alameda Countywide Clean Water Program. September, 2002. 33pp.
- Scherbatskoy, T., Shanley, J. B. and Keeler, G. J. 1998. Factors controlling mercury transport in an upland forested catchment. *Water Air and Soil Pollution*. pp.427.

- Schueler, T. R. 1996. Impervious Surface Coverage: The Emergence of a Key Environmental Indicator.
- Sher, 2001. California Mercury Reduction Act of 2001. BILL NUMBER: SB 633.
- Steding, D. J. and Flegal, A. R. 2002. Mercury concentrations in coastal California precipitation: evidence of local and trans-Pacific fluxes of mercury to North America. *Journal of Geophysical Research*. pp.11-1.
- Steding, D. J., Dunlap, C. E. and Flegal, A. R. 2002. New isotopic evidence for chronic lead contamination in the San Francisco Bay estuary system: Implications for the persistence of past industrial lead emissions in the biosphere. *PNAS*. pp.11181
- SMCSTOPPP, 2002. PCBs Use and/ or release sites in San Mateo County. San Mateo County Stormwater Pollution Prevention Program, February 25, 2002. xxpp.
- Sznopek, J.L. and Goonan, T.G., 2000, The materials flow of mercury in the economics of the United States and the world. Denver, CO. June 2000. 28pp.
- Tsai, P. and Hoenicke, R. 2001. San Francisco Bay atmospheric deposition pilot study Part 1: mercury. Oakland, CA. 45p.
- Tsai, P., Hoenicke, R., Hansen, E. and Lee, K. 2001. San Francisco Bay atmospheric deposition pilot study part 2: trace metals. Richmond, CA. pp.1.
- Tsai, P., Hoenicke, R. and Yee, D. 2002. Atmospheric concentrations and fluxes of organic compounds in the Northern San Francisco Estuary. *Environmental Science and Ecology*. pp.4741.
- UNEP, 1999. Guidelines for the identification of PCBs and materials containing PCBs. United Nations Environment Program. Geneva, Switzerland. August 1999. pp.34.
- Wisconsin DNR, 1997. The Wisconsin Mercury Source Book. Wisconsin Department of Natural Resources. <http://www.epa.gov/glnpo/bnsdocs/hgsbook/section1.pdf>
- Yang, Y. and Baumann, W. 1996. Study of polychlorinated biphenyls in street dust by supercritical fluid extraction-gas chromatography/mass spectrometry. *Fresenius' Journal of Analytical Chemistry*. Germany. pp.56.
- Yee, D., 2005. Written communication: Results from the National Atmospheric Deposition Program, San Jose (CA72).
<http://nadp.sws.uiuc.edu/sites/siteinfo.asp?net=MDN&id=CA72>

3. Hg and PCB Pollution in the Urban Environment

3.1 Introduction

Trace amounts of Hg and PCBs can be found in soils, sediments and organic carbon blanketing the entire urban environment. For example, Hg and PCBs are found on rooftops, in house dust, in park and garden soils, in sediments on pavement surfaces, in road dust, and in the sediments of urban stormwater conveyances. However, the distribution of mass is heterogeneous and influenced by factors such as proximity to intensive use areas such as factories or recycling yards, population density, traffic patterns, wind redistribution, rainfall patterns, and surface roughness. In the context of source control, the previous section described the ultimate sources of Hg and PCBs in the urban environment and the likely magnitudes of “leakage” from uses onto the watershed surfaces. Once pollutants are on watershed surfaces, rainfall, gravity, and wind can act upon the leaked mass, move it around, and transport a portion of it into the stormwater conveyance system. This section describes the concentrations and particle characteristics of Hg and PCBs in various urban media based on an extensive search of the literature. This is used to build a series of hypotheses on what might be found in components of the Bay Area environment. These concentrations and hypotheses are then compared to concentrations stormwater conveyances of the Bay Area, information that will assist in decisions of treatment control options at different scales. For consistency the following definitions are used throughout:

Parent material:	The geological bedrock formations from which soils are derived
Soils:	Unconsolidated material derived from weathering of underlying or adjacent bedrock formations and vegetative debris
Street dusts:	Unconsolidated material derived from erosion of local soils, wear of road materials, vehicular wear debris, atmospheric fallout/rainout, and vegetative debris
Sediments:	Unconsolidated material that have recently undergone transport or are undergoing transport by water in a natural or constructed fluvial setting

3.2 Review of International Literature on Hg and PCBs in Urban Media

3.2.1 Hg in Soils

By far the majority of information found in the literature is on concentrations of Hg and PCBs in bulk near surface soils. An extensive literature search was carried out without bias for depth of soil sampled (e.g. 0-2 cm, 0-5 cm etc), method of collection (e.g. mini corer, trowel), and laboratory methods (e.g. instruments, detection limits, appropriately described QA procedures). This data was used to develop a conceptual model (hypothesis) for Hg soil concentrations that might be expected in the Bay Area small tributaries and sewersheds under differing land use conditions. We did not review change in concentrations with depth because we were only interested in soil mercury that is easily mobile during rainfall events and derived from pollutant sources above ground.

3.2.1.1 Bulk Concentrations

Hg is found in soils even in remote parts of the world where the only pathways of arrival are weathering from parent geological materials (i.e. mercury is a naturally occurring element in soil (e.g. Bradford et al., 1996) and atmospheric deposition (e.g. Fitzgerald, et al., 1998). The lowest concentrations (<0.04 mg/kg) in our review of world soils data were associated with rural and remote locations 10s or 100s of km from any known sources (Table 3-1). Typical maximum observed concentrations out of the influence of urban or industrial pollution appear to range from 0.04-0.32 mg/kg with a median of 0.05 mg/kg. In urban areas with no discernable industrial impacts, maximum observed concentrations appear to range from 0.15-0.44 mg/kg with a median of 0.16 mg/kg. Hg in soils of agricultural areas impacted by urban or industrial emissions range between 0.42-31 mg/kg with a median of 0.84 mg/kg. The outlier (Wiersma et al., 1986) probably represents a special case where industrial emissions are deposited on the floodplain of the Rhine River (likely fine and much enriched particles preferentially deposit). In urban areas highly influenced by industrial activities, Hg concentrations range between 0.5-7 mg/kg with a median of 2.4 mg/kg. In urban areas on the industrial fringe and industrial areas, Hg concentrations range between 0.35-230 mg/kg with median of 0.86 mg/kg (Table 3-1). It is surprising that the median ends up lower for industrial land use than for urban land use with industrial influence but this is likely due to the nature of sampling and limited descriptions of land use and pollutant sources provided by many of the authors. Under these circumstances it might be more appropriate to group all urban areas with industrial influence and industrial areas together – in this case the range would be 0.35-230 mg/kg with a median of 2.3 mg/kg. Regardless of how we manipulate the data, there is a clear continuum from remote areas with low concentrations gradating through to industrial areas with very high soils concentrations (Figure 3-1). Concentrations vary by 3-4 orders of magnitude (we will see below that variation for PCBs is much greater).

3.2.1.2 Frequency Distribution

A common feature of soils pollution data is that the data are distributed log normal, and therefore the medians are less than the means. This is true in all cases where means and medians were provided by the authors (Table 3-1). This is caused by a combination of source characteristics and, to a lesser extent, dispersion processes and has long been recognized (e.g. Klein, 1972). It is particularly true for substances such as mercury that are associated with particular industrial processes as well as more general urban uses but it is less true for copper for example, that is more evenly distributed in the urban environment (Klein, 1972). The same concept emerges if we look at the frequency distribution of maximum concentrations from the studies listed in Table 3-1. For example, in urban areas highly influenced by industrial activities, median Hg is 2.4 mg/kg and mean is 2.8 mg/kg. Or if we group all urban areas with industrial influence and industrial areas together, the median is 2.3 mg/kg and the mean is 21 mg/kg. Thus, the soils Hg data exhibit a non-normal distribution at the scales of individual urban systems and on-mass among systems where high concentrations are rare but indicators of and associated with a local large point source and moderate to low concentrations are associated with distributed local sources and long range atmospheric transport.

3.2.1.3 Geological Background and Enrichment Factors

Another confounding factor in the interpretation of Hg soils data is the influence of local geological sources (another obvious contrast between Hg and PCBs given there are no natural sources of PCBs). In a study of metropolitan area of Berlin, background soils concentrations associated with geological sources were very low (<0.04 mg/kg) (Birke and Rauch, 2000). In this case all elevated concentrations were attributed to various sources of pollution. This is similar to a study of the Turner Valley in Canada where background soil concentrations were also very low – again all of the elevated concentrations were attributed anthropogenic sources (in this case a gas works) (Kohut et al., 2000). In other cases it has been noted that even though there are significant local geological sources, the fingerprint of industrial pollution overwhelmed the geological fingerprint (e.g. Horvat et al., 2003). In yet another case, it was possible to distinguish geological anomalies associated with bedrock, tectonics, and technogenic haloes around industrial, urban, and mining activities (Koval et al., 1999). Thus, the issue of interpretation of Hg data over a variable background is an important one to consider and is particularly pertinent in the San Francisco Bay Area given the serpentine rocks of the coast range of California are enriched in Hg (e.g. Bailey, 1964). Enrichment factors (EFs) also occur as a function of natural weathering, i.e. basic soil forming processes. If one assumes that these basic soil forming processes are similar across the majority of a given city or the industrial/urban areas Bay Area, then variation in naturally caused EFs would be minimal compared to anthropogenic signals. An exception in the Bay Area could occur when comparing for example Guadalupe River with Coyote Creek. There may be lenses of naturally high soil Hg concentrations in the common floodplains of the alluvial

Table 3-1. Hg concentrations (mg/kg) in soils in and near cities or industrial areas from around the world. Note where no range was given by the authors, the maximum was conservatively calculated as the mean or median + 1 standard deviation.

SOILS	Location	Mean	Median	Range	Max (mg/kg)	Ref
Forest	Ilawa Glowna, Poland			0.037	0.037	15
Open space (1-30 km from coal fired power plant)	Four Corners, NM	0.016±0.0067		0.006-0.45	0.045	4
Residential and commercial	Maricopa County, Arizona	0.04±0.01		0.02-0.05	0.05	23
Background Illinois soils	Illinois, USA	0.033±0.020			0.053	5
Rural unpolluted	Tarragona County, Spain	0.04±0.02			0.06	19
Forest (Next to road)	Dortmund, Germany			0.08-0.12	0.12	17
Residential	Southeastern Michigan	0.08±0.07			0.15	18
Residential	Tarragona County, Spain	0.08±0.07			0.15	19
Industrial (Oil refining and petrochemical)	Tarragona County, Spain	0.08±0.08			0.16	19
Agricultural and horticultural	The Netherlands	0.08		0-0.32	0.32	29
Urban (Traffic and heating Hg sources)	Bruno, Czech Republic	0.35			0.35	26
Forest	Southern Sweden			0.15-0.37	0.37	7
Agricultural - with industrial influence (Asphalt plant and other industry)	Grand Rapids, MI	0.11±0.09	0.09	0-<0.42	0.42	8
Commercial	Southeastern Michigan	0.2±0.24			0.44	18
Agricultural with industrial influence (Mines, steel, chemical, coking, power generation, smelter)	Suszec Commune, Poland	0.07±0.02	0.06	0.02-0.46	0.46	14
Industrial (Paper mill and chlor-alkalai)	Amursk, Russia			0.004-0.464	0.46	10
Arctic near mining industry	Finland, Norway and Russia			0.03-0.49	0.49	25
Industrial	Southeastern Michigan	0.2±0.3			0.5	18
Residential - with industrial influence (Asphalt plant and other industry)	Grand Rapids, MI	0.1±0.1	0.07	0->0.51	0.51	8
Industrial - with industrial influence (Asphalt plant and other industry)	Grand Rapids, MI	0.14±0.1	0.11	0-<0.51	0.51	8
Airport	Grand Rapids, MI	0.33±0.18	0.17		0.51	8
Urban gardens with industrial influence	Nagpur, Central India	0.43±0.13			0.56	3
Industrial (Near a paper mill)	Coastal Motril, SE Spain			0.117-0.760	0.76	20
Agricultural with industrial influence (Paints, batteries, electrical manufacture etc)	Changhua County, Taiwan	0.25±0.2	0.18	0.00-0.84	0.84	12
Industrial (Coal + black oil fired power plants)	Khabarovsk, Russia			11-950	0.95	10
Urban with industrial influence	Xuzhou, China	0.29±0.32	0.18	0.02-1.3	1.3	28

Table 3-1 cont. Hg concentrations (mg/kg) in soils in and near cities or industrial areas from around the world. Note where no range was given by the authors, the maximum was conservatively calculated as the mean or median + 1 standard deviation.

SOILS	Location	Mean	Median	Range	Maximum	Ref
Railway junction	Ilawa Glowna, Poland			0.014-2.28	2.3	15
Urban with industrial (Mining of sulphide minerals and coal combustion)	Oslo, Norway	0.518	0.48	0.01-2.3	2.3	27
Urban with industrial influence (Paper, wood and metal)	Jakobstad, Finland		0.093	0.011-2.309	2.3	24
Industrial (Chemical, paper, aluminum, cement etc)	Baikal, Russia	0.029		<0.002-2.4	2.4	11
Urban with industrial influence (Iron, steel and non-ferrous)	Aviles, Spain	0.57		0.17-2.41	2.4	21
Urban with industrial influence (Cement and other unspecified)	Central Jordan	1.81±0.72		0.6-3.05	3.1	1
Residential - High-density with industrial influence (Metals, chemical, paint, asphalt and electrical)	Berlin, Germany	0.48	0.34	3.5	3.5	2
Agricultural - with industrial influence (Metals, chemical, paint, asphalt and electrical)	Berlin, Germany	0.13	0.06	3.8	3.8	2
Residential -low-density with industrial influence (Metals, chemical, paint, asphalt and electrical)	Berlin, Germany	0.19	0.1	5	5	2
Urban parks and green areas with industrial influence (Gas and chemical)	Sicily, Italy			0.04-6.96	7	16
Industrial (Coke, chemical, mercury and metallurgical plants)	Donets Basin, Ukraine	<19			19	22
Urban with industrial influence (Mining and metallurgical)	Mieres, Spain	4.24	2	0.5-25	25	13
Agricultural and horticultural - industrial influence from floodplain flooding	The Netherlands	0.16	0.07	0-31	31	29
Industrial (Metals, chemical, paint, asphalt and electrical)	Berlin, Germany	0.66	0.21	71.2	71	2
Industrial (Area of mercury production)	Guizhou Province, China			19-156	156	6
Industrial (Former gas plant site)	Turner Valley, Canada			0.07-230	230	9

1 Banat et al., 2005	7 Johansson et al., 1995	13 Loredó et al., 2003	19 Nadal et al., 2004	25 Reimann et al., 1997
2 Birke and Rauch, 2000	8 Klein, 1972	14 Loska et al., 2004	20 Navarro et al., 1993	26 Strnad et al., 1994
3 Chutke et al., 1995	9 Kohut et al., 2000	15 Malawska and Wilkomirski, 2001	21 Ordonez et al., 2003	27 Tjhuis et al., 2002
4 Crockett and Kinnison, 1979	10 Kot and Matyushkina, 2002	16 Manta et al., 2002	22 Panov et al., 1999	28 Wang et al., 2005
5 Dreher and Follmer, 2004	11 Koval et al., 1999	17 Munch, 1993	23 Parker et al., 2000	29 Wiersma et al., 1986
6 Horvat et al., 2003	12 Lin et al., 2002	18 Murray et al., 2004	24 Peltola and Astrom, 2003	

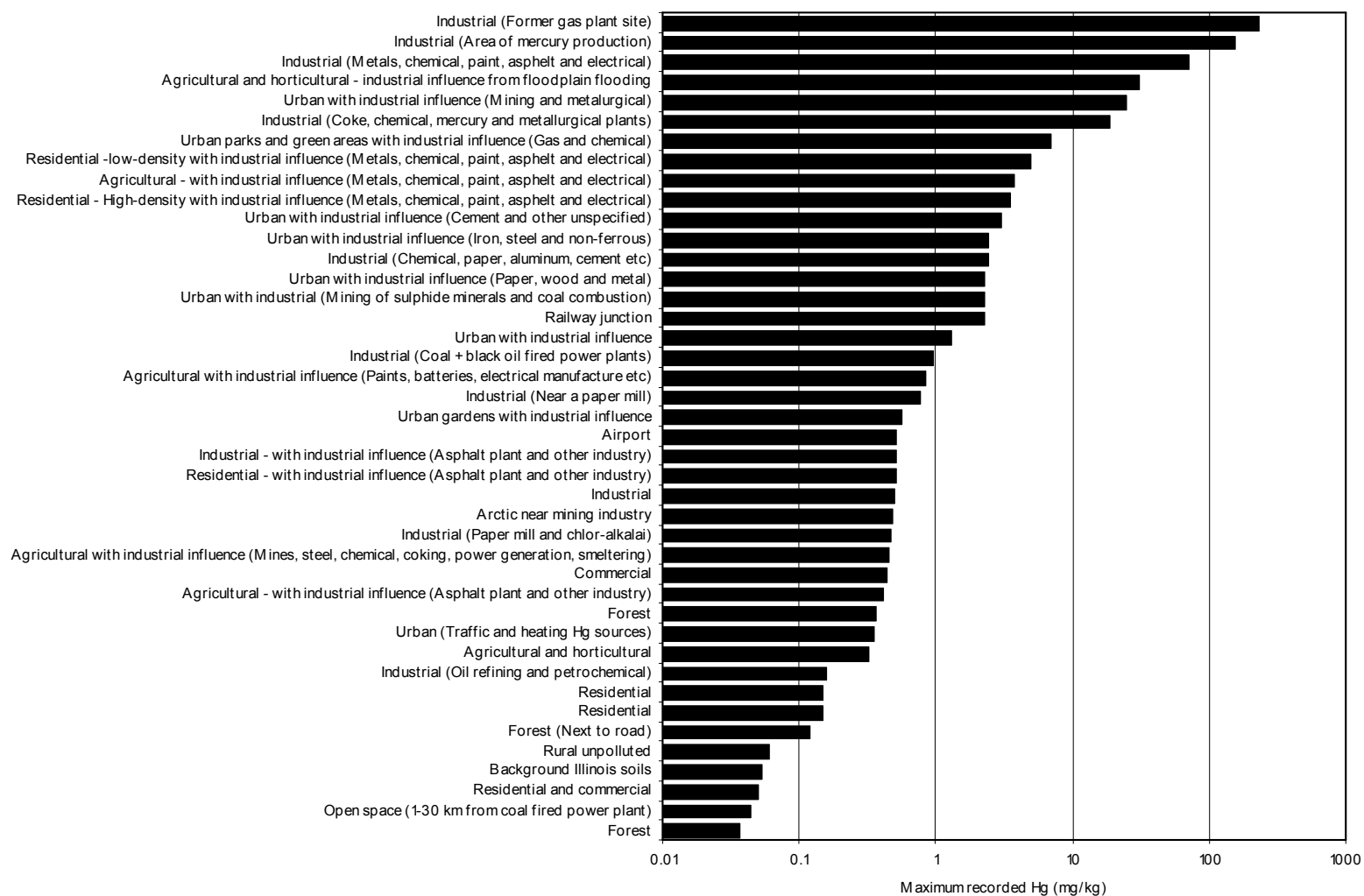


Figure 3-1. Hg concentrations in soils from studies in other parts of the world. Data and sources listed in Table 3-1.

fan associated with these two fluvial systems. That said - in the cases reviewed during our literature search, the anthropogenic signal always outweighed the natural signal.

To help interpret when anthropogenic factors have caused elevated soils concentrations rather than natural factors, some authors have calculated an EF by comparing local background concentrations to concentrations in the area of interest (e.g. Manta et al., 2002; Loska et al., 2004). An $EF > 1$ indicates enrichment over natural geological sources and an $EF < 1$ indicates depletion. Other authors have simply referenced local low level concentrations and assumed they represent background (e.g. Malawska and Wilkomirski, 2001; Murray et al., 2004). In the study of heavy metals in an urban watershed of Michigan, EF was greater than 2 in areas influenced by chemical industries (Murray et al., 2004). EFs as high as 35 were calculated for the city of Palermo, Italy in areas influenced by gas and chemical plants (Manta et al., 2002) and from 7-90 in a railway junction study area in Poland (Malawska and Wilkomirski, 2001). The use of EFs might be a useful tool in the Bay Area especially in floodplain soils of the City of San Jose and other urban areas downstream from Hg mineralized zones of coast range serpentines that might be naturally enriched in mercury. Indeed these areas might have naturally elevated Hg concentrations compared to urban areas built on floodplains downstream from other parent geological formations. It is possible that these natural soil concentrations might confound attempts to reach target levels of 0.2 mg/kg described in the TMDL (Looker and Johnson, 2004).

3.2.1.4 Dispersal in Relation to Wind Direction

Gaseous and particulate Hg can disperse from sources by advective wind or wind generated locally by traffic. The effectiveness of wind dispersal is influenced by the velocity and constancy of both velocity and direction as well and the height above ground of the Hg release. A number of studies have purposefully investigated the dispersal of Hg from point sources (e.g. Crockett and Kinnison, 1979; Münch, 1993; Navarro et al., 1993; Loredó et al., 2003) and inadvertently (Malawska and Wilkomirski, 2001). Crockett and Kinnison (1979) in their investigation of the Four Corners coal fired power plant found no indication of dispersal anywhere within a 0-30 km radius. Münch (1993) measured Hg in the vicinity of a heavily traveled forest road and found that roadside vegetation limited dispersal to <10 m (indicated by background concentrations). Navarro et al. (1993) in their investigation of Hg dispersal from Hg mining spoil heaps found that concentrations had decreased from <25 mg/kg to <5 mg/kg within 2 km. Similarly, in a study of Hg dispersion from a paper mill, concentration in agricultural land decreased from 0.76 mg/kg near the mill to 0.12 mg/kg at the distance of 3 km (Navarro et al., 1993). They found a strong linear correlation between distance and Hg content in soils ($r^2=0.86$). In a study of pollution associated with a railway junction, Malawska and Wilkomirski (2001) found that Hg in surface soils had decreased from a maximum of 2.28 mg/kg to 0.037 mg/kg at the control location 2 km distant. In fact within their 2 km² study area, concentrations varied substantially from 0.016-2.28 mg/kg indicting limited dispersal.

Other studies have mapped in great detail the dispersion of Hg throughout urban environments and thus inadvertently the magnitude and distance of dispersal (e.g. Klein,

1972; Birke and Rauch, 2000; Tjihuis et al., 2002) while others have mapped the dispersion of Hg throughout wider areas that include urban, industrial and agricultural areas (e.g. Lin et al., 2002). Klein (1972) mapped Hg concentration in an urban area using a 1 mile sampling grid. He commented on the unique pattern that Hg exhibited relative to Zn. In his study area of 19x16 miles there were 9 Hg hotspots with Hg concentrations >0.3 mg/kg. In many cases these hotspots were elongated in the direction for the prevailing winds (from the west and south) and concentrations could vary from >0.3 mg/kg to <0.06 mg/kg in a distance of 2-3 km. In another systematic study of a city, a 1 km sampling grid was adopted across an area of about 500 km² (Tjihuis et al., 2002). The greatest concentrations were found in the dense urban center with hotspots in other areas. Although it is difficult to resolve absolute size of the 20 circle sizes representing concentrations ranging from 0.005 – 2.3 mg/kg it is certain that concentrations around isolated hotspots varied by as much as >1.2 to <0.05 within 2 km in an easterly direction and sometimes within 1 km. Thus, it appears under some industrial conditions when Hg is discharged from chimneys far above the ground, there may be no local on ground soil Hg signature, but for general non-point urban and ground level point sources, dispersion appears to be limited to <3 km and perhaps 1-2 km commonly.

3.2.2 Hg in Rooftop particles

Mercury and PCBs have an atmospheric pathway associated with long range atmospheric transport and sources dispersed locally by wind turbulence or gaseous emission. The dry-fall and wet-fall of the atmospheric Hg and PCB burden is a source to urban areas and remote parts of the world. Some of this material falls on roof tops where it can be transported off roofs during periods of rainfall. In addition, roofs made of or coated with zinc, iron, and copper can be a source of these metals and roofs coated with asphalt can be a source of PAHs and mercury. Only one study was found that described roof top particulate concentrations of Hg (Van Metre and Mahler, 2003) (Table 3-2; Figure 3-2) and no studies were found on roof top particle concentrations of PCBs.

3.2.2.1 Bulk concentrations

Bulk concentrations of Hg on roofs were measured 12 m and 102 from a major expressway at Camp Mabry in Austin Texas (Van Metre, 2003). Two roofing material types were studied (galvanized and asphalt). Concentrations were greater on roof closer to the expressway (0.16-0.23 mg/kg) than farther distant (0.09-0.31 mg/kg) for asphalt roofs. Similarly, concentrations were greater on galvanized roofs closer to the expressway (0.06-0.08 mg/kg) than farther distant (0.06-0.18 mg/kg). Concentrations appeared to be greater on asphalt relative to galvanized roofs, a difference that could be associated with contrasting trapping efficiencies of each type of roofing material or that the asphalt is a source unto itself; the authors suggest this warrants further investigation (Van Metre, 2003). Rooftops in this watershed cover 29% of the total area. Van Metre (2003) estimated that Hg runoff from roofs contributed to 46% of the total Hg load in the local watershed.

Table 3-2. Hg concentrations (mg/kg) in particles in street sweepings, on roof tops, in street dust, and in selected BMPs in other parts of the world. Note where no range was given by the authors, the maximum was conservatively calculated as the mean or median + 1 standard deviation.

Sweepings, Roofs, Street dust, and BMPs	Location	Mean	Median	Range	Maximum	Ref
Street sweepings - Residential/commercial	Pensacola, Florida	0.0188		0.0006-0.0502	0.050	4
Street sweepings (no industrial/urban)	Urbana, Illinois		0.09±0.008		0.098	2
Catch basin cleanings - Residential	Snohomish County and Seattle, WA		0.08	0.07-0.14	0.14	7
Catch basin cleanings - Industrial	Snohomish County and Seattle, WA		0.08	0.04-0.15	0.15	7
Catch basin cleanings - Commercial	Snohomish County and Seattle, WA		0.13	0.07-0.16	0.16	7
Roof tops - Galvanized metal adjacent to motorway	Camp Mabry, Austin, TX			0.06-0.18	0.18	8
Swales - Commercial and Agricultural	Pensacola, Florida	0.0689		0.261-0.1818	0.18	4
Roof tops - Asphalt shingle adjacent to motorway	Camp Mabry, Austin, TX			0.09-0.31	0.31	8
Retention ponds - Residential/commercial	Pensacola, Florida	0.1108		0.0092-0.3945	0.39	4
Street Dust - Residential	Wellington, New Zealand			0.061-0.5	0.50	3
Street Dust - Industrial (Refractory, pipe manufacturing, and other)	Nagpur, Central India	0.55±0.01		0.41-0.65	0.65	1
Street Dust - Bus stations with industrial influence (Refractory, pipe manufacturing, and other)	Nagpur, Central India			0.15-0.74	0.74	1
Street Dust - Airport with industrial influence (Refractory, pipe manufacturing, and other)	Nagpur, Central India	0.76			0.76	1
Street Dust - Highways in industrial city (Refractory, pipe manufacturing, and other)	Nagpur, Central India			0.65-0.80	0.80	1
Street Dust - Residential/commercial with industrial influence (Refractory, pipe manufacturing, and other)	Nagpur, Central India	0.53±0.34		0.05-0.89	0.89	1
Street Dust - Railway	Nagpur, Central India			0.16-1.02	1.0	1
Street Dust - Urban with industrial influence (Mining and metallurgical)	Mieres, Spain	4.24			4.2	5
Street Dust - Urban with industrial influence (Iron, steel and non-ferrous)	Aviles, Spain	2.56		1.2-10.8	11	6
Street Dust - Commercial - near industrial shipping areas	Wellington, New Zealand			0.078-40	40	3

1 Chutke et al., 1995 3 Kennedy, 2003 5 Loreda et al., 2003 7 Serdar, 1993
2 Hopke et al., 1980 4 Liebens, 2001 6 Ordonez et al., 2003 8 Van Matre and Mahler, 2003

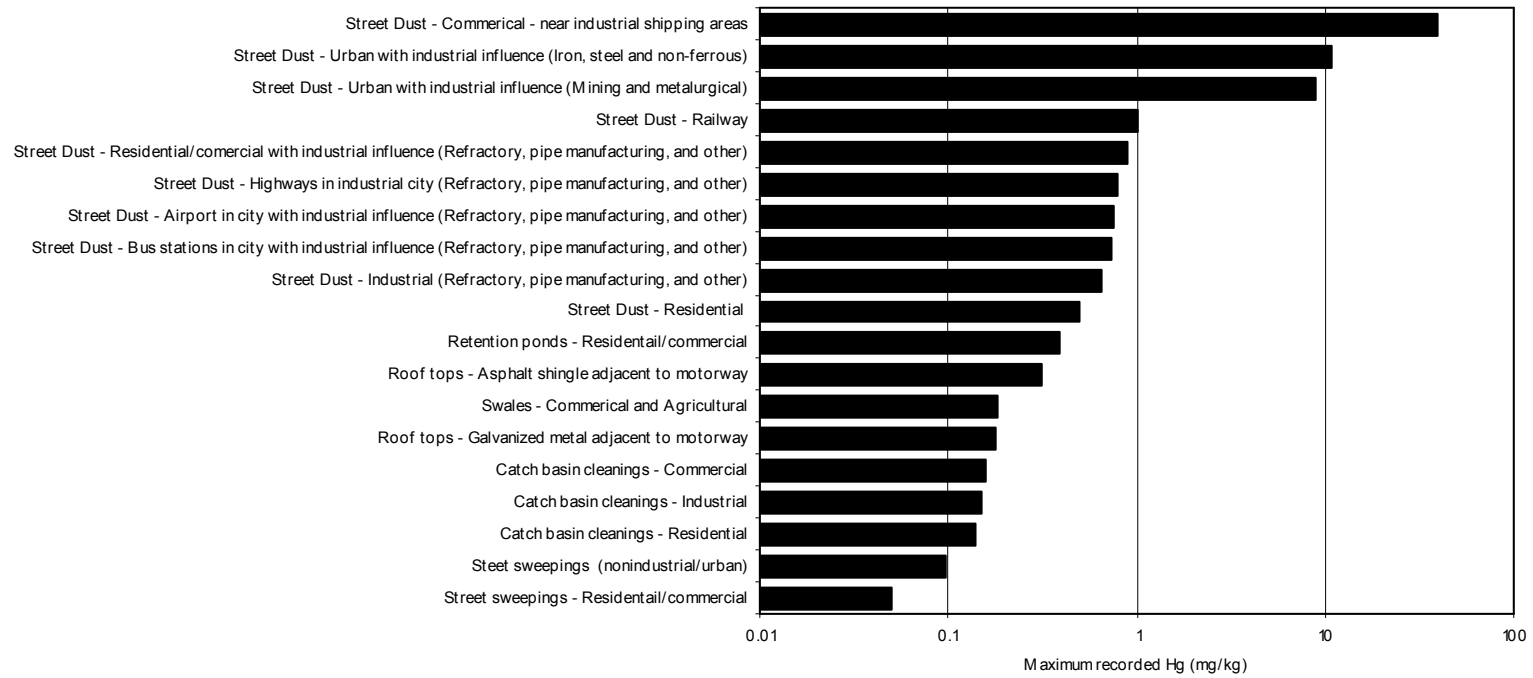


Figure 3-2. Graphical comparisons of Hg concentrations in particles in street sweepings, on roof tops, in street dust, and in selected BMPs in other parts of the world.

3.2.2.2 Variation with Grainsize and Organic Carbon

There was no information on particle size characteristics in the paper by Van Metre (2003) and we have found no information elsewhere. Organic carbon in roof samples in the study by Van Metre (2003) ranged between 7.6-16%, a surprising result (given most soils have <10% organic matter) unless there is an enrichment process that results in preferential transport of organic matter rather than inorganic matter into the atmosphere. Reworking of the data is possible from their paper (Figure 3-3). It is arguable if organic carbon is an important vector for runoff of Hg from roofs in general but there may be a weak relationship found for specific roof types such as asphalt.

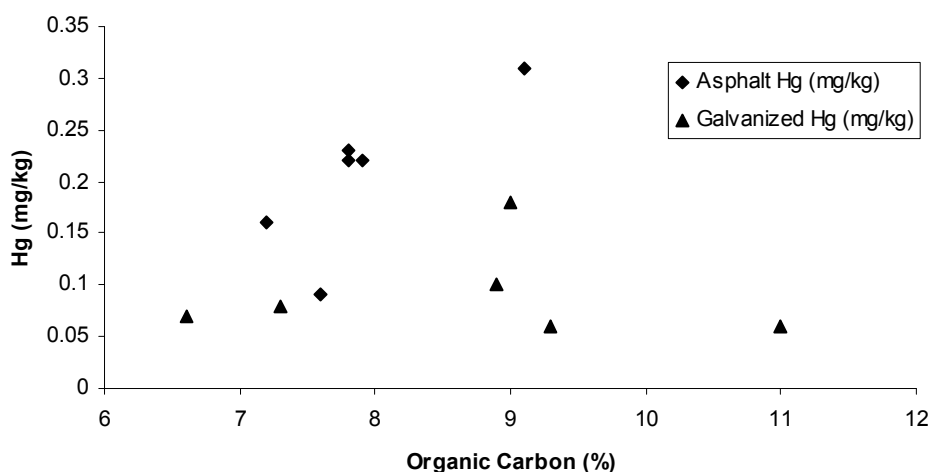


Figure 3-3. Scatter plot of % organic carbon in particulate matter washed of roof tops and Hg (after Van Metre, 2003).

3.2.3 Hg in Road and Street Dust

3.2.3.1 Sources and Bulk Concentrations

Street dust and concentrations of Hg attached to street dust can be derived from a number of sources including dry and wet fall atmospheric deposition, wear debris from cars, spillage during transport, wind blown trash, and local soils. In a literature review of trace elements in street and house dust, Fergusson and Kim (1991) concluded that the majority of street dust is derived from local soils. Even in 1991, there was extensive information available on the concentrations of a range of metals in street and house dust, but only one paper then existed on Hg in street dust and that was for a study based on analysis of street sweeping material (Hopke et al., 1980) rather than direct sampling of street dust materials (Fergusson and Kim, 1991). It is important to make this segregation

of the data because of the way street sweeping machines selectively gather materials bias towards larger particles sizes that carry lower concentrations of trace pollutants relative to fine inorganic particles and fine organics.

After extensive searching, a total of four papers were found that describe Hg concentrations in street dust based on direct sampling (Chutke et al., 1995; Kennedy, 2003; Loreda et al., 2003; Ordonez et al., 2003). These studies provide data that span a wide variety of land uses including residential, commercial, and industrial areas as well as airports, railway stations, and bus stations and a variety of road types from suburban roads to highways. Concentration in these studies ranged from 0.05-40 mg/kg (Table 3-2). In general, concentrations in street dust appear to follow a pattern of lower concentrations in residential areas with little or no industrial influence and higher in areas where industrial sources are known (Figure 3-2). A comparison of data on street dust (Table 3-1) to data on soil concentrations (Table 3-2) suggests that street dusts are likely enriched compared to soils, a phenomenon discussed by Fergusson and Kim (1991) and likely due to preferential erosion of fine materials from local soils in addition to enrichment by traffic related sources.

The term Enrichment Factor (EF) is again used for this dust : soil phenomenon (Fergusson and Kim, 1991). Again, an EF of >1 represents enrichment of dust relative to soil. They developed a histogram of EF for 44 elements using cerium as a reference non-pollution element. The EF for Hg (based on data from street sweeping material (Hopke et al., 1980) was approximately 2.6 (carefully measured from a graph in the Fergusson and Kim (1991). The EF for Hg were similar for Ni (2.6), Cr (2.7), and As (2.8) but differed substantially from Cu and Zn (11) and lead (92). In their review, Fergusson and Kim (1991) also demonstrated a strong log-log relationship ($r^2=0.81$) between concentrations found in local soils compared to street dusts when all 44 elements were groups together. This suggests it is possible to estimate concentrations in street dust for given land uses if local soil concentrations are known.

To test if there is a relationship between Hg in street dusts and Hg in immediately local soils, Hg data on concurrently collected street dusts and soils were extracted from the literature and graphed (Figure 3-4). This graph illustrates a large range in EFs for each study / land use category from <1 (depletion in street dusts relative to soils) to $>>1$ (enrichment). However, using only the best estimates from each study (using either a mean or median) a pattern of greater enrichment in street dusts in industrial areas relative to residential areas appears. It is not clear if this pattern would be validated if more studies were found or is the pattern is a random accident associated with only three pieces of literature. The pattern does seem logical and may be explained by a greater variety of sources of Hg in road dust in industrial settings relative to urban settings (e.g. higher atmospheric concentrations and deposition rates, direct spillage during haulage). It can also be explained by heavy vehicles causing greater damage to soils (generation of finer particles) on industrial sites and the wind or vehicle related transport of fine enriched particles from these sites onto the local roads. In the residential setting, sources would be fewer and vehicular damage to soils would be lesser.

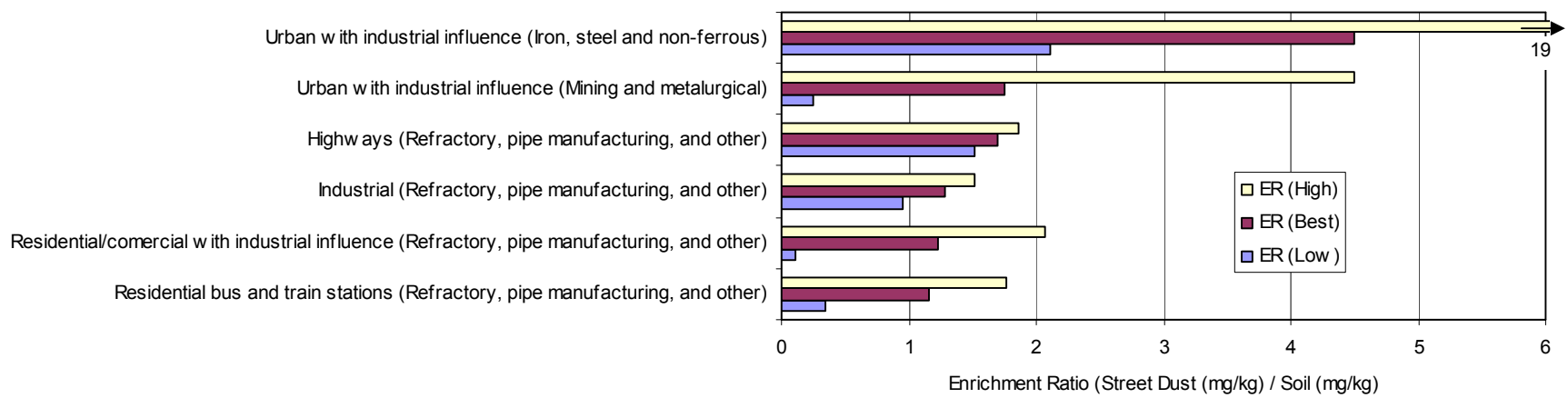
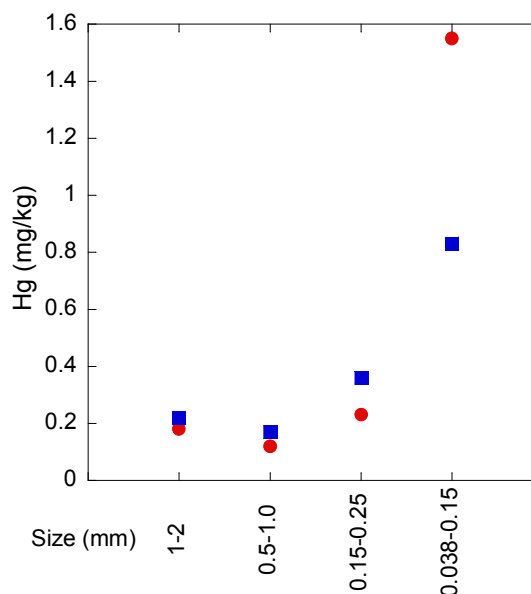


Figure 3-4. Enrichment ratios based on post calculation of soil and street dust concentrations found in Chutke et al. (1995), Loredó et al. (2003), and Ordonez et al. (2003)

3.2.3.2 Variation with Grainsize and Organic Carbon

There is only limited literature on variation of Hg with particle size and at present we have found no information on Hg in street dust relative to organic carbon. The following information was largely obtained from a literature and data review completed in New Zealand (Kennedy, 2003). Kennedy remarked upon the existence of both weak and strong relationships between Hg and grain size (see Kennedy, 2003 and references therein). It appears that in locations with higher concentrations (such as commercial areas sampled in Wellington), a strong particle size effect can be found (Figure 3-5). Kennedy (2003) citing generally the work of Pitt remarked that this was also the case for San Jose, California in the Bay Area). Samples with low Hg concentrations did not show a strong relationship with grainsize.

Figure 3-5 Concentrations found in street dust particles in relation to particle size in a commercial land use zone of Wellington, New Zealand. Locations: Lambton Quay (Red diamond) and Tory Street (Blue square) (data extracted from Kennedy, 2003).



3.2.4 Hg in Street Sweepings

3.2.4.1 Bulk Concentrations

Two papers were found during our survey of the literature describing Hg concentrations in street sweeping materials (Hopke et al., 1980; Liebens, 2001). Data spans residential and commercial land uses with little or no industrial influence. Concentrations in these studies ranged between 0.0006-0.0502 mg/kg and appear to be lower than found in catch basins, roof tops, swales, retention ponds and street dust (Table 3-2). Concentrations in street sweepings are about 5-10 times lower than in street dusts from similar types of land uses in other study areas (Table 3-2) perhaps supporting the hypothesis that street sweeping might under sample finer particles that have greater concentrations relative to

coarser particles (see previous section). In general however, there is a lack of literature on Hg in street sweepings and studies need to be done on the effectiveness of modern high efficiency street sweepers to provide conclusive evidence on Hg concentrations in street sweeping materials.

3.2.4.2 Hg Variation with Grainsize and Organic Carbon

From the limited literature available (Hopke et al., 1980; Liebens, 2001), it appears that Hg is preferentially found in street sweeping in smaller particle size fractions. Presently we are not aware of any studies of organic carbon in relation to Hg in street sweepings. Hopke et al. (1980) studied dust collected by vacuum at a roadway intersection representative of a moderately sized non-industrial urban community in Urbana, Illinois. He observed slight weighting of Hg concentrations in finer particle sizes and remarked that this could reflect the increase in surface: volume ratio of particles with decreasing size. Liebens (2001) measured particle size variations in street sweepings but did not measure Hg concentrations on separate particles sizes in the study of residential and commercial areas of Pensacola, Florida. Liebens (2001) found no statistical difference between Hg in street sweepings from residential and commercial areas and commented that the likely reason was because there were similar particles size characteristics. These two studies provide only limited insight into Hg-particle size relation in street sweepings – further work is needed to help inform decisions on the effectiveness of street sweepers to reduce urban runoff loads of Hg.

3.2.5 PCBs in Soils

The majority of information on concentrations of PCBs in urban media found in the literature is bulk near surface soils (similar to Hg). An extensive literature search was carried out without bias for depth of soil sampled (e.g. 0-2 cm, 0-5 cm etc), method of collection (e.g. mini corer, trowel), laboratory methods (e.g. instruments, detection limits, appropriately described QA procedures), and number of congeners measured (e.g. sum of 5, 9, 40 congeners). This data was used to develop a conceptual model (hypothesis) for PCB soil concentrations that might be expected in the Bay Area small tributaries and sewersheds.

3.2.5.1 Bulk Concentrations

PCBs are found in soils even far from urban and industrial areas. In our extensive review of the literature on PCB soil concentrations we found lowest concentrations (0.001 mg/kg) associated with rural areas (Table 3-3). Typical rural and agricultural concentrations in areas with little or no known urban or industrial influences appear to range from 0.001-0.67 mg/kg with a median of 0.020 mg/kg. In urban areas influenced by varying amounts of industrialization PCB concentrations in soils appear to range from

0.0013-6.8 mg/kg with a median of 0.092 mg/kg. The highest concentration was found in a study of soils at varying distances from residential buildings known to have PCBs in joint sealants. In industrial areas or agricultural or urban areas with a known high level of industrial influence, concentrations can range between 0.18-510,000 mg/kg with a median of 10.7 mg/kg. In a similar fashion to Hg, within the bounds of the description of land use and known sources, there appears to be a general increase in concentrations (rural < residential < urban with industrial sources < industrial) (Figure 3-6). PCBs differ from Hg in the degree of variation in concentrations. PCB soil concentrations span about 8 orders of magnitude whereas Hg soil concentrations only span 3 orders of magnitude. This is likely due to a combination of use characteristics and the dispersion of Hg via long range atmospheric transport.

A recent local report on the Ettie Street pump station watershed in Oakland is the only detailed local piece of work that we know of that described PCB concentrations collected in soils and sediments (Kleinfelder, 2005). They collected data from a range of sampling media including driveways, fence lines next to buildings, sidewalks, curbs, gutters, and storm grates. The data collected along fence lines is most comparable to the world literature on soils (Table 3-3). PCB concentrations along selected fence lines in the Ettie Street pump station watershed ranged from 0.13-11.1 mg/kg. These fall within the full range of concentrations seen in other watersheds around the world but compare more closely with soil concentrations in industrial watersheds.

3.2.5.2 Frequency Distribution

PCBs show a non-normal frequency distribution of concentrations in soils in a similar manner to Hg. In the two cases where authors reported both medians and means, the means were greater than the medians (Table 3-3) (Alcock et al., 1993; Bracewell et al., 1993). Priha et al. (2005) in their study of soil PCBs in the vicinity of residential buildings where sealants containing PCB were used also demonstrated a non-normal distribution for all samples taken within 3 m of buildings (medians < means). From 3-10 m distance, median and mean soils concentrations were <0.5 mg/kg and indistinguishable given the reporting accuracy. In the study of PCBs in soils in a 5 km radius around military radar installations in western Canada, Stow et al. (2005) developed a detailed understanding of frequency distribution. In this case, out of 1361 samples, only 1.2% were >50 mg/kg (max = 590 mg/kg), 8.8% were between 10-50 mg/kg, and the remaining 90% were <10 mg/kg (Stow et al., 2005). When we analyze the data from different studies in Table 3-3, the same kind of frequency distribution is observed. For example, in rural and agricultural areas with little or no known urban or industrial influences, the median described above is 0.020 mg/kg and the average for the same data is 0.034 mg/kg. For urban areas influenced by varying amounts of industrialization the median PCB concentration is 0.092 mg/kg and the average for the same data is 0.82 mg/kg. Thus the soils PCB data exhibit a non-normal distribution at the scales of individual urban systems and on-mass among systems in a similar manner to Hg. High concentrations are rare but indicators of and associated with local large point sources. Moderate to low concentrations are associated with distributed local sources and to some extent, long range atmospheric transport.

Table 3-3. PCB concentrations (mg/kg) in soils in and near cities or industrial areas from around the world. Note where no range was given by the authors, the maximum was conservatively calculated as the mean or median + 1 standard deviation.

SOILS	Location	Mean±SD	Med	Range	Maximum	Reference
Rural	Catalonia, Spain	0.000657±0.000357			0.0010	15
Agricultural and residential	Uberlandia, Brazil			0.0005-0.00125	0.0013	21
Residential with industrial influence	Catalonia, Spain	0.001965±0.001322			0.0033	15
Urban forest	Bayreuth, Germany	0.0035			0.0035	8
Urban - park area	Bayreuth, Germany	0.0055			0.0055	8
Rural	Iassy County, Romania	<0.008			0.0080	5
Residential and commercial	Maricopa County, Arizona	0.007±0.002		0.004-0.01	0.010	13
Grassland - former gasworks site	Bayreuth, Germany	0.0129			0.013	8
Rural river banks - downstream from power generation	Laja River Basin, Chile			0.0089-0.0168	0.017	3
Rural	Southern Romania	<0.020			0.020	6
Gardens	Taiwan, Vietnam, and Thailand			0.0016-0.022	0.022	18
Residential with industrial influence (Vicinity of Chemical industries)	Catalonia, Spain	0.012038±0.011650			0.023	15
Residential - influenced by industry	Catalonia, Spain	0.010342±0.016784			0.027	15
Rural and residential within 5km from airports	Croatia		0.005	<0.001-0.039	0.039	19
Urban roadside	Bayreuth, Germany	0.0919			0.092	8
Urban grassland sites with industrial influence (chemical and steel)	Linz, Austria		0.0142	0.0064-0.0950	0.10	20
Uplands	Taiwan, Vietnam, and Thailand			0.0014-0.130	0.13	18
Urban parks	Iassy County, Romania	<0.134			0.13	5
Urban parks and gardens with some industrial influence	Seine River Basin, France			0.00009-0.150	0.15	11
Urban - house gardens	Bayreuth, Germany			0.0154-0.1578	0.16	8
Industrial - Including chlorinated compounds manufacture	Southern Romania	<0.175			0.18	6
Paddy field	Taiwan, Vietnam, and Thailand			0.00061-0.320	0.32	18
Rural bogs	Scotland	0.096	0.0885	0.015-0.321	0.32	4
Industrial - Near electrical transformer stations	Croatia			0.007 - >0.400	0.40	19
Agricultural	United Kingdom	0.065	0.030	0.014-0.669	0.67	1
Urban public gardens - influenced by industry	Bahrain			0.2-0.7	0.70	2

Table 3-3 continued. PCB concentrations (mg/kg) in soils in and near cities or industrial areas from around the world. Note where no range was given by the authors, the maximum was conservatively calculated as the mean or median + 1 standard deviation.

SOILS	Location	Mean±SD	Med	Range	Maximum	Reference
Roadsides	Taiwan, Vietnam, and Thailand			0.0014-0.960	0.96	18
Industrial - Near a chemical plant	Southern Romania	<1.100			1.1	6
Industrial - Incinerator, steelworks and railway	Pontypool, South Wales			0.0146-4.620	4.6	9
Residential - influenced by industry	Bahrain			1.1-4.8	4.8	2
Agricultural - influenced by industry	Bahrain			0.4-4.9	4.9	2
Industrial (Chemical plant)	Genoa, Italy	<5.000			5.0	10
Residential (0-12 m from building with joint sealant)	Finland			0.52-6.83	6.8	14
Industrial (Including oil refineries, aluminum smelters, chemical plants, etc)	Bahrain			0.3-10.7	11	2
Industrial city urban gardens (50 industries of a variety of types)	Serpukhov, Russia			1.2-30.0	30	12
Airports	Croatia		0.533	3-41.327	41	19
Industrial near Dewline Radar Stations	Canadian Arctic	5.1		1-590	590	16
Industrial (Production of PCB mixtures)	Poland			0.6-783.3	783	17
Industrial (Transformer manufacturer)	USA			17-17800	17800	7
Industrial (Electrical component factory)	Japan	510000			510000	7

1 Alcock et al., 1993	6 Covaci et al., 2003	11 Motelay-Massei et al., 2004	16 Stow et al., 2005	21 Wilcke et al., 1999
2 Alhaddad et al., 1993	7 Erickson, 1992	12 Orlinskii et al., 2001	17 Sulkowski et al., 2003	
3 Barra et al., 2005	8 Krauss and Wilcke, 2002	13 Parker et al., 2000	18 Thao et al., 1993	
4 Bracewell et al., 1993	9 Lovett et al., 1998	14 Priha et al., 2005	19 Vasilic et al., 2004	
5 Covaci et al., 2001	10 Miniero et al., 1994	15 Schumacher et al., 2004	20 Weiss et al., 1994	

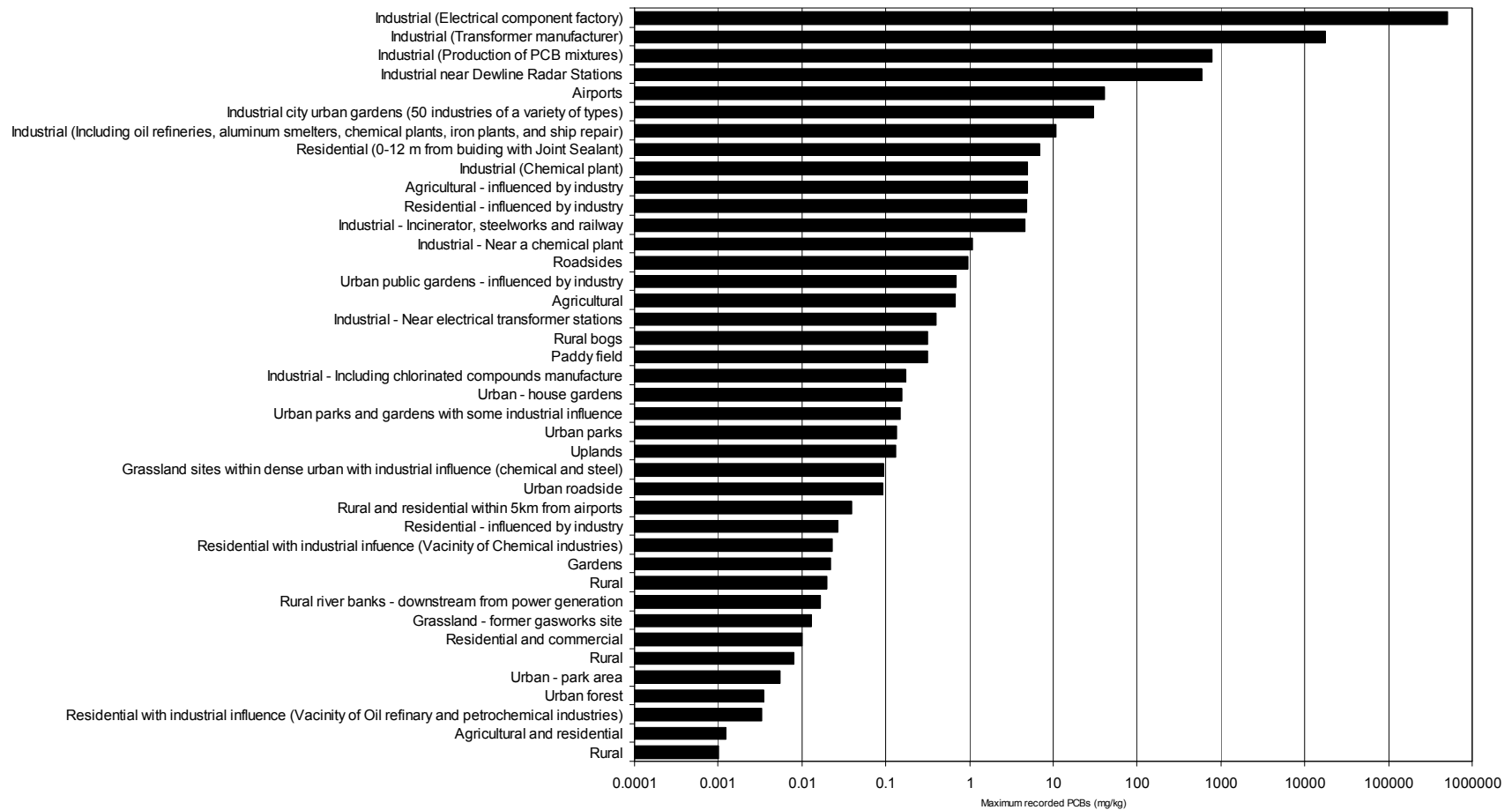


Figure 3-6. Concentrations of PCBs found in soils from studies in other parts of the world with a focus on urban and industrial areas (for references see Table 3-3).

3.2.5.3 Dispersal in Relation to Wind Direction

Gaseous and particulate PCBs can disperse from sources by advective wind or wind generated locally by traffic. The effectiveness of wind dispersal is influenced by the velocity and constancy of both velocity and direction (e.g. Irvine and Loganathan, 1998). Bracewell et al. (1993) hypothesized that the “U” shaped spatial distribution observed in Scotland was caused by the redistribution of PCB pollution through volatilization and re-deposition by prevailing WSW winds. Several studies have purposefully investigated the dispersal of PCB from point sources (Vasilic et al., 2004; Stow et al., 2005; Priha et al., 2005). In their study of PCBs in soils associated with airports in Croatia, Vasilic et al. (2004) observed PCB concentrations within airport premises ranging from 0.003-41 mg/kg (median 0.5 mg/kg). They measured soils concentrations from 0-5 km distance from the fence line ranging from <0.001-0.039 mg/kg. This suggests that movement of PCBs away from the airport was minimal even at distances of 100s m to perhaps 1000 m. Stow et al. (2005) in their study of halos around radar stations (relative to dominant wind direction) found that concentrations had diminished from a peak of 1-17 mg/kg (mean = 3.5 mg/kg) to 0.35 mg/kg within 250 m, to 0.02 mg/kg at 2 km, to 0.01 mg/kg at 3 km, and to 0.005 mg/kg at 5 km distance. Irvine and Loganathan (1998) investigated redistribution of PCBs in road dust associated with natural and traffic generated wind. They found that traffic wind was quite capable of redistributing PCBs in street dusts over distances of <7m per vehicle pass but did not state the total distance explicitly. However, they did comment that two former transformer manufacturing companies 600-900 m distant from the sampling sites showed no discernable halo effect. Several studies inadvertently show distribution of PCBs in soils over relatively short distances in urban areas that can give an idea of dispersion. Lovett et al. (1998) mapped soil concentrations in an urban area influenced by an incinerator and other industrial facilities. Interpreting concentrations based on a shaded map and a map distance scale, it is apparent that concentrations varied from >0.1 mg/kg to <0.05 mg/kg in just 100-140 m and to <0.030 mg/kg at a distance of 400 m in the dominant wind direction. This was constant with an earlier study on soil concentrations that concluded that the halo around the incinerator extended out about 200 m (references cited in Lovett et al., 1998). Lovett et al. (1998) also mapped PCB concentrations in air relative to a PCB hotspot and wind direction. They found that concentrations in air at distances of 100-200 m were greatest when wind had passed over the incinerator.

Overall, from the few studies available, it appears that strong PCB polluted soils halos only extend 100s m from point sources and perhaps most typically <300 m. The PCB dispersion picture emerging from the review of the studies available contrasts with that of mercury where the hotspots appear to be more blurred and halos appear to be larger (<3 km and perhaps 1-2 km commonly (see earlier section)).

3.2.6 PCBs in Road and Street Dust

3.2.6.1 Sources and Bulk Concentrations

Street dust and concentrations of PCBs attached to street dust can be derived from a number of sources including dry and wet fall atmospheric deposition, wear debris from cars, leakage of transmission fluids, spillage during goods transport, wind blown trash, and local soils. Our extensive search of the literature revealed just 3 papers describing PCBs in street dust (Yang and Baumann, 1996; Irvine and Loganathan, 1998; García-Alonso and Pérez-Pastor, 2003) and a recent local report on work completed so far in the Ettie Street pump station watershed (Kleinfelder, 2005). These studies however, do describe street dust from wide variety of situations within differing urban land uses: urban streets, streets near railway stations, a street near a gas station, under railway bridges in industrial areas, and on motorways all within Berlin, Germany (Yang and Baumann, 1996). Irvine and Loganathan (1998) studied PCB concentrations on streets servicing an industrial area and in two unpaved parking lots an industrial area of New York. García-Alonso and Pérez-Pastor (2003) studied a residential area of Madrid, Spain 2 km from the central city. The study area was characterized by open areas, university buildings and garden extensions where the only local source was considered to be roads and rush hour traffic. Twelve samples were taken from pavement areas next to a relatively busy road. The Ettie Street study is well known locally. Kleinfelder (2005) sampled a number of street surfaces adjacent to suspected PCB source areas in the industrial zones of the watershed.

Overall, PCB concentrations in bulk street dust in studies from other parts of the world were observed to range from 0.009-3.8 mg/kg (Table 3-4). This variation is a little over 400x and similar to Hg in street dusts (800x). Magnitude of variation is likely bias low because of a small number of studies found. In a similar manner to Hg and PCBs soils data and Hg street dust data, PCBs in street dust appear to be lower concentrations in urban areas with little or no industrial influence and be greater in urban areas with industrial influence, railways, near airports and in industrial areas (Table 3-4). The highest concentration observed from the literature we were able to amass was from a German highway, a fact that the authors speculated might be from the influence of diesel soot and tire wear debris on concentrating PCBs. Data collected recently in the Ettie Street pump station watershed (Kleinfelder, 2005) appear to fall in amongst the international data with about the same variation in magnitude. Comparing the street dust data (Table 3-4, Figure 3-7) with the soils data (Table 3-3, Figure 3-6) it is suggested that the street dust data is enriched relative to the soils data. However, unlike for Hg, none of the studies we retrieved during the literature survey contained analysis of both street dusts and local soils. Therefore, at this time we can say nothing about the magnitude or process of enrichment; no enrichment factors (EFs) can be generated.

Table 3-4. PCB concentrations (mg/kg) in street dusts from studies conducted in other parts of the world and comparisons to concentrations observed in the Ettie Street pump station watershed, Oakland, California.

	Location	Mean±SD	Median	Range	Maximum	Author
Industrial - Ettie St. Site 89	Oakland, CA	0.029			0.03	1
Industrial - Ettie St. Site 90	Oakland, CA	0.028			0.03	2
Industrial - Ettie St. Site 43	Oakland, CA	0.056			0.06	2
Urban unpaved parking lot with industrial influence	Buffalo, NY			0.053-0.063	0.06	2
Urban without industrial influence	Madrid, Spain	0.032		0.0090-0.066	0.07	3
Industrial - Ettie St. Site 17	Oakland, CA	0.098			0.10	3
Industrial - Ettie St. Site 60	Oakland, CA	0.122			0.12	3
Industrial - Ettie St. Site 117	Oakland, CA	0.160			0.16	3
Industrial - Ettie St. Site 70	Oakland, CA	0.199			0.20	3
Urban streets	Germany			0.140-0.220	0.22	3
Industrial - Ettie St. Site 92	Oakland, CA	0.220			0.22	3
Industrial - Ettie St. Site 91 (Driveway)	Oakland, CA	0.286			0.29	3
Industrial - Ettie St. Site 61	Oakland, CA	0.293			0.29	3
Urban with industrial influence	Buffalo, NY			0.090-0.310	0.31	3
Airport	Germany			0.320	0.32	3
Industrial - Ettie St. Site 71	Oakland, CA	0.582			0.58	3
Industrial - Ettie St. Site 50	Oakland, CA	0.7503			0.75	3
Industrial - Ettie St. Site 117 (Driveway)	Oakland, CA	1.220			1.22	3
Industrial - Ettie St. Site 57	Oakland, CA	1.292			1.29	3
Industrial	Buffalo, NY			0.733-1.698	1.70	3
Railway	Germany			0.200-2.000	2.00	3
Industrial - Ettie St. Site 56 (Driveway)	Oakland, CA	2.215			2.22	4
Motorways (PCBs attached to tires wear debris and diesel soot)	Germany			0.210-3.800	3.80	4
Industrial - Ettie St. Site 2 (Driveway)	Oakland, CA	3.814			3.81	4
Industrial - Ettie St. Site 64-65 (Driveway)	Oakland, CA	7.348			7.35	4

1 Garcia-Alonso and Perez-Pastor, 2003

2 Irvine and Loganathan., 1998

3 Kleinfelder, 2005

4 Yang and Baumann., 1996

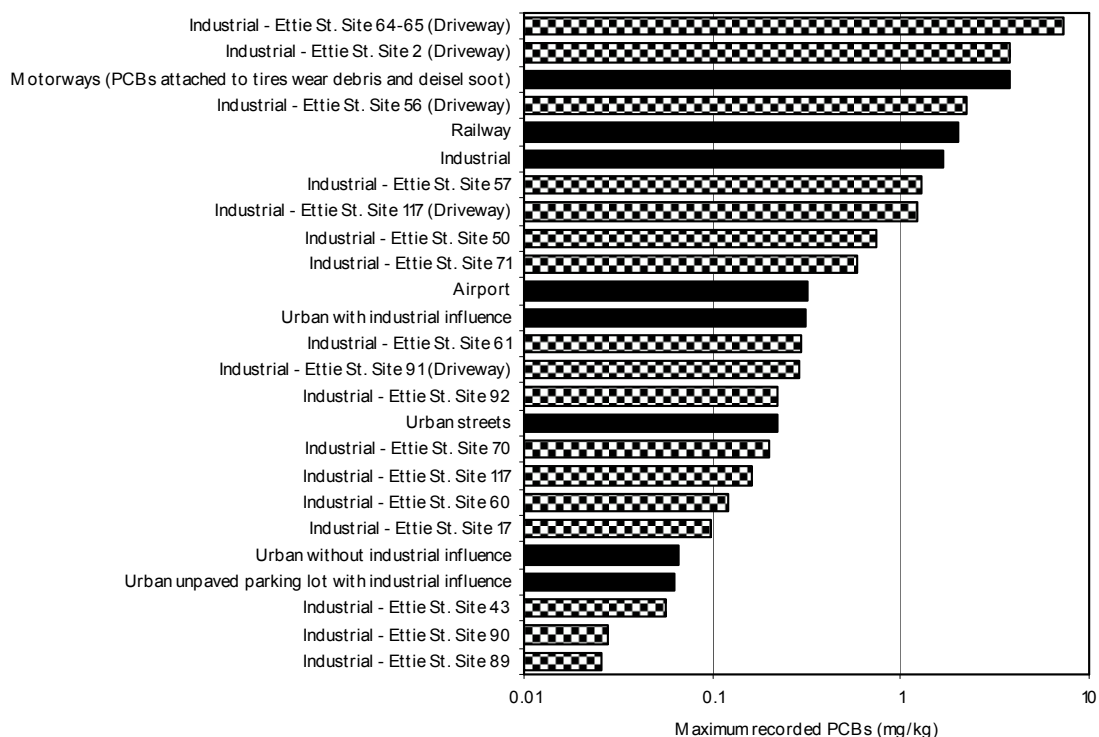


Figure 3-7. PCB concentrations in street dusts from studies conducted in other parts of the world and comparisons to concentrations observed in the Ettie Street pump station watershed, Oakland, California.

3.2.6.2 PCB Variation with Grainsize and Organic Carbon in Street Dusts

During our literature search, no papers were found that that discussed, even qualitatively, relationships between PCBs and grain size or organic carbon. This remains a large data gap in knowledge and limits the evaluation of the effectiveness of control techniques for PCBs transported from street surfaces into local drainages.

3.2.7 PCBs in Street Sweepings

During our literature search, no papers were found on PCB concentrations in street sweepings. This remains a large data gap in knowledge and limits the evaluation of the potential enhancement of street sweeping as a management technique for reducing the load of PCBs entering storm drains.

3.3 Hg and PCB Pollution Characterization in Bay Area Stormwater Conveyances

In an effort to characterize urban stormwater pollution in the Bay Area, BASMAA agencies collaborated during 2000 and 2001 to present a series of reports on the concentrations of Hg and PCBs in bed sediment collected from storm drains and creeks (Gunther et al., 2001; KLI, 2001; KLI, 2002; Salop et al., 2002). Samples were collected from open channels, catch basins, manholes, pump stations, outfalls, and detention basins. Only the top 2-3 cm of depositional areas in each sampling location was retained for analysis. All analyses of Hg and PCBs were carried out on bulk samples.

3.3.1 Hg Concentrations in Bulk Sediment

Concentrations of Hg in Bay Area stormwater conveyances ranged 2 orders of magnitude between 0.02-4.26 mg/kg (Figure 3-8) with a median of 0.16 mg/kg and a mean of 0.32 mg/kg. The frequency distribution (median < mean) of Hg in Bay Area stormwater conveyance sediments is similar to that found in soils data for both Hg and PCBs and typical of soil or water environmental pollutant data in general. When the data are sorted for land use, a pattern emerges using median concentrations (Table 3-5):

Industrial > Commercial/Residential > Mixed > Open

Thus in general, highest concentrations are found in areas closer to industry and known point sources. This is consistent with the world literature on Hg in soils and street dusts (see previous sections).

3.3.2 Hg Concentrations in Grainsize Fractions

It is important to understand the relationship between Hg and grainsize fractions at each watershed scale so that treatability can be assessed (mainly gravity based settling). Unfortunately, the work completed by BASMAA agencies (Gunther et al., 2001; KLI, 2001; KLI, 2002; Salop et al., 2002) did not include any analyses of either Hg or PCBs on separate grain sizes. The only data that we presently know of in the Bay Area for bed sediment Hg in relation to grain size has been carried out in the Guadalupe River watershed - a mining impacted watershed that flows to South San Francisco Bay.

In the Guadalupe River and some of its tributaries, bed sediment samples were collected by TetraTech Inc. and analyzed for grainsize and mercury on grainsize fractions to fulfill the needs of the Guadalupe River Hg TMDL (Austin, 2006). In addition, SFEI was funded by SCVURPPP in WY 2005 to collect bed load sediment Hg data and bed load sediment data to determine the Hg bed load at Hwy 101 (McKee et al., 2005). Although the sampling location is downstream from the main urban areas of San Jose (the largest

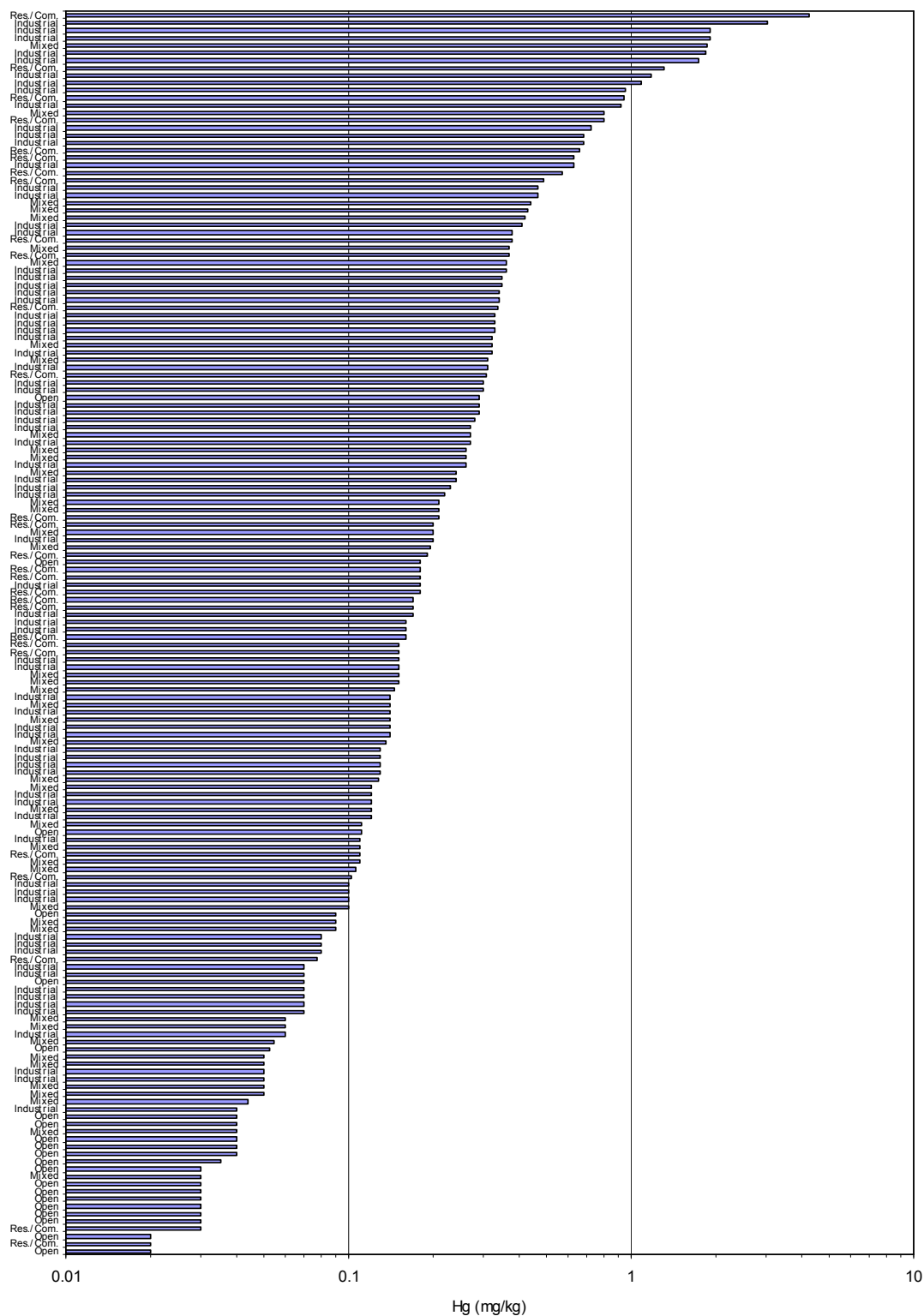


Figure 3-8. Hg concentrations measured in bed sediments from stormwater conveyances in the Bay Area. Data from Gunther et al. (2001), KLI (2001, 2002), and Salop et al. (2002).

Table 3-5. Hg and PCB concentrations in Bay Area stormwater conveyances in relation to land use designations upstream from the sampling location.

		Hg (mg/kg)	PCB (mg/kg)
Industrial	Minimum	0.040	0.0040
	Maximum	3.0	27
	Mean	0.40	0.90
	Median	0.24	0.094
Res./Com.	Minimum	0.020	0.00020
	Maximum	4.3	17
	Mean	0.48	0.77
	Median	0.20	0.063
Mixed	Minimum	0.030	0.00024
	Maximum	1.9	3.3
	Mean	0.22	0.14
	Median	0.14	0.019
Open	Minimum	0.02	0.00020
	Maximum	0.29	0.030
	Mean	0.061	0.0041
	Median	0.040	0.0011

city in the Bay Area), it represents a 414 km² watershed with extreme pollution associated with Hg mining in the historic New Almaden Mining District. Never-the-less, it does help to confirm that for large watersheds in the Bay Area, greater concentrations are found on finer particle size fractions (Figure 3-9). At this time, we have not requested the full set of TMDL Hg data from TetraTech, Inc. but we know from presentations they have made on the data, that it shows similar Hg-grainsize relationships.

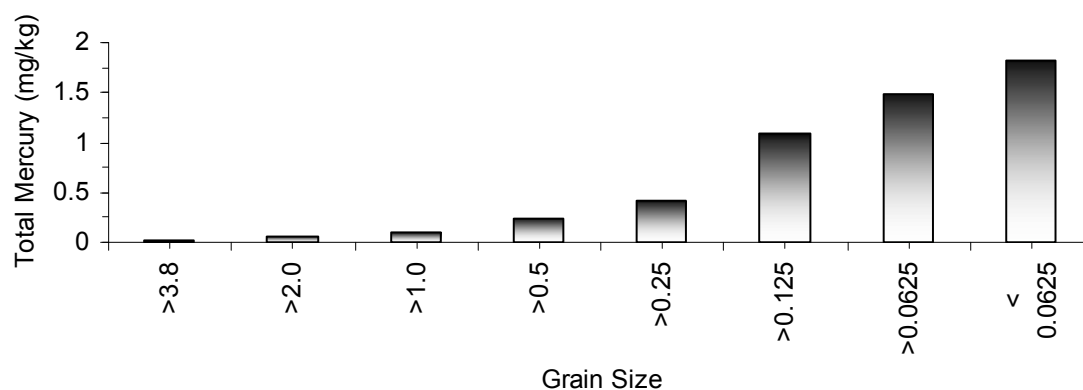


Figure 3-9. Hg concentration in bed load sediments of the Guadalupe River at Highway 101, San Jose (McKee et al., 2005).

3.3.3 PCB Concentrations in Bulk Sediment

Concentrations of PCBs in Bay Area stormwater conveyances ranged 5 orders, much greater variation than Hg and more typical of soil and street dust PCB concentrations described by the world literature. PCB concentrations varied in magnitude between 0.0002-27 mg/kg (Figure 3-10) with a median of 0.056 mg/kg and a mean of 0.59 mg/kg. The frequency distribution (median < mean) of PCBs in Bay Area stormwater conveyance sediments is similar to that found in soils data for both Hg and PCBs and typical of soil or water environmental pollutant data in general. When the data are sorted for land use, a pattern emerges that is similar to that for Hg using median concentrations (Table 3-5):

Industrial > Commercial/Residential > Mixed > Open

Only open space was statistically significant from the other land uses. However, this general pattern is consistent with the world literature on Hg in soils and street dusts (see previous sections).

3.3.4 Spatial Variation of Hg and PCB Sediments of the Bay Area

A number of programs in the Bay Area collect and analyze Hg and PCBs in sediments (Flegal et al., 1994; Hunt et al., 1998; Daum et al., 2000; Gunther et al., 2001; KLI, 2001; Heim, 2002; SFEI, 2003). Although there is much variation, greatest concentrations are found in stormwater conveyance sediment near the Bay margin for Hg (Figure 3-11) and PCBs (Figure 3-12).

3.3.5 Difficulties with Interpretation – Confounding Factors

There has been much discussion and debate on how to interpret the stormwater conveyance bed sediment data collected by the BASMAA agencies. The original authors completed a series of statistical analyses on the data (Gunther et al., 2001; KLI, 2001; KLI, 2002; Salop et al., 2002). In the case of Hg it was generally agreed that residential/commercial, industrial, and mixed land use could not be distinguished from one another and that open space was statistically significant from urban land uses (Table 3-6). In the case of PCBs, industrial and residential/commercial could not be distinguished and residential/commercial and mixed could not be distinguished but again industrial, residential/commercial, and mixed could be distinguished from open space (Table 3-6). Strong statistical differences ($P < 0.000$) were found when the data were lumped together into 2 general land use categories (urban and non-urban) (Figure 3-13). Based on the review of literature of Hg and PCBs in soils and street dusts in relation to areas of known pollution, this result should have been entirely expected.

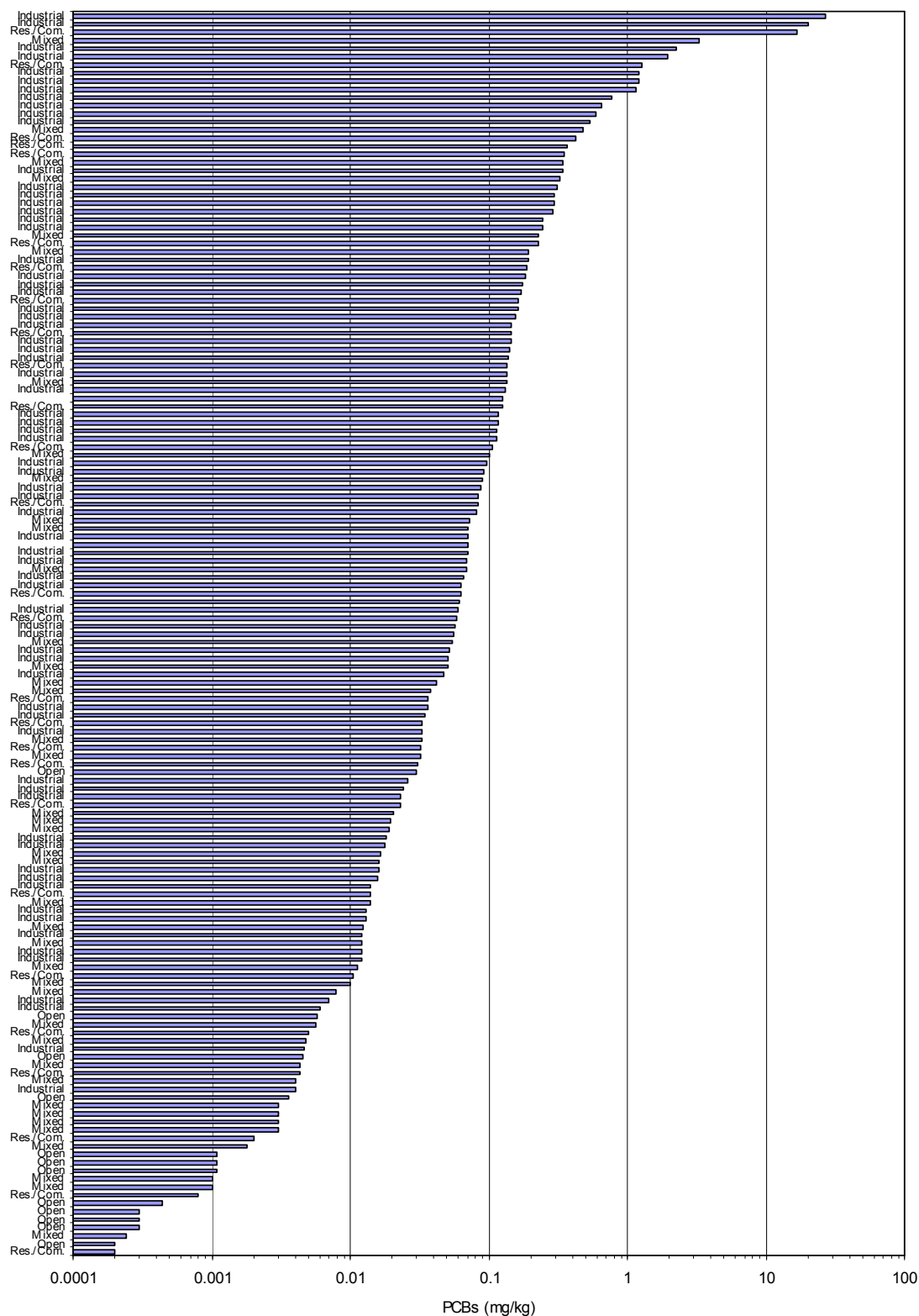


Figure 3-10. PCB concentrations measured in bed sediments from stormwater conveyances in the Bay Area. Data from Gunther et al. (2001), KLI (2001, 2002), and Salop et al. (2002).

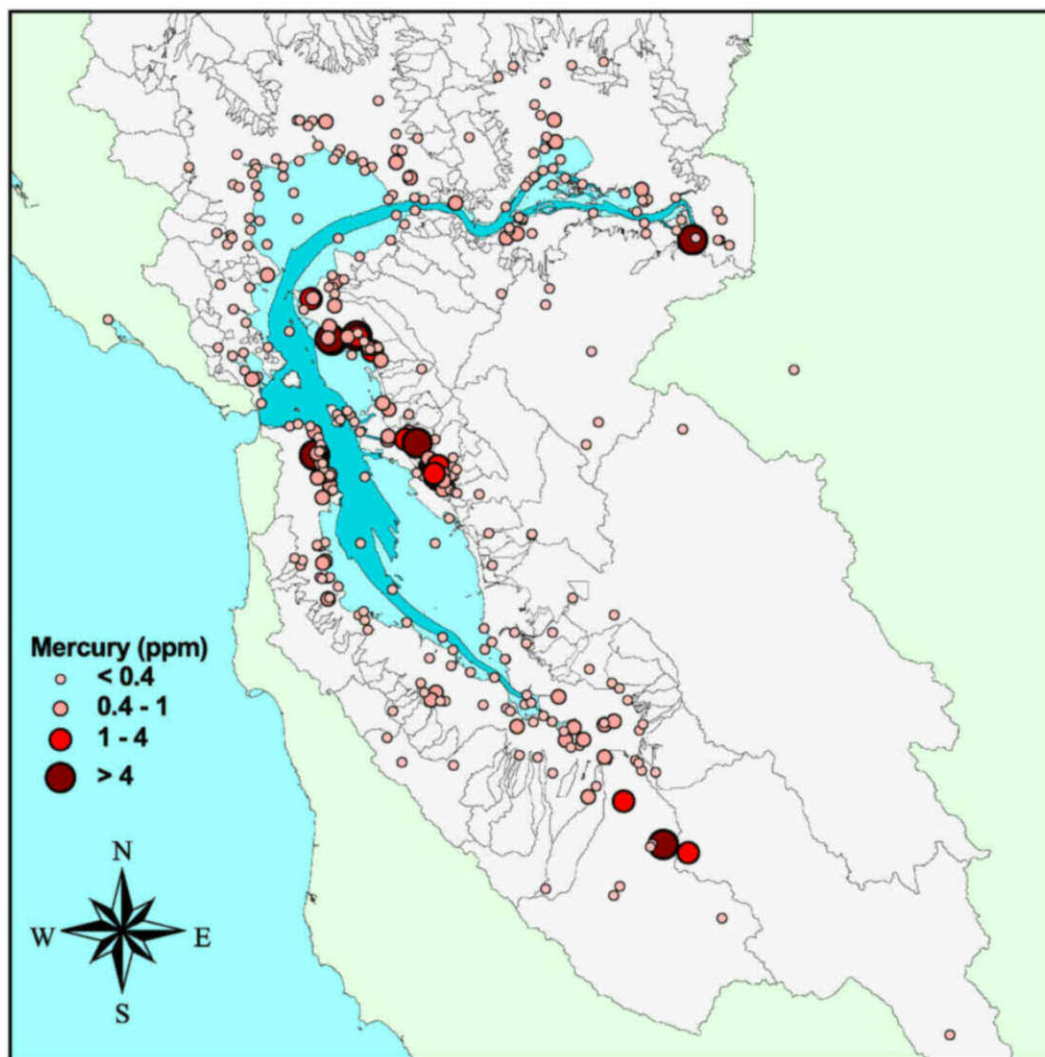


Figure 3-11. Average mercury concentrations in Bay Area sediment Data were compiled from Flegal et al. (1994), Hunt et al. (1998), Daum et al. (2000), Gunther et al. (2001), KLI (2001), and Heim (2002).

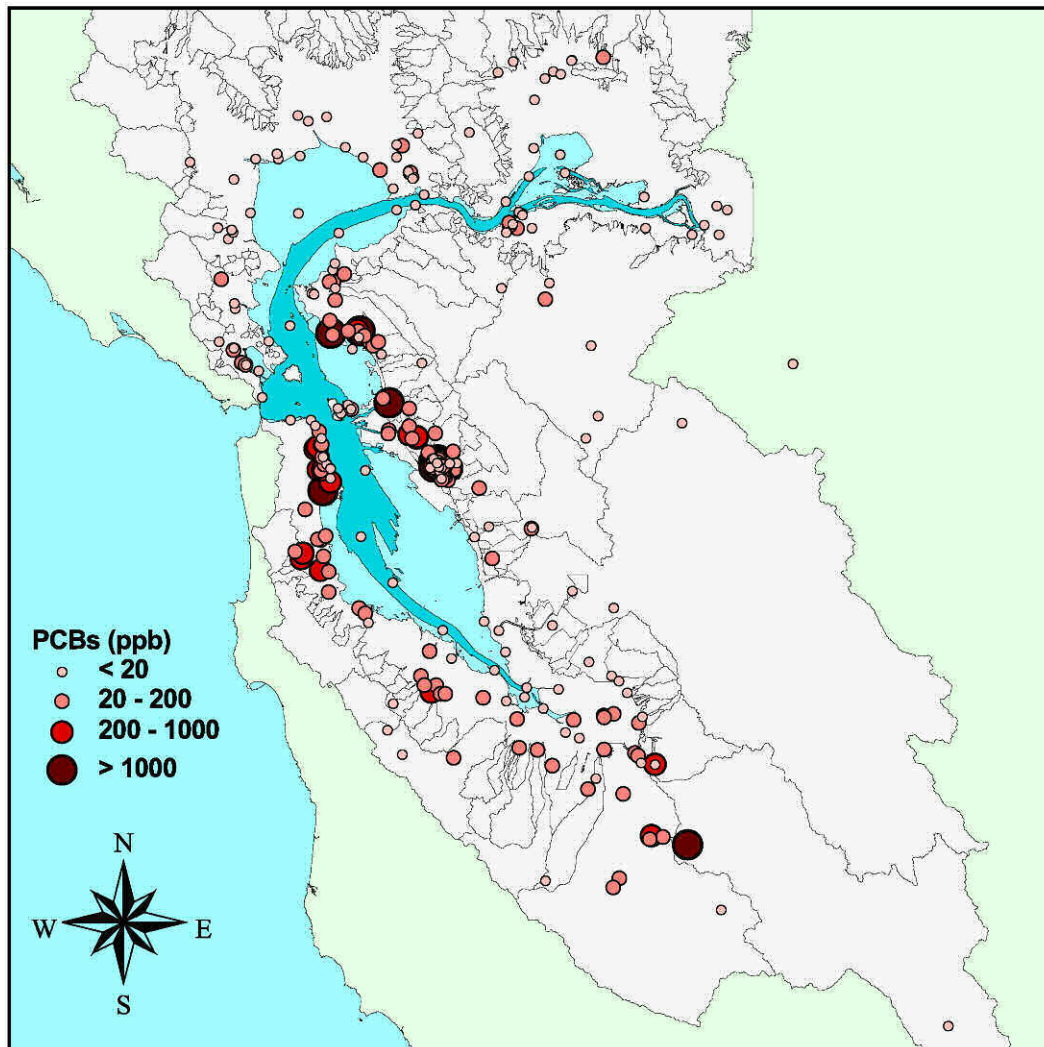


Figure 3-12. Average PCB concentrations in Bay Area sediment. Data compiled from RMP monitoring (SFEI, 2003), Hunt *et al.* (1998), Flegal *et al.* (1994), Daum *et al.* (2000), KLI (2001), and Gunther *et al.* (2001)

Table 3-6. Statistical comparisons of Hg and PCB concentrations (normalized to fine sediment (<0.0625 mm) in stormwater conveyance bed sediments based on 4 major land use categories (After KLI, 2002).

Sample Size	Probability ¹	Unadjusted Mean Value of Concentrations ² (PCB in the Case of Normalized Values)	Adjusted Mean Value of Concentrations ³ (PCB in the Case of Normalized Values)	Adjusted Mean Value of Concentrations ³ (Hg in the Case of Normalized Values)	Adjusted Mean Value of Concentrations ³ (Hg in the Case of Normalized Values)
Total PCBs ¹	<0.0001	1.000	1.000	1.000	1.000
Total Mercury	<0.0001	1.000	1.000	1.000	1.000

1. Summation of all 54 categories.
2. Significant at 0.05 level (two-tailed test) and adjusted p-value < 0.05. Significant results are indicated.
3. Unadjusted mean values are categorized into categories that are most significantly different (p < 0.05) based upon the adjusted p-value.

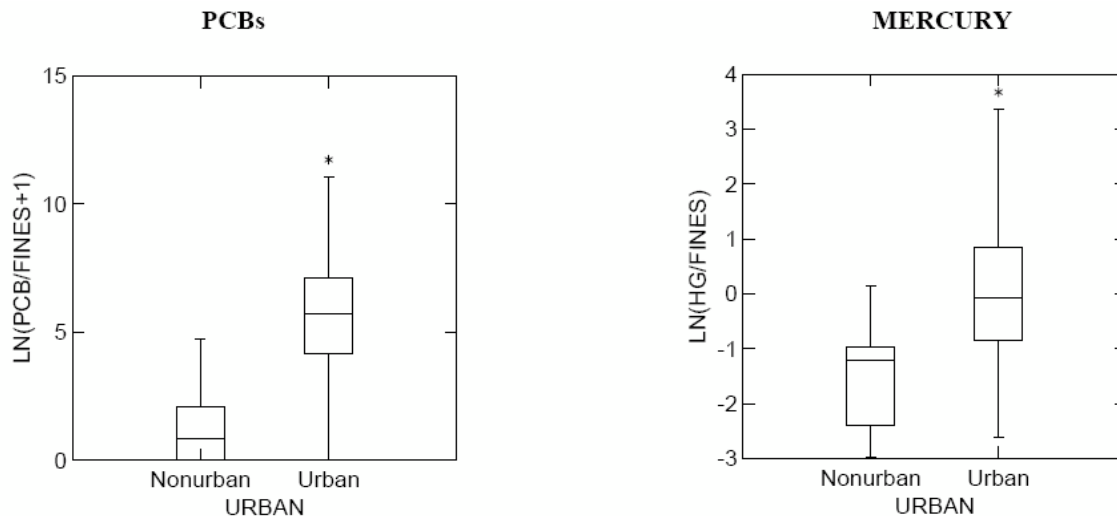


Figure 3-13. Comparison of Hg and PCB concentrations (normalized to fine sediment (<0.0625 mm) in stormwater conveyance bed sediments in urban and non-urban land uses (After KLI, 2002).

It is well known that concentrations found in aquatic sediment represent the integration of sources of sediment and pollutants from upstream landscapes. Sediment quality surveys have their origin in the mining industry where they are used track geochemical signatures of valuable mineral deposits - a useful inexpensive technique and one that is now been applied by a number of worker to try to track urban and industrial pollution (e.g. McCallum and Hall, 1998; Pettigrove, 2003). However, aquatic sediments also integrate the processes of supply and are confounded by complex patterns of erosion and deposition (McKee et al., 2003). In addition to the confounding factors discussed by McKee et al. (2003) (downstream grainsize trends, variations in sediment storage within stream or between streams, no direct relationship between suspended sediment concentrations and bed sediment concentrations, and the process of erosion and

deposition during floods relative to supply), the two most important confounding factors are upland dilution by clean sediment and variation of grainsize between samples. In addition, the lack of statistical significance could be simply due to the inherent noise in these types of data set resulting in a lack of power in the analysis. Another potential cause for the lack of statistical difference could simply be the difficulty in designating land use in highly urbanized areas that may have undergone redevelopment over time.

Dilution of polluted areas can occur when the upland area is large or when the sediment supply from the upland area is large relative to the sediment supply from the urban and industrial areas. Dilution is likely one of the reasons why there was no statistical difference found between industrial and residential/commercial areas in the BASMAA data set. In terms of study design for tracking pollution with bed sediment, this is a particular problem in the Bay Area because many of the polluted areas occur on the Bay margin mostly downstream from commercial, residential, and open space land uses which all are likely to dilute the pollutant signal by supplying cleaner sediment. In areas where very high concentrations were observed (e.g. Ettie Street pump station watershed), there is relatively little clean sediment supply from upland areas. In addition, clean sediment supply from uplands is not annually constant and there is likely varying amounts of sediment supply from bed and banks that can cause year to year variations.

The other main confounding factor with bed sediment data is the issue of variation of the pollutant with grainsize. Referring back to Figure 3-9, Hg in the Guadalupe River varies systematically with grainsize. However, the current pattern of Hg-grainsize variation is unlikely to remain constant over time as TMDL implementation changes sediment and Hg supply characteristics. In fact, Hg variation with grainsize will likely be a good indicator of how influential load reduction techniques are for that watershed. It is very problematic to compare bulk data between watersheds for the same reasons. Bulk data represents the integration of the concentrations on all the grain sizes. Comparing bulk data from one watershed to another assumes the same pollutant grainsize distribution, an assumption that is likely not true. The only way of comparing sediment pollutant data is to compare similar grainsize fractions. This is another reason why there was no statistical difference between industrial and residential/ commercial land use classes in the BASMAA data set. Ironically, the same factor might have enhanced the statistical difference between urban and non-urban land uses if the non-urban samples had a sediment size distribution biased towards coarser grain sizes (likely for upland areas).

3.4 Summary and Data Gaps

3.4.1 Summary

Hg and PCBs are found in soils and sediments even in remote areas of the world derived from long range atmospheric transport. Hg and PCB are found in higher concentrations in rural areas near urban and industrial areas, and even higher concentrations in urban and industrial areas. It appears that the halo of soil pollution around Hg hotspots extends up to

<3 km (more typically 1-2 km) whereas the halo of soil pollution around PCB hotspots extends hundreds of meters and perhaps most typically <300 m.

Within land use categories, street dusts are more enriched with Hg than the local soils, illustrating a process of enrichment likely caused mostly by a fining of grain sizes and the way wind, water, vehicle tracking, and foot tracking selectively moves fine polluted sediment from source areas onto street surfaces. Enrichment factors for Hg appear to be greater in industrial areas relative to other urban land use classes. There were no data found in the literature on the enrichment of PCBs between street dusts and adjacent soils but it is suggested that street dust would be enriched with PCBs relative to soils in a similar manner to Hg.

Concentrations in all media (soils, street dusts, roof runoff sediments, and bed sediments) display a frequency distribution that is skewed towards low concentrations – the median is always less than the mean. This is true for individual polluted areas as well both because of the use distribution characteristics at a particular location, and because of the halo of dispersion that is influenced mostly by wind but also by tracking of soil and dust particles on people shoes and on vehicles. This is illustrated by a conceptual model that shows how sampling (in this case systematic but it is true for random sampling designs as well) biases results towards low concentrations (Figure 3-14). Thus, a better measure of the pollution of a use area is the maximum concentration.

Bay Area bed sediment data collected by BASMAA agencies shows the same frequency distribution as data collected in other soils and sediment media described by the world literature. All land use classes show low concentrations but only urban and industrial land use classes show very high concentrations. The median concentrations in each land use class systematically decrease from Industrial > Commercial/Residential > Mixed > Open). This is consistent with the soils and street dust data compiled from the world literature and reflects proximity to pollutant sources. In addition, the greater statistical variation among land use classes shown for PCBs relative to Hg is constant with the relative sizes of dispersion halos and the inter-related greater magnitude of variation seen for PCBs across land use categories in all media (soils, street dusts, and sediments).

3.4.2 Data gaps

By far the most important data gap for the Bay Area is the lack of soils and street dusts characterization data. Such data will help to prioritize source areas for treatment and will help refine or make decisions about treatment methodology. Such data must include analysis of Hg and PCBs in relation to grain size. There is presently no knowledge on Hg and PCB concentrations relative to sediment density – a fact that will continue to hinder determinations of treatability in relation to BMPs that employ settling as the main treatment process. It is unclear which is better to investigate particle size or particle density. The rationale for selecting particle size or particle density needs to be developed. If particle size relationship is to be investigated, then the operational approach to measure this parameter needs to be clearly presented, and PCBs must be quantified on the particle

size isolated by the same method used to quantify particle size. If particle size distribution and PCBs are measured on fractions separated by different methods, the results may be questionable. If PCBs are presumed to be associated with organic carbon, then PCBs will likely be found in the fine fraction but again this needs to be confirmed for stormwater conveyances systems so that treatment technologies can be designed effectively.

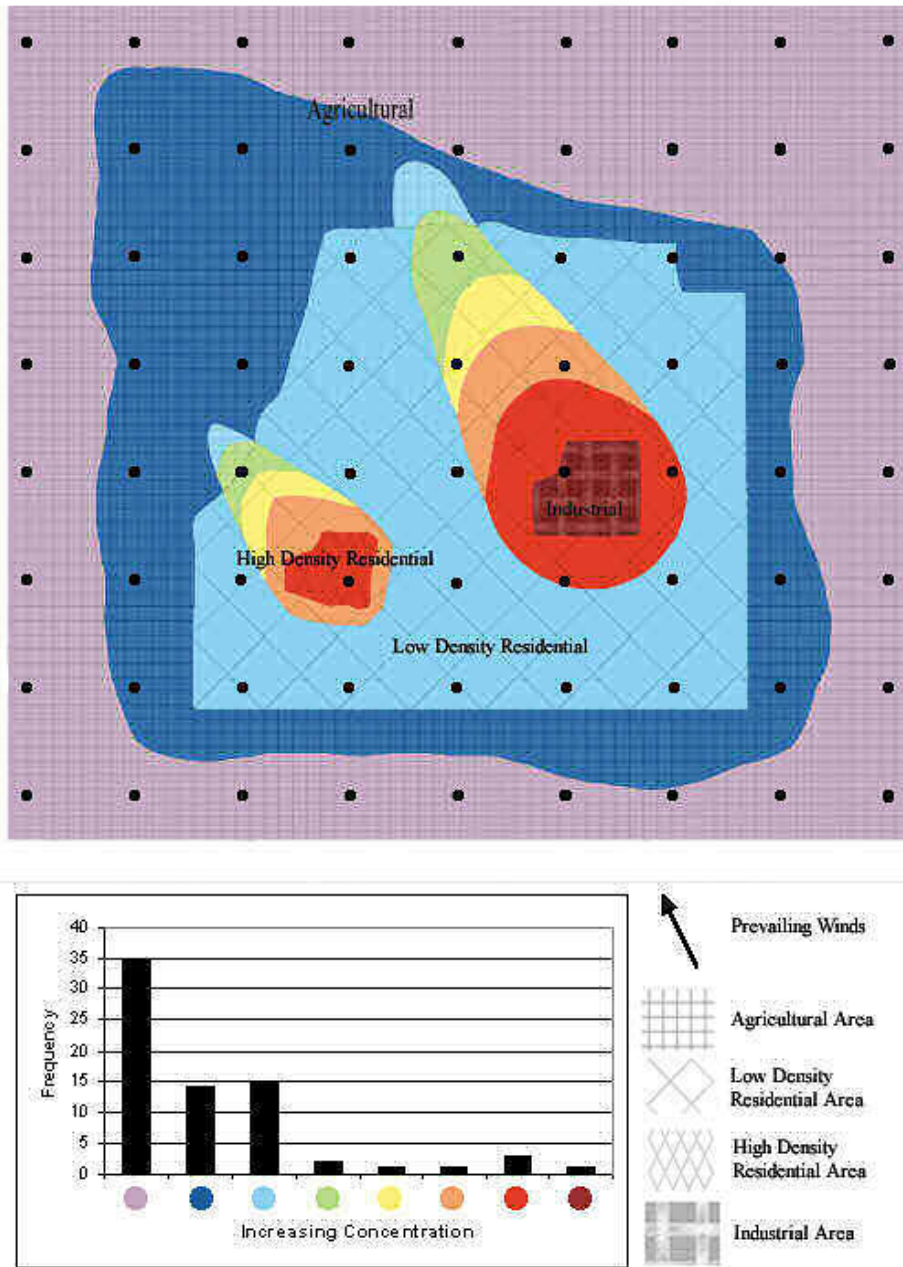


Figure 3-14. Conceptual model of industrial and urban pollution and the frequency distribution of pollution generated by a systematic sample design.

3.5 References

- Alcock, R. E., Johnston, A. E., McGrath, S. P., Berrow, M. L. and Jones, K.C. 1993. Long-term changes in the polychlorinated biphenyl content of United Kingdom soils. *Environmental Science and Technology*, 27 pp.1918-1923.
- Al-Haddad, A., Madany, I. M. and Abdullah, F. J. 1993. Levels of PCBs and PAHs in Bahrain soil. *Environment International*, 19 (3), 277-284
- Austin, C., 2006. Guadalupe River Watershed Mercury Total Maximum Daily Load (TMDL) Project Report. San Francisco Bay Regional Water Quality Control Board. Oakland, CA. January, 2006. 150 pp.
http://www.waterboards.ca.gov/sanfranciscobay/TMDL/guadalupe_river_mercury/GuadalupeMercuryProjectReport012406.pdf (cited December 2007).
- Bailey, E. H. and Everhart, D. L. 1964. Geology and quicksilver deposits of the New Almaden District Santa Clara County California. U.S. Geological Survey. Washington D.C. pp.1-206.
- Banat, K. M., Howari, F. M. and Al-Hamad, A. A. 2005. Heavy metals in urban soils of central Jordan: should we worry about their environmental risks? *Environmental Research*, 97, pp.258-273.
- Barra, R., Popp, P., Quiroz, R., Bauer, C., Cid, H. and von Tumpling, W. 2005. Persistent toxic substances in soils and waters along an altitudinal gradient in the Laja River Basin, Central Southern Chile. *Chemosphere*, 58, 905-915.
- Birke, M. and Rauch, U. 2000. Urban geochemistry: investigations in the Berlin metropolitan area. *Environmental Geochemistry and Health*, 22, pp.233-248.
- Bracewell, J. M., Hepburn, A. and Thomson, C. 1993. Levels and distribution of polychlorinated biphenyls on the Scottish land mass. *Chemosphere*, 27 (9), 1657-1667
- Bradford, G.R., Chang, A.C., Page, A.L., Bakhtar, D., Frampton, J.A., and Wright, H., 1996. Background concentrations of trace and major elements in California soils. Kearney Foundation Special Report. Kearney Foundation of Soil Science, Division of Agriculture and Natural Resources, University of California. 52pp.
- Chutke, N. L., Ambulkar, M. N. and Garg, A. N. 1995. An environmental pollution study from multielemental analysis of pedestrian dust in Nagpur city, Central India. *The Science of the Total Environment*, 164, pp.185-194.
- Covaci, A., Hura, C., and Schepens, P, 2001. Selected organochlorine pollutants in Romania. *Science of the Total Environment* 280, 143-152.
- Covaci, A., Gheorghe, A., and Schepens, P., 2003. Levels of persistent organochlorine pollutants (POPs) in soils from South Romania. *Fresenius Environmental Bulletin* 12, 94-102.
- Crockett, A. B. and Kinnison, R. R. 1979. Mercury residues in soil around a large coal-fired power plant. *Environmental Science and Technology*, 13 (6), pp.712-715.
- Daum, T., S. Lowe, R. Toia, G. Bartow, R. Fairey, J. Anderson, and J. Jones. 2000. Sediment contamination in San Leandro Bay, California. San Francisco Estuary Institute. Richmond, CA.
- Dreher, G. B. and Follmer, L. R. 2004. Mercury content of Illinois soils. *Water, Air, and Soil Pollution*, 156, pp.299-315.
- Erickson, M.D., 1992. Analytical chemistry of PCBs. CRC Press, Inc./ Lewis Publishers, Boca Raton, Florida. 508pp.

- Fergusson, J. E. and Kim, N. D. 1991. Trace elements in street and house dusts: sources and speciation. *The Science of the Total Environment*, 100, pp.125-150.
- Fitzgerald, W. F., Engstrom, D. R., Mason, R. P. and Nater, E. A. 1998. The case for atmospheric mercury contamination in remote areas. *Environmental Science and Technology*, 32, pp.1-7.
- Flegal A.R., R.W. Risebrough, B.A. Anderson, J. Hunt, S. Anderson, J. Oliver, M. Stephenson, and R. Packard. 1994. San Francisco Estuary Pilot Regional Monitoring Program: Sediment Studies. Submitted to San Francisco Bay Regional Water Quality Control Board.
- García-Alonso, S., and Pérez-Paster, R.M., 2003. Occurrence of PCBs in ambient air and surface soil in an urban site of Madrid. *Water Air, and Soil Pollution* 146, 283-295.
- Gunther, A.J., P. Salop, D. Bell, A. Feng, J. Wiegel, R. Wood. 2001. Initial characterization of PCB, mercury, and PAH concentration in the drainages of Western Alameda County, CA. Produced for the Alameda Countywide Clean Water Program.
- Heim, W.A., K. Coale, and M. Stephenson. 2002. Draft Report. Assessment of ecological and human health impacts of mercury in the Bay-Delta watershed. CALFED Bay-Delta Mercury Project.
- Hopke, P. K., Lamb, R. E. and Natusch, D. F. S. 1980. Multielemental characteristics of urban roadway dust. *Environmental Science and Technology*, 14 (2), pp.164-172.
- Horvat, M., Nolde, N., Fajon, B., Jereb, V., Logar, M., Lojen, S., Jacimovic, R., Falnoga, I., Liya, Q., Fagneli, J. and Drobne, D. 2003. Total mercury, methylmercury, and selenium in mercury polluted areas in the province Guizhou, China. *The Science of the Total Environment*, 301, pp.231-256.
- Hunt, J.W., B.S. Anderson, B.M. Phillips, J. Newman, R.S. Tjeerdema, K. Taberski, C.J. Wilson, M. Stephenson, H.M. Puckett, R. Fairey, and J. Oakden. 1998a. Bay Protection and Toxic Cleanup Program Final Technical Report: Sediment quality and biological effects in San Francisco Bay. California State Water Resources Control Board. Sacramento, CA.
- Irvine and Loganathan. 1998, *Water, Air and Soil Pollution*, 105, 603-615.
- Johansson, K., A. Andersson, and T. Andersson 1995. Regional accumulation pattern of heavy metals in lake sediments and forest soils in Sweden. *The Science of the Total Environment*, 160/161, pp. 373-380.
- Kennedy, P. 2003. Metals in Particulate Material on Road Surfaces. Report for Ministry of Transport, New Zealand, 99pp.
http://www.transport.govt.nz/research/Documents/stormwater_wc_contaminant_loadings.pdf
- Klein, D. H. 1972. Mercury and other metals in urban soil. *Environmental Science and Technology*, 6, pp.560-562.
- Kleinfelder, Inc. 2005. Sediment sampling report Ettie Street pump station watershed, Oakland, California. City of Oakland PWA – ESD. July 2005. pp.1-31.
- KLI, 2001. Joint stormwater agency project to study urban sources of mercury and PCBs. Report prepared by Kinnetic Laboratories, Inc. for Santa Clara Valley Urban Runoff Pollution Prevention Program, Contra Costa Clean Water Program, San Mateo Countywide Stormwater Pollution Prevention Program, Marin County Stormwater Pollution Prevention Program, Vallejo Flood Control and Sanitation District, Fairfield-Suisun Sewer District. 44pp + appendices.

- KLI, 2002. Joint stormwater agency project to study urban sources of mercury, PCBs, and organochlorine pesticides. Report prepared by Kinnetic Laboratories, Inc. for Santa Clara Valley Urban Runoff Pollution Prevention Program, Contra Costa Clean Water Program, San Mateo Countywide Stormwater Pollution Prevention Program, Marin County Stormwater Pollution Prevention Program, Vallejo Flood Control and Sanitation District, Fairfield-Suisun Sewer District. 71pp.
- Kohut, C., Dudas, M.J., and Nason, G.E., 2000. Distribution of mercury in soils at a decommissioned gas plant. *Canadian Journal of Soil Science* 80, 473-482.
- Kot, F. S. and Matyushkina, L. A. 2002. Distribution of mercury in chemical fractions of contaminated urban soils of Middle Amur, Russia. *Journal of Environmental Monitoring*, 4, pp.803-808.
- Koval, P. V., Kalmychkov, G. V., Gelety, V. F., Leonova, G. A., Medvedev, V. I. and Andrulaitis, L. D. 1999. Correlation of natural and technogenic mercury sources in the Baikal polygon, Russia. *Journal of Geochemical Exploration*, 66, pp.277-289.
- Krauss and Wilcke, 2002, *Soil Sci. Soc. Am. J.*, 66, 430-437
- Liebens, J. 2001. Heavy metal contamination of sediments in stormwater management systems: the effect of land use, particle size, and age. *Environmental Geology*, 41, p.341-351.
- Lin, Y-P., Teng, T-P. and Cheng, T-K. 2002. Multivariate analysis of soil heavy metal pollution and landscape pattern in Changhua county in Taiwan. *Landscape and Urban Planning*, 62, pp.19-35.
- Loredó, J., Pereira, A., and Ordóñez, A., 2003. Untreated abandoned mercury mining works in a scenic area of Asturias (Spain). *Environment International* 29, 481-491.
- Loska, K., Wiechula, D. and Korus, I. 2004. Metal contamination of farming soils affected by industry. *Environment International*, 30, pp.159-165.
- Lovett, A. A., Foxall, C. D., Ball, D. J. and Creaser, C. S. 1998. The Panteg monitoring project: comparing PCB and dioxin concentrations in the vicinity of industrial facilities. *Journal of Hazardous Materials*. 61, pp.175-185.
- Malawska, M. and Wilkomirski, B. 2001. An analysis of soil and plant (*Taraxacum officinale*) contamination with heavy metals and polycyclic aromatic hydrocarbons (PAHs) in the area of the railway junction Ilawa Główna, Poland. *Water, Air, and Soil Pollution*, 137, pp.339-349.
- Manta, D. S., Angelone, M., Bellanca, A., Neri, R. and Sprovieri, M. 2002. Heavy metals in urban soils: a case study from the city of Palermo (Sicily), Italy. *The Science of the Total Environment*, 300, pp.229-243.
- McCallum, D. W. and Hall, K. J. 1998. Limitations of sediment quality surveys: a case study of an urban watershed in British Columbia, Canada. *Water Science and Technology*, 38 (11), pp.201-208.
- McKee, L., Leatherbarrow, J., Pearce, S., and Davis, J., 2003. A review of urban runoff processes in the Bay Area: Existing knowledge, conceptual models, and monitoring recommendations. A report prepared for the Sources, Pathways and Loading Workgroup of the Regional Monitoring Program for Trace Substances. SFEI Contribution 66. San Francisco Estuary Institute, Oakland, Ca.
- McKee, L., Oram, J., Leatherbarrow, J., Bonnema, A., Heim, W., and Stephenson, M., 2005. Concentrations and loads of mercury, PCBs, and PBDEs in the lower Guadalupe River, San Jose, California: Water Years 2003, 2004, and 2005. A Technical Report of

- the Regional Watershed Program: SFEI Contribution 424. San Francisco Estuary Institute, Oakland, CA. 47pp.
- Miniero et al., 1994. PCDDs, PCDFs, and PCBs in the soil of an industrial site. 2. Preliminary assessment. *Fresenius Environmental Bulletin*, 3 (6), 339-344
- Motelay-Massei, A., Ollivon, D., Garban, B., Teil, M. J., Blanchard, M. and Chevreuil, M. 2004. Distribution and spatial trends of PAHs and PCBs in soils in the Seine River basin, France. *Chemosphere*, 55 (4), 555-565
- Munch, D. 1993. Concentration profiles of arsenic, cadmium, chromium, copper, lead, mercury, nickel, zinc, vanadium and polynuclear aromatic hydrocarbons (PAH) in forest soil beside an urban road. *The Science of The Total Environment*, 38 (1-3), pp.47.
- Murray, K. S., Rogers, D. T. and Kaufman, M. M. 2004. Heavy metals in an urban watershed in Southeastern Michigan. *Journal of Environmental Quality*, 33, pp.163-172.
- Nadal, M., Schuhmacher, M. and Domingo, J. L. 2004. Metal pollution of soils and vegetation in an area with petrochemical industry. *Science of the Total Environment*, 321, pp.59-69.
- Navarro, M., Lopez, H., Sanchez, M. and Lopez, M. C. 1993. The effect of industrial pollution on mercury levels in water, soil, and sludge in the coastal area of Motril, Southeast Spain. *Archives of Environmental Contamination and Toxicology*, 24, pp.11-15.
- Ordonez, A., Loreda, J., Miguel, E. D. and Charlesworth, S. 2003. Distribution of heavy metals in the street dusts and soils of an industrial city in Northern Spain. *Archives of Environmental Contamination and Toxicology*, 44, pp.160-170.
- Orlinskii, D., Pripitina, I., Popova, A., Shalanda, A., Tsongas, T., Hinman, G. and Butcher, W. 2001. Influence of environmental contamination with PCBs on human health. *Environmental Geochemistry and Health*, 23, 317-332.
- Panov et al., 1999. On pollution of the biosphere in industrial areas: the example of the Donets coal Basin. *International Journal of Coal Geology* 40, 199-210.
- Parker, J. T. C., Fossum, K. D. and Ingersoll, T. L. 2000. Chemical characteristics of urban stormwater sediments and implications for environmental management, Maricopa County, Arizona. *Environmental Management*, 26 (1), 99-115.
- Peltola, P. and Astrom, M. 2003. Urban geochemistry: a multimedia and multielement survey of a small town in Northern Europe. *Environmental Geochemistry and Health*, 25, pp.297-419.
- Pettigrove, V. and Hoffmann, A. 2003. Impact of urbanisation on heavy metal contamination in urban stream sediments: influence of catchment geology. *Australasian Journal of Ecotoxicology*, 9, pp.119-128.
- Priha, E., Hellman, S. and Sorvari, J. 2005. PCB contamination from polysulphie sealants in resident areas-exposure and risk assessment. *Chemosphere*, 59, 537-543.
- Reimann, C., Boyd, R., de Caritat, P., Halleraker, J. H., Kashulina, G., Niskavaara, H. and Bogatryrev, I. 1997. Topsoil (0-5 cm) composition in eight arctic catchments in Northern Europe (Finland, Norway and Russia). *Environmental Pollution*, 95 (1), pp.45-56.

- Salop, P., Abu-Saba, K., Gunther, A., and Feng, A., 2002. 2000-01 Alameda County watershed sediment sampling program: Two-year summary and analysis. Report prepared for the Alameda Countywide Clean Water Program. September, 2002. 33pp.
- Schuhmacher, M., Nadal, M. and Domingo, J. L. 2004. Levels of PCDD/Fs, PCBs, and PCNs in soils and vegetation in an area with chemical and petrochemical industries. *Environmental Science and Technology*, 38, (7) 1960-1969.
- Serdar, D. 1993. Contaminants in vector truck wastes. Washington State Department of Ecology. Olympia, WA. 40p.
- SFEI. 2003. RMP Annual Results. San Francisco Estuary Institute. Oakland, CA. http://www.sfei.org/rmp/2001/RMP_2001_water.pdf
- Stow, J. P., Sova, J. and Reimer, K. J. 2005. The relative influence of distant and local (DEW-line) PCB sources in the Canadian Arctic. *Science of the Total Environment*, 342, 107-118.
- Strnad et al., 1994. Pollution of heavy metal and polycyclic aromatic-hydrocarbons in the town of Brno (Czech-Republic). *International Journal of Environmental Analytical Chemistry* 54, 233-248.
- Sulkowski et al., 2003, *Fresenius Environmental Bulletin*, 12 (2), 158-164
- Thao, V. D., Kawano, M. and Tatsukawa, R. 1993. Persistent organochlorine residues in soils from tropical and sub-tropical Asian countries. *Environmental Pollution*, 81 (1), 61-71.
- Tijhuis, L., Brattli, B. and Saether, O. M. 2002. A geochemical survey of topsoil in the city of Oslo, Norway. *Environmental Geochemistry and Health*, 24, pp.67-94.
- Van Metre, P. C. and Mahler, B. J. 2003. The contribution of particles washed from rooftops to contaminant loading to urban streams. *Chemosphere*, 52 (10), pp.1727-1741.
- Vasilic et al., 2004, *Intl. J. of Environmental Analytical Chemistry*, 84 (14-15), 1093-1103
- Wang, X. S., Qin, Y. and Sang, S. X. 2005. Accumulation and sources of heavy metals in urban topsoils: a case study from the city of Xuzhou, China. *Environmental Geology*, 48, pp.101-107.
- Weiss, P., Riss, A., Gschmeidler, E. and Schentz, H. 1994. Investigation of heavy metal, PAH, PCB patterns and PCDD/F profiles of soil samples from an industrialized urban area (Linz, Upper Austria) with multivariate statistical methods. *Chemosphere*, 29 (9-11), 2223-2236.
- Wiersma et al., 1986. Cadmium, lead, mercury, and arsenic concentrations in crops and corresponding soils in the Netherlands. *Journal Agric. Food Chemistry* 34, 1067-1074.
- Wilcke, W., Lilienfein, J., Lima, S. D. and Zech, W. 1999. Contamination of highly weathered urban soils in Uberlandia, Brazil. *J. of Plant Nutrition and soil Science*, 162, 539-548.
- Yang and Baumann. 1996, *Fresenius' J. of Analytical Chemistry*, 354 (1), 56-60.

4. Transport of Hg and PCBs in Stormwater

4.1 Introduction

Past reviews of other systems have established that Hg and PCBs are mainly transported in the particulate phase (Johnson and Looker, 2003, McKee et al. 2003, Hetzel, 2004). With the paucity of data on Hg and PCB transport in urban stormwater systems, this section of the white paper starts with that premise and seeks to provide a robust overview of the transport and treatability of particulate matter in the urban stormwater drainage network. The reason to provide this overview is to understand the nature of Hg and PCB transport in the stormwater system in order to select the appropriate unit processes to address their attenuation or removal. The overall conceptual model of the current pollution of the Bay is summarized in Figure 4-1.

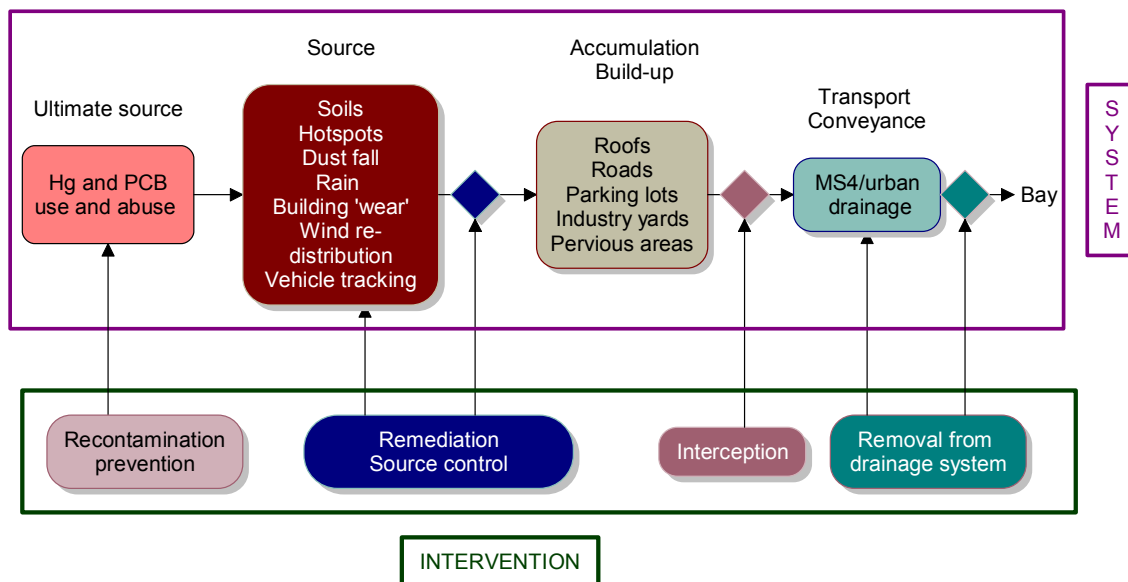


Figure 4-1. Overall conceptual model of system.

The most widely held perception of Hg and PCB transport to the Bay involves the following:

1. Most Hg and PCB are transported by particles in suspension.
2. Most particles in stormwater runoff ($\approx 90\%$) are suspended and settleable (usually operationally defined as $< 62.5 \mu\text{m}$).
3. Hg and PCB are mostly associated with particulate organic carbon (POC).
4. Large particles ($>62.5 \mu\text{m}$) are in bed load and do not transport significant Hg and PCB. They move slowly through the drainage network and are readily trapped in catchbasins, drains etc. This bed load comprises $\sim <10\%$ of the total sediment load in stormwater runoff.

This section critically examines these concepts in order to confirm or develop a new conceptual model on the transport and treatability of Hg and PCBs.

4.2 Processes of Mobilization

4.2.1 Rainfall, runoff, erosion

Rainfall is polluted with Hg, and to a lesser extent, PCBs (see Section 2 for detailed discussion). In addition, runoff generated by rain falling on impervious surfaces mobilizes dissolved and particulate matter accumulated there. Large rainstorms also generate runoff on pervious areas which can erode and entrain sediment and supply pollutants and sediments to stormwater as well as replenish pollutants on the impervious surfaces. Sediments (as thoroughly discussed in Section 3) are commonly polluted with Hg and PCBs.

Sediment transported from urban areas is derived from a number of sources within the urban environment. Land surface sources include building and roadway construction sites where there is ineffective mitigation to prevent sediment erosion, roadway median strips and edges where there is often limited or no vegetation cover, shaded areas (e.g., on the lee side of industrial buildings or under bridges) where vegetation does not grow, and freshly tilled and sprayed (herbicide) landscape areas and industrial yards where heavy vehicles may damage the structure of the soil making it more susceptible to erosion. Dust and dirt particles on impervious surfaces (e.g., roads, driveways, parking lots and other paved areas, and roofs of houses, commercial buildings, factories and warehouses) derived from the vehicle wear, building wear, road abrasion, vehicle tracking (see below) and wind blown loess (see below) will also be transported easily during rain events. Instream sources include failing banks and revetments (perhaps associated with increased peak discharge or water velocity in channels), bed erosion, and illegal dumping of inorganic or organic waste off bridges and in the near stream environment. Although some of this sediment may be unpolluted by Hg and PCBs (e.g., subsoils eroded during earthworking operations or just remote from pollutant sources) and act as a dilution for Hg and PCB pollution usually in an urban environment, this sediment is polluted to a moderate to large degree with Hg or PCBs (see Section 3 for a thorough discussion) and thus be a source of Hg and PCB in runoff. Major areas of concern are:

- Polluted industrial yards
- Soils or dirt accumulated on impervious surfaces near buildings receiving fallout and washoff of paints and other building materials containing PCBs (e.g., caulking) and Hg

4.2.2 Vehicle tracking of pollutants on tires

Vehicle tracking may be an important mechanism by which these polluted soils can be transported to impervious surfaces where they can be easily washed off. Tires on heavy vehicles can transport polluted sediment from industrial yards either directly by picking up soils in its tread or indirectly through dirt splatters on mudguards and bodywork. This material can be transferred to roadways and gutters after this material is dislodged by higher speeds, abrasion, road-water splash, shaking, drying and wind turbulence. The deposited material can be re-

dispersed by the wind turbulence to other impervious surfaces (pavements, roofs), vegetation, or pervious surfaces.

4.2.3 Dust resuspension and deposition

Dust resuspension and deposition is another major vector transferring pollutants from source to impervious surfaces. Dry soils on industrial yards can become suspended by wind – either natural or vehicle induced – and dispersed to other areas. Vehicle tracking can exacerbate this process by damaging the cohesive structure of soils in industrial yards. This dust can deposit on adjacent impervious surfaces. Dust resuspension and deposition is also a major mechanism limiting the build up of dirt on roads. Traffic- and wind-induced turbulence limits the build-up of dirt and pollutants on roadways. Wind may limit build up on other impervious surfaces remote from traffic (such as roofs). The build-up and resuspension limiting processes are described more fully in Section 5.

4.3 Transport processes (deposition and transformations in the drainage network)

4.3.1. Suspended sediment in urban drainage

4.3.1.1 Generic concentration variability

The observed concentrations of TSS found in other studies in the USA are summarized in Figure 4-2. Median concentrations are mostly on the order of 20-100 mg/L, although there is a great deal of variability.

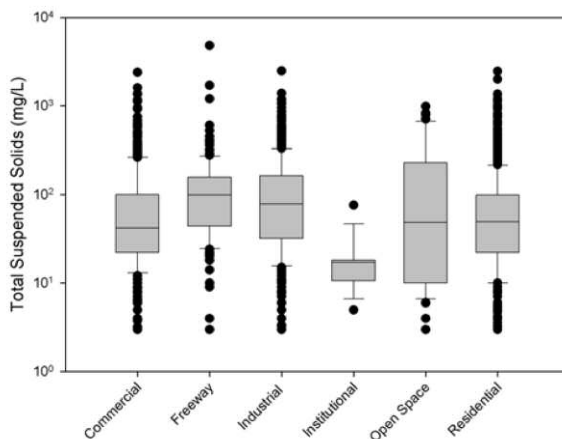


Figure 4-2. Typical urban stormwater (medians - USA nationwide). From Pitt et al., 2004.

4.3.1.2. San Francisco Bay Watersheds

TSS has been measured in urban San Francisco Bay watersheds in a number of different studies (Table 4-1). The concentration and loads of SSC have been measured in a wide range of Bay watersheds by the USGS. These tend to be upland or large-scale watersheds. The flow-weighted mean concentrations (Table 4-2) are much higher than the EMCs measured in the urban watersheds. This may reflect the more accurate measure of SS (SSC versus TSS), higher sediment loads from the upland (steeper, rural) watersheds, and/or more comprehensive databases (measurements taken over a longer period including more intense storms).

Table 4-1. Suspended sediment EMCs for urban land use in San Francisco Bay urban watersheds (mg/L). From Davis et al. (2000).

Land Use	Alameda	Santa Clara	BASMAA	SCCWRP
Residential	192	76	90	102
Commercial	192	76	98	118
Industrial	114	152	113-157	174
Transportation	192	-	-	-
Open	11	85	-	371
Agricultural	-	-	-	2068

Table 4-2. Flow-weighted-means of SSC for Bay watersheds. (Summarized from McKee et al., 2003).

Data	Concentrations (mg/L)
USGS data (Table 3.4, p57, McKee et al., 2003)	535-953
Calculated average concentrations from annual sediment loads and runoff volumes to the Bay	400-750

There are major differences between SS loads and concentrations from Bay watersheds. This may be due to different topography/geology/land use, e.g., lowland, built-out urban versus upland rural, containing some urban development (earthworking). The data suggests potentially high SS loads from Bay watersheds, in contrast with commonly accepted loads from fully built-out urban areas. However, the difference may also be due in part to measurement methodology. High sediment loads associated with upland or headwater catchments has some important implications:

1. High sediment loads are likely to be associated with relatively low pollutant concentrations (except in areas of Hg mining) that may dilute pollution in downstream urban sediments
2. High SS loads from headwater catchments may help removal of Hg and PCBs in stormwater drainage networks. High loads of relatively unpolluted suspended sediments are carried in the high gradient headwater portions of the stream network. Polluted sediments presumably mainly input from older urban areas on the lower gradient reaches of the streams on the flood plains and reclaimed land. If the carrying capacity is exceeded

on these downstream reaches, sedimentation will occur irrespective of source. If this is correct, the higher the upstream load, the greater the likelihood of exceeding the sediment carrying capacity, and the greater the potential removal of polluted sediments in the lower gradient reaches.

3. High sediment loads represent a source of Hg to the Bay (i.e., the higher the SS load, the greater the Hg load although note that when this occurs from a relatively unpolluted areas this could be useful for supplying clean (low particle concentration) sediment to the Bay even though there may be a high load). This would suggest there may be benefit in maximizing sediment removal to reduce loads in highly polluted areas and minimizing sediment removal in less polluted areas. High sediment loads are largely coming from areas which may have low Hg levels in the suspended sediment. These sediments represent a pollution recovery mechanism for Hg in the Bay, and accomplish this in two ways. Firstly, burial of polluted sediments is a major recovery mechanism, and burial may be a major mechanism in South Bay. Secondly, incoming sediments with low Hg levels (e.g., 0.06 mg/kg) could dilute Bay sediments with high Hg levels (e.g., 0.33 mg/kg) to reach target concentrations (0.2 mg/kg).
4. Similarly for PCBs, while targeting and removal of polluted sediments will be a priority for BMPs, the highest sediment loads are largely coming from areas which will have low PCB levels in the suspended sediment (e.g. upland open land use, earthworking for new urban development). As with unpolluted upland sediments and Hg pollution, these sediments represent a pollution recovery mechanism for PCBs in the Bay.

4.3.2. The physical nature of particulates in stormwater

4.3.2.1. Introduction

The nature and characteristics of particulate material determine the transport and treatability of sediment-bound Hg and PCB. Behavior and treatability of particles transported by rainfall-runoff is in turn a function of particle size, density, concentration, charge, amphoteric behavior and gradation (the mix of different particle sizes). Stormwater runoff is a heterogeneous discharge composed of entrained particles from eroded watershed soils or generated by urban activities and detritus (in large part due to tire and pavement abrasion and construction) and dissolved/complexed inorganic/organic constituents generated from the interaction of typically acidic rainfall with various urban and infrastructure surfaces and residual detritus. Properties of particles in runoff vary widely from site to site as a function of loading, hydrology, geology and infrastructure. To a large degree, particles are substrates of quartz, calcite, aluminosilicates and inorganic particles often incorporating aggregates of clay minerals and organic matter. In soils dominated by swelling clays (montmorillonite family), the basic building blocks of aggregates are tactoids of clay formed by the flocculation of these clays by cations. Secondary building blocks are the aggregates formed by the bonding of these tactoids by cementing agents such as iron oxyhydroxide and organic matter. In soil environments having undergone more severe weathering (i.e. greater loss of silica), the mineral phase is dominated by iron and aluminum oxyhydroxides onto which organic matter will sorb. Microorganisms can also be bound to these surfaces. Particulate organic matter, especially plant material from deciduous vegetation either whole or disintegrated by vehicle tracking, grass cuttings, paper and plastic litter, and cigarette butts can form a significant fraction of the total organic matter.

4.3.2.2. Particle size distributions observed in stormwater

“Particle” sizes vary from “gross pollutant size” that includes trash measured in centimeters to colloidal material measured in nanometers. Five classes of particles can be recognized. These are colloids, suspended material, settleable material, sediment material, and gross solids, which are differentiated based on size and gravitational settling.

Table 4-3. Particle size classes in urban stormwater (from Strecker et al., 2005)

Solids Fraction	Size range
Gross solids	Generally > 10 mm
Sediment	> $\approx 75 \mu\text{m}$
Settleable	≈ 25 to $75 \mu\text{m}$
Suspended	≈ 1 to $25 \mu\text{m}$
Colloidal	< $1 \mu\text{m}$

Gutter samples and parking lots: Particle sizes have been measured in urban stormwater in a number of studies. Pitt et al. (1995) measured particle sizes in 121 stormwater inlet studies in three states, using both manual and automatic sampling methods. The samples were from gutter flows in a residential neighborhood (southern New Jersey), a long term parking area (Birmingham, Alabama) and a mixture of parking areas and gutter flows in residential and commercial areas (several cities, Wisconsin). The New Jersey samples were collected by dipper, the others by automatic samplers. Particles were sized with the Coulter Counter and verified with microscope, sieve and settling columns. The median particle sizes ranged from 0.6 to 38 μm , and averaged 14 μm . The 10 percentile sizes ranged from 0.5 to 11 μm and averaged 3 μm . The 90 percentile sizes ranged from <1 to 90 μm . There is a great deal of variation between runoff events and this variation was greater than the differences between geographical location, but it is clear that most particles were found to be in the non-settleable fraction (nominally <25 μm).

Urban drain: PSD has been measured for 46 runoff samples in Madison, Wisconsin by the USGS and Wisconsin Department of Natural Resources at the inlet to a stormwater detention pond (described in Burton and Pitt, 2002). This study included the measurement of bedload, which was about 10% of the total load. Table 4-4 list proportions of sediment observed in various percentiles. “Non-settleable” (“<25 μm ”) particles form the majority of the sample and range from ~50-90% of samples. The proportion of >62.5 (or >75) μm ranges from ~5-45%. There is also a great deal of variability; medians (50th percentiles) for the 46 runoff events range from 2-26 μm , while the 90th percentiles range from 35-1100 μm , reflecting that larger particles are transported, but their proportion is small.

A number of recent studies collected all runoff and all particulate matter in runoff events from highways in large reservoirs. These studies separated the solids by extended settling, dried the resulting sandy material and determined the particle size through mechanical shaking and sieving. Unlike the studies described above, these studies found most of the particle mass was made of larger particles (sediment sized, bedload) and the suspended fraction was only 10-20% (e.g., Lin et al., 2004, Lin and Sansalone, 2003). The results of these studies are biased towards

Table 4-4. Particle sizes measured in 46 stormwater runoff events in Madison, Wisconsin (sizes are approximate only – read off graph) (Reported in Burton and Pitt, 2002).

Percentile	median	Range
10th	0.8	<1.3
25th		0.8-5
50th	9	2-26
75th		10-400
90th		35-1100

the land use investigated (highways) and snow hydrology and are probably not likely representative of Bay Area conditions. In addition, these studies dried the samples prior to determining the particle size, a methodology that might have influenced particle characteristics.

The following table summarizes some particle size data from these and other studies. This data compilation relied on reviews by Pitt (2002), James (2002) and Clark et al. (2003). The data has been separated into two groups in the table. The first group has median particle size < 100 µm. The second group has median particle size > 100 µm. Those studies which form the first group sub-sampled the stormwater using automated samplers or sampled supernatant only. Those which form the second group collected the total runoff volume, but the studies were limited to highway runoff. Considering the highway runoff work by Lin et al. and Sansalone is likely uncharacteristic of conditions in stormwater conveyance systems of the Bay Area, the data presented in Table 4-5 support a hypothesis that Bay Area stormwater will be dominated by particles <100 µm. This hypothesis is consistent with a review of USGS suspended sediment data collected in the Bay Area (McKee, personal communication).

Table 4-5. Particle sizes in stormwater runoff.

Median (µm)	Range (µm)	Note
3.8	100% < 62.5	General urban (Clarifica, 2003)
4.5	96% < 62.5	General urban (Clarifica, 2003)
9	2-26 (storm medians)	Urban (Burton and Pitt, 2002)
14	0.6-38	Range of medians from gutter flows from 121 runoff events in 3 cities (Pitt, 1995, 1996)
18		Public Works yard (Corsi et al., 1999)
15	75% < 50	Motorway runoff (Andral et al., 1999)
20	25% 10, 75% 50	(Vignoles and Herremans, 1995)
<25		Shopping centre parking (Randall et al., 1982)
50	36-70 (storm medians)	Residential, commercial (Engstrom, 2004)
78	62% < 100	Detention basin, surface layer (Jacopin et al., 1999)
87	28-148	First flush 21 events of bridge runoff (Drapper et al., 2000)
	80% < 88	Highway runoff (Kobringer, 1984)
	70%-90% <100	Residential (Ball and Abustan, 1995)
	69% <62.5	Residential, commercial and industrial (Auckland Regional Council, 2004)
	20% > 125	
172	2% < 25 µm 20% <75 µm	Highway, Baton-Rouge (Lin et al., 2004)
550	300-500 (storm medians)	Cincinnati Highway (Sansalone et al., 1998)
633	90% > 117 50% > 555	Baton Rouge Highway Total sediment captured from runoff (grit + suspended + settleable + sediment) (Lin, 2003)
11		Suspended fraction (mass-based) Baton Rouge (Lin, 2003)

While there is a great deal of variation, it is suggested that there are two distinct groups. The difference between the two groups has led to some controversy about the nature of PSD in stormwater. Generally the colloidal, suspended, and settleable particles are regarded as the most important or “mobile” particles, i.e., the $<75\ \mu\text{m}$ fraction in urban stormwater. They are the ones that usually define stormwater water quality – along with the dissolved fraction. Most sampling strategies for stormwater characterization, loading and BMP assessment only measure these, and do not consider the larger particles. However, it is also recognized that a large proportion of the total mass that is transported from the primary urban upland source areas, such as roads and other pavements, are in the settled or sediment and gross solids size fraction ($>75\ \mu\text{m}$). Catchbasins, grit basins, pipe blockage, sediment build-up in storm drainage and sediment forebays, all attest to the importance and magnitude of this particle size fraction. Most of the mass that builds up on pavements is in the larger particles sizes, with the suspended/settleable fractions typically being $<20\%$ (Pitt, 1996, Burton and Pitt, 2002, James, 2002). In fact, the PSD measured by Sansalone and coworkers is similar to distributions measured in street dust (James, 2002).

The differences between the two groups may be due in part to the energy available for mobilization of particles. Highways may represent optimum conditions for mobilizing larger particles and increasing bed load. Efficient drainage is important to remove runoff as quickly as possible, and there is a lot of energy from high-speed wheel tracking available for mobilizing large particles. Such energy is not as available on other pavement surfaces (e.g., suburban streets, parking lots, driveways, foot paths).

The differences between these two groups of studies may also be at least partly due to the methods used to collect and analyze sediment. There has been some controversy about measuring suspended sediment in urban stormwater centering on the following three problems:

1. Sampling (isokinetic sampling, depth-integrated sampling)
2. Laboratory testing protocols (TSS versus SSC)
3. Particle size measurement methods

These are briefly examined in the “**Particle size measurement**” below. Due to these problems, many studies have not have representatively sampled and measured the PSD in stormwater. However, it is highly unlikely that the two groups of studies have drastically misrepresented PSD, i.e., that the first group has greatly underestimated largest particles, or that the second group has greatly over-estimated the larger particles. We assume that the difference between the two groups is largely due to a difference in source characteristics and in the energy available for mobilization at different locations within the stormwater drainage system.

4.3.2.3. Density

Specific gravity is typically $2.5\text{-}2.7\ \text{g/cm}^3$ for larger particles (sand-sized or larger), but is more highly variable for smaller particles. Specific gravities for stormwater particles have been reported as $1.1\text{-}2.5\ \text{g/cm}^3$ (Butler et al., 1996) and $1.5\text{-}2.5\ \text{g/cm}^3$ (Pitt, correct citation unknown); the lower densities being associated with organic matter. Bulk densities range from 1 to $1.5\ \text{Mg/m}^3$ for organo-

Particle size measurement

Sampling: Isokinetic sampling achieves sample bottle inlet velocities equal or greater than the flow velocity of the water body. Lesser inlet velocities result in unrepresentative sampling, especially of larger particles. This is typically experienced with simple bottle sampling and some automatic samplers. However, many automatic samplers have sampling velocities far greater than commonly encountered stream velocities. Depth integration sampling recognizes that particle size gradients can occur in all but the most turbulent storm flows. Automatic samplers cannot representatively sample all particle sizes from the whole water column.

TSS versus SSC laboratory protocols: The TSS protocol analyzes a subsample of the collected sample, while the SSC protocol analyzes the whole sample and is regarded as the best estimate of suspended sediment (SS). The former usually suffers from the difficulty of not representatively sampling larger particles, but is the most common practice in stormwater studies – often because other analyses are performed on the sample. For example, serious discrepancies can occur when 30% of particles are $> 62.5 \mu\text{m}$ (Gray et al., 2000). Given that smaller particle sizes appear to dominate urban stormwater (Table 4-5) and carry much of the mercury and PCB load (see section 3), the method of analysis for suspended sediments may often be inconsequential, however, the SSC method offers the best guarantee of consistency when samples occasionally include larger particles.

Particle size measurements: Particle size methodologies are notoriously incomparable (Kayhanian et al., 2005a) and some techniques are biased toward smaller particles. There is no standard method for measuring particle sizes in urban stormwater (Bent et al., 2001). Sample preparation is critical. Any technique that manipulates the sample (long holding times, agitation – apart from gentle shaking such as inversion, concentration by settling, centrifuging or filtration) will probably change the PSD. Any concentration technique (e.g., separating solids from runoff and drying in order to sieve) completely changes particle sizes, because flocculation and aggregation occurs. Subsequent disaggregation techniques cannot be relied upon to reproduce the natural particle size distribution¹. Standard methods for soils overcome this difficulty by measuring the mineralogical particle size after mechanical grinding, oxidation, adding a dispersing agent, and dispersing suspensions. However this does not measure the natural particle size.

Techniques that measure PSD in un-amended stormwater samples also suffer from methodological difficulties. Long holding times (e.g., sometimes utilized in automatic samplers) may result in flocculation (Kayhanian et al., 2005), while agitation may result in disaggregation. Techniques that rely on direct or indirect observation of particle in the water column often lack QAQC procedures adequate to the study objectives, and may not measure the true PSD.

Definition of particle sizes and their measurement

Particle sizes can be defined in two ways (e.g. Walling and Woodford, 1993). First, there is the naturally-occurring particle size. Larger particles in urban drainage can be agglomerations of small particles (e.g. Gartner et al., 2001). In urban drainage systems, it is the naturally-occurring particle size that determines the transport properties of the particles (e.g. Syvitski, 1991). Secondly, there is the laboratory or mineralogically defined particle sizes, whose measurement involves prior disaggregation. Disaggregation usually involves separation of the particles from the water and chemical disaggregation (e.g., using peroxide to oxidize organic matter and adding dispersing agents (e.g., Calgon and physical dispersion, ultrasonics). The degree of disaggregation will depend on the rigor of the disaggregation step, and it is highly likely that this rigor varies from study to study. It is naturally occurring particle sizes that occur in an unmanipulated field sample we are most interested in when trying to understand settling velocities within BMPs.

¹ This is almost invariably observed for silt-sized or smaller particles, or samples that contain a significant proportion of these sized particles. This tends not to happen for sand or greater sized particles with little mud. This apparently was the situation in Lin's (2003) detailed studies.

mineral aggregates in surface layers of agricultural soils (Fred Hetzel, Personal Communication). We anticipate that particles in urban runoff will exhibit bimodal or even multimodal density distributions depending on source characteristics.

4.3.2.4. Mobility

The characteristics of the heterogeneous mixture of material found in urban stormwater are determined by both the energy and the type of source materials available for transport. As described earlier, urban runoff can contain very non-uniform particle gradations that vary from colloidal-size to gravel-size material with runoff residence times that are relatively short compared to other waters (Sansalone et al., 1998). For example, source area watersheds, such as urban transportation land uses with relatively short residence times that are designed for rapid conveyance of runoff, and where vehicle tracking creates highly turbulent conditions (expelling water from treads by pressure and centrifugal forces, tire swash) are capable of transporting a much wider gradation of particles than larger, hydraulically less efficient watersheds. As this gradation is transported downstream in a larger urban watershed, larger particles are selectively removed in the flow path resulting in a selectively finer gradation as flow moves through the urban watershed.

Whereas natural stream waters may have residence times of hours to days and wastewaters on the order of hours, stormwater may interact and entrain particles for only minutes to an hour before reaching the catchment outlet (Sansalone and Buchberger, 1997).

4.3.2.5. Settling

Gravity separation or settling of gross solids, sediment ($> 75 \mu\text{m}$), and settleable ($25\text{-}75 \mu\text{m}$) particles occurs throughout the stormwater drainage network whenever there are significant changes in transport capacity of the runoff. Smaller particles can also settle because of flocculation and aggregation processes. Settling velocity is the rate at which a particle settles in quiescent water typically measured as cm/s. Settling is determined by particle density, particle shape and water viscosity and density (in turn affected by temperature). It is also affected by particle-particle interactions (which in turn are controlled by SS concentrations) which gives rise to four general classes of particle settling (Strecker et al., 2005, based on Metcalfe and Eddie, 2004) (Type I through IV).

Type I is discrete particle settling and is the most common assumed behavior employed when estimating settling rates. It occurs at low concentrations where particle-particle interaction is insignificant, which as a general rule is when suspended solid concentrations are less than 100 mg/L. However, this is dependent upon the size gradation of the particles. For example, 100 mg/L of clay particles would undergo flocculation and Type II settling, while 100 mg/L of sand particles would behave as discrete particles (Lin et al., 2004b).

Type II settling is flocculent settling and generally occurs at concentrations above 200 mg/L and is described more fully in the following section. However the same dependence on size gradation as discussed for Type I settling is still applicable. Unlike Type I settling, Type II settling is dependent on sedimentation depth.

Type III settling is hindered or zone settling, which occurs when particle concentrations are generally above 500 to 1000 mg/L.

Type IV settling is compression settling where particle concentrations are measured in percent. The greatest concentrations measured in the Bay Area in USGS gauging studies range between 1-7% for rivers and creeks with watershed areas larger than a few square kilometers (see review: McKee et al., 2003).

Particle settling velocities are one of the most important factors determining the transport, transit time, fate and treatability of SS. Particle size and settling velocity are often used interchangeably via Stokes Settling Theory, which assumes particles are spherical, homogeneously-dense particles, and consequently particle size is often reported as a surrogate for settling velocity. However, particles are rarely perfect spheres and are mostly irregular in terms of shape and density. This is particularly true if agglomeration or flocculation has occurred. Therefore, typical settling velocities calculated from particle size information and based on the specific gravity for quartz (2.65) are probably incorrect, and settling may be slower (less dense) or more rapid (due to flocculation).

The most comprehensive study measuring settling velocities was the National Urban Runoff Program carried out in the United States in the early 1980s which collated data on stormwater particle settling characteristics from a set of 46 settling column tests (Driscoll 1986, USEPA, 1983). There were a wide range of particle sizes, and hence settling velocities in any individual urban runoff sample. The distribution of settling velocities could be adequately characterized by a log-normal distribution. There was substantial storm-to-storm variability in median (or other percentiles) settling velocity at a specific site. The range indicated was about one order of magnitude in any percentile of the distribution in a specific storm. Uncertainty in the coefficient of variation of the site-averaged settling velocity distribution (95% confidence interval) was smaller, but still appreciable (about a factor of 5). No significant differences between site-to-site mean distributions were identified. The within-site variability was on the same order as potential site-to-site differences. The foregoing indications, with regard to storm-to-storm and site-to-site differences, supported the pooling of all available data to define “typical” characteristics of particle settling velocity distributions in urban runoff, and the assumption that such results are generally transferable to other urban runoff sites. Table 4-6 illustrates best estimates for the distribution of particle size velocities in urban runoff from any site. These tests results were collated into five groups of settling velocities containing a range of particle sizes for planning purposes (Table 4-6). Nominal particle sizes are also given in Table 4-6 based on Stokes Law. The EPA design settling velocities ignore mobilization and settling of larger particles.

4.3.2.6. Flocculation (Type II settling)

In stormwater runoff at the upper end of the urban watershed, the colloidal and suspended fractions are mixed with the settleable and sediment fractions in a relatively shallow water column (mm to cm). With residence times of these particles generally less than several hours and with unsteady flow, floc development will probably not occur in runoff during its passage through the urban catchment (Strecker et al., 2005).

Table 4-6. USEPA Design settling velocity categories and equivalent particle size mean and range assuming Stokes Law settling (Driscoll, 1986). Also mean particle diameters are shown for particles with higher proportions of organic matter and hence low densities (ρ).

Settling velocity, mean (m/hour)	Equivalent diameter mean (μm) $\rho=2.65$	Equivalent diameter range (μm) $\rho=2.65$	Equivalent diameter mean (μm) $\rho=1.1$	Equivalent diameter mean (μm) $\rho=1.5$	Equivalent diameter mean (μm) $\rho=2$
0.009	2	<3	8	3	2
0.1	6	3-7	26	12	8
0.5	14	7-20	58	26	18
2	28	20-60	115	52	37
20	90	60-125	365	163	115

However, flocculation can occur naturally in stormwater runoff if it is retained or ponded. Depending on parameters such as turbulence, pH, ionic strength, and particle properties, natural flocculation can begin within several hours to 12 hours of initial runoff. Natural flocculation may be enhanced by non-urban upland inputs of suspended sediment (by increasing the concentration of suspended particles). With the exception of small sewersheds of the Bay Area (where little characterization data are available), most if not all watersheds that include upper areas in either the East Bay hills, Santa Cruz or peninsular mountains will have concentrations of suspended sediment well in excess of 100 mg/L, which are in excess of concentrations needed to initiate natural flocculation processes (about 50-100 mg/L depending on particle size).

In summary, the rapidity of transport processes and turbulence mean that there is little opportunity for flocculation in the urban drainage system. However, concentrations are high enough for such processes to occur if runoff is ponded (e.g., wet pond BMPs) or held in quiescent conditions (e.g., sample storage).

4.3.3. Deposition and resuspension of sediment in drainage networks

4.3.3.1. Transport processes in natural drainage systems

Transport processes of sediment involve the continual deposition and resuspension of sediment as it travels down through the drainage network. An earlier review of urban stormwater examined these processes in the San Francisco Bay watershed setting. The following (summarized here for convenience from McKee et al., 2003) outlines these processes. The reader is referred to the original review for more detail.

Sediment from its various sources can be stored temporarily or permanently in various sinks within the fluvial system. The magnitude and distribution of sediment (and related pollutant) storage will vary from reach-to-reach within a creek and also between watersheds depending on factors such as stream slope, valley confinement, geology, soils, land use, the presence of

reservoirs, and climate. These factors, in turn, affect the temporal and spatial sediment discharge from local tributaries to the Bay.

Over the long term, streams will transport almost all sediment that is supplied; however, over the short term, streams will use the sediment that is supplied to construct a path within the channel and valley confines. The storage of sediment by a stream contributes to channel morphology, including instream features such as bars, riffles and dunes, and out-of-stream features such as floodplain, (wetlands, alluvial fans) and terraces.

Modified channels, especially flood control channels that have been re-graded and widened, can fill with large volumes of sediment as the channel tries to return to equilibrium. This filling process has been observed in many modified channels in California. For example, in 1959 the San Lorenzo River in Santa Cruz County was modified as part of a flood control project by the Army Corps of Engineers (Griggs and Paris, 1982). The channel was widened and dredged to increase the slope and capacity of the river, however this modification drastically increased the channel's gradient by 32%. In an effort to return to its original gradient, the river deposited large amounts of sediment in the channel, raising the channel bottom 0.9-1.2 m (3-4 ft) above the original channel bed. In 1982 it was estimated that 350,000 m³ (450,000 yd³) of sediment must be removed in order to restore the channel to its original flood control design (Griggs and Paris, 1982).

In natural systems sediment storage also occurs in floodplain deposits during floods when the discharge is greater than the channel can convey. The channel uses the floodplain to disperse excess flow, resulting in decreased velocity and power and deposition. However, these natural processes pose a natural hazard to urban and agricultural communities that utilize the flat rich soils of the floodplains. As such, channels that have been deepened, widened, or leveed are most often disconnected from the floodplain. Because discharge is retained in the banks, the stream will have more power, which results in erosion of banks, undermining or complete failure of revetments, or flood and erosion problems further downstream (e.g., Collins, 2001). A decrease in stream access to the floodplain and the increase in erosive power potentially result in greater sediment (and pollutant) discharge from local small tributaries in the Bay Area.

During small storms, much of the sediment and pollutants entering the Bay may be eroded from temporary storage in channels. During larger events, a greater proportion of sediment and pollutant loads will be derived from source areas outside of the near channel environment and perhaps anywhere within the drainage basin.

As discussed, for any given natural watershed, only part of the sediment (and related pollutants) eroded on the hill slopes or supplied to the stream will end up reaching the ultimate receiving waterbody (the Bay). Much of the remainder will be stored in various locations on the valley slopes, near and in channels, and on the floodplain.

Novotny and Chesters (1989) describe methods of calculating soil loss and sediment delivery in the context of nonpoint source effects upon water quality. The "delivery ratio" describes the relation between basin sediment yield and upland erosion generation potential; $Y=DR(A)$, where Y is the basin sediment yield, A is the upland erosion generation potential, and DR is the

delivery ratio. The delivery ratio captures the different physical sediment storage processes occurring in a watershed, and ideally represents processes occurring on a 5-10 year time period. It is necessary to use some ratio between upland erosion and downstream sediment delivery and transport, but problems arise when estimating a single delivery ratio for a watershed, including: the time span considered; the spatially lumped character of the delivery ratio over an entire watershed; and the seasonality and hydrological variability of the parameter (Novotny and Chesters, 1989). Problems with seasonality arise because of the intermittent nature of sediment movement and the variable correlation between individual runoff events and sediment delivery.

Despite the lumped nature of the sediment delivery ratio there is a relationship between delivery ratio and watershed size. Larger watersheds retain a greater proportion of eroded sediment than smaller ones. Watersheds draining into the San Francisco Bay vary in size from essentially <1 km² up to 1662 km² (0.4-642 mi²), although the 10 largest watersheds have an area greater than 105 km² and comprise about 75% of the total watershed area around the Bay. Although watersheds in the Bay Area vary in slope, geology, intensity of tectonic deformation, and rainfall, as a first approximation, sediment delivery ratios for the 10 largest watersheds are likely to range from 55%-7% for the smallest and largest basins, respectively. The relationship has some scatter, but on average, the sediment delivery ratio may be approximately 20%.

4.3.3.2. Transport and deposition processes in urban drainage systems

For urban drainage systems where channels that have been deepened, widened, or levied, the stream no longer has access to the floodplain for sediment storage. In some channels, this has resulted in greater stream power, and hence less storage in the channel as well. This will largely be true in urban areas upstream of the flood plains and reclaimed areas on the Bay margins. However, on these flatter floodplain areas, many of these modified channels have been redirected, re-graded and widened to meet stormwater drainage requirements and this can provide additional channel storage as described above. The dynamics of this storage of sediments are complicated by periodic removal processes other than hydrological. Notwithstanding this storage and removal facility, sediment delivery ratios are likely to be substantially greater in urban systems than for a natural system.

Despite the large changes in the interaction of the stream channel with its banks and floodplain, some runoff processes in urban areas mimic the natural processes. The upland sources have been replaced by runoff from roads and roofs. Coarse sediment mobilized here is deposited when hydraulic energies decrease; such as in drop inlets, catchbasins, pipes and flood channels. As described earlier, these processes result in routine removal of accumulated sediment in these structures throughout the San Francisco Bay watersheds. Even roof runoff shows this phenomenon. Washoff from roofs contains mostly fine or dissolved material, which is easily mobilized. But often there are big reductions in slopes when runoff drains off the roof to the gutter. Gutters usually have much gentler slopes and even pond water during and after rainfall. Therefore particulate material sometimes accumulates in the gutter, effectively reducing the sediment delivery ratio to less than 100%.

In summary, sediment transport is a complex process of erosion, deposition and resuspension. These processes can be captured semi-quantitatively in the concept of the sediment delivery

ratio. In more natural systems, the SDR is often less than 50%. In urban systems, the usual patterns of erosion, deposition and resuspension are greatly disrupted by the large landscape and drainage changes. Consequently, SDRs would be expected to be larger than natural systems, because of the lower opportunity of deposition and higher energies for resuspension. However, particular opportunities for deposition can occur through deliberate intervention (structural BMPs such as detention basins) or unintentionally (through constructing flood conveyance channels with more gentle slopes than found up-channel).

4.3.4. Deposition at the Bay margins

4.3.4.1. The settling process

In pre-European times, most stormwater runoff did not carry sediment to the main body of the Bay, but only as far as the wetlands on the Bay margins. The streams cut direct channels to the Bay only after several years of high rainfall. At present, artificial channels have been cut directly to the Bay as part of the land drainage system. The lower portions of these channels are tidal. At high tide these channels are at 'zero' gradient and are effectively backwaters or ponds. At low tide they are effectively an extension of the stream or drain channel because stream flows are carried in low tide channels to the tidal front. At some point the channels widen and the channel takes on more of an estuarine quality, eventually widening out to the main body of the Bay.

The fate of suspended sediment and attached pollutants depends on the state of the tide, the size of the storm and the morphology of the estuary at the tidal front. At high tide, within a short distance of the discharge meeting the tidal front, currents will be low enough to provide ideal settling environments (Hume and McGlone, 1986). The coarser particles settle by gravity because there is a large drop in the velocity of the water carrying these particles. For smaller storms, or where stormwater discharges into a large estuarine area, in addition to settling by coarse particles, a proportion of the finer particles will begin to flocculate and the resulting flocs settle to the bed. For small watersheds and small storms, these processes occur in the near-shore areas, possibly still in the channels leading to the Bay. Therefore, the immediate fate of a proportion of the pollutants after entering the estuary is deposition by settling in the upper reaches of the estuary (Williamson and Morrissey, 2000). As the tide retreats, stormwater will flow in channels incised within intertidal flats. This may result in scouring of any previously deposited fine sediments (e.g., during the previous high tide) and deposition of coarse, urban-derived sediments in these channels. On reaching the tidal front, stormwater will be mixed and spread out over lower intertidal and subtidal areas, with the mixing/settling field therefore tending to spread down-estuary. A similar picture holds for the rising tide, except the mixing/settling field moves up-estuary (Williamson and Morrissey, 2000).

In larger watersheds or during large storms, greater discharges will push stormwater further down the drainage channel irrespective of tide. During very large storms, fine particles and dissolved pollutants will be carried well out into the Bay as the large volume of fresh water displaces saline water from the estuary arm or channel and/or as higher buoyancy of the large fresh water inflow spreads over the top of the saline waters of the Bay. Coarse particles may still settle in the upper reaches because of the decrease in velocity or turbulence. At mid-high tides, the discharge field will tend to spill out over the top of the adjacent intertidal areas, which

provide ideal quiescent settling areas. Stormwater solids tend to build up on the low tide channel margins (Green et al., 2001). As the tide ebbs, most of the flow is concentrated in the low tide channel. Settling will occur only where channels widen significantly and the stormwater discharge can spread out over the intertidal and at low tides, subtidal areas.

4.3.4.2. The redistribution process

Once settled, polluted sediment is intermittently resuspended and redispersed. Direct observations of small wave (5–20 cm) action show very high turbidity in shallow waters behind the tidal front (Williamson and Wilcock, 1994; Green and Bell, 1995). Very fine sediments (clays and fine silts) can be transported in suspension for large distances (100's m) until reaching quiescent areas, whereas the coarser fraction of the suspended material (medium silts to fine sands) settles within short distances (e.g., <10 m) of the point of resuspension. The continual advance and retreat of the tide means that pollutants can be spread widely over the intertidal zone.

Some of the sediment resuspended by waves will escape from the estuary on the ebb tide. However, since many estuarine arms discharge into effectively enclosed basins, much of the “escaped” material will return on the next flood tide. Because of estuarine processes ‘**settling lag**’ and ‘**scour lag**’, fine sediments tend to march up-estuary. The concept originated with Postma (1967). **Settling lag** results in an up-estuary increase in fine-sediment concentration and deposition. As a suspended grain is carried up-estuary, the tidal current slows until it eventually falls below the critical speed needed to keep it in suspension (the critical deposition speed). However, because the grain has a finite settling speed, it takes time to settle to the bed. During that time, the grain is carried further up-estuary, beyond the point at which the transporting current fell below the critical deposition current speed. When the tide turns, and assuming the tidal currents are symmetrical, the deposited grains will not be re-entrained until much later in the tidal cycle because they are further up-estuary due to the settling lag. So they will be suspended for a shorter period, and will move less distance down-estuary. So we have a “ratchet” effect here: 2 steps up, one step back. It is a continuous process, and causes suspended sediment concentrations to increase in upper estuary, and therefore deposition is also increased. Note that the finer the sediment, the more this mechanism operates, because it takes longer for the sediment to settle once the flow falls below the critical deposition speed.

The **scour lag** is subtly different. Whereas settling lag is the time taken for a sediment particle to reach the bottom once the transporting current falls below the critical depositional speed, scour lag is the delay due to the difference in the current necessary to erode the particle from the bed and that occurring at final deposition. That is, the critical erosion speed that is bigger than the critical deposition speed. So if a particle is being transported up estuary on the flood, when the transporting current speed falls below the critical deposition speed, the particle falls to the bed (as this takes time, the settling lag mechanism is operating). After the tide turns, because the erosion speed is greater than the deposition speed, then it is later in the tidal cycle before the grain is picked up, so it travels less distance back out into the estuary. Again, this is a “two steps forward one back” mechanism. Again, this favors fine sediments, because for coarse sediments, the critical erosion speed is the same as the critical deposition speed.

4.3.4.3. Observations from Auckland urbanized estuaries

Pollution by Zn, Cu and Pb occurred with the onset of urbanization in the 1960s in Auckland estuaries (Swales et al., 2003). Figure 4-3 and 4-4 show the depth of pollution in cores collected down narrow estuaries in Auckland New Zealand. Pakuranga Creek Estuary is about 100 ha (250 ac) in size and receives runoff from a 25 km² (10 mi²) watershed. Lucas Creek has a similar size estuary and watershed areas, but urbanization began in the 1990s. Both estuaries are highly intertidal, nearly draining fully at spring low tides. The results are examined here because the pollution profiles give insight into the scale of the processes occurring in narrow estuaries, such as the channels leading to the main body of the San Francisco Bay.

The studies are able to differentiate the accumulation of urban-derived pollutants within the estuary. The figure for Pakuranga Creek shows very high deposition rates in the uppermost estuary (about 1 m since 1960), then a decrease in deposition rates down the estuary, until site 5 where pollution barely reaches the lower estuary. The limited data for Lucas show the same phenomena, with probably more than 45 cm of polluted sediment at the uppermost site deposited since about 1990, and a 12-15 cm of polluted sediment at the lower site (after allowing for bioturbation). Hence both studies show substantial accumulations of pollutants in the uppermost reaches of estuaries. Note that the very high variability in pollutant concentrations in Pakuranga cores is due to periods of intense construction and little erosion controls on earthworking operations, interspersed with periods of lower constructional activity where runoff from mature urban areas predominates (Swales et al., 2003). The eroded subsoils contain low levels of trace metals while sediments from mature urban areas are high in trace metals.

In summary, although hydrological and tidal processes spread pollutants out widely over estuarine areas, the dominant processes of settling of coarse particles, flocculation and settling and scour lag result in a large accumulation in the upper estuary.

4.4 Chemical nature of particulates

4.4.1. Particulate Organic Carbon (POC)

Many SFB studies have proposed the importance of particulate organic matter in the transport of Hg and PCBs (e.g., McKee et al., 2003, Johnson and Looker, 2003, Hetzel, 2004). Hg²⁺ is adsorbed by particulate organic matter while PCBs both adsorb and fractionate or dissolve into organic matter. Observed distribution coefficients, K_d , the ratio of particulate to dissolved fractions, are determined in part by organic matter content of particulate matter. Therefore it is important to describe the concentration, properties and characteristics of particulate organic matter in urban stormwater.

Particulate organic matter (POC) is typically a heterogeneous mixture of different types of material for two reasons: (1) POC is derived from a heterogeneous source materials (e.g., plants, animals, microbes, and man-made materials such as plastic, petroleum products, rubber, bitumen); and (2) POC can be modified by a wide variety of physical, chemical and biological processes including combustion and pyrolysis (e.g., pressure/thermal alteration of plant material to form coal, combustion of wood to form soot) (Allen-King et. al., 2002).

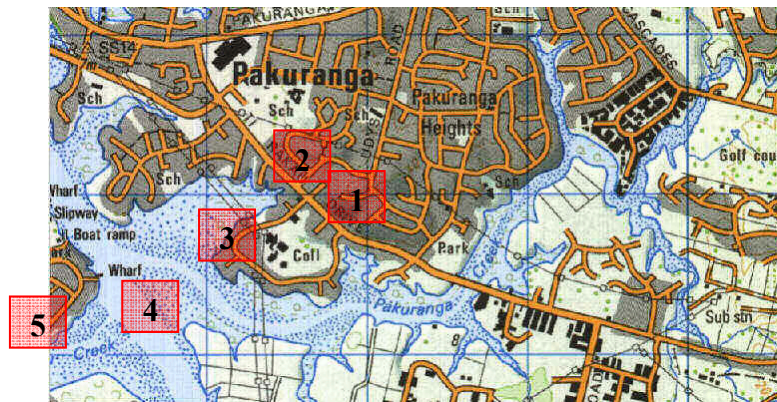
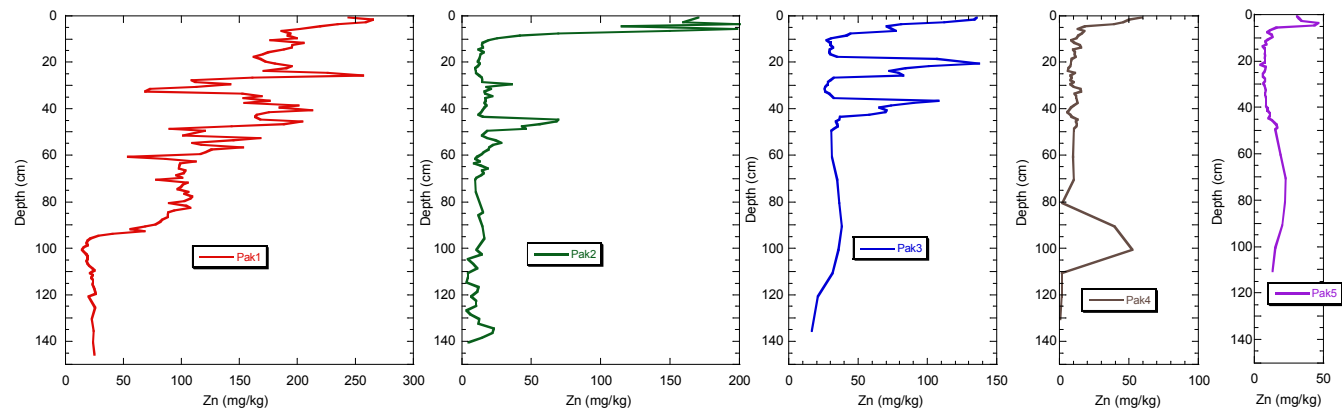
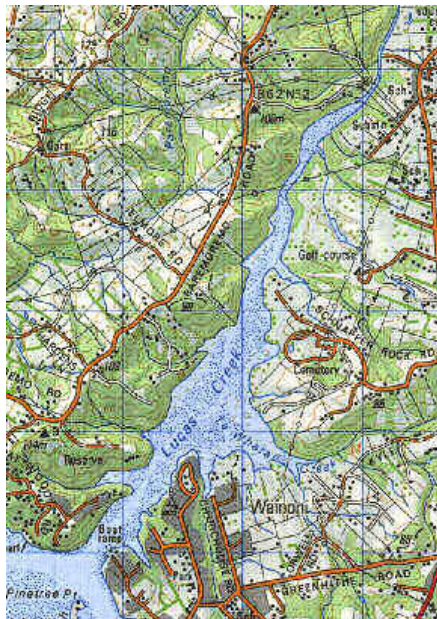


Figure 4-3. Zn profiles for single cores from Pakuranga Creek, Auckland New Zealand. Pak1 core is uppermost in the estuary, while Pak 5 is lowermost. The map shows the estuary and sampling sites where 1=Pak1 and 5 = Pak5.



POC can also be viewed as two separate physical types. The first type is where molecular or colloidal organic matter is adsorbed to mineral grains, and this will probably be dominated by organic carbon such as humic acids and microbe exudates. Until recently, the reactive portion of sediment organic matter was regarded to be of this type which was viewed as compositionally uniform (termed “amorphous”), with a uniform ability to bind pollutants. The second type is where it occurs as discrete particles of carbonaceous

material, such as plant fragments, algae, bacteria, wood, coke, coal and pitch. Some of this material is very resistant to breakdown and inert to many chemical and biological processes.

In terms of mercury adsorption, the first type is probably the most important phase. The coating of small particles by organic matter creates a large adsorption substrate for Hg. This phase is also very important for PCB adsorption. The phenomena of adsorption of hydrophobic organic pollutants onto this type of organic matter has been well described, and has been mathematically expressed very simply using equilibrium partitioning theory. These simple expressions involving the POC concentrations and the highly familiar K_{oc} - the octanol-water partitioning coefficient, were very successful at predicting changes in composition in sediments (Karickhoff et al., 1979, Karickhoff 1981, USEPA, 1993, Ankley et al., 1996).

Normalization with POC

The proportion of organic matter is a factor in determining the amount of adsorbed hydrophobic organic pollutants. This is not true for inorganic ions such as Hg^{2+} , which can adsorb to other sediment phases such as iron oxides. Because of the strong association between sediment organic matter and hydrophobic organic pollutants, it is quite common to normalize PCB concentrations with the proportion of organic matter in sediments. This is especially appropriate in receiving water sediments where researchers wish to reduce the variability in concentrations to detect, for example, time trends. Because organic content is from one of the major factors determining variation, normalization with organic matter content removes this source of variation. Some sediment quality criteria are normalized to total organic carbon (TOC), for example, 1% TOC (Long et al., 1995).

Sediment pollutant concentrations in a number of SFB studies have been normalized to the proportion of mud or 'fines' (< 62.5 μm fraction) (KLI and EOA, 2002, Salop et al., 2002). The primary reason is to estimate concentrations in suspended sediments transported to the Bay. A secondary reason is that it helps "factor out" variation due to particle size and detect spatial, land use or regional differences. However, the normalization process itself is questionable because larger sizes are probably also polluted. It becomes particularly questionable when the proportion of fines is low (e.g., the highest mercury concentration in the KLI dataset was 40 mg/kg with 1% fines – which involves a 100 x's scaling factor). In addition, to compare the normalized data, the assumption must be made that the distribution of concentrations throughout the range of size fractions is similar between samples (an assumption that is unlikely to be true). True inter-location comparisons can only be made by comparing the concentrations in specific grain size fractions or better still, by taking organic carbon content into account.

A number of workers and organizations have normalized Hg (e.g., Horowitz, 1987, Mason and Sullivan, 1998, Mason and Lawrence, 1998, Kofka et al., 1999) and PCBs (e.g., Long et al., 1995, USEPA, 1993) to organic carbon in both freshwater and marine environments. The use of organic carbon helps in the detection and definition of spatial differences between samples, but cannot be used directly to estimate concentrations in suspended sediment. Normalization by this technique is especially appropriate for PCBs because PCBs fractionate into organic matter (OM). It may be still appropriate for Hg, but less so because other substances adsorb Hg^{2+} , e.g., hydrous iron oxides which may have a similar concentration to OM in particles. Of relevance to this normalization technique is the fact that TOC is often highly inversely correlated to particle size, and this is probably a surface area effect (Horowitz, 1987). However, the latter relationship does not hold if significant proportions of particulate organic matter are present (wood, coals, soot, plant material), as the greater mass of these tend to be in sizes larger than 62.5 μm .

For some time there are many instances where unexpectedly strong sorption of hydrophobic organic pollutants such as PAHs (Youngblood et al., 1975, Prah et al., 1983, Socha et al., 1987), and PCBs by sediments occurs (Gustafsson et al., 1997, Allen-King et al., 2002 and references cited within). This is due to the presence of significant quantities of thermally-altered POC from geological or anthropogenic origins, such as coals and soots. The coal-related or combustion-related particles have important properties with respect to the fate and bioavailability of PCB and other hydrophobic organic pollutants like PAH. Unlike the amorphous organic matter, this thermally-altered POC is referred to a condensed or glassy carbon. An example of the importance of this thermally-altered POC is described below (see “Super-sorbents” box).

While the phenomenon has been shown to be very important at one site in San Francisco Bay, it is not clear how important it is in the transport of PCBs to the Bay from local watersheds (Hetzl, pers. comm., Ghosh et al., 2003). Thermally-altered POC is widespread in the environment, however. Natural sources include shales and coal. One of the most common and important man-made form is black carbon, produced by the incomplete combustion and/or pyrolysis of either terrestrial organic matter (e.g., vegetation) or fossil fuels. It can occur either as carbonized residues of initial starting materials (chars) or as particulate carbon that has re-condensed from the gaseous phase (soot). It has been widely found in marine and estuarine environments². Anthropogenic sources in the San Francisco Bay area would include char from historical land clearing or crop residues burning in agricultural soils, soot particles in older highly industrialized regions, and tire rubber, where it is a major component. Land areas which have been infilled or reclaimed using waste materials may have significant quantities of black carbon and other glassy carbon forms such as coal dust (Allen-King et al., 2002). Therefore it will be very important to take it into account in this study in case specific watersheds have significant glassy carbon quantities in their runoff.

4.4.2. Hydrous ferric oxide

In terms of cation adsorption, short range order hydrous iron oxide is a very important phase in sediments. It is formed by the hydrolysis of iron minerals and oxidation of ferrous iron, especially in iron-rich anaerobic water. It is a common component of soils and sediments. Freshly precipitated FeOOH has a relatively high surface area per unit weight, as high as 600 m² per gram (Dzombak and Morel, 1990). The adsorption properties of short range order FeOOH for heavy metals and other substances are well known since early laboratory studies (e.g., Benjamin and Leckie, 1978).

² Just how widespread it is can be illustrated by the following extract from Allen-King et al. 2002. Global black carbon production has increased over the past centuries as a consequence of increased biomass burning and fossil fuel consumption. Post-1900 sediments and soils contain the products of fossil-fuel consumption (oil- and coal derived black carbon), as well as the residues derived from plant combustion prior to 1900. Recent marine sediments show a peak in carbon abundance that is likely to be of anthropogenic origin. It is found to comprise between 12% and 31% of the organic carbon in deep-sea sediments and up to 50% of the extant noncarbonate carbon on marine shelf sediments.

“Super-sorbents”

A number of studies have established that condensed or glassy POC in receiving water environments can contain the bulk of hydrophobic pollutants such as PCB and PAH. For example, in sediments from Hunter’s Point in San Francisco Bay, (a PCB “hot-spot”) and other harbors in Milwaukee and New York, carbonaceous particles constitute 5-7% of the mass and 60-90% of PCBs and PAH (Ghosh et al., 2003). The carbonaceous material is primarily coal, coke, charcoal, pitch and wood. At Hunter’s Point, most PCBs were in particles in the 62.5-260 μm size range. Many marine and estuarine sediments are known to contain significant quantities of soot or other forms of black carbon (Allen-King et al., 2002). In contrast, the mud fraction contained little carbonaceous material and only a small fraction of PCB (and PAH). This PCB appeared to be adsorbed to particle surfaces.

While this association is important in the receiving waters, some researchers suggest that PCB is not known to be associated with production of carbonaceous particles, and so PCBs are not necessarily transported to the receiving waters in this form. These carbonaceous particles in the receiving waters may preferentially accumulate PCBs over long times (Ghosh et al., 2003).

Nevertheless, there is the high likelihood of significant quantities of thermally-altered POC occurring in some urban landscapes e.g., urban areas built on agricultural lands, older industrial areas, land that has been infilled with waste materials (Allen-King et al., 2002). PCB polluted sites may also contain carbonaceous material such as coal dust. Redistribution and sorption of PCBs could occur within the drainage network in the bed sediments. (This redistribution is unlikely to occur in the water column with suspended POC because of the relatively short transit periods). Therefore we need to consider the association of PCBs with large carbonaceous particles in the transport of PCBs in the urban storm drain network. However, it is unlikely that this is a Bay-wide phenomenon; if that were so then there would not be a PCB fish advisory in the Bay.

4.4.3 Pollutant size distribution

It is basic knowledge that the smaller the particle, the higher the surface area: volume ratio, and the higher the degree of pollution on a mass of pollutant per unit particle mass basis. Therefore it is often assumed that the majority of pollutant loads are carried by the smaller (suspended) particles in stormwater.

There is a great deal of published information on the heavy metals Cu, Pb and Zn and to a lesser extent Cd, Cr and Ni. Concentrations of these metals typically increase with decrease in particle sizes (see references later). We would expect a similar picture for Hg because similar phases and chemical processes are at work (e.g., Hg^{2+} adsorption onto organic matter, mineralogical surfaces).

However, it is incorrect to assume that particles larger than 62.5 μm are not polluted with heavy metals. Naturally occurring particles can be agglomerations of smaller particles. Very high concentrations of heavy metals can be found in large particles on roads for example. This may be due to the source of the heavy metals – e.g., there may be large particles of rubber (which contain ZnO), rust, Zn galvanizing. Although an equivalent large particle source of Hg is unlikely, it is possible that PCBs can be found in larger particles, as described later in this section.

It is informative to examine heavy metal concentrations measured in different particles sizes from different studies. Vignoles and Herremans (1995) found a strong relationship between particle size and metal pollution (Table 4-7). Most metals (>50%) were associated with particles less than 10 μm . However, most of the variation occurs in the lowest size fractions and there is very little variation in concentrations the medium silt to very fine sand sizes (20-100 μm). Johnson et al., 2003 found similar variation between particle sizes for stormwater runoff samples from roads and parking lots; most of the variation occurred between <10 μm samples and larger samples (Table 4-8). Again there was little variation in concentrations in the medium silt to fine sand ranges (10-250 μm). They also show that there is a great deal of variation (expressed as coefficient of variation) in the pollution with a particular particle size range. The observed Zn and Cu concentrations were very much higher than those in Table 4-7 above. Sansalone and Buchberger (1997) also found little variation in the silt to fine sands (in this case 25-150 μm) in highway runoff sediments (Table 4-9). Concentrations were much lower than recorded by Johnson et al (2003). This study did not measure concentration in very small particle sizes (<25 μm). Much of the variation of metal concentration with particle size in this study occurred above 150 μm . The results of the above studies contrast markedly with results from highway runoff. For example, Caltrans (2002) found little variation in pollution concentrations (mg/kg) in suspended sediments in highway runoff (Table 4-10).

Table 4-7. Concentrations of Cu, Pb and Zn in different particle size ranges. Proportion (%) of total metal in each range is given in brackets (Vignoles and Herremans, 1995).

Size μm	Proportion of SS (%)	Cu (units?)	Pb (units?)	Zn (units?)
<10	(35)	171 (63)	822 (73)	1232 (60)
10-20	(14)	81 (11)	247 (8)	801 (16)
20-32	(10)	42 (4)	158 (5)	331 (5)
32-40	(9)	46 (4)	163 (4)	398 (5)
40-50	(6)	57 (3)	181 (2)	469 (3)
50-100	(11)	62 (8)	129 (4)	419 (6)
>100	(14)	42 (7)	104 (4)	272 (5)

Table 4-8. Concentrations of Cu, Pb and Zn in different particle size ranges, with coefficient of variation (CoV) given in brackets (Johnson et al., 2003).

Size range μm	Cu mg/kg (CoV)	Pb mg/kg (CoV)	Zn mg/kg (CoV)
0.45-2	2894 (1.21)	199 (1.4)	13540 (1.56)
2-10	4668 (1.60)	868 (0.78)	13641 (1.88)
10-45	735 (0.97)	229 (0.50)	1559 (0.74)
45-106	1312 (1.16)	226 (0.85)	2076 (0.88)
106-250	2137(1.45)	375 (1.03)	3486 (0.79)
>250	50 (-)	117 (0.58)	266 (0.88)

Table 4-9. Concentrations of Cu, Pb and Zn in different particle size ranges (Sansalone and Buchberger, 1997 – summarized in Li et al., 2005).

Size range μm	Cu mg/kg	Pb mg/kg	Zn mg/kg
25-38	364	265	1189
38-45	353	236	996
45-63	364	266	1027
63-75	333	258	1057
75-150	312	248	1014
150-250	204	195	574
250-425	78	65	325
425-850	48	53	314
850-2000	45	37	259

Table 4-10. Concentrations of Cu, Pb and Zn in different particle size ranges (Caltrans, 2002 – summarized in Li et al., 2005).

Size μm	Cu mg/kg	Pb mg/kg	Zn mg/kg
<20	37.4	21.2	263
20-300	33.7	21.4	311
300-850	30.1	22.5	231
850-2000	15.2	16.6	121
>2000	25.7	19	262

Several conclusions can be made on these heavy metal PSD studies. Firstly the classical variation with particle size does not always hold in high energy zones close to sources areas where source characteristics dominate rather than transport processes, although the highest concentrations are usually found in small particles. Secondly there is a great deal of variation in the concentration – PSD relationship between studies. Thirdly, the variation in heavy metal concentrations as the particle sizes gradation increase through the 62.5 μm size is small; e.g., particles in the fine sand size (~ 62.5 -250 μm) usually have similar concentrations to particles in the silt range (~ 16 -62.5 μm).

There is less information on organic pollutants. There is some information on the hydrophobic organic pollutant PAH, which shares some similarities to PCBs in terms of binding characteristics but differs in source characteristics. The increase in concentration with decreasing particle size is less true for PAH than for metals. Some studies show a general increase with particle size or highest concentrations in a particular particle size range >62.5 μm (Williamson and Mills, 2002). This may also be due to large source particles (e.g., pieces of coal tar used as base material in older bitumen roads). It could also be due to the preferential dissolution of PAH into larger discrete organic particles (“super-sorbents”) such as char, coal, kerogen, coke. This phenomenon occurs with PCBs as well, and is described more fully in Section 4.1. Krein and Schorer (2000) investigated heavy metals and PAHs in road runoff and found, as expected, an inverse relationship existed between particle size and particle-bound heavy metals concentration. However, particulate-bound PAHs were found to be bimodally distributed. Three-ring PAHs were mostly found in the fine sand fraction, while six-ring PAHs were mostly concentrated in the fine silt fraction.

Larger particles can assume greater prominence than smaller particles when mass loads are considered. The mass of pollutant transported in urban stormwater in a particular particle size range is a product the mass of sediment and the concentrations of the pollutant in that particle size range. Smaller particles may be the most important vehicles for mass transport at low flows, because larger particles are not being mobilized. However larger particles are more important if conditions are suitable to mobilize them (e.g., high flows, steep slopes), because they can have a far greater mass than the smaller particles. In other words, on some occasions, the relative mass of the particles is a more important factor than the relative concentration of the pollutants within the particles. An example of this is the study by Sansalone et al., 1997, who found the greater mass of pollution by trace metals (Hg or PCB were not measured) was in sediment-sized particles larger than the suspended fraction (Figure 4-4). These measurements were made on particles collected from highway runoff (described in Table 4-5). However, if the greater mass is in finer particles, then these will carry the greater amount of pollutants (e.g., Table 4-7 above)

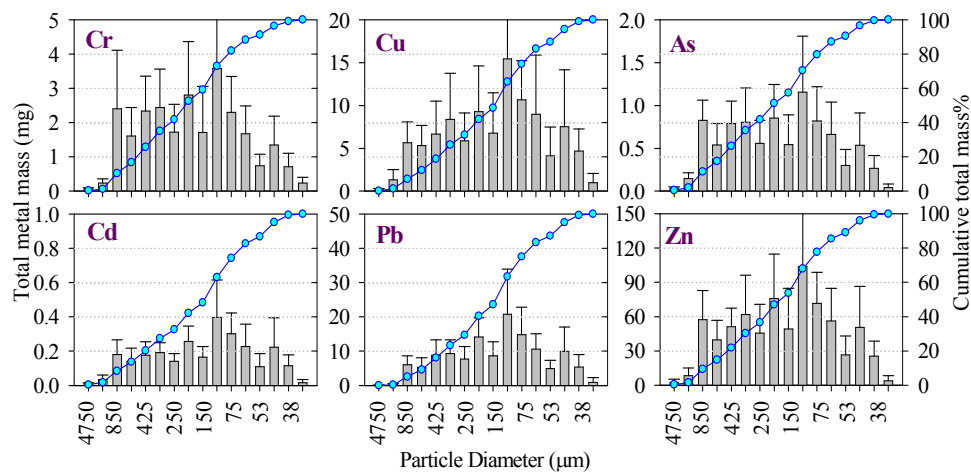


Figure 4-4. Distribution of metal mass across entire gradation in wet weather runoff. Bars represent the mean of 12 rainfall-runoff events and the range bars represent the standard deviation. Data from a Baton Rouge watershed comprised of 1,088 m² of urban transportation land use pavement (from Strecker et al., 2005).

In summary, while studies have demonstrated that highest concentrations of trace pollutants are often found in finer particles, there is a great deal of variation in the metal concentration – PSD relationship. Also, concentrations in fine sand-sized particles (> 62.5-250 µm) may be as high as found in silt-sized particles. This re-emphasizes the point that the authors of the urban sediment Hg and PCB study made when presenting their results normalized to the proportion of % fine (< 62.5 µm) sediments; that the survey results need to be treated with caution (KLI and EOA 2002). While concentrations

may be higher in finer particles, they may not represent the greatest mass. The greater mass may occur in larger particles in situations where larger particles dominate sediment loads, such as highway runoff. In addition, recent studies have found high concentrations and loads of some trace pollutants such as PAH in large particles, albeit with low densities and settling characteristics. Overall, there is no *a priori* way to predict the likely concentration – PSD relationship for Hg and PCBs, and this will need to be measured in representative watersheds draining to San Francisco Bay.

4.5. Transport of Hg in urban stormwater

Hg is predominantly adsorbed to particulate matter. Reported Log K_d range from about 3.7 to 6.6, depending on [POC], [DOC], [SS] and how dissolved Hg is measured and defined. When $\log K_d > 5$, particulate Hg always predominates, while with $\log K_d < 4$, the [SS] needs to exceed 100 mg/l for the particulate forms of mercury to predominate (McKee et al., 2003 and references cited therein).

Mercury is most strongly adsorbed by organic matter in soils and sediments, with humic acid possibly the dominant organic phase. The next strongest adsorption phase is hydrous ferric oxide (FeOOH) (Gabriel and Williamson, 2004). Horowitz (1995), however, on the basis of correlative evidence considers FeOOH to be a stronger adsorbing phase than organic matter in river sediment. Ionic mercury is also adsorbed by other mineral surfaces, such as on clays, although in natural systems these may well be coated with organic matter, FeOOH etc.

The PSD of Hg in urban stormwater runoff is not known either in general or in particular for the Bay area. Measures of Hg in street dust and urban soils tends to support the paradigm of increasing concentration with decreasing particle size (See Section 4.3) and this will probably be the case of Hg in stormwater particles. However, given the uncertainties, it will be important that this study directly measures the physical properties (e.g., particle size, settling characteristics) of the particles that transport Hg.

Inorganic mercury can be transformed to methyl mercury in the urban drainage network. Methylation is the process by which Hg^{2+} is converted to organic methyl mercury species (CH_3Hg^+ and $(\text{CH}_3)_2\text{Hg}$) by sulfate reducing bacteria in anoxic environments (Jones and Slotten, 1996; Alpers and Hunerlach, 2000). Organic forms such as monomethyl mercury (MMHg) and di-methyl-mercury (DMHg) are more easily taken up by organisms and stored in their tissues. Therefore the methylation process strongly impacts the effect of mercury in the environment. Methylation often occurs in wetland areas on the margins of estuaries where there is a source of sulfate from seawater, an abundance of organic carbon, and a low or fluctuating concentration of dissolved oxygen. This is because methylation is dependent on environmental factors such as dissolved oxygen, dissolved inorganic carbon, temperature, salinity, pH, redox, and the forms and concentrations of sulfur and mercury (Jones and Slotten, 1996; Alpers and Hunerlach, 2000).

4.6. Transport of PCB in urban stormwater

4.6.1. Speciation of PCB

The chemical and physical properties of individual PCB congeners vary according to the extent of chlorination and arrangement of chlorine atoms around the molecule. The hydrophobic nature of PCBs gives them characteristic properties of low water solubility, and a relatively high octanol-water coefficient (KOW) indicative of preferential sorption to organic matter. PCB congeners with higher numbers of chlorine atoms are less water soluble, less volatile, and have higher affinities for sorption to organic phases (*i.e.*, higher K_{ow}) compared to less-chlorinated PCBs. Therefore, highly chlorinated PCB residues have a greater tendency to partition into organic matter, persist in soil and sediment in the environment, and bioaccumulate in lipids of wildlife and humans (McKee et al., 2003). Their resistance to biological breakdown and low vapor pressure mean that they persist in the environment a relatively long time.

In tributaries and storm drains of watersheds polluted by PCBs, mobilization of PCB residues by erosion and leaching of particulate material is often the dominant transport mechanism (McKee et al., 2003). Several studies have determined that significant correlations exist between PCB concentrations and POC, suspended particulate matter (SPM), and total suspended solids (TSS) (Steuer et al., 1999a, 1999b). For example, particulate PCB concentrations in water samples collected during flood-flow conditions from the tidal reaches of the Guadalupe River and Coyote Creek which drain into the Lower South San Francisco Bay, comprised approximately $87 \pm 2.3\%$ and $90 \pm 6.4\%$ of total PCB concentrations measured, respectively (SFEI Annual Results, e.g., SFEI, 2002). Furthermore, samples from these locations have PCB congener patterns indicative of Aroclor 1260 (Leatherbarrow et al., 2002), which sorbs to particulate phases more readily than lower-molecular weight Aroclors.

In contrast to the expected preferential sorption of PCBs to particulate phases, several studies have measured higher proportions in the dissolved fraction in water samples with low suspended particulate concentrations (Chevreuil et al., 1990; Marti and Armstrong, 1990) and low organic carbon content (Jiang et al., 2000). This is not anticipated in San Francisco watersheds because they have:

- high [SS], typically 374-4472 mg/L
- stream and drain and Bay sediments sampled throughout the Bay area have congener patterns of high molecular weight Aroclors, 1254, 1260.

Studies at the Guadalupe River (McKee et al., 2004) have led to hypotheses on hydrologic processes mobilizing and transporting of chlorinated hydrocarbon residues from various sources within the watershed. First, greater concentrations on the rising stages of floods relative to falling stages suggest that runoff from urban areas is more polluted than runoff from the non-urban upper watershed. Second, a first flush phenomenon transported material from a unique source of low molecular weight PCBs (tri- and tetra- chlorinated) during the first storms of the season – possibly of atmospheric

origin. Third, base flow conditions transported material that originated from within the stream channel – dominated by pent and hexa-chlorinated PCBs, whereas higher flows were dominated by less weathered particulate material (dominated by tetra and penta-chlorinated PCBs) that probably originated from terrestrial watershed sources stored further from the stream channel.

The PSD of PCB in urban stormwater runoff is not known either in general or in particular for the Bay area. Measures of PCB in urban soils tends to support the paradigm of increasing concentration with decreasing particle size (See Section 2) and this may be the case of PCB in stormwater particles in most urban catchments. However, black carbon and other forms of glassy carbon (e.g., coal dust) may be important sediment phases for PCBs in some watersheds. It is important to consider these in the transport processes of PCBs to the Bay. They are not part of the fine sediment fraction, although their low density may mean that they are part of the suspended load and that they will be mobilized and transported readily. Their low density may also mean that they are difficult to remove in BMPs that rely on settling (Table 4-6). These larger particles will not be included in any separated mud ($<62.5\ \mu\text{m}$) fraction, if this is to be collected for separate analysis. Given these unknowns and uncertainties, it will be important that this study directly measures the physical properties (e.g., particle size, settling characteristics) of the particles that transport PCB.

4.7 Other pollutants

Other pollutants of concern for San Francisco Bay have been summarized in Section 1 (Table 1-4) in terms of their priority for attention. Many of these, e.g., PBDE, pyrethroid pesticides, Cu, organochlorine pesticides, other trace metals, dioxan/furans, PAH and organochlorine pesticides will be associated with particulate matter. A few pollutants of concern (notably Se and some endocrine disruptors) will mostly be in the dissolved phase.

4.8 Likelihood of various sources entering Bay

4.8.1. The conceptual model for particulate transport

When considering impacts on the Bay, we would normally or intuitively only consider the suspended and dissolved fractions and assume that the larger particles are mostly trapped in the urban drainage network, especially if there is a decrease in gradient downstream. These particles might be flushed out to the Bay in large storm events (very infrequent flood events).

However, it is important to consider the whole particle size gradation in this project from several points of view:

1. The PSD of Hg and PCB is not known for the San Francisco Bay (SFB) watersheds. Larger particles may also be polluted with Hg and PCBs.

2. The purpose of this White Paper is to summarize expectation using knowledge of urban stormwater quality and come up with conceptual models and understanding of unit processes to underpin BMP selection. This knowledge on urban stormwater includes the SS PSD controversy. The major uncertainties about the measurement of suspended sediment in stormwater described above occur because of uncertainty whether TSS protocols and automatic samplers are representatively sampling the particle size distribution. These protocols and methods are now being questioned, particularly in relation to the TMDL process, and the setting of Waste Loads Allocations (James, 2002).
3. Soil hot spots of Hg and PCBs will comprise a wide particle size distribution (PSD) and may be dominated by particles $>75\text{ }\mu\text{m}$. Additionally, the greater mass of particles on roads is in the larger sizes ($>75\text{ }\mu\text{m}$), and suspended sediment-sized or less particles are typically only a few to 20% of the total mass.
4. Most of the mass transported initially by runoff from impervious surfaces to the storm drainage system can be in the larger particle sizes (Figure 4-5).
5. Some of the BMPs or pollutant removal strategies being considered in this project trap larger particles more efficiently and these form the bulk of the trapped material. Examples are catch basins, removal of sediment from urban streams and other stormwater conveyance systems, and road sweeping.
6. The semi-quantitative models we are developing for the SFB are mass-balance based. At the outset, we need to take into account all sources, sinks and routes. Those that are insignificant can be discounted later.

The apparent controversy on the size of particles mobilized in urban stormwater runoff cannot be resolved in this project. However, the controversy can be accommodated into a conceptual model and the various processes quantified. In terms of the main elements of the controversy:

- whether or not large particles are an important component of the load;
- whether most protocols are representatively sampling particle size distribution;

Overall, the following picture emerges, and can be put forward as a working hypothesis (Table 4-11; Figure 4-5). Particles larger than commonly measured may well be mobilized in some instances. However, this is probably associated with impervious areas with relatively high energy (high drainage efficiency, turbulence). It is also possible that in other areas with lower energy, relatively few larger particles are mobilized. In addition, it is plausible that some studies may have not representatively sampled the particle size gradation. However, the natural processes in the drainage network mimic to some extent the problems associated with automatic samplers and TSS methods. Natural processes will attenuate or remove larger particles, and the smaller particles (settleable, suspended and colloidal) will dominate the discharge to the Bay.

4.8.2. Conceptual underpinning of sediment delivery ratios

Table 4-11 Conceptual model of particle mobilization and transport.

Step	Location	Predicted median particle size ¹
A	Polluted sites will be dominated by the particle size distribution of local soils.	50-500 µm
B	Build-up of particulate matter on some of the more important impervious surfaces (roads, pavements, parking lots, industrial yards) is dominated by the larger particles (sediment, grit).	>500 µm
C	Runoff is dominated by these larger particles on roads and highways.	200-500 µm
	Smaller particles may dominate runoff in other area with smaller slopes and less traffic.	50-100 µm
D	Larger particles settle in traps (e.g., catch basins, sediment forebays) or when flow velocities decrease significantly (e.g., decrease in slope).	50-100 µm in overflow
	Bed load may still be significant portion of sediment and pollutant load.	>500 µm in bedload
E	Larger particle continue to be trapped in the stormwater drainage system if slope continue to decrease (this is expected given the morphology of the bay watersheds). This is aided by any influx of particulate matter from upland open and rural areas which help exceed the transport capacity.	5-75 µm
	Bedload transported slowly and accumulates in drainage.	
		> 500 µm
F	Smaller suspended and settleable particles are discharged to Bay.	10-75 µm
	Settleable settle near discharge to tidal waters	25-75 µm
	Suspended and colloidal more widely dispersed (some flocculates)	<1-25 µm
	Bedload also discharges to tidal creek and wetland headwaters.	> 500 µm

¹Nominal particle sizes based on inorganic particle density

One of the aims of this study is estimate how much pollutant enters the Bay once it enters the urban stormwater drainage system. In the large complex system such as the San Francisco Bay watersheds the study is attempting to deal with the quantitative estimate of delivery of Hg and PCB from the ultimate sources (e.g., mercury in lamps, PCB in soils), their attenuation during transport within the whole urban drainage system, at the source, mobilization, transport and trapping (either intentional removal (e.g, structural BMPs) or sediment processes (being trapped in channel storage) and their delivery to the Bay.

When considering the efficiency of transport of Hg and PCB from sources to the Bay, it is useful to utilize the sediment delivery ratio (SDR) concept. For the Bay urban areas, two different types of sediment sources need to considered:

1. Sediment transport from upland areas which are mostly non-urban. These areas would tend to have low Hg (except in mining areas) and low PCB concentrations. Sediment supply may be augmented from soil erosion on urban construction sites, because most present day urbanization is occurring in these upland areas. Major sediment sources are earth movements and there is a great deal of exchange of material between streams and their beds and banks.
2. Sediment transport from the lowland urban areas. These areas tend to have higher Hg and – in some places - high PCBs. Significant sources are sediments eroded in overland flow on roads and other impervious surfaces.

This introduces an additional complexity to the difficulty in estimating single sediment delivery ratios described in section 3.3 above.

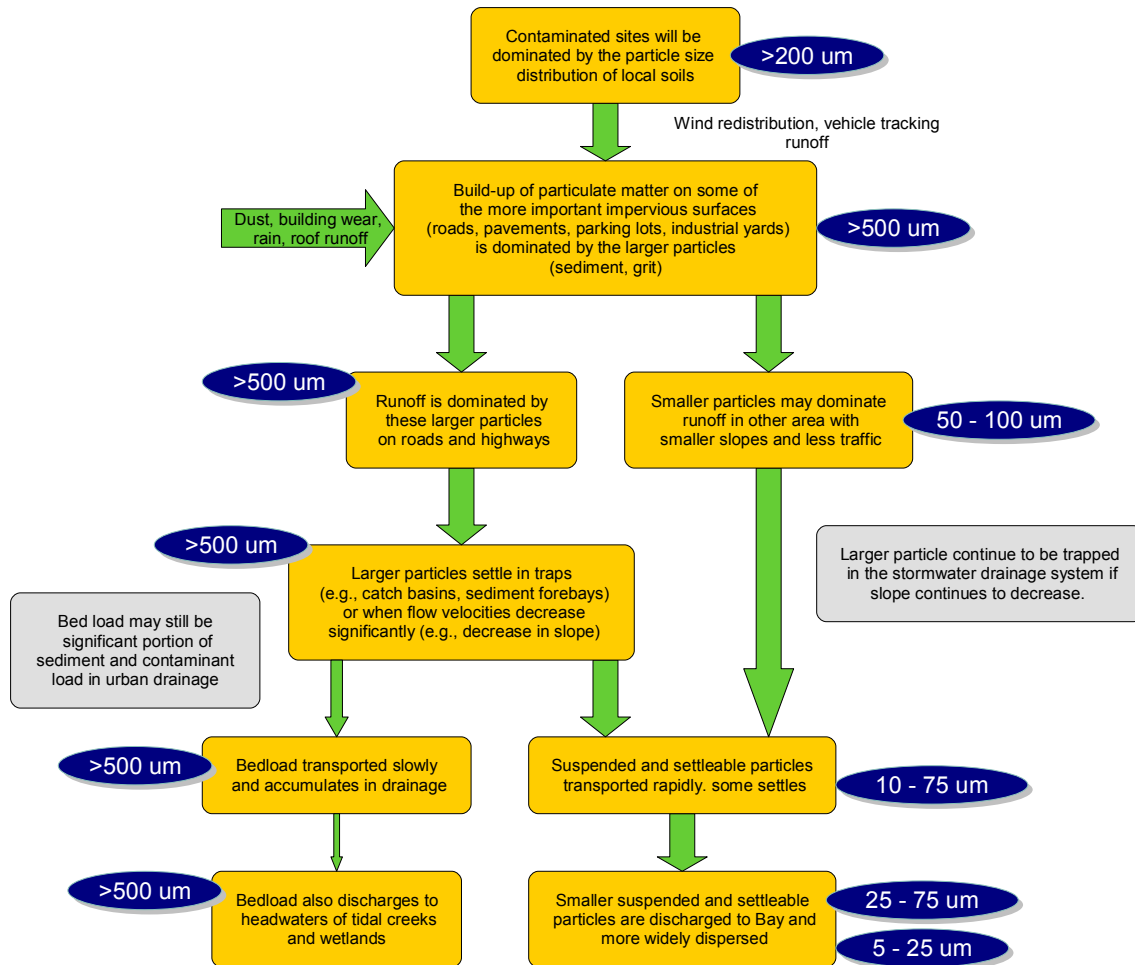


Figure 4-5. Conceptual model of SS transport

When considering systems that are dominated by upland sources, the SDR will be close to those for natural systems. However, we are mainly interested in urban-dominated watersheds or sub-catchments because the issue being addressed here is the transport of sediment polluted with Hg and PCBs which are largely sourced in these areas. As described in section 3.3, the SDR are likely to be relatively high. A major task of the quantitative model development will be the derivation and selection of these SDRs.

4.9. Summary and information gaps in mobilization and transport of Hg and PCB

The forgoing review reveals that a great deal of information exists about the physical and chemical nature of particulate matter in urban stormwater. This review has summarized

this information to underpin the study and provide an understanding of the processes transporting Hg and PCBs to the Bay, and the processes leading to removing this sediment during stormwater drainage maintenance or through placement of structural BMPs. The review provides a qualitative model for sediment transport, and this will form the basis for the quantitative assessment of transport and removal of Hg and PCBs in the San Francisco Bay urban drainage system.

The forgoing (re)emphasizes the highly variable nature of urban stormwater. There is little information to predict the nature of association of Hg and PCBs with particulate characteristics. A major conclusion is that it will be necessary to obtain local data on the physical characteristics of the particulate matter carrying Hg and PCB, and hence its transport and treatability characteristics. The information summarized in this review will provide a context and check on monitoring data gathered in this project. The key missing information is summarized in Table 4-12.

Table 4-12. Information gaps/monitoring requirements.

Information gap	Reason
Hg and PCB distributions across particle size gradation	Understand transport and attenuation of Hg and PCBs
[POC]	Predict treatability of Hg and PCBs
	Help explain variation in PSD, and sources
	Importance of glassy carbon
	Help predict treatability of PCBs
[FeOOH]	Help explain variation in PSD for Hg
PSD of SS, Hg, PCB and POC in urban stormwater as opposed to settled sediment	“Ground truth” findings from sediment analysis
PSD of SS, Hg, PCB and POC variation across source areas and land use	Relative importance if land use on loads and speciation
PSD of SS, Hg, PCB and POC variation down the stream continuum	Measurement of any transformations in the drainage network (and their implications for BMP choice)

4.10. References

- Abu-Saba, K., Grieb, T., Looker, R.E., McCord, S. (2005 – draft only). A conceptual model for mercury in San Francisco Bay. A collaborative project of the Clean Estuary Partnership.
- Allen-King, R. M.; Grathwohl, P.; Ball, W. P. 2002. New modeling paradigms for the sorption of hydrophobic organic chemicals to heterogeneous carbonaceous matter in soils, sediments, and rocks. *Adv. Water Resour.* 2002, 25, 985-1016.
- Alpers, C.N., and Hunerlach, M.P., 2000. Mercury contamination from historic gold mining in California. U.S. Geological Survey Fact Sheet FS-061-00. 5pp.
- Andral, M.C., Roger, S., Montrejaud-Vignoles, M., Herremans, L. 1999. Particle Size Distribution and Hydrodynamic Characteristics of Solid Matter Carried by Runoff from Motorways, *Water Environ. Res.*, 71, 4, 398.
- Ankley, G. T., Berry, W. J., DiToro, D. M., Hansen, D. J., Hoke, R. A., Mount, D. R., Reiley, M. C., Swartz, R. C., and Zarba, C. S. 1996. Use of equilibrium partitioning to

- establish sediment quality criteria for nonionic chemical: A reply to Iannuzzi et al., *Environ. Toxicol. Chem.* 15(7), 1019-1024.
- Auckland Regional Council 2004. Management and treatment of stormwater quality. Auckland Regional Council Technical Publication 237
- Ball, J.E., and Abustan, I. 2000. Modelling the Export of Phosphorous from Urban Catchments. *Australian J. Water Resour.* 4, 33.
- Roger Bannerman, Greg Fries, Judy Horwath 2003. "Source Area and Regional Storm Water Treatment Practices: Options for Achieving Phase II Retrofit Requirements in Wisconsin." Proceedings of the National Conference on Urban Storm Water: Enhancing Programs at the Local Level Chicago, IL February 17-20, 2003.
- Benjamin MM, Leckie JO. 1978. Competitive adsorption of Cd, Cu, Zn and Pb on amorphous iron oxyhydroxide. *J Colloid Interface Sci* 83:410-419.
- Bent, Gardner C., John R. Gray, Kirk P. Smith, G. Douglas Glysson, 2001. A Synopsis of Technical Issues for Monitoring Sediment in Highway and Urban Runoff, USGS, OFR 00-497.
- Burton, G.A. Jr and R. Pitt (2002), *Stormwater Effects Handbook: A Tool Box for Watershed Managers; Scientists and Engineers*, CRC Press, Inc., Boca Raton, FL.
- Butler, D., May, R.W.O., and Ackers, J.C., 1996a, Sediment transport in sewers Part 1—Background: Proceedings of the Institution of Civil Engineers-Water Maritime and Energy, v. 118, no. 2, p. 103–112.
- Caltrans (2002). Caltrans Tahoe highway runoff characterization and sand trap effectiveness studies-2001-2002 monitoring season. CTSW-RT-02-044, California Department of Transportation.
- Chevreuil, M., L. Granier, A. Chesterikoff, and R. Letolle. 1990. Polychlorinated biphenyls partitioning in waters from river, filtration plant and wastewater plant: the case for Paris (France). *Water Research.* 24 (11). pp. 1325-1333.
- Clarifa Inc, 2003. Preliminary assessment for improved design criteria for construction sediment control ponds. Report prepared for Toronto City.
- Clarke, S., Pitt, R., Field, R., Fan, E., Heaney, J., Wright, L., Burian, S. 2003. Annotated bibliography of urban wet weather flow literature from 1996 through 2002. <http://unix.eng.ua.edu/~rpitt/Publications/Wetweatherlit>
- Collins, L.M., 2001. Wildcat Creek watershed: a scientific study of physical processes and land use effects. San Francisco Estuary Institute. 85 pp.
- Corsi, R.; Greb, S.R.; Bannerman, R.T.; and Pitt, R.E. 1999. Evaluation of the Multi-Chambered Treatment Train, a Retrofit Water-Quality Management Device. U.S. Geological Survey Open-File Report 99-270. Middleton, WI.
- Davis, J.A., McKee, L.J., Leatherbarrow, J.E., and Daum, T.H., 2000. Contaminant loads from stormwater to coastal waters in the San Francisco Bay region: Comparison to other pathways and recommended approach for future evaluation. San Francisco Estuary Institute, September 2000. 77pp.
- Drapper, D., Tomlinson, R., and Williams, P. 2000. Pollutant concentrations in road runoff: South Queensland Case Study. *Journal Environmental Engineering*, 126-313??
- Driscoll, E., Methodology for Analysis of Detention Basins for Control of Urban Runoff Quality, United States Office of Water Environmental Protection Nonpoint Source Branch Agency, Washington, DC. EPA440/5-87-001, 1986

- Dzombak DA, Morel FMM. 1990. Surface Complexation Models. Hydrous Ferric Oxide. John Wiley and Sons, London.
- Engstrom, A. M. 2004. Characterizing water quality of urban stormwater runoff: Interactions of heavy metals and solids in Seattle Residential Catchments. M. Sc. Thesis, (Civil Engineering), University of Washington.
- Gabriel, M.C., Williamson, D.G. 2004. Principal biogeochemical factors affecting the speciation and transport of mercury through the terrestrial environment. *Environmental Geochemistry and Health* 26: 421–434.
- Gartner J.W., Cheng R.T., Wang P.-F., Richter K. 2001. Laboratory and field evaluations of the LISST-100 instrument for suspended particle size determinations. *Marine Geology* 175, 199-219.
- Ghosh, U.; Zimmerman, J. R.; Luthy, R. G. 2003. PCB and PAH speciation among particle types in contaminated harbor sediments and effects on PAH bioavailability. *Environ. Sci. Technol.*, 37, 2209-2217.
- Gray, John R., G. Douglas Glysson, Lisa M. Turcios and Gregory E. Sellwarz, 2000. Comparability of Suspended-Sediment Concentration and Total Suspended Solids Data, USGS WRIR 00-4191
- Green, M.O. & Bell, R.G. 1995. Wave influence on suspended-sediment fluxes in an estuary (Manukau Harbour, New Zealand). In *Proceedings 12th Australasian Conference on Coastal and Ocean Engineering*, Melbourne. National Conference Publication. No. 95/5, Institution of Engineers, Canberra, p 59-64.
- Green, M. Williamson, R.B. 2001. Prediction of contaminant accumulation in estuaries. Auckland Regional Council Technical Report No. 163.
- Green, M.O., Williamson, R.B., Timperley, M., Collins, R., Senior, A., Adams, A., Swales, A. and Mills, G., 2004. Prediction of Contaminant Accumulation in the Upper Waitemata Harbour – Methods. Auckland Regional Council Technical Report TP 261, 97 pp.
- Griggs, G.B., and Paris, L., 1982. Flood control failure: San Lorenzo River, California. *Environmental Management*, v. 6 (5), p. 407-419.
- Gunther, A., Salop, P., Bell, D., Feng, A., Wiegel, J., and R. Wood, 2001. Initial Characterization of PCB, Mercury, and PAH Contamination in the Drainages of Western Alameda County, CA. Prepared for the Alameda Countywide Clean Water Program. Hayward, CA.
- Gustafsson, O., Gschwend P.M. 1997. Soot as a strong partition medium for polycyclic aromatic hydrocarbons in aquatic systems. In: Eganhouse RP, editor. *Molecular markers in environmental geochemistry*. Washington: American Chemical Society.
- Hetzel, F. 2004. PCBs in San Francisco Bay: Total Maximum Daily Loads Report. San Francisco Bay Regional Water Quality Control Board. Oakland, CA.
- Horowitz, A.J. and Elrick, K.A. 1987. The relation of stream sediment surface area, grain size and composition to trace element chemistry. *Applied Geochemistry* 2, 437-451.
- Horowitz, A.J. 1991. A primer on sediment-trace element chemistry, 2nd Edition. USGS Open File Report 91-76.
- Horowitz, A.J. 1995. The use of suspended sediment and associated trace elements in water quality studies. USGS IAHS Special Publication, Open File Report 91-76.

- Hume, T. M. & McGlone M. S. 1986. Sedimentation patterns and catchment use changes recorded in the sediments of a shallow tidal creek, Lucas Creek, Upper Waitemata Harbour, New Zealand. *NZ Journal of Marine and Freshwater Research* 20: 677–687.
- Jacopin, C.; Bertrand-Krajewski, J.L.; and Desbordes, M. (1999a) Characterisation and Settling of Solids in an Open, Grassed, Stormwater Sewer Network Detention Basin. *Water Sci. Technol. (G.B.)*, 39, 2, 135.
- James, R.B. 2002. Measurement and BMP removal of suspended material in stormwater runoff. National TMDL Science and Policy 2002 Conference, November 13-16, Phoenix, AZ.
- Jiang, X., D. Martens, K.W. Schramm, A. Kettrup, S.F. Xu, and L.S. Wang. 2000. Polychlorinated organic compounds (PCOCs) in waters, suspended solids, and sediments of the Yangtse River. *Chemosphere*. 41 (6). pp. 901-905.
- Johnson, Pauline, R. Pitt, S.R. Durrans, M. Urrutia, and S. Clark, 2003. Metals Removal Technologies for Urban Stormwater, Water Environment Research Federation.
- Johnson W., and Looker R., 2003. Mercury in San Francisco Bay: Total Maximum Daily Loads Report. California Regional Water Quality Control Board San Francisco Bay Region. June 6th 2003.
- Jones, A.B., and Slotton, D.G., 1996. Mercury effects, sources, and control measures. A special study of the San Francisco Estuary Regional Monitoring Program. San Francisco Estuary Institute, Oakland, Ca. September 1996. 16pp.
- Kayhanian, M., Young, M.T., and Stenstrom, M.K. 2005. Limitation of current solid measurements in stormwater runoff. StormCon, The North American Surface Water Quality Conference & Exposition, Orlando, FL, USA, July 18-21, 2005.
- Karickhoff, S. W. (1981). "Semi-empirical estimation of sorption of hydrophobic pollutants on natural sediments and soils," *Chemosphere* 10, 833-846.
- Karickhoff, S. W., Brown, D. S., and Scott, T. A. (1979). "Sorption of hydrophobic pollutants on natural sediments," *Water Res.* 13, 241-248.
- KLI and EOA (Kinnetic Laboratories, Inc. and Eisenberg, Olivieri, and Associates) 2002. Joint Stormwater Agency Project to Study Urban Sources of Mercury, PCBs and Organochlorine Pesticides. Prepared for SCVURPPP, CCCWP, SMCSPPP, MCSPPP, VFCSD, and FSSD. April 2002.
- Kolka, R.K. Grigal, D. F. , Verry, E. S. and Nater, E. A. (1999). Mercury and Organic Carbon Relationships in Streams Draining Forested Upland/Peatland Watersheds. *Journal of Environmental Quality* 28, 766-75.
- Kobriger, N.P. 1984. Volume I. Sources and Migration of Highway Runoff Pollutants. FHWA/RD-84/057. Federal Highway Administration, Rexnord, EnviroEnergy Technology Center, Milwaukee, WI.
- Krein, A. and Schorer, M. 2000. Road runoff pollution by polycyclic aromatic hydrocarbons and its contribution to river sediments. *Water research* 34 (16) 4110–4115.
- Krishnappan, B.G.; Marsalek, J.; Watt, W.E.; and Anderson, B.C. (1999) Seasonal Size Distributions of Suspended Solids in a Stormwater Management Pond. *Water Sci. Technol. (G.B.)*, 39, 2, 127.
- Leatherbarrow, J.E., R. Hoenicke, and L.J. McKee. 2002. Results of the Estuary Interface Pilot Study, 1996-1999. RMP Technical Report. SFEI Contribution XX. San Francisco Estuary Regional Monitoring Program. San Francisco Estuary Institute. Oakland, CA.

- Lee, J.H.; Bang, K.W.; "Characterization of Urban Stormwater Runoff." *Water Resources (G.B.)*, Vol. 34 No. 6 (2000) p. 1772-1780.
- Li, Y., Kayhanian, M., Lau S., and Stenstrom, M.K. 2005. Particle-based pollutant removal BMPs. StormCon, The North American Surface Water Quality Conference & Exposition, Orlando, FL, USA, July 18-21, 2005.
- Li, Y., Lau S., Kayhanian, M., and Stenstrom, M.K. (in press). Particle Size Distribution in Highway Runoff. *Journal of Environmental Engineering*, ASCE.
- Lin, H., Kim, J.-Y., Ma, J., Sansalone, J.J., 2004a. Separation unit operations for non-colloidal particulate matter in rainfall-runoff. StormCon, The North American Surface Water Quality Conference & Exposition, Palm Desert, CA, USA, July 26-29, 2004.
- Lin, H., Ying, G., Sansalone, J.J., 2004b. Settling behaviour of non-colloidal particulate matter in rainfall-runoff. StormCon, The North American Surface Water Quality Conference & Exposition, Palm Desert, CA, USA, July 26-29, 2004.
- Lin Hong 2003. Granulometry of non-colloidal particulate matter transported by urban rainfall-runoff. Ph. D Thesis.
- Long, E. R.; MacDonald, D. D.; Smith, S. L.; Calder, F.D. (1995). Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management*. 19, 81-97.
- McKee, L., Leatherbarrow, J., Newland, S., and Davis, J., 2003. A review of urban runoff processes in the Bay Area: Existing knowledge, conceptual models, and monitoring recommendations. A report prepared for the RMP Sources, Pathways and Loading Workgroup. San Francisco Estuary Regional Monitoring Program for Trace Substances. SFEI Contribution Number 66. San Francisco Estuary Institute, Oakland, Ca.
- McKee, L., Leatherbarrow, J., Eads, R., 2004. Concentration and loads of mercury, PCBs, and OC pesticides associated with suspended sediments in the lower Guadalupe River, San Jose, California. A technical report of the Regional Watershed Program: SFEI Contribution # 66. San Francisco Estuary Regional Monitoring Program for Trace Substances, San Francisco Estuary Institute, Oakland, Ca.
- Marti, E.A. and D.E. Armstrong. 1990. Polychlorinated biphenyls in Lake Michigan tributaries. *Journal of Great Lakes Research*. 16 (3). pp. 396-405.
- Mason, R.P., and Sullivan, K.A., 1998. Mercury and methyl mercury transport through an urban watershed. *Water Resources* 32, 321-330.
- Mason, R.P. and Lawrence, A.L. (1998). Concentration, distribution and bioavailability of mercury and methyl mercury in sediments of Baltimore Harbor and Cheseapeake Bay, Maryland, USA. *Environmental Toxicology and Chemistry* 18, 2438-2447.
- Metcalf & Eddie 2004. *Wastewater Engineering. Treatment and Reuse*. Revised by Tchobanoglous, G., Burton, F.L., Stensel, H.D. McGraw Hill, San Fransisco.
- Novotny, V., and Chesters, G., 1989. Delivery of sediment and pollutants from nonpoint sources: A water quality perspective. *Journal of Soil and Water Conservation*. p. 568-576.
- Pitt, R. 1996. Accumulation, washoff and size distributions of stormwater particulates. In "Solids in Sewers: Characteristics, effects and Controls of sewer solids and associated pollutants". Scientific and Technical Report of the International Association on Water Quality (IAWQ). London. 1996.

- Pitt, R., Field, R., Lalor, M., and Brown, M. 1995. Urban stormwater toxic pollutants: Assessments, sources and treatability. *Water Environment Research*, 67(3), 260-275.
- Pitt, R. A. Maestre and R. Morquecho (2004). "Nationwide MS4 Stormwater Phase I Database." Watershed 2004, Dearborn, MI, July 2004, WEF, Alexandria, VA. [Online] <http://unix.eng.ua.edu/~rpitt/Research/ms4/Paper/recentpaper.htm>
- Postma, H., (1967). Sediment transport and sedimentation in the estuarine environment. In: Lauff, G. M. (Edr.), *Estuaries*. AAAS Publ. 83, 158-179.
- Prahl, F. G.; Carpenter, R. PAH-phase associations in Washington coastal sediment. *Geochim. Cosmochim. Acta* 1983, 47, 1013-1023.
- Randall, C.W., Ellis, K., Grizzard, T.J., Knoke, W.R. 1982. Urban runoff pollutant removal by sedimentation. In: *Proceedings of the Conference on Stormwater Detention Facilities, Planning, Design, Operations, and Maintenance*, Henniker, NH, ed. by W DeGroot. American Society of Civil Engineers, New York.
- Salop, P., Abu-Saba, K., Gunther, A., and Feng, A., 2002a. 2000-01 Alameda County Watershed Sediment Sampling Program: Two-Year Summary and Analysis. Prepared for the Alameda Countywide Clean Water Program, Hayward, CA.
- Sansalone, J.J. and Buchberger, S.G. (1997). "Partitioning and First Flush of Metals and Solids in Urban Highway Runoff." *J. Environmental Engineering, ASCE*, 123(2):134-143.
- Sansalone, J.J., Koran, J., Smithson, J.M., and Buchberger, S.G. (1998). "Physical Characteristics of Urban Roadway Solids Transported during Rain Event." *J. Environmental Engineering, ASCE*, 124(5):427-440.
- Socha, S. B.; Carpenter, R. Factors affecting pore water hydrocarbon concentrations in Puget Sound sediments. *Geochim. Cosmochim. Acta* 1987, 51, 1273-1284.
- Steuer, J.S., S.A. Fitzgerald, and D.W. Hall. 1999a. Distribution and transport of polychlorinated and associated particulates in the Milwaukee River system, Wisconsin, 1993-95. 1999a. U.S. Geological Survey. Water-Resources Investigations Report 99-4100. Prepared in cooperation with the Wisconsin Department of Natural Resources and the Milwaukee Metropolitan Sewage District. Middleton, WI.
- Steuer, J.S., D.W. Hall, and S.A. Fitzgerald. 1999b. Distribution and transport of polychlorinated biphenyls and associated particulates in the Hayton Millpond, South Branch Manitowoc River, 1993-1995. U.S. Geological Survey. Water-Resources Investigation Report 99-4101. Prepared in cooperation with the Wisconsin Department of Natural Resources. Middleton, WI.
- Strecker, E., Huber, W., Heaney, J., Bodine, D., Sansalone, J., Quigley, M., Leisenring, M., Pankani, D., and Thayumanavan, A. (2005). "Critical Assessment of Stormwater Treatment and Control Selection Issues." Prepared for the Water Environment Research Foundation, WERF 02-SW-1. [Draft accepted for publication]
- Swales, A.; Williamson, R.B., Van Dam, L. Stroud, M. 2003. Reconstruction of Urban Stormwater Contamination of an Estuary Using Catchment History and Sediment Dating Profiles. *Estuaries* 25, 43-56.
- Syvitski, J.P.M. (Ed), 1991. Principles, methods, and application of particle size analysis. Cambridge University Press, New York.
- Roger C. Sutherland and Seth L. Jelen (2002). "Quantifying the Optimum Urban Runoff Pollutant Load Reduction Associated with Various Street and Catchbasin Cleaning

- Practices." Global Solutions for Urban Drainage, Ninth International Conference on Urban Drainage, September 8-13, 2002.
- Thomas J. Martinelli, Rob Waschbusch, Roger Bannerman, and Anna Wisner (2002). "Pollutant Loadings to Stormwater Run-Off from Highways: The Impact of a Sweeping Program." Wisconsin Department of Transportation Division of Transportation Infrastructure Development Bureau of Highway Operations. WI-11-01.
- U.S. Environmental Protection Agency, 1993. Final Report of the Nationwide Urban Runoff Program, Water Planning Division, Washington, D.C.
- U.S. Environmental Protection Agency. (1993). "Sediment quality criteria," Federal Register, FR Doc. 94-1133, Robert Perciasepe, Assist. Admin. for Water, Washington, DC.
- Vignoles, M and Herremans, M. (1995) Metal pollution of sediments carried in runoff water in Toulouse city. In: Novatech 95, 2nd International Conference in Innovative Technologies in Urban Storm Drainage, May 30 – June 1, Lyon, France. pp611-614.
- Walling, D.E., and Woodford, J.C., 1993. Use of a field-based water elutriation system for monitoring the in situ particle size characteristics of fluvial suspended sediment. *Water Research* 27, 1413-1421.
- Williamson, R.B., Wilcock, R.J. 1994. The distribution and fate of contaminants in estuarine sediments: Recommendations for monitoring and environmental assessment. Auckland Regional Council Technical Report TP 47.
- Williamson, R.B. and Mills, G.N. 2002 Sediment Quality Guidelines for Auckland Estuaries. Contribution to the Regional Discharges Project, Auckland Regional Council.
- Williamson, R.B.; Morrissey, D.J. 2000. Stormwater contamination of urban estuaries. Predicting the build-up of heavy metals in sediments. *Estuaries* 23, 56-66.
- Youngblood, W. W.; Blumer, M. Polycyclic aromatic hydrocarbons in the environment-Homologous series in soils and recent marine sediments. *Geochim. Cosmochim. Acta* 1975, 39, 1303-1314.

5. REVIEW OF BEST MANAGEMENT PRACTICE CONTROL OPTIONS

5.1 INTRODUCTION

This section outlines the options for pollution prevention, source control and treatment Best Management Practices (BMPs) targeting Hg and PCBs in the San Francisco Bay area. The discussion of BMP options is presented within the context of the conceptual model for the source-to-Bay transport of pollutants (Figure 5-1). Specifically some control practices address product usage, disposal, and recycling (pollution prevention), some address the build up of pollutants on impervious source areas (source control) and some address removal of pollutants from the drainage network (treatment BMPs and maintenance practices).

Past work has established that Hg and PCBs are largely transported in the particulate phase (Looker and Johnson, 2001, McKee et al., 2003, 2004, Hetzel, 2004). With the paucity of data on Hg and PCB transport and control in urban stormwater systems, this report focuses on the treatability of particulate matter in the urban stormwater drainage network.

The review also provides a preliminary assessment of the potential relative effectiveness of the various options for reducing particle-associated loads of pollutants like PCBs and Hg to San Francisco Bay.

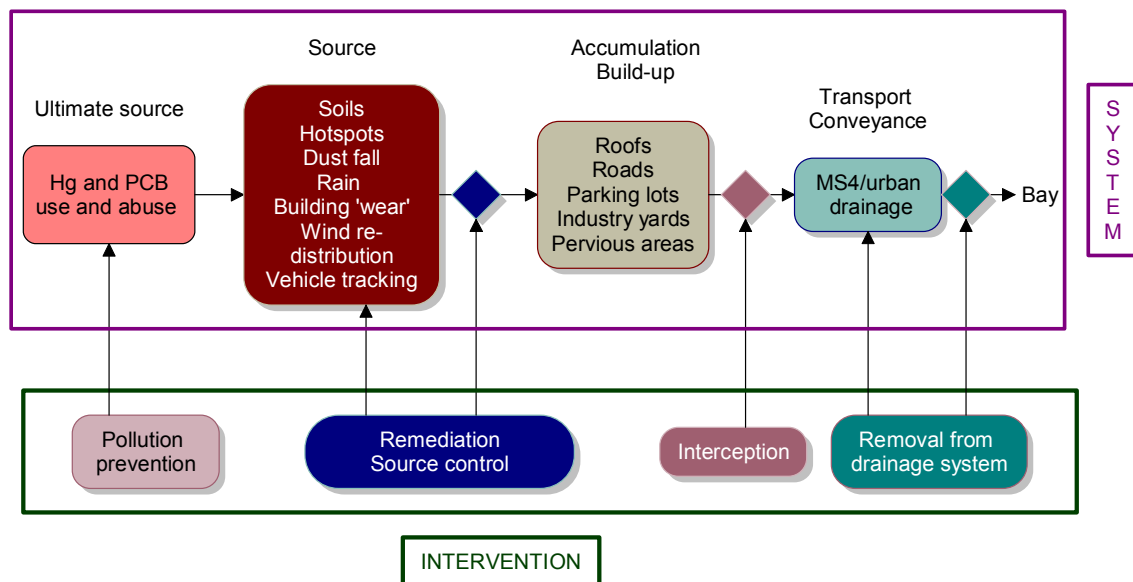


Figure 5-1. Overall conceptual model of Hg and PCB sources, mobilization and transport in the urban system; and opportunities for intervention and control.

5.2 POLLUTION PREVENTION

The goal of pollution prevention is to prevent materials from entering the environment and becoming pollutants. Ideally such controls would result in recycling and reuse of the materials, or replacement of the material with a more environmentally benign option (i.e., product substitution). For example, mercury is used in dental amalgams, fluorescent lamps, switches and thermostats, and thermometers (EIP, 2000). Pollution prevention is also applicable to PCBs, where PCBs are used in certain equipment (e.g., transformers) and in caulking materials used in building materials and other construction (EIP, 1997, Herrick et. al., 2004, Kohler et. al., 2005). Municipal stormwater and solid waste programs in the Bay Area are making an effort to inform the public regarding disposal and recycling opportunities, and have recycling programs to accept these materials.

5.2.1 Implication of Enhanced Pollution Prevention in Affecting PCB and Hg Loads to San Francisco Bay

Pollution Prevention Programs are effective in minimizing the introduction of products containing potential pollutants into the air, water, and soil environment. The effectiveness of these programs with respect to stormwater runoff is not easily evaluated given that it is difficult to determine to what extent such products would enter the storm drain system compared to other disposal options (e.g., landfills, sanitary sewer system). With respect to the PCB and Hg TMDLs, such programs have helped to establish (and limit) the baseline load condition. Future enhancements of these programs that target products that specifically are liable to enter the storm drain system (e.g., automotive switches containing Hg) could help reduce Hg loads to the Bay.

5.3 SOURCE CONTROL OPTIONS

5.3.1 Soil Remediation and Site Cleanup

Soil remediation in the context of this report refers to remediating soil pollution in areas where past practices have resulted in elevated concentrations of PCBs and Hg in soils that could potentially be mobilized during storm events and transported into the storm drain system. This control option also includes the identification and elimination of outdoor storage of Hg or PCBs that could potentially contribute to loads. Most of these sites are located in industrial areas or along industrial corridors (e.g., train corridors) where these materials were used in the past. Stormwater programs in the Bay Area are actively attempting to identify such sites (Larry Walker Associates, 2005, Salop and Akashah, 2004), and are conducting source identification case studies (Salop et. al., 2002; EOA, 2004). Where these case studies have indicated elevated concentrations of Hg and/or PCBs in sediments, the stormwater programs are referring these areas to the Water Board for enforcement and cleanup.

5.3.1.1 Implication of Enhanced Soil Remediation and Site Cleanup in Affecting PCB and Hg Loads to San Francisco Bay

Given the legacy of uses of PCBs and Hg, the cleanup of industrial sites represents removal of sources that may be responsible for much of the urban load currently entering the Bay (Note discussion of known mining sources in the Bay Area is outside this review). The scales of the cleanup will be the site scale corresponding to the historical area of usage. In principle, such cleanup represents the most long-term cost effective option compared to continued attempts at intercepting these pollutants in the downstream storm drain system. Assuming the site is cleaned up, the potential for areas of elevated sediment concentrations to contribute to loads to the Bay depends on the concentrations and extent of pollution, and the degree to which the polluted sediments can be mobilized and enter the storm drain system. Of course, the success of this option ultimately will depend on the extent to which identification, enforcement and cleanup of actual sites can be achieved. Overall, this option maybe potentially effective for long-term reduction in PCB and Hg loads to the Bay.

5.3.2 Street Sweeping

Street sweeping as a BMP for pollutant removal has been controversial for many years. Given all the material (including trash, vegetative debris, and sediments) that is removed by street sweepers, many people believe that street sweeping is also effective for removing pollutants. However, one of the definitive conclusions of the U.S. EPA-sponsored Nationwide Urban Runoff Program (NURP), which collected runoff water quality data from catchments having different sweeping regimes, found that street sweeping was generally an ineffective technique for improving the quality of urban runoff (EPA, 1983).

Street sweeping as an effective stormwater quality BMP has undergone a renaissance with the advent of more sophisticated cleaners (mechanical/vacuum cleaners), and a better understanding of the following controlling factors that can affect effectiveness.

- street texture – dirt pick up is more effective on smooth streets
- street loading – dirt pick up is more efficient at higher street loading
- large particle armoring – may prevent smaller particle removals
- moisture inhibits pick up
- wind/turbulence redistribution – sweeping should include the whole impervious area and not just the gutter
- dust and pollutant build up rates - can be much more rapid than street cleaning frequencies
- parking inhibits regular cleaning
- driver and device operation abilities are a big factor in efficiencies, especially in respect to speed and clogging

The following describes how some of these factors affect street sweeping performance.

Particle Size Effects – It is generally considered true that sweepers that can remove finer particles will result in improved runoff water quality. Earlier mechanical street sweepers removed mostly coarse particles (about 70%), while rain removed significant amounts of finer particles (about 50%) (Pitt, 2002). A more recent study (Valiron, 1992) confirmed conventional sweepers achieved only 15% removal of those particles less than 40 μm compared to 80% removal of particles greater than 2 mm. If most of the mass of Hg and PCBs is associated with fine particles, this implies removal efficiencies of 15-80% and likely closer to 15%. However, without knowing the size distribution of the sweepings and the concentrations in each size class, it is impossible to predict the efficiency but <50% does not seem unlikely. In Section 3 (Figure 3-5), we found that Hg was about 5 times higher on <150 micron particles however we have no information on PCB concentrations in relation to particle size.

Climate effects - In humid areas, frequent rain minimizes accumulation of dust and dirt. However, in drier climates where rains are relatively infrequent, streets become quite dirty during late summer and fall. Street sweeping studies in San Jose and Castro Valley have shown reductions in suspended solids and heavy metal concentrations in runoff (Pitt, 2003, based on Pitt, 1979, Pitt and Shawley, 1982).

Pick Up Efficiency - Various pick-up efficiencies have been measured or claimed in various reports and studies. It is impossible to specify a general street sweeping effectiveness because that depends on the many factors described above, as well as particle size measured. Reported efficiencies range from 0% to about 80% for total solids (TS). High rates are associated with modern efficient sweepers and high frequency of cleaning (Minton, et al., 1998; Curtis and Meosotis, 2002).

Modeling – Models also have been used to estimate the theoretical effectiveness of street sweeping. Calibrated simulation models have taken these factors into account, such as the Simplified Particulate Transport Model (SIMPTM) developed by Sutherland and Jelen (1993) or the Source Loading and Management Model (SLAMM) developed by Pitt and Voorhees (2000). These models have been calibrated and applied by some researchers to estimate loads and concentrations from stormwater catchments, as well as to evaluate BMP effectiveness, including street sweeping. Modeling build up and removal by street sweeping and/or washoff shows much promise for improved runoff quality due to sweeping. For example, regular pavement sweeping (weekly) and annual catch basin cleaning were predicted to remove 75% TSS in runoff (Sutherland & Jelen, 1996). In contrast, Bannerman et al. (2003) predicted TSS removals of 17% for high efficiency street sweeping.

Pollutant Accumulation - The seasonal rainfall in San Francisco Bay results in a strong seasonal first flush because of the build-up of pollutants during the summer and autumn months. Note that this build-up is not progressive, because eventually, wind and vehicle turbulence limits the accumulation. Efficient sweeping (sweepers capable of removing small particles) strategically applied before the winter rains in November may bring

about significant reduction in the pollution of the first few runoff events, if the logistical problems of timing, deployment and parked cars can be overcome.

Street dust build-up rates are highly variable. Pitt (1996) summarizes rates for California and the rest of North America. Table 5-1 is a condensed summary. Two things are worth noting in Table 5-1: (1) the maximum observed loading is much higher for rough roads, and (2) the time period for reaching maximum loading is 1-2 months. The maximum loading condition is approached asymptotically with time and Sartor and Boyd (1972) showed that street loading substantially rebounded within only 1-2 weeks following rain events and sweeping.

Table 5-1. Street dust loading and deposition rates (Pitt, 1996).

Location	Initial loading g (curb-meter)⁻¹	Deposition rate g (curb-meter)⁻¹ day⁻¹	Maximum observed loading g (curb-meter)⁻¹	Time to maximum loading (Number of days)
Smooth and intermediate textured Streets				
San Jose, CA	35	4	>140	>50
San Jose, CA	80	4	230	70
Castro Valley, CA	85	10	290	70
Rough and Very Rough Textured Streets				
San Jose, CA	510	6	>710	>50
San Jose, CA	220	6	430	30

The benefits of street sweeping are offset by the reasonably rapid build-up. Routine street sweeping programs can only be conducted, at best, at a bi-weekly or monthly frequency and the sweeping must be staged over the areas to be swept.

At the end of the day, we lack definitive studies that measure the actual benefits of street sweeping on urban runoff quality. Studies have often failed to measure benefits to stormwater quality (Bannerman pers. comm.). Detecting differences due to sweeping is difficult because stormwater quality is so variable and because of the difficulties in measuring particulate matter in street runoff (Martinella et al., 2002).

5.3.2.1 Implication of Enhanced Street Sweeping in Affecting PCB and Hg Loads to San Francisco Bay

The effectiveness of street sweeping is affected, and in some respect constrained, by a number of factors as discussed above. Of these factors, the frequency of sweeping, the extent to which the technology can pick up the particle sizes to which Hg and PCBs are attached, and ultimately the significance of streets as a source of Hg and PCBs will affect the overall benefits of this option. A key consideration is that enhancements of these programs are considered relatively expensive. Salop and Akashah (2004) have

conducted an initial evaluation of the potential for enhancing these programs by the local member agencies of the Alameda Countywide Clean Water Program. Their initial results indicated that the potential benefits of conversion of all equipment to higher efficiency models are limited (e.g., 1.1 kg/yr of PCBs and 2.2 kg/yr of Hg or about 3% of the target TMDL load reduction), although particle size was only considered minimally and additional monitoring to sample the Hg and PCB in material collected by street sweeping is planned to confirm this estimate (Salop and Toll, 2004). Overall the local information collected to date, along with the national literature, suggests that street material collected by sweepers represents only a fraction of the total mass accumulated on streets, and reasonable enhancements of such programs will not substantially change this proportion picture. Street sweeping is conducted throughout the Bay Area and therefore may be considered a watershed scale control, although more intense street sweeping is commonly conducted at smaller scales such as downtown urbanized areas. In conclusion, the potential for load reduction of Hg and PCBs to the Bay by enhanced street sweeping is considered low and perhaps medium relative to other options if industrial areas are targeted.

5.3.3 Street Washing

Street washing using low pressure water can be very effective at removing coarse and fine particles from streets. Street wash wastewater would need to receive treatment before discharge to the Bay. This could be achieved by either:

1. Diversion to municipal wastewater treatment plants. A diversion system that allows operators to direct wash water to wastewater sewers or capturing the washing downstream and pumping into tanker trucks could be used. Diversion requires installation of new infrastructure, as well as the need to deal with downstream implications. Although usually designed to maintain enough velocity to prevent sedimentation of a lighter normally lower particle density load, some of the heavy particles in stormwater could settle in wastewater sewers, so the infrastructure will need to include catch basins or other similar devices to trap the large particles. Capturing and transport in tankers could occur in either downstream detention basins or pump stations, or by temporarily plugging the conveyance system and vacuuming the wash water into the tanker truck. Due to the effort and expense of capturing the wash water, this technique would be limited to “hot spot” areas. Likely candidate areas would be industrial catchments draining to pump stations, such as the Ettie Street pump station.
2. Capturing and treatment (e.g., settling) in downstream treatment BMPs. While this BMP seems redundant because the whole process is more economically achieved during rainfall, there are some definite advantages in terms of retrofitting. The treatment BMP can be sized to a fraction of the upstream watershed – and the washing processes can be staged to occur in dry weather.

5.3.3.1 Implication of Street Washing in affecting PCB and Hg Loads to San Francisco Bay

The effectiveness of street washing depends on factors similar to street sweeping including pollutant build up, cleaning frequencies and rainfall event frequencies, as well as the efficiency of downstream transport and the efficiency of downstream diversion or treatment. Similarly to street sweeping, the scale of application of street washing would be a street block, therefore effectiveness on a regional scale is partly dependant on the choice of location. However, on a per curb-mile basis, street washing is likely to be more effective than sweeping as it can more effectively remove pollutants from uneven areas and cracked pavements. Street washing is not constrained by two critical shortcomings of street sweeping: vehicle speed and the ability to sweep along curbs. However, the effectiveness of street washing is partly constrained by down-gradient transport (flow volume/rate/ slope dependant and capture methods. The overall conclusion is presumed that street washing would yield limited benefits in controlling loads of Hg and PCBs to the Bay at a regional scale but in certain hotspot applications it might prove to be effective (although presently there is no data on PCBs or Hg to support that assertion).

5.3.4 Storm Drain System Maintenance

Storm drain maintenance activities include cleaning catch basins, storm drain inlets, storm drains, pump stations, trash racks, and other facilities associated with the storm drain system. The following summarizes literature and local information on the efficacy of these practices in removing sediment, and by implication sediment-associated pollutants like PCBs and Hg.

Catch basins have been recently reviewed by Pitt (2002) and the following has been summarized from his review. Catch basins are an inlet box with a grating at the road curb, with a discharge pipe to the storm sewer. Catch basins include sumps to trap particulate matter, ranging in depth from 0.5 – 1 m. Inlets that do not include a sump are typically referred to as “drop inlets.” And may be thought of as self-cleaning although there may be a lot of material lying on the bottom at the beginning of the wet season.

Catch basins mostly trap larger particles (sand-sized and greater) that are not usually regarded as part of the stormwater quality problem. They do trap finer particles that are part of the stormwater suspended load, as well. A number of studies have demonstrated that catch basins can reduce pollutants in stormwater (Table 5-2). The proportion of catch basins vs. storm drain inlets in the Santa Clara Valley is unknown (EOA, 1999), and it is likely that this is the case with most, if not all, of the remainder of stormwater programs in the Bay Area.

A survey of the 80 stormwater agencies in the Bay Area conducted in 2004 found that all of the agencies perform at least some storm drain system maintenance activities (Table A-1) (CEP, 2004). Of the 80 agencies, all but 15 perform street sweeping and inlet cleaning. Forty-nine agencies reported conducting storm drain line/ditch cleaning, 30 perform pump station cleaning, and 39 perform in-stream sediment removal. Other

maintenance practices include sediment removal from junction boxes (18 agencies) and from culverts, silt basins, lagoons, and detention ponds (13 agencies). The reported volume and mass of sediment removed through maintenance practices is summarized in Table A-2.

Table 5-2. Effectiveness of Catch Basins on Trapping Pollutants and on Water Quality

Location	Effect	Reference
Bellevue, WA	Reduce total residue, Pb in runoff by 10-25% Reduce COD, TKN, TP, Zn in runoff by 5-10%	Pitt & Shawley (1982)
Bellevue, WA	Trap coarse particles, up to 45% of total street particulates	Pitt (1985)
London, UK	Trapped sediment only 10% particles < 100 µm	Butler et al. (1995)
London, UK	Increases pollution of first flush runoff with COD, NH ₃	Butler et al. (1995)
Stafford, NJ	Reduce TSS in runoff on avg. 22% (0-50%) Reduce SS in runoff on avg. 32% (0-55%)	Pitt and Field (1998)

5.3.4.1 Implication of Storm Drain System Maintenance in Affecting PCB and Hg Loads to San Francisco Bay

The Joint Stormwater Agency Project evaluated concentrations of mercury and PCBs in sediments collected from stormwater conveyance systems (manholes, catch basins, open channels, outfalls, and pump stations) throughout the Bay Area (KLI and EOA, 2002). The mean PCB concentration (normalized to the fine sediment fraction) was 4,455 ppb for industrial land uses and 2,224 ppb for residential/commercial land uses. The mean Hg concentration (normalized) was 2.4 ppm for industrial and 4.6 ppm for residential/commercial land uses. (The mercury TMDL is predicated on a Bay mass fraction concentration of Hg of 0.2 ppm, and current research is indicating that the corresponding PCB fraction will be 1-10 ppb.)

Salop and Akashah (2004) conducted an initial evaluation of the potential for enhancing storm drain system maintenance programs by the local member agencies of the Alameda Countywide Clean Water Program. Their initial results indicate that the potential benefits of increasing the frequency of storm drain facility cleanouts from annually to semi-annually as 0.3-2.0 kg/yr of PCBs and 0.1-0.4 kg/yr of Hg or about 1-6% and 0.1-1% of the target TMDL load reduction for the Bay Area). Additional monitoring is planned for the fall of 2005 to refine these estimates (Salop and Toll, 2004). These initial estimates, if confirmed, and extrapolated to the Bay Area, indicate that implementing enhanced storm drain system maintenance could provide modest to significant reductions in Hg and PCB loads to the Bay although we recognize that this

conclusion (based on the storm drain conveyance pollutant budgets in Section 2 of this white paper) differs from that of Salop and Akashah (2004).

The scale of application of storm drain maintenance is fundamentally the facility scale (e.g., storm drain inlet, pump station). In the case of storm drain flushing the scale is the reach of line being flushed. Enhanced storm drain maintenance could target industrial areas or areas adjacent to known hotspots, in which case the scale of application becomes a catchment scale.

5.3.5 Channel De-Silting

Much of the runoff from the storm drain system in the Bay Area discharges to streams, rivers, and drainage channels prior to entering the Bay. These channels tend to drain urban areas and in some cases upland open space. Agencies responsible for maintaining these facilities conduct de-silting in order to maintain the hydraulic conveyance. De-silting tends to be conducted in areas near the Bay margin where the topography is relatively flat. The Santa Clara Valley is a good example of this condition, where the Santa Clara Valley Water District is responsible for flood management.

Permitting requirements for channel de-silting involve obtaining Clean Water Act (CWA) Section 404 Permits from the Corps of Engineers and the associated CWA Section 401 Water Quality Certifications from the Water Board, and Department of Fish and Game Code Sections 1601 Streambed Alteration Permits. Flood control agencies such as the Santa Clara Valley Water District are entering into long-term maintenance agreements with regulatory agencies in order to facilitate timely de-silting maintenance.

The amount of sediment removed by agencies varies substantially year to year depending on the location, amount of rain, funding, and permitting status. For example, the total mass of sediment removed through de-silting conducted by the Alameda County Flood Control and Water Conservation District (ACFCWCD) was about 100,000 cubic yards in 1999 and 2000. In some years, very little de-silting was conducted (Salop and Akashah, 2004). The SCVWD also has an active de-silting program and removed 115,100 cubic yards of sediment from stream channels in 1999.

5.3.5.1 Implication of Channel De-Silting on PCB and Hg Loads to San Francisco Bay

There is limited monitoring data in these de-silting sites for Hg and PCBs (Salop et. al., 2002 and KLI and EOA, 2002) and where it exists, it mostly yields low concentrations. Sediment data collected by the ACCWP for creeks, flood control channels, and an in-channel stilling basin indicated PCB concentrations in the range of 0.3 ppb to 472 ppb with a mean of 44 ppb (excluding two known “hot spots”) (Salop et al., 2002). Mercury concentrations ranged from 0.04 ppm to 4.3 ppm with a mean of 0.5 ppm.

Based on local data, Salop et al. (2004) estimated a range of about 50-150 ppb PCBs in older urban areas (pre-1950) and a range of 0.2-0.5 ppm for Hg. These data appear to indicate that de-silting drainage channels can remove significant PCBs and Hg mass. This could be enhanced by targeting areas where there are known hotspots (especially for PCBs).

The scale of PCB and Hg pollution in channels is the reach scale, the length of the reach dependent on the number of outfalls that may have contributed pollution in the past or are continuing to contribute. Targeting on a spatial basis is likely to be impractical given de-silting is usually carried out to maintain flood conveyance capacity. De-silting also could possibly be carried out to create deeper pools that could enhance settling of fine material.

5.4 TREATMENT BMPs

In addition to maintenance of the storm drain system, urban runoff treatment is another option for intercepting Hg and PCB loads to the Bay. This section describes and evaluates the potential effectiveness of treatment options for removing particle-associated pollutants such as PCBs and Hg from urban runoff.

5.4.1 Unit operations and processes

The selection of a stormwater treatment system should be based on a fundamental understanding of water quality (solid-phase and aqueous chemistry) and hydrology, but this approach is not common. The more common design approach is to select treatment BMPs that are expected, or have been shown in some manner, ability to treat the pollutants of concern (or a surrogate pollutant such as TSS) consistent with some stipulated performance measure (e.g., 80% removal) with little attention paid to the physical, chemical, and biological unit operations and processes (UOPs) that occur within the BMPs. A better approach is to first select UOPs applicable for the target constituents based on the constituent form (i.e., dissolved, colloidal, particulate), chemical speciation (e.g. ionic metal species, phosphorus species, etc.), and granulometric characteristics (e.g., particle size, specific gravity, surface area). Then, to individually select the components of a treatment system based on the UOPs that are effective for treating target constituents.

All UOPs can be organized according to four fundamental process categories: 1) hydrologic controls, 2) physical treatment operations, 3) biological processes, and 4) chemical processes. Table 5-3 provides a summary of the fundamental process categories, and related UOPs and treatment system components. Treatment System Components incorporate one or several UOPs, and include conventional BMPs, such as swales, ponds, tanks, and so forth, in addition to pretreatment devices (e.g., trash racks, catch basin screens, etc.), and tertiary enhancements (e.g., soil amendments, carefully selected vegetative species, and custom hydraulic controls such as weirs, etc.).

Table 5-3. Unit Processes and Associated Treatment Controls* (highlighted UOPs are relevant to Hg and PCB treatment)

Fundamental Category	Process	Unit Operation or Process (UOP) Target Pollutants	Typical Treatment System Components
Hydrologic Operations		Flow and Volume Attenuation	Extended detention basins Retention/detention ponds Wetlands Tanks/vaults Equalization basins
		Volume Reduction All pollutant loads	Infiltration/exfiltration trenches and basins Permeable or porous pavement Bioretention cells Dry swales Dry well Extended detention basins WHAT ABOUT LID: Green roofs, Greenways, Less pavement, etc.
Physical Treatment Operations		Size Separation and Exclusion (screening and filtration) Coarse sediment, trash, debris	Screens/bars/trash racks Biofilters Permeable or porous pavement Infiltration/exfiltration trenches and basins Manufactured bioretention systems Engineered media/granular/sand/compost filters Hydrodynamic separators Catch basin inserts (i.e., surficial filters)
		Density, Gravity, Inertial Separation (grit separation, sedimentation, flotation and skimming, and clarification) Sediment, trash, debris, oil and grease	Extended detention basins Retention/detention ponds Wetlands Settling basins, Tanks/vaults Swales with check dams Oil-water separators Hydrodynamic separators
		Microbially Mediated Transformation (can include oxidation, reduction, or facultative processes) Metals, nutrients, organic pollutants	Wetlands Bioretention systems Biofilters (and engineered bio-media filters) Retention ponds Media/sand/compost filters
Biological Processes		Uptake and Storage Metals, nutrients, organic pollutants	Wetlands/wetland channels Bioretention systems Biofilters Retention ponds
		Chemical Sorption Processes Metals, nutrients, organic pollutants	Subsurface wetlands Engineered media/sand/compost filters Infiltration/exfiltration trenches and basins
Chemical Processes		Coagulation/Flocculation Fine sediment, nutrients	Detention/retention ponds Coagulant/flocculent injection systems
		Ion Exchange Metals, nutrients, organic pollutants	Engineered media, zeolite, peat's, surface complexation media
		Chemical Disinfection Pathogens	Custom devices for mixing chlorine or aerating with ozone Advanced treatment systems

* Adapted from Strecker et. al., (2005)

In addition to those listed in Table 5-3, diversion of dry weather runoff and the low flow portion of wet weather runoff to a wastewater treatment plant is also an option since the physical, biological and chemical treatment processes presented in Table 5-3 are equivalent to those performed by POTWs. In some cases, routing stormwater to POTWs may be more cost effective than independently treating stormwater. The use of alum to aid flocculation and capture needs further evaluation, however, initial data in the Tahoe basin indicate that this is a potentially effective treatment.

5.4.2 Hydrologic Control

Flow alteration is a significant unit operation for stormwater treatment and historically has been the single major unit operation for stormwater management for decades in the United States and many parts of the world. Water quality and quantity cannot be separated; alterations to the hydrograph affect water quality. In large part, flow alteration is implemented as a hydrologic control. Flow alteration includes modifications to components of the hydrologic cycle such as runoff, infiltration, detention, storage and evaporation. In general, the goals of these physical operations (recognized as hydrologic controls) have been to reduce volume, reduce peak flows, generate more uniform flow rates, and attenuate temporal aspects of flow. To varying degrees, these hydrologic controls can have a significant impact on water quality. Applications of hydrologic modification are ubiquitous in the built environment and are intentional or inadvertent, as well as beneficial or detrimental. Examples of intentional applications that have potential water quality and quantity benefits include infiltration, detention and flow equalization, while detrimental applications include impervious paving or loss of vegetation.

There are two fundamental hydrologic unit operations: flow attenuation and volume reduction (or minimization of volume increases). Flow attenuation refers to the hydrologic operations responsible for reducing peak event discharges (e.g., "peak shaving"). The primary mechanisms involved in flow attenuation include interception, conveyance, and detention, and, to a lesser degree, infiltration. Volume reduction hydrologic operations, responsible for reducing the total volume of runoff, are retention, infiltration, and evapotranspiration. Runoff can also be retained in storage vessels such as underground tanks and vaults and reused (e.g., irrigation water). If pollutant loads are a high concern, volume reduction should be a major unit operation in any selected treatment system design.

Hydrologic source control is ideally suited to new development, and has evolved into a concept of low impact development (Davis, 2005). Potential hydrologic controls in urban retrofit conditions may include routing roof runoff to rain gardens, or incorporation of roof gardens in redevelopment projects. The basis for granting load reduction credits for hydrologic source controls in the context of TMDLs is unclear.

5.4.3 Treatment Performance

5.4.3.1 EPA International Stormwater BMP Database

The treatment performance of the more widely accepted BMPs are summarized in the EPA International Stormwater Best Management Practices (BMP) Database (EPA, 2005) that contains the performance data from over 200 BMP monitoring sites, including over 40 sites in California. The following discussion of treatment performance is based on interpretation of these data.

There are various measures of treatment performance. Treatment performance has traditionally been presented in the form of percent reductions in effluent vs. influent concentrations or loads. Analysis of the database indicates a wide variability in this measure, possibly related to the fact that the same type of BMP could show poor removals if located at a site where influent concentrations were relatively low, compared say to a site where influent concentrations were high. A more statistically robust (lower variability) measure of treatment performance appears to be effluent quality. The following discussion will therefore focus on effluent quality as a measure of effectiveness (see also CASQA BMP handbooks).

Total suspended solids (TSS) represent the most widely reported stormwater constituent in the International Stormwater BMP Database. Information regarding Suspended Solids Concentrations (SSC), particle size distributions or settling velocities among the studies included in the database is very limited. The following discussion will therefore focus on TSS data.

BMP data in the database are organized as follows:

- DB – Detention Basins (dry)
- GS – Grass Swales or Biofilter
- HD – Hydrodynamic Device
- MF – Media Filter
- RP – Retention Pond (Wetpond)
- WB – Wetland Basin
- WC - Wetland Channel

It is important to note that the retention ponds (RP) as classified in the Database are wetponds, not infiltration basins. The number of sites within each BMP category varies from 3 to 21 (Table 5-4).

The following summarizes some of the key statistics from analysis of the database, based on pooling the data by site and then pooling the data by storm event.

5.4.3.2 Site Mean Effluent TSS Concentration

Figure 5-3 shows, for each BMP category, the statistical distribution of the average effluent concentration for individual BMPs in that category. This type of pooling weighs the data taken from each site equally, irrespective of the number of storms monitored at each site. The box and whisker plots show the median and the 95% confidence band about the median, the 25th and 75th percentiles, and outliers. Table 5-6 shows the corresponding numerical values, and the number of sites in each BMP category. Based on a non-parametric analysis, the median of average effluent TSS concentrations is significantly lower than the median average influent for media filters (MF), retention ponds (RP), wetland basins (WB) and wetland channels (WC).

For most BMP categories, the median effluent TSS concentration ranged from 19 to 25 mg/L (Table 5-4). The distribution of average effluent concentrations were slightly higher for detention basins (DB), which drain after each event and generally lacked a significant littoral zone and hydrodynamic devices (HD), flow-through systems that rely on centrifugal and gravitational forces to provide treatment. It should be noted that detention basins have been shown to provide considerable reduction (up to 30%) in effluent volume, which may translate into lower total mass loading of TSS downstream.

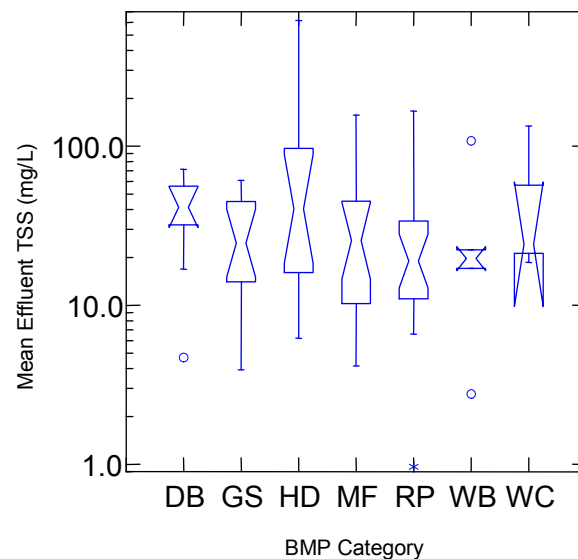


Figure 5-3. Distribution of Effluent TSS from Pooled Site Mean Data

5.4.3.3 Storm Event Effluent TSS Concentrations by BMP Category

A second way of pooling data in the database is to pool all the storm event EMC data. This type of pooling weights the data from each storm event equally, irrespective of site. The distribution of effluent EMCs were lowest for media filters, retention ponds and wetland basins (Figure 5-4). Median effluent TSS EMCs for hydrodynamic devices, retention ponds, wetland basins and wetland channels were significantly less than median influent EMCs. The distribution of influent TSS values for hydrodynamic devices was much higher than for other categories in this dataset, and may not be representative of typical urban runoff. In general, lower effluent TSS concentrations were observed for wetland basins (WB) and wet ponds (RP) that provide extended storage of stormwater flows.

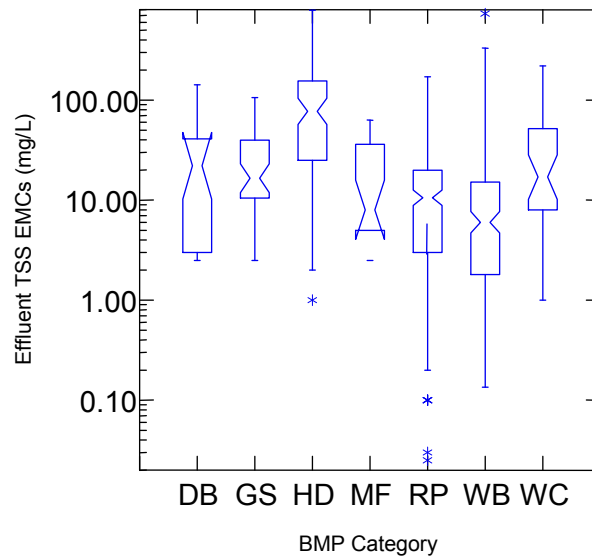


Figure 5-4. Distribution of Effluent TSS from Pooled Storm Event EMC Data

Table 5-4. Median TSS Effluent Concentrations (mg/l) from Pooled Site Mean Data and Pooled EMC data (EPA, 2005).

BMP Category		Number of BMPs	Median of Avg. Effluent (95% Confidence Interval) ¹			Significant Difference Between Average Influent and Effluent ²	Median of Effluent EMCs (95% Confidence Interval) ¹			Significant Difference Between Influent and Effluent EMCs ²
			Median	LCL	UCL		Median	LCL	UCL	
DB	Detention Basin	9	41	30	55	NO	22	10	47	NO
GS	Grassy Swale	14	24	15	40	NO	16	12	23	NO
HD	Hydrodynamic Device	13	40	18	89	NO	77	57	104	YES
MF	Media Filter	18	25	14	44	YES	8	4	16	NO
RP	Retention Pond (wet pond)	21	19	12	28	YES	10	9	13	YES
WB	Wetland Basin	6	19	16	23	YES	6	5	8	YES
WC	Wetland Channel	3	24	10	59	YES	17	10	28	YES

¹Calculation of confidence interval based on McGill et al (1978), from the natural log of the quantiles.

²Based on non-parametric analysis of difference in median values.

5.4.3.4 Caltrans Database

Another source of treatment performance data is that collected by Caltrans as part of the BMP Retrofit Pilot Program (Caltrans, 2004). Table 5-5 summarizes the influent and effluent data for the pooled Event Mean Concentrations. In contrast to Table 5-4 where the median values of the pooled EMCs are given, the data in Table 5-5 is the mean of the pooled EMCs. The table indicates that, for all of the BMPs tested, the effluent was significantly lower than the influent. On the basis of effluent quality, the better performing BMPs are the filters and wet basins (mean effluent about 10-20 mg/l) compared to extended detention basins, grassy swales, and buffer strips (30-50 mg/l).

These results are consistent with the findings from the EPA database. The Caltrans study identified Austin and Delaware sand filters as effective in the removal of particles and some inorganic constituents of stormwater, however we are not aware of any that have been tested for Hg and PCBs neither are we aware of any installed in the Bay Area.

Table 5-5. Mean Influent and Effluent TSS Concentrations Based on Pooled Storm Event Data (Caltrans, 2004)

BMP Types	Number of BMPs	Mean Influent EMC (mg/l)	Mean Effluent EMC (mg/l)	Significant Difference?
Unlined Extended Detention Basins	5	137	39	yes
Grassy Swales	6	94	47	yes
Buffer Strips	4	100	31	yes
Sand Filter (Austin)	5	88	8.6	yes
Sand Filter (Delaware)	1	102	19	yes
Wet Basin	1	210	14	yes

5.4.4 Summary

Suspended solids can be removed effectively by settling. For most well designed treatment BMPs, the median effluent TSS concentrations are in the 10-50 mg/L, provided the concentration and characteristics (e.g., particle size distributions) of influent suspended solids do not significantly deviate from “typical” stormwater. Well designed treatment systems that incorporate wet pools and wetland vegetation typically exhibit good effluent quality for suspended solids. Based on currently available data, these BMPs can typically achieve effluent concentrations of less than 20 mg/L. Well designed swales and media filters also perform well in achieving low effluent suspended solids concentrations.

The presence of a permanent wet pool is a feature of a wet pond/wetland system. Incorporating even a small permanent wet pool can significantly improve the sediment removal performance of TSS by providing long periods of retention during smaller storms. Long retention times during small events allow for appreciable sediment removal compared to dry facilities that typically have very limited detention times during small events. Generally, settleable solids comprised of inorganic particles in the 25 to 75 μm range are effectively removed by quiescent gravitational sedimentation (Hong Lin, 2004). Gerb and Bannerman (1997) indicate that particles greater than about 30 μm are effectively removed in a wet pond they investigated, although they also note substantial reduction (about 74%) of fine sediments.

For vegetated swales and media filters, gravity settling and filtration are the primary removal mechanisms for suspended sediments. Direct filtration can usually be effectively accomplished if influent concentrations are less than 50 mg/L, which generally requires some level of pretreatment in urban runoff, where solids concentrations are frequently above 100 mg/L and can exceed 1,000 mg/L depending on the site, loading, and hydrology. Generally, suspended inorganic particles less than 25 µm require some natural or enhanced coagulation/flocculation followed by sedimentation and/or filtration.

5.4.4.1 Implication of Treatment on PCB and Hg Loads to San Francisco Bay

Analysis of TSS effluent data obtained from the EPA International Stormwater BMP Database (EPA, 2005) and the Caltrans BMP Retrofit Program (Caltrans, 2004) indicate that media filters, wet ponds, and wetland basins can achieve effluent TSS concentrations generally less than 25 mg/l.

A number of these BMP types (swales, wet ponds, and wetlands) are surface features and are ideally suited for new development and significant redevelopment. Retrofitting options are limited, especially in downtown urbanized areas. Candidate retrofit sites for these types of BMPs are typically parks, schools, and utility corridors that provide some open space; these types of sites are difficult to site BMPs because of public and/or agency acceptance issues. Media filters can be placed above or below ground and therefore could be a more likely candidate for retrofitting in ultra-urban environments. Where retrofitting is feasible, these data indicate that there could be significant reductions in TSS, and particle-associated pollutants such as PCBs and Hg, assuming that the catchment is source of these pollutants.

A survey of the 80 stormwater agencies in the Bay Area conducted in 2004 found that a total of 416 Treatment Control BMPs had been implemented within their jurisdictional boundaries (Table A-3). The most frequently implemented Treatment Control BMP was Vegetated Swales (173 sites), followed by Extended Detention Basins (88 sites), Drain Inserts (75 sites), Vortex Separators (23 sites), Media Filters (16 sites), and Wetponds and Infiltration Trenches (11 sites each). The remainder of the Treatment Control BMP types had three or fewer installations reported (CEP, 2004).

The extent to which such BMPs could effectively treat PCBs and Hg would depend on: (1) the particle fractions that would likely be treated, and (2) the predominant particle size ranges containing PCBs and Hg. With respect to the former, work conducted by Sansalone and his student Hong Lin at Louisiana State University (Hong Lin, 2003) on runoff from highways indicated that settleable solids (defined as solids that settled in an Imhoff cone in 1 hour) ranged from 25-75 microns, and suspended solids were in the range of 1-25 microns (Lin, 2003). Although there are obviously concerns regarding the representativeness of these data, it does indicate that suspended solids may tend to be in the range of about 25 microns or less. If this were the case in the Bay Area, particles greater than 25 microns would be effectively removed in these facilities, and if the mass of PCBs and/or Hg were predominantly in particles >25 µm, treatment would be

effective. If particles are smaller and mass of PCBs and Hg are on the smaller particles, BMP options would need to focus on another group of BMPs more apt to small particle removal (for example sand filters; Karamalegos et al., 2005).

5.4 SUMMARY

Pollution prevention, source controls, and treatment controls were evaluated in terms of potential for reducing loads of PCBs and Hg to the Bay. The evaluation is preliminary based on current information, and will be revised based on additional planned monitoring. Table 5-6 summarizes the relative effectiveness evaluation and the rationale for the assignment of relative effectiveness. The table does not take into account cost effectiveness.

Based on this evaluation, soil remediation and site cleanup may be good options for further evaluation. Channel de-silting may be a viable option for removing mass of Hg based on limited data and perhaps for PCBs (there is even less data). Channel de-silting should be further evaluated by permit holders before a final decision is made. Treatment options are considered effective based on the review, however, again, further monitoring is required to better understand the treatability of Hg and PCBs (including evaluation of how particle concentrations change during the treatment process). Storm drain maintenance is considered to be moderately effective, at least in industrial areas where past practices have involved the use of the PCBs and Hg. Street sweeping and washing are considered less effective because of the number of factors that constrain the implementation of these measures. Pollution prevention, although considered a very successful program, appears to have limited opportunities for enhancement with respect to reduction on loads of PCBs and Hg although ongoing legislative changes at the international, federal and state level will continue to reduce new imports of Hg in to the Bay Area and might have some impact of atmospheric deposition of Hg associated with long range transport (see section 3.4.1.2 of this report).

Table 5-6. Preliminary Evaluation of Relative Effectiveness of Enhanced BMP Implementation for Reducing Loads of PCBs and Hg to San Francisco Bay.

BMP Category	BMP type	Relative Effectiveness (particle associated pollutants)	Comments
Pollution Prevention	recycling, product substitution, etc.	Low	Established successful programs that have prevented introduction of products containing pollutants to aquatic, air, and soils environment. Difficult to distinguish benefits to individual media. Have helped to set baseline loads, but have limited potential for achieving additional significant reductions in Hg (except further legislative use bans) and PCB loads to Bay.
Source Control	Soil Remediation and Site Cleanup	High	Legacy nature of PCBs and Hg use by industries supports this option. Initial studies conducted by stormwater agencies indicate elevated concentrations at some older industrial areas. Potential for load reduction would depend on proximity of site to storm drain system, potential for sediments to be mobilized into storm drain, efficacy of enforcement and cleanup. Masses removed from such sites may be significant relative to target load allocation. May be more applicable to PCBs
Treatment Controls	Street Sweeping	Low to Medium	Ongoing funded maintenance activity. Incorporation of improved street sweeping fleet, more frequent sweeping, and targeted sweeping over time could improve removals of particulates and associated pollutants. Enhanced programs carry high price tag. No definitive field data that shows street sweeping as commonly employed actually improves runoff water quality. Need to understand particle size relationships between particles on the street and particles in the hopper of the street sweeper (in relation to Hg and PCBs)
	Street Washing	Low to High	Same issues as with street sweeping with additional issues relating to the transmission to the areas of capture and the capture process. Probably of limited use on a regional basis but may be highly effective in for removing mass on streets and pavements adjacent to polluted hotspots if gradients are conducive and volume and capture methods are experimentally developed.
	Storm Drain Maintenance	Medium	Ongoing funded maintenance. Data indicate elevated concentrations at pump stations, especially in industrial areas. Targeting areas near hotspots and increasing frequency might enhance effectiveness.
	Channel De-silting	Low-High	Data indicate elevated concentrations of Hg medium to low concentrations of PCBs in stream sediments. De-silting reaches with elevated concentrations could result in mass removed that would be significant relative to load reduction targets. Actual load reductions would depend on extent and mobility of polluted sediments. Significant permitting issues requiring multiple permits and permitting agencies.
	Settling and Filtration Systems (media filters, wet ponds, wetlands)	Medium to High	Treatment effectiveness for TSS demonstrated. Effectiveness for treating PCBs and Hg depends on particle size and density of particles associated with PCBs and Hg. Retrofitting opportunities for wet ponds and wetland BMPs in urban areas are limited. Locating subsurface media filter type BMP is more feasible.

5.5 REFERENCES

- Bannerman, R., Fries, G. and J. Horwach. (2003). "Source Area and Regional Storm Water Treatment Practices: Options for Achieving Phase II Retrofit Requirements in Wisconsin." Proceedings of the National Conference on Urban Storm Water: Enhancing Programs at the Local Level Chicago, IL February 17-20, 2003.
- Butler, D., Y. Xiao, S.H.P.G. Karunaratne, and S. Thedchanamoorthy (1995). "The gully pot as a physical and biological reactor." *Water Science & Technology*. Vol. 31, No. 7, pp. 219-228.
- Caltrans, 2004. BMP Retrofit Pilot Program. Final Report, CTSW-RT-01-050, January.
- CEP, 2004. Draft Feasibility Assessment for Managing Stormwater Loads in San Francisco Bay. Clean Estuary Partnership 001-09223-00. August 6, 2004.
- Curtis, Meosotis, 2002. Street Sweeping for Pollutant Removal. Department of Environmental Protection, Montgomery County Maryland, Watershed Management Division. February. P. 7.
- Davis, Allen, 2005. Green Engineering Principles Promote Low Impact Development, *Environmental Science & Technology*, August 15, pp. 339A-344A.
- EIP Associates, 1997. Polychlorinated Biphenyls (PCBs) Pollution Prevention Plan, prepared for City of Palo Alto, October.
- EIP Associates, 2000. Mercury Reduction Menu, prepared for City of Palo Alto, April.
- EOA, 1999. Catch Basin Retrofit Feasibility Study Technical Memorandum, prepared for Santa Clara Valley Urban Runoff Pollution Prevention Program, July.
- EOA, 2004. Review of Potential Measures to Reduce Urban Runoff of PCBs to San Francisco Bay, prepared for Santa Clara Valley Urban Runoff Pollution Prevention Program, March.
- GeoSyntec Consultants, Oregon State University, University of Colorado, Louisiana State University, Aquatus Environmental, July 2004. Critical Assessment of Stormwater Treatment and Control (BMP) Selection Issues, Interim Draft Guidance Manual, submitted to Water Environment Research Foundation.
- EPA, 2005. International Stormwater Best Management Practices (BMP) Database. Cited September 2005. <http://www.bmpdatabase.org>
- Greb, S., and R. Bannerman, 1997. Influence of Particle Size on Wet Pond Effectiveness, *Water Environment Research*, Vol. 69, Number 6, pp1134-1138.
- Herrick, Robert F., M.D. McClean, J.D. Meeker, L.K. Baxter, G.A. Weymouth, July 2004. An Unrecognized Source of PCB Contamination in Schools and Other Buildings, *Environmental Health Perspectives*, pp 1051-1053, Vol. 112, Number 10.
- Hetzel, F. 2004. PCBs in San Francisco Bay: Total Maximum Daily Loads Report. San

Francisco Bay Regional Water Quality Control Board. Oakland, CA.

Johnson, B. and R. Looker (2003). Mercury in San Francisco Bay Total Maximum Daily Load (TMDL) project report. California Regional Water Quality Control Board San Francisco Bay Region, Oakland, CA, 94 pp.

Karamalegos, A.M., Barrett, M.E., Lawler, D.F., and Malina, Jr., J.F., 2005. Particle Size Distribution of Highway Runoff and Modification Through Stormwater Treatment. Report prepared for Texas Department of Transportation (TxDOT) by Center for Research in Water Resources, The University of Texas at Austin.
<http://www.crwr.utexas.edu/reports/pdf/2005/rtp05-10.pdf>

Kohler, Martin, J. Tremp, M. Zennegg, C. Seiler, S. Minder-Kohler, M. Beck, P. Lienemann, L. Wegmann, P. Schmid, 2005. Joint Sealants: An Overlooked Diffuse Source of Polychlorinated Biphenyls in Buildings, *Environmental Science and Technology*, pp. 1967-1973, Vol. 39, No. 7.

KLI and EOI (Kinnetic Laboratories, Inc. and Eisenberg, Olivieri, and Associates) 2002. Joint Stormwater Agency Project to Study Urban Sources of Mercury, PCBs and Organochlorine Pesticides. Prepared for SCVURPPP, CCCWP, SMCSPPP, MCSPPP, VFCSD, and FSSD. April 2002.

Larry Walker Associates, 2005. PCB TMDL Implementation Plan Development (Draft), prepared for Clean Estuary Project, June.

Lin Hong 2003. Granulometry of non-colloidal particulate matter transported by urban rainfall-runoff. Ph. D Thesis.

McKee, L., Leatherbarrow, J., Newland, S., and Davis, J., 2003. A review of urban runoff processes in the Bay Area: Existing knowledge, conceptual models, and monitoring recommendations. A report prepared for the RMP Sources, Pathways and Loading Workgroup. San Francisco Estuary Regional Monitoring Program for Trace Substances. SFEI Contribution Number 66. San Francisco Estuary Institute, Oakland, Ca.

McKee, L., Leatherbarrow, J., Eads, R., 2004. Concentration and loads of mercury, PCBs, and OC pesticides associated with suspended sediments in the lower Guadalupe River, San Jose, California. A technical report of the Regional Watershed Program: SFEI Contribution # 66. San Francisco Estuary Regional Monitoring Program for Trace Substances, San Francisco Estuary Institute, Oakland, Ca.

Martinelli, Thomas J., Waschbusch, R.J., Bannerman, R.T. and Wisner, Ann, 2002. *Pollutant Loading to Stormwater Runoff from Highways: The impact of Freeway Sweeping Program, Wisconsin Department of Transportation*, Research Project ID # 0092-4582.

Minton, Gary R.; Lief, Bill; Sutherland, Roger, 1998. High-Efficiency Sweeping or Clean a Street, Save a Salmon! Stormwater Treatment Northwest. Vol. 4, No. 4. November.

Pitt, R. *Demonstration of Nonpoint Pollution Abatement Through Improved Street Cleaning Practices*. U.S. EPA. Grant No. S-804432. EPA-600/2-79-161. 270 pages. Cincinnati, August 1979.

- Pitt, R. and G. Shawley. 1982. *A Demonstration of Non-Point Source Pollution Management on Castro Valley Creek*. Alameda County Flood Control and Water Conservation District (Hayward, CA) for the Nationwide Urban Runoff Program, U.S. Environmental Protection Agency, Water Planning Division, Washington, D.C., June 1982.
- Pitt, R. 1985. *Characterizing and Controlling Urban Runoff through Street and Sewerage Cleaning*. U.S. EPA. Contract No. R-805929012. EPA/2-85/038. PB 85-186500/AS. 467 pages. Cincinnati, June 1985.
- Pitt, R. 1996. "Accumulation, washoff, and size distributions of stormwater particulates." In *Solids in Sewers: Characteristics, Effects, and Controls of Sewer Solids and Associated Pollutants*. Scientific and Technical Report of the International Association of Water Quality (IAWQ). London.
- Pitt, R.E., Field, R. 1998. "An evaluation of storm drainage inlet devices for stormwater quality treatment". Water Environment Federation 71st Annual Conference and Exposition, WEFTEC Technology Forum. Orlando FL., October 1998.
- Pitt, R.E. and Voorhees, J. 2000. *The Source Loading and Management Model (SLAMM), A Water Quality Management Planning Model for Urban Stormwater Runoff*. University of Alabama, Department of Civil and Environmental Engineering, Tuscaloosa, AL.
- Pitt, R.E. 2002. "Emerging Stormwater Controls for Source Areas." In *Management of Wet Weather Flows in Watershed*. Sullivan, D. and Field, R., eds., CRC Press, Boca Raton, FL.
- Salop, Paul, D. Hardin, K. Abu-Saba, A.J. Gunther, 2002a. Analysis of 2001 Source Investigations in Ettie Street Pump Station and Glen Echo Creek Watersheds, Oakland, California, prepared for Alameda Countywide Clean Water Program.
- Salop, Paul, K. Abu-Saba, A.J. Gunther, and A Feng. 2002b. 2000-01 Alameda County Watershed Sediment Sampling Program: Two-Year Summary and Analysis, prepared for Alameda Countywide Clean Water Program.
- Salop, Paul and M. Akashah, 2004. A Review of Source Control Options for Selected Particulate-Associated TMDL Pollutants, prepared for Alameda Countywide Clean Water Program, August.
- Salop, Paul and J. Toll, 2004. Source Control Options Related to TMDL Implementation Actions, Sampling and Analysis Plan, prepared for Alameda Countywide Clean Water Program, September.
- Sartor, J. and G. Boyd, 1972. *Water Pollution Aspects of Street Surface Contaminants*, US EPA Report EPA-R2-72-081.
- Sutherland, R.C. and S.L. Jelen (1996). "Sophisticated stormwater quality modeling is worth the effort." In: *Advances in Modeling the Management of Stormwater Impacts*. Edited by W. James. Computational Hydraulics International. Guelph, Ontario.
- Sutherland, R.C & Jelen, S.L. (1993). *Simplified Particle Transport Model-Users Manual*, Version 3.1, 66pp.

- Strecker, Eric, W. Huber, J. Heaney, D. Bodine, J. Sansalone, M. Quigley, M. Leisenring, D. Pankani, A. Thayumanavan, 2005. Critical Assessment of Stormwater Treatment and Control Selection Issues, Water Environment Research Foundation Report 02-SW-1.
- U.S. EPA, 1983. Results of the Nationwide Urban Runoff Program, Executive Summary, December.
- Valiron, F (1992). "Usual techniques for stormwater pollutant removal in urban areas." (in French) *Provisory report for the Seine-Normandie Water Agency*, 61 p.

ATTACHMENT A – INVENTORY OF Bay Area BMPs

Table A-1. Survey Responses: Maintenance Practice Implementation by Agency.

Agency	Maintenance Practice					
	Street/Parking Lot Sweeping	Inlet Cleaning	Storm Drain Line/Ditch Cleaning	Pump Station Cleaning	In-Stream Sediment Removal	Other
ACFCWCD	no	yes	yes	yes	yes	
Alameda County	yes	yes	no	yes	yes	
Albany	yes	yes	yes	no	yes	
Atherton	yes	yes	yes	yes	yes	Junction boxes
Belmont	yes	yes	yes	yes	yes	Junction boxes
Belvedere	yes	yes	no	no	no	Culverts, silt basins, lagoons, and detention ponds
Berkeley	yes	yes	yes	no	yes	Junction boxes
Brisbane	yes	yes	yes	yes	no	Junction boxes
Burlingame	yes	yes	yes	yes	yes	Junction boxes
Caltrans	no	no	no	no	no	
City of Alameda	no	no	no	no	no	
City of San Mateo	yes	yes	yes	no	yes	Junction boxes
Clayton	yes	yes	yes	no	no	
Colma	yes	yes	yes	no	no	
Contra Costa County	yes	yes	yes	no	yes	
Corte Madera	yes	yes	no	yes	no	Culverts, silt basins, lagoons, detention ponds
Cupertino	no	no	no	no	no	
Daly City	yes	yes	no	no	yes	
Danville	yes	yes	yes	no	yes	
Dublin	yes	yes	yes	no	yes	
East Palo Alto	yes	yes	yes	no	yes	
El Cerrito	yes	yes	yes	no	yes	
Emeryville	yes	yes	yes	no	no	
Fairfax	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
Fairfield	yes	yes	yes	no	yes	Detention basins, junction boxes, and box culverts
Foster City	yes	yes	yes	no	no	

Agency	Maintenance Practice					
	Street/Parking Lot Sweeping	Inlet Cleaning	Storm Drain Line/Ditch Cleaning	Pump Station Cleaning	In-Stream Sediment Removal	Other
Fremont	yes	yes	no	no	no	
FSSD	no	no	no	yes	no	
Half Moon Bay	yes	yes	yes	no	yes	Junction boxes
Hayward	yes	yes	yes	no	yes	
Hercules	yes	yes	yes	no	yes	
Hillsborough	yes	yes	yes	no	no	Junction boxes
Lafayette	yes	yes	yes	no	yes	
Larkspur	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
Livermore	yes	yes	yes	yes	yes	
Marin County	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
Martinez	yes	yes	yes	no	yes	
Menlo Park	yes	yes	yes	yes	yes	Junction boxes
Mill Valley	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
Millbrae	yes	yes	yes	yes	yes	Junction boxes
Milpitas	no	no	no	no	no	
Moraga	yes	yes	yes	no	yes	
Mountain View	no	no	no	no	no	
Newark	yes	yes	yes	yes	no	
Novato	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
Oakland	yes	yes	yes	yes	yes	Junction boxes
Orinda	yes	no	no	no	no	
Orinda	yes	yes	yes	no	yes	
Pacifica	yes	yes	yes	no	yes	
Palo Alto	no	no	no	no	no	
Piedmont	yes	yes	yes	no	yes	
Pinole	yes	yes	yes	no	no	
Pinole	yes	yes	yes	no	no	
Pittsburg	yes	no	no	no	no	

Agency	Maintenance Practice					
	Street/Parking Lot Sweeping	Inlet Cleaning	Storm Drain Line/Ditch Cleaning	Pump Station Cleaning	In-Stream Sediment Removal	Other
Pleasant Hill	yes	yes	no	no	no	
Pleasanton	yes	yes	yes	yes	yes	
Portola Valley	yes	yes	yes	no	yes	Junction boxes
Redwood City	yes	yes	yes	yes	yes	Junction boxes
Richmond	yes	yes	yes	yes	no	
Ross	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
San Anselmo	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
San Bruno	yes	yes	yes	yes	yes	Junction boxes
San Carlos	yes	yes	yes	yes	yes	
San Jose	no	no	no	no	no	
San Leandro	yes	yes	yes	no	no	
San Mateo County	yes	yes	yes	yes	yes	Junction boxes
San Pablo	yes	yes	yes	no	yes	
San Rafael	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
San Ramon	yes	yes	yes	no	yes	
Santa Clara	no	no	no	no	no	
Santa Clara Valley Water District	no	no	no	no	no	
Sausalito	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
South San Francisco	yes	yes	yes	yes	yes	Junction boxes
Suisun City	yes	yes	yes	no	yes	
Sunnyvale	no	no	no	no	no	
Tiburon	yes	yes	no	yes	no	Culverts, silt basins, lagoons, and detention ponds
Union City	no	no	no	no	no	
Vallejo	no	no	no	no	no	
Walnut Creek	yes	yes	yes	no	yes	
Woodside	yes	yes	yes	no	no	Junction boxes
WVCWP	no	no	no	no	no	

TableA-2. Survey Responses: Sediment Removed through Maintenance Practices

Maintenance Practice	Total Sediment Volume 2000-2001 (cu.yd.)	Total Sediment Volume 2001-2002 (cu.yd.)	Total Sediment Volume 2002-2003 (cu.yd.)	Total Sediment Mass 2000-2001 (tons wet)	Total Sediment Mass 2001-2002 (tons wet)	Total Sediment Mass 2002-2003 (tons wet)	Total Sediment Mass 2000-2001 (tons dry)	Total Sediment Mass 2001-2002 (tons dry)	Total Sediment Mass 2002-2003 (tons dry)
Street Sweeping Reported Separately	153,743	184,278	182,018			9,400			20
Other Practices Reported w/out Sweeping	15,692	10,460	12,073	9	9	80	9		
Sweeping Reported with Other Practices	3,190	3,401	3,301			803			
Total Sediment Removed - All Practices	172,625	198,140	197,392	9	9	10,283	9	0	20

Table A-3. Survey Responses: Number and Type of Treatment Control BMPs Installed by Agency.

Clean Water Program/ City/Town	Treatment Control BMPs															
	Extended Detention Basin (TC-22)	Wetpond (TC-20)	Constructed Wetland (TC-21)	Infiltration Basin (TC-11)	Infiltration Trench (TC-10)	Retention/Irrigation (TC-12)	Vegetated Swale (TC-30)	Vegetated Buffer Strip (TC-31)	Bioretention (TC-32)	Sand Filter (TC-40)	Media Filter (MP-40)	Water Quality Inlet (TC-50)	Wet Vault (MP-50)	Vortex Separator (MP-51)	Drain Inserts (MP-52)	Multiple System (TC-60)
Alameda County Clean Water Program																
Alameda														5	3	
Albany																
Berkeley												1			1	
Caltrans	2								2							
Dublin	1						2				1				4	
Emeryville							2					2			1	
Fremont		1					18					2			3	
Hayward	7						2							7	3	
Livermore							9							1		
Newark														3	5	
Oakland							1								1	
Piedmont																
Pleasanton		2					7				1			1		
San Leandro	1						1							1	2	
Union City							4							2		
Unincorporated Alameda County							1									
AC Flood Control/Water Control District		2	1													
Zone 7																
Contra Costa Clean Water Program																
Clayton																
Concord																
Danville																
El Cerrito																
Hercules																

Clean Water Program/ City/Town	Treatment Control BMPs															
	Extended Detention Basin (TC-22)	Wetpond (TC-20)	Constructed Wetland (TC-21)	Infiltration Basin (TC-11)	Infiltration Trench (TC-10)	Retention/Irrigation (TC-12)	Vegetated Swale (TC-30)	Vegetated Buffer Strip (TC-31)	Bioretention (TC-32)	Sand Filter (TC-40)	Media Filter (MP-40)	Water Quality Inlet (TC-50)	Wet Vault (MP-50)	Vortex Separator (MP-51)	Drain Inserts (MP-52)	Multiple System (TC-60)
Lafayette																
Martinez																
Moraga																
Orinda																
Pinole																
Pittsburg																
Pleasant Hill							1	1				1		1	1	
Richmond	1	2	1													
San Pablo																
San Ramon																
Walnut Creek	1			1			1								1	
Unincorporated Contra Costa County																
CCC Flood Control/Water Control District																
Fairfield-Suisun Urban Runoff Management Program																
Fairfield	33						44									
Suisun	8						2				6					
FS Sewer District																
Marin County Stormwater Pollution Prevention Program																
Belvedere	1															
Corte Madera							1									
Fairfax																
Larkspur																
Mill Valley																
Novato	1						1									
Ross																
San Anselmo																

Clean Water Program/ City/Town	Treatment Control BMPs															
	Extended Detention Basin (TC-22)	Wetpond (TC-20)	Constructed Wetland (TC-21)	Infiltration Basin (TC-11)	Infiltration Trench (TC-10)	Retention/Irrigation (TC-12)	Vegetated Swale (TC-30)	Vegetated Buffer Strip (TC-31)	Bioretention (TC-32)	Sand Filter (TC-40)	Media Filter (MP-40)	Water Quality Inlet (TC-50)	Wet Vault (MP-50)	Vortex Separator (MP-51)	Drain Inserts (MP-52)	Multiple System (TC-60)
San Rafael																
Sausalito																
Tiburon							1									
Unincorporated Marin County							15									
MC Flood Control District																
Santa Clara Valley Urban Runoff Pollution Prevention Program																
Cupertino	1				1		2									
Los Altos																
Los Altos Hills																
Milpitas							3								3	
Mountain View	3												2		30	
Palo Alto	2						7									
San Jose	2				1		4									
Santa Clara							1									
Sunnyvale					2		1									
West Valley Communities	1				1		10								3	
Unincorporated Santa Clara County																
SCV Water District																
San Mateo Stormwater Pollution Prevention Program																
Atherton					2		1									
Belmont	1			2			5									
Brisbane	2															
Burlingame							1	1							2	
Colma	6						4								4	
Daly City	1	2														
East Palo Alto	1						1								1	

Clean Water Program/ City/Town	Treatment Control BMPs															
	Extended Detention Basin (TC-22)	Wetpond (TC-20)	Constructed Wetland (TC-21)	Infiltration Basin (TC-11)	Infiltration Trench (TC-10)	Retention/Irrigation (TC-12)	Vegetated Swale (TC-30)	Vegetated Buffer Strip (TC-31)	Bioretention (TC-32)	Sand Filter (TC-40)	Media Filter (MP-40)	Water Quality Inlet (TC-50)	Wet Vault (MP-50)	Vortex Separator (MP-51)	Drain Inserts (MP-52)	Multiple System (TC-60)
Foster City																
Half Moon Bay	2						1					1				
Hillsborough							2									
Menlo Park					1											
Millbrae							2									
Pacifica							1									
Portola Valley	3				3		3									
Redwood City	2						2									
San Bruno	2													2		
San Carlos							3									
San Mateo	2		1				5								7	
South San Francisco											8					
Woodside							1									
Unincorporated San Mateo County	1	2														
Vallejo																
Vallejo Sanitation and Flood Control District																
Total	88	11	3	3	11	0	173	2	2	0	16	7	2	23	75	0

6. Summary – Overview of Knowledge (Strengths and Weaknesses)

6.1 Introduction

This section summarizes key knowledge (strengths and weaknesses) in an effort to provide justification for work and data needs. For detailed information and sources of that information, the reader should refer back to the individual sections of the report.

San Francisco Bay is listed as a water body impaired with Hg and PCBs under Section 303(d) of the federal Clean Water Act. Recent Total Maximum Daily Loads (TMDL) reports describe source assessments, numeric targets, a linkage analysis, load allocations, and a preliminary implementation plan for each substance. This current effort primarily aims to provide information to assist BASMAA and the Water Board refine the implementation plans for Hg and PCBs based on our best understanding of the sources, transport, and deposition of these substances throughout the urban environment and stormwater conveyance system. In addition, where possible, this effort will refer to benefits for other substances and beneficial uses.

Much (perhaps around 60%) of the Hg in the Bay is derived from mining wastes – a legacy of the gold rush of the mid 19th century and Hg mining in the Guadalupe River watershed (1850-1970). The main management solutions for the Hg problem are the reduction of urban runoff and reduction of Guadalupe River legacy mining loads to the Bay. The TMDL for Hg calls for a 48% reduction in urban runoff loads and a 98% reduction in Guadalupe River mining loads.

Most of the PCBs in the Bay were derived from urban runoff loads, wastewater loads, and shoreline activities over the period of peak usage from 1950-80. Wastewater loads have reduced from 1000s kg/y to ~2.3 kg/y (Jay Davis personal communication). The proposed solution to reduce impairment caused by PCBs in the Bay described in the TMDL is to reduce urban runoff load.

6.2 Summary of Knowledge

Peak urban, commercial and industrial usage of Hg and PCBs occurred between 1950 and 1990. Distributed (general) uses in the urban environment of both chemicals likely followed population trends. Santa Clara the most populous county in the Bay Area with a 1990 population of 1.5 M increased in population by 5.2 times over this period. The other large counties Alameda (1.3 M), Contra Costa (0.8 M), and San Mateo (0.6 M) increased in population during this period by 1.7 times, 2.7 times and 2.8 times respectively. In terms of population, 1.2 M people were added to Santa Clara County during this period, whereas about 0.5 M were added to Alameda, Contra Costa, and San Mateo counties. Thus, we would intuitively expect the greatest impact from general urban use of Hg and PCBs in Santa Clara County. Although population most definitely influenced the use of these substances, industrial uses were substantial and left a legacy of polluted soils (hotspots).

Industrial use of Hg and PCBs occurred mainly on the fringe of the Bay bracketing the main highways and railway routes in Contra Costa, Alameda, Santa Clara, San Mateo, and South San Francisco. Industrial areas make up about 6% of the Bay Area local tributaries today and it was probably about 9% during the 1950-90 period. This was the primary use area for both chemicals and (based on hypotheses generated from a review of world literature) likely has a general soils

concentration that is greater than in urban areas and open space/agricultural areas. In addition, the industrial zone contains many known hotspots that can have soil, and sediment concentrations several orders of magnitude higher than the general industrial zone median.

Sources that are distributed in association with population trends and general urban usage are farther from the Bay and will have a lower connectivity because of lower imperviousness / runoff coefficients and greater opportunity for permanent storage in bed, bank, and bar deposits in upland and midland creeks and stormwater conveyances. Industrial source were and still are proximal to the Bay and are highly “geomorphically” connected due to greater imperviousness / runoff coefficients and shorter travel distances to the Bay in stormwater conveyances where deposition is undesirable.

Historically, the greatest uses of Hg were batteries>paint>laboratory>”other uses”. Today, the annual average usage has dropped to about 7,000 t down from a 1950-90 average of 13,000 t. Today’s main uses are “other uses”>batteries>instruments>dental>laboratory>lighting. Historically, PCBs were mostly used in transformers and large capacitors (~60%) and plasticizers (25%). Today there is no new use but there is still legacy use that is gradually being phased out. Hg and PCBs differ substantially in their spatial usage patterns. The largest two uses of Hg (batteries and paint) were dissipative uses, whereas the largest use of PCBs were associated with power distribution and factories with high electricity demand.

Based on an extensive literature survey, we are able to hypothesize Hg and PCB concentrations in a range of urban setting and urban media (Table 6-1). Across all media, both Hg and PCBs show a log-normal concentration frequency distribution (median<mean) indicative of the spatial distribution in the urban landscape. The median is the best statistic for comparisons between media and the maximum concentration is the best indicator of the magnitude of pollution associated with a hotspot or use area. Hg concentrations are enriched in street dust relative to local soils due to preferential erosion of fine materials from local soils in addition to traffic related sources. It appears that the enrichment factor (EF) is greater in industrial areas relative to residential and commercial areas probably because of greater traffic related sources (e.g. Hg in diesel, oils and greases and PCBs in heavy oils predating 1979). Based on one piece of literature finer particles (< 0.15 mm) in street dust are enriched relative to coarser particles. Based on our own data from Guadalupe River, we know that finer particles are enriched relative to coarser particles in large stormwater conveyances. We have no data for PCBs but they would likely show the same patterns for EF and particle size distribution.

Past work has indicated that Hg and PCBs are largely transported in stormwater in the particulate phase. Urban sediments mobilized by rainfall-runoff are polluted with Hg or PCBs and thus a source of Hg and PCB in stormwater runoff. Major areas of concern are polluted industrial yards, soils or dust accumulated on impervious surfaces near buildings receiving fallout and washoff of paints and other building materials containing PCBs (e.g., caulking) and Hg. Sediment may also be relatively unpolluted by Hg and PCBs (e.g., subsoils eroded during earthworking operations) and act as a diluent for Hg and PCB pollution.

The major mobilization process in urban areas is rainfall-runoff from impervious surfaces. Accumulation of sediment on these surfaces is enhanced by vehicle tracking and dust re-suspension and deposition. Suspended sediment concentrations in built-up areas are typically 20-100 mg/L nationally, 50-200 mg/L in San Francisco urban watersheds, and 500-1000 mg/L in San Francisco Bay watersheds. Higher loads of SS are probably dilutants for Hg and PCB levels in urban drainage.

Table 6-1. Summary of Hg and PCB concentrations in soils, roof tops, street sweepings, and street dust (from world literature survey), and stormwater conveyance sediments (BASMAA).

Media	Land use category		Hg (mg/kg)	PCB (mg/kg)
Soils	Open / agriculture / remote	Minimum	0.037	0.0010
		Maximum	0.32	0.13
		1st quartile	0.048	0.012
		3rd quartile	0.090	0.031
		Median	0.053	0.020
		Mean	0.10	0.034
	Urban with no discernable industrial impacts	Minimum	0.15	
		Maximum	0.44	
		1st quartile	0.15	
		3rd quartile	0.37	
		Median	0.16	
		Mean	0.25	
	Agricultural with industrial influence	Minimum	0.42	
		Maximum	31	
		1st quartile	0.46	
		3rd quartile	3.8	
		Median	0.84	
		Mean	7.3	
	Urban with industrial influence	Minimum		0.0013
		Maximum		6.8
		1st quartile		0.010
		3rd quartile		0.16
		Median		0.092
		Mean		0.82
	Residential on industrial fringe and industrial	Minimum	0.35	
		Maximum	230	
		1st quartile	0.51	
		3rd quartile	3.5	
		Median	2.3	
		Mean	21	
	Industrial	Minimum		0.18
		Maximum		510,000
		1st quartile		4.6
		3rd quartile		590
		Median		11
		Mean		40,713
Street Sweepings		Minimum	0.050	
		Maximum	0.098	
		1st quartile	-	
		3rd quartile	-	
		Median	-	
		Mean	0.074	

Table 6-1 continued.

Media	Land use category		Hg (mg/kg)	PCB (mg/kg)
Roof tops		Minimum	0.18	
		Maximum	0.31	
		1st quartile	-	
		3rd quartile	-	
		Median	-	
		Mean	0.25	
Street Dust		Minimum	0.50	0.03
		Maximum	40	7.3
		1st quartile	0.75	0.12
		3rd quartile	7.0	1.3
		Median	0.85	0.29
		Mean	6.5	1.1
Stormwater Conveyance Sediments (BASMAA)	Open	Minimum	0.020	0.00020
		Maximum	0.29	0.030
		1st quartile	0.030	0.00030
		3rd quartile	0.053	0.0039
		Median	0.040	0.0011
		Mean	0.061	0.0041
	Mixed	Minimum	0.030	0.00024
		Maximum	1.86	3.3
		1st quartile	0.090	0.0047
		3rd quartile	0.26	0.071
		Median	0.14	0.019
		Mean	0.22	0.14
	Res./Com.	Minimum	0.020	0.00020
		Maximum	4.26	17
		1st quartile	0.16	0.018
		3rd quartile	0.51	0.18
		Median	0.20	0.063
		Mean	0.48	0.77
	Industrial	Minimum	0.040	0.0040
		Maximum	3.04	27
		1st quartile	0.12	0.033
		3rd quartile	0.35	0.23
		Median	0.24	0.094
		Mean	0.40	0.90

Suspended sediment in urban runoff can be usefully categorized as colloidal (<1 μm), suspended (<25 μm), settleable (25 – 75 μm), (bed) sediment (>75 μm), and gross solids (>10 mm), where these nominal sizes are based on a particle specific gravity of 2.65 although there is evidence of lower density particles in some settings. A wide range of particle sizes are observed in urban stormwater, but studies fall into two major particle size groupings. Most studies find much of the mass in suspended and settleable solids. A few studies have found that much larger particle sizes dominate. We have ascribed the differences to differences in mobilization energy and problems with suspended sediment measurements. Particle densities range from about 1 to 3 g/cm^3 depending on mineralogy and organic matter content. The low density particles may be important in transporting PCBs. They will be difficult to settle and separate from stormwater.

Settling is a much more important property in the transport and fate of particulates than particle size because of the effect of particle density. The few studies conducted indicate particle settling rates typical of suspended and settleable solids. Unhindered or discrete settling occurs at low concentrations of particulates, but flocculation can occur at the concentrations typically encountered in Bay watersheds (50 – 200 mg/L). The flocculation process is usually slow, however, compared with the time of transit in the drainage system, unless water is stored or ponded.

The urban drainage network morphology interferes with the natural drainage system processes (e.g., by preventing access of streams to the flood plain, increasing stream power), which usually means that sediments are transported more efficiently and quickly than in natural systems. However, natural system processes are still operating in the upper reaches of Bay watersheds. Sometimes urban drainage systems encounter engineered flood channels across slope. In these channels, stream power is considerably reduced and sediment accumulation occurs.

Deposition processes at the Bay margins depend on storm size and the state of the tide. At low tide, sediments can be carried far out into the Bay via the low tide channels. At high tide, most sediment is expected to settle in the upper reaches. Natural redistribution processes, such as settling and scour lag, tend to shunt fine particles up-estuary, so there is a tendency for pollutants to accumulate in the upper estuary.

Organic matter is a key component in Hg and PCB sequestering in particles and their transport and treatability. Two major types are observed. Amorphous organic matter is largely humic acids, bacteria etc and is related to particle surface area (and hence particle size). Black (glassy) carbon is formed by mantle diagenetic processes (e.g., coal, kerogen) or thermal processes (char, coke, activated charcoal, soot). Black carbon is highly effective in dissolving and rendering PCBs unavailable to biota. While it is unlikely to be widespread in San Francisco Bay watersheds or the Bay itself, it will be important to take into account in our subsequent monitoring studies. Hydrous ferric oxide (FeOOH) has a relatively high concentration and surface area in particles and may be important in the transport of Hg. It is relatively easy to measure and to check its role in Hg transport in urban watersheds.

The particle size distribution of Hg and PCB in urban stormwater is unknown. The particle size characteristics of other similar pollutants (other trace metals, PAH) show a wide variety of particle size associations, including an inverse relationship with particle size, and higher concentrations with large particles. In some cases, even when greater concentrations are associated with a specific particle size range, more mass can be associated with another size range (for example the work of Sansalone). Concentrations across the medium silts to fine sands (>10 – 250 μm) are often reasonably homogeneous, and most variation with particle size is more apparent at smaller (<10 μm) and larger (>250 μm) particle sizes. While the inverse relationship between concentration and particle size can be observed, the greatest mass of pollutant is usually carried by the particle sizes with the greatest mass, which are usually not the smallest, and most polluted, particles. Chemical processes will tend to favor the particulate nature of Hg and PCBs, and this is confirmed in most studies. Some studies have found a dominant soluble phase for Hg or PCBs, but this is unlikely under the chemical characteristics of San Francisco runoff.

Only a small portion of the total mass entering the system will find its way into stormwater conveyances and out into the Bay. Based on our best estimates, it appears that the largest sources of Hg to Bay Area stormwater conveyance channels are Watershed Surface Sediment Erosion >

Atmospheric Deposition > Instruments > Bed and Bank Erosion > Switches and Thermostats > Industrial Hotspots > Fluorescent Lighting (Figure 6-1). Based on our best estimates, it appears that the largest sources of PCBs to Bay Area stormwater conveyance channels are Watershed Surface Sediment Erosion > Industrial Hotspots > Transformers and Large Capacitors > Railway Lines > Building Demolition and Remodeling > PCBs Still in Use (Figure 6-2). In terms of treatment options, we recognize the uses/use areas can be grouped according to their characteristics (for example uses that can be recycled). The groups are shown in Figures 6-1 and 6-2 and are referred to in Table 6-2.

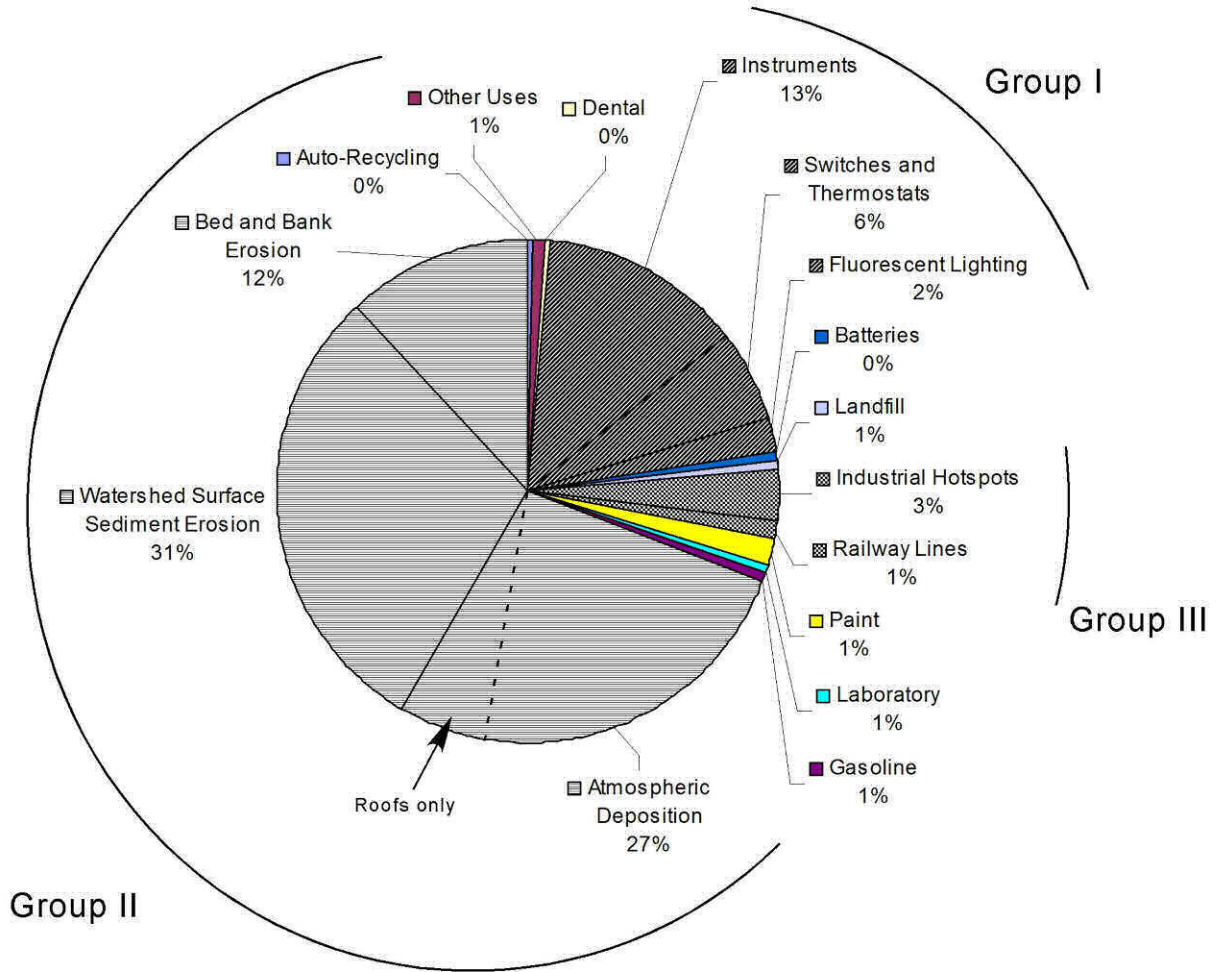


Figure 6-1. The mass of Hg entering stormwater conveyances from uses/use areas based on best estimates.

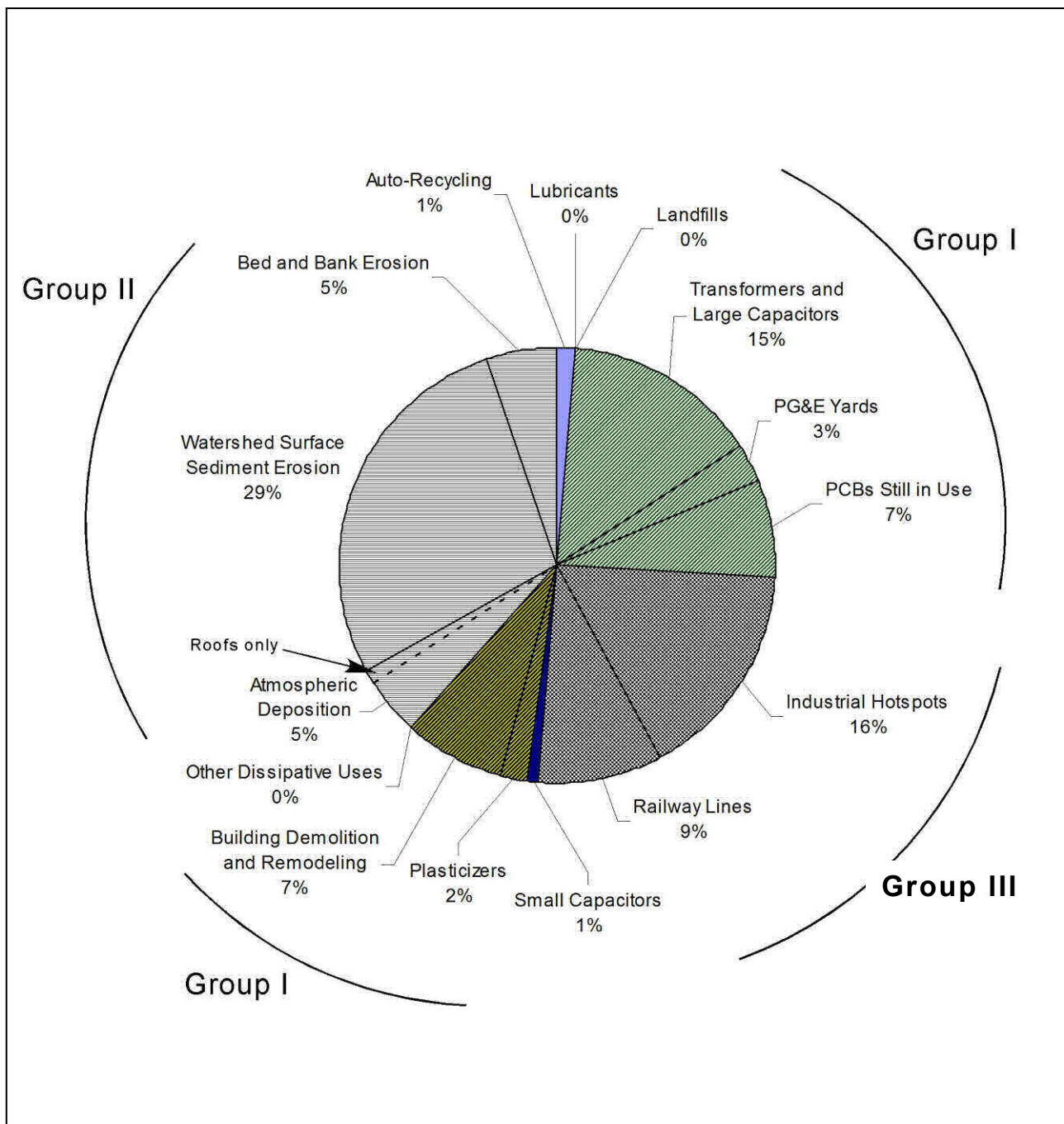


Figure 6-2. The mass of PCBs entering stormwater conveyances from uses/use areas based on best estimates.

Table 6-2. Summary of sources, controls, and implications for monitoring.

Source Grouping	Sources	Hg	PCB	Candidate Control (s)	Monitoring
I Product usage	Instruments, switches, and fluorescent lighting	√√		Recycling	Documentation
	Transformers, large capacitors		√√	Inspections, product phase-out, on-site treatment	Documentation
	Drying oils/Plasticizers or Softening agents (sealants, caulking/paints, coatings)/building demo and remod		√√	Education, recycling, treatment	Documentation
II Diffuse	Atmospheric deposition	√√	√	Street sweeping, storm drain maintenance, and treatment (e.g. roof disconnection)	Mass of pollutant removed per unit level of implementation (e.g., curb mile swept, inlet cleaned) for various levels of implementation
	Bed and bank pollution and erosion	√√	√	Hydro-modification / Channel de-silting	Measure volume and concentration in sediment removed
	Soil erosion from historical pollution	√√	√√	Hydro-modification and erosion control	Systematic data collection in soils in urban areas (ind, res, open)
III Industrial	Industrial hotspots/railway lines	√	√√	Site cleanup and/or treatment	Systematic monitoring of (1) bed sediment in industrial sewersheds, and (2) soils monitoring on railway sidings

Based on a literature review that included a mix of national and local information, pollution prevention, source controls, and treatment controls were evaluated in terms of potential for reducing loads of PCBs and Hg to San Francisco Bay. The evaluation is preliminary based on current information, and will be revised based on additional planned monitoring. The following are the conclusions based on this review.

- **Recycling** - Established successful programs that have prevented introduction of products containing pollutants to aquatic, air, and soils environment. Difficult to distinguish benefits to individual media. Have helped to set baseline loads and have some potential for achieving additional reductions in Hg (further legislative use bans) but are of limited use for further reductions of PCB loads to Bay.
- **Soil Remediation and Site Cleanup** - Legacy nature of PCBs and Hg use by industries supports this option. Initial studies conducted by stormwater agencies indicate elevated concentrations at some older industrial areas. Potential for load reduction would depend on proximity of site to storm drain system, potential for sediments to be mobilized into storm drain, efficacy of enforcement and cleanup. Masses removed from such sites may be significant relative to target load allocation. May be more applicable to PCBs.
- **Street Sweeping** - Ongoing funded maintenance activity. Incorporation of improved street sweeping fleet, more frequent sweeping, and targeted sweeping over time could improve removals of particulates and associated pollutants. Enhanced programs carry high price tag. No definitive field data that shows street sweeping as commonly employed actually

improves runoff water quality. Need to understand particle size relationships between particles on the street and particles in the hopper of the street sweeper (in relation to Hg and PCBs).

- ***Street Washing*** - Same issues as with street sweeping with additional issues relating to transport to the areas of capture and the capture process.
- ***Storm Drain Maintenance*** - Ongoing funded maintenance. Data indicate elevated concentrations at pump stations, especially in industrial areas. Targeting areas near hotspots and increasing frequency might enhance effectiveness.
- ***Channel De-silting*** - Data indicate elevated concentrations of Hg and/or PCBs in some stream sediments. De-silting of reaches with elevated concentrations could result in mass removed that would be significant relative to load reduction targets. Actual load reductions would depend on extent and mobility of polluted sediments. Significant permitting issues requiring multiple permits and permitting agencies.
- ***Treatment*** - Treatment effectiveness for TSS demonstrated. Effectiveness for treating PCBs and Hg depends on particle size and density of particles associated with PCBs and Hg. Retrofitting opportunities for wet ponds and wetland BMPs in urban areas are limited. Locating subsurface media filter type BMP is more feasible.

6.3 Data and Information Gaps

Based on our review of the literature, and critical discussion in the preceding five sections of this white paper, we have identified and prioritized a number of data gaps (Table 6-3). We aim to further test the importance of these data gaps during the next phase of the project (desktop modeling). We anticipate that some of these data gaps will be addressed during the field data collection phase of this grant.

Table 6-3. Data and information gaps discovered during the review contained in this white paper (sections 1-5).

Media	Date or information needed	Hg	PCBs	Reason	Priority
Fluorescent lighting	Mass associated with illegal disposal in dumpsters or on creek sides	√√		May substantially increase the estimate of mass entering storm drain conveyances and elevate the importance of policing such illegal disposal	Medium
Switches and thermostats	Mass lost during transport and recycling	√√		Mass may be currently over estimated (based on estimates for fluorescent lighting)	Medium
Batteries	Sales of each type of battery. Average weight of a battery based on total sales. Percent of batteries that are discarded or have access to stormwater conveyance systems	√√		Mass may be currently underestimated	Low
Instruments	Amount of breakage during use, transport, and recycling	√√		Mass supply to stormwater conveyances based on fluorescent lighting – high uncertainty. May have over estimated the importance of this source.	High
“Other Hg uses”	No explicit information available on the types of products in this category.	√√		Only made a wild stab at the loss to stormwater conveyances based on 0.01-0.1%.	Low
Drying oils/plasticizers or softening agents	There is limited literature and no local literature or studies on the amount of PCBs still distributed in buildings constructed during the 50s and 60s in the Bay Area.		√√	Presently the estimates are very rough for runoff surrounding buildings with caulking, building maintenance, and demolition.	High
“Hotspots”	Number, areas, and soil concentrations	√√	√√	The number of hotspots is the most important data gap. There could conceivably be 1x or 100x more that we collated in our inventory.	High
Railway lines	No local soils data for Hg, only limited soils data for PCBs	√√	√	Mass may be over estimated because literature is biased towards yard areas rather than the much larger area associated with general sidings where concentrations are likely to be lower.	High
Auto-dismantlers	No local soils data for Hg, only limited soils data for PCBs. Knowledge of locations limited because of much illegal activity.	√√	√	Mass associated with this is highly uncertain.	Low
Media					
Atmosphere	PCB dry deposition		√√	Dry deposition is a large fraction of total atmospheric supply of PCBs. We have no local data and the world literature shows it can be 2-10x greater than wet deposition on watershed surfaces.	Low
Soils	Concentration in surface 5 cm across the range of land use types	√√	√√	Need confirmation of the magnitude of the mass of Hg and PCBs associated with general soil erosion because this has a big impact on treatment options.	High
	Concentration in surface 5 cm in industrial sewersheds	√√	√√	Will help to prioritize industrial storm drain catchments for clean up or treatment.	Medium
Street sweepings	Concentrations in hopper materials	√√	√√	Need to determine if street sweeping is an effective method of removal per unit implementation (curb mile swept, inlet cleaned).	Medium
Street dusts	Concentrations on surfaces	√√	√√	Need to determine if street sweeping is effective and if street washing would be effective. Also be used to track mobile sediments derived from hotspots.	High
Conveyance sediments	Concentrations in relation to particles sizes and density and organic matter and FeOOH	√	√√	Need to determine if gravity settling type treatment BMPs will be effective and if so provide data for design.	Medium
	Concentrations in relation to particles size measured in deposited sediments	√	√√	Need to determine which industrial storm drain catchments near the Bay margin (usually completely engineered) have greatest pollution, produce the most sediment, and should be prioritized for clean-up or treatment.	High