SAN FRANCISCO ESTUARY PROJECT

STATUS AND TRENDS REPORT
on
POLLUTANTS
IN THE SAN FRANCISCO ESTUARY

FINAL DRAFT

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Disclosure Statement

This report was produced for the San Francisco Estuary Project under Cooperative Agreement CE-009496-01 between the United States Environmental Protection Agency (USEPA) and the Aquatic Habitat Institute (AHI), Richmond, California. Section IV.D., Pollutant Effects on Beneficial Uses, was initially drafted by Robert B. Spies of the Lawrence Livermore National Laboratory. Items relating to urban and nonurban runoff (including the existing management structure, present and future loads, and control strategies) were drafted by the Association of Bay Area Governments under contract with AHI. Text describing pollution prevention was drafted by the Subcommittee on Pollutants in the San Francisco Estuary, including representatives of Citizens for a Better Environment, USEPA, the Bay Area Dischargers Association and AHI. The remainder of the report was drafted by AHI. All conclusions and recommendations expressed in the report are those of the authors and do not necessarily reflect the views of the USEPA.

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This report has been reviewed by more than 50 readers; each of them has contributed to improving the document. We are grateful for their help. Three formal drafts have preceded this public version of the report.

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This report was produced for the San Francisco Estuary Project under a Cooperative Agreement between the U.S. EPA and the Aquatic Habitat Institute.

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STATUS AND TRENDS REPORT ON POLLUTANTS IN THE SAN FRANCISCO ESTUARY

EXECUTIVE SUMMARY[1]

The San Francisco Bay-Delta Estuary is among the largest in North America. It comprises the San Francisco, San Pablo, and Suisun bays, and the Delta of the Sacramento-San Joaquin rivers, two large rivers that drain the Central Valley of California. The Bay-Delta region is the fourth largest metropolitan area in the United States, consisting of twelve counties with a population of seven million people.

The Estuary provides important transportation routes for commerce and supports recreational activities such as boating and sport fishing. Also, it supplies water for drinking, industry, irrigation, and for other beneficial uses. It is a rich ecological system that provides a habitat for a diversity of organisms. For example, each year two-thirds of the State's salmon and nearly half of the waterfowl and shorebirds migrating along the Pacific Flyway pass through the Estuary.

Human activities have greatly affected many aspects of the Estuary including geography, hydrology, and ecology. The activities relevant to the discussion of pollutants in the Estuary include the introduction of sediments and metals from mining operations, the discharge of domestic sewage, the diversion of freshwater, and the release of persistent, toxic pollutants in industrial discharges and surface runoff.

Before 1970, upgrades in Bay Area sewage treatment systems and the California Porter-Cologne Act established precedents for water quality improvements. In 1972, environmental protection activities accelerated when Congress passed the Federal Water Pollution Control Act (Clean Water Act): "The objective of this Act is to restore and maintain the chemical, physical, and biological integrity of the nation's waters...it is the national goal that the discharge of pollutants into the navigable waters be eliminated...it is the national policy that the discharge of toxic pollutants in toxic amounts be prohibited."

Despite much progress, the goals and objectives of the Clean Water Act have not been fully achieved. There is concern that many toxic pollutants being discharged into the Estuary are affecting the health of aquatic life and wildlife, thereby reducing the viability of biological communities. Also, there is concern about the potential human health effects resulting from the consumption of fish and wildlife.

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[1] This Summary was written by the Subcommittee on Pollutants and Quality Assurance of the San Francisco Estuary Project, not by the authors shown on page ii.

This report explores our knowledge of Pollutants in the Estuary, identifies the pollutants of concern, discusses processes affecting their fate, transport, and effects, and identifies gaps in the understanding of these processes. The Subcommittee on Pollutants and Quality Assurance of the San Francisco Estuary Project recommends actions in Appendix III to protect the Estuary, and to establish a program of monitoring, data management, and research to improve the scientific basis for managing the Estuary.

STATUS

Each year, an estimated 5,000 to 40,000 metric tons of at least sixty-five toxic pollutants are disposed in the Estuary. The pollutants of concern include trace elements such as copper, nickel, silver, and zinc, and synthetic organic compounds such as organochlorine pesticides, polychlorinated biphenyls (PCBs), and polynuclear aromatic hydrocarbons (PAHs). These pollutants are produced and mobilized by numerous industrial, agricultural, natural, and domestic activities within the catchment. Pollutants are conveyed to the Estuary by rivers. Also, they reach the Estuary through storm drains, other runoff from urban and nonurban lands, wastewater treatment plants, industrial facilities, atmospheric deposition, discharges from maritime vessels, underground seepage, and disposal of dredged material.

To date, most pollution control efforts have focused on direct discharges of sewage and industrial waste. While these efforts continue, controlling pollutants released into urban and non-urban runoff has also become a priority. Pollutants in urban runoff originate from transportation activities (e.g., crankcase oil, combustion by-products, and tire wear) among other sources. Pollutants are deposited onto urban surfaces from the atmosphere and flushed from urban surfaces (e.g., pavement, yards, and construction sites) into storm drains by rainfall, landscape irrigation, and wash-down practices. Aside from San Francisco and a small part of Sacramento where most urban runoff is treated along with municipal wastewater, polluted runoff from urban areas flows untreated into the Estuary.

Nonurban runoff is defined as surface runoff from agricultural lands, range lands, and forests. Rainfall and irrigation water flush pesticides and other agricultural chemicals into drains, and the runoff flows untreated into the Estuary. Nonurban runoff also includes pollutants leached from soils by rain or irrigation (e.g., selenium), drainage from mine sites, and sediment from eroded soils.

Pollutants are distributed within the Estuary by a combination of physical, chemical, and biological processes. In the Estuary, many persistent pollutants become bound to particulate matter that settles near discharge points, and accumulates in areas of sediment deposition together with pollutants from past industrial activities. Some of these areas have been identified as "toxic hot spots". Pollutants can become concentrated in organisms directly from the

water column and by ingestion of contaminated food. These two processes can lead to high concentrations of pollutants in tissues even though concentrations in the water are low. Studies indicate that pollutants are having adverse effects:

- Water from some creeks and rivers in the Bay-Delta catchment, and some Bay sediments were toxic in bioassays at certain times of the year. High concentrations of agricultural pesticides in San Joaquin River water coincided with an acutely toxic response of the bioassay organism, Ceriodaphnia (a water flea).

- Stormwater runoff from the City of Sacramento was acutely toxic to Ceriodaphnia.

- Persistent pollutants appeared to cause sublethal effects on some species. For example, PCBs appeared to reduce reproductive success in starry flounder.

- Occurrences of micronuclei in red blood cells of starry flounder from throughout the Bay suggested an effect of genotoxic chemicals.

- Concentrations of PCBs and DDE in the eggs of Black-crowned Night Herons were correlated with decreased embryo size and eggshell thickness, respectively.

- Concentrations of silver in some South Bay clams appeared sufficiently high to exert toxic effects.

- Concentrations of silver, copper, and cadmium in South Bay clams were linked to local sources and to the variation of water circulation driven by changes in flow from the Sacramento and San Joaquin rivers.

- The California Department of Health Services issued warnings regarding the consumption of certain fish and ducks taken from the Estuary, due to mercury and selenium contamination, respectively.

This list is not intended to be complete nor does it imply any particular order of importance. In many cases, the hydrologic and chemical complexity of the Estuary has made it difficult to identify and to relate the specific effects to individual pollutants and their sources. There may be additional pollutants of concern. Also, there may be additional effects, particularly effects resulting from many pollutants acting together.

TRENDS

Ecological resources of the Estuary have declined due to the cumulative effects of pollutants, overfishing, destruction of wetlands and other habitats.

diversion of freshwater, introduction of exotic species, dredging, dredge spoil disposal, and other factors.

Trends in the inputs of many pollutants to the Estuary are associated more with specific human activities and the use of chemicals than with the number of people living in the catchment. Biodegradable pollutants from sewage have been increasingly controlled through municipal wastewater treatment. As a result, biochemical oxygen demand and concentrations of sewage bacteria decreased substantially despite increases in population. Sediment and mercury inputs resulting from hydraulic mining and gold processing decreased once these activities were banned. Also, levels of DDT and PCBs generally declined only after these substances were banned.

Persistent pollutants of concern in the Estuary have been increasingly influenced by chemical use and freshwater flow patterns. In contrast to trends in some biodegradable pollutants, trends in persistent pollutants are affected more by the use of chemicals than by treatment methods. Concentrations of toxic metals in sediments and certain organisms are high in some urban-industrial portions of the Estuary, and concentrations of most metals do not appear to be decreasing. These concentration of these metals correspond with the continued use of these materials within the catchment despite the treatment of wastewater. Unless patterns of chemical use and land development change, pollutant loads discharged into the Estuary via runoff are likely to increase. Also, increased diversion of freshwater inflow may further increase the concentration of some pollutants of concern in the Estuary.

Historically, efforts to reduce the input of pollutants to the Estuary have focused on treating direct discharges rather than examining the usage of toxic chemicals. Given the environmental problems described above and the great expense associated with new treatment technologies to control persistent pollutants, pollution prevention techniques represent a promising option for achieving reductions of pollutant loads. Reduced use of toxic chemicals resulted in a 70%-90% reduction in chromium and lead discharges at a local petroleum refinery, and was associated with lower copper discharge rates from a number of metal plating and electronics manufacturing plants.

NEXT STEPS

Scientific inquiries have identified pollution problems, and have identified directions for improvement. **Appendix III** recommends a range of management actions intended to restore and protect the Estuary while we improve our understanding of its complex ecosystem.

I. PREFACE

In recognition of the special need to protect water quality and natural resources of our Nation's estuaries, Congress, in its amendments to the Clean Water Act, passed the Water Quality Act of 1987 and established the National Estuary Program (NEP). Congress charged the U.S. Environmental Protection Agency (EPA) with administering the NEP, and required that designated Estuary Projects develop Comprehensive Conservation and Management Plans (CCMP) within five-year planning periods.

As enabled by the Water Quality Act, the Governor of California nominated the San Francisco Bay/Sacramento-San Joaquin Delta for inclusion into the NEP. In response, the Administrator of EPA established the San Francisco Estuary Project (SFEP) in April 1988. EPA and the State convened a Management Conference consisting of over one hundred participants who represent all levels of government, environmental groups, farming interests, business, and industry. The Conference outlined the following four goals:

1. Develop a comprehensive understanding of environmental and public health values attributable to the Bay and Delta and how these values interact with social and economic factors.

2. Achieve effective, united, and ongoing management of the Bay and Delta.

3. Develop a CCMP to restore and maintain the chemical, physical, and biological integrity of the Bay and Delta, including restoration and maintenance of water quality, a balanced indigenous population of shellfish, fish, and wildlife, and recreation activities in the Bay and Delta, and assure that the beneficial uses of the Bay and Delta are protected.

4. Recommend priority corrective actions and compliance schedules addressing point and nonpoint sources of pollution. These recommendations will include short and long-term components based on the best scientific information available.

The Conference then identified five Management Issues as those that had the greatest bearing on the ecological integrity of the Estuary: (1) Decline of Biological Resources, (2) Increased Pollutants, (3) Freshwater Diversion and Altered Flow Regime, (4) Increased Waterway Modification, and (5) Intensified Land Use. SFEP is scheduled to complete the CCMP by November 1992; it will then be submitted to the Governor and the EPA Administrator. Once approved, the CCMP will guide local, State, and federal agencies in efforts to improve protection of the Estuary.

The first step in the CCMP process is to increase the understanding of each Management Issue. As part of this characterization effort, SFEP is

preparing a series of Status and Trends Reports (STRs) that are being overseen by individual Subcommittees. For each STR, the respective Subcommittee seeks to develop a scientific consensus on what is known and not known about the Management Issue.

STRs are being prepared on: (1) **Wetlands and Related Habitats**; (2) **Aquatic Resources**; (3) **Wildlife**; (4) **Pollutants**; (5) **Dredging and Waterway Modification**; and (6) **Land Use and Population**. In addition, SFEP is studying: **land use** to assess the relationship between land use change and regulation on the future environmental health of the Estuary; **freshwater flows** to estimate the effect of flow variability on fluid dynamics and some ecological processes, and to evaluate the freshwater needs of various biological parameters; **Quality Assurance/Quality Control** procedures to examine opportunities for enhanced sampling techniques and laboratory analysis; and the current system of **laws and regulations** to analyze the management of the Estuary's resources.

The characterization effort will culminate in the completion of a "State of the Estuary" report that summarizes available information from the individual technical reports and provides an objective assessment of current conditions in the Estuary. This report will set the stage for developing the CCMP with its attendant management recommendations.

This Status and Trends Report on Pollutants in the San Francisco Estuary is the product of more that a year's effort by members of SFEP's Subcommittee on Pollutants and Quality Assurance, the Aquatic Habitat Institute, and SFEP staff. Consistent with Section 320 of the Water Quality Act, this STR utilized existing data to assess the relationship between estuarine borne pollutants and the impact on water quality and biota. The Subcommittee chose not to explore the effects of pollutants on drinking water supplies and problems related to water purification.

This edition of the STR was preceded by three earlier drafts that were scrutinized by more than fifty individuals with a wide range of viewpoints and expertise. In conjunction with the review of this document, the Subcommittee developed a set of Goals and Management Actions to Address Pollution in the San Francisco Estuary; these recommendations are contained in Appendix III. Immediate Actions are defined as Management Actions that the Subcommittee agrees should be initiated before the CCMP is completed; Potential Actions are defined as recommendations that will be evaluated for possible inclusion into the CCMP.

To solicit additional input on the pollutants issue, SFEP will make this report available to the public and conduct a series of public workshops. Subsequently, SFEP will revisit the Goals and Management Actions, revise them as appropriate, and either facilitate their implementation or incorporate them into the CCMP. Using this approach, SFEP intends to develop a CCMP that responds to the findings of its technical reports and the desires of Project participants.

II. INTRODUCTION

The purpose of this report is to present a clear description of our present knowledge of chemical pollution and chemical pollutant effects in the San Francisco Bay-Delta Estuary (the Estuary). Much has been learned about pollutants and their effects in the Estuary since the 1950s, particularly with regard to the mass of pollutants entering the system (Gunther *et al.*, 1987), the potential effects that pollution and individual chemical pollutants may have on the beneficial uses of the system (Filice, 1954a, b; Phillips, 1987; Luoma and Phillips, 1988; Phillips and Spies, 1988), and the dynamic interactions of pollutants with the sediments, water column, and biota in the Estuary (Luoma and Cloern, 1982; Nichols *et al.*, 1986; Rozengurt *et al.*, 1987; Luoma and Phillips, 1988). Nonetheless, our understanding of the details of pollutant interactions within the ecosystem is incomplete. Additional research and monitoring are needed if we are to build upon the information that is presently available and develop a full understanding of the problem.

This report provides an overview of historic information as well as the present status and possible future trends related to the distribution, fate, and effects of chemical pollutants in the San Francisco Estuary. The findings of this report will contribute to the Comprehensive Conservation and Management Plan to be developed for the Estuary through the San Francisco Estuary Project of the U.S. Environmental Protection Agency (USEPA). This Introduction contains a brief description of the San Francisco Estuary ecosystem and discusses present concerns regarding the effects of pollutants on the ecology of the Estuary. In addition, the Introduction describes the contents and structure of the remainder of the report, as well as a summary of the institutional management structure in place to address pollution, pollutants, and pollutant effects in the Estuary.

A. ECOLOGICAL SETTING

The San Francisco Estuary is among the largest estuaries in North America, with a surface area of 1,240 square kilometers (km^2; 479 square miles [mi^2]) and a drainage basin of 152,500 km^2 (58,764 mi^2) (Conomos *et al.*, 1985). The Estuary consists of two distinct regions: the Sacramento-San Joaquin Delta (the Delta) and San Francisco Bay (the Bay) (Figure 1).

The Delta is an inland delta where waters of the Sacramento River, the San Joaquin River, and several smaller rivers converge into a network of channels covering 3,000 km^2 (1,158 mi^2) (Madrone Associates, 1980). The Delta is bounded on the east by the outwash plain of the Sierra Nevada, and on the west by the Coast Range. The northern boundary of the Delta is on the Sacramento River at Sacramento, and the southern boundary is on the San Joaquin River near Vernalis. The drainage basins of the rivers that empty into the Delta comprise about 37% of the land area of the State of California (Figure 2), and carry between 40% and 50% of the freshwater runoff in the State of California (Madrone Associates, 1980).

Figure 1. The San Francisco Estuary, including the Sacramento-San Joaquin Delta and San Francisco Bay.

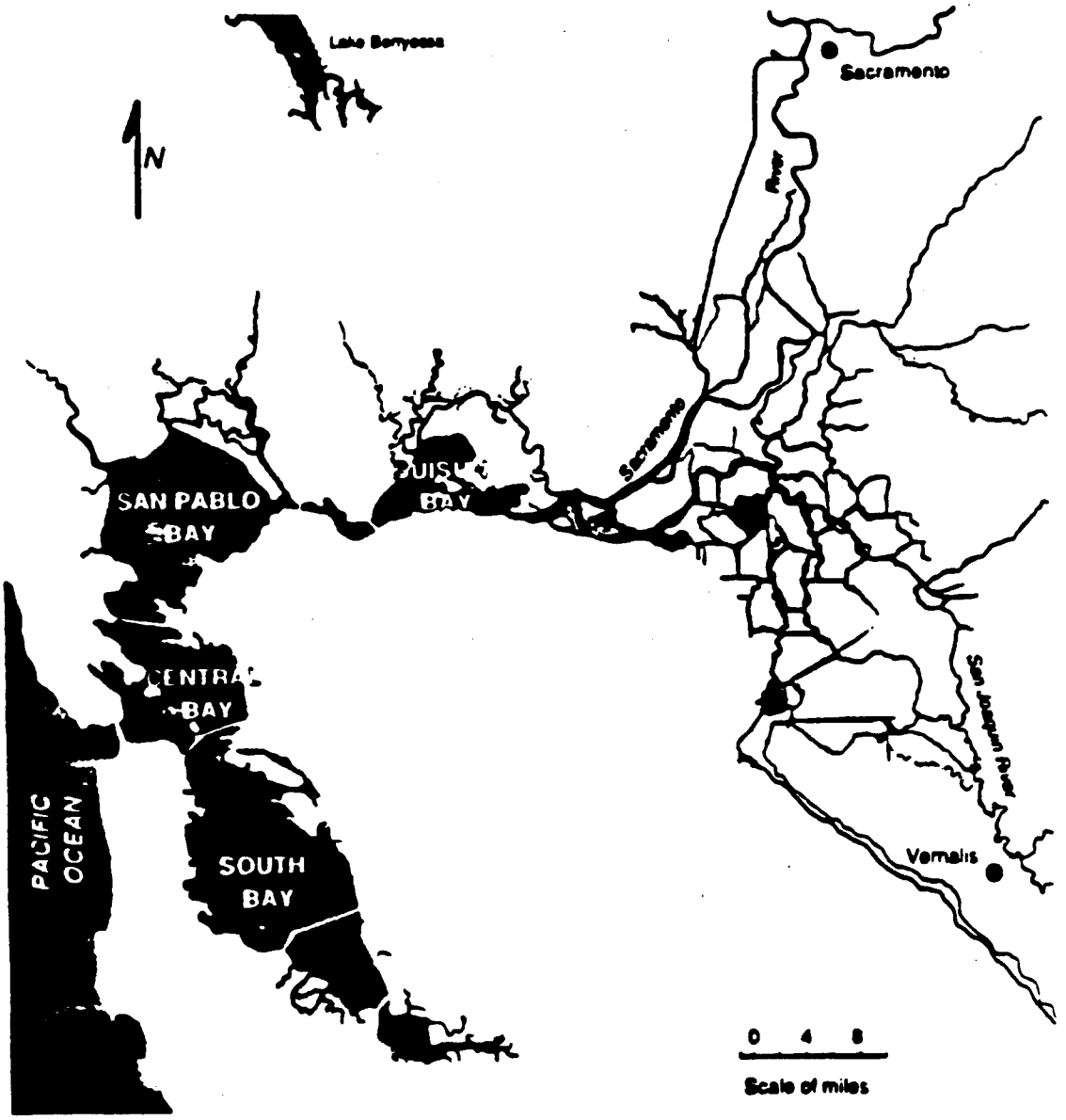
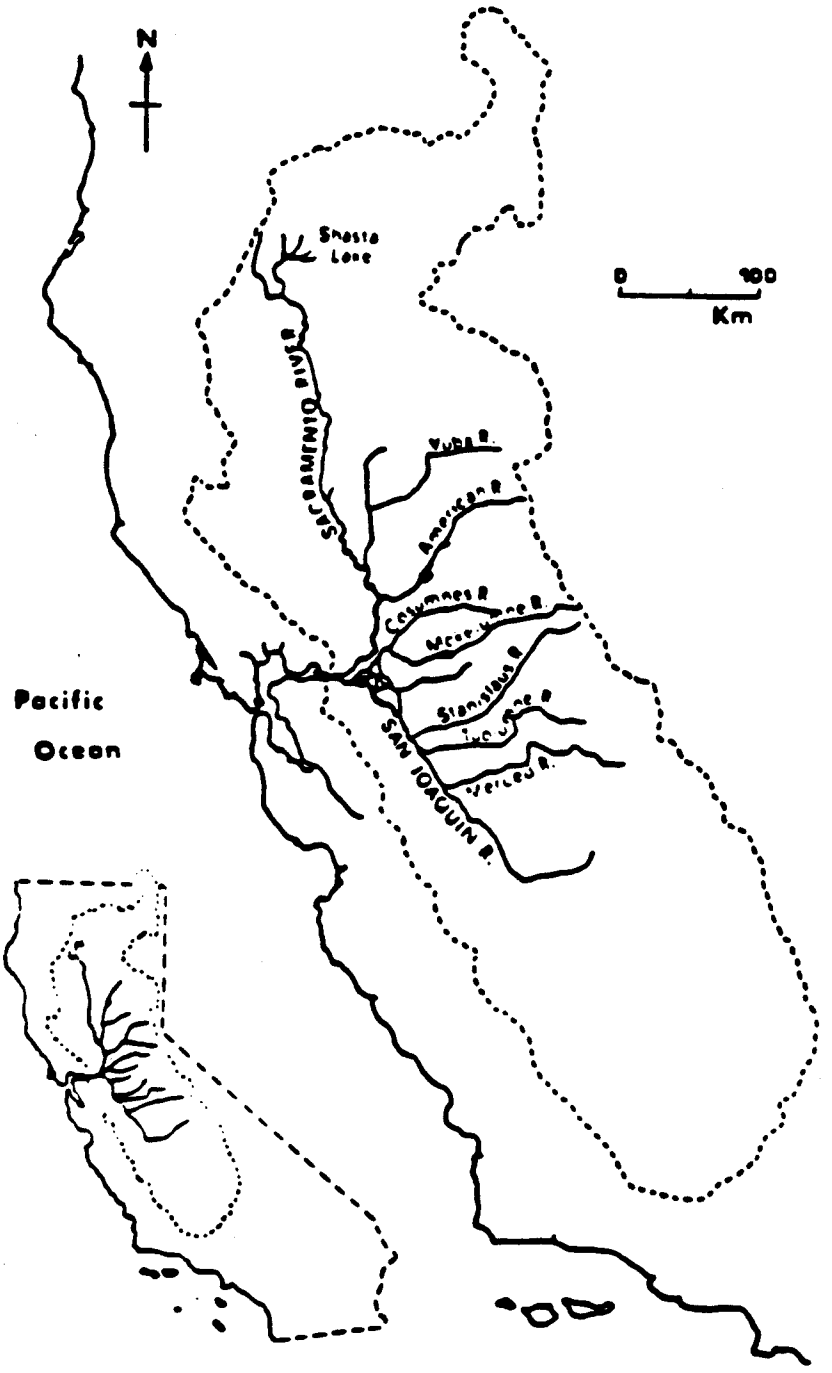


Figure 2. The major tributaries of the San Francisco Estuary. Dashed line indicates the drainage basin of the rivers that empty into the Delta. From Wright and Phillips (1988).



San Francisco Bay is a large coastal embayment that receives fresh water from the Delta and many other small tributaries. Four sub-embayments are described for San Francisco Bay. These are, from north to south: 1) Suisun Bay, 2) San Pablo Bay, 3) Central Bay, and 4) South Bay (Figure 1). The Golden Gate is the boundary between the Bay and the Pacific Ocean. All the subembayments of the Estuary undergo seasonal changes in the proportional mixing of fresh water with salt water. The salt content of estuarine waters is referred to as "salinity," and is expressed as grams of sea-salt per kilogram (kg) of water (parts per thousand). Full strength sea water has a salinity of about 34 parts per thousand, while the salinity of fresh water is zero.

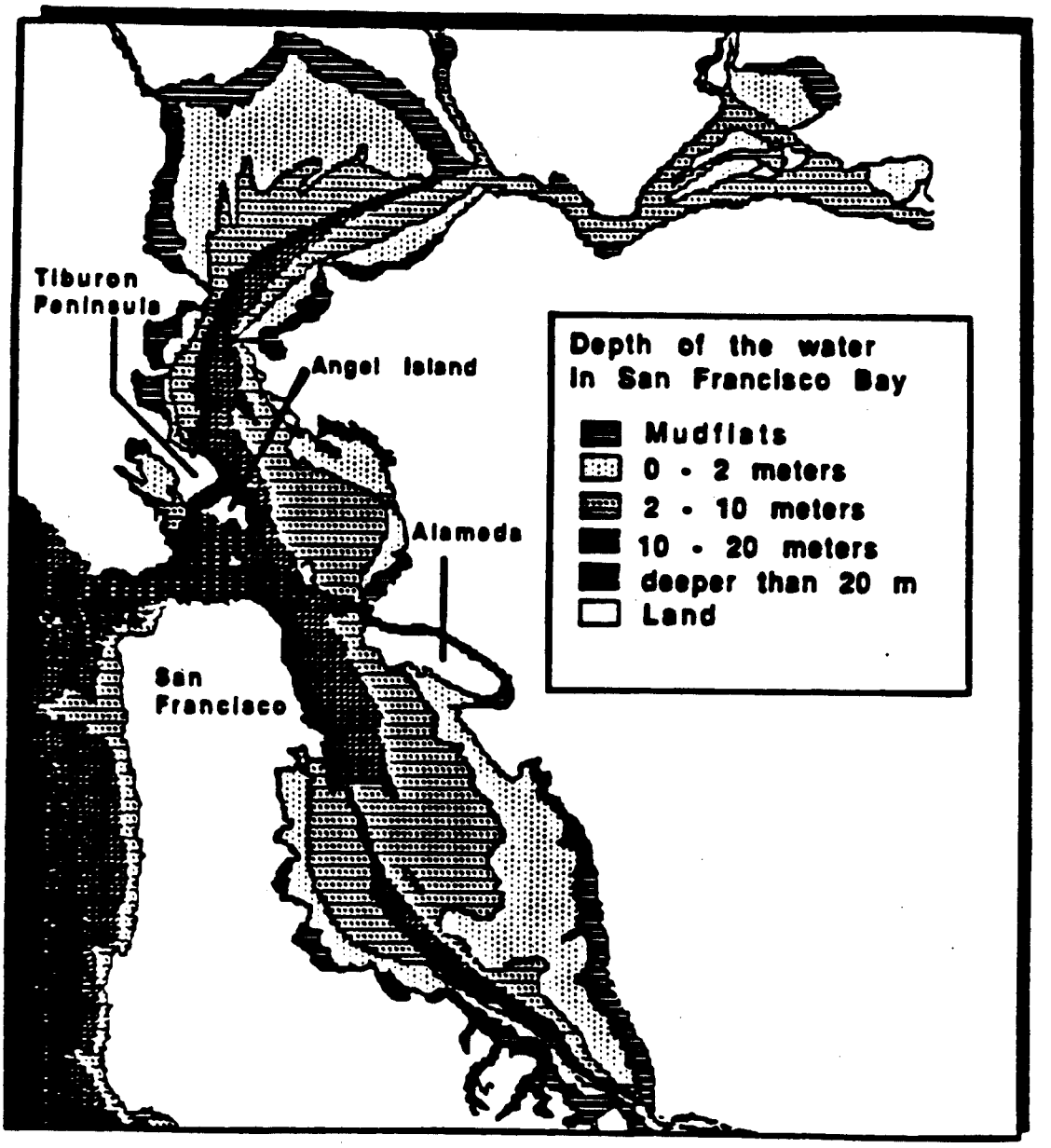
San Francisco Bay is relatively shallow, with an average depth of 6 meters (m) at mean lower low water (MLLW) and a median depth of about 2 m (Figure 3) (Conomos *et al.*, 1985). Suisun, San Pablo, and South bays are characterized by broad shallows incised by narrow channels whose depths are maintained by flow and tidal scouring. Central Bay is the deepest part of the Estuary, including a 110-m deep channel at the Golden Gate.

The Estuary is a highly dynamic environment. The two principal causes of this variability are tidal action and climate. The tidal cycle is 24 hours (hr), 50 minutes, with two high tides and two low tides in each cycle. Tidal amplitude or tidal range (the change in depth between mean low water and mean high water) at the Golden Gate is 1.7 m. The morphometry of the Estuary causes an increase in the tidal range in the South Bay (2.6 m) and a relative decrease in the northern reach (e.g., 1.3 m at Suisun Bay). These tidal ranges are large relative to average water depths, creating a tidal prism that is 24% of the total volume of the Bay. Morphometry and bathymetry also affect the timing of maximum tidal currents (e.g., the South Bay is flooding while the northern reach is ebbing), allowing for a tidally driven exchange of water between the north and south portions of the Bay. The typical tidal excursion in the Bay (the horizontal distance a parcel of water moves during a flood tide or an ebb tide) is on the order of 10 km (Conomos, 1979).

Tidal influence also extends throughout the Delta, where complex physiography causes variation in the timing and range of tides in different places. At Sacramento, in the northern Delta, the tidal range is 0.9 m; near Tracy, in the southern Delta, the range is 0.7 m. Upstream tidal points in the Delta may lag behind tides at the Golden Gate by as much as 10 hr.

The climate of the Bay-Delta region causes substantial seasonal change in the Estuary, principally due to seasonal variation in rainfall and wind. The Delta region and most of the Bay drainage basin have a Mediterranean climate; hot, dry summers and cool, moist winters. Consequently, surface runoff is much greater in the winter than in the summer. Seasonal differences in river flow cause large differences in freshwater inflow to the Estuary. Winter freshwater flows range from 1,000 to 10,000 cubic meters per second ($m^3 s^{-1}$; 35,300 to 353,000 cubic feet per second; cfs). Summer freshwater flows range from 100 to 400 $m^3 s^{-1}$ (3,600 to 14,300 cfs) (Conomos *et al.*, 1985).

Figure 3. Bathymetry of San Francisco Bay.



Total surface runoff in the basin varies substantially from year to year. Estimates of Delta outflow for "water years" (October to September) 1955/56 to 1983/84 varied by more than a factor of 25. In water year 1976/77 (year two of a 2-yr drought), for example, total Delta outflow was estimated to be 3.1 km^3, while in water year 1982/83 total Delta outflow was approximately 79 km^3 (Figure 4) (DWR, 1987).

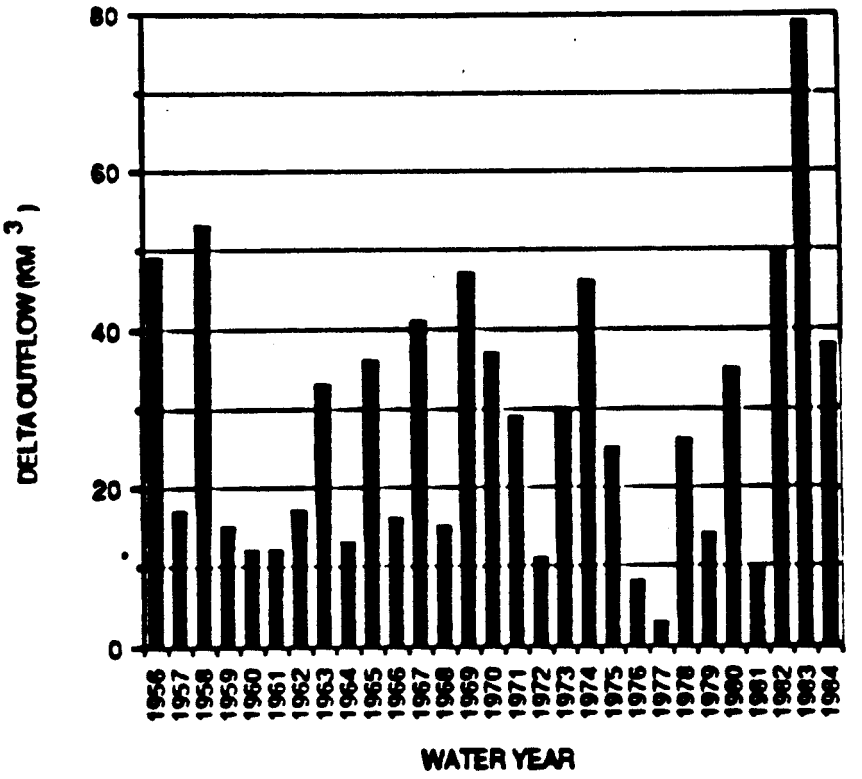
The seasonal and year-to-year extremes in runoff cause wide variation in the physical, chemical, and biological characteristics of the Estuary. Freshwater inflow to the Estuary, for example, affects within-year and between-year salinity, turbidity, and circulation patterns (Conomos, 1979). During low-flow periods, for example, measurable salinity may intrude upstream as far as the confluence of the Sacramento and San Joaquin Rivers. During the periods of highest outflow low salinity water (about 1 part per thousand) may be found as far downstream as San Pablo Bay Conomos, 1979; Davis, 1982).

Delta outflow also influences several other ecological parameters such as temperature gradients in the brackish water portion of the Estuary (Rozengurt *et al.*, 1987), flushing time of the Estuary, mixing and transport of suspended sediments and transport and dilution of pollutants.

The northern reach of the Bay (Suisun, San Pablo, and Central Bays) receives 90% of the Bay's total freshwater inflow and exhibits the gravitational circulation pattern typical of partially mixed estuaries; surface currents of lower salinity flow toward the sea, while bottom currents of higher salinity flow landward. Mixing between upper and lower water masses results in a gradual transition from the low salinity of the freshwater sources to the higher salinity of Central Bay and the ocean. At a point known as the "*null zone*," the "turbidity maximum," or the "entrapment zone," mixing of fresh and saline water creates an accumulation of nutrient-laden suspended particles, phytoplankton, and zooplankton. Surface salinity in the *null zone* ranges from approximately 1 to 6 parts per thousand (Arthur and Ball, 1979). The position of the null zone in the Estuary may change by up to 50 km in response to Delta outflow (Conomos, 1979). High freshwater flows move the null zone "downstream," toward San Pablo Bay. During periods of lower freshwater flow tidal intrusion may move the null zone upstream into Suisun Bay or the western Delta.

The southern reach of the Bay-Delta Estuary (South Bay) receives less than 10% of the fresh water entering the Estuary. Unlike the northern reach, freshwater inflow in the South Bay are rarely sufficient to drive gravitational circulation. Salinity in the South Bay is often close to that of the ocean, and during extended droughts, evaporative loss of fresh water can cause South Bay salinity to exceed that of the ocean (Conomos, 1979). During much of the year the South Bay can be considered a tidally oscillating lagoon (Conomos *et al.*, 1985). When Delta outflow is high, in the winter, significant transport of low salinity water into Central Bay can cause reductions in the salinity in South Bay Such conditions can generate a gravitational circulation cell between South and Central bays (Smith, 1987; see Section IV.C.).

Figure 4. Estimated volumes (km^3) of total Delta outflow from water year 1956 (i.e., October 1955 to September 1956) to water year 1984. Data from DWR (1987).



Climate in the Bay Area is transitional between coastal and inland climates and is heavily influenced by the continuous interaction of oceanic and continental air masses. These air mass interactions result in strong winds during the summer months and less windy conditions at other times of the year. Given the fact that much of the Estuary is very shallow (more than 70% of the Estuary is less than 5 m deep) seasonal winds have a strong influence on water circulation and water column conditions in the Bay. Prevailing summer winds are from the west and northwest, reinforced by an inland movement of air caused by solar heating of air masses in the Central Valley (Conomos *et al.*, 1985). In winter, wind speed and direction are influenced by storms that pass through central California. Prevailing summer winds generate waves with maximum periods of 2-3 s and wave heights exceeding 1 m. Winter storms can generate waves with a period of 5 s (Conomos *et al.*, 1985). These wind-generated waves resuspend sediment in the shallow portions of the Bay, creating turbid conditions and dispersing particulate matter throughout the Bay. In the South Bay, the steady northwesterly winds of the summer generate surface currents flowing to the southeast. These surface currents establish compensatory, opposite currents in deeper waters that flow to the northwest and induce mixing between the deeper waters of the Central Bay and the South Bay.

In summary, California's climate and the physiography of the Estuary combine to create an extremely dynamic environment. Particularly notable in this regard are the shallow depths of the Estuary, which intensify the effects of tides and allow for resuspension and transport of sediments, and the marked seasonal and annual variation in freshwater inflow, which affects water chemistry throughout the system and influences biological productivity at all trophic levels. The existence of several sub-units within the Estuary representing distinct habitats (the low-salinity channels of the Delta; the gravitational circulation and variable salinity of the northern reach; the deep, saline waters of the Central Bay; and the shallow, saline waters of the South Bay) contributes to the complexity of the ecosystem.

B. MODIFICATIONS TO THE ESTUARY

The following human activities collectively have caused fundamental changes in the Bay-Delta ecosystem (Nichols *et al.*, 1986):

1) Hydraulic mining during the Gold Rush period, which resulted in heavy sedimentation in the basin;
2) The introduction of exotic species (finfish, shellfish, and their associated symbiotic and parasitic fauna);
3) Diking of tidal marshes;
4) Filling of the margins of the Bay;
5) Water management, including the storage of surface runoff in the basin and the diversion of large quantities of fresh water from the Delta;
6) The discharge of pollutants into the Estuary;
7) The harvesting of fish and shellfish; and

8) The dredging of channels for navigation and the disposal of dredged material.

The precise effects of any of these modifications on the Bay-Delta Estuary and its beneficial uses are complex and difficult to determine (Nichols *et al.*, 1986; Phillips, 1987; Rozengurt *et al.*, 1987; Luoma and Phillips, 1988; Phillips and Spies, 1988; Spies and Rice, 1988; see Sect. IV.D.). This report is confined, in large part, to a discussion of the extent to which the ecosystem of the San Francisco Estuary has been affected by the discharge and mobilization of chemical pollutants.

Wastes have flowed into the Estuary since the gold rush period and have caused considerable change in water quality. The disposal of untreated, oxygen-demanding wastes (organic matter and reduced chemical compounds that consume oxygen when released to surface waters) combined to have an adverse effect on commercial shellfish harvesting in the Estuary throughout the 20th century. By the 1960s, untreated sewage discharge had become severe enough to cause hypoxia and anoxia in the South Bay and other portions of the Estuary. Since that time significant public and private investment for the treatment of sewage and other oxygen-demanding wastes drastically reduced these problems.

Recently, public and scientific concern has shifted from a focus on identified, observable pollutant effects (e.g., aesthetics, anaerobic conditions, and fish kills) to concern about the potential toxic impact of persistent pollutants on the beneficial uses of the Estuary. Declines in the populations of striped bass (*Morone saxatilis*) and chinook salmon (*Oncorhynchus tshawytscha*) in the Estuary have contributed to these concerns, as have demonstrations of toxicity in water and sediment bioassays from different regions of the Estuary and the presence of high concentrations of pollutants in some portions of the Estuary. Pollutants of particular concern include mercury, selenium, silver, copper, various pesticides, PAHs, and PCBs. Data regarding the concentrations, distribution, fate, and effects of these pollutants are limited in both quality and quantity (Phillips, 1987); appropriate methods for analysis of many toxic pollutants have been available for only a few years. Nonetheless, sufficient data exist to allow a reasonable discussion of the sources, distribution, and potential effects of pollutants in the Estuary and to identify research priorities for the short term (1-3 yr) and the long-term (5 yr or more).

C. CONTENTS AND STRUCTURE OF THIS REPORT

The report is divided into several sections. After a discussion of the existing management structure for water quality in the Estuary (Section II.C., below), Section III provides a chronological summary of the major developments that have affected the gross loading and distribution of pollutants in the Estuary. Section IV discusses the current status of pollutants in the Estuary, employing the most recent data to highlight the major issues. The pollutants of greatest concern and their sources are reviewed: factors such as seasonal fluctuation in loading and spatial distribution of loads are also

discussed when possible. The section also describes the fates of pollutants in the Estuary, and their possible effects on beneficial uses. Areas of inadequate knowledge ("information and understanding gaps") are highlighted.

Section V concerns future trends and management options. The Section begins with an examination of the factors that will influence the future abundance, distribution, and effects of pollutants in the Estuary. The strength of the present monitoring and research data base in supporting management strategies in the Estuary is reviewed, as are the available strategies that could be employed for the management of the effects of pollutants of concern. Section VI provides the major conclusions of the report, summarizes critical gaps in our understanding, and provides recommendations for obtaining answers to these important unknowns. Detailed material relating to several of the items covered in the text is provided in Appendices.

D. EXISTING MANAGEMENT STRUCTURE

This Section describes two features of the management structure: the general jurisdiction of the government agencies that directly regulate pollution and its effects in the Estuary and the roles of other government agencies and interest groups that influence management decisions and policies.

1. The Principal Regulatory Agencies

A. U.S. ENVIRONMENTAL PROTECTION AGENCY

Two agencies, the U.S. Environmental Protection Agency (USEPA) and the California State Water Resources Control Board (SWRCB), possess principle authority to regulate sources of pollution to the Estuary. This authority derives largely from the 1972 (and subsequent) amendments to the *Federal Water Pollution Control Act* (or *Clean Water Act*). *Clean Water Act* provisions are administered by USEPA, but actual implementation in most States is performed by State agencies such as the SWRCB, the San Francisco Bay Regional Water Quality Control Board (SFBRWQCB), and the Central Valley Regional Water Quality Control Board (CVRWQCB). Since *Clean Water Act* provisions require close collaboration among USEPA and State agencies, many responsibilities of State agencies are included in the following description of USEPA's role in regulation of water quality in the Estuary.

The 1972 *Clean Water Act* established the National Pollutant Discharge Elimination System (NPDES) program for the regulation of discharges of municipal and industrial wastewaters. Municipal and industrial facilities are required to obtain an NPDES permit that specifies allowable limits, based on available wastewater treatment technologies, for pollutant levels in their effluent. In California, USEPA has delegated the implementation of this program to the SWRCB and the Regional Boards.

The 1987 amendments to the *Clean Water Act* also directed USEPA to regulate stormwater discharges through the NPDES permit program. Recently

proposed regulations require NPDES permits for stormwater discharges associated with certain industrial and commercial activities, and for discharges from separate municipal storm sewer systems serving over 100,000 persons (USEPA, 1988a,b). Applicants for these permits will be required to submit information on present and planned stormwater management practices. This regulatory program is primarily aimed at controlling illicit connections to sewer systems, construction site runoff, and storm water runoff from industrial sites, commercial, and residential sites.

Section 303 of the *Clean Water Act* requires that states establish water quality standards for all waters of the United States. Standards consist of designated uses of water bodies and water quality standards to protect those uses. In California, these are known as "beneficial uses" and "water quality objectives." Where multiple uses are designated, objectives must protect the most sensitive use. States must review their standards once every 3 yr. State standards are subject to USEPA approval. Section 303(c)(2)(B) of the *Clean Water Act* , added in the 1987 revisions, requires that States adopt numerical criteria by February 1990 for toxic pollutants for which USEPA has published water quality criteria, "the discharge or presence of which in affected waters could reasonably be expected to interfere with those uses designated by the State."

The *Clean Water Act* also requires that States identify "water quality limited segments" (waters in which required effluent limits are not stringent enough to result in attainment of water quality objectives). States must prioritize these waters and establish "total maximum daily loads" (TMDLs) to attain the applicable water quality objectives "with seasonal variations and a margin of safety that takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality" (Section 303(d)(1)(C)). TMDLs established by States are subject to USEPA approval.

Section 304(l), also added to the *Clean Water Act* in the 1987 revisions, requires States to submit to USEPA lists of waters that cannot reasonably be expected to attain water quality objectives after application of required effluent limits. The State must develop individual control strategies (ICSs) for each listed segment that will produce a reduction of toxic pollutants from point sources necessary to meet the water quality objectives adopted under Section 303(c)(2)(B) (described above). ICSs must be designed to achieve water quality objectives within 3 yr after their establishment. ICSs are also subject to USEPA approval.

The 1987 amendments to the *Clean Water Act* also require States to develop control strategies for "nonpoint" sources, such as storm drains, canals, and ditches that transport polluted runoff (USEPA, 1987). These sources will be required to comply with Water Quality Objectives. States are required to develop Nonpoint Source Assessment Reports and associated Management Plans and to establish a grant program for implementing these management programs. The Assessment Reports identify categories of nonpoint source pollution, identify surface water bodies that would not attain water quality

standards without control of nonpoint source loads, describe the development of "best management practices" (BMPs) to control nonpoint sources, and review existing control programs. The Management Plans identify BMP implementation programs, provide schedules for implementation, and determine funding sources. In California, both a Management Plan and a Source Assessment Report have been developed for nonpoint pollution (SWRCB, 1988a); these are currently under review by USEPA.

USEPA has published guidance to assist the States in complying with many of the above requirements. EPA publishes water quality criteria that States may adopt as part of their standards. USEPA's *Water Quality Standards Handbook* (USEPA, 1983) contains guidance on designation of uses and development of site-specific water quality objectives. USEPA has also published guidelines on how to conduct use-attainability analyses and wasteload allocations. A *Technical Support Document for Water Quality Based Toxics Control* contains guidance on setting effluent limits to meet water quality objectives for toxics, including narrative objectives.

B. THE STATE WATER RESOURCES CONTROL BOARD AND REGIONAL WATER QUALITY CONTROL BOARDS

In California, the SWRCB shares authority for the implementation of *Clean Water Act* and the *Porter-Cologne Water Quality Control Act* with nine Regional Water Quality Control Boards. The Regional Boards conduct planning, permit, and enforcement activities under the guidance and direction of the SWRCB. The San Francisco Estuary lies within the jurisdiction of the SFBRWQCB and the and the CVRWQCB. SFBRWQCB regulates water quality in the watershed of the Bay, and CVRWQCB regulates water quality throughout the Central Valley, including the Delta.

The Regional Boards prepare "Water Quality Control Plans," which document approaches to implementing State and Federal policies in the context of actual water quality conditions in each Region. The Plans specify beneficial uses of receiving waters, water quality objectives imposed to protect the designated beneficial uses, and strategies and schedules for achieving water quality objectives. These Plans are subject to triennial review and are revised periodically to keep pace with changing conditions in receiving waters and advances in understanding of the impact of pollution on the Estuary.

The State and Regional boards are currently developing statewide water quality control plans for inland surface waters and enclosed bays and estuaries. It has been suggested that these plans will include water quality objectives governing approximately 68 of USEPA's list of 126 priority pollutants, as called for by Section 303(c)(2)(B) of the *Clean Water Act*. The State and Regional boards are also working on a statewide clean water strategy and water quality assessment, which will identify water bodies where designated uses are not being attained, and where toxic pollutants may be affecting uses or are at levels that may warrant concern.

In 1987 the State Board commenced proceedings (known as the "Bay-Delta Hearings") to develop water quality objectives to provide reasonable protection of beneficial uses of the Estuary and to consider alternate allocations of water rights to achieve such objectives. Policies are being developed to regulate toxic pollutants and salinity (freshwater inflow). Recent State legislation (Senate Bill 475) established a "Bay Waters Protection and Toxic Cleanup Program", which requires the State and Regional boards to identify toxic "hot spots" and plan for their cleanup or mitigation.

C. U.S. ARMY CORPS OF ENGINEERS

The 1972 *Clean Water Act* also established provisions for the management of dredged material. The U.S. Army Corps of Engineers (USCOE) was given authority to issue permits for the discharge of dredged material into inland waters of the nation. Applicants for permits are required to satisfy several conditions intended to prevent unacceptable adverse effects on the aquatic environment. Review of permit applications includes an assessment of the potential for harmful levels of pollutants to be released to the aquatic environment during both dredging and disposal. The Regional Boards also take an active role in regulation of pollutant movement through dredging activities. The management of dredged material and an evaluation of pollutant mobilization due to dredging activities are discussed in detail in the *Status and Trends Report on Dredging and Waterway Modification in the San Francisco Estuary* (Gunther *et al.*, 1989).

2. Other Governmental Agencies

A. NATIONAL OCEANIC AND ATMOSPHERIC ADMINISTRATION

The primary role of the National Oceanic and Atmospheric Administration (NOAA) is to provide information as a basis for management of the resources of the Estuary. Federal legislation requires NOAA to undertake programs in estuarine and coastal assessment, research, and synthesis/prediction. NOAA derives its authority to collect and archive oceanographic, geophysical, climatic, and pollution data from the *National Ocean Survey Act*, the *National Climate Program Act*, the *Magnuson Fishery Conservation and Management Act* (*MFCMA*), and the *Marine, Protection, Research, and Sanctuaries Act*. Assessment activities include the monitoring of ambient levels of pollutants in the sediment and water column. Research on the effects of pollutants on estuarine habitat, estuarine organisms, and their subsequent effects on human health also constitute a primary focus for NOAA research programs. Based on mandates found in the *Fish and Wildlife Coordination Act* and the *MFCMA*, NOAA uses this information to provide extensive recommendations to State and Federal agencies on regulatory decisions.

3. User Organizations

Government agencies solicit public review of regulatory decisions and policies. A wide variety of organizations representing specific interest groups are active participants in these reviews.

Many environmental groups, including the Audubon Society, Citizens for a Better Environment, CLEAN South Bay, and the Oceanic Society, provide comments on proposed management activities. These groups seek to encourage regulators to meet their obligations under existing environmental legislation and endorse legislation and regulatory action that strengthens those obligations to protect the environment.

Two major associations represent the interests of dischargers to the Estuary in public review processes. The Bay Area Dischargers Association (BADA) consists of the five largest municipal wastewater dischargers to the Bay (Central Contra Costa Sanitary District, East Bay Dischargers Authority, East Bay Municipal Utility District, the city of San Jose, and the city and county of San Francisco). The Bay Area League of Industrial Associations performs a similar function for industrial dischargers to the Bay. These groups, because of their technical experience and knowledge with respect to the control of pollutants, seek to provide regulatory agencies with the best available data and information so that regulatory actions are sound, and in the public interest.

III. SUMMARY OF HISTORICAL TRENDS

The historical record relating to pollutants in the San Francisco Estuary consists of two distinct sets of "data." One set is information from prior to 1950. This information is largely descriptive and anecdotal in form and has been summarized by several authors (Skinner, 1962; Hedgpeth, 1979; Nichols *et al.*, 1986). The second set began to be accumulated in the 1950s when Filice (1954a, b, 1958, 1959) initiated scientific studies of pollution in the Estuary. Because of the sharp line separating anecdotal information from scientific data we have used 1950 as a dividing line in our discussion of historical trends for pollutants in the Estuary.

A. DEVELOPMENTS PRIOR TO 1950

Prior to the arrival of the first European settlers, in about 1769 (Hedgpeth, 1979), the Bay Area was inhabited by a large population of native Americans (Perry, 1989). Shell middens show that the aboriginals consumed large quantities of shellfish (Nichols, 1979); other native fish and wildlife are presumed to have been abundant at this time. It is also probably true that chemical inputs to the Estuary were largely from natural sources (i.e., most of the metals were presumably derived from weathering of rocks and minerals in the basin). Organic chemicals in the Estuary were most likely compounds of biogenic origin (e.g., terpenes from coniferous forests or hydrocarbons from oil seeps). Some pyrolytic and pyrogenic organic compounds may have been present due to deposition of particulate aerosols produced in fires. Pyrolytic and pyrogenic compounds are those created by thermal breakdown (pyrolysis) or breakdown and subsequent condensation (pyrogenesis) after incomplete combustion of complex organic compounds, and may include the polycyclic aromatic hydrocarbons (PAHs).

The Spanish settlement that is now San Francisco was an isolated trading post until gold was discovered in the foothills of the Sierra Nevada in 1848. Between 1848 and 1850 the European population of San Francisco grew from about 400 to 25,000 (Nichols *et al.*, 1986). This rapidly growing population almost immediately began to alter the basic ecology of the Bay-Delta region and the Estuary. Among the more significant changes imposed on the Estuary were 1) massive changes in sediment distribution, 2) mobilization of chemicals in excess of that from natural sources, and 3) the introduction of exotic biota, such as striped bass (*Morone saxatilis*), soft-shelled clams (*Mya arenaria*), eastern oysters (*Crassostrea virginica*), and other species (Nichols *et al.*, 1986).

The most significant physical change wrought by the Gold Rush was the change in sediment distribution in the Bay-Delta. Between 1853 and 1884 miners employed high-pressure flows of water to excavate and expose ore deposits. Hydraulic mining technology excavated tens of millions of cubic yards of rock, rubble, and sediment each year. Massive quantities of debris were deposited in creeks and rivers, blocking flows and obstructing channels throughout the drainage. This mining debris had a long-lasting effect on the

physiography of the region as it was transported, gradually, toward the Estuary. Nichols *et al.* (1986) estimated that, by 1900, the depths of Suisun Bay, San Pablo Bay, and Central Bay had declined by an average of 1.0, 0.75, and 0.25 m, respectively. The State Legislature prohibited the use of hydraulic mining for gold in 1884.

Changes in sediment distribution and bathymetry altered the dynamics of water and sediment transport in the Estuary. While the degree of this alteration remains unknown, the predominance of shoal water in the Estuary today is at least partly due to soils excavated during the Gold Rush period and transported to the Estuary over time.

This newly deposited material probably contained trace amounts of mercury. An estimated 3,500 tonnes of mercury was used to extract gold from the Sierra Nevada ores, contributing to the existing enrichment of this element throughout the Bay-Delta (Phillips, 1987).

The affluence of the Gold Rush also led to the importation of several estuarine species from the East coast; the striped bass, the eastern oyster, and the soft-shelled clam. The opportunistic striped bass and soft-shelled clam adapted well to the Estuary, while the oyster did not become established permanently. Many of these "exotic" species, and their attendant symbionts, commensals, predators, and other associated species, have flourished in the Estuary, displacing native species (Carlton, 1979; Nichols *et al.*, 1986).

Commercial harvesting of salmon, sturgeon, sardines, flatfish, crabs, shrimp, clams, and oysters began soon after the Gold Rush period. By 1900, however, catches of salmon, sturgeon, and the introduced striped bass had declined (Nichols *et al.*, 1986). Gradually these commercial fisheries were halted to protect stocks for sportfishing. Harvests of two introduced species of shellfish, eastern oysters and soft-shelled clams, also began to decline in the early 1900s. Declines in these fisheries may have been associated with overfishing, habitat destruction from sedimentation, or chemical pollution.

Skinner (1962) and Miller (1986) concluded that discharge of untreated wastes and reduced water quality contributed to the decline of these fish and shellfish species. By the end of the 19th century, anoxic conditions and contamination by fecal bacteria were common near points of sewage discharge (Miller, 1986). The significance of these water quality changes compared to other factors affecting Bay-Delta biota (e.g., overfishing, sedimentation, land reclamation, and competition with introduced species) is unknown.

Water quality degradation was also thought to contribute to declines observed in water bird populations in the early 1900s (Skinner, 1962). As early as 1890, naturalists noted the threat posed by discharges of domestic sewage, which reduced the amount of suitable habitat by altering substrates and community structure near outfalls. Anaerobic conditions near sewage outfalls favored the growth of bacteria causing avian botulism and cholera. Discharge of untreated wastewater made the Bay a focal point for these waterfowl diseases.

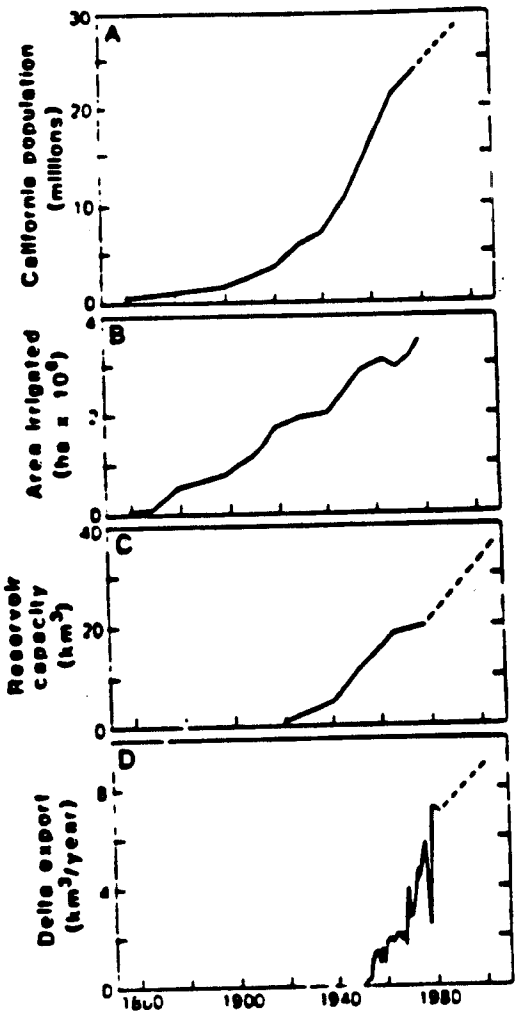
Skinner (1962) also described impacts of industrial activities on water bird populations. The first major oil refinery in the region was the Union Oil facility (1896), followed by the Chevron (1902), Tosco (1912), Shell (1915), Pacific (1966) and Exxon (1968) facilities. In the early 1900s recovery of the waste from refining processes was poor, and oily discharges to the Bay were common. Additional oil was discharged to the Estuary as the result of bilge pumping and flushing of oil tanks from the heavy shipping traffic in the Bay. Oily deposits on the shores of Marin and San Mateo Counties were common in the early 1900s due to the flushing of the tanks of oil ships . Automotive garages were an additional source of oil pollution. It was common practice early in this century for garages to dump waste oils directly into sewage systems that discharged into the Bay. In 1925 it was estimated that garages in Berkeley and Oakland disposed of 3,000 gallons per day [gal d^{-1}] (11,000 liters [L] d^{-1}) in this manner (Skinner, 1962).

The 1940s witnessed two major developments affecting the abundance and fate of pollutants in the Estuary: water development and the use of synthetic organic pesticides. Both these practices were related to the importance of agriculture in the State's economy. The alteration of the freshwater flow regime to the Estuary has affected hydrodynamics and, thus, transport of pollutants throughout the Estuary (Smith, 1987). Between 1860 and 1980 the land area under irrigation in California increased by about 30,000 hectares per year [ha yr^{-1}] (Nichols *et al.*, 1986) (Figure 5b). Local and Federal agencies constructed dams, reservoirs, and canals (Figure 5c), and the Sacramento River, source of approximately 80% of the fresh water entering the Estuary (USGS, 1986), was tapped by the U.S. Bureau of Reclamation to supply water to the Central Valley Project. (Later, in 1967 the California Department of Water Resources State Water Project drew additional water from the Sacramento River.) These water projects served as a major water supply for San Joaquin Valley farms and Southern California municipalities. Exports of this water from the Delta began in the 1940s and have increased steadily, in parallel with increases in population (Figure 5d).

The use of persistent organic chemicals to control weeds and pests on Central Valley farmlands has resulted in their eventual transport to the Estuary (Crosby and Li, 1987; Gunther *et al.*, 1987). Pesticides applied in the farmlands of the Delta are washed directly into waters of the Estuary.

Urban development of the lands surrounding the Estuary accompanied the population growth that began in 1848, and has continued at a rapid pace. The spread of urban lands has been especially striking in the southern reach of the Bay (south of the San Francisco-Oakland Bay Bridge), which is presently encircled by urban land. Historical trends in land use are described in detail in the "Status and Trends Report on Land Use." Runoff from urban lands carries a variety of pollutants of concern (USEPA, 1983; Gunther *et al.*, 1987).

Figure 5. Statewide trends from 1860 to 1980 in a) population growth; b) irrigated agriculture; c) water project storage capacity; and d) water export from the Delta. From Nichols *et al.* (1986).



B. DEVELOPMENTS SINCE 1950

Substantial progress has been made in the management of pollutant loads to the Estuary in the last 40 yr. Regulatory efforts during this period focused on municipal and industrial effluents, due to the relative ease of their control and the clear identity of the responsible parties.

Data collected in the early 1950s provided the first clear evidence that untreated effluent discharges were exerting detrimental effects on organisms. Filice (1954a) studied the composition of the benthic community at the mouth of Castro Creek in San Pablo Bay. At that time Castro Creek received waste from the San Pablo Sewer District, the Standard Oil Company (now Chevron) refinery, and a number of other industries. Filice (1954a) described the water in the creek as having "... a low pH (4.6), toxic chemical waste, a rich bacterial and detrital content, and practically no oxygen." He also demonstrated that these waste streams were contaminating local areas to a significant extent, and that the intertidal area over which the wastes flowed had a depauperate fauna. It is likely that untreated discharges in other portions of the Estuary were causing similar problems, since benthic samples collected around other municipal and industrial outfalls along the shoreline east of Castro Cove in 1951 and 1952 also provided evidence of detrimental effects (Filice, 1954b, 1958, 1959).

Some publicly owned treatment works (POTWs) began primary treatment of municipal wastewater (screening, primary sedimentation, sludge digestion, and disinfection) in the early 1950s. This was the beginning of efforts to control the effects of untreated municipal effluent on the Estuary. Secondary treatment (microbial degradation and secondary sedimentation) was instituted in the mid-1960s. The State *Porter-Cologne Water Quality Act* of 1969 and the Federal *Clean Water Act* of 1972 both required that industrial and domestic waste discharges meet specific, minimum requirements for maintaining water quality in the Estuary. Implementation of these laws led to rapid improvements in the quality of municipal and industrial effluents, and of receiving waters of the Bay, during the 1970s.

These improvements are well documented. Consolidation and relocation to deeper waters of the many POTW discharges into the Bay began in the late 1960s and continued through the 1970s. Many smaller municipalities formed regional discharge authorities to collect and treat sewage from member communities for combined discharge to deep waters. Also, many old, inefficient POTWs were replaced by larger, more advanced sub-regional plants. Since the 1960s the number of municipal plants in the Bay Region has declined from 82 to 58; 46 of these discharges empty into the Bay (Condit, 1987). At present 63% of POTW effluent receives secondary treatment and is discharged to deep waters of the Bay. The remainder receives tertiary treatment (secondary plus filtration) with 28% released into shallow waters of the lower South Bay by the San Jose/Santa Clara, Palo Alto, and Sunnyvale plants and nine percent 9% discharged to shallow waters of the North Bay or to streams that flow into the Bay (BADA, 1987). Additional improvements in treatment also took place in the 1970s. By 1980 nearly all the municipal sewage from the Bay Area received

secondary treatment. Since 1960 over $3 billion has been spent on treatment upgrades, outfall consolidation, and outfall relocation (Condit, 1987).

These efforts resulted in dramatic improvements in effluent quality. From 1955 to 1985 the total population served by POTWs grew from 2.5 million to 5.1 million (Figure 6a), and effluent flows from POTWs increased, from about 236 million gallons per day (MGD; 900 million L d^{-1}) in 1955 to nearly 580 MGD (2,200 million L d^{-1}) in 1985 (Figure 6b). Despite the increase in wastewater volume, pollutant loads decreased. For example, the biochemical oxygen demand load (BOD) from municipal discharges decreased from 181,000 kilograms (kg) d^{-1} in 1955 (a maximum of 221,000 kg d^{-1} in 1965), to 26,000 kg d^{-1} in 1985 (Fig 6c). (BOD is a measure of degradable matter in a waste stream determined by measuring the amount of oxygen used during a fixed period of incubation of wastewater.) Suspended solids loads also declined dramatically, from 126,000 kg d^{-1} in 1955 (a maximum of 146,000 kg d^{-1} in 1970) to 29,000 kg d^{-1} in 1985 (Figure 6d).

Advances in wastewater treatment led to dramatic improvements in "conventional" measures of Bay water quality (e.g., turbidity, dissolved oxygen, coliform count, and odor). Foul odors and unsightly evidence of raw sewage discharges that were once prevalent in the Bay no longer exist. Declines in ambient BOD of surface waters and increases in concentrations of dissolved oxygen were observed during the 1970s (Luoma and Cloern, 1982). Bacteriological quality of Bay waters also improved substantially during this period. Total coliform counts, which averaged 800 organisms per 100 milliliter (ml) in the South Bay in 1964, had declined to an average of 4 organisms per 100 ml in 1977 (Luoma and Cloern, 1982). As one result of this improvement in bacteriological water quality, a 50-yr-long ban on the harvesting of shellfish in the waters of San Mateo County was partially relaxed in 1982. Harvesting during the wet season is still banned throughout the Bay.

The advances in wastewater treatment that occurred prior to the 1970s undoubtedly had the effect of reducing the loads of elemental and organic pollutants entering the Estuary. The precise extent of this reduction is unknown because the data are insufficient for a thorough analysis of trends. The first reliable measurements of trace metal pollution in the Estuary were performed in the mid-1970s, and, except for measurements of "oil and grease" concentrations, there are no historical data on organic pollutants in discharges to the Estuary. Pretreatment programs, which first took effect in the late 1970s, have further reduced the load of pollutants entering the Estuary from municipal sources. Pretreatment programs are aimed at reducing the magnitude of loads carried in industrial discharges that feed into municipal treatment plants ("indirect" industrial discharges).

The data in Table 1 show the reductions in trace element loads that resulted from pretreatment and improvements in treatment processes instituted by 15 major Bay Region municipal dischargers between 1975 and 1985 (SFBRWQCB, 1988a). These data should be considered only gross indicators of trends in municipal trace element loads because the reliability of the data is

Figure 6. Trends in a) population growth, b) effluent volumes, c) BOD loads, and d) suspended solids loads from municipal dischargers in the Bay Region, 1955 to 1985. Data from Condit (1987).

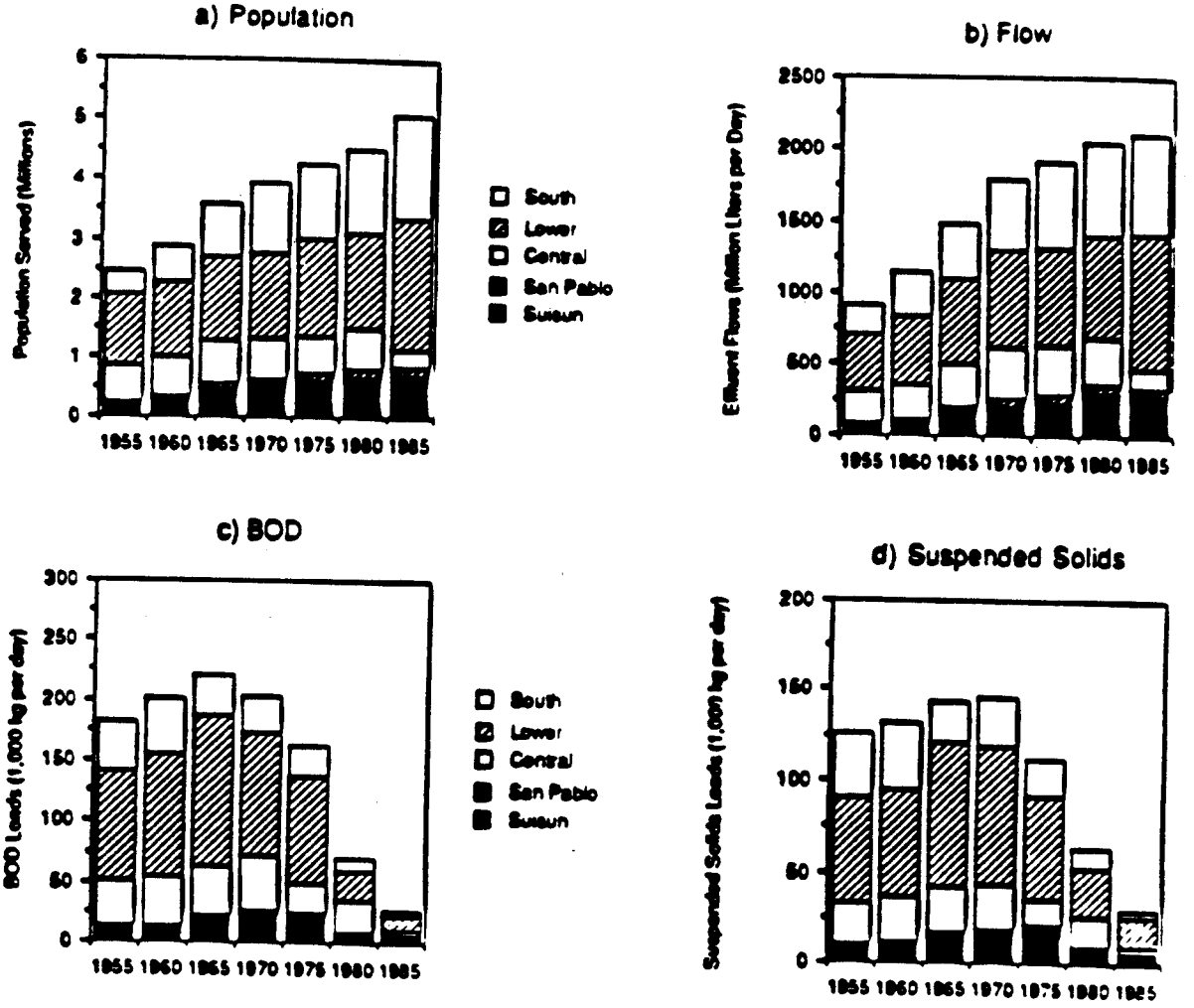


Table 1. Trends in total trace element loads (kg d^{-1}) from 15 POTWs under the jurisdiction of the San Francisco Bay Regional Water Quality Control Board, 1975 to 1985. Data represent maximum average loads ("below detection limit" [BDL] values were set to the limit of detection; see section IV.B.1 for further discussion of interpretation of BDL values). From SFBRWQCB (1988a).

	1975	1980	1985	% Change 1975-1985
Flow	1556	1673	1787	+15
Arsenic	11.6	9.7	6.5	-44
Cadmium	16.4	8.5	6.6	-60
Chromium	185	38	15	-92
Copper	343	169	52	-85
Lead	115	52	25	-78
Mercury	3.01	1.28	0.82	-73
Nickel	87	88	55	-37
Silver	25.4	10.4	9.6	-62
Zinc	653	188	117	-82

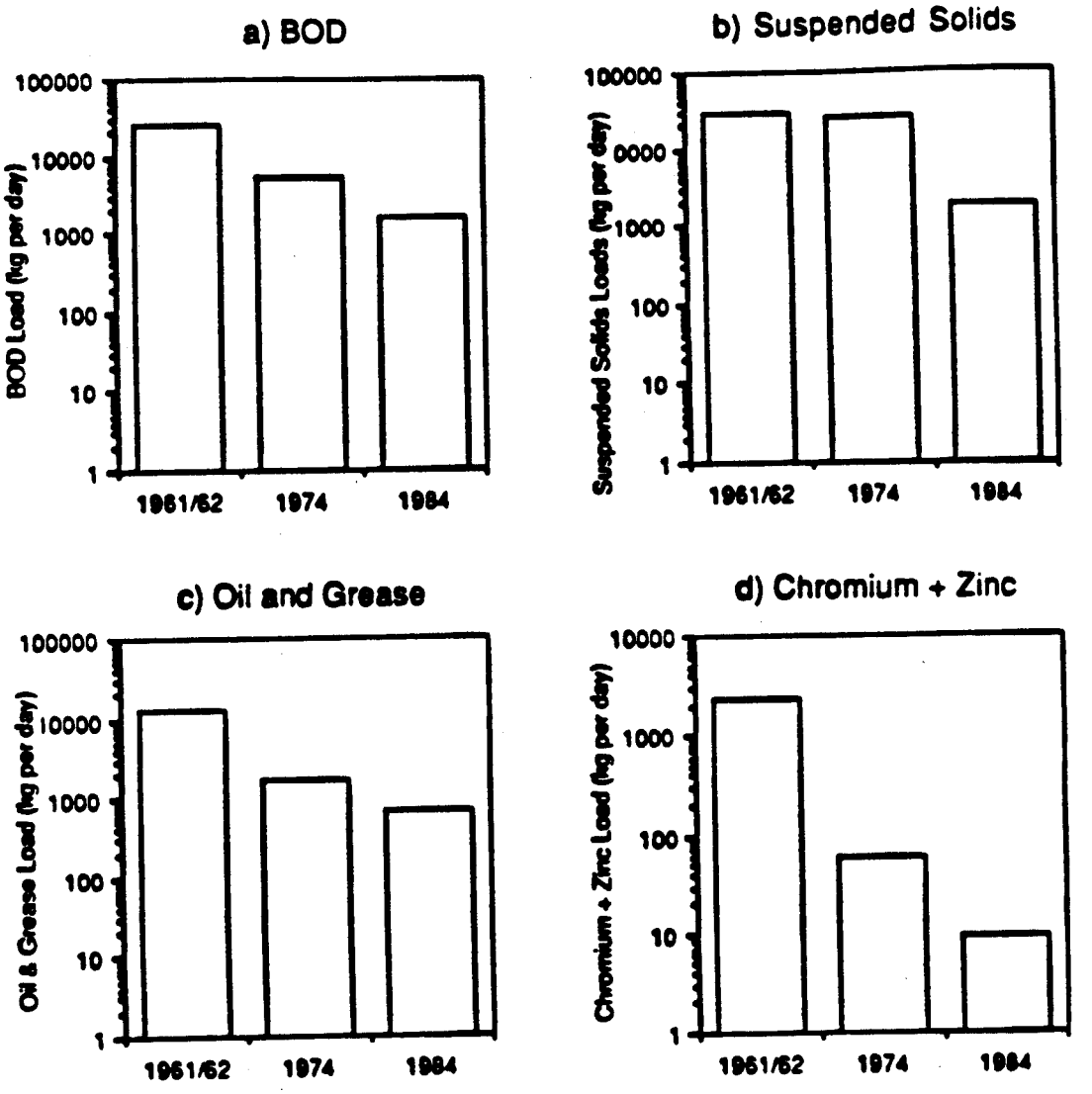
not well documented. Load reductions differ for different trace elements; the largest decrease was observed for chromium (from 185 kg d^{-1} in 1975 to 15 kg d^{-1} in 1985) and the smallest for nickel (from 87 kg d^{-1} in 1975 to 55 kg d^{-1} in 1985). Nickel is the only element for which there was an increase in overall loads during any of the 5-yr intervals measured.

Treatment of industrial wastewater discharged to the Estuary has also improved since the 1960s. Petroleum refineries are the largest class of industrial dischargers to the Estuary. Loads of BOD, suspended solids, oil and grease and two trace metals from refineries have decreased at least an order of magnitude from 1961 to 1984 (Figure 7). During this time BOD declined from 25,000 to 1,600 kg d^{-1} (Figure 7a); suspended solids loads from 29,000 to 1,900 kg d^{-1} (Figure 7b); oil and grease loads from 13,000 to 680 kg d^{-1} (Figure 7c); and combined loads of chromium and zinc from 2,400 to 9.0 kg d^{-1} (Figure 7d) (SFBRWQCB, unpublished data). Again, these data should be considered only gross indicators of trends in loads because the reliability of the data is not well documented.

Progress has also recently been made in "pollution prevention" and source reduction as a means for reducing industrial pollutant loads to the Estuary. For example, discharges of chromium, lead, and nickel from the Chevron oil refinery were reduced substantially by applying several kinds of pollution prevention, including 1) substituting less toxic raw materials, 2) redesigning and reformulating products to contain or require fewer toxic materials, 3) improving plant operating efficiency to generate fewer waste materials, and 4) closed-loop recycling and recycling of wastes at the plant (CBE, 1989). The impact of pollution prevention measures employed at the Chevron Richmond Plant was to reduce chromium, nickel, and lead loading to the Estuary from that plant by 67%, 86%, and 97%, respectively, compared to loading data from 1982 to 1983.

Although advances in the management of water quality in the Estuary have occurred over the last 40 yr, significant problems remain. While "conventional" pollutants have been successfully controlled, and present-day concerns over eutrophication, or nutrient enrichment, in surface waters have declined, concerns remain regarding toxic chemical pollutants and their possible effects on the well-being of populations of shellfish, fish and wildlife in the Bay-Delta ecosystem (Luoma and Cloern, 1982; Luoma and Phillips, 1988; Phillips and Spies, 1988). Reductions of loads of toxic pollutants achieved to date have largely been coincident with reductions of BOD and suspended solids loads. Although evidence indicates that loading rates of toxic pollutants have declined in the last 20 yr, present rates of mobilization of toxicants in the Estuary due to human activities remain high relative to natural rates of mobilization and continue to pose a potential hazard to biota. The remainder of this report reviews the current state of knowledge concerning the abundance, sources, fate, transport, metabolism, and possible effects of toxic pollutants in the Estuary.

Figure 7. Trends in loads of a) BOD, b) suspended solids, c) oil and grease, and d) chromium and zinc from Bay Area refineries, 1961/62 to 1984. Data from the San Francisco Bay Regional Water Quality Control Board (unpublished).



IV. THE CURRENT STATUS OF CHEMICAL POLLUTION IN THE ESTUARY

The current condition of the San Francisco Estuary has been defined by natural changes in the system, as well as by the combined physical, geologic, chemical, and biological alterations imposed by humans since the time of the Gold Rush. All of these changes have combined to establish an estuarine ecosystem that is dynamic, ecologically unique, and not completely understood. Perhaps least understood are the relationships between the abiotic and biotic compartments in the Bay and the Delta that have been subjected to the presence of large masses and increased concentrations of toxic and potentially toxic chemical pollutants.

Between about 1975 and the present, much information on the distribution, fate, and effects of pollutant chemicals in the San Francisco Estuary has been gathered. Those data are the results of studies conducted by a number of local, State, and Federal laboratories, and they provide a basis for evaluating the current condition of the Estuary. They serve also as a baseline for evaluating future improvements in the condition of the Estuary.

However, significant gaps remain in our present understanding of pollutants in the Estuary. These gaps in knowledge and understanding exist in large part because methodology for appropriate measurement of pollutants and their effects has been lacking. For example, reliable methods for the analysis of many important environmental pollutants, such as lead, cadmium, DDT, PCBs, PAHs, and many industrial and agricultural chemicals, have been available for only a few years, and bioassay techniques useful in drawing inferences about the ecological effects of chemical pollutants are still being developed. The available data and existing data gaps can be used to help define which studies and monitoring efforts should be conducted in the future, and which options may be available, now and in the future, for the effective management and eventual control of toxic chemical pollutants in the Estuary.

The review and synthesis of the available data in this report provide a context in which further advances in our understanding of the Estuary, its ecology, and its dynamics can be made. The sections that follow provide an overview of the current status of pollutants in the Estuary, focusing on pollutant loads, fates, and effects on beneficial uses. This section concludes with a description of the most important gaps in our knowledge and understanding of pollutants in the Estuary.

A. POLLUTANTS OF GREATEST CONCERN

This section presents a list of pollutants most likely to exert effects detrimental to the beneficial uses of the Estuary. This list of pollutants of concern is not complete; rather, it consists of those pollutants known to be present in the water, sediments, and biota of the Estuary that may pose a threat to the health and well being of the Estuary and its biota. Other pollutants certainly exist in the

system; however, their identity, as well as their abundance, distribution, and potential ecological effects, are not sufficiently well known for evaluation in this report.

The remainder of this report will place greatest emphasis on an assessment of these "pollutants of concern." The list of pollutants (Table 2) was prepared by the Aquatic Habitat Institute staff and participants from the San Francisco Estuary Project. Appendix I describes the derivation of this list and the criteria used to make selections of pollutants. The Appendix includes a matrix presenting the grading of each pollutant against these criteria. Briefly, the criteria used for selecting pollutants of concern were:

1) The potential for each pollutant to cause toxicity or to affect beneficial uses of the Estuary;
2) The extent of the database for each pollutant within the San Francisco Estuary;
3) Whether the pollutant exhibits widespread enrichment in the Estuary;
4) Whether the pollutant exhibits localized enrichment in the Estuary; and
5) Whether the pollutant exerts or may exert detrimental effects on the biological resources of the Estuary.

Pollutants identified as being of particular concern due to high regional or local concentrations or other factors are shown in boldface in Table 2.

The available data on pollutant concentrations in water, sediment, and biota of the Estuary in comparison to appropriate regulatory criteria are shown in Table 3. Water quality objectives specifying permissible concentrations of pollutants in water are a principal regulatory tool employed by the Regional Water Quality Control Boards. It is surprising to note, therefore, the paucity of reliable data on pollutant concentrations in Bay-Delta waters (Table 3a); this point has been discussed in detail by Phillips (1988). Very few data exist on the concentrations of most organic pollutants in waters of the Estuary, including critical pollutants such as the organochlorines and the petroleum hydrocarbons. This lack of information is at least partly due to technical difficulties involved in the measurement of the very low concentrations of these pollutants in natural waters. Techniques for the accurate measurement of trace element concentrations, on the other hand, are well developed. Nevertheless, there are only eight local studies in which such techniques have been employed, and these rarely covered the entire Estuary. Most of these studies were conducted in waters downstream of the Carquinez Strait. In addition, the studies generally measured pollutant concentrations in solution, constraining comparison of available data to water quality objectives (Table 3a), which apply to concentrations of pollutants in unfiltered samples ("total" concentrations). In 1989 the San Francisco Bay Regional Water Quality Control Board initiated a study employing ultra-clean techniques to determine trace element concentrations in the Bay (T. Mumley, SFBRWQCB, personal communication).

Tables 3b and 3c present summaries of pollutant concentrations measured in biota of the Estuary. Bioaccumulation of pollutants of concern has been studied in many species of fish and shellfish in the Bay and Delta; Tables 3b and 3c focus on data collected for mussels (*Mytilus* spp.) because of the

Table 2. List of pollutants thought to be of concern in the San Francisco Estuary. Pollutants shown in bold type are thought to be of particular concern.

Trace Metals

Cadmium
Copper
Mercury
Nickel
Selenium
Silver
Tin (Tributyl)
Antimony
Arsenic
Chromium
Cobalt
Lead
Zinc

Organochlorines and other pesticides

Chlordane and its metabolites
DDT and its metabolites
Polychlorinated biphenyls
Toxaphene
Aldrin
Chlorbenside
Dacthal
Dieldrin
Dioxins
Endosulfan
Endrin
Heptachlor and its epoxide
Hexachlorobenzene (HCB)
Hexachlorobutadiene
Hexachlorocyclohexane (HCH)
Methoxychlor
Polychlorinated terphenyls
2,4,6-trichlorophenol
Malathion
Parathion

Hydrocarbons

(i) Monocyclic aromatic hydrocarbons (MAHs)

Benzene
Ethylbenzene
Toluene
Xylene

(ii) Cycloalkanes

(iii) Polynuclear aromatic hydrocarbons (PAHs)

Acenaphthene
Acenaphthylene
Anthracene
Benz(b)fluoranthene
Benz(k)fluoranthene
Benz(g,h,I,)perylene
Benzo(a)pyrene
Benzo(e)pyrene
Benzo(a)anthracene
Benzthiazole
Chrysene
Dibenzo(a,h)anthracene
2,6-Dimethylnaphthalene
Fluoranthene
Fluorene
1-Methylnaphthalene
2-Methylnaphthalene
1-Methylphenanthrene
2-(4-morpholinyl)benzthiazole
Naphthalene
Phenanthrene
Pyrene
2,3,5-Trimethylphenanthrene
Indeno(1,2,3-c,d)pyrene

Table 3a. Summary of reliable data on contaminant levels in waters of the Estuary. Measured data are compared to existing regulatory criteria. All data in ug L^{-1}. Reported concentrations were measured in grab samples. Water quality objectives refer to "total" concentrations (concentrations measured in unfiltered samples).

Pollutant	Range of dissolved concentrations measured in waters of the Estuary (sources of data [see footnote])	Range of "total" concentrations measured in waters of the Estuary (sources of data [see footnote])	SFBRWQCB water quality objective downstream of Carquinez Strait	SFBRWQCB water quality objective upstream of San Pablo Bay	CVRWQCB water quality objective (maximum concentrations)	USEPA water quality criterion for saltwater	USEPA water quality criterion for freshwater
Cadmium	0.007 - 0.520 (2, 4, 5, 6, 8)	0.062 - 0.380 (4, 8)	9.3 (4D) 43 (1H)	1.1 (4D)* 3.9 (1H)*	-	9.3 (4D) 43 (1H)	1.1 (4D)* 3.9 (1H)*
Copper	0.069 - 5.300 (2, 4, 5, 6, 8)	0.770 - 9.970 (4, 8)	None	6.5 (4D) 9.2 (1H)	10	- 2.9 (1H)	12 (4D)* 18 (1H)*
Mercury	0.006 - 0.011 (8)	0.010 - 0.095 (8)	0.025 (4D) 2.1 (1H)	0.025 (4D) 2.4 (1H)	-	0.025 (4D) 2.1 (1H)	0.012 (4D) 2.4 (1H)
Nickel	0.181 - 8.040 (2, 4, 5, 8)	1.210 - 15.900 (4,8)	7.1 (1D) 140 (Inst)	56 (1D) 100 (Inst)	-	8.3 (4D) 75 (1H)	160 (4D)* 1400 (1H)*
Selenium	0.013 - 4.700 (1)	<0.012 - 0.174 (8)	-	-	-	71 (4D) 300 (1H)	5 (4D) 20 (1H)
Silver	0.002 - 0.310 (3, 4)	0.007 - 0.345 (3, 4)	2.3 (Inst)	1.2 (Inst)	10	- 2.3 (1H)	0.12 (4D)* 4.1 (1H)*
Tributyltin	-	0.004 - 0.570 (7)	-	-	-	0.01 (4D)** 0.266 (1H)**	0.0264 (4D)** 0.149 (1H)**
Arsenic	-	-	36 (4D) 69 (1H)	190 (4D) 360 (1H)	10	36 (4D) 69 (1H)	190 (4D) 360 (1H)
Chromium Chromium (VI)	0.130 - 0.190 (8) -	0.540 - 3.600 (8) -	- 50 (4D) 1100 (1H)	- 11 (4D) 16 (1H)	-	- 50 (4D) 1100 (1H)	- 11 (4D) 16 (1H)
Lead	0.001 - 0.350 (3, 4, 5, 8)	<0.060 - 6.650 (4, 8)	5.6 (4D) 140 (1H)	3.2 (4D) 82 (1H)	-	5.6 (4D) 140 (1H)	3.2 (4D)* 82 (1H)*
Zinc	0.021 - 7.500 (2, 4, 5, 6, 8)	0.770 - 22.300 (4, 8)	58 (1D) 170 (Inst)	58 (1D) 170 (Inst)	100	86 (4D) 95 (1H)	110 (4D)* 120 (1H)*
PAHs	-	-	15 (1D)	-	-	-	-
PCBs	-	0.0004 - 0.0066 (9)	-	-	-	0.03 (4D) 10 (1H)	0.014 (4D) 2 (1H)

Dashes indicate that either reliable data or regulatory guidelines do not exist

1 - Cutter (1989a, b); 2 - Eaton (1979a), 3 - Flegal and Gordon (unpublished data); 4 - Girvin *et al* (1978), 5 - Gordon (1980), 6 - Kuwabara *et al* (1989), 7 - Stallard *et al* (1987), 8 - Stukas (1986), 9 - Andorfini *et al* (1975a)

4D - Four day average, 1H - One hour average, 1D - One day average, Inst - Instantaneous value

* Standard varies with hardness of the receiving water, hardness of 100 mg/l assumed

** Proposed criteria

Table 3b. Summary of data on trace element levels in *Mytilus* in the San Francisco Estuary. Measured data are compared to median international standards (described in text), and to statewide statistics generated by the California State Mussel Watch. All data in ug g^{-1} dry weight. Median international standards from Phillips [P.T] (1988).

		From Long *et al.* (1988): Statistics for concentrations in *Mytilus* spp., early 1970s to mid-1980s					From Phillips [P.T] (1988): State Mussel Watch data for transplanted *M. californianus*, 1977-1987	
Pollutant	Median international standard for consumption of shellfish*	Mean	Standard deviation	Median	Range	Number of observations	Statewide 85th percentile	Number of samples from the Estuary exceeding the 85th percentile (total number analyzed)
Cadmium	7	7.41	4.39	6.00	0.8 - 34.4	332	10.83	35 (94)
Copper	140	10.02	3.64	9.40	2.2 - 30.7	305	21.85	1 (94)
Mercury	3.5	0.4	0.25	0.33	0.09 - 3.22	311	0.44	26 (85)
Nickel	-	-	-	-	-	-	5.3	1 (4)
Selenium	2.1	-	-	-	-	-	4.48	4 (21)
Silver	-	0.97	1.94	0.64	0.02 - 22.5	317	0.7	42 (94)
Arsenic	9.8	-	-	-	-	-	23.82	0 (12)
Chromium	-	2.72	2.30	2.20	0.1 - 14.8	288	3.93	17 (94)
Chromium (VI)	7	-	-	-	-	-	-	-
Lead	14	6.23	34.9	2.90	<0.2 - 519	331	11.01	3 (94)
Zinc	490	-	-	-	-	-	336.3	3 (94)

Dashes indicate data were not available.

* Wet weight standards converted to approximate dry weight equivalents by multiplying by 7.

Table 3c. Summary of data on organochlorine concentrations in *Mytilus* spp. in the San Francisco Estuary. Measured data are compared to statewide statistics generated by the California State Mussel Watch. All data in ng g^{-1} dry weight.

	From Long *et al.* (1988): Statistics for concentrations in *Mytilus* spp., early 1970s to mid-1980s					From Phillips [P.T] (1988): State Mussel Watch data for transplanted *M. californianus*, 1977-1987	
Pollutant	Mean	Standard deviation	Median	Range	Number of observations	Statewide 85th percentile	Number of samples from the Estuary exceeding the 85th percentile (total number analyzed)
Total Chlordane	-	-	-	-	-	192	4 (60)
Total DDT	330	1840	70	10 - 22470	189	1483	8 (54)
Total PCB	650	600	520	60 - 4600	193	1420	2 (61)
Toxaphene	-	-	-	-	-	<100	6 (54)
Aldrin	-	-	-	-	-	<1.0	5 (59)
Chlorbenside	-	-	-	-	-	6.2	11 (54)
Dacthal	-	-	-	-	-	9.3	1 (52)
Diazinon	-	-	-	-	-	<23	0 (56)
Dieldrin	-	-	-	-	-	67.0	17 (56)
Total Endosulfan	-	-	-	-	-	17.0	0 (52)
Endrin	-	-	-	-	-	<6.0	7 (52)
Heptachlor	-	-	-	-	-	<1.0	7 (54)
Heptachlor epoxide	-	-	-	-	-	1.4	8 (54)
Hexachloro-benzene	-	-	-	-	-	0.2	22 (54)
α HCH	-	-	-	-	-	4.9	10 (54)
β HCH	-	-	-	-	-	<3.0	4 (54)
δ HCH	-	-	-	-	-	<2.0	3 (54)
γ HCH	-	-	-	-	-	3.6	13 (54)
Methoxychlor	-	-	-	-	-	<15	0 (54)
Methyl Parathion	-	-	-	-	-	<4.0	0 (51)

Dashes indicate data were not available

Table 3d. Summary statistics on pollutant concentrations (μg g^{-1} dry weight) in surficial sediments of San Francisco Bay, based on data collected by many investigators from 1970 through 1987. Adapted from Long *et al.* (1988).

Pollutant	Mean	Standard Deviation	Median	Range	Number of Samples
Cadmium	1.06	1.16	0.71	0.02 - 17.3	999
Copper	51	58	46	1 - 1500	879
Mercury	0.5	0.67	0.38	<0.01 - 6.80	1097
Silver	1.13	1.52	0.58	<0.01 - 16	336
Chromium	89	96	63	8 - 769	396
Lead	56	300	38	1 - 10000	1314
Polynuclear Aromatic Hydrocarbons (total) (1)	4.1	10.1	1.7	0.02 - 80.9	101
DDT and metabolites (2)	0.1	0.28	0.007	0.00025 - 1.96	153
Polychlorinated biphenyls (total)	0.115	0.172	0.05	0.006 - 0.824	52

(1) Includes only 7 PAHs measured in each of several studies: phenanthrene; anthracene; fluoranthene; pyrene; benz[a]anthracene; chrysene; and benzo[a]pyrene.

(2) Does not include data on extremely contaminated sediment from the Lauritzen Canal. The overall mean including the additional 13 samples from the Lauritzen Canal is 7.5 μg g^{-1} dry weight.

availability of comparative data. There are few U.S. regulatory criteria for trace metals in shellfish. Consequently, Table 3b compares trace element concentrations measured in *Mytilus* spp. with non-regulatory criteria. Concentrations of pollutants that approach or exceed these non-regulatory criteria may suggest the need for further investigation of potential effects.

The first column of Table 3b lists a median of standards set by other countries for these pollutants (Phillips [P.T.], 1988). The next columns in Table 3b present baywide means and standard deviations calculated by Long *et al.* (1988) in their review of available data on pollutant abundance in San Francisco Bay. With the exception of cadmium, most mean trace element concentrations in *Mytilus* spp. were below the Median International Standards (MIS). Mean cadmium concentrations in *Mytilus* spp. (7.4 $\mu g\ g^{-1}$ dry weight) exceeded the MIS slightly (approximately 7.0 $\mu g\ g^{-1}$). The last columns of Table 3b list data from the California State Mussel Watch Program and show the frequency of concentrations exceeding the 85th percentile (or "elevated data level [EDL]) of values reported throughout the State. It should be noted that although these EDLs provide an indication of elevated toxicant concentrations, they do not necessarily represent concentrations that may be damaging to either mussels or other organisms (Phillips [P.T.], 1988). Concentrations of cadmium, mercury, and silver in transplanted *Mytilus californianus* frequently exceed their respective EDLs. It should also be noted that the EDL for selenium is much higher than the MIS. A similar overall pattern is seen for trace element accumulation by *Mytilus edulis*, although fewer data have been collected.

Table 3c presents a summary of concentrations of organic pollutants in *Mytilus* spp. (based on data from Long *et al.* [1988] and Phillips [P.T.] [1988]). There are few regulatory guidelines that may be used to judge the significance of organic chemical concentrations in shellfish. Statewide statistics generated by the State Mussel Watch were therefore used as a basis for comparison in Table 3c. Elevated concentrations of many organochlorines occur in mussels from the Estuary; these high concentrations were most frequent in the case of the pollutants hexachlorobenzene, dieldrin, hexachlorocyclohexane, and chlorbenside. These data and data from other species show that organochlorines are present throughout the Bay-Delta area (Phillips, 1988). Few measurements have been made of the concentrations of other hydrocarbons in tissues; there is a conspicuous lack of data on concentrations of PAHs in all media sampled from the Estuary.

A substantial body of data concerning concentrations of several pollutants in sediments of the Estuary has been generated in the last 20 yr. However, there are no regulatory guidelines that may be used to judge the significance of these data. Long *et al.* (1988) performed a statistical analysis of data from samples collected at 1,232 sites in 20 different studies in San Francisco Bay. Table 3d provides descriptive statistics on the pollutants included in this analysis.

These summary statistics should be interpreted with caution; they provide an indication of pollutant concentrations present in the Estuary, but the data

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Lead	56	300	38	1 - 10000	1314
Polynuclear Aromatic Hydrocarbons (total) (1)	4.1	10.1	1.7	0.02 - 80.9	101
DDT and metabolites (2)	0.1	0.28	0.007	0.00025 - 1.96	153
Polychlorinated biphenyls (total)	0.115	0.172	0.05	0.006 - 0.824	52

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These summary statistics should be interpreted with caution; they provide an indication of pollutant concentrations present in the Estuary, but the data

come from samples that were collected using a variety of sampling techniques, and methods of varying quality were used in analysis. Spatial coverage was also non-uniform, in that most of the samples were taken from areas about to be dredged (Long *et al.*,1988). The data do show that much variation exists in the concentrations of pollutants in sediments from the Estuary. Some of this variation is clearly attributable to sampling location. Peripheral portions of the Estuary (e.g., shorelines, harbors, and nearshore channels) had generally higher concentrations of pollutants than the central portions. Relatively little information is available on the concentrations of organochlorines and PAHs in sediment.

B. POLLUTANT LOADS

The characterization of pollutant loads to the Estuary is essential to understanding pollutant fate and potential effects upon biota. The cost-effective management of pollutants also requires an understanding of the relative significance of different pollutant loads. An adequate characterization of major loads would enable managers to focus research and control efforts on loads that present the greatest ecological threats.

This section is a first step toward characterizing pollutant loads to the Estuary from municipal and industrial effluent, urban runoff, nonurban runoff, major tributaries (the Sacramento and San Joaquin rivers), dredging and disposal of dredged material, atmospheric deposition, spills, and marine vessel discharges. Classes of pollutants found in the major inputs and the present state of knowledge regarding the magnitudes of loads entering the Estuary will be reviewed. Major gaps in understanding of the magnitude and variability of each of the major input categories will be assessed, and approaches to gaining an improved understanding of pollutant loads will be highlighted.

The mass loading of pollutants (mass of pollutant per unit time) is considered in this section. Mass loading data can be used to estimate the quantities of pollutants entering the ecosystem, and therefore can provide the best single measure for assessment of the relative contributions of various inputs to the pool of bioavailable pollutants present in the Estuary. Loading data for continuous input streams (i.e., municipal and industrial effluents and rivers) are presented in kg d^{-1}; data for discontinuous input streams (i.e., urban and nonurban runoff, atmospheric deposition, vessel discharges and spills) are presented in tonnes yr^{-1}. Loading data in common units (tonnes yr^{-1}) are presented at the end of this section for direct comparison among inputs.

The next step in characterizing pollutant inputs, and an important subject of future research, will be determining the origins, or primary sources, of toxic materials in the waste streams that flow into the Estuary. Human activities, such as mining, industrial processes, transportation, agriculture, and household maintenance are usually the primary sources of pollutants. Detailed investigation of primary sources will be necessary for pollutants found to be entering the Estuary in excessive amounts.

This section on pollutant loads assesses loading to the Estuary in a manner similar to that of Gunther *et al.* (1987). The boundaries of the Estuary are shown in Figure 1. Unlike many other reports on this estuarine ecosystem, in this report the entire Delta is considered part of the Estuary. With the exception that pollutant remobilization due to dredging and dredged material disposal is included as a source, this discussion includes only pollutant input to the Estuary, and does not address the problem of pollutant transport or fate after entry to the ecosystem. For example, no attempt is made to determine the flux of pollutants from the Delta to the Bay or from Central Bay to the Pacific Ocean. Dredging and dredged material disposal activities transfer sediment-associated pollutants from one site in the Estuary to another. Since these activities may

remobilize pollutants that have been buried and otherwise isolated from the water column, dredging and dredged material disposal are treated in the same manner as the other pollutant sources. A detailed discussion of fate and transport issues is provided in Section IV.C., below.

1. Municipal and Industrial Effluents

Considerable effort has gone into defining and controlling pollutant loads from municipal and industrial discharges. The following discussion of pollutant loading to the Estuary from these inputs is based largely upon data on volumes and chemical characteristics of discharges collected by municipal and industrial dischargers, as required under the NPDES program (see Section II.C.). The Aquatic Habitat Institute (AHI) has assembled a computerized data base that contains effluent monitoring data gathered from 1984 through 1987.

The effluent monitoring data base was constructed in two phases (Gunther *et al.*, 1987). In phase one, data on toxic pollutant loads were collected for all major and minor discharges in the Bay and Delta from 1984 to 1986. These data provided the basis for a detailed assessment of municipal and industrial loads to the Estuary (Gunther *et al.*, 1987). Subsequently, data on pollutant loads by major discharges in the Bay region during 1987 were added to the data base. Data on loads from minor outfalls and discharges in the Delta were not included for 1987. Data from all 4 yr (1984-1987) are summarized in this section. The following analysis adds to that Gunther *et al.* (1987) by inclusion of 1987 data and by focusing upon spatial and temporal trends. For information on the acquisition, processing, and evaluation of these data, refer to Gunther *et al.* (1987).

Pollutant loads from municipal and industrial discharges are better characterized than those from any other input. Problems arise, however, in that standard methods for wastewater and water column analysis of pollutants are unable to quantify accurately the low concentrations of many pollutants, generating many "below detection limit" (BDL) results (BDL means that a particular analytical measurement either failed to detect the presence of a particular pollutant, or that the amount measured was lower than the amount necessary for an accurate estimate of concentration). Table 4 lists the frequencies of detection for the major classes of pollutants commonly analyzed in effluents (frequencies of detection for each pollutant appear in Table 1, Appendix II).

The only class of pollutants detected regularly in effluents were the trace elements. Of the pollutants of concern (see Section IV.A.), complete data sets exist for arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc (Table 1, Appendix II). Only a few of the volatile organics (EPA Method 624) have been detected frequently, including chloroform (69%), bromodichloromethane (48%), dichloromethane (45%), tetrachloroethene (39%), toluene (22%), and 1,1,1-trichloroethane (22%). Of these, however, only toluene is included in the list of pollutants of concern (Table 2). Other pollutants,

Table 4. Frequencies of detection of the major classes of pollutants measured in Bay-Delta effluents. Frequencies of detection for each pollutant appear in Table 1 in Appendix II).

Pollutant Class	Total # Analyses	Total # Detected	Frequency of Detection
Trace Elements	17468	9886	0.57
Organochlorines (EPA Method 608)	2713	17	0.01
Polynuclear Aromatics (EPA Method 610)	1232	10	0.01
Volatile Organics (EPA Method 624)	10104	1209	0.12
Semi-volatile Organics (EPA Method 625)	18983	396	0.02

including semi-volatile organics, organochlorine pesticides, PCBs, and PAHs, were detected infrequently in Bay-Delta effluents (Table 4).

Other considerations frustrate the use of even the best data on toxic organics for load calculations. Sampling for these pollutants in general occurs only twice each year. The general absence of accompanying quality control test results also constrains quantitative interpretation of these data (Gunther *et al.*, 1987). Quality control data would provide assurance that analytical results were not influenced by contamination of samples or poor recovery of the analytes. Only data for 10 trace elements (listed above) were used in the assessment of spatial and temporal trends of pollutant loading from municipal and industrial effluents.

The Estuary has been separated into 10 major segments for analysis of spatial patterns in municipal and industrial loading (Figure 8). Segments within the Bay have been separated by the bridges shown in the diagram; the Delta has been divided into 4 segments. The following series of figures illustrates loading of individual pollutants to the Estuary. The columns within each segment represent the average load (kg d^{-1}) to that segment for the pollutant of interest. Each figure also includes several inset graphs, depicting annual trends in loading from major point-source dischargers. Solid black dots on the figures show the approximate location of the major outfalls.

A. VOLUMES OF WASTE DISCHARGED

Flow data were not gathered from 1987, so the average flows portrayed for each segment of the Estuary (Figure 9) were based on data from 1984 to 1986 (similar figures for the trace elements are based on data from 1984 to 1987). It should also be noted that discharges of cooling and other recycled waters to the Estuary are not included.

Figure 9 identifies most of the largest dischargers in the Estuary between 1984 and 1986. An average of over 3.2 billion liters per day of treated effluent was released to the Estuary from 1984 to 1986. The North Delta receives waste water from the largest volume discharger in the Estuary, the Sacramento Regional Wastewater Treatment Plant, which released an average of 507 million liters per day (MLD; 134 MGD). Lower South Bay received a total of 625 MLD (165 MGD), including 447 MLD (118 MGD) from San Jose/Santa Clara (the second largest discharger in the Estuary), 106 MLD (28 MGD) from Palo Alto, and 68 MLD (18 MGD) from Sunnyvale. The third largest discharger, East Bay Municipal Utility District (EBMUD), contributed most (329 MLD; 87 MGD) of the total volume (450 MLD; 119 MGD) discharged to Central Bay.

Upper South Bay received the greatest volume of any segment (696 MLD; 184 MGD), including 280 MLD (55 MGD) from the San Francisco Southeast (SFSE) plant. Several municipal treatment plants, including Dublin-San Ramon, Hayward, Livermore, Oro Loma, San Leandro and Union Sanitary District (comprising the East Bay Dischargers Authority [EBDA]) released a total of 257 MLD (68 MGD) through a combined outfall in Upper South Bay.

Figure 8. Segments of the Estuary employed for analysis of spatial patterns in loading.

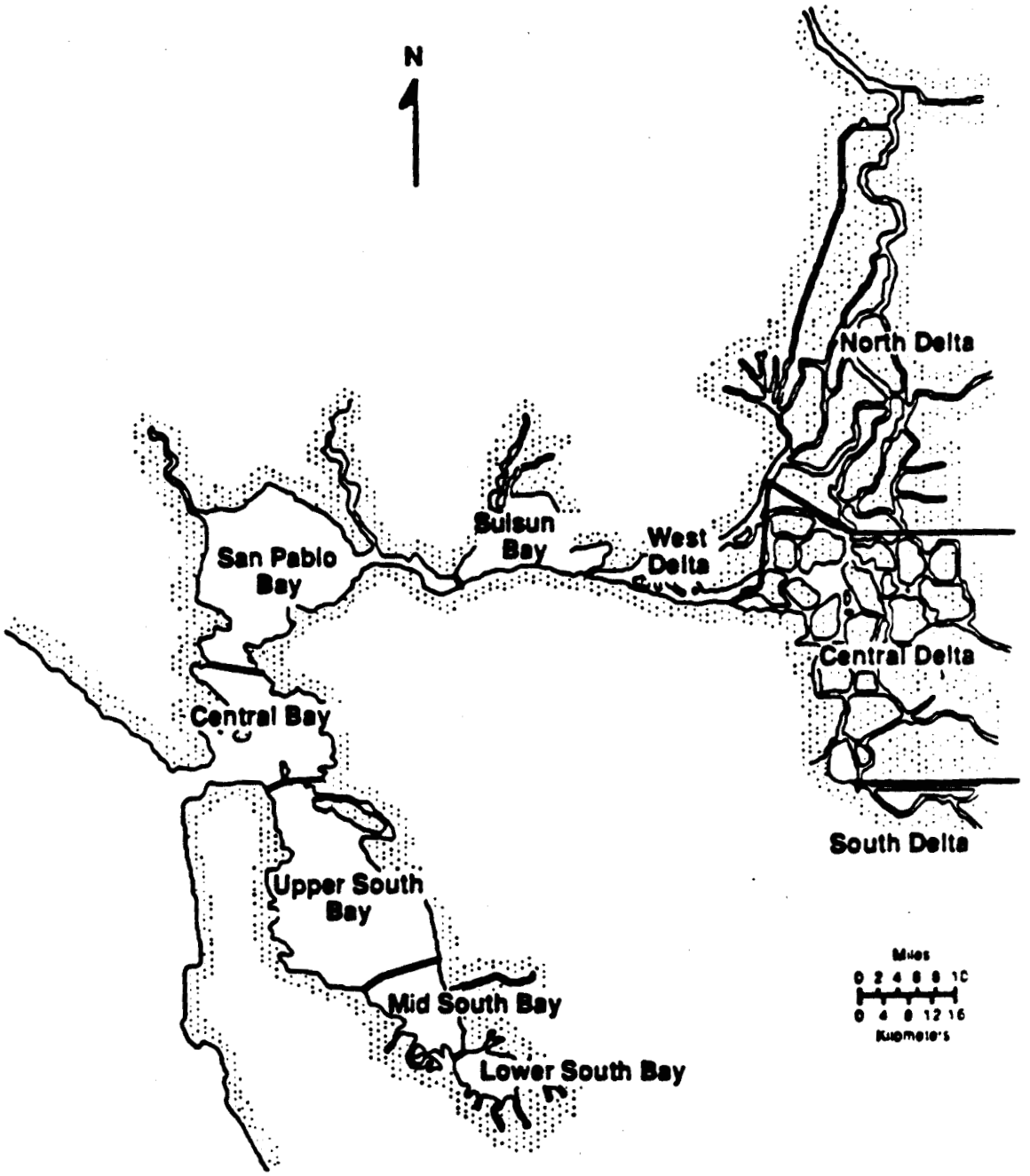
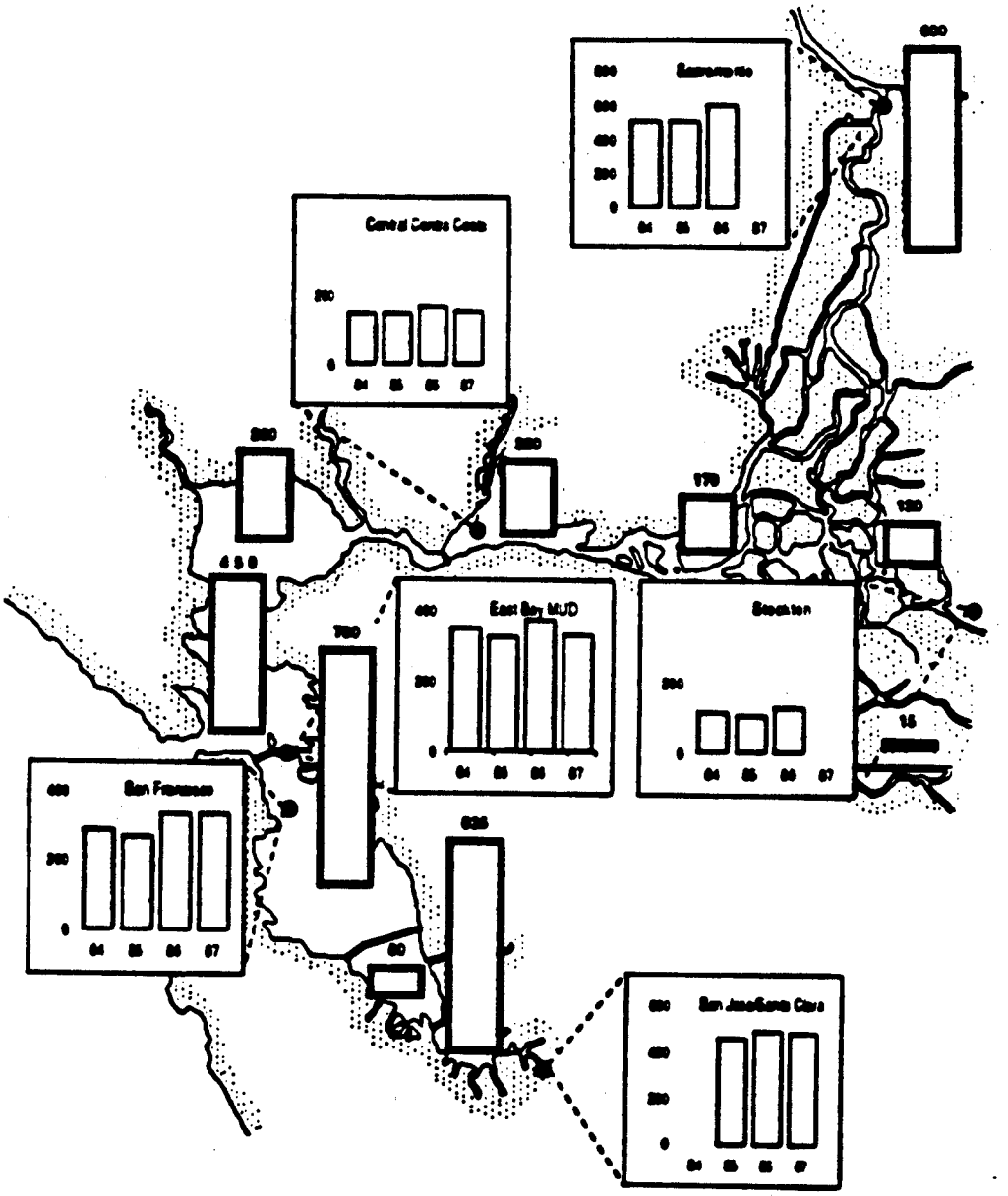


Figure 9. The spatial distribution of effluent flows into the Estuary. Volumes shown are in millions of liters per day. Free-standing columns represent average volumes released into each segment from 1984 to 1986. Inset graphs show trends in flow rates from the largest dischargers in the Estuary.



One trend evident from inspection of Figure 9 is the increased rate of discharge in 1986 relative to the other years. Trends in monthly average flow rates for the largest dischargers (Figure 10) suggest an explanation for this phenomenon. For most of these dischargers, flow peaked in the winter months, the most pronounced peaks occurring in early 1986, when increased flows were observed for every discharge except Stockton. These "pulses" can be attributed to rainfall; the heaviest rains of the 4-yr period occurred in February 1986. Two of these treatment plants, Sacramento and SFSE, have combined sewage/stormwater collection systems; increased flows of stormwater runoff account for the large output of these plants in early 1986.

Other plants included in Figure 10 do not have combined sewers, yet also showed increased output during periods of rainfall. In many municipalities water seeps into sewage collection systems, in a process known as "infiltration/inflow." The effect of wet season increases in flow upon pollutant loads will be addressed below in a discussion of zinc loading.

B. TRACE ELEMENTS

Mass loadings of trace elements are summarized in Figures 11-19. The prevalence of BDL results in the data base makes it impossible to calculate average pollutant loads without making unwarranted assumptions about these semi-quantitative data. Average mass loads from municipal and industrial effluents are therefore presented as ranges in this report. The lower end of each range was calculated assuming that BDL values indicated zero pollutants; the upper ends were calculated assuming that BDL indicated a concentration equal to the detection limit. In these figures, the top of the shaded portion of each column represents the load calculated with BDL values set to zero (the minimum average load), and the top of the unshaded portion is the load calculated with BDL values set to the detection limit (the maximum average load). Actual average loads lie within these limits. Pollutants that are detected infrequently have wide ranges of mass load relative to the minimum average load. The size of the unshaded portion of each column relative to the shaded portion therefore indicates the degree to which pollutant concentrations have not been quantified. Thus, for example, Figure 19 for zinc consists of almost entirely shaded columns, representing a well-quantified pollutant.

Monthly average mass loads are the basis for the statistics presented in this analysis. Mass loads were calculated by multiplying flow rates by the trace element concentrations in treated effluents. Mass loads for 1984 to 1986 were derived from monthly average concentrations and monthly average flows. More detailed data were available for 1987. Therefore, mass loads for that year were derived by combining concentration data with the average daily flow on the sample date. Monthly average mass loads were then calculated by averaging the individual load data obtained during each month. Effluent samples collected by different dischargers for chemical analysis varied from grab samples collected over a 30 day period. Although long-term composite samples provide a better estimate of loads, monthly average mass loads based on these different types of data are treated identically in the following overview.

Figure 10. Trends in monthly average flow rates for the largest dischargers to the Estuary from January 1984 to December 1987.

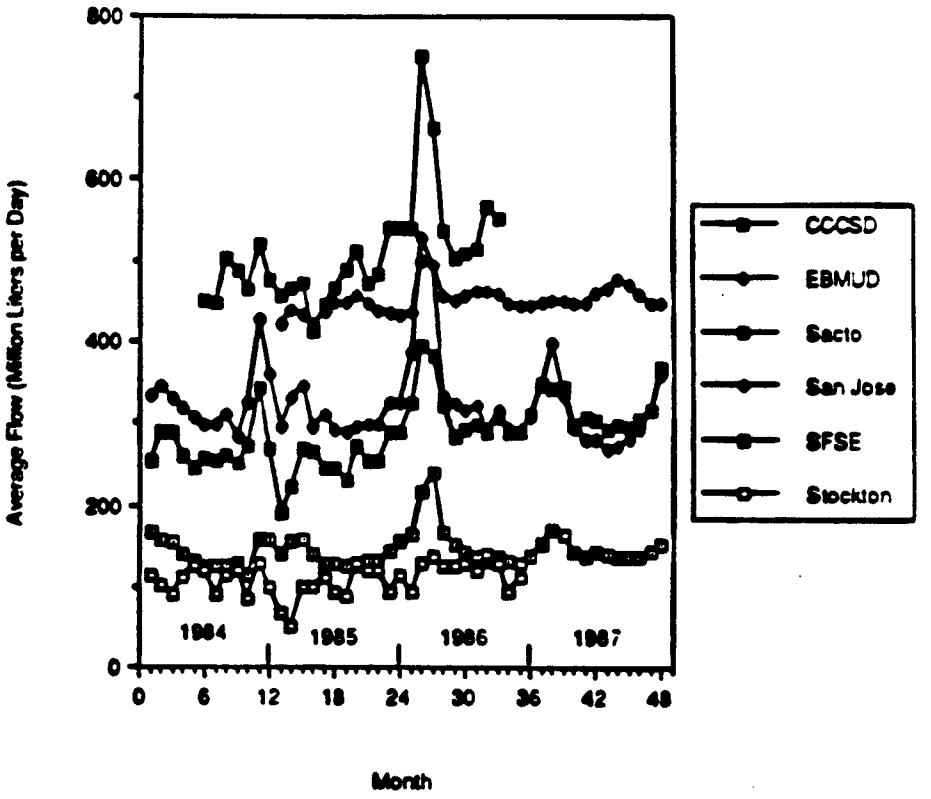


Figure 11. The spatial distribution of arsenic loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of arsenic in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

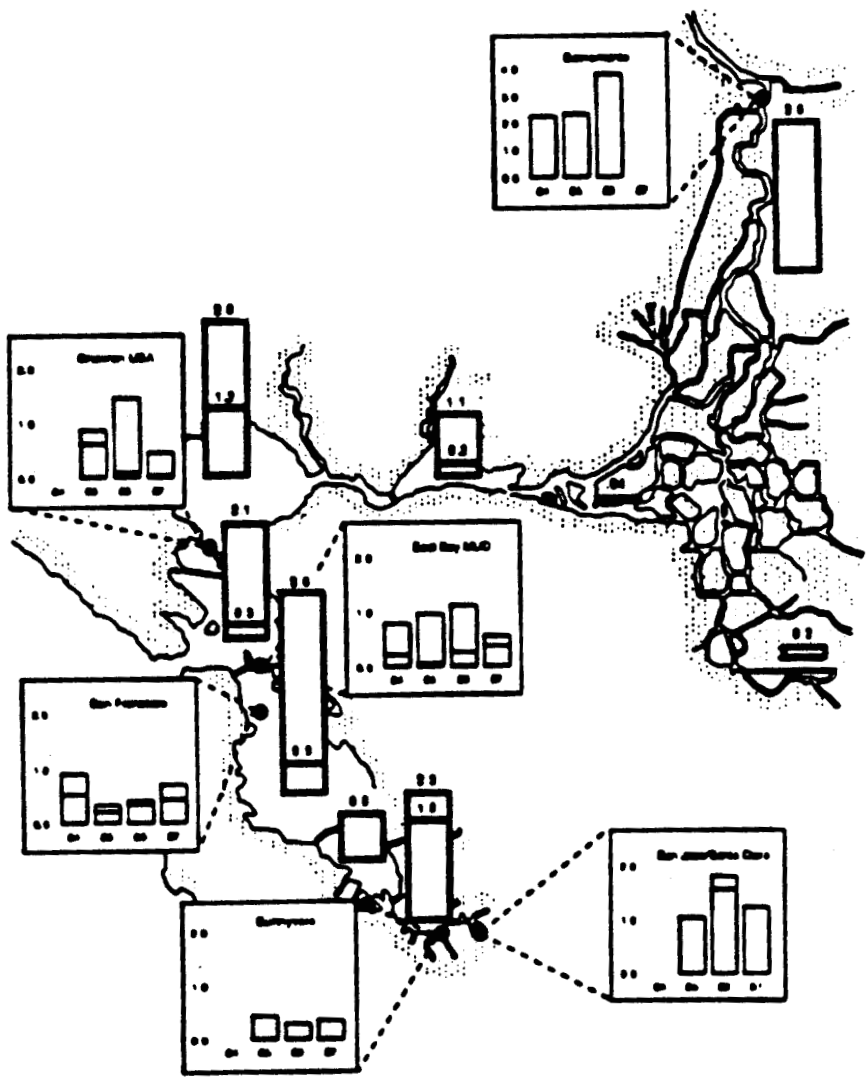


Figure 12. The spatial distribution of cadmium loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of cadmium in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

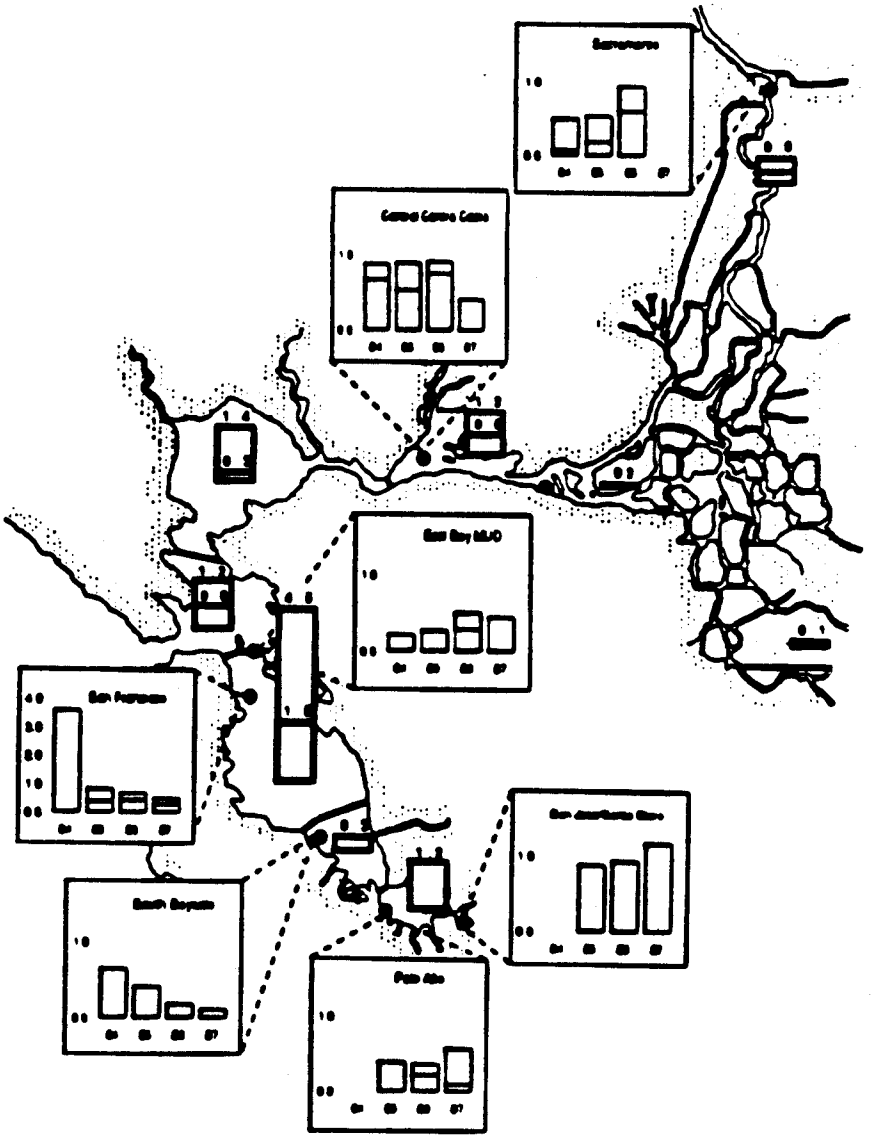


Figure 13. The spatial distribution of chromium loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of chromium in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

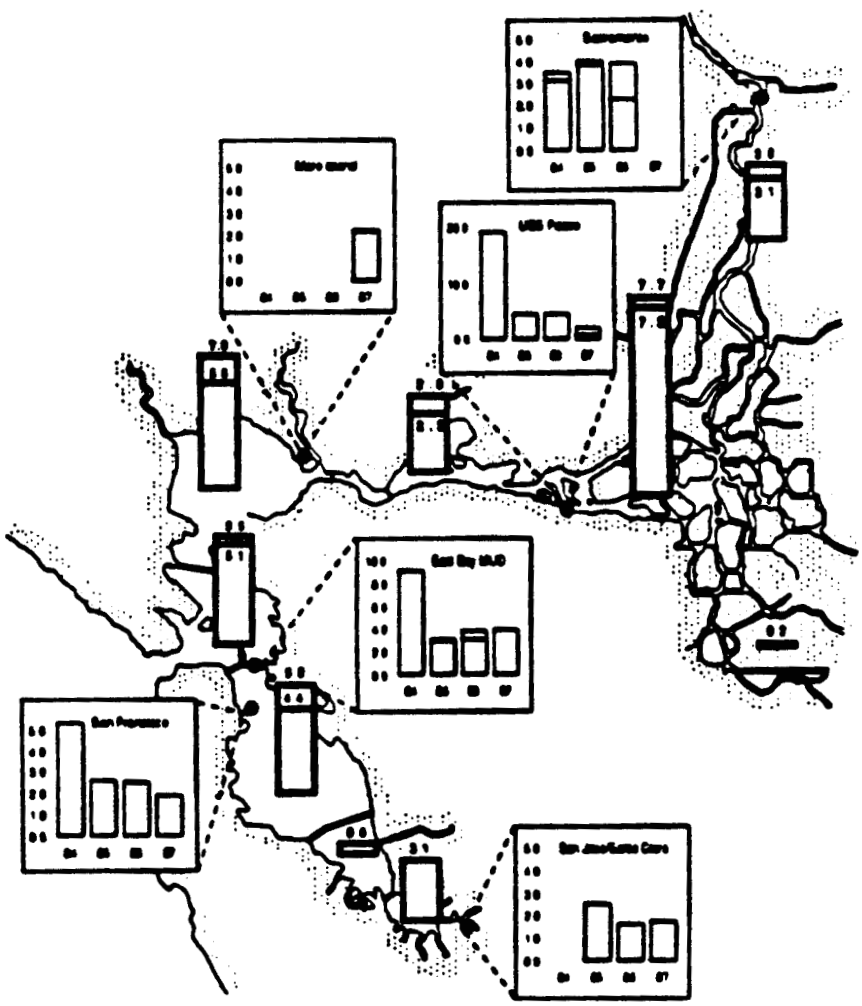


Figure 14. The spatial distribution of copper loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of copper in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

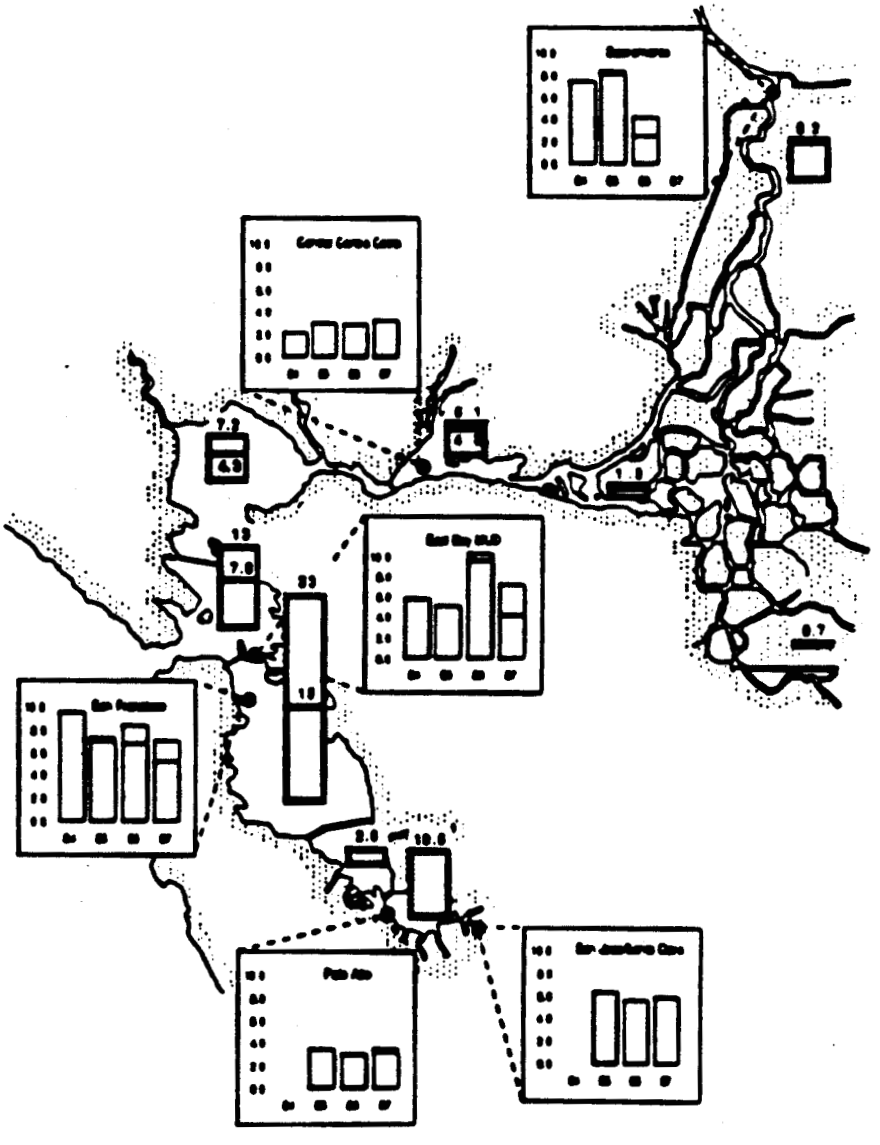


Figure 15. The spatial distribution of lead loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of lead in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

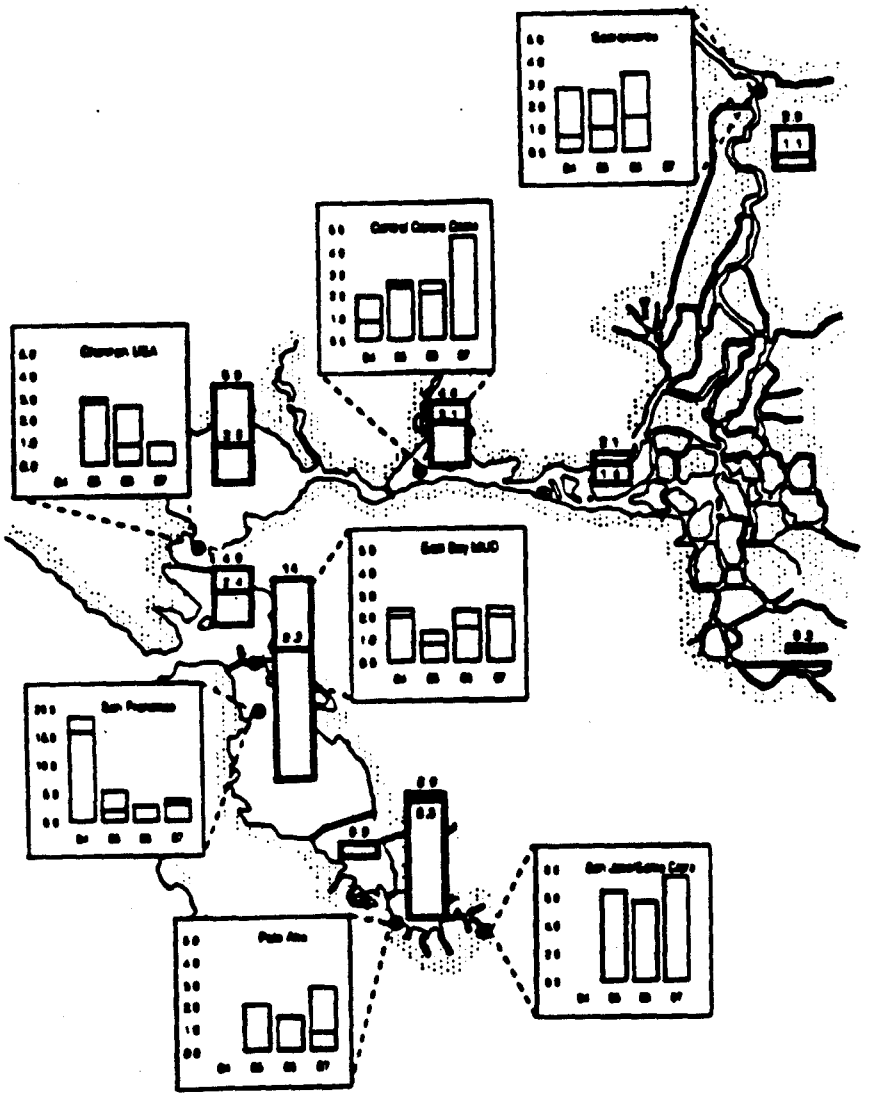


Figure 16. The spatial distribution of mercury loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of mercury in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

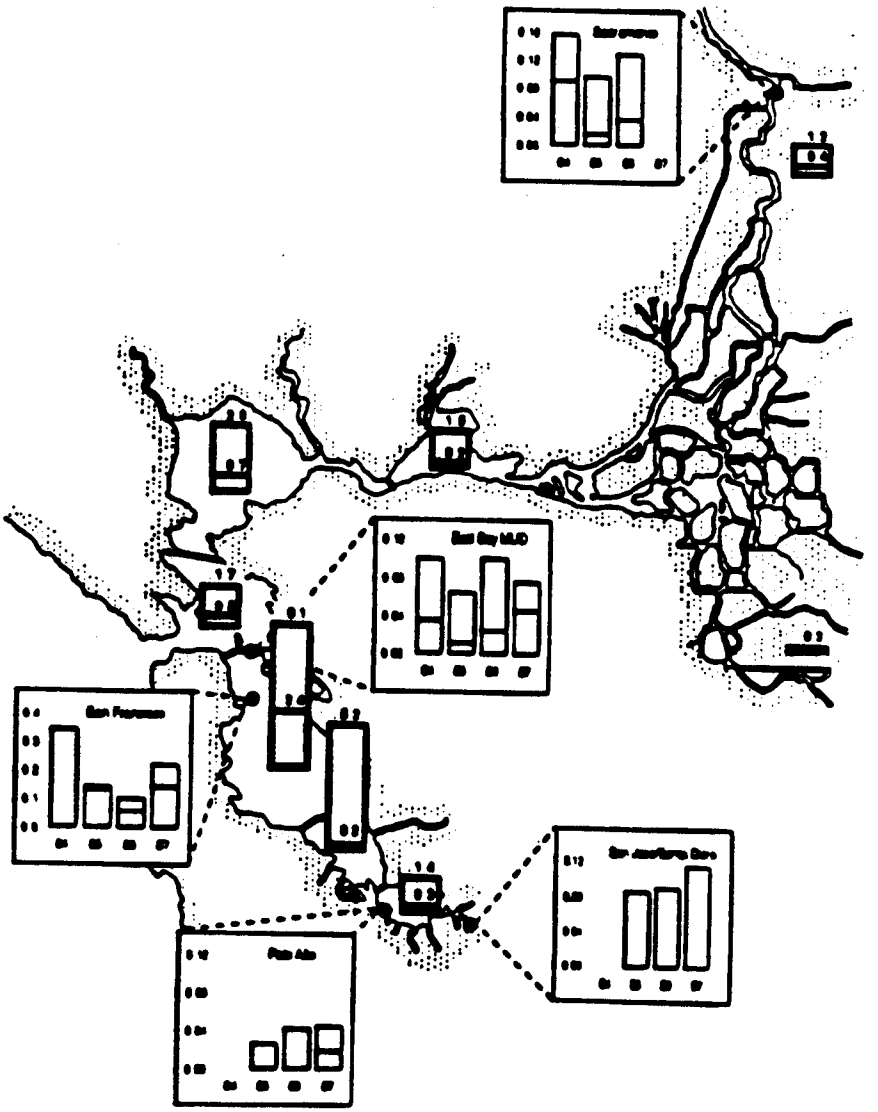


Figure 17. The spatial distribution of nickel loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of nickel in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

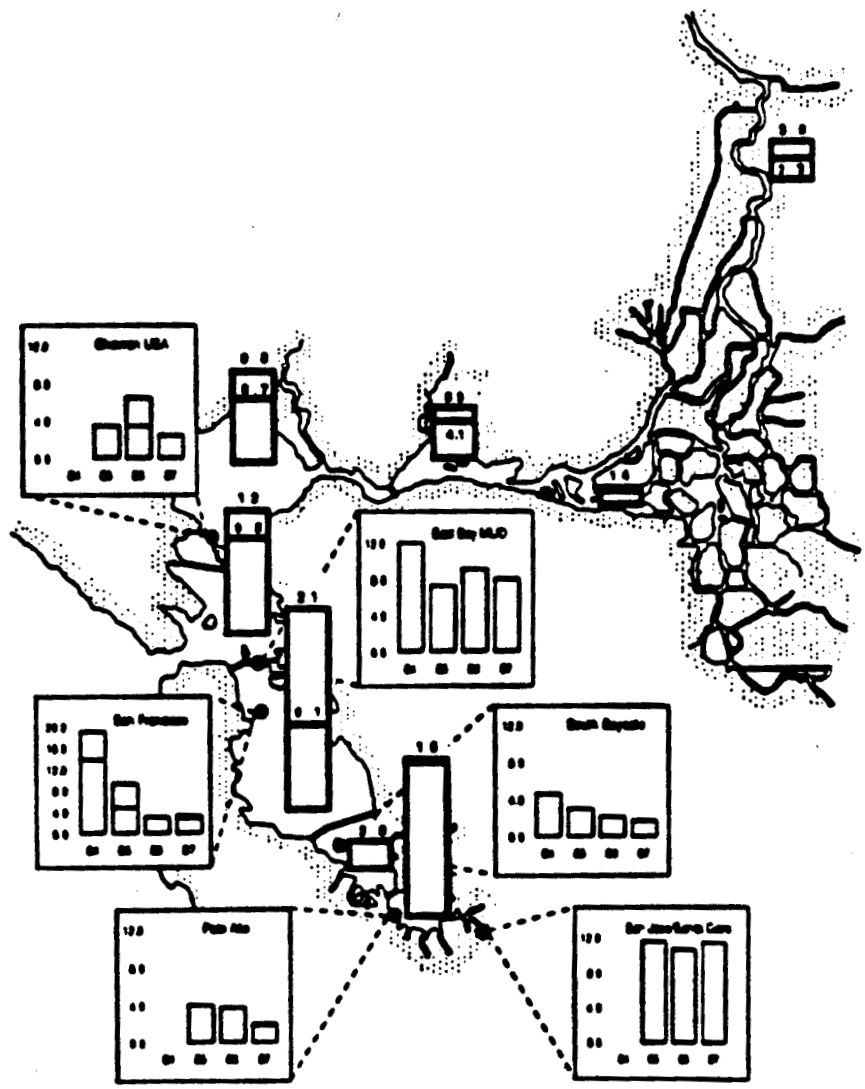


Figure 18. The spatial distribution of silver loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of silver in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.

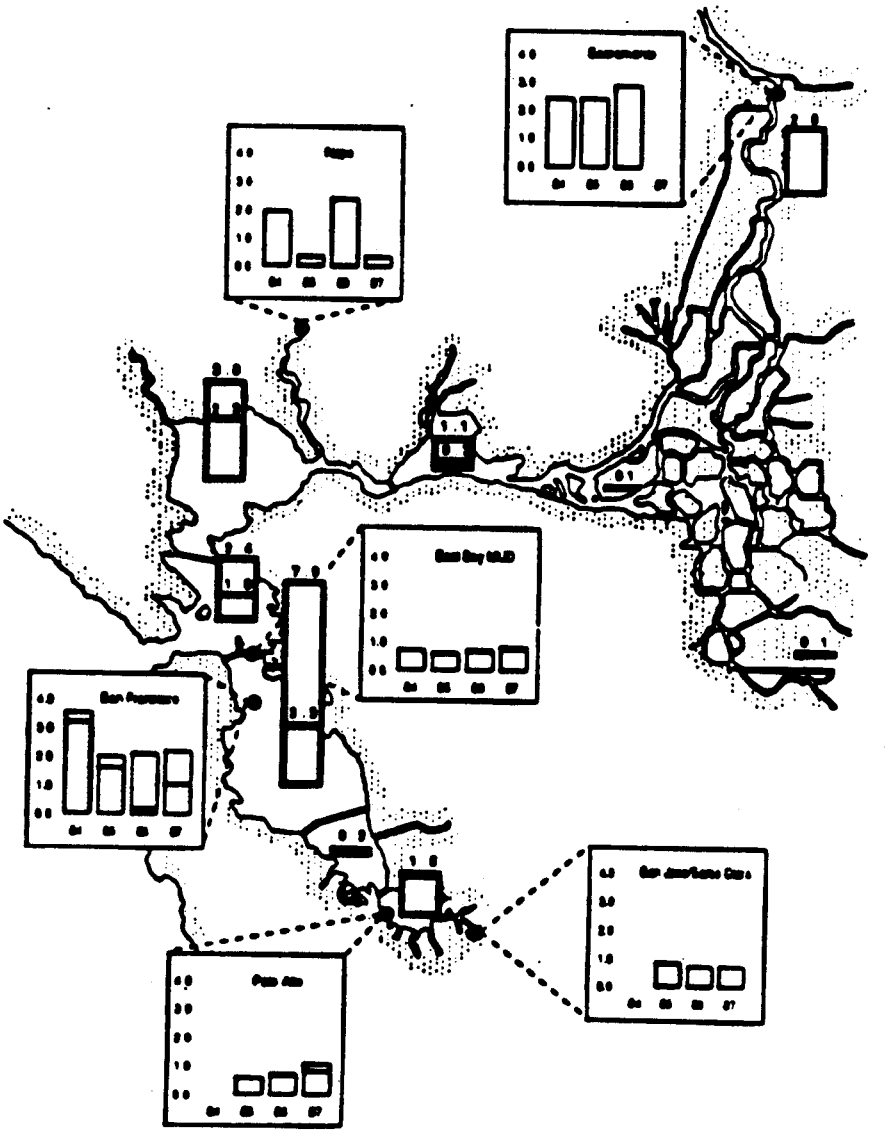
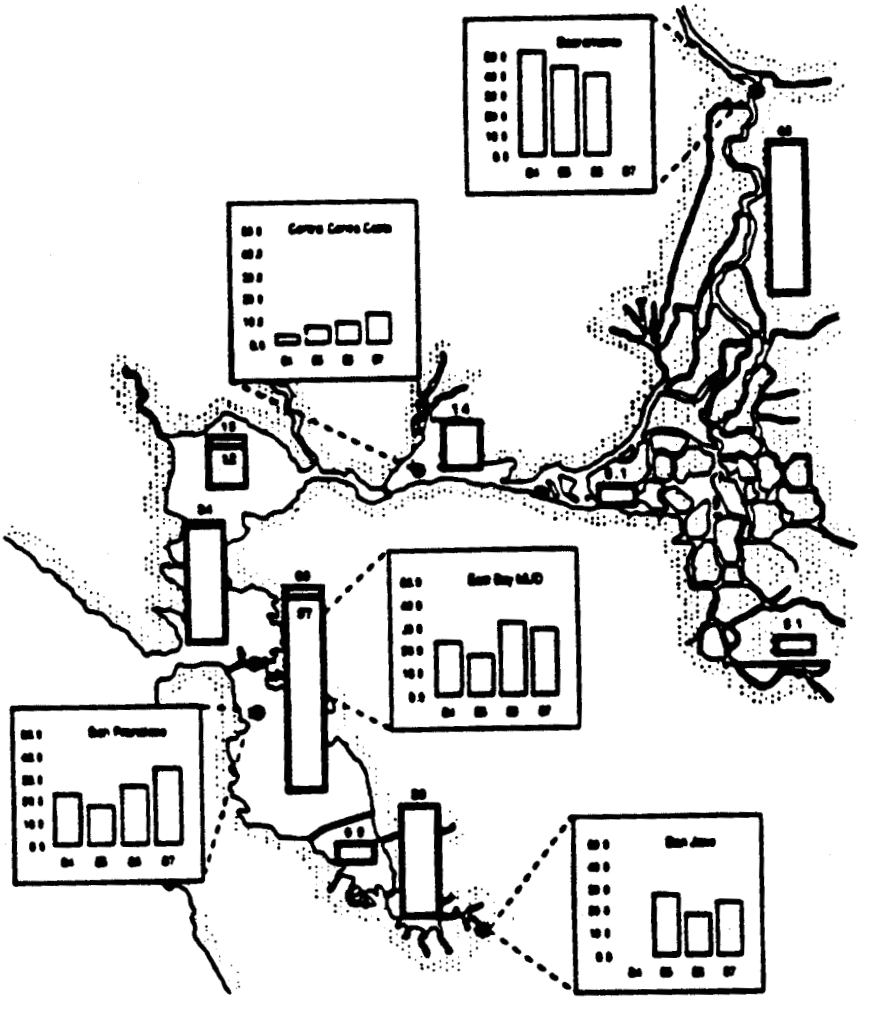


Figure 19. The spatial distribution of zinc loads (kg d^{-1}) into the Estuary, 1984-1987. Free-standing columns represent average loads released into each segment. Inset graphs show trends in loads from the largest discharges of zinc in the Estuary. The height of the shaded portion of each column represents an average load calculated with BDL values set to zero; the total height of each column represents an average load calculated with BDL values set to the limit of detection.



Comparisons of the relative magnitude of loads refer to the minimum average load, or the shaded portion of the columns in Figures 11-19. Unless otherwise noted, the data discussed herein are averages for the 4-yr period of study. Overall average loads of trace elements from municipal and industrial discharges to the Estuary are presented in Table 6 (section IV.B.1.C.).

Arsenic

Arsenic loads to the Estuary have been quantified poorly (Figure 11). The largest measured contributor was San Jose/Santa Clara, but the trace element data for this plant include only four samples each year. The limited data for San Jose/Santa Clara suggest a contribution of 1.3-1.4 kg As d^{-1} to Lower South Bay. The total load of 1.8-2.3 kg As d^{-1} to Lower South Bay is the largest quantified load to any segment. Chevron USA (a refinery), Chevron Chemical (a chemical manufacturer), and Napa Sanitation District contributed most of the total load of arsenic released into San Pablo Bay (1.2-2.8 kg As d^{-1}).

Cadmium

Cadmium loads were greatest in the southern portion of the Estuary (Figure 12). Upper South Bay received the greatest average load (1.5-4.5 kg Cd d^{-1}). The minimum average load was influenced by very high loads from SFSE in 1984 (3.6 kg Cd d^{-1}). Loads from this plant decreased to about 0.5 kg Cd d^{-1} in subsequent years. This pattern for SFSE was repeated for all of the trace elements except arsenic and zinc. Analytical methodology accounted for much of the uncertainty associated with the average Cd load to Upper South Bay, since detection limits were rather high. San Jose/Santa Clara released a fairly consistent 1.0 kg Cd d^{-1} into Lower South Bay. The apparent increase in loads from Sacramento was due to a combination of higher flows at the times of sampling and relatively constant cadmium concentrations.

Chromium

Chromium exhibited an unusual spatial pattern of loading (Figure 13). The largest inputs occurred in the West Delta (7.3-7.7 kg Cr d^{-1}) and San Pablo Bay (5.6-7.0 kg Cr d^{-1}). Most of the load of Cr released into the West Delta (7.3-7.6 kg d^{-1}) was discharged by USS Posco (a steel manufacturer). USS Posco is the largest single discharger of Cr in the Estuary. The high concentrations of Cr in USS Posco's effluent in 1984 (890 µg L^{-1} in September 1984; 810 µg L^{-1} in October 1984) contributed significantly to an average mass load of 19 kg Cr d^{-1} in that year. Average Cr concentrations in this effluent decreased, in subsequent years, from an average of 250 µg L^{-1} in 1984 to 10 µg L^{-1} in 1987. EBMUD, the second largest discharger of chromium in 1984-1987, also had high effluent Cr concentrations during 1984. These concentrations resulted in an average load of 9.0 kg Cr d^{-1}. This was more than double the load observed in any of the next 3 yr. SFSE released larger amounts of Cr in 1984 than in later years. The overall loading of chromium to the Estuary in 1984 was exceptionally

high. A power plant on Mare Island released 2.1-2.2 kg Cr d^{-1} into Mare Island Strait in 1987, the highest rate of release into San Pablo Bay. This input was first discovered in 1987. Although flows from the plant averaged only 10 MLD (2.6 MGD), extremely high Cr concentrations (often reaching 1,000 µg L^{-1} and higher) made this one of the largest discharges of chromium in 1987.

Copper

Copper loading was greatest in the southern reach of the Estuary (Figure 14). The largest discharger was SFSE (7.2-8.0 kg Cu d^{-1}). As observed for chromium, Cu loading from SFSE was greater in 1984 than in subsequent years. Copper loads in other discharges to Upper South Bay, however, were poorly quantified. Lower South Bay received considerable Cu loads from San Jose/Santa Clara (5.9 kg Cu d^{-1}) and Palo Alto (3.2 kg Cu d^{-1}). Sacramento effluent had an average Cu load of 6.2-6.7 kg d^{-1} entering the North Delta. In general, the distribution of copper loads paralleled the pattern observed for flow (Figure 9), reflecting rather consistent Cu concentrations in municipal effluents.

Lead

Lead loads were greatest in the southern reach of the Estuary (Figure 15). San Jose/Santa Clara discharged 6.5 kg Pb d^{-1} into Lower South Bay, accounting for most of the 8.3-8.9 kg Pb d^{-1} entering that segment. Upper South Bay received the largest Pb load of any segment. This was the result of an average load of 5.8-7.6 kg Pb d^{-1} from SFSE. The lead load from SFSE during 1984 was very high (15.8-18.4 kg Pb d^{-1}), decreasing to approximately 2.5 kg Pb d^{-1} in the following 3 yr. Central Contra Costa Sanitation District released the third largest amount of lead in the Estuary (2.5-2.9 kg Pb d^{-1}). However, infrequent sampling of this effluent (four samples per year) made it difficult to interpret the significance of the apparent rise in loading from this discharger during the period of study.

Mercury

Mercury loads to the Estuary were poorly quantified (Figure 16). The average mass load from SFSE (0.17-0.20 kg Hg d^{-1}) was the largest in the Estuary. Temporal trends in mercury loading from SFSE followed the pattern observed for most of the other trace elements, with substantially greater amounts released in 1984 in comparison to 1985-1987.

Nickel

Segments of the Estuary downstream of Suisun Bay received the bulk of the nickel load (Figure 17). An average of 16 kg d^{-1} Ni was discharged into Lower South Bay. The largest single discharger of nickel was San Jose/Santa Clara (11.3 kg d^{-1} Ni). EBMUD was the second largest, discharging 9.1 kg d^{-1} of nickel into Central Bay. SFSE ranked third (6.1-8.7 kg d^{-1} Ni), again showing a higher Ni loading in 1984 than in 1985-1987. Chevron USA released

substantial quantities of nickel (3.1-4.1 kg d^{-1}) into San Pablo Bay. Unlike the loading patterns observed for arsenic (Figure 11) and lead (Figure 15), Ni loads from Chevron USA did not decline appreciably from 1985 to 1987. However, beginning in 1988, pollution prevention procedures instituted by Chevron USA reduced nickel loads to 1.6 kg Ni d^{-1} (T. Mumley, SFBRWQCB, personal communication) (see Section V.B.2.B. for discussion).

Silver

Silver loading showed a distinct spatial pattern (Figure 18). San Pablo Bay received the largest load (2.3-3.8 kg Ag d^{-1}), most of this from a small municipal treatment plant (30 MLD; 8 MGD) serving the City of Napa. This discharge is released into the Napa River, not directly into San Pablo Bay. High concentrations of silver in this effluent (> 100 µg Ag L^{-1} on two occasions during 1984-1987) account for the large mass load from this facility. Infrequent sampling of this effluent casts some doubt on the quality of the data; only eight samples were taken between 1984 and 1987 because the wastewater is reclaimed during the dry season. More frequent sampling would be required to confirm the significance of silver loads from this plant.

The second greatest discharger of silver was SFSE (1.5-2.4 kg d^{-1}). SFSE discharged Ag at a greater rate in 1984 than in later years. Variation in loads measured by SFSE in 1985-1987 resulted from concentrations being close to the analytical limits of detection.

Loads of Ag from Palo Alto (0.7-0.8 kg d^{-1}) were comparable to those from San Jose/Santa Clara (0.8 kg Ag d^{-1}) and EBMUD (0.7 kg Ag d^{-1}), even though Palo Alto released a smaller volume of effluent than the latter two plants. The large range in Ag loading to the Upper South Bay (2.2-7.9 kg d^{-1}) is a product of infrequent detection of silver because the methods used had relatively high limits of detection.

Zinc

Zinc is the element for which the best data are available in Bay-Delta effluents (Figure 19). Zinc concentrations in effluents are typically much higher than other trace elements, eliminating the problems of below-detection-limit data. The largest discharger was Sacramento, which released an average of 45 kg Zn d^{-1} into the North Delta; the maximum annual average of 52 kg Zn d^{-1} occurred in 1984. The next largest dischargers, SFSE and EBMUD, both discharged 25 kg Zn d^{-1}. Zinc loads from SFSE were relatively constant during the 4-yr period, lacking the pronounced peak observed for other trace elements in 1984.

The Upper South Bay received the largest Zn load, including contributions from Union Sanitary District (6.8-6.9 kg Zn d^{-1}), San Francisco's North Point plant (treating stormwater runoff in wet weather; 6.6 kg Zn d^{-1}), and Oro Loma (5.9-6.3 kg Zn d^{-1}), as well as the load from SFSE. The Central Bay

and Lower South Bay received approximately equal quantities of zinc, with EBMUD (25 kg d^{-1}) and San Jose/Santa Clara (23 kg d^{-1}) accounting for most of the Zn loads for their respective segments.

Monthly trends in zinc loads from three of the largest dischargers are shown in Figure 20. Monthly average flow data have also been plotted for comparison. Zinc loads from these plants were variable, ranging from less than 10 to over 60 kg Zn d^{-1} in each case. Some of the peaks in zinc loads coincided with increased flow; however, in many instances elevated loads occurred without increases in flow. It appears that the rainfall-induced pattern of temporal variability in flow rates described previously, with peak flows occurring during the wet-season months, does not translate directly into a similar pattern for pollutant loads. It is possible, however, that high flows associated with rainfall create temporarily high trace element loads at the onset of storms (or "first-flush") that is not detected by effluent monitoring programs. Much of the variation in Zn load derives from factors that contribute to variation in pollutant concentrations in effluent. Among these factors are concentrations in treatment plant influent, performance of the plant in pollutant removal, and variation introduced in the process of analyzing pollutant concentrations in effluent.

Selenium

Few data are available on selenium concentrations in effluents. Bay Area refineries began monthly analysis of selenium in early 1985; weekly selenium analysis began in early 1986 and has continued to the present. Average Se loads calculated for 1986 and 1987 are more reliable than those for 1985 because of more frequent sampling and the use of more accurate analytical methods (Table 5). Annual fluctuations in selenium loads calculated for Chevron are due to changes in analytical methods and the reporting of results. In 1987 Chevron employed more sensitive analytical methods and took a large number of samples (samples were collected daily in several months); the average load calculated for that year provides an accurate representation of Chevron's actual Se discharge.

Reliable Se data do not exist for other dischargers; the data base for this element is incomplete. Some selenium analyses have been performed on municipal effluents under the NPDES program; however, the value of those data is limited due to infrequent sampling and the use of methods with high limits of detection (1 μg L^{-1} or higher).

Fortunately, a recent study of selenium cycling in the Estuary by the California Department of Water Resources has yielded reliable data on contribution of Se from POTWs and refineries (Cutter, 1989a,b). Techniques achieving a detection limit of 5 nanograms per liter (ng L^{-1}) for each of three different forms of selenium were employed. The only shortcoming of these data is that few samples were collected, particularly for municipal dischargers. In general, 10 samples were collected from each refinery from February 1987 to June 1988, and 4 samples were collected from each POTW from December 1987 to March 1988.

Figure 20. Monthly average loads (kg d^{-1}) of zinc from three major discharges, January 1984 to December 1987. Monthly average flows (MLD) are plotted for comparison.

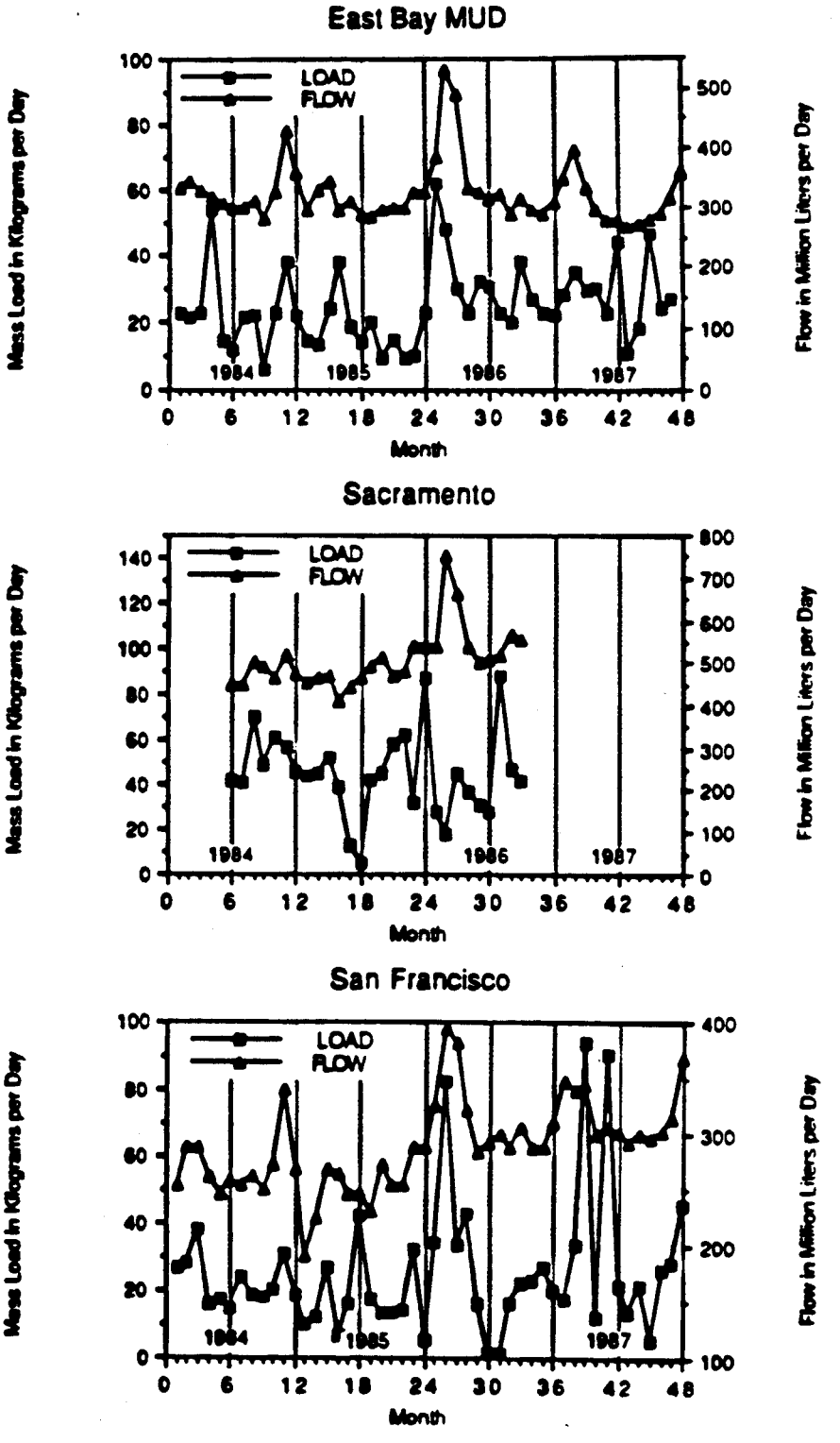


Table 5. Average selenium loads (kg d^{-1}) measured by Bay Area refineries from 1985 to 1987. Minimum averages calculated with BDL values set to zero; maximum averages calculated with BDL values set to the limit of detection.

	1985		1986		1987	
	Min	Max	Min	Max	Min	Max
Chevron	2.17	2.43	0.30	2.13	1.00	1.00
Exxon	0.20	0.22	0.41	0.46	0.72	0.72
Pacific	0.12	0.12	0	0.02	0.02	0.02
Shell	1.32	1.32	1.78	2.19	1.96	1.96
Tosco	0.88	0.89	0.48	0.53	0.43	0.46
Union	1.17	1.17	1.70	1.70	1.28	1.28

Figure 21 shows selenium loads computed from average concentrations reported in the DWR study (Cutter, 1989a,b) and 1987 annual average flows (these were the most recent flow data compiled for this report). Loads calculated in this manner for the refineries agree with loads recently estimated by the refineries (Table 5). The DWR data showed that selenite (Se^{4+}) is the predominant form discharged by refineries. The loads calculated for POTWs are based on few data points and should be considered preliminary estimates only. The POTW data show, however, that measurable concentrations of selenium are released by major municipal facilities, principally as selenate (Se^{6+}). Concentrations of Se in the POTW effluents were generally below 1 $\mu g\ L^{-1}$. Four samples from Palo Alto, however, averaged 1.32 $\mu g\ L^{-1}$ Se, and four samples from San Jose/Santa Clara averaged 0.96 $\mu g\ L^{-1}$ Se. The magnitude of the loads estimated for refineries and municipal dischargers is consistent with concentrations measured in the water column of the Estuary (Cutter, 1989a).

C. OVERVIEW OF LOADS FROM MUNICIPAL AND INDUSTRIAL DISCHARGES

Table 6 provides a compilation of loads calculated for each segment. Selenium is not included in the table because of the limited data available. In summary, Upper South Bay received the largest average loads of cadmium, lead, mercury, silver, and zinc. The loads of cadmium, lead, mercury, and silver calculated for Upper South Bay were heavily influenced by high loads from SFSE in 1984. Elevated concentrations of these elements in SFSE effluent in 1984 were scattered throughout the year; it is unlikely that isolated incidences of contamination accounted for the extremes in loading.

The Lower South Bay received the largest loadings of arsenic, copper, and nickel. San Jose/Santa Clara contributed most of the pollutant load to this segment. Calculated loads for San Jose/Santa Clara, however, are uncertain because of the infrequent sampling performed by this facility.

San Pablo Bay received the largest average load of silver, primarily due to high concentrations in the effluent from Napa during 1984 and 1986. However, estimated loads from Napa are also uncertain due to infrequent sampling. The West Delta received the largest average load of chromium, principally from USS Posco. Chromium loads from USS Posco were greater in 1984 than in succeeding years. Mass loads into the Central Delta are not included in the table because concentration data were not available for the largest discharger into that segment, the City of Stockton (109 MLD; 29 MGD).

With the exception of selenium, no local data are available on the speciation of pollutants in Bay/Delta effluents. Trace elements in treated effluents are likely to be predominantly in dissolved form. Secondary treatment is designed to remove suspended solids from waste water. Particle-associated trace elements are also removed in this process. Less soluble elements, such as chromium, are removed more efficiently than more soluble elements, such as nickel. As a result of this reduction in particulate loads, soluble species of trace

Figure 21. The spatial distribution of average loads (kg d^{-1}) of three forms of selenium into the Estuary. Data from Cutter (1989a,b). Columns represent averages of 10 samples collected from refineries and 4 samples from municipal discharges in 1987 and 1988.

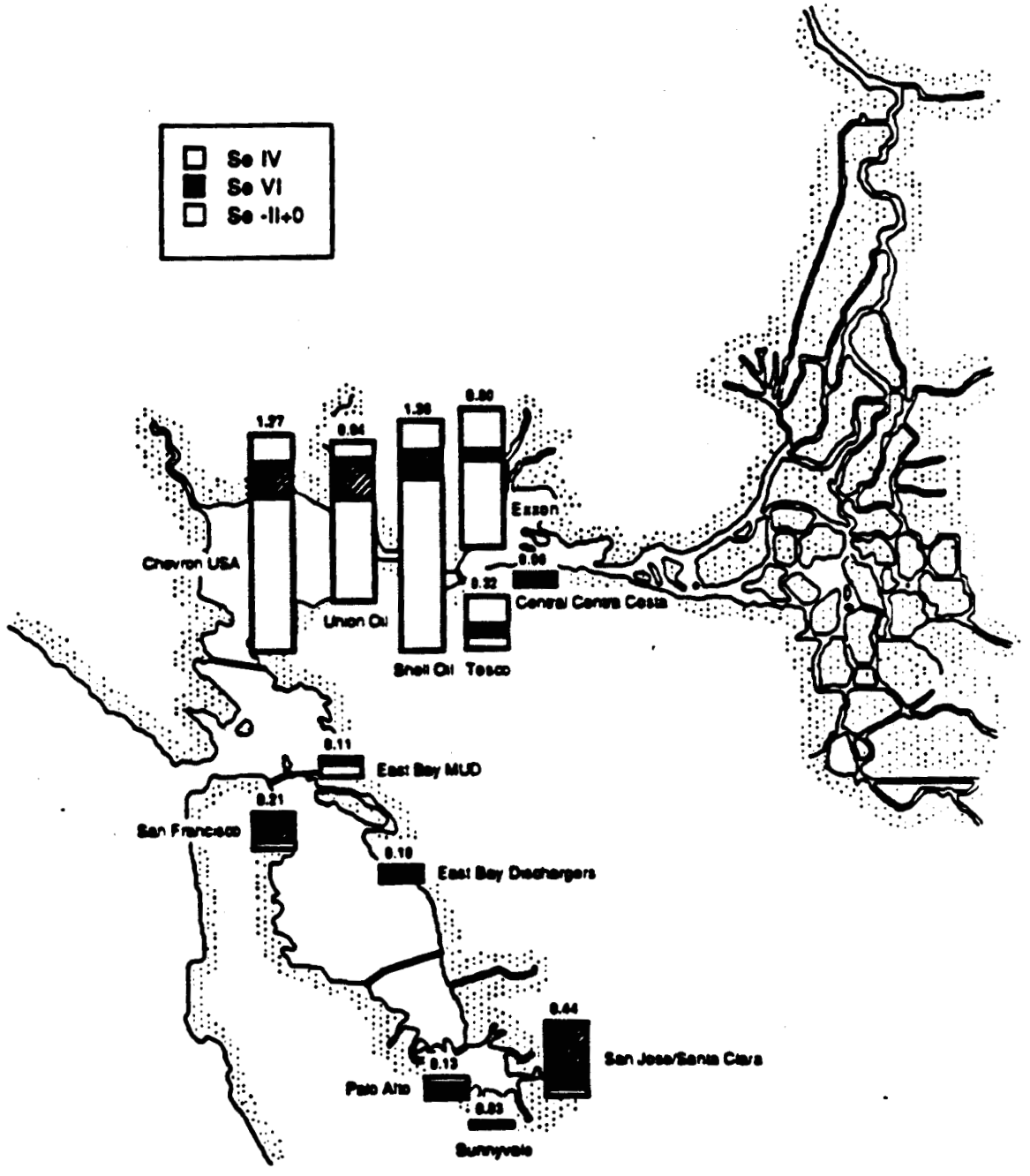


Table 6. Summary of flows and pollutant loads to various segments of the Estuary from municipal and industrial discharges, 1984-1987. Flow data in million L d^{-1}, load data in kg d^{-1}.

	Flow	Arsenic		Cadmium		Chromium	
		Low	High	Low	High	Low	High
North Delta	602	0.0	2.6	0.3	0.6	3.1	3.8
Central Delta	121	0.0	0.0	0.0	0.0	0.0	0.0
South Delta	15	0.0	0.2	0.0	0.1	0.0	0.2
West Delta	167	0.0	0.0	0.0	0.0	7.3	7.7
Suisun Bay	220	0.2	1.1	0.6	1.2	2.3	2.9
San Pablo Bay	261	1.2	2.8	0.3	1.4	5.6	7.0
Central Bay	450	0.3	2.1	0.6	1.2	5.1	5.5
Upper South Bay	696	0.5	3.6	1.5	4.5	4.4	5.8
Middle South Bay	79	0.0	0.8	0.3	0.3	0.6	0.6
Lower South Bay	625	1.8	2.3	1.2	1.4	3.1	3.1
Estuary Total	3236	4.0	15.5	4.8	10.7	31.5	36.6

	Copper		Lead		Mercury	
	Low	High	Low	High	Low	High
North Delta	6.2	6.8	1.1	2.9	0.04	0.12
Central Delta	0.0	0.0	0.0	0.0	0.00	0.00
South Delta	0.0	0.7	0.0	0.2	0.00	0.02
West Delta	1.2	1.9	1.4	2.1	0.00	0.00
Suisun Bay	4.3	5.1	3.1	4.6	0.02	0.15
San Pablo Bay	4.3	7.2	2.5	6.9	0.07	0.28
Central Bay	7.8	13.0	2.4	4.0	0.05	0.17
Upper South Bay	15.0	33.0	9.2	14.0	0.24	0.61
Middle South Bay	2.6	2.6	0.9	1.0	0.02	0.52
Lower South Bay	10.5	10.5	8.3	8.9	0.03	0.14
Estuary Total	52	81	29	45	0.5	2.0

	Nickel		Silver		Zinc	
	Low	High	Low	High	Low	High
North Delta	2.3	3.8	0.0	2.5	45	45
Central Delta	0.0	0.0	0.0	0.0	0	0
South Delta	0.0	0.0	0.0	0.1	5	5
West Delta	1.4	1.9	0.0	0.0	6	6
Suisun Bay	4.1	5.3	0.3	1.1	14	14
San Pablo Bay	6.7	9.8	2.3	3.8	12	15
Central Bay	9.5	12.0	1.0	2.4	34	35
Upper South Bay	9.1	21.0	2.2	7.9	57	60
Middle South Bay	2.8	2.8	0.2	0.2	6	6
Lower South Bay	16.0	16.0	1.5	1.6	33	33
Estuary Total	52	73	7.5	19.6	212	219

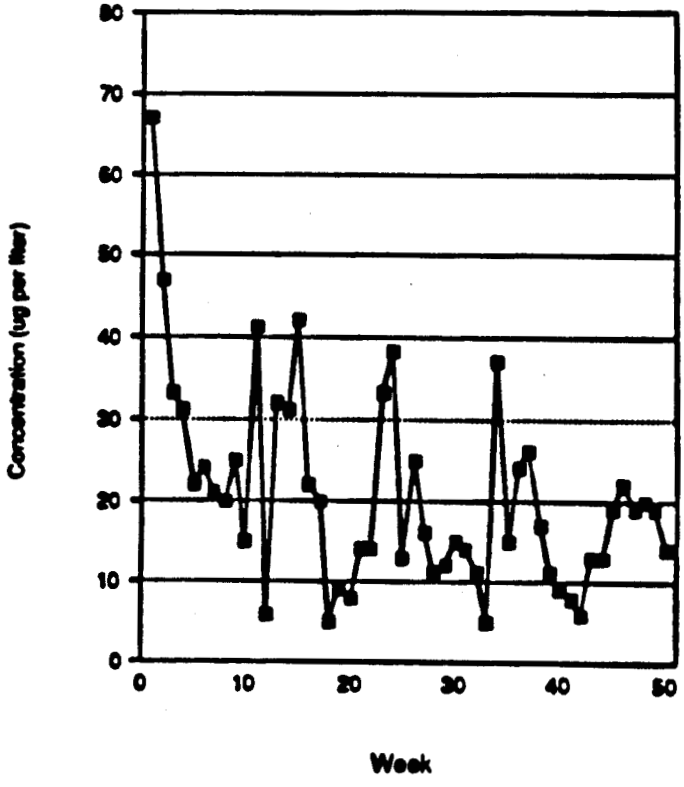
elements comprise a greater percentage of the species present in the final effluent (Morel and Schiff, 1983).

A number of factors make interpretation of loads calculated from the trace element data difficult. One of these is the infrequent sampling undertaken by some of the major dischargers (most notably San Jose/Santa Clara) between 1984 and 1987. Few dischargers analyze pollutants in effluents 64 on a weekly basis. Weekly variation in copper concentrations from the Sunnyvale plant demonstrate the extent of variability that may occur in effluent characteristics. Copper concentrations in 1987 varied from 5 to 67 μg Cu L^{-1} (Figure 22). Given such wide variation, it is possible that erroneous conclusions about trends in loads from discharges may be drawn. Perhaps the major gap in loading data derives from sampling that has not provided an adequate description of temporal variability in effluent element concentrations.

The most severe constraint on the interpretation of loads of the numerous pollutants analyzed in effluents is the use of analytical methods with detection limits that are close to, or exceed, the actual concentrations present. Many of the analyses reported by Bay-Delta dischargers are of little use in an calculating mass loading because of the prevalence of BDL results.

Much of the variation in loads that are relatively well-quantified (e.g., Zn) derives from temporal variation in effluent pollutant concentrations. Factors that contribute to this variation are concentrations in treatment plant influent, performance of the plant in pollutant removal, and variation introduced in the process of analyzing pollutant concentrations in effluent. More rigorous quality assurance programs would reduce at least that portion of the variation that is due to chemical analysis of pollutant concentrations in effluent. Quality assurance data, however, are generally unavailable.

Figure 22. Weekly concentrations of copper measured in effluent from the Sunnyvale Water Pollution Control Plant, 1987.



2. Urban and Nonurban Runoff

In contrast to the more constant discharges of municipal and industrial effluents, urban and nonurban runoff are episodic and seasonally variable. The following discussion describes urban and nonurban runoff and the pollutants of concern associated with them, provides estimates of pollutant loads to the Estuary where possible, and describes important gaps in knowledge. This discussion is based upon the review of Gunther *et al.* (1987); the reader is referred to that document for a more detailed discussion of these topics.

Estimates for urban and nonurban runoff pollutant loads relate to runoff within the basin of San Francisco Bay and within the boundaries of the Delta. Urban and nonurban runoff upstream of the Delta contribute to mass transport by the major rivers of the Central Valley; estimates of these riverine loads are given in the next section.

A. URBAN RUNOFF

This section of the report identifies the pollutants associated with urban runoff and estimates the magnitude and variability of pollutant loads to the Estuary from this input. Important gaps in knowledge, including the limitations of existing pollutant measurements, are discussed.

Since the review of Gunther *et al.* (1987), two additional studies have been conducted. First, the CVRWQCB monitored the concentration of pollutants in runoff from an urban watershed in Sacramento (Montoya *et al.* 1988; Montoya, 1989). Second, local agencies from Santa Clara County investigated pollutant concentrations in runoff to the South Bay (Woodward-Clyde, unpublished). Both studies used extensive quality assurance programs to produce reliable data; the data from the Santa Clara Valley study are preliminary and are only described briefly.

The substitution of impervious for permeable surfaces during urban development reduces the ability of a region to absorb precipitation. This causes an increase in the volume and rate of surface runoff from storms. Municipalities have made extensive investments in systems of channels and conduits to manage urban storm waters. Pollutants deposited upon urban surfaces can be dissolved or suspended in runoff from urban areas. In the San Francisco Bay Basin Plan, surface runoff from developed areas is considered a major load of pollutants to the Estuary (SFBRWQCB, 1986).

Pollutants in urban runoff originate from different urban land uses. These include commercial, light industrial, and residential (including lawns and gardens) uses, as well as managed open-space areas such as parks, cemeteries, planted road dividers, and construction sites (Pitt and Shawley, 1981; BCDC,1987). Urban applications of insecticides for mosquito and domestic pest control, as well as herbicides to maintain golf courses, right-of-ways, and residential landscaping, contribute toxic pollutants to urban runoff. About 68% of malathion applied in the Bay-Delta region is used on urban flora

(Pait *et al.*, 1989); some of this may be transported to the Estuary. The presence of some trace metal pollutants, particularly lead, zinc, and copper, results from automobiles, including exhaust emissions, metal abrasion, and the wear of tires and brake-shoe linings (Montoya, 1987).

Other sources of trace elements in urban runoff include atmospheric fallout and illegal discharges to urban surfaces. A recent study of the Trinity River in Fort Worth, Texas, for example, showed that illegal dumping of waste crankcase oil in storm drains contributed significantly to elevated PAH concentrations in the river and sediments (Irwin, 1988). Numerous reports to Regional Water Quality Control Boards of oil slicks, unnaturally discolored waters, odorous foreign compounds, and suspicious containers in and near surface waters, suggest that illegal dumping of harmful compounds occurs in the Basin of the Estuary as well (B. Montoya, CVRWQCB, personal communication). Based on a survey of three communities in the Bay Area, Russell and Meiorin (1985) estimated that at least 50% of used motor oil from households is illegally dumped, either on the ground, in storm sewers, or in landfills.

Pollutants in Urban Runoff

Data on the concentration of pollutants in urban runoff in the Bay-Delta region are sparse. Gunther *et al.* (1987) reviewed seven studies conducted in the region. Reliable data were sparse due to lack of quality assurance or due to inadequate sampling techniques. National sampling under the Nationwide Urban Runoff Program (NURP) detected all 13 priority trace elements in urban runoff. All but 3 elements had frequencies of detection greater than 10% (Table 7) (USEPA, 1983). National data showed that organic pollutants in runoff include solvents, pesticides, fuel oils, combustion products, and lubricants, as well as synthesized polymers and resins.

Measurements of "oil and grease" concentrations in urban runoff in the Bay-Delta region have been performed either by extraction in Freon 113 followed by infrared spectrophotometry (Stenstrom *et al.*, 1984), or extraction in dichloromethane, evaporation, and gravimetric determination (Silverman and Stenstrom, 1985; Montoya, 1989). These methods overestimate toxic hydrocarbon concentrations due to the co-extraction of non-toxic oils, waxes, and fats. Few studies have utilized more precise, more expensive techniques to measure the concentrations of hydrocarbons in urban runoff. Those studies have measured various PAHs and chlorinated hydrocarbons in runoff. Most such studies have been conducted outside the Bay-Delta area.

The PAHs detected with a frequency greater than 10% in the NURP program were n-naphthalene, pyrene, fluorene, and phenanthrene. Higher molecular weight PAHs are detected more frequently in urban runoff because low-molecular-weight PAHs and monocyclic aromatic hydrocarbons (MAHs) are more easily volatilized from urban surfaces (see Gunther *et al.*, 1987).

Table 7. Event mean concentrations of trace metals in urban runoff from the Nationwide Urban Runoff Program and the City of Sacramento. City of Sacramento values are median of 11 storm events. All concentrations in $\mu g\ L^{-1}$. Dashes indicate that data were not collected.

	USEPA (1983)	Sacramento, 1986-87 (Montoya *et al.*, 1988)	
Pollutant	**Conc.Range (Frequency of Detection [%])**	**Wet Season**	**Dry Season**
Antimony	2.6 - 23 (13)	—	—
Arsenic	1 - 51 (52)	5	4
Beryllium	1 - 49 (12)	—	—
Cadmium	0.1 - 14 (48)	1	0.2
Chromium	1 - 190 (58)	17	3
Copper	1- 100 (91)	29	10
Lead	6 - 460 (94)	84	3
Mercury	0.6 - 1.2 (9)	—	—
Nickel	1 - 182 (43)	18	1
Selenium	2 - 77 (11)	—	—
Silver	0.2 - 0.8 (7)	—	—
Thallium	1 - 14 (6)	—	—
Zinc	10 - 2400 (94)	247	89

Many chlorinated hydrocarbons can also be found in urban runoff. These include industrial solvents (such as trichloroethene and carbon tetrachloride), pesticides (including DDT), and polychlorinated biphenyls (PCBs). Analyses of samples from the Trinity River (Texas) showed that sites receiving substantial runoff from either downtown Fort Worth or Dallas had significantly greater PCB concentrations than other sites. At one downstream sampling site, a storm drain canal, concentrations of PCBs were found in spiny softshell turtles two orders of magnitude above the concentrations found in the same species at a reference site (Irwin, 1988). PCBs have been detected in urban runoff in the Bay-Delta region at concentrations less than 0.5 $\mu g\ L^{-1}$. In the Sacramento River, the lower American River, and in four streams of the South Bay, PCBs were measured at concentrations less than 0.19 $\mu g\ L^{-1}$ (Silverman and Stenstrom, 1985; BCDC, 1987; Montoya, 1987; Woodward-Clyde, unpublished).

In urban storm waters throughout the country, the NURP detected 12 of 17 chlorinated hydrocarbon pesticides analyzed with 3 (α-BHC, γ-BHC, and α-endosulfan) detected in more than 10% of the samples (Table 8). Alpha- and gamma-BHC have been found in all Sacramento storm drains sampled by the Sacramento Water Quality Control Laboratory (SWQCL). Concentrations ranged from 0.003 to 0.243 $\mu g\ L^{-1}$ (SWQCL, unpublished: in Montoya, 1987). Preliminary data showed that both endosulfan and chlordane occurred in sediments associated with urban runoff in the Santa Clara Valley at concentrations of from 3 to 10 $\mu g\ kg^{-1}$ for endosulfan and 13 to 30 $\mu g\ kg^{-1}$ for chlordane (dry weight; Woodward-Clyde, unpublished data).

There is a strong correlation in urban runoff samples between total suspended solids (TSS) and pollutants, including hydrocarbons (Pope *et al.*, 1978; Hoffman *et al.*, 1982), several trace metals (Wilber and Hunter, 1977; Montoya, 1989), and PAHs (Hoffman *et al.*, 1985). Trace metals and hydrocarbons in urban runoff are derived primarily from street dust (Stenstrom *et al.*, 1982). Preliminary data from the urban runoff discharge study in Santa Clara Valley (Woodward-Clyde, unpublished) and data from the City of Sacramento (Montoya, 1987) show that PAHs are concentrated in sediment samples adjacent to storm drains and areas receiving runoff. More data are needed on the speciation of pollutants in urban runoff.

Although pathogenic bacteria are not included in the list of "pollutants of concern" in the Estuary (Table 2), it should be pointed out that local problems have been documented that may be associated with urban runoff. In a San Mateo County study, five sites were chosen for sampling because of their distance from effluent discharges. Shellfish at these sites were examined for coliform contamination. Elevated bacteria counts were observed at four sites; one site had a Most Probable Number (MPN) of 54,000 fecal coliforms per 100 ml^{-1} (Jarvis *et al.*, 1983).

Large storms that produce wet weather overflows of sanitary sewers may cause significant loading of coliform bacteria to receiving waters as well. A

Table 8. Pesticides detected in the Nationwide Urban Runoff Program (USEPA, 1982). All values in $\mu g\ L^{-1}$.

Pesticide	Frequency of Detection (%)	Range of Concentrations
aldrin	6	0.1
α-BHC	20	0.0027-0.1
β-BHC	5	0.1
γ-BHC	11	0.052 -0.1
δ-BHC	6	0.1
chlordane	5	0.1 - 10
DDT	1	0.1
dieldrin	2	0.008 -0.1
α-endosulfan	13	0.1 -0.2
heptachlor	5	0.1
heptachlor-epoxide	1	0.1
isophorone	4	10

monitoring study completed by the East Bay Municipal Utilities District along portions of the East Bay shoreline showed that total coliform concentrations in surface waters increased from less than 10 MPN (100 ml)$^{-1}$ to between 1,000 and 100,000 MPN (100 ml)$^{-1}$ during two storm periods (EBMUD, 1986). After storms, coliforms in shellfish increased by two orders of magnitude, and total coliforms in sediments increased by 2 to 3 orders of magnitude.

Pollutant Loading to the Estuary in Urban Runoff

Estimating the loading of pollutants to the Estuary in urban runoff requires information regarding land use, precipitation, runoff coefficients (the fraction of rainfall that becomes runoff), and pollutant concentrations in runoff (see Gunther *et al.* [1987] for a more detailed discussion). Pollutant concentrations and runoff coefficients, and to a lesser extent land use, are poorly known in the Bay-Delta, making estimates of loads quite uncertain. Utilizing data from a variety of studies, Gunther *et al.* (1987) estimated the load to the Estuary of several pollutants in urban runoff (Table 9). These estimates are presented as ranges, reflecting the uncertainty inherent in the calculations. The ranges in loading estimates were derived by systematically varying the assumptions used in the calculations to create reasonable "high" and "low" estimates. The estimated ranges of Gunther *et al.* (1987) bracket the estimate developed by NOAA (1988), which is also presented in Table 9.

The uncertainty in the estimates of Gunther *et al.* (1987) arises from uncertainty in estimates of pollutant concentration in runoff, inadequate estimates of runoff coefficients, and a lack of information on mode of transport of pollutants in surface runoff (i.e., particulate or dissolved). The estimates used in this report were derived from sources outside the Bay-Delta region. The national data base developed by the NURP demonstrated significant site-specific variation, underscoring the need for local measurements of pollutant concentrations and runoff volumes to calculate these loads more accurately. Such measurements will also allow more detailed analysis of runoff in different drainages of the estuarine catchment.

Runoff coefficients used in this report were also estimated from other areas; such coefficients for particular regions could also be determined empirically in local studies, as these are known to vary within and among drainages.

The particulate nature of many of the pollutants in urban runoff implies that these materials may settle out in streambeds or storm drains prior to reaching the Estuary, possibly reducing the magnitude or changing the timing of pollutant loading. Precipitation patterns, irrigation during dry weather, and the scouring of accumulated pollutants from urban surfaces in early season storms (see below) result in temporal variations in loading that are poorly understood at present.

Table 9. Estimate for the loading of some toxic pollutants to the San Francisco Estuary in urban runoff. All values in tonnes yr^{-1}. After Gunther *et al.*, (1987); NOAA (1988). Dashes indicate where data were not available.

Pollutant	Estimated Load[a]	Estimated Range[b]
arsenic	6	1 - 9
cadmium	2	0.3 - 3
chromium	12	3 - 15
copper	42	7 - 59
lead	179	30 - 250
mercury	0.1	0.03 - 0.15
zinc	189	34 - 268
total hydrocarbons	8,260[c]	1,143 - 11,016[d]
PCBs	—	0.006 - 0.4
PAHs	—	0.5 - 5

[a] NOAA (1988)
[b] Gunther *et al.* (1987)
[c] derived using concentrations of oil and grease
[d] minimum value derived using concentration data for petroleum hydrocarbons, maximum value using oil and grease data (see Gunther *et al.* [1987])

Recent estimates of trace metal mass emissions in urban runoff in the Sacramento Valley show that runoff contributes greater loads of "total" trace elements to receiving waters than effluent discharges in this region (Table 10). Montoya *et al.* (1988) estimated mass emissions from several inputs, making conservative assumptions regarding runoff coefficients and utilizing data on pollutant concentrations measured in runoff from part of the City of Sacramento. For the six metals studied, estimates of runoff loads exceeded loads from NPDES permittees. This was especially true for lead and zinc. Montoya (1987) documented significant variability for runoff coefficients among years for two watersheds in Sacramento. These changes reflect urbanization of the area, but they also demonstrate additional variation of uncertain origin.

The findings of Gunther *et al.* (1987) and Montoya *et al.* (1988) for lead and zinc showed that loading for these metals would exceed that for other metals in urban runoff. Gunther *et al.* (1987) found that, in spite of uncertainty associated with calculated mass emissions, urban runoff clearly contributed greater overall loads of lead to the Estuary than effluent discharges. The declining use of leaded gasoline will probably reduce lead loads in urban runoff. Both studies show clearly that urban runoff is an important contributor of toxic pollutants to the Estuary.

Temporal Variation in Urban Runoff

Regional precipitation patterns influence the loading of pollutants *via* urban runoff to the Estuary. Pollutants accumulate within a drainage both between storms and between rainy seasons, much of the build-up can be flushed out by the first rain. Many urban runoff studies have documented the occurrence of this "first flush" effect, in which pollutant concentrations are elevated during the early part of a runoff event, and in early season storms as compared to storm events later in the season (Pope *et al.*, 1978; Rimer *et al.*, 1978; Hoffman *et al.*, 1985).

The data of Montoya (1989) from eleven storm events in Sacramento document a seasonal "first flush" of trace metals (particularly lead and zinc) (Table 11). Concentrations of pollutants in runoff declined sharply after the first storm event, suggesting that pollutants accumulate during the dry season in Sacramento. Pollutant concentrations were lower later in the season, showing that accumulated pollutants had been scoured from the drainage. A "first flush" is not characteristic of all urban drainages, however, and generalizations must be made cautiously.

Montoya (1989) also found that the highest pollutant concentrations were measured during the first 0.5 inch of rainfall for storm events that occurred early in the season. This was not the case for storms that occurred later in the season. This suggests that the scouring of pollutants from urban surfaces by early season storms influences temporal trends in pollutant loading.

Table 10. Comparison of estimated mass emissions of trace metals by urban runoff and NPDES permittees in the Sacramento Valley (Montoya *et al.*, 1988). Values in tonnes yr^{-1}.

Element	Urban Runoff Mass Emissions	NPDES Discharge Mass Emissions
Cadmium	0.2	0.05
Chromium	3.6	1.8
Copper	8.1	2.7
Lead	12.7	0.5
Nickel	2.7	1.8
Zinc	59.5	15.5

Table 11. Event mean concentrations for a Sacramento storm drain for the 1986-87 wet season (Montoya, 1989). All values in µg L^{-1} except as noted. Cumulative rainfall for the season was 0.73 in during event 1, 0.73-2.6 in during events 2-5, and 2.6-11.0 in during events 6-11.

Pollutant	Storm Event 1	Events 2-5 (standard error)	Events 6-11 (standard error)
Arsenic	11	6.5 (2.3)	1.3 (0.5)
Cadmium	2	1.2 (0.6)	0.3 (0.2)
Chromium	240	18.8 (4.3)	13.2 (4.5)
Copper	158	39.8 (10)	19.3 (3.4)
Lead	274	107 (24.7)	72.8 (12.9)
Nickel	48	22.3 (4.6)	12 (1.9)
Zinc	614	288.3 (57.2)	189.3 (22.2)
"Oil and Grease" (ppm)	ND	6.8 (1.5)	3.7 (1.8)[a]

[a] n = 3
ND - Not detected.

Pope *et al.* (1978) found that the trace elements cadmium, chromium, copper, and nickel exhibited slightly different "first flush" characteristics. These pollutants reached peak concentrations at different times during a storm, suggesting that these metals may have been associated with particles of specific size fractions. The differential susceptibility of the various size-fractions of particles to scouring from surfaces may explain the observed differences in suspension and transport from urban surfaces in response to changing velocities (or volumes) of runoff.

Recent studies in Sacramento showed that urban runoff volumes during the dry season can also be significant. Domestic/commercial landscape irrigation, groundwater infiltration, pumped groundwater discharges, construction projects, and wash-off practices contribute to urban runoff during dry periods (Montoya *et al.*, 1988). Gunther *et al.* (1987) did not consider such dry season flows.

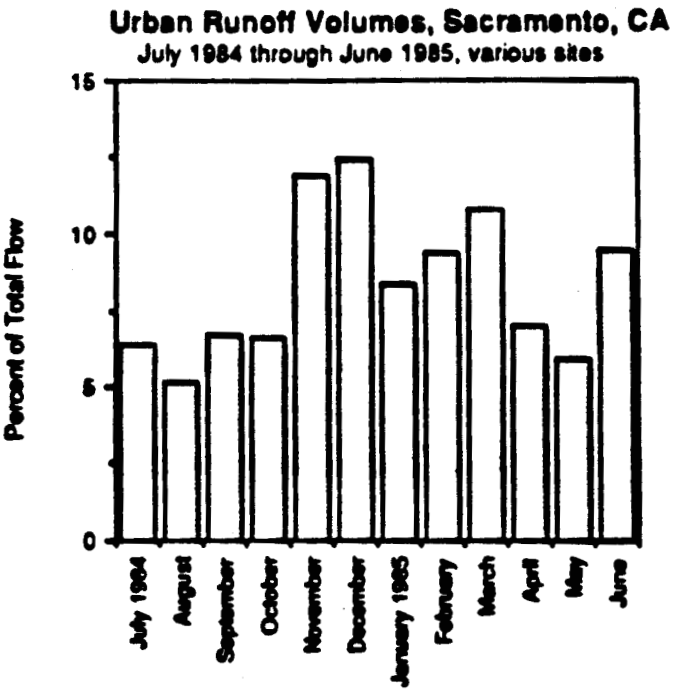
As an example of the importance of dry season flows, Montoya (1987) examined runoff pumped by the City of Sacramento from its underground conveyance system over river levees. These pumping activities provide a convenient metering system for runoff volumes. These pumping records showed that just under half the annual runoff volume discharged from the Sacramento area in 1984-1985 occurred during the May-October dry season (Figure 23) (Montoya, 1987) . The volume of dry season runoff could be a result of water use patterns peculiar to Sacramento, and thus might not be characteristic of other drainages in the Estuary.

All trace elements measured in Sacramento (Montoya *et al.*, 1988) have been measured at lower concentrations in dry season runoff than in wet season runoff (Table 7). In a study by Sacramento County five of eight metals analyzed were found at higher concentrations during the wet season. However, zinc, arsenic, and mercury levels were higher in dry season runoff (SWQCL, unpublished: in Montoya, 1987). This study measured wet season concentrations using grab samples, which are not really appropriate for estimating pollutant loads from storm waters (Hoffman *et al.*, 1983; Gunther *et al.*, 1987).

Preliminary data from Santa Clara Valley also suggest that concentrations of eight trace elements (arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc) were higher in storm waters than in dry season flows. In the case of Santa Clara Valley, the increase in lead loads for the wet season was the most dramatic (Woodward-Clyde, unpublished). Total coliform counts in storm waters from the Santa Clara Valley were anywhere from 2 to 10 times higher than in dry periods, with an exception for an elevated reading during one of six dry-weather samples (Woodward-Clyde, unpublished).

Toxicity bioassays conducted recently by the CVRWQCB have shown that some stormwater runoff from the City of Sacramento is acutely toxic to

Figure 23. Monthly urban runoff volumes (as percent of total urban runoff flow) for a portion of Sacramento, July 1984-June 1985 (Montoya, 1987).



invertebrates at even a 3:1 dilution (i.e., 25% storm water). This suggests that urban runoff from Sacramento may cause toxicity in the Sacramento and lower American rivers (Foe, 1987a). Bioassay results show that undiluted dry season runoff from the City of Sacramento is also acutely toxic to the water flea, *Ceriodaphnia* (Foe, 1987a).

Gaps in Knowledge

Gaps in knowledge regarding urban runoff exist for two reasons. First, there are inadequate data on concentrations of pollutants in runoff samples. The reason that the available data are inadequate is two-fold: 1) Analyses performed on available samples have been performed using techniques with inappropriately high limits of detection; and 2) Analytical protocol has not included adequate quality assurance procedures to guarantee that the data are accurate.

Second, samples taken for analysis of pollutants in urban runoff fail to provide an accurate picture of the temporal distribution of pollutant loads throughout runoff events. At least part of the reason for this gap in knowledge has been a lack of appropriate, common sampling design employed among and between agencies investigating urban runoff pollution. Another part of the reason is that the procedures used for gathering samples during runoff events have not, heretofore, included volume-rated samples, especially during the early portions of a storm event.

Filling these gaps in knowledge would make it possible to improve our overall understanding of the physical and chemical processes that determine pollutant loads in urban runoff. For example, accurate and reliable measurements of pollutant concentrations in runoff samples would enable investigators to make reasonable estimates of the overall load of pollutants entering the system, providing reasonable estimates of the volume of flow are available. Arriving at reasonable estimates of volume of runoff flow, however, can only come as the result of improved measurements of time-dependent flow in runoff drains, and/or better measurements of runoff coefficients.

The most critical data needs are

1) Improved measurement of metals concentrations in runoff water samples (particulate and dissolved);
2) Reliable measurements of organic pollutants in runoff water samples (particulate and dissolved);
3) Accurate estimate of runoff coefficients for a variety of land-use types;
4) Accurate volume-rated sampling of runoff water for chemical analysis; and
5) Accurate estimates of dry-season runoff for the municipalities in the Bay-Delta Estuary.

B. NONURBAN RUNOFF

This section describes current knowledge of the loading of pollutants to the Estuary *via* runoff from nonurban lands. Concern regarding contamination from nonurban runoff has grown due to the detection of agricultural chemicals in water, sediment, and biota of the Bay-Delta; toxic effects in waterfowl from agricultural drainage into the Kesterson National Wildlife Refuge; and acute toxicity demonstrated in bioassays using water from or near agricultural drains. The following discussion reviews data concerning pollutants of concern in nonurban runoff, presents previous estimates of loading, and reviews temporal variability of loads. Important data gaps are also described.

Nonurban runoff refers to runoff from agricultural lands, pasture lands, and natural range and forests. Nonurban runoff includes precipitation runoff from croplands and pastures and irrigation return flows (*via* either surface or subsurface agricultural drains). Pollutants of concern in nonurban runoff are derived from erosion and leaching from soils of trace elements, synthetic organic pollutants (particularly pesticides), and solvents used for pesticide application. Although erosion and sedimentation are natural processes, they are accelerated by activities such as construction, mining, cultivation of steep slopes, and over-grazing of pastures. Surface and subsurface agricultural drains carry nutrients, minerals, metals, and pesticides into adjacent aquatic environments (DWR, 1988). In addition to the legal use of pesticides, illegal dumping can also result in contaminated nonurban runoff entering the Estuary.

Pollutants in Nonurban Runoff

Although the nonurban runoff category includes several land use types (such as agriculture, forest, range, and pasture), actual data on pollutant concentrations in runoff exist only for agricultural drains outside of the Estuary. These data can be used to estimate potential loads of pollutants from agricultural lands within the Estuary drainage. Extensive portions of the Delta (520,000 acres), for example, are devoted to agriculture (DWR, 1987), and hundreds of agricultural drains discharge into waters of the Delta (DWR, 1987).

Montoya *et al.* (1988) measured trace elements in agricultural drainage from several sites in the Sacramento Valley. He detected concentrations lower than those present in urban runoff in Sacramento (Table 12). The major agricultural drains of the San Joaquin Valley, such as Salt and Mud sloughs, are sources of pollutants to the San Joaquin River. A report of the San Joaquin River Basin Technical Committee showed that in 1985, Mud and Salt sloughs contributed 12% of the flow in the San Joaquin River at Vernalis. However, the water flowing from these sloughs contributed 81% of the selenium, 69% of the boron, 46% of the total dissolved solids, and 44% of the molybdenum entering the river (CVRWQCB, 1988).

Table 12. Concentrations ($\mu g\ L^{-1}$) for grab samples of six trace metals in six Sacramento Valley agricultural drains, and estimated loads (tonnes yr^{-1}) to the Valley in agricultural drainage. Number of samples varied from 9 to 38. Coefficient of variation in parentheses. After Montoya *et al.* (1988).

Drain	As	Cd	Cr	Cu	Ni	Zn
RD108	NA	0.2(200)	4.7(51)	7.6(32)	8.7(47)	14(48)
Colusa Basin Drain	<0.5-2.3	0.1(300)	12(64)	9.6(36)	8.6(62)	25(112)
Sacramento Slough	<0.5-3.4	0.1(100)	8.6(45)	8.6(59)	7.9(72)	21(86)
RD1000[a]	NA	0.1(100)	3.1(58)	8.7(126)	3.1(100)	26(158)
Natomas East Main Drain[a]	NA	0.2(100)	6.5(37)	7.6(33)	4.5(100)	34(76)
Toe Drain	NA	0.1(100)	12(34)	11(20)	22(30)	21(19)
Loads	4.5	0.3	17.2	15.9	15.5	40.5

[a] contains some urban runoff and effluent discharge

Pesticides applied to the soil surface can move in the liquid phase either by runoff of water at the soil surface or by leaching through the soil. Pesticides that have been identified in drain waters of the Central Valley include atrazine, simazine, fenthion, dacthal, diazinon, bidrin, phorsulfon, chloropropham, molinate, and thiobencarb (DWR, 1988). The fate of pesticides and other pollutants associated with agricultural practices on Delta islands is much different than in the Central Valley. The lag time of pollutant transport from land to the aquatic environment is much shorter in the Delta. Most of the islands are diked, and the land is below the level of the surrounding waters. Drainage water is pumped directly from these islands to the waters of the Estuary, leaving little time for the degradation of pesticides or the settling of particles that may have pollutants adsorbed to them. The toxic effects of these discharges, however, have not been investigated.

Loading of Pollutants to the Estuary via Nonurban Runoff

Loads of seven trace elements and chlorinated hydrocarbon pesticides to the Estuary *via* nonurban runoff have been estimated for 1982 (Table 13) (NOAA, 1988). Trace metal loads were estimated from predictions of sediment yield (erosion) for different nonurban land uses and an average soil concentration for each metal. Chlorinated pesticide loads were calculated from pesticide usage statistics using a "loss" factor (see Gunther *et al.* [1987] for a more detailed discussion). As 1982 was a year of high precipitation, Gunther *et al.* (1987) estimated that typical nonurban pollutant loads could be an order of magnitude lower than the values presented in Table 13.

It is interesting to note that the source category "other nonurban land," which includes range and pasture land, is predicted to contribute larger loads of trace elements to the Bay and Delta than agricultural land (NOAA, 1988). This is due in part to the fact that there is about four times as much range and pasture land as agricultural land draining directly to the Bay and Delta. It is also due to the fact that the more extreme topography of the range and pasture results in greater erosion.

There are other sources of uncertainty in these estimates. For example, the model used by NOAA (1988) is highly sensitive to uncertainty in the form of inadequate characterization of soil properties and inadequate data on trace metal concentrations in the soils of the estuarine catchment. The model can also be affected by incomplete pesticide usage information, inadequate data regarding irrigation volumes and erosion in Delta farmlands, and variations in runoff with different levels of precipitation and soil moisture (Gunther *et al.*, 1987). For example, while the loss of pesticides has been estimated at 1-5% of that applied (Wauchope, 1978; Green and Kahn, 1987), the actual mass lost is

Table 13. Loads of selected trace metals and chlorinated hydrocarbon pesticides (CHP) to the Bay and Delta for 1982. All values in tonnes yr^{-1}, except as noted. After NOAA (1988). Since precipitation was above average in 1982, these loads could be an order of magnitude higher than average loads (Gunther *et al.*, 1987).

Source Category	As	Cd	Cr	Cu	Pb	Hg	Zn	CHP
Cropland	34	2	269	122	61	0.38	293	0.42
Forest Land	11	0	133	49	33	0.16	135	0
Other Nonurban Land	74	5	1,134	410	265	1.2	1,023	0
Irrigation Return Flows	0	0	2	0	0	2	2	0.08
TOTAL	119	6	1,537	581	358	1.73	1,453	0.5

uncertain due to variations in climate, timing of application, soil characteristics, and agricultural practices. In addition, the degradation of non-chlorinated pesticides in surface runoff occurs rapidly, due to ultraviolet radiation, microbial decomposition, and volatilization, although these processes are significantly reduced when pesticides percolate deeper into the soil (Green and Kahn, 1987). Seiber (1987) points out that there are still too many uncertainties regarding important environmental processes (such as leaching) to predict pesticide losses accurately.

Montoya *et al.* (1988) recently estimated the loading of six trace metals to the Sacramento River *via* agricultural drainage in the Sacramento Valley (Table 12). These estimates were developed using measured concentrations of trace elements in drain waters and volumes of drain flow, a more empirical approach than that of NOAA (1988). The agricultural portion of the loads calculated by NOAA (1988) (see "cropland" in Table 13) is consistently an order of magnitude greater than that of Montoya *et al.* (1988), even though the amount of agricultural land upon which NOAA estimated its loads (8,411 km2; about 2 million acres) is less than twice as much the 1.4-million-acre estimate used by Montoya *et al.* (1988). The high precipitation in 1982 could also have inflated the loads of pollutants estimated by NOAA (1988).

Temporal Variation in Nonurban Runoff

Temporal variation in the loading of pollutants to the Estuary *via* nonurban runoff would be expected due to variation in precipitation and irrigation practices. Loads carried by nonurban runoff are heavily influenced by variation in these factors, as evidenced by studies conducted in the Central Valley.

In an examination of variability in trace metal concentrations during a storm event, Montoya *et al.* (1988) sampled the Colusa Basin Drain over a 4-d period in early January 1987. Concentrations of cadmium, copper, and zinc varied two-to-six fold, relatively small variations compared to those documented for urban runoff. This is probably due in part to the more permeable nature of nonurban surfaces, which would tend to dampen the scouring effects of runoff, possibly eliminating the sharp peaks in pollutant concentrations often noted in urban storm events. For example, the flow in the Colusa Basin Drain increases for 2-4 d after rainfall, reflecting a gradual translation of runoff and soil percolation into drain flow. Some of the agricultural drains in the Central Valley are artificially controlled, which could also influence pollutant concentrations by accelerating the settling of particulates. Analysis of other drains and other seasons is needed, as significant variations in concentrations and flow volumes

would necessitate the use of event mean concentrations to estimate pollutant loads accurately.

During a storm event in February 1986, concentrations of copper, chromium, and nickel in the lower San Joaquin River (17, 72, and 60 $\mu g\ L^{-1}$, respectively) were above the average concentrations for that year (3, 3, and 8 $\mu g\ L^{-1}$, respectively [CVRWQCB, 1988]). These concentrations could be produced by resuspension of river sediments under a more turbulent flow regime.

Irrigation practices also influence pollutant loads from agricultural runoff. In the Sacramento Valley three distinct peak periods for agricultural discharge occur during the year. These are the rainy season, the rice growing season (May-June), and rice field dewatering (August-September). Pesticide application during the growing season results in transport of these chemicals to receiving waters. Foe (1988a,b) found that samples taken from the Colusa Basin, Sacramento Slough, and the Sacramento River caused significant mortality in the test species during rice growing season, but not at other times of the year.

Montoya *et al.* (1988) found higher concentrations of copper in samples from the Colusa Basin Drain and Sacramento Slough during rice growing season, which may be related to the use of copper sulfate as an algicide in rice fields. Chromium concentrations were higher from non-rice season samples, possibly implicating soils as a source for this element (Montoya *et al.*, 1988).

Gaps in Knowledge

We cannot understand the contribution of nonurban runoff to pollutant loading in the Estuary until reasonable estimates are available for all critical parameters. These include:

1) Runoff coefficients;
2) Volumes and patterns of pesticide usage in the drainage basin;
3) Physicochemical fate of pollutants in soils and on particles in water; and
4) Patterns of irrigation employed on crops in the basin.

Data gaps remain regarding pollutant loads to the Estuary from nonurban runoff. These involve both the adequacy of the existing database and the methods utilized to predict the loads of pollutants.

There are few data regarding the concentration of pollutants in nonurban runoff. Recent work by the CVRWQCB has begun to describe the concentrations of trace elements in agricultural drainage in the Central Valley. The San Joaquin Valley Drainage Program has also characterized trace element transport in the waters of the San Joaquin Valley (Shelton and Miller, 1988;

Gilliom *et al.*, 1989). Similar efforts are necessary if temporal and spatial trends in concentrations and loads from agricultural lands within the Estuary are to be estimated. Direct measurement of runoff from other nonurban lands will be more difficult. There is almost no information on concentrations of organic chemicals in nonurban runoff. Data on the use of pesticides in the catchment are still incomplete.

Efforts to predict the loads of pollutants from nonurban runoff using more complex models are also hampered by lack of data, including trace element concentrations in soils, soil moisture measurements, and other parameters used in modeling efforts (see Gunther *et al.* [1987]). Field data will also be necessary to calibrate and verify models, particularly if they are to be used to examine the impacts of various control strategies.

3. Riverine Inputs

Gunther *et al.* (1987) discussed mass transport of toxic pollutants to the Estuary in the major tributaries of the Delta. This section includes a brief review of that discussion and an analysis of data that have been published since 1987 that relate to monitoring efforts on the Sacramento and San Joaquin rivers. Estimation of riverborne loads to the Delta is the most straightforward means of accounting for the transport of pollutants to the Estuary from the many sources situated upstream of the Delta. Estimates of riverine inputs provide a basis for assessing the relative magnitude of loads from other inputs of pollutants in the Estuary.

Pollutant loads transported by the Sacramento River past the City of Sacramento and by the San Joaquin River past the City of Vernalis are considered to have entered the Estuary (see Figure 1). Loads to the Estuary should not be confused with loads to San Francisco Bay. An analysis of pollutant transfer between components of the Estuary (e.g., from the Delta to the Bay), although clearly an important subject, is beyond the scope of this report. These transfers are difficult to quantify because of the complex hydrology and chemistry of the Estuary.

The Sacramento and San Joaquin rivers are the dominant features of the basin that drains into the Estuary. Each of these rivers drains extensive agricultural areas. Long-term average flows recorded by the U.S. Geological Survey (USGS) (Fogelman *et al.*, 1986a,b) are 70 billion L d^{-1} (18.5 billion gallons per day [BGD]) in the Sacramento River (including flows diverted through the Yolo Bypass) and 9 billion L d^{-1} (2.4 BGD) in the San Joaquin. These flows comprise 80% and 11%, respectively, of the total volume entering the Delta. In each of these rivers, flow rates can increase sharply in the wet season during periods of heavy runoff.

Agricultural drainage in the Sacramento Valley may contribute over 30% of the total flow of the Sacramento River during May and June (Cornacchia *et al.*, 1984). Similarly, it is estimated that agricultural drainage contributes more than 20% to the total time-averaged flow in the San Joaquin River (Nichols *et*

al., 1986), and most of the flow during the summer (DWR, 1986a). Many types of pesticides are applied to agricultural lands in the Central Valley. In the San Joaquin Valley alone, nearly 23 million kg of about 500 different pesticides were applied in 1982; this is almost 10% of the total applied annually to major crops in the United States. (Clifton and Gilliom, 1986). Agricultural practices have also influenced trace element movement in the watershed, especially that of selenium (Gilliom *et al.*, 1989). Agricultural drainage water from the western San Joaquin Valley contaminated the Kesterson Wildlife Refuge, where high selenium concentrations were the apparent cause of reproductive impairment in waterbirds (Letey *et al.*, 1986; Ohlendorf *et al.*, 1986a).

Although the Sacramento and San Joaquin rivers probably contribute the largest total loads of many pollutants entering the Estuary, the present understanding of these loads is deficient in many respects (Gunther *et al.*, 1987). Data collected by USGS under the San Joaquin Valley Drainage Program (Shelton and Miller, 1988; Gilliom *et al.*, 1989) and by the California Department of Water Resources (DWR) in a study of selenium cycling in the Estuary (Cutter, 1989a) provide the best basis for calculating riverine pollutant loads. Although some information regarding the transport of pollutants in the Sacramento River has been collected in the San Joaquin Valley Drainage Program, it has not as yet been published, leaving an almost complete lack of data on loads transported to the Estuary by one of the most significant inputs. Selenium is the only pollutant whose transport by the Sacramento River has been sufficiently characterized to allow comparison with loads from other inputs. Other data published to date for the Sacramento River concern only dissolved forms of trace elements, and these have been collected infrequently (Gunther *et al.*, 1987). Aqueous concentrations of trace organics have not been characterized in these two rivers, at least partially due to the difficulty in detecting such compounds in water (see Section IV.A.).

Sampling under the San Joaquin Valley Drainage Program at Vernalis provides detailed information on concentrations, sources, and transport of trace elements (particularly selenium) in the San Joaquin River system (Gilliom *et al.*, 1989). Sampling under this Program at Vernalis provides the most useful data available for assessment of riverine transport of pollutants into the Estuary. Data published to date were generated from an analysis of samples collected twice per month from June 1985 to March 1987 (Shelton and Miller, 1988). Table 14 lists average concentrations and mass loads of six trace elements measured at Vernalis. Ranges of average concentrations reflect the occurrence of several "below detection limit" results for forms of some elements. Data on dissolved concentrations were collected for arsenic, copper, nickel, and zinc. Copper, nickel, and zinc were transported predominantly in particle-associated form; arsenic, on the other hand, was transported primarily in dissolved form. The average load of "total" forms of each element at Vernalis during this period greatly exceeded the loads estimated for all municipal and industrial effluents combined (see Table 6 above).

Massive runoff in early 1986 had a profound effect on the mass transport of pollutants. In general, maximum loading of both total and dissolved forms of

Table 14. Average and maximum concentrations ($\mu g\ L^{-1}$) and mass transport ($kg\ d^{-1}$) of six trace elements measured on the San Joaquin River at Vernalis. Data are for "totals" except where specified as dissolved. Summary based on over 40 samples measured by Shelton and Miller (1988) from June 1985 to March 1987. Flows coinciding with collection of these samples averaged 14 billion $L\ d^{-1}$, with a maximum of 87 billion $L\ d^{-1}$ on 17 March 1986.

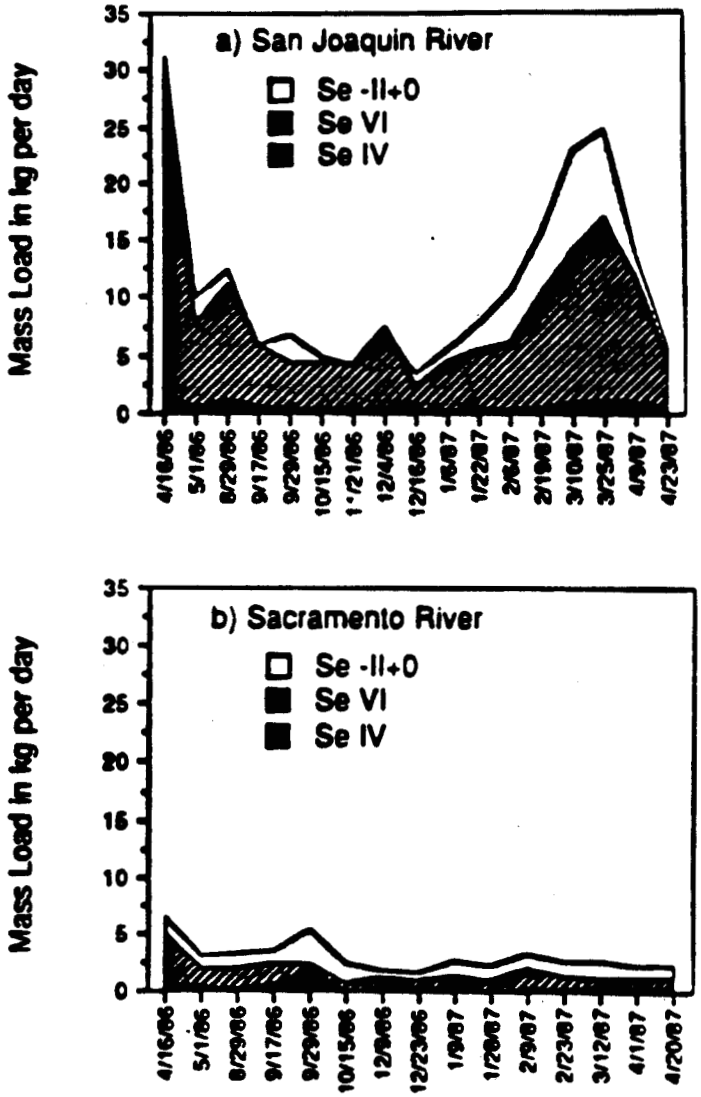
Analyte	Average Concentration ($\mu g\ L^{-1}$)	Average Load ($kg\ d^{-1}$)	Maximum Load and Corresponding Concentration ($kg\ d^{-1}$ [$\mu g\ L^{-1}$])	Date Maximum Load Observed
Total Arsenic	2	33	174 (2)	3/17/86
Dissolved Arsenic	2	21	111 (2)	2/25/86
Total Chromium	10	184	2091 (24)	3/17/86
Total Copper	10	222	4704 (54)	3/17/86
Dissolved Copper	2	48	871 (10)	3/17/76
Total Lead	4.6-6.6	136-153	4355 (50)	3/17/86
Total Nickel	11	141	1651 (85)	2/18/86
Dissolved Nickel	2	29-36	271 (7)	3/3/86
Total Zinc	27-29	449-476	6969 (80)	3/17/86
Dissolved Zinc	8	190-204	4181 (48)	3/17/86

the elements listed in Table 14 occurred on 17 March 1986, when the highest flow rate measured during the entire sampling effort was recorded (87 billion L d^{-1}; 23 BGD). The few exceptions to this generalization were dissolved arsenic and dissolved and total nickel, which exhibited maxima earlier in the same high flow period. Relatively high concentrations of chromium (24 µg L^{-1}), copper (54 µg L^{-1}), lead (50 µg L^{-1}), and zinc (80 µg L^{-1}) coincided with the 17 March peak flow, resulting in the extremely high maximum rates of transport for these elements (see Table 14). Except for arsenic, all the elements listed in Table 14 tend to be associated with particles in aquatic environments; and high flows in the river produced elevated concentrations of suspended sediment and sediment-associated pollutants. Arsenic does not adsorb to particles to the same extent as the other elements, but nevertheless, arsenic transport also peaked on 17 March (even though arsenic concentrations were not particularly elevated on that date). Loading rates of all of these elements were high from mid-February to mid-April 1986, accounting for a significant proportion of the total transport measured that year.

Transport of selenium by the Sacramento and San Joaquin rivers is better understood than that of any other element. As mentioned above, DWR sponsored a study of selenium cycling in the Estuary. Rigorous analytical techniques combined with frequent sampling yielded valuable information on riverine transport of Se (Cutter, 1989a). Selenium in aquatic environments occurs primarily in dissolved form, and total dissolved selenium concentrations measured in the San Joaquin River at Vernalis from April 1986 to April 1987 averaged 1.28 µg L^{-1}. Selenate (Se^{6+}) was 69%, selenite (Se^{4+}) was 25%, and "organic selenide" was 6% of this total. Total selenium concentrations in the Sacramento River at Freeport averaged 0.08 µg L^{-1}. Selenate was 46%, selenite 12%, and organic selenide 42% of the total.

Figure 24 depicts loads calculated using selenium concentration data collected by Cutter (1989a) from April 1986 to April 1987, and the corresponding daily average flows measured by USGS (Hunter *et al.*, 1988a,b; Mullen *et al.*, 1988a,b). Selenium transport in the San Joaquin River showed two peaks (Figure 24a). The first peak occurred on 16 April 1986, when high flows (51 billion L d^{-1};13.5 BGD) carried a large load into the Estuary in spite of a below-average concentration (0.61 µg L^{-1}) on that date. A second peak, occurring in early 1987, resulted from a combination of a high concentration (a maximum of 2.87 µg L^{-1} in March) and moderate flow (9 billion L d^{-1}; 2.4 BGD). Selenate was the predominant form present in the San Joaquin River, comprising an average of 69% of the total during this period. Selenium transport in the Sacramento River was lower and less variable than that in the San Joaquin (Figure 24b). Maximum transport occurred in April 1986, when both flow (57.5 billion L d^{-1}; 15.2 BGD) and concentration (0.11 µg L^{-1}) exceeded means for the 1-yr period. Again, selenate was the predominant form, accounting for an average of 48% of the total selenium present in the Sacramento River. In contrast to the San Joaquin River, however, organic selenides comprised a substantial proportion (40%) of the total.

Figure 24. Trends in selenium transport in (a) the San Joaquin River and (b) the Sacramento River, April 1986 to April 1987. Forms measured include selenate, selenite, and "organic selenides." Data derived from Cutter (1989a), Hunter *et al.* (1988a, b), and Mullen *et al.* (1988a, b).



Although Cutter's work (1989a) provided valuable data on riverine transport of pollutants, the loads estimated for his period of study may not represent loads that might be expected in years with different patterns of precipitation. Flows during Cutter's study were somewhat anomalous, following a major freshet by one month and continuing through a period of very low flow. Furthermore, there was virtually no flow from the Yolo Bypass during the study. The Yolo Bypass is thought to have a major influence on the seasonal and annual loading of dissolved and particulate substances to the Estuary (Schemel and Hager, 1986).

Selenium is the only pollutant selected for consideration in this report whose transport by both the Sacramento and San Joaquin rivers has been assessed in a comprehensive manner. Data on concentrations of other pollutants in the Sacramento River have been collected infrequently, and are not directly comparable to concentrations measured in most other inputs to the Estuary (i.e., only dissolved forms have been analyzed). This lack of information on the Estuary's principal source of freshwater leaves a significant gap in our understanding of loads to the Estuary as a whole.

Pollutant loads carried by the San Joaquin River have been studied in a relatively thorough fashion. It is evident from this monitoring that loads carried by the rivers are large compared to those from other inputs. These data also suggest that rates of transport may fluctuate considerably during the course of a year, driven by hydrologic variation in the basin. The degree and pattern of fluctuation in transport rates varies among pollutants, depending on their speciation.

4. Dredging and Dredged Material Disposal

Considerable attention has recently been focused upon pollutant mobilization due to dredging and dredged material disposal in the Bay-Delta. The principal concerns are that dredging activities may serve to redistribute pollutants that have been buried or are otherwise sequestered in sediments. Under certain conditions, these pollutants may become bioavailable and may possibly exert toxic effects upon estuarine biota. The extent to which this occurs in the Estuary has been subject to a great deal of speculation, but little study to date.

A recent review of pollutant concentrations in the sediments of San Francisco Bay (Long *et al.*, 1988), and other more site-specific investigations (e.g., Spies *et al.*, 1987a; USCOE, 1988; Word *et al.*, 1988; Rice *et al.*, in press) suggest that ports and harbors surrounded by urban and industrial activity are likely sites of sediment contamination. In particular, concern has been expressed about the elevated concentrations of pollutants in the sediments of localities subject to dredging activity, such as Mare Island Strait, Richmond Harbor, and Oakland Harbor. The various factors involved in assessing such dredged material contamination and, more importantly, the remobilization of pollutants from dredged material disposed in the San Francisco Bay and Delta

have been recently reviewed in the *Status and Trends Report on Dredging and Waterway Modification* (Gunther *et al.*, 1990).

The reports of Gunther *et al.* (1987, 1990) highlight the fact that the physicochemical state of pollutants in sediments varies, and the degree to which pollutants are associated with sediment particles will determine their susceptibility to mobilization. Current techniques are unable to provide a definitive picture of the physicochemical state of trace metals associated with sediments. This, combined with the fact that the form of metals may vary considerably among individual locations, has limited our understanding of the relative contribution of dredging and dredged material disposal to the overall distribution of bioavailable pollutants in the Estuary. Serne and Mercer (1975) and Eaton (1975b) conclude that the bulk of the trace metals in sediments of the Estuary (with the exception of cadmium) are lattice-bound and, hence, relatively unavailable for exchange or release. It is possible that these conclusions are a function of the sampling methods of these investigators. Thompson *et al.* (1980) demonstrate that sampling methods that produce anoxic sediments, such as long periods of storage, or sampling of subsurface sediments, can increase the concentrations of metals found in the sediments as sulfides. Despite the use of careful fractionation techniques, sulfides may be counted in the residual fraction, thereby inflating the estimate of residual metal concentrations (S.N. Luoma, USGS, personal communication).

Organic pollutants, on the other hand, are usually found in association with dissolved and particulate organic carbon in sediments (Adams, 1988). There are few local data to describe the extent of their remobilization and potential bioavailability.

The behavior of sediment-associated pollutants following their disposal is complex and influenced by temperature, oxidation/reduction potential (Eh), pH, salinity, biological transformation, and the disposal method used (see Gunther *et al.*, 1990). Serne and Mercer (1975) and Brannon *et al.* (1978) carried out studies showing that the aquatic chemistry of pollutants, rather than bulk sediment pollutant concentrations, is the most important factor determining remobilization. These studies highlight the need to understand the kinetics of pollutant release when making an assessment of potential bioavailability of pollutants from disposed materials.

The ultimate fate of sediment-associated pollutants in the San Francisco Bay and Delta is linked with the dynamic nature of the ecosystem. Suspended sediments are transported throughout the Estuary *via* complex currents created by tides, wind, and river inflow; erosion and deposition of the order of one meter in depth may often occur over several months (Gunther *et al.*, 1990). Under such conditions it is not surprising that the ultimate fate of disposed dredged material (and its associated pollutant load) in such a system is poorly understood. Investigations involving both numerical and physical modeling have provided some insights (Gunther *et al.*, 1990), but are presently limited by the lack of field verification. Only one tracer study has been undertaken in the Estuary (USCOE, 1976). That study demonstrated the apparent role of winds in

moving sediments in shallow regions, and the possibility of the return of disposed dredged material to navigation channels. There is clearly an urgent need for further studies to elucidate the fate of sediment-bound pollutants in the Bay.

Despite the considerable problems involved in the accurate quantification of pollutant loads derived from dredging activities and the disposal of dredged material in the Bay, some calculations have been made. It should be realized that such estimates are difficult to compare with those from other pollutant inputs. This is because "double counting" may have been involved: i.e., the pollutants in dredged sediments may have originally been derived from other quantified inputs, either recently or in the past. In their assessment of the loading of toxic pollutants to the San Francisco Bay and Delta, Gunther *et al.* (1987) assumed that between 1 and 10% of the pollutants transported in dredged materials may be remobilized in the Estuary (Table 15). Based upon this premise, they showed that dredging and disposal of sediments contributed a relatively minor load of trace metals, PAHs, and PCBs to the Estuary. However, these authors advised that such estimates should be treated with extreme caution, as the available data did not, in many instances, allow accurate quantification of loadings from other inputs, such as riverine loads of pollutants sorbed to suspended particles.

Segar (1988) has recently suggested that the loading estimates produced by Gunther *et al.* (1987) are too low. His estimates (Table 15) utilize the same average concentration of pollutants as those employed by Gunther *et al.* (1987) and were derived using the following assumptions:

1) losses of dredged material and its associated pollutants by burial at the Alcatraz site (the principal disposal site in San Francisco Bay) are 20% of the total amounts dredged and disposed in any one year;
2) of the remaining 80% of the pollutants not lost through mounding at Alcatraz, only a portion of each of the trace metals is sorbed, while the rest is residual in nature. Data for the percentages of metals present in the residual phase in sediments were abstracted from Eaton (1979b) and Serne and Mercer (1975), and were as follows: cadmium, 5%; copper, 60%; lead, 50%; mercury, 80%; nickel, 85%; and zinc, 60%;
3) all of the PCBs and PAHs are present in sorbed forms; and
4) all of the pollutants not present in the mounded material at Alcatraz or in the residual phase may be both remobilized and bioavailable.

Gunther *et al.* (1990) consider such calculations to be highly conservative, providing an estimate of the maximum potential load. They make the point that a body of information exists that suggests low rates of remobilization and/or bioavailability of pollutants from dredged material (see Anderlini *et al.*, 1975a,b; O'Connor, 1989).

It is clear that there is insufficient information to quantify accurately pollutant loadings derived from dredging and disposal of dredged material. All estimates produced so far should be treated with great caution. In addition, Gunther *et al.* (1990) note that pollutant loadings due to dredging activities may

Table 15. Estimated inputs (tonnes yr^{-1}) of selected pollutants to the San Francisco Estuary due to disposal of dredged material. Bioavailable dredged material loadings represent the potentially readily bioavailable pollutants calculated by adjusting total loading estimates from Gunther *et al.* (1987), assuming a) 20% of the pollutant is buried in accumulated sediments at the Alcatraz dumpsite; and b) Residual phase pollutants are not potentially bioavailable. Residual phase estimated as 60% of total copper, 80% of total mercury, 5% of cadmium, 50% of lead, 60% of zinc, 85% of nickel, and 0% of PCBs and PAHs (Segar, 1988).

Pollutant	Dredged Material (Gunther *et al.*, 1987)	Dredged Material (Segar, 1988)	Bioavailable Dredged Material (Segar, 1988)
Copper	1 - 10	100	32
Mercury	0.01 - 0.1	1	0.16
Cadmium	0.02 - 0.2	2	1.5
Lead	1 - 10	100	40
Zinc	3 - 30	300	96
Nickel	2 - 20	200	24
PCBs	0.00067 - 0.0067	0.067	0.054
PAHs	0.05 - 0.47	4.7	3.8

differ considerably depending upon the type of dredging performed. For instance, maintenance dredging can be assumed to involve sediments that have been recently deposited, thus reflecting the general background contamination within the Estuary (unless an unusual contamination incident has occurred). On the other hand, new work dredging removes deeper sediments that have often been buried for prolonged periods. New work may re-introduce to the Estuary pollutants that otherwise would have remained unavailable for uptake by living organisms. On the other hand, not all sediments cut by new work dredging operations contain historic deposits of pollutants, and it is difficult to estimate whether any particular new dredging product will re-introduce pollutants to the Estuary.

Finally, any mobilization of pollutants by dredging of recently deposited sediments must be considered in the context of natural resuspension of sediments in the Estuary. During 1986-1987, aquatic disposal of 5.9 million cubic yards (yd^3) of sediment occurred in the Bay (Gunther *et al.*, 1990), whereas natural resuspension of sediments is estimated to be 120 to 160 million $yd^3\ yr^{-1}$ (USCOE, 1976, 1988).

Although harbors and shipping channels tend to be more contaminated than other portions of the Estuary, most sediments in the system exhibit rather high concentrations of pollutants. Natural resuspension, therefore, may be considered a "load" of pollutants, similar (although less concentrated) to loads estimated for dredged material disposal. This factor is not accounted for in the estimates of Segar (1988) or Gunther *et al.* (1990).

5. Additional Inputs

Other inputs of pollutants to the Estuary include atmospheric deposition, marine vessel discharges, accidental spills, and seepage from waste disposal sites. Pollutants from these inputs may impact the Estuary locally. This section will outline issues relating to these inputs.

A. ATMOSPHERIC DEPOSITION

Atmospheric deposition is considered to contribute minor loads of most toxic pollutants to the San Francisco Estuary (Gunther *et al.*, 1987). However, few definitive data exist on this subject. Toxic substances enter the air through aerosol formation (often *via* combustion), evaporation, suspension of dust, and volatilization from water to the air. The following discussion concerns loads directly deposited into waters of the Estuary; materials deposited on land enter the Estuary in runoff, and are accounted for in the preceding discussions of urban and nonurban runoff.

Very few data exist on the atmospheric deposition of toxic pollutants in the Bay-Delta region. Gunther *et al.* (1987) employed deposition rates measured in other parts of the country to estimate loads to the Estuary; the uncertainty of these estimates cannot be overemphasized. Ranges were presented to reflect some of this uncertainty. They estimated that atmospheric

deposition contributes 0.14-0.35 tonnes yr^{-1} of cadmium, 1.9-3.1 tonnes yr^{-1} of copper, and 6-21 tonnes yr^{-1} of lead to the Estuary. Atmospheric deposition appears to contribute potentially significant loads of PAHs (0.8-4.8 tonnes yr^{-1}) and total hydrocarbons (2.1-45 tonnes yr^{-1}).

Atmospheric deposition of PCBs to the Estuary was estimated to be between 0.12 and 0.87 tonnes yr^{-1} (Gunther *et al.*, 1987). This estimate (and that for PAHs) was based on flux rates calculated for the Great Lakes region (Eisenreich *et al.*, 1981), where atmospheric deposition is thought to contribute as much as 90% of the total input of PCBs to Lakes Superior and Michigan (see Gunther *et al.* [1987]. Ritts (unpublished) estimated loading to the Estuary of PCBs from atmospheric deposition based on flux rates measured in Cloverdale, California, a semi-rural town in the Bay drainage area (average atmospheric PCB concentrations of 40 ng m^{-3}, maximum 157 ng m^{-3}) and urban areas in the Midwest (averaging 7 ng m^{-3} in the atmosphere). The regional average rate of deposition calculated by Ritts was 0.024 tonnes yr^{-1}, the estimate ranging from 0.003 to 1.860 tonnes yr^{-1}.

Estimates of pollutant loads to the Estuary from atmospheric deposition could be improved by measurement of local flux rates for both wet and dry deposition. Measurement of dry deposition (both gaseous diffusion and particle deposition) is particularly challenging, however, making quantification of this input particularly difficult.

B. MARINE VESSEL DISCHARGES

Sewage and gray water (waste water from kitchen and bathing uses) discharged from vessels can have localized effects on water quality and public health in marinas and harbors with minimal water flushing. These wastes can be sources of coliform bacteria, toxic soap residues, biochemical oxygen demanding substances, suspended solids, oil and grease, and nutrients (BCDC, 1987).

Studies of vessel wastes in San Francisco Bay indicate that discharges from houseboats and other live-aboard vessels have caused concern in several portions of the Estuary, including Richardson Bay, Alviso Slough, Redwood Creek, and the Delta (BCDC, 1984, 1987; SFBRWQCB, 1986). Many houseboats are now required to connect to land-based sewage treatment facilities, but other live-aboard vessels are not connected to such systems, and are thought to lack proper sanitation devices (BCDC, 1984). Untreated sewage and gray water from these vessels is discharged directly into the Estuary. Concern over the impacts of vessel wastes in Richardson Bay led the SFBRWQCB to petition the USEPA to designate Richardson Bay as a "vessel sewage no discharge area," with vessel waste discharges regulated by the Regional Board and local governments (BCDC, 1984). Subsequently, residents of live-aboard vessels constructed a large floating holding tank with a pump-out mechanism to collect their wastes. The feasibility and reliability of this system is currently being reviewed.

Pollutant loads from vessel wastes have not been estimated, but probably contribute relatively insignificant amounts of toxic pollutants to waters of the Estuary. More studies would be required to gain a better understanding of the contribution of vessel wastes to contamination of the Estuary.

C. ACCIDENTAL SPILLS

Accidental spills can contribute significant loads of pollutants to the Estuary. Spills of petroleum hydrocarbons occur frequently and are of particular concern. Such spills can cause direct toxicity to fish and wildlife, disrupt food chains, destroy aquatic habitat, affect public health, and reduce the aesthetic appeal of the Estuary.

Most spills are small and unpredictable, resulting from damaged ships, operator errors, handling accidents at terminals, and accidents involving materials carried on shoreline highways. Two major petroleum spills have occurred in the last 20 yr. The collision of two oil tankers near the Golden Gate Bridge in 1971 caused a release of approximately 3.2 million L (845,000 gal) of fuel oil, which fouled shorelines and impacted wildlife. On 23 April 1988 approximately 1.4 million L (370,000 gal) of crude oil was accidentally released from an above-ground holding tank at the Shell Oil refinery in Martinez. The oil flowed into an adjacent marsh, and then into the Carquinez Strait and Suisun Bay. Fifty miles of adjacent shoreline were eventually affected. Oil was spread as far east as Ryer and Roe Islands, and as far west as Pt. San Pablo downstream of the Carquinez Strait. An unusually high clean-up rate of 80-90% was reached in large part because of oil containment in marshlands (SFBRWQCB, 1988b; Lt. Perry, Marine Environmental Response Division of the Marine Safety Office, U.S. Coast Guard, personal communication).

The U.S. Coast Guard (USCG) is the primary source of data on spills in the Estuary. Gunther *et al.* (1987) reviewed USCG data collected from 1984 to 1986. Although a lack of quality assurance detracts from the USCG data, they were used as a basis for generalized estimates of pollutant loads from spills. An average of 117,000 L yr^{-1} (about 31,000 gal yr^{-1} or about 94 tonnes yr^{-1}) of petroleum hydrocarbons was released to the Estuary from spills during 1984-1986. Long-term trends in pollutant loads from spills are heavily influenced by infrequent, catastrophic events such as the Shell Oil spill of 1988.

D. LEAKAGE FROM WASTE DISPOSAL SITES

Within the past decade, many cases of groundwater and surface water contamination due to leakage from both hazardous waste and municipal waste disposal sites have been documented locally and nationally. Efforts have been made by Federal, State, and local governments to determine the extent of this form of pollution. It is generally recognized that all older land disposal sites leak. Sites constructed without leak-prevention measures leak worst (SWRCB, 1989). Sites in close proximity to the Estuary or its tributaries may contribute loads of toxic pollutants.

The two main threats to water quality are leakage and gas migration. Landfill compaction may redistribute leachate, gas pressure changes within fill areas may affect leachate levels, and gas condensate collection drains may add to leachate levels. Both gas and leachate can contain decomposition products, whereas hazardous household wastes tend to be found only in leachate.

In this discussion, the term "waste sites" is used in a generic sense to refer to former municipal solid waste disposal sites (Class III landfills), hazardous waste disposal sites, or industrial waste disposal sites. There are thousands of waste sites throughout the catchment of the Estuary. Nearly 2,000 of these sites have been identified in the immediate Bay Area, over 90% of which are sites where hazardous materials have been deposited (SWRCB, 1988c; CWMB, 1989). Although simple characterization of most of the waste sites has been done, water contamination has only been monitored on solid waste landfills and particularly noxious hazardous waste sites.

There are about 21 active solid waste landfills in the Bay area, some of which pose threats to surface water due to their locations (ABAG, 1985). However, there are dozens of closed waste sites, many of which can no longer be identified. At least 175 landfill sites have not been evaluated for potential hazardous waste migration into the Bay and Delta.

Landfill sites (Class III disposal sites) may contain less toxic material than hazardous material disposal sites. Until recently, management of these sites has been less rigorous. As a result, more leakage has been documented from landfills throughout the country than for hazardous waste disposal sites due to the strict controls that now exist on hazardous material transportation and disposal. California Regional Water Quality Control Boards have conducted Solid Waste Assessment Tests on some landfills. A general picture can be discerned from observations of similar conditions among several landfill sites.

Most municipal landfills have received small amounts of hazardous household products. These generally include paints, insecticides, household cleaners, solvents, thinners, and automotive products. Some landfills have accepted wastes from commercial waste haulers and self-haul vehicles, which could also add to the hazardous wastes deposited. It is estimated, however, that 1% or less of the wastes deposited in these landfills are hazardous (ABAG, 1989).

Some toxic organics (including trichloroethene, vinyl chloride, acetone, and methyl ethyl ketone) are commonly found in municipal landfills. Deterioration of discarded metal objects and disposal of sewage sludge can contribute arsenic, lead, copper, chromium, and nickel to landfills. Lead is also contained in older paints and leaded gasoline (CH2M Hill, 1988). Factors such as pH, oxidation-reduction reactions, and clay sorption affect the mobility of metals once they enter soils near landfills.

Volatile organic compounds (VOCs) are present at all sites, but not always at concentrations of concern. VOCs are components of refined petroleum products and are partially water soluble. They can be found in motor fuels, paint thinner, and solvents disposed of in landfills (CH2M Hill, 1988). Concentrations in leachate can be an order of magnitude greater than levels found in groundwater (EMCON Associates, 1987a; Tomko, 1987; CH2M Hill, 1988; WMNA, 1989). Presence of hydrocarbons in leachate has been verified at several closed municipal landfill sites around the Bay (Tomko, 1987; CH2M Hill, 1988).

At many closed landfills in the Bay Area, settling and subsidence within waste cells have created "outward gradients" such that ground water and/or leachate is not contained within the confining structure and has mixed with salt water. Water quality indicators at many Bay landfill sites confirm that salt water mixing has indeed occurred (e.g., Mountain View, Sunnyvale, and Albany) (EMCON, 1987b; CH2M Hill, 1988; Nichols/Berman, 1988).

6. Summary of Data on Pollutant Inputs

Table 16 summarizes data presented in preceding sections on pollutant loads contributed by major categories of inputs. Our present understanding of pollutant loads is developing rapidly; however, the fact that only a few of the pollutants of concern from Table 2 can be presented in Table 16 demonstrates that much work remains to be carried out. Pollutant inputs in Table 16 are listed in order of the certainty with which their loads can be estimated. Load estimates for municipal and industrial effluents and the major tributaries are based on repeated measurements of pollutant concentrations as these inputs enter the Estuary. Load estimates for the other major inputs are based on predictive models; the predicted pollutant loads are less precise than the estimates based on actual measurements.

Definition of pollutant loads from major inputs represents only the first step toward understanding the biological significance of these loads. The potential contribution of each input to toxic effects in resident biota depends on many other parameters (Section V, below). For this reason, direct comparisons of the loads estimated for the various inputs should be made with caution.

Table 16 shows that very few data are available on the speciation of loads to the Estuary. Municipal and industrial discharges are likely to contribute primarily dissolved forms of trace elements. Riverine loads also consist of varying proportions of dissolved and particle-associated forms, depending on the pollutant under consideration. Urban and nonurban runoff, in contrast, consist primarily of particle-associated forms. Differences in the speciation of loads from different inputs may result in varying contributions to the pool of bioavailable pollutants in the Estuary. Changes in speciation and sorption/desorption reactions that occur when pollutants enter the estuarine environment will make prediction of bioavailability based on speciation in waste streams difficult.

Table 16. Summary of load estimates (tonnes yr^{-1}) for major sources of contaminants. Data for each source vary in precision and temporal coverage, as discussed earlier in the text. Estimated loads for dissolved forms are in italics.

Pollutant	Form	Municipal and Industrial Effluent	San Joaquin River	Sacramento River	Urban Runoff	Total Nonurban Runoff	Atmospheric Deposition	Dredged Material	Spills
Arsenic	Total	1.5 - 5.5	12		1.0-9.0	10-120			
	Dissolved		*7.7*						
Cadmium	Total	1.8 - 4.0			0.3-3.0	0.52-6.0	0.14-0.35	0.02 - 0.2	
Chromium	Total	12 - 13	66		3.0-15	130-1500			
Copper	Total	19 - 30	80		7.0-59	51-580	1.9-3.1	1.0 - 10	
	Dissolved		*17*						
Lead	Total	11 - 16	51 - 55		30-250	31-360	6.0-21	1.0- 10	
Mercury	Total	0.2 - 0.7			0.026-0.15	0.15-1.7		0.01 - 0.1	
Nickel	Total	19 - 27	51					2.0- 20	
	Dissolved		*11 - 13*						
Selenium	*Se VI*	*0.5*	*3.0*	*0.5*					
	Se IV	*1.2*	*0.3*	*0.1*					
	Se -II+0	*0.4*	*0.9*	*0.5*					
Silver	Total	2.7 - 7.2							
Zinc	Total	77 - 80	164 - 175		34-268	130-1450		3.0- 30	
	Dissolved		*69 - 73*						
PCBs					0.006-0.40			0.00067 - 0.0067	
PAHs					0.50-5.0		0.8-4.8	0.05 - 0.47	
Total Hydrocarbons		(a)			1100-11000		2.1-45		94

(a) Although "oil and grease" data are collected by point sources, they were not compiled for this report. It is likely, however, that oil and grease loads from point sources are comparable to Total Hydrocarbon loads from urban runoff.

Seasonal variation also influences the ultimate biological significance of loads from each input. As described above, each major input is influenced to some degree by seasonal and annual variation in precipitation. Loads from nonurban runoff and riverine inputs occur predominantly during rainy periods. Surprisingly, urban runoff in Sacramento is less of a seasonal phenomenon than might be expected due to irrigation and other urban water uses during the dry season, and this may be true of other parts of the Estuary as well. Loads appear to be higher during the wet season due to higher pollutant concentrations. Available data also suggest that early season storms carry larger loads than storms later in the season. Wet season pulses of mass transport from the Sacramento and San Joaquin rivers, nonurban runoff, and urban runoff may result in elevated rates of accumulation on a local or regional basis during these periods. The characteristics of individual watersheds will vary, however, and generalizations from particular studies must be made cautiously.

The concentrations of pollutants in each input may also determine their potential for contributing to toxic effects. Mass loads of pollutants carried by the rivers, for example, result from relatively small concentrations of pollutants present in large volumes of water. These loads may be dilute enough to be tolerated by sensitive organisms. The likelihood of toxic effects of wet season riverine loads may also be lowered by the presence of high concentrations of particulates, which can sequester pollutants, and by lower biological activity during the winter. High concentrations sometimes found in effluent discharges or urban runoff, on the other hand, may exceed levels that can be tolerated by organisms in the proximity of these inputs.

C. FATE OF POLLUTANTS IN THE ESTUARY

1. Introduction

The fate of pollutants in the San Francisco Estuary is governed by partitioning, transformation, and transport processes in water, sediments, and biota. An understanding of these processes, and their interactions, is necessary in order to understand distribution and transport of pollutants, and to determine the exposure of biota to pollutants in different parts of the Estuary.

The fate of pollutants in aquatic systems may be viewed three ways. First, fate can be a description of the final disposition of a pollutant in an ecosystem. For example, the disposition of a molecule of the element cadmium in an aquatic system is eventual incorporation into sedimentary deposits, most likely in the form of a hydroxide or some other insoluble precipitate. The final disposition of a molecule of phenol, a highly volatile and reactive compound, is most likely complete degradation to CO_2 and water, although that degradation may occur in either the liquid phase (hydrolysis) or in the atmosphere (photolysis or hydrolysis). The ultimate disposition of pollutants provides information on the eventual steady state of the system. It does not show the effects that pollutants may exert as they move among and between environmental compartments.

A second way to view environmental fate is the mathematical description of pollutant partitioning among compartments (e.g., water, suspended sediments, deposited sediments, or biota). This approach is referred to as "fate modeling," and describes the proportional distribution of pollutants in ecosystems rather than their ultimate fate. Fate models may be used to predict the distribution of pollutants among the compartments of the system, including the biota. To be successful, fate modeling requires a description of the environment in question: the characteristics of the water and water masses; particle distribution, organic carbon content, and mineral composition of suspended and deposited sediments; chemical and physical interactions at the air-water interface and at the sediment-water interface; bioaccumulation factors; and rates of pollutant metabolism and elimination by biota.

A third view of environmental fate is a description of potential pathways for pollutant movement and distribution in a given ecosystem. Like fate modeling, the descriptive approach, which we shall refer to as "fate assessment," provides information of the distribution of a pollutant among environmental compartments. This information is empirically derived and does not have the predictive capacity of a fate model.

The fate of pollutants in the San Francisco Estuary has been inferred from laboratory data rather than from field studies. There are two reasons for this. First, as shown in Section IV.A., existing data provide a generally inadequate representation of the distribution or abundance of pollutants in the water, sediments, or biota of the Estuary. Many of the available data are from

separate studies conducted at different times utilizing a variety of methods; comparison of data among studies is problematical.

Second, pollutants enter the Estuary in different chemical forms, change forms in the ecosystem, and move throughout the system in response to forces of tides, freshwater inflows, and winds that are either poorly understood or have not been studied in great detail.

This section will describe the processes of partitioning, transport, and transformation of pollutants in estuarine and aquatic systems. Data from the Estuary will be highlighted, and gaps in our understanding of pollutant fates in the Estuary will be described. The goal of this discussion is not an exhaustive review of these topics; it is to describe the nature of these processes and how they may be used to manage pollutants in the Estuary.

Developing a pollutant fate assessment in the Estuary requires information from several sources:

1) Knowledge of the physical chemistry of the individual pollutants, including solubility in fresh, saline, and brackish waters; vapor pressures; chemical speciation in the system; ion exchange potential; and partitioning to mineral solids and organic solids;
2) Knowledge of the dynamics of the system as related to tidally-driven and wind-driven currents, sediment deposition and resuspension, and seasonal differences in flow and circulation; and
3) Knowledge of the relationships among pollutant concentrations in water and on suspended and deposited sediments and the potential for pollutant accumulation and metabolism among the biota of the system.

Some of these data are known for the San Francisco Estuary; however, our understanding of the relative importance of different ecological factors to the fate of pollutants in the Estuary is still being developed. More research on the parameters that determine environmental fate of pollutants in the Estuary is needed.The discussion that follows highlights the major parameters that must be understood for a fate assessment of pollutants in the Estuary.

2. Partitioning

The fact that pollutants can be undetected in one compartment of an ecosystem while reaching high concentrations in another shows that pollutants can move, or "partition" from one phase or compartment to another. Partitioning is driven primarily by the physicochemical characteristics of a given substance and by biological processes. The partitioning of compounds among compartments in an ecosystem is a continuous process and may be highly responsive to changes in environmental conditions (NRC, 1981).

A. THE INFLUENCE OF POLLUTANT CHARACTERISTICS

The physical characteristics of a chemical determine its behavior in the environment. Environmental parameters, in turn, affect the rate and direction of transport of a chemical in the environment. The physical characteristics that most influence environmental fate are vapor pressure, water solubility, and octanol/water partition coefficient (K_{ow}). K_{ow} is a measure of the hydrophobicity of a substance, as determined by its relative solubility in octanol and water. Environmental variables that influence partitioning include temperature, salinity, organic carbon content of sediments and particles, particle (or grain) size, hydrodynamics, and climate (NRC, 1981).

A key partitioning process is the sorption of pollutants to particles. Sorption can influence the bioavailability, transport, and transformation of substances. Sorption reactions are often fully reversible, and the transport of a sorbed pollutant to a location with different environmental conditions might alter its physicochemical status, influencing its bioavailability or mobility in the Estuary. Much of the discussion of pollutant fate that follows relates to adsorption and desorption of pollutants on particulate matter in the Estuary.

Research shows that concentrations of organochlorines and aromatic hydrocarbons are extremely low in estuarine waters. This is expected based upon the low water solubility of these compounds and their tendency to associate with (adsorb to) suspended and deposited sediments (Phillips, 1980). In the San Francisco Estuary, PAHs are essentially non-detectable in the water column; however, concentrations may range from below detection limits up to 81 $\mu g\ g^{-1}$ (dry weight) in sediments (Long *et al.*, 1988). In the southern reach of the Elizabeth River, Virginia, PAHs may reach concentrations of 170 $\mu g\ g^{-1}$ (dry weight), but are barely detectable in the dissolved state in the water column (Bieri *et al.*, 1982).

Evaluation of the fate of monocyclic aromatic hydrocarbons (MAHs) in the Estuary (deVlaming, 1988) showed the importance of solubility and volatility in environmental fate. MAHs have higher water solubilities than PAHs; therefore, one might expect them to appear in higher concentrations in the water column. However, deVlaming (1988) found low concentrations of these substances in the water column, due primarily to the fact that MAHs have high vapor pressures and tend to partition out of the water to the atmosphere.

Sorption of organic pollutants to particles is related to transport of these compounds. (In this context, particles include dead or living phytoplankton and other microorganisms which sorb organic pollutants in a manner similar to that of organically enriched sediments.) Pollutants also adsorb to non-settleable particles ("microparticles"), such as humic and fulvic acids and colloidal aggregates. In San Francisco Bay, particles and their associated pollutants are repeatedly suspended, transported, and deposited as part of the natural cycle of sediment transport, and as part of the process of dredging and disposal of dredged material. Adsorption of organic pollutants, particularly those neutral,

lipophilic forms such as the PCBs and chlorinated pesticides, to the suspended and deposited sediments makes it possible to predict, from partitioning theory, the distribution and transport of these compounds in the Estuary. Such predictions require information on physical and chemical characteristics of sedimentary particles, circulation patterns, sediment particle size, sediment organic carbon content, and the K_{ow} of the compound in question.

Understanding the factors that influence the partitioning of organic species among different particulate phases is vital for improving our predictive capability regarding the fate of sorbed pollutants, as the transport of settleable and non-settleable particles will be very different.

The solubility of many trace metals is generally higher than for organic pollutants, and a larger portion of trace metals in the Estuary will partition to the water column. Trace metals occur in solution both as simple and complex ions; however, even the most contaminated coastal waters are generally undersaturated with respect to most trace elements. This is because trace metals will also partition to suspended particles, including settleable and non-settleable fractions. Characteristics of individual metals and environmental conditions govern the partitioning (Phillips, 1980).

Gunther *et al.* (1990) reviewed the physicochemical status of trace metals in sediments in the San Francisco Estuary. They concluded that pollutants in sediments should be analyzed using sequential leaching techniques that define pollutant forms empirically. Single extractants may remove more than one form of pollutant (Gambrell *et al.*, 1976; Luoma and Davis, 1983), and these measurements may vary among investigators. These forms include organically bound, sorbed to ion exchange sites, inorganic precipitates such as sulfides and carbonates, and "residual" metals bound in the crystalline lattice of clay minerals.

The precise nature of the sorption of trace metals can greatly influence their fate. Thus, silver associated with manganese oxides will partition into deposit-feeding clams 100 times more rapidly than if bound to iron oxides or marsh grass detritus (Luoma, 1983).

B. THE INFLUENCE OF ENVIRONMENTAL CONDITIONS

The influence of environmental conditions on pollutant partitioning is complicated. When environmental conditions change, pollutants in one state or compartment may move to another, and these changes can have important ecological and management implications. Environmental factors affecting sorption reactions include salinity, concentration of suspended solids, biological cycles, pH, and redox conditions.

For example, it has been shown that cadmium sorbed to particles in river water will partition to solution upon mixing with more saline water in an estuary (Edmond *et al.*, 1985; Elbaz-Poulichet *et al.*, 1987). This is because in the more saline environment free cadmium ions (Cd^{+2}) complex with chloride, changing

the sorption equilibria (Comans and van Dijk, 1988). Kuwabara *et al.* (1989) found a similar situation in San Francisco Bay. Salomons and Kerdjik (1986) caution, however, that the behavior of cadmium will vary among estuaries depending upon local conditions; complexation of free metal ions may may be influenced by anthropogenic discharges, where organic materials are more readily available to form complexes with trace metals (Sunda and Lewis, 1978) In addition, not all sorption reactions may be as easily reversible as those for cadmium (Di Toro *et al.*, 1985), and the rate of reactions can influence measurements of reversibility (Comans and van Dijk, 1988).

The complexity of the behavior of a pollutant in the San Francisco Estuary is demonstrated by recent work with selenium. Cutter (1989a) examined the concentration of three dissolved selenium species and particulate selenium across salinity gradients in the Estuary during high (April) and low (September) flow periods in 1986. He concluded that there was large temporal and spatial variation in the concentrations of different forms of selenium in the Estuary, with a distinct difference observed between the northern reach and South Bay. Selenium in the South Bay exhibited a conservative mixing profile consistent with dilution of point source inputs. The behavior of selenium in the northern reach of the Estuary was a more complex function of changing riverine inflows, seasonal biological cycles, point source inputs, and dissolution of particulate selenium. Cutter (1989a) pointed out that predictions of the fate of selenium become less precise under such conditions.

Gunther *et al.* (1990) reviewed data on the effect of Eh (oxidation-reduction potential) and pH upon the partitioning of trace metals in sediments. They reviewed experiments that demonstrate the mobilization of trace metals from various sediment components in response to changes in Eh and pH. Increases in Eh will also precipitate amorphous hydroxides of iron and manganese, as these elements are dissolved under reducing (low Eh) conditions. The precipitation of these hydroxides can "scavenge" other trace metals from solution, as observed by Elbaz-Poulichet *et al.* (1984) for lead in the Gironde Estuary of France. These scavenged metals will thus be found sorbed to particles even though, under equilibrium conditions, these elements would be found in solution. Flocculation of particles into aggregates can also affect adsorption/desorption reactions. Although environmental conditions might dictate adsorption or desorption, flocculation can limit the surface area exposed to surrounding water and thus the rate at which equilibria can be reached (Lick, 1988).

While we have a general understanding of partitioning in aquatic environments, specific information regarding a given substance and ambient environmental conditions is critical to predicting the pollutant fates. In the high-energy, variable ecosystem that is the San Francisco Estuary, changes in environmental conditions over relatively small spatial and temporal scales make it difficult to predict the behavior of pollutants in the Estuary. It is likely, however, that there are key environmental factors that interact with characteristics of the substance in question to play a dominant role in its fate in

the ecosystem. Thus, Elbaz-Poulichet *et al.* (1984) were able to ascertain the major processes controlling lead cycling in the Gironde Estuary. As Salomons and Kerdijk (1986) point out, however, differing conditions among estuaries can cause the same pollutant to behave differently.

A clear research need is to further elucidate dominant adsorption, desorption, and complexation processes for pollutants of concern in the San Francisco Estuary. These processes include the characteristics of important pollutants relevant to sorption, such as (1) the speciation of pollutant loads to the Estuary, (2) the fractions to which a given pollutant sorbs, (3) the kinetics of sorption/desorption reactions, and (4) the influence of key environmental parameters such as salinity and biological cycles. These studies will help us understand what controls the fate of pollutants in the Estuary.

Given the number of pollutants in the Estuary and the variation in environmental conditions, research on the fate of pollutants must focus on particular compounds. These should be selected because they are especially toxic or prevalent in the Estuary, or because of their value as "model" compounds, in that understanding their chemical behavior might allow useful generalizations to be made regarding a variety of pollutants or pollutant classes. The results of these studies would help regulatory agencies develop a systematic framework for predicting the fate of pollutants in the Estuary.

C. PARTITIONING AS A REGULATORY TOOL

In theory, once the key physical and chemical characteristics of a substance and the ambient environment are defined (a fate assessment), it should be possible to predict the partitioning of that substance in an ecosystem (a fate model). A variety of fate models are available for use in the Estuary, such as those provided by the Center for Exposure Assessment Modeling of the USEPA (Ambrose *et al.*, 1990). Many such models use the concept of fugacity, in which the "escaping tendency" of a substance is used to determine equilibrium concentrations in various environmental compartments (Mackay, 1979; Mackay and Paterson, 1981), and sediment quality criteria are developed via the equilibrium partitioning approach (Shea, 1988; Pavlou, 1988).

In the latter approach the concentration of a pollutant in water is governed by its concentration in sediment and reversible exchange reactions between the two phases. Given a pollutant water concentration of toxicological significance (e.g., national water quality criteria), one can predict the sediment concentration needed to produce a particular water concentration by:

$$K_P = \frac{C_{sed}}{C_{wat}}$$

where C_{sed} is the concentration of the pollutant in deposited or suspended sediment, C_{wat} is the concentration in interstitial water or the water column, and K_P is the partitioning coefficient. As K_P varies with organic carbon content of the sediments for many pollutants, this relationship is often converted by normalizing the sediment concentration to total organic carbon and expressing K_P as K_{OC} (Pavlou, 1988).

Uncertainty prevents the precise determination of sediment quality criteria using the equilibrium partitioning approach. Pavlou (1988) stated that estimates derived by equilibrium partitioning should be considered a "first cut." The extent to which future research will be able to reduce this uncertainty is unclear. The complexity of the partitioning of trace elements in sediments suggests that sediment quality criteria developed for metals will remain uncertain for a long time (e.g., Unger's studies [1988] of tributyltin in the Chesapeake). Although the sediment-water partitioning of many organic species is better understood, uncertainty remains regarding criteria developed for these substances as well. It remains to be seen whether a particular K_{oc} value adequately describes the partitioning of an organic pollutant in different sediments.

Another source of uncertainty in the equilibrium partitioning approach relates to the water concentrations (c_{wat}) used in the calculations. The predicted safe levels for sediments (Pavlou, 1988) assumed that values for c_{wat} were readily available, which is not the case for many pollutants. The use of imprecise estimates for c_{wat} in these cases will add additional uncertainty to determining safe levels for pollutants in sediments.

If uncertainties cannot be reduced, the equilibrium partitioning approach can still be used to develop sediment quality criteria. Such development will require simplifying assumptions, however, which must be implemented in a clear and systematic fashion. This type of methodology has been used by the USEPA to develop water quality criteria in the face of uncertainties such as varying species sensitivity to a particular pollutant.

3. Transport

A. INTRODUCTION

Complex currents in the Estuary, produced by the interaction of tides, freshwater inflows, and winds, transport dissolved particles and particle-associated pollutants far from their original point of introduction. The processes of partitioning and transport (as well as transformation; see Section IV.C.4) are not independent. It would be entirely possible for a substance to be chemically transformed to a form that would dissolve (partition) more readily into water and, thus, be more easily transported in the Estuary (e.g., the biologically-mediated transformation of mercury to methylmercury).

When discussing transport one must distinguish net transport over a complete tidal cycle and transport within a tidal cycle. Thus, a parcel of water typically travels 10 km during a tidal cycle in the Estuary (the "tidal excursion"), but this may produce only limited net movement (Conomos, 1979). Net movement of water parcels is often referred to as net or residual currents (Kjerfve, 1989).

B. CIRCULATION AND MIXING

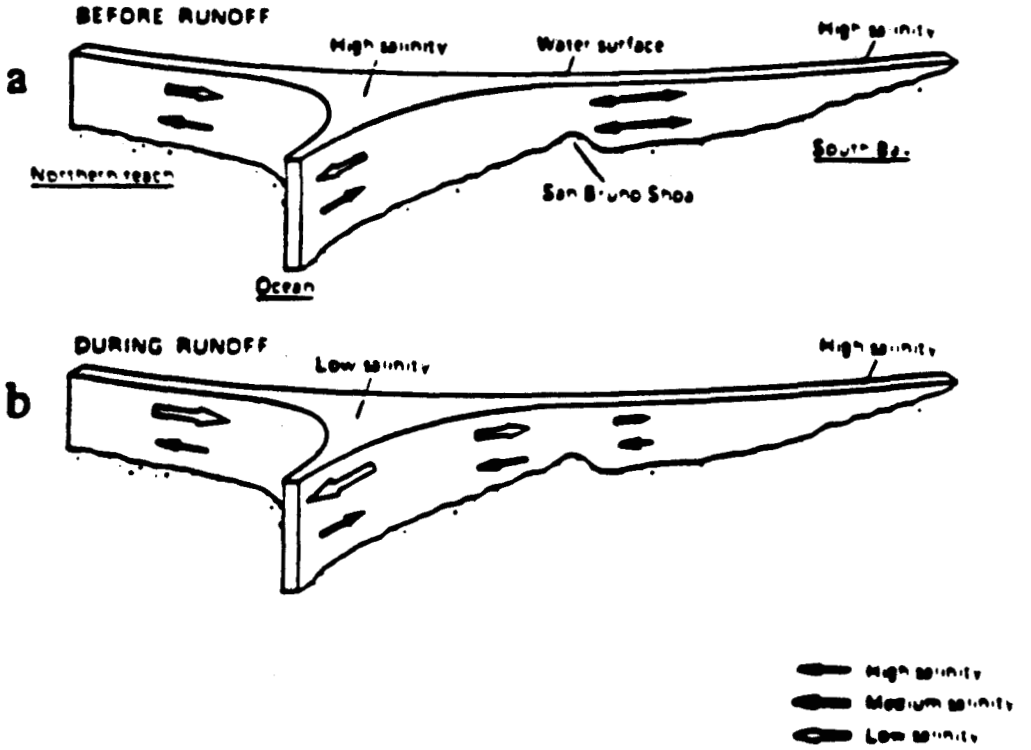
Circulation and mixing in the Estuary are driven by currents induced by tides, riverine inflow, and winds. In the San Francisco Estuary, the tidal cycle is 24 hr and 50 min long, with two high tides and two low tides in each cycle. Over each month, there are periods when the range from high to low tide is larger (spring tides) and periods when it is smaller (neap tides). The morphometry and bathymetry of the Estuary result in amplification of the tidal range in the South Bay and reduction of the tidal range in the northern reach relative to the Golden Gate. Morphometry and bathymetry also affect the timing of maximum tidal currents (e.g., the South Bay is flooding while the northern reach is ebbing), which allow for the exchange of water between these two portions of the Estuary.

General circulation patterns in the northern reach of the Estuary (San Pablo Bay, Suisun Bay, and the Delta) are distinct from those in the South Bay. In the northern reach, river inflow generates gravitational circulation, in which seaward-flowing surface currents of low salinity water are found in conjunction with landward-flowing bottom currents of high salinity water. At the farthest landward penetration of saline water is a region of mixing known as the "null zone," which is characterized by an accumulation of particles including phytoplankton. Landward of the null zone net currents are seaward at all depths. Vertical mixing induced by high spring tides will weaken gravitational circulation.

There is rarely sufficient freshwater inflow into the South Bay to establish the salinity gradients that drive gravitational circulation. Water movements in the South Bay are slower, and winds play an important role in circulation. The steady summer northwesterlies move surface waters to the southeast in the South Bay, generating compensatory currents flowing to the northwest in deeper waters. During periods of high freshwater inflow in the winter, however, the significant influx of fresh water into Central Bay can generate a gravitational circulation cell between South and Central bays. It does not only extend south of San Bruno Shoal under conditions of high flow (Figure 25a). This gravitational circulation cell initially operates in reverse of the cell in the northern reach; lower salinity surface water in Central Bay is drawn into South Bay while higher salinity water moves at lower depths into Central Bay. After large runoff events, the salinity of Central Bay rises quickly as ocean water intrudes with the tides, and the circulation cell reverses itself (Figure 25b) (Smith, 1987).

The volume of freshwater inflow has a profound effect upon circulation patterns in all embayments. High freshwater flows result in more intensive mixing and more rapid circulation in the northern reach and South Bay. High freshwater inflow moves the null zone toward San Pablo Bay, altering the circulation and mixing regimes of the northern reach. The null zone appears to advance downcurrent rapidly in response to peak flow, while its retreat back upstream in the Estuary may take several months (Smith, 1987).

Figure 25. Idealized summer (a) and winter (b) patterns of landward-seaward net currents in the channels of South Bay. During summer the lack of a salinity difference between Central and South bays leads to oscillatory net currents (a). During winter, when large freshwater inflows lower the salinities of Central Bay below those of South Bay, gravitational exchanges between the two embayments are induced (b). After Smith (1987).



The temporal variability of circulation and mixing processes in the Estuary causes variability in pollutant transport. The mean hydraulic residence time, the average time a parcel of water remains in the Estuary (or any of the smaller embayments), provides some indication of pollutant transport, particularly for dissolved species (the transport of particle-bound pollutants is complicated by flocculation, deposition, and erosion [see section C below]). Walters *et al.* (1985) calculated mean residence times (τ) for various embayments in the Estuary (Table 17). Residence times in the different embayments varied significantly depending upon whether high-flow or low-flow estimates were used. During high-flow periods τ was estimated at 1.2 d for the northern reach; under low-flow conditions τ was estimated to be 60 d. The different hydrological regimes between the South Bay and the northern reach are reflected in Table 17, as the larger estimates of τ for the South Bay reflect the more sluggish currents of this embayment. Walters *et al.* (1985) did not consider evaporative losses from the South Bay or freshwater inflow from sewage treatment plants (annual average of 15.2 $m^3 s^{-1}$ [1,321 million L d^{-1}; 342 MGD]), both of which would influence estimated residence times, particularly during summer months.

These estimates for entire embayments provide important information with respect to the time scale of chemical and biological processes and fate of pollutants (including partitioning and transformation). During high-flow conditions a chemical transformation that occurs on the timescale of one week will be unimportant in the northern reach (τ = 1.2 d). Under low-flow conditions such a process might be very important (τ = 60 d). The estimates of τ for entire embayments may underestimate the actual residence time for water (and solutes) in the large shallow reaches of Suisun, San Pablo, and South Bays. This fact has important implications with respect to the fate of pollutants discharged into channels from deep-water outfalls, as compared to runoff in bayshore discharges.

The utility of residence time estimates for entire embayments has been questioned by Smith (1987), who noted that processes within an embayment in the Estuary could cause significant deviations from the basin-wide estimate for τ. The frequency distribution for residence times in an embayment is probably positively skewed; some water parcels demonstrate residence times much greater than the mean (e.g., $\tau_{mean} > \tau_{median}$). Very few data exist on salinity and currents in the broad, shallow reaches of the Estuary, as the majority of data have been collected in channels (Walters *et al.*, 1985; Smith, 1987). This is, in part, a practical problem associated with the difficulty of measuring currents in shoal areas (<2 m) with a relatively large tidal range (2 m) and large wind waves (>1 m) (Walters *et al.*, 1985). Highly sensitive current meters are now available to make these measurements (L. Smith, USGS, personal communication).

Local variations in bathymetry may also affect circulation and mixing in particular embayments. This is apparent in the South Bay, where data suggest

Table 17. Estimated mean hydraulic residence times (d) for the San Francisco Estuary (Walters *et al.*, 1985). Low flow and high flow are 10^2 and 10^4 $m^3 s^{-1}$, respectively.

Embayment	High Flow	Low Flow
Suisun Bay	0.5	35
San Pablo Bay	0.8	25
Northern reach	1.2	60
South Bay	120	160
South Bay (north of Dumbarton Bridge)	80	120
Extreme South Bay (south of Dumbarton Bridge)	40	70

three distinct mixing zones (Ambler *et al.*, 1985; Powell *et al.*, 1986; Smith, 1987). These zones are the northern portion of South Bay north of San Bruno Shoal, south from San Bruno Shoal to the San Mateo Bridge, and south of the San Mateo Bridge. A large, clockwise gyre may form in the central portion of the South Bay. Such a gyre is estimated by numerical models of this region, although there are few data on currents in the shallows (see above) to verify these predictions. The residence times estimated by Walters *et al.* (1985) indicate such mesoscale variation within South Bay (Table 17). A large clockwise gyre is also predicted to exist in the shallows of San Pablo Bay (Conomos, 1979; Walters *et al.*, 1985).

Mixing zones in subembayments affect pollutant transport in the Estuary, particularly during low-flow conditions. Thus, dissolved pollutants entering the South Bay south of San Bruno Shoal will not be transported to Central Bay as readily as those entering South Bay north of San Bruno Shoal. Walters *et al.* (1985) identified detailed hydrodynamic studies at San Bruno Shoal and Pinole Shoal in San Pablo Bay as an important research need.

Finally, residence times for the entire Estuary depend upon exchange between the Bay and the ocean; however, few data are available to assess this phenomenon. In order to understand this exchange, models must include the circulation and mixing patterns of Central Bay and the local coastal waters. Vertical current distribution in Central Bay shows net landward flow at lower depths, indicating the importance of gravitational circulation in ocean-Bay exchanges (Conomos *et al.*, 1970; Smith, 1987).

In summary, circulation and mixing in the Estuary are a complex function of tides, river inflows, winds, and the morphometry and bathymetry of the system. Distinct circulation patterns can be identified for the northern reach and the South Bay, although circulation and mixing in both portions of the Estuary vary over the short term under the influence of freshwater inflow. Under low-flow conditions residence times in Suisun and San Pablo bays are about one month. Estimated residence times for South Bay are on the order of several months. During-high flow conditions, residence times are probably reduced to days in the northern reach and weeks in the South Bay.

These estimates are important for understanding the movement of pollutants. More research is necessary to understand the currents in the broad, shallow areas of each embayment. Of particular interest is the question of how winds, tidal fluctuations, and freshwater inflows affect these currents and the resultant exchange of water between the shallows and the channels. More detailed understanding of ocean-Bay exchange at the Golden Gate is needed, along with knowledge regarding the effect of local morphometry and bathymetry upon mixing regimes at key locations in the Estuary. This new information will help us understand the movement of water and associated pollutants in the Estuary.

C. PARTICLE TRANSPORT

Many pollutants of concern partition to particles, including settleable and non-settleable fractions. The transport of these particle fractions cannot be understood solely by studying water movements; pollutant fate assessment must include some understanding of the processes of deposition, erosion, aggregation, and sorption/desorption. Estuaries are among the most complex systems in which to model particle transport, and there are no simplifying assumptions that can be applied to all these processes. Each process must therefore be studied and integrated into a model of particle transport (DiToro *et al.*, 1988).

General patterns of particle transport and residence times are known for the San Francisco Estuary (Cloern and Nichols, 1985; Smith, 1987; Gunther *et al.*, 1990). The sediments of the Estuary are highly dynamic, and transport follows seasonal trends. Higher sediment inflows during the winter combined with generally weaker winds results in particle deposition in quiescent locations such as the broad mudflats of San Pablo Bay. The northwesterly winds of summer induce currents that erode and resuspend these sediments, which can then be transported throughout the system (Sustar, 1982). Winter storms may also result in sediment erosion and transport, and microclimatic variations across the Estuary influence the overall pattern of sediment transport. The dynamic nature of the Estuary and the number of processes that must be described makes developing predictive models of particle transport difficult.

The USCOE developed a model (*TABS-2*) that couples hydrodynamic, disposal, and sediment transport models in an integrated scheme (Pankow, 1988). The hydrodynamic portion of the model has been verified against data from the San Francisco Bay Physical Model, and the transport element includes routines that estimate deposition and erosion of sediments.

This model is still in development. The transport element has yet to be verified against data from the Estuary, and the model has only been used to simulate transport in the vicinity of the Alcatraz disposal site over a 4-hr period. In addition, the hydrodynamic and transport elements are only two-dimensional, and thus cannot directly represent the vertical stratification of currents that exists in the Estuary. The model (as applied to date in San Francisco Bay) only reproduces tidal currents and does not include wind-driven currents. Wind-driven currents (and wind waves) are very important in the broad, shallow reaches of the Estuary, and appear to be important in determining the erosion and transport of sediments from these regions (USCOE, 1976).

In addition, there are some aspects of sediment behavior that are not well understood, but are important for the modeling of particle transport. For cohesive sediments (such as the muds that characterize much of the Estuary), the boundary between suspended and deposited materials is often unclear. Erosion of deposited cohesive material is influenced by the stress history of an individual deposit. Two sediment deposits of similar density might exhibit significantly different erosion characteristics in response to a given shear stress

Water content, sediment composition, and turbulent stress affect sediment erosion. Although these processes can be described quantitatively, they cannot be predicted, requiring study of individual sediments. The importance of major storm events relative to that of several days of moderate winds in producing erosion and deposition is also poorly understood (Lick, 1988).

In summary, the transport of dissolved and particle-associated pollutants in the Estuary can be understood only in general terms. The details of these processes are complex and not well understood. Nonetheless, it is known that movement of pollutants through the ecosystem is controlled by many factors that vary over different timescales. There are seasonal, monthly, and daily changes in water movements that affect circulation; pollutant influx to the Estuary has seasonal and storm-event related components, and sorption/desorption or complexation reactions can occur over minutes to days. In addition, at any one time these factors may vary from location to location within the Estuary.

Understanding particle transport is very important for relating pollutant sources to pollutant abundance, distribution, and effects in the field. The broad scope of particle transport questions that remain to be answered implies that research will only be helpful to natural resource managers if it is applied toward elucidating the key factors that dominate the transport of pollutants of concern. An understanding of these key factors will allow the development of generalized, predictive models for use in assessing alternative regulatory strategies.

4. Transformation

In addition to partitioning and transport, transformation reactions play a role in determining the fate of chemical pollutants in the environment. Transformations may be chemical reactions, or they may be mediated by living organisms such as microorganisms, invertebrates and fishes. Aerobic and anaerobic bacteria are usually considered to be most important mediators for degradation of organic pollutants, while the more important reactions for metals transformation are chemical in nature. Microbial action may cause the complete degradation of organic compounds, or it may cause alterations that change important properties of a substance. Thus, compounds can be made more volatile, more easily metabolized, or more susceptible to further biochemical degradation.

Even among the relatively recalcitrant PCBs, studies in the Hudson River and in New Bedford Harbor have shown that natural populations of microorganisms can perform reductive dechlorination of some PCB congeners. (Brown *et al.*, 1987). Where these processes occur, they may generate PCB residues that contain fewer chlorine atoms than the the PCB molecules that were initially released into the environment. PCB dechlorination processes, however, have only been shown to occur at sites containing high concentrations of PCBs (i.e., more than 200 ug g^{-1} sediment). In addition, very little, if any, research has been conducted on the dechlorinated PCBs to

determine their potential toxicity in estuarine environments. Presumably the process of dechlorination generates PCBs that are more soluble, and which may migrate through sedimentary deposits to the aerobic zone. Once in the aerobic zone they may be attacked by aerobic bacteria capable of complete decomposition of the PCB molecule. The same potential exists in the environment for otherwise "recalcitrant" compounds like DDT, toxaphene, and chlordane.

While bacteria may remove some organochlorines from sediments, some interesting problems may occur in the bacterial metabolism of chlorinated compounds like aldrin and heptachlor. Metabolites of these compounds include epoxidated products that are not subject to substantial further degradation in the environment. Thus, aldrin may be metabolized to aldrin epoxide (dieldrin), a compound of much greater environmental persistence and toxicity. Heptachlor is metabolized to heptachlor epoxide which, like dieldrin, is both more toxic and less degradable than the parent compound (see review in Nelson *et al.*, 1987).

Microbial transformations can produce more volatile forms of trace metals, often through the process of biomethylation (Ridley *et al.*, 1977). Mercury-resistant bacteria have been shown to volatilize both organic and inorganic forms of mercury. Nakamura *et al.* (1988) demonstrated that bacteria isolated from the sediments in Minamata Bay, Japan, could volatilize mercuric chloride, and methyl-, ethyl-, and propyl-mercury. Volatilization of mercury has also been demonstrated in less polluted environments (Spangler*et al.*, 1973). Microorganisms can also methylate inorganic mercury, significantly increasing its toxicity and mobility in aquatic environments (Jensen and Jernelöv, 1969). Methylation of mercury occurs in San Francisco Bay sediments *in situ* (Olson and Cooper, 1975) and in the laboratory (Olson and Cooper, 1976). High organic content in the sediments was associated with greater biomethylation capacity. Harrison and Laxen (1978) concluded that microorganisms are alkylating lead in the mudflats of Morecambe Bay in the United Kingdom. These investigators found significant quantities of alkylated lead in coastal air masses, and their studies indicated a source distinct from urban areas.

Non-biological transformations also take place in aquatic systems. These include hydrolysis, oxidation, and photochemical reactions. The photochemical reactivity of several trace metals has recently been reviewed by Waite (1988), who documented photo-induced changes in speciation for organically bound metals (copper and cobalt) and the decomposition of alkylated forms of zinc and tin (including tributyltin). The photo-decomposition of tributyltin (Bis[tributyltin] oxide) is rather slow (half-life of 89 d), although the rate of this reaction increases in the presence of humic substances. Waite (1988) points out that photo-induced changes in speciation or binding of trace metals in waters could influence the mobility, bioavailability, and toxicity of heavy metals in aquatic environments and deserves further study.

5. The Biological Uptake of Pollutants

In order to assess the effects of pollutants on biota in the Estuary it is necessary to understand the bioaccumulation process. In general, accumulation by the biota of materials introduced to the Estuary is indicative of human impact, and the potential exists that the pollutants accumulated by organisms in the Estuary are exerting toxic effects. However, it is important to note the following: 1) not all biological effects of chemical pollutants will be associated with accumulation of the pollutant (e.g., the acutely lethal surface-active effects of Cu or Zn on the gills of fish); 2) accumulation of a pollutant by an aquatic organism does not, necessarily, imply that an effect is occurring, or will occur; and 3) bioaccumulation to high concentrations in some tissues is, in fact, a means for detoxification of a pollutant (e.g., the binding of Cd with metallothionein, or the accumulation of PCBs in depot lipids) (O'Connor and Rachlin, 1982; O'Connor and Kneip, 1986; Nelson *et al.*, 1988).

Nonetheless, uptake of pollutants demonstrates that the organisms have been exposed to the pollutant in question, and, therefore, that the pollutant is present in bioavailable form in the environment (Phillips, 1980).

Biological uptake of pollutants has two components: 1) bioconcentration, or the accumulation of pollutant materials directly from the water column and 2) bioaccumulation, or the total accumulation of pollutant materials *via* the processes of bioconcentration and trophic transport. We shall refer to the general process of biological uptake as the latter unless the discussion is specifically related to uptake from water.

Uptake in freshwater, marine, and estuarine species has three main implications:

1) Even though ambient water concentrations of various pollutants may be low, concentrations in an organism's tissues may reach the point where they are toxic to that organism.
2) Even if such concentrations do not affect the organism itself, they may adversely affect the health of predator organisms, including humans.
3) The concentration in the tissues of some organisms may be used (within certain definable limits) as an indicator of the concentrations present in the surrounding waters. This concept has been widely adopted worldwide and is embodied in programs such as "Mussel Watch," which uses mussels as sentinel (or "indicator") organisms to assess levels of organic and trace metal pollution. The role of various organisms as biological indicators has been the subject of extensive reviews by Phillips (1977a, b; 1978a, b; 1980; 1988).

Previous sections of this report (IV.B. and IV.C.2.) discussed the variety of forms in which pollutants may exist in the San Francisco Estuary. It is important to realize that sorption and chemical speciation play a role in defining the extent to which pollutants are available to be accumulated by living organisms (i.e.

their "bioavailability"). Measurements of total pollutant concentrations in the water column or in sediments are often poor indicators of the extent to which such substances may be accumulated by biota. Thus, as stated by Waldichuk (1985) "...there is no true substitute for chemical analysis of the tissues of exposed [sessile] marine organisms for metal concentrations, if one wishes to determine the biological availability of metals at a given site." A similar rationale is implicit in defining the bioavailability of organic pollutants as well.

The following section will discuss bioaccumulation in terms of the factors determining the bioavailability of pollutants and will also discuss food-web transfer of pollutants in the Estuary. Particular attention will be paid to local species at greatest potential risk from certain pollutants of concern, such as selenium and PCBs. The effects of pollutants upon organisms and other "beneficial uses" of the Estuary is reviewed in Section IV. D.

A. THE IMPORTANCE OF PARTITIONING AND SPECIATION TO BIOAVAILABILITY

The physicochemical form of pollutants in waters and sediments may vary greatly. Determining the availability of pollutants to living organisms is a crucial element in assessing the potential toxic effects on living resources in the Estuary.

Trace Elements

Determining the significance of trace metal contamination in aquatic environments is a complicated problem in that the bioavailability and toxicity of any particular metal is not entirely dependent upon its concentration in the surrounding environment. The toxicity of trace metals in the marine environment is determined by (1) the toxicity of the individual metal in question, (2) the synergistic or antagonistic aspects of toxicity when trace metals are present in combination, and (3) the prevailing physicochemical parameters, such as pH, Eh, salinity and temperature, which determine the chemical and physical form of the metal.

Numerous studies of the physicochemical speciation of metals in estuaries support the contention that bioavailability changes constantly as metals undergo adsorption, precipitation, dissolution, and complexation processes (Fletcher *et al.*, 1983). Luoma (1983) has reviewed the factors that determine the uptake of trace metals by aquatic organisms and highlights the environmental and biological processes affecting bioavailability.

The proportion of free metal ions is one of the more important parameters governing bioavailability, especially for cadmium, copper, iron, manganese, and zinc (e.g., see Zitko *et al.*, 1973; Pagenkopf *et al.*, 1974; Sunda and Guillard, 1976; Andrew *et al.*, 1977; Sunda *et al.*, 1978; Anderson and Morel, 1982). As mentioned previously, factors such as salinity and suspended solids concentration can affect partitioning, and thus the bioavailability, of free metal ions. Most metal complexes are not bioavailable, although there are some

exceptions. Separating the precise role of dissolved and complexed metals in bioavailability has proven difficult (Luoma, 1983).

Sediments represent a significant pool of trace metals. The behavior of sediment-associated metals is complex and may be affected by such parameters as oxidation-reduction potential (Eh), pH, and salinity (see Gunther *et al.* [1990] for a more detailed discussion). A significant portion of the trace metals present in sediments (with the notable exception of cadmium) are often lattice bound, thus making them unavailable for uptake by living organisms. However, accumulation of trace elements from sediments and suspended particles has been demonstrated in a number of instances. Luoma and Jenne (1977) demonstrated that deposit feeding clams accumulated significant amounts of ^{110}Ag, ^{109}Co, and ^{65}Zn *via* ingestion. Uptake varied with sediment type, and differences in bioavailability of the metals were related to the strength of metal binding to particles. Weakly bound metals were more available, and more strongly bound metals less so.

Polychaetes live in sediments and therefore may be exposed to high concentrations of pollutants. Bryan and Hummerstone (1973a, b, c) showed that the sand worm *Nereis diversicolor* accumulated copper, zinc, cadmium, and manganese from sediments. However, bioaccumulation was dependent upon a number of factors, including salinity, animal size, the ability of the organism to regulate metal uptake, and the acquisition of tolerance in chronically polluted situations. In the case of manganese, accumulation appeared to be related to a rapidly exchanging pool of the element associated with the salinity of the interstitial waters in the sediment.

Luoma and Bryan (1978, 1979, 1981) examined the relationship among sediment concentrations of silver, cadmium, cobalt, copper, lead, and zinc and concentrations in tissues of the burrowing clam *Scrobicularia plana* and the polychaete *Nereis diversicolor*. They determined a significant (but weak) correlation between sediment concentrations of all metals except copper and the tissues of *S. plana* in 17 estuaries. Stronger correlations were observed among *N. diversicolor* and copper and lead concentrations in sediments. It would appear that detritus feeding organisms are capable of accumulating trace elements from sediments; accumulation in such circumstances is due to dietary intake.

Luoma *et al.* (1985) showed significant temporal variation in the uptake of silver, copper, and zinc in the deposit feeding clam *Macoma balthica* in South San Francisco Bay. These variations included distinct seasonality for copper and silver at one station, but not at others. These studies were able to differentiate the influence of biological cycles and variation in discharge. Luoma *et al.* (1985) concluded that increases in bioaccumulation were correlated with hydrodynamic processes, fluctuating geochemical factors, and biological cycles, as influenced by seasonal and annual rates of Delta outflow.

Filter feeders accumulate pollutants, including trace elements, not only from solution, but also from inorganic particles (Preston *et al.*, 1972; Raymont,

1972; Boyden and Romeril, 1974). Variations in accumulation may be related to changes in metal availability. Bivalve molluscs (e.g., *Mytilus*) take up metals rapidly from solution and from food. Dietary accumulation was the dominant route of uptake in instances where there was no direct discharge of metals (Phillips, 1976a,b). Bivalves can respond to metals present in sediments, but as Phillips (1980) points out, the importance of this response in determining total body burdens of metals probably varies with species and the element.

Luoma (1983) described the importance of metal speciation in determining bioavailability. Methylated forms of mercury, tin, and arsenic have been observed to accumulate in many organisms; and the accumulation and toxicity of methyl mercury has been well documented, particularly in instances of large-scale contamination such in Minimata Bay, Japan. Oxidation states also affect the bioavailability of certain metals, including mercury, selenium, arsenic, and chromium (Luoma, 1983). As a rule of thumb, Hg^0 is more available than Hg^{2+} (due to the lipid solubility of the former); Se^{2+} is more available than Se^{6+}; and Cr^{6+} is more available than Cr^{3+}.

Of the many factors influencing the uptake of trace elements by the biota, Luoma (1983) has suggested that the most important are (1) metal concentrations in solution, (2) speciation of metals in solution, (3) metal concentrations in food, (4) partitioning of metals within food materials, (5) the influence of other cations, (6) temperature (especially for non-exchangeable metals), and (7) pH and redox potential. Luoma (1983) also suggested that separation of the relative importance of these parameters is the most important problem we face in predicting trace metal bioavailability in nature.

Organic Pollutants

Many of the organic pollutants of concern, such as PAHs, PCBs, dibenzofurans, and dioxins, have low water solubilities but are very soluble in lipids. Thus, they may concentrate in the lipid-rich tissues of aquatic biota. The ratio of the concentration of organic pollutants in the tissues of an organism to the concentration in water is the bioconcentration factor (K_{bc}). K_{bc} is inversely proportional to water solubility, so in the case of PAHs, less-soluble compounds (such as benzo(*a*)pyrene) are accumulated more readily than more-soluble compounds (e.g. naphthalene). However, one cannot assess the impact of lipophilic organic pollutants in aquatic environments without understanding their interactions with particulate matter.

Metals and organic pollutants with low solubility tend to attach to suspended particles which may, then, settle in areas with low current velocities. Numerous places have been found in the Estuary where pollutants have accumulated to high, potentially toxic concentrations (Luoma and Cloern, 1982). It has often been assumed that certain organic chemicals, when adsorbed to particles or settled in the sediments, are not bioavailable. However, as Adams (1988) suggests, the problem is complicated by the wide variety of organic chemicals (many of anthropogenic origin) that need to be assessed, and the variety of different sediment types with which they may interact. The

feeding habits of organisms also play a key role in the bioavailability of sediment-bound organic pollutants, particularly at the micro-environment level (Karickhoff and Morris, 1988). In the case of burrowing organisms, or those whose membranes are in direct contact with sediment-bound organic pollutants, desorption kinetics play an important role in chemical uptake and the concentration of pollutants in sediment pore waters is likely to be an important factor (Adams, 1988).

Karickhoff and Morris (1988), in their recent review of the relationship of pollutant sorption to bioavailability, stressed that uncharged sorbates, such as PCBs and DDT, are not irreversibly bound to sediments, but when such compounds have been in contact with sediments for prolonged periods, a significant portion of the sorbed material becomes very resistant to release. This phenomenon is apparently associated with physical inaccessibility, as chemicals become "buried within" sediment particles (e.g., in an aggregate structure or within a sorbant component such as organic matter). Nonetheless, a "labile component" of adsorbed organics will exist in a contaminated sediment, and may be available for release into nearby aqueous phases. Current knowledge of pollutant fluxes among sediments, water, and biological membranes is relatively poor and is an area where further research effort needs to be placed.

Anderson *et al.* (1988), in summarizing bioaccumulation of sediment-associated chemicals, suggest that there are two essential conditions that must be taken into consideration when assessing bioavailability. These are that (1) determinations must be made on a site-specific basis, and (2) measurement of bioavailability must include consideration of all possible avenues of exposure. These authors list the following factors as important in the determination of bioavailability:

1) Organic carbon is an extremely important factor controlling the bioavailability of hydrophobic organic chemicals (such as PCBs, DDT, and certain PAHs) adsorbed to settled or suspended sediments. Clay content can also be an important controlling factor.
2) Bioavailability is affected by dissolved organic carbon (DOC) and total organic carbon (TOC). TOC is the dominating factor in the water column, while DOC is important in interstitial waters.
3) The bioconcentration of neutral organic chemicals in fish, invertebrates, sediments and detritus can, in theory, be normalized by the use of organic carbon.
4) Interstitial waters are the primary route of exposure for many benthic organisms, and concentrations of organic pollutants in interstitial water may provide a means of calculating safety factors for benthic invertebrates.

As described in Section IV. C. 2, further research is needed in this area, and currently available techniques for measuring pollutants in interstitial waters may be insufficient to detect low concentrations of many organic pollutants, or to

properly ascertain what fraction of the measured pollutant was, truly, in the interstitial waters.

It is clear from the preceding discussion that the bioavailability of organic pollutants is complicated by considerations of physicochemical speciation, and the nature and route of exposure. This has recently been demonstrated by Murray and Richardson (1988), who examined the uptake of PAHs by mussels (*Mytilus edulis*) in Port Phillip Bay, Australia. Automatic integrating samplers (Seastar®), capable of separating "dissolved" and "particle-bound" fractions of PAHs in more than 100 L of water over a 96-hr period, were placed alongside mussels at several sites, including Hobsons Bay (influenced by runoff from the Yarra River, including the City of Melbourne), Corio Bay (near an oil refinery), and Port Phillip Bay proper (a control site). Hydrocarbons present at the Hobsons Bay site included combustion-derived PAHs and high-boiling-range petroleum hydrocarbons characteristic of lubricating oil. At the Corio Bay site, hydrocarbons present in the samples spanned the whole crude oil range, along with some synthetic materials and halogenated aromatics.

Duplicate measurements using mussels and integrating samplers undertaken in the Corio Bay studies showed excellent agreement, indicating that both methods of sampling produced consistent results. In addition, the range in lipid concentration factors (K_{bcl}) from site to site was very narrow for total petroleum hydrocarbons and for aromatic and biogenic hydrocarbons. There was no marked difference between K_{bcl} and K_{bcd} (measured on a dry weight basis) between total and aromatic hydrocarbons. However, bioconcentration factors for individual PAHs were not the same as for petroleum hydrocarbons and biogenic hydrocarbons, and were not the same at all of the sites. The reason for this was most likely the association of these compounds with particulate matter.

Bioaccumulation of toxic pollutants (both trace metals and organic compounds) is not a straight-forward matter and is complicated by physicochemical factors such as speciation and partitioning; by biological variables including the type of organism involved, its habitat, and its feeding habits; and also by environmental factors such as season, or temperature, which may alter the fate, distribution patterns, and bioavailability of individual pollutants markedly (Phillips, 1980). At present, it remains extremely difficult to estimate potential bioavailability of pollutants solely from analyses of water, sediments, or suspended particles; there still remains no substitute for tissue analysis in the estimation of bioavailability.

B. FOOD-WEB TRANSFER

Considerable concern has been expressed regarding the possible magnification of pollutant concentrations in the tissues of living organisms by passage through food webs. The net result of "biomagnification" is that organisms at higher trophic levels (e.g., fish, birds, and marine mammals) may accumulate large concentrations of toxic pollutants in their bodies and, thus, may be subject to an increased risk of health impairment. In cases where the

organisms are also a source of human food, the implications are obvious: the episode of mercury poisoning at Minimata Bay, Japan, provides ample testimony to this fact.

Food-web transfer of pollutants is a complex affair, and the pitfalls in understanding the problems involved have been aptly summarized by Harvey *et al.* (1974) in their statement:

> Marine food webs are very complicated and there is much disagreement among marine biologists on prey-predator relationships. Many marine organisms are very opportunistic and flexible in their diets. Any attempt to use a chemical tracer for food-web information will be confounded by these facts, as well as by differences between individuals of a species. Thus, we must accept 'order of magnitude' comparisons...

There have been many investigations of food-web transfer of toxic pollutants undertaken (see the review by Phillips, 1980). The pollutants of greatest concern in this regard are the highly persistent lipophilic organics (e.g., PCBs and DDT). Dietary uptake of toxic pollutants can be important for chemicals with a high K_{bc} and a long half life (*i.e.*, $\geq$ 50 d), especially when these compounds are not metabolized by the organism (Anderson *et al.*, 1988). Although the results of many studies involving lower trophic levels have been equivocal, high-ranking predators appear to be the organisms most susceptible. For example, studies in several parts of the world have shown that seals accumulate PCBs and DDE to high concentrations from their diets and may suffer reproductive failures as a consequence of accumulation (see Reijnders, 1986).

The ability of many organisms to regulate metals in their tissues limits the amplification of these pollutants *via* food-web transfer. However, it is generally accepted that the phenomenon occurs in the case of methylmercury, which appears to behave in a similar fashion to organochlorine pollutants. Phillips (1980) speculates that this may be due to the similarity in the cycling of these compounds in ecosystems, which reflects a similar persistence (although based upon different methods of sequestration).

In general, the concentrations of pollutants of concern in the tissues of upper-trophic-level organisms (including fish, birds, and marine mammals) in the San Francisco Estuary may be concluded to be due to food-web transfer; this conclusion is based upon the few data available, as well as numerous studies of food-web transfer of organic pollutants in predatory fishes from other systems. O'Connor and co-workers (Pizza, 1983; Pizza and O'Connor, 1983; O'Connor, 1984; O'Connor and Pizza, 1987a, b) determined that PCBs were accumulated from the food by striped bass, and that as much as 90% of a dose could be assimilated into the tissues. However, elimination of a substantial fraction of a dietary dose of PCBs was rapid; within 120 hr as much as 50% of the assimilated PCB was eliminated. The remainder became associated with lipids and was eliminated slowly. Similar conclusions have been made by workers studying PCB accumulation in lake trout (Thomann and Connolly, 1984) and spot (*Leiostomus xanthurus*; Rubinstein *et al.*, 1984).

The available data for trace metals in the Estuary have recently been reviewed by Luoma and Phillips (1988). These authors suggest that pollutant enrichment at a site in one benthic species is generally accompanied by enrichment throughout a benthic community. In the case of silver in South San Francisco Bay, extension of existing trace element contamination to higher trophic levels is indicated. Copper and silver enrichment has been observed in Palo Alto and Redwood creeks and is reflected in the liver and kidney concentrations in diving ducks (Ohlendorf *et al.*, 1986b). A similar situation has been observed in the case of selenium in Suisun Bay and Carquinez Strait (Ohlendorf *et al.*, 1987). However, as is the case in many other estuaries around the world, few definitive studies of heavy metal transfer in food webs have been undertaken.

With regard to local analyses of organic pollutants, little information (apart from that relating to bivalve concentrations) is available (Phillips and Spies, 1988). Ohlendorf and Miller (1984) suggested that significant uptake of DDE, PCBs, and hexachlorobenzene occurs in four species of waterfowl (northern pintails, *Anas acuta*; northern shovelers, *Anas clypeata*; canvasbacks, *Aythya valisineria*; and lesser scaups, *Aythya affinis*) during their annual stay in the catchment, although residue levels in general were low. Hoffman *et al.* (1986) have shown that PCB concentrations in whole eggs of black-crowned night-herons (*Nycticorax nycticorax*) are negatively correlated to embryo weights, but despite the reproductive problems involved with this species, they were unable to link the problems conclusively with organochlorine contamination in the Bay. Ohlendorf and Marois (1990) found DDE concentrations in black-crowned night-heron eggs from San Francisco Bay that were higher than those associated with reduced reproductive success of this species. In addition, mean shell thickness of night-heron eggs from the Bay during 1982-1984 was significantly less than pre-DDT thickness and was negatively correlated with DDE concentration.

To date, there has been little work performed with seal populations in the Bay. Risebrough *et al.* (1978) reported elevated concentrations of PCBs in the tissues of a few harbor seals (*Phoca vitulina*) found dead in the Bay, but again there is no evidence to link these concentrations with other local data. A small study currently underway (D. Kopec, personal communication) will assess concentrations of organic pollutants in blood lipid of live seals in Central and South bays.

It is our conclusion, based upon studies from many different water bodies across the country, that bioaccumulation of pollutants in the biota of the San Francisco Estuary probably occurs primarily as the result of food-web transport, and that direct accumulation of dissolved pollutants from the water column is far less important (Pizza and O'Connor, 1987; O'Connor and Pizza, 1987). More research should be performed to assess the specific pathways of pollutant transport in benthic, pelagic, and planktonic communities in the Estuary; however, such studies will represent a specification of vectors and pathways rather than clarification of the basic process of food-web transport.

C. BIOACCUMULATION IN THE ESTUARY

Bioaccumulation of toxic pollutants of concern in the San Francisco Bay-Delta has been the subject of two recent reviews (Phillips, 1987, 1988). These reports provide a detailed overview of this subject, and the interested reader is referred to them for further details. The following general discussion highlights several pollutants that accumulate in biota of the Estuary, based on the findings of Phillips (1987, 1988). Data on the accumulation by mussels of the pollutants discussed below were presented in Tables 3b and 3c.

Mercury is present in the Estuary catchment due to past mining activities, natural cinnabar deposits, and effluent discharges. Mercury is found at high concentrations in invertebrates and fish in the Estuary (e.g., Northeastern San Pablo Bay, Islais Creek [*Mytilus edulis*], and Redwood Creek [*Crassostrea gigas*]) (Girvin *et al.*, 1975; Risebrough *et al.*, 1978). The Toxic Substances Monitoring Program has measured elevated levels of mercury in the axial muscle of fish (SWRCB, 1986), and an advisory has been issued with respect to the consumption of striped bass.

Selenium has been detected in fish, birds, and invertebrates in the Estuary. Higher levels of Se tend to be found in the northern reach (Risebrough *et al.*, 1978; CDFG, 1987). There is evidence that Se causes reproductive toxicity in bird populations at Kesterson National Wildlife Refuge (Ohlendorf *et al.*, 1986a, c). Selenium concentrations similar to those measured in birds from Kesterson (30-40 $\mu g\ g^{-1}$ dry weight in liver) have been measured in greater scaup (*Aythya marila*) and surf scoters (*Melanitta perspicillata*) from the South Bay, giving rise to concern about toxicological impacts (Ohlendorf, 1986c). No adverse effects of Se on waterfowl in the Bay have been detected.

Silver is discharged into the South Bay, and bioaccumulation of silver by invertebrates including *Mytilus edulis* and *Macoma balthica* close to effluent discharges has been documented (Risebrough *et al.*, 1978; Luoma and Cain, 1979; Luoma *et al.*, 1985). Silver has also been found in ducks in the South Bay (Ohlendorf *et al.*, 1986b).

Pesticides are transported into the Estuary from agricultural areas in the Central Valley, and also from agricultural lands in the Delta. Concentrations of DDT remain detectable in estuarine sediments and biota, although these have probably decreased substantially since the ban on DDT use in 1970. Chlordane, dieldrin and toxaphene are all found in invertebrates, fish, and birds of the Estuary and its catchment, as are several other persistent pesticides. Concentrations range from low levels to concentrations exceeding National Academy of Sciences guidelines or U.S. Food and Drug Administration action levels in fish from the Central Valley (SWRCB, 1986). Significant toxicity has also been found to be present in ambient river waters of the upper catchment, presumably linked to pesticide usage. To date, the pollutants exerting this toxicity have not been positively identified, but studies are continuing.

Polychlorinated biphenyls (PCBs) are present in biota at elevated concentrations throughout the Estuary. The available data indicate multiple sources of PCBs, and there is also suggestive (but not yet conclusive) evidence of impacts on fish populations in the Bay (Phillips and Spies, 1988). Few local data exist for dioxins and dibenzofurans. A very recent national survey of dioxins in fish has found significant levels of these pollutants in samples from the San Francisco Estuary and its catchment (Pollock, *et al.*, 1989). This is thought to be due to the chlorination of paper and pulp mill effluents, which is a source of dioxins; however, PCBs may also be implicated.

Petroleum hydrocarbons are not well characterized in the Bay; however, they are thought to be of concern due to the presence of six major refineries on the Bay margins, hydrocarbon loading from urban runoff, and the potential for spills. Monocyclic aromatic hydrocarbons (MAHs) were previously thought to be implicated in effects on striped bass (*Morone saxatilis*) populations in the Estuary, but this evidence remains equivocal (deVlaming, 1988). It is probable that PAHs are of greater impact in the Bay and Delta, but very little is known of the abundance and distribution of this class of pollutants in the Estuary or of their bioaccumulation by local species.

At present, there are few data which accurately document bioaccumulation of pollutants in the Bay and Delta; in addition, few studies have examined spatial and temporal trends of pollutant abundance in biota (Phillips, 1987). Further, the possible transfer of pollutants through food webs in the area has not been investigated to date, and studies of toxic pollutant accumulation by birds and marine mammals (especially seals) are urgently required (see Phillips, 1988). Of particular concern in the Estuary is the accumulation of certain organic pollutants by important fish species, including striped bass (*Morone saxatilis*) and starry flounder (*Platichthys stellatus*). The possible effects of such accumulation will be dealt with in more detail in the following Section IV. D. covering the effects of pollutants in the Estuary.

6. Summary

The fate of pollutants entering the San Francisco Estuary is governed by partitioning, transformation, and transport processes in water, sediments, and biota. Few empirical data are available regarding the fate of pollutants in the Estuary. However, processes controlling pollutant fate and transport may be inferred from studies of similar pollutants in other estuarine ecosystems, and the importance of some of these processes is suggested by observation of localized areas of higher pollutant concentrations in sediments and biota.

Partitioning, or the movement of pollutants among compartments in the system, is driven by the physicochemical nature of a given substance, various environmental factors, and biological processes. The partitioning of compounds among "compartments" in an ecosystem is a continuous process responsive to changes in environmental conditions. In theory, knowledge of the important characteristics of a substance and the ambient environment would allow one to predict the partitioning of that substance in an ecosystem. Various mathematical

and conceptual models have been developed to predict intercompartmental transport of pollutants. A key process in partitioning is the sorption of pollutants to particles, which can influence the bioavailability, transport, and transformation of substances. Sorption reactions are often reversible, and thus the transport of a sorbed pollutant to a location with different environmental conditions might alter its physicochemical status, influencing its bioavailability or mobility in the Estuary.

In addition to partitioning, the fate of pollutants in the Estuary is controlled by transport processes. Complex currents in the system, produced by the interaction of tides, freshwater inflows, and winds, will transport dissolved and particle-associated pollutants far from their original point of introduction. General circulation patterns in the Estuary can be described for the northern reach (San Pablo Bay, Suisun Bay, and the Delta) that are distinct from those in the South Bay. In the northern reach, river inflow generates gravitational circulation, in which seaward-flowing surface currents of low-salinity water are found in conjunction with landward-flowing bottom currents of high-salinity water. In contrast, there is rarely sufficient freshwater inflow into the South Bay to establish the salinity gradients that drive gravitational circulation, and water movements in this embayment are slower. Winds play an important role in circulation in the South Bay, especially during the summer. Circulation and mixing in both portions of the Estuary are significantly influenced by freshwater inflows.

It is clear that the transport of dissolved and particle-associated pollutants through the Estuary is a complex issue. The movement of pollutants through the ecosystem is controlled by many factors that vary over remarkably different timescales. There are seasonal, monthly, and daily changes in water movements that affect circulation; pollutant influx to the Estuary has seasonal and storm-event-related components; and sorption-desorption or complexation reactions can occur over minutes to days. In addition, these factors may vary from location to location in the Estuary at any one time.

The final aspect of the fate of pollutants is their interaction with biota in the Estuary; the potential adverse effects of these substances cannot be manifest unless organisms are exposed directly to the pollutant. Concentrations of pollutants in an organism's tissues may reach the point where they are toxic to that organism, or its predators, even though ambient water concentrations of various pollutants may be very low. The concentration of pollutants in an organism's tissues may be used (within certain definable limits) as an indicator of the concentrations present in the surrounding waters, which is the concept underlying biomonitoring or "Mussel Watch" programs.

Bioaccumulation of pollutants is a complex function of physicochemical factors (e.g., speciation and partitioning); biological variables (e.g., species, habitat, physiology, and feeding habits); and environmental factors (e.g., season and local hydrodynamics) that may alter the distribution and bioavailability of individual pollutants. It remains extremely difficult to estimate potential bioavailability of pollutants solely from analyses of water, sediments,

or suspended particles; there still remains no substitute for tissue analysis in the estimation of bioavailability. An improved understanding of the interplay among physiological, biochemical, geochemical, and ecological factors is required before predictions of bioavailability, based upon knowledge of water and sediment chemistry, can become a reality.

7. Gaps in Knowledge

Additional studies are required in order to specify the distribution or abundance of pollutants in the water, sediments, and biota of the Estuary, and to validate existing models and predictions. The major gaps in knowledge regarding the fate of pollutants in the Estuary are:

1) An insufficient understanding of the actual partitioning of pollutants between and among the actual compartments of the Estuarine ecosystem. Studies should be undertaken using appropriately chosen, surrogate compounds in order to determine the rates and directions of pollutant transport between water and suspended solids, between water and tissue, between suspended and deposited solids, and between deposited solids and the water column.
2) An inadequate understanding of the speciation of chemical pollutants, especially metals and organometallic compounds, in the Estuary. Speciation of pollutants will, in all likelihood, prove to be the major factor determining bioavailability of some classes of compounds, and an improved understanding of metals speciation under a variety of environmental conditions will vastly improve our ability to predict their behavior in the system.
3) An inadequate understanding of particle and sediment transport in the Estuary, especially with regard to the seasonally-variable residence times of suspended particles in the sub-embayments that comprise the Estuary. Studies should be undertaken to determine masses and volumes of particle transport and to develop adequate predictive models of residence times so that it will be possible to estimate pollutant mass transport in the system.
4) Few data are available to describe the microbial populations of the Estuary and the extent that they play a role in the transformation and degradation of pollutants. Additional information is needed in this area, particularly for basic processes affecting microbial populations, and factors affecting microbial transformation.
5) The lack of comprehensive data on the distribution of pollutants in the biota. The toxic metals and their bioaccumulation have been studied in some depth, at some locations; however, very few data are available on concentrations of organic pollutants in the biota of the Estuary, including classes of compounds known to have significant impacts on the health of aquatic biota, such as the PAHs.

D. POLLUTANT EFFECTS IN THE ESTUARY

1. Introduction

This section contains a condensed version of AHI's review of pollutants in San Francisco Bay. This summary provides the reader an overview, but it has been simplified and not extensively referenced. The full review of pollutants in the Estuary may be found in Phillips (1987). Following this summary the results of more recent work undertaken in the Bay-Delta are reviewed. In addition, the scientific bases for regulatory and other management actions will be examined.

The purpose of this section is to determine what we know about the effects of pollutants in the Estuary. Certain knowledge of effects resulting from pollutants in the Estuary is difficult to obtain; effects are often inferred as a result of field or laboratory studies and extrapolation of those results to field situations. Pollutants exist in the Estuary in complex mixtures which, acting together, may increase or decrease toxicity. The total effect of mixtures may be equal to the sum of toxicity of the components (additive), greater than the sum of the components (synergistic) or less than the sum of the components (antagonistic). These interactions, combined with natural variability in biological and geochemical parameters, makes establishing cause and effect relationships difficult. Pollutant-induced ecological changes, demonstrated at the community or population level in the field, are the ultimate measures of significant effects.

Ambient aquatic toxicity (biotoxicity) procedures involve exposing representative test organisms to environmental water and measuring gross biological parameters (i.e., growth, reproduction, mortality). Biotoxicity testing has been carried out with waters from the Delta and the Bay. Biotoxicity tests provide insight into the general health of specific water bodies, and have been particularly useful in defining locations where pollutants are present at acutely toxic concentrations.

On the other hand, sublethal effects to organisms, that may affect reproduction or growth, may not be measurable in short-term bioassays, and lower concentrations of pollutants over longer periods might result in more significant population effects than indicated by short-term bioassays. Also, there may be interactive effects of pollutants that are not predictable from bioassays of single compounds or elements. Other traditional methods of study, such as community-level surveys, measurement of pollutants in tissues of organisms, and *in situ* and sediment bioassays may indicate, but cannot by themselves establish, significant system-wide effects of pollutants.

The measurement of sub-lethal changes in organisms taken from the Bay-Delta, combined with the measurement of pollutants in the same organisms, may provide the clearest indication of significant biological effects. In this respect, pollutant-induced biochemical, morphological or genetic changes in field-collected organisms, the so-called "biomarkers" of toxic effects, are receiving more attention and may provide a promising route for evaluating effects in contaminated aquatic systems.

2. Summary of Previous Work

This section is a brief review of the patterns of pollutant distribution within the estuary, the apparent bioavailability of pollutants, and their possible effects. This is a condensation of the earlier review by Phillips (1987). For a more detailed discussion of these topics and extensive references to the scientific literature, the reader is referred to this review. Information regarding the concentrations of pollutants in the Bay-Delta have been summarized in Table 3.

A. SILVER

Concentrations of Ag in the northern part of the Estuary are low, but in the South Bay higher concentrations can be observed, particularly in mollusc tissues (for which the greatest amount of data are available). Concentrations of greater than 20 μg Ag g^{-1} (dry weight) are commonly encountered in various species of molluscs, and seasonal maxima may approach 200 μg Ag g^{-1} in the tissues of *Macoma balthica* near the discharge of the Palo Alto sewage plant. The mudflats near the Palo Alto sewage plant and the vicinity of Redwood Creek have molluscs with the greatest body burdens of this element. High concentrations of silver have also been found in the livers of diving ducks (scaup and scoters). It is not known whether these tissue concentrations in South Bay animals are having a deleterious effect.

B. COPPER

The highest Cu concentrations in the Estuary are seen in both the northern and southern reaches. Major sources of copper to the system are probably the Sacramento River and discharges in the South Bay and the Central Bay. Copper concentrations in *Macoma balthica* are among the highest ever observed in estuarine benthos, and are clearly of concern. Seasonal maxima of Cu bioaccumulation approach 500 μg Cu g^{-1} (dry weight) in *Macoma balthica* collected near Palo Alto. Livers from ducks in the South Bay have been reported to have Cu concentrations in the range of 50 μg g^{-1}. Some investigators have suggested that this element, along with Cd, may be particularly bioavailable within San Francisco Bay. Copper is an element of special concern because it is toxic to many organisms at very low concentrations (low μg L^{-1} range, dissolved). At the same time Cu is an essential trace element and is required by all organisms in small amounts. The range between natural concentrations in sea water and those toxic to marine animals is very small, making it difficult to evaluate allowable concentrations in discharges. There are no known, observed effects of copper on biota within the Bay; however, the fact that this element may be uniquely bioavailable in this part of the Estuary is of concern.

C. SELENIUM

Seleniferous soils in the Central Valley contribute to elevated concentrations of Se in the San Joaquin River Basin, the San Joaquin River

and the Delta. Compared to other sources in the Estuary the Sacramento River is not the major source of Se. The available data show that refinery discharges in the mid-Estuary and in the South Bay are major sources of Se to the Bay. The chemistry of selenium is complex in the environment and within organisms. Selenium often interacts strongly with other elements (e.g., antagonistically with Hg) to affect their toxicity. In combination with a general lack of knowledge in both of these areas, it is difficult to determine the significance of selenium residues in the tissues of Bay-Delta organisms. Low Se concentrations occur in molluscs from the northern part of the South Bay; maximum concentrations occur in samples from the Central Bay and in the extreme South Bay. The greatest enrichment of Se is seen near the Carquinez Straits. The well known deformities of birds in the Kesterson Wildlife Refuge are closely linked to Se contamination of agricultural wastewater.

D. MERCURY

There are few reliable data on Hg concentrations in waters of the Estuary. It is therefore difficult to assess both the major sources of the element, and the patterns of abundance. However, there are more data available on Hg concentrations in sediments and organisms. The sediment data show that there are either ubiquitous sources of Hg to the system, or that Hg from specific, historical sources has become well mixed over time. Higher concentrations of Hg in small creeks may indicate the importance of urban runoff as a source. In bivalve tissues, Hg enrichment was seen in the northern and southern extremes of the Estuary. There are few reliable data on concentrations in fish, but in the catchments of both the South Bay and the Sacramento and San Joaquin Rivers, Food and Drug Administration (FDA) action limits for Hg in fish (0.5 $\mu g\ g^{-1}$) have been exceeded. Some of this mobilization of Hg may have been due to gold mining and ore purification practices during the last century. Due to the potential toxicity of Hg to aquatic organisms, and its extreme toxicity to humans, further study of this element in the Bay-Delta is warranted.

E. CADMIUM

Because of its toxicity to marine life, and the fact that seafood may be a major source in the human diet, cadmium is of particular interest in estuaries such as San Francisco Bay. Measurement of Cd in bivalves provides both an indication of the amounts of this element that are bioavailable and the potential risk to humans consuming the shellfish. Except for elevated Cd concentrations in mollusc samples from the Carquinez Straits, concentrations of Cd measured in several bivalves in the Estuary show a uniform distribution. This is possibly due to natural biogeochemical phenomena associated with salinity changes. One area of the South Bay (Redwood Creek) has consistently shown very high concentrations of Cd (up to 60 $\mu g\ g^{-1}$, dry weight) in oyster tissues. These concentrations are about 4 to 7 times above concentrations found in specimens from Tomales Bay (a non-urbanized reference site). Reported concentrations of Cd in mussels range from approximately 1 to 35 $\mu g\ g^{-1}$ (dry weight) within the Bay. Most areas of the Bay had mussels with Cd concentrations ranging from 2.5 to 12.9 $\mu g\ g^{-1}$ (Risebrough *et al.*, 1978).

Such concentrations are above the median international shellfish standard of 1 $\mu g\ g^{-1}$ (dry weight). However, these concentrations provide no indications of major sources. There is a possibility that Cd within the Estuary may be unusually bioavailable, as tissue concentrations are much higher than might be suspected, based only on sediment concentrations for this element.

F. LEAD

Although lead is of considerable toxicological concern to urban human populations, this is not the case for marine life. Concentrations reported to be toxic to marine organisms are at least several orders of magnitude greater than reliable concentration data reported for sea water (less than 0.1 $\mu g\ L^{-1}$). Reliable data indicate that the concentration range for dissolved Pb in the Central Bay is 0.018 to 0.033 $\mu g\ L^{-1}$ (Gordon, 1980), although elevated concentrations of Pb have been measured in some water samples taken in the extreme South Bay.

The concentrations of Pb in the Bay-Delta approximate those in other large estuaries, and indicate that there is not a widespread problem with Pb contamination. There are, however, local enrichments that give rise to a patchwork of locally elevated Pb concentrations. While concentrations of Pb in the tissues of most bivalves in the Estuary show a slight elevation over those from the reference site in Tomales Bay, high concentrations of Pb have been found in bivalves from Islais Creek, Albany Hills, beneath the Bay Bridge, in Alameda and Oakland Harbors, and in the Redwood Creek area. Extremely high concentrations (36 μg Pb g^{-1} dry weight), were found in *Mytilus edulis* in Islais Creek.

G. ZINC

Zinc shows moderate enrichment in the waters of the northern and southern portions of the Estuary. On an estuary-wide basis, Zn concentrations are somewhat elevated in comparison to the nearby Pacific Ocean. There is little system-wide variation in Zn concentrations in sediments of the Bay-Delta, and most of this Zn is tightly bound in the sediment matrix and is not bioavailable. As may be expected from the water concentration data, there is no apparent system-wide pattern of Zn concentrations in the tissues of bivalves. Except for a few localized areas of elevated concentrations, Zn concentrations in organisms indicate moderate enrichment within the Estuary.

H. CHROMIUM

Chromium has been measured infrequently in the Estuary. Direct measurements of Cr in water and measurements of Cr on suspended particulate matter show that the Sacramento and San Joaquin Rivers are sources of this element to the Estuary. Chromium sources to the Sacramento and San Joaquin rivers may be from acid mine drainage in their drainage basins. Chromium concentrations are high throughout Suisun Bay, probably as

the result of industrial discharges. Concentrations of Cr in sediments from the Estuary (generally above 200 µg Cr g^{-1} dry weight), are higher than most other Pacific Coast areas surveyed by the NOAA Status and Trends Program. The concentrations of Cr in mussels from the central and northern reaches of San Francisco Bay are generally below 1 µg g^{-1} (dry weight). In the South Bay more than a third of the sites surveyed had Cr concentrations in mussels greater than 4 µg g^{-1} (Risebrough *et al.*, 1978). More recent data collected by the California Mussel Watch Program showed no Cr enrichment in the South Bay relative to the North Bay. There are indications of Cr enrichment in the clam *Corbicula* sp. from Antioch and other portions of the lower Delta, as well as in fish livers collected in the San Joaquin and Sacramento Rivers.

I. NICKEL

Nickel is only of moderate toxicity, and toxic concentrations of Ni in water are much greater than the concentrations measured in waters from the Estuary. However, the USEPA guidelines recommend a 4-day maximum concentration of 8.3 µg Ni L^{-1}, and concentrations of dissolved Ni approaching this have been measured in the Estuary. There is a clear gradient of Ni concentration in waters from the Golden Gate into South Bay. These data suggest a source of Ni in the extreme South Bay. Other data show increased Ni concentrations in the northern reach of the Bay, and suggest that Delta outflow is a significant source of Ni, as well as the fact that there are probably other sources within the Bay. There is no pattern of Ni in Estuary sediments. In general, the mean of values for Bay sediments is about 100 µg Ni g^{-1} (dry weight), which is very close to the crustal value of 95 µg g^{-1} in clays. Some experimental data show that only a small portion of the Ni in sediments is available to the biota.

The data on Ni concentrations in biota of the Bay (mainly bivalves) show few clear distribution patterns on a system-wide basis. However, data from the bivalve *Tapes japonica* show enrichment of Ni in the South Bay. This is in direct contrast to Ni accumulation in other bivalve species. As with many other trace elements, the general pattern of tissue enrichment for Ni was patchy, with localized areas (such as Islais Creek) having elevated concentrations.

J. TIN

Tin was considered to be of only minor toxicological concern for aquatic biota until about 1976, when it was discovered that the organic tin (specifically tributyl tin oxide (TBT; TBTO; the active ingredient in some anti-fouling paints), was toxic to aquatic animals at concentrations in the ng L^{-1} range. Goldberg (1987) showed that the concentrations of TBT in Bay and Delta marinas often exceeded 50 ng L^{-1}, a concentration known to be toxic to molluscs. In general, TBT concentrations are lower outside harbors and marinas. There has recently been considerable work on the cycling and microbial degradation of various forms of organic tin, concentrating mainly on the fate of TBT. The use of alkylated tins as an antifouling paint on the hulls of pleasure craft has recently been banned by the California State Legislature.

K. ORGANOCHLORINE COMPOUNDS

Polychlorinated biphenyls (PCBs), and DDT (dichlorodiphenyl-trichloroethane) and its metabolites (DDE, DDD) are among the most persistent organic pollutants released into the aquatic environment by man. They accumulate in the fatty tissues of organisms, and can be found at high concentrations (tens to hundreds of parts per million) in fish, birds and marine mammals. Such concentrations have been shown to interfere with various biochemical reactions, and with hormonal regulation. One common effect of PCBs, DDTs and other lipophilic organic pollutants is interference with normal reproduction. Despite the restrictions placed on the manufacture and use of PCBs, they continue as a problem in the aquatic environment due to their resistance to degradation. Most of the available local data are sediment concentrations; low solubility makes these compounds difficult to measure in water.

Earlier studies indicated that several tributary creeks, as well as certain sewage outfalls, contained PCBs. The current picture may be different due to improvements in sewage treatment over the past 10 years. It is suspected that street runoff is a source of PCBs, because PCBs are transported in the atmosphere and are deposited on the land surface in both wet and dry fallout. Whatever the sources, it is clear from the NOAA Status and Trends data that San Francisco Bay shows significant enrichment of PCBs in sediments. Islais Creek has been identified specifically as an area of PCB contamination, as well as a probable source of PCBs to the surrounding areas of the Bay.

The best data on PCB distribution within the Estuary are measured concentrations in the tissues of mussels and other bivalve molluscs. These data show that PCBs occur throughout the Bay, with some of the higher concentrations occurring in the South Bay. The most likely explanation of sources to the Estuary, given the limitations of the current data, is that there are numerous point and non-point sources at the periphery of the Estuary, and that occasional major spills or discharges may be occurring.

The fish in the Bay-Delta contain high concentrations of PCBs, and there is evidence that PCBs and chemical compounds related to PCBs may be affecting reproductive success in some fish species. Extremely high concentrations of PCBs have been measured in the tissues of harbor seals (*Phoca vitulina*) (up to 12,000 μg PCB g^{-1} lipid in liver). Such concentrations are cause for great concern because of the possible reproductive effects of PCBs. There are also data that link increasing concentrations of PCBs in black-crowned night herons in the South Bay to decreasing embryo weights (Hoffman *et al.*, 1986). Cause and effect studies need to be carried out, especially in Bay-Delta species from higher trophic levels. Such studies are needed to determine if those relatively few PCBs known to have direct health effects are responsible for the observed effects in starry flounder or black-crowned night herons. Other compounds such as dibenzofurans or dioxins could also be responsible for the observed effects.

DDT and DDT derivatives are widely distributed in San Francisco Bay, but do not show strong regional differences indicative of large point sources. Rather, it appears that DDT inputs prior to 1970 have, to a large extent, persisted in the system, and are now well mixed and redistributed. There is some evidence that the current concentrations of DDT in Bay-Delta organisms are cause for concern. Concentrations of DDE greater than 8 ppm (wet wt.) in the eggs of black-crown night herons form 1982 to 1984 are high enough that reproductive impairment might be expected (Ohlendorf and Fleming, 1988). Mean egg shell-thickness of night-heron eggs from San Francisco Bay were significantly less than shell-thickness measured prior to the use of DDT. Shell-thickness was negatively correlated with the concentration of the major DDT metabolite DDE (Ohlendorf and Marois, 1990).

Due to the multitude of uses to which organochlorine compounds have been put in the Estuary and its tributaries (particularly agricultural uses), it is not surprising that residues of persistent chlorinated compounds other than PCBs and DDTs are present in the Estuary. In general, problems with these compounds seem to be few; however, in isolated areas such as the Lauritzen Canal (Richmond), higher concentrations due to local inputs may be cause for concern. Recently, dioxins and dibenzofurans have been identified in the tissues of rainbow trout from the Sacramento River. The extent of this contamination and its significance are yet to be determined.

L. HYDROCARBONS

The accurate analysis and identification of anthropogenic hydrocarbons in the Bay-Delta has, until very recently, presented great problems. This is due mainly to technical difficulties of analysis; separating individual compounds from complex mixtures, correctly identifying them, and determining if they are natural or anthropogenic. There are few reliable data on the input, distribution and concentration of petroleum hydrocarbons in the Bay-Delta. Most of the available data on hydrocarbons are in the form of "oil and grease" measurements, which are generally not well correlated with concentrations of individual petroleum hydrocarbons or PAHs.

The available data show that there are areas of gross hydrocarbon contamination within the Bay; e.g., Islais Creek, Shell Marsh and in the vicinity of some refinery outfalls and fueling docks in northern San Francisco Bay. The data from the NOAA Status and Trends Program, as well as that from the studies of Spies *et al.* (1985) and Chapman *et al.* (1987), show that the San Pablo Bay region generally has less sediment contamination from polynuclear aromatic hydrocarbons (PAHs) than sites in the Central Bay near Berkeley and at Oakland. However, there are indications of considerable spatial variation on small scales. The tissue data cannot be used to make generalizations about petroleum hydrocarbon distribution in the Estuary.

The data suggest that fish from Central Bay have higher concentrations of PAHs and PAH metabolites in their livers than those from San Pablo Bay. Data on the concentrations of mononuclear aromatic hydrocarbons (MAHs) in

striped bass from the Delta reveal a heterogeneous distribution pattern in this highly mobile fish species. It is, at best, unclear if MAHs have a significant effect on this species.

M. BENTHIC COMMUNITIES

As is the case in other estuaries, a large part of the San Francisco Bay is characterized by soft-bottom benthic communities. The benthic habitat in some areas of the Estuary has low numbers of species. This is probably related to the more rigorous demands that lower and fluctuating salinities put on the physiology of most animals. Seasonal as well as year-to-year changes have a strong impact on the pattern of species diversity and organism abundance. In San Francisco Bay, biological interactions among major species of benthic invertebrates are apparently a major cause of changes in organism abundance (Nichols, 1985). Sediment texture (and associated covariables such as organic carbon content, porosity and sulfide content), water depth and, in the case of intertidal organisms, exposure to air, are also factors that are known to change benthic community composition from place to place.

San Francisco Bay has been used extensively for shipping since about 1850. Non-native organisms attached to the hulls of ships and in jettisoned ballast have been a continual source of new species to the Bay-Delta ecosystem. In fact, about 95% of the benthic invertebrates in the Bay originated from elsewhere. Native species apparently do not compete well against these cosmopolitan fauna, and in certain areas of the Bay "exotic" species have become dominant in the benthos (e.g., *Corbicula fluminea*, *Potamocorbula* sp.). That these invasions are still occurring, and may be dramatic and rapid, is evident from the current invasion of the asiatic bivalve *Potamocorbula amurensis* . From its first detection in 1987 in Suisun Bay, this organism has spread rapidly into the Carquinez Straits, and is now the dominant benthic invertebrate in these areas (Carlton *et al.*, unpublished).

Prior to the upgrading of point source discharges to the Bay-Delta, Filice (1954a,b; 1959) documented alterations of benthic communities for hundreds of meters around such outfalls. Changes of this nature are still evident in some enclosed waterways such as Islais Creek (Chapman *et al.*, 1986).

Differentiating anthropogenic effects from the other sources of variability has been a stumbling block in the useful application of community surveys for the assessment of the health of the Bay-Delta, especially in regions removed from the immediate vicinity of point sources. In such localities, effects of discharges on community structure account for less variability, and those from natural sources more. Thus, distinguishing pollutant effects becomes problematical, both in terms of the numbers of samples required to establish statistically valid relationships, and the need to account for other sources of variability through more vigilant measurement of natural factors. These difficulties increasingly lead marine scientists to look towards alternative approaches for detecting possible effects of anthropogenic chemicals. Since the late 1970s efforts to evaluate impacts on benthic communities have

departed from the classical approach of measuring species diversity. The difficulty of measuring the effects of pollutants on benthic communities in many places in the Bay by traditional benthic survey methods can probably be best summed up by a recent quote: " ...the effects of periodically elevated concentrations of pollutants on the behavior and survival of individuals or populations on this mudflat are not yet distinguishable from natural variation" (Nichols and Thompson, 1985). Instead of traditional survey methods, emphasis has been placed upon detecting subtle changes in physiological and biochemical parameters in organisms exposed to low concentrations of such substances, or upon the development of multi-faceted approaches to measurement of benthic community characteristics.

N. FISHERIES

Phillips (1987) summarized the status of fisheries stocks that were in decline, and examined the link between fisheries declines and pollutant effects. There is some evidence of decline in the stocks of chinook salmon (winter and spring runs) , striped bass, oysters and Dungeness crab (*Cancer magister*). However, as discussed above, the roles of various natural and anthropogenic factors in these declines are yet to be determined. Unequivocal evidence does not exist for population level effects of anthropogenic chemicals upon any fish stock in this or any other estuary in the world.

The effects of pollutants on striped bass and starry flounder from the Estuary have been investigated. The striped bass fishery has declined by approximately 25% since 1959 (Stevens *et al.*, 1985). This has been documented in various indices of abundance (such as the index of young-of-the-year striped bass), and calculations have been based upon data gathered by the California Department of Fish and Game. Earlier fisheries data suggest that declines in landings started about 1900 and extended to about 1930 (Smith and Kato, 1979). These early declines are less well documented than the declines recorded since 1959. It is interesting to note that the fishery in Chesapeake Bay on the Atlantic Coast has also been in decline since about 1960.

The major factors that may have played a role in the decline of the striped bass population are increasing diversion of freshwater from the head of the Estuary, disease, and fishing pressure (Stevens *et al.*, 1985; Brown et al., 1988). Decreasing freshwater inflow and increasing diversion of freshwater from the Estuary have been correlated with the young-of-the-year striped bass index; freshwater flow may be linked to the survival of early life history stages of this species. Unfortunately, fisheries biologists do not yet understand how such decreases translate into year class strength. There is not usually a correlation between abundance of early life history stages and young-of-the-year in fish stocks. This may be due to the influence of poorly understood factors in the ecology of larval fish: it is suspected, for instance, that at low larval fish abundances potential predators concentrate on other prey, while the reverse may be true at higher abundances.

The relative importance of major controlling factors on striped bass populations and how they interact under various environmental conditions is not known. Year-to-year differences in the striped bass abundance indices make it difficult to test existing hypotheses. However, it has been suggested that pollutants may be affecting striped bass populations by reducing reproductive success and causing low survival of larval and juvenile stages. Certainly the results of statistical tests done on the data base collected by the Cooperative Striped Bass Study indicate this. However, the large number of below-detection values for some of the implicated pollutants, the large number of factors measured relative to the number of individual fishes analyzed, and the probability that unmeasured factors are having a substantial influence on the endpoints all indicate that the final answer to this puzzle is yet to be found.

The second species upon which the effects of pollutants have been investigated in some detail is the starry flounder (*Platichthys stellatus*). Polychlorinated biphenyls and other organic pollutants have been measured in the livers and spawned eggs of flounder collected from various sites, mainly in northern San Pablo Bay and in the vicinity of Berkeley. Captured fish have subsequently been spawned in the laboratory, and the viability and survival of the fertilized eggs and larvae have been assessed. It has been determined that:

1) Fish collected from the Central Bay have higher concentrations of PCBs, higher activities of pollutant-induced hepatic P-450 mixed-function oxidases (MFOs) in their livers, and poorer reproductive success than fish collected from northern San Pablo Bay; and
2) Some measures of poor reproductive success are correlated to hepatic MFO activity in spawning females and PCB concentrations in spawned eggs (see Section IV.D.4. for further discussion of enzyme induction in fishes).

Although correlative in nature, these data implicate organic pollutants such as PCBs and PAHs (which are also inducers of the MFO system) as a cause of poor reproductive success. However, the significance of these observed reproductive effects, if any, on the starry flounder fishery in the Estuary is unknown. It should be mentioned that the PCB concentrations may correlate with other chemical compounds, but because other chemicals are either metabolized or not easily measured, the effects that are here being attributed to PCBs may, in fact, be due to other compounds. Laboratory studies are required in order to distinguish test various hypotheses regarding the responsible toxic compounds.

Pathological conditions in striped bass, English sole and starry flounder have been investigated in relation to their incidence, distribution within the Bay, and their relation to pollutants and waste discharge. The immune system of fishes has been shown to be susceptible to pollutant exposure in the laboratory, and in grossly contaminated environments (Weeks *et al.*, 1986; Weeks and Warriner, 1984, 1986). It is clear from these studies that in some estuarine systems (e.g., the Southern Branch of the Elizabeth River, Virginia) depression of immune response might cause increased disease susceptibility among fish.

However, the degree of histologically detectable lesions and abnormalities, and the incidence of cestode infestation in striped bass from the San Francisco Estuary show no simple relationship to pollutants measured in their tissues (mainly PCBs in liver) (Whipple *et al.*, 1983). In the case of starry flounder, liver and kidney lesions show no clear relationship to hepatic MFO activity (Spies *et al.*, 1985). Fin erosion, a condition common in fishes from contaminated environments, was found in juvenile striped bass and other species near the outfall of the Chevron refinery and was induced by the refinery effluent in laboratory exposures.

An outbreak of shell necrosis in the commercial shrimp *Crangon fanciscorum* occurred in the extreme South Bay during 1984 (Kinnetic Laboratories, 1987). The outbreak was associated with several population changes : 1) recruitment was unusually protracted during the summer of 1983; 2) the mean size of shrimp was reduced and this was most evident in the small mean size of ovigerous females from October 1983 through August 1984; 3) the size of the population (as determined by trawling) was greatly depressed from 1984 through the end of the study in late 1986; and 4) the frequency of parasitism by the isopod *Argeia pugettensis* in 1984 was reduced compared to previous years.

There were extreme environmental perturbations during this period: 1) a major *El Nino* event was occurring off the coast of California, and this was accompanied by high water temperatures and low primary productivity; 2) during the winter of 1982 and the spring of 1983 the Estuary experienced an unusually high and protracted inflow of freshwater, while 1984 was a year of minimal inflow; 3) salinities in the extreme South Bay were very low during the winter of 1982 and spring of 1983; 4) large increases in the concentrations of metals in water (unfiltered samples) occurred in the extreme South Bay during late 1983 and early 1984; these (with approximate maximum concentrations) included Cd (28 $\mu g\ L^{-1}$), Cu (100 $\mu g\ L^{-1}$), Cr (30 $\mu g\ L^{-1}$), Pb (11 $\mu g\ L^{-1}$), Hg (2.9 $\mu g\ L^{-1}$), Ni (40 $\mu g\ L^{-1}$), Ag (8 $\mu g\ L^{-1}$), and Zn (88 $\mu g\ L^{-1}$); and 5) there was a bay-wide elevation of PCB during 1984.

The causes of the shell necrosis and the associated population changes are not known, but several explanations have been advanced. The maximum concentrations of many of the metals are in the range that would indicate that dissolved metals present in these samples perhaps had the potential to impact sensitive aquatic species, including crustaceans. Since many of the metals concentrations were elevated at the same time there is the potential also for synergistic toxic effects. Although shrimp analyzed for metals content show that several metals (e.g., Cd, Pb and Hg) were generally higher in collections from the South Bay, trends between years did not generally reflect the patterns seen in the water data. Another factor is that this species of shrimp spawns in the northern parts of the Bay offshore, outside the Estuary. *C. franciscorum* spawn in the Gulf of the Farallones, but under certain circumstances may also spawn in the South Bay. In the latter case exposure of sensitive life history stages of this species to pollutants may have had a negative effect on part of the population. Unfavorable conditions in the spawning grounds or unfavorable currents during

larval migrations may have also played a role in producing the delayed recruitment and, by implication, the smaller-than-normal mean size of ovigerous females during the 1983-1984 period.

The following information, concerning the human consumption of aquatic organisms, is appropriate to include in this section:

1) The California Department of Health Services employs a standard of 0.5 μg mercury g^{-1} for consumption of sea food. This concentration is exceeded by some species of fish in Clear Lake, the Guadalupe River, and in several Central Valley locations. Health advisories have therefore been issued for the consumption of fish from Clear Lake and for striped bass from the Bay-Delta.
2) Similarly, some fish from the Grassland Area of Merced County contain concentrations exceeding 2.0 μg selenium g^{-1}, and health warnings have been issued for these fish.
3) The median international standard for Cd in shellfish is 1 μg g^{-1}, and this often exceeded by shellfish from the Bay, particularly from shellfish in the vicinity of Redwood Creek in the South Bay. Interestingly, shellfish from Bodega Head have been found to exceed this standard and this is probably due to the persistent upwelling features of the northern California Coast.
4) The Toxic Substances Monitoring Program of the State Water Resources Control Board has shown that fish from various Central Valley locations occasionally exceed Food and Drug Administration's criteria for organochlorines for human consumption.

O. BIRDS

The San Francisco Estuary is a critical staging and wintering area for migratory bird populations of the Pacific Flyway. An average of 693,000 waterfowl, primarily ducks, are counted in the Estuary during the annual mid-winter waterfowl surveys (Takekawa *et al.*, 1988). More than 57% of the diving ducks observed during mid-winter censuses in California are counted in the Estuary (Takekawa *et al.*, 1988). The Estuary is also an important staging area for shorebirds (avocets, stilts, plovers, sandpipers, and phalaropes). In April, 1988, 838,000 shorebirds were counted during the period of the spring migration. This makes the Estuary one of the most valuable shorebird habitats in the Western Hemisphere (Stenzel *et al.*, 1989). The Estuary also provides habitat for other resident and migratory species, including grebes, pelicans, cormorants, herons, egrets, and gulls. All of the waterfowl, shorebirds, and other classes of water birds mentioned above depend on the Estuary's aquatic food web.

Declines in some bird species have been noted. Numbers of canvasbacks (*Aythya valisineria*), for example, have recently fluctuated around critically low levels on the West Coast (Takekawa *et al.*, 1988). However, as with fish species, it is difficult to determine the role of toxicants in such declines. Water birds have been shown to accumulate some pollutants (i.e., selenium,

mercury, DDE, and PCBs) to potentially deleterious concentrations, as described above (Hoffman *et al.*, 1986; White *et al.*, 1987, 1988, 1989; Ohlendorf and Fleming, 1988; Ohlendorf *et al.*, 1988; Ohlendorf and Marois, 1990). Elevated concentrations of selenium have caused the California Department of Health Services to issue advisories on the consumption of greater scaup (*Aythya marila*), lesser scaup (*Aythya affinis*), and surf scoter (*Melanitta perspicillata*) taken from the Bay (CDFG, 1989).

Studies of reproductive impairment provide the strongest evidence to date of toxicant effects on bird populations. Evidence for such effects in the Estuary is limited to the finding of a positive correlation between PCB concentrations in black-crowned night herons (*Nycticorax nycticorax*) with decreasing embryo weights (Hoffman *et al.*, 1986). Extensive study of selenium effects on water birds at the Kesterson Reservoir, located in the San Joaquin Valley about 50 miles south of the Delta, has established a relationship between selenium accumulation and reproductive effects (Ohlendorf, 1989). Many of the water bird species found in the Estuary breed elsewhere (i.e., most waterfowl and shorebirds), making studies of the effects of toxicants on their reproduction problematical. Research is currently underway on the sublethal effects of bioaccumulated toxicants on several species of diving ducks that winter in Suisun Bay (Harry Ohlendorf, USFWS, personal communication). Morphology, histopathology, immune response, and cytochrome P-450 activity will be assessed.

P. MAMMALS

There is no available evidence for chlorinated hydrocarbons having effects upon marine mammals within the Bay, although this problem has not yet been investigated thoroughly. Elevated concentrations of PCBs and DDTs in marine mammals from northern California are a source for concern. In view of the reproductive effects seen in seals from the North Sea (Reijnders, 1986), the potential impact of these pollutants on seal populations in the Estuary should be thoroughly investigated.

Q. BIOASSAYS

Bioassays have significant use as direct means to determine whether a particular waste stream discharge will have toxic effects in the receiving waterbody and to measure the toxic strength of waste. They are used to a lesser extent to test ambient toxicity of water and sediment; sediment bioassays have found use recently for the evaluation of so-called "in-place" pollutants.

The application of bioassay-based evaluations to the routine testing of waste discharges to the Estuary has led to an overall reduction in the potential for toxic effects on the biota due to point-source discharges. Bioassay tests that address the problem of both acute and chronic toxic effects are required of all major dischargers to the Estuary; procedures are in place to implement a program of toxicity reduction to discharges shown to have toxic effects. However, it must be remembered that bioassay testing, including standard

procedures such as the 96-hour acute toxic bioassay used for wastewater monitoring, has numerous problems with respect to predicting wastewater discharge effects in receiving waters. While acute and chronic assays, combined with toxicity reduction evaluations are useful in testing for negative impacts, the following shortcomings should be kept in mind with the use of these simple tools:

1) Often the species used are not sensitive.
2) The physical and chemical conditions affecting toxicity in the test may be significantly different to those in the real estuarine environment.
3) Most assays do not account for additive effects of other pollutants, stress on the test animals, and fluctuating environmental conditions which would be the norm in the field situation.
4) The uptake routes in the assays are generally only from water, while in the field other pathways may also be important.

In recent years there has been a trend away from use of single (and sometimes insensitive) species such as the three-spine stickleback, toward multiple species testing utilizing more sensitive species. There has also been an increasing use of sediment bioassays to gauge the toxicity of sediments, particularly those being considered for dredging and dumping elsewhere within the Bay or offshore. Various bioassays employing molluscs, fish and crustaceans have been used to characterize sediments. Some of these tests have included effects on behavioral, reproductive and developmental processes. Many of the tests with sediments from heavily contaminated, moderately contaminated and lightly contaminated areas of the Bay have shown significant toxic responses. Some data indicate that physical characteristics of the sediments may have significant effects upon bioassay organisms and this needs to be considered in interpreting the assay results. This aspect requires further study.

R. CONCLUSIONS AND RECOMMENDATIONS

The following general conclusions were reached by Phillips (1987):

1) The quality of the existing data base is inadequate for determining the distribution of most pollutants in the Bay-Delta.
2) Some of the constraints on the current knowledge of pollutant distribution are due to analytical technique development. These will improve as the science of environmental chemistry progresses. However, some of the present lack of knowledge is due to poor quality of analytical procedures and inadequate design of monitoring programs.
3) Areas of extreme contamination have not been well studied with respect to sources or effects of the pollutants in place.
4) The studies that have most conclusively shown a link between toxic effects and specific pollutants in the Bay have been correlative in nature. Laboratory studies are needed to demonstrate if such relationships are indeed cause and effect.

5) While additional studies are required to define cause-and-effect relationships between pollutants and effects in the Estuary, there is a considerable body of evidence supporting the conclusion that toxic pollutants in the Estuary are having a negative impact on the biota.

Specific conclusions regarding individual pollutants were as follows:

1) The amounts of silver in the South Bay are significant, and may be having effects upon biota. Additional research is needed on the sources, abundances and toxic effects of silver in this area.
2) Copper and cadmium appear to be unusually bioavailable in the Bay. Additional work is needed to determine the causes of this phenomenon, in view of the regulatory implications of present concentrations of these elements in organisms.
3) There is at present a very poor understanding of the sources of selenium in the Bay-Delta. More study is required.
4) Further studies are needed regarding the biological effects of tin (especially TBT) and other organometals in the Estuary.
5) The environmental chemistry of arsenic in the Estuary is poorly understood, especially with regard to the bioavailability of different arsenic species in the water column and in sediments; additional studies of the various chemical forms of this element in the Estuary are needed.
6) PCBs are probably having an effect upon some species of fish and birds in the Bay-Delta. These may be subtle effects on reproduction which may result in an inability of populations to maintain themselves in this area. More congener-specific studies are needed, since the toxicological effects of PCBs may be related to only a small number of these compounds.

In addition, many more studies are needed on pollutants and classes of pollutants that are known to be present in Estuarine waters, but have not been part of the suite of chemicals analyzed. One class of unstudied toxic compounds that has been identified in the list of pollutants of concern but not included in routine chemical analyses are the polycyclic aromatic hydrocarbons (PAHs). This diverse class of pollutants arises from a number of different sources and has the potential to cause significant harm to aquatic biota. PAHs should be included in all routine pollutant survey and monitoring studies carried out in the Estuary.

3. Bioassays of Sediments and Water from the Bay-Delta

A. SEDIMENT BIOASSAYS

As noted in previous sections of this report, the greater proportion of pollutants that enter the Estuary are associated with particulate matter, and their ultimate fate is deposition in the sediments. Once in sedimentary deposits, pollutants establish a dynamic equilibrium with the water column and may serve as a source of pollutant materials to the water and to the biota. Sediment-sorbed pollutants may also be transported throughout the Estuary along with the large mass of sediment (about 160 million cubic yards per year) that is resuspended as the result of wind and wave action.

Several reports are available with which to estimate the degree of contamination in the sediments of the Estuary; however, there is no data base suitable for mapping sediment pollution in the Estuary (CBE, 1987). The vast majority of sediment pollutant data derives from sampling programs aimed at evaluating sediments slated for dredging and disposal in the Estuary. As a result, sediment pollution data are skewed to higher concentrations; most of the analyses come from urban-industrial areas with high rates of sediment accretion coupled with major sources of pollutant materials from domestic, industrial and non-point source pollutant discharges.

CBE (1987) identified 39 "toxic hot spots" in the Estuary based upon their evaluation of sediment pollution data. Using a variety of health- and bioassay-related criteria, CBE (1987) described hot spots for sediment concentration from all four major sub-embayments. CBE (1987) concluded that urban and industrial pollutant discharges were the greatest sources of sediment contamination, and that the most effective way to prevent further sediment contamination would be to reduce or eliminate pollutants in discharges.

Long *et al.* (1988) summarized the results of a NOAA-sponsored effort to describe the status of sediment contamination in the Estuary. They determined, as did CBE (1987), that pollutants were widespread in sediments throughout the Estuary, but that there was considerable variation and patchiness. Long *et al.* (1988) acknowledged that the patchiness and natural variation of sediment contamination in Estuarine sediments makes it difficult to "piece together an overall picture of [pollutant] trends." The effects of sedimentary pollutants on the biota of the Estuary were generally indecipherable, in that trends suggested by chemical data were, in some cases, not supported by bioassay data and did not allow the development of a "preponderance of evidence" (Long *et al.*, 1988).

Gunther *et al.* (1987, 1990) summarized various sources of pollution to the Estuary based upon available monitoring data. In their analysis they noted that the majority of pollutants entering the Estuary came from riverine inputs, urban runoff and non-urban runoff. Based upon bulk-sediment chemical analysis (Long *et al.*, 1988; Gunther *et al.*, 1987, 1990) the most contaminated portions of the Estuary were near industrial and military facilities, including Islais Creek, Richmond Inner Harbor, Oakland Inner Harbor, Redwood Creek, China

Basin and near the Hunter's Point Naval Station. The available data suggest possible recent decreases in sediment concentrations of cadmium, DDT and PCBs at some sites in the Bay (Long *et al.*, 1988). No such changes are indicated for pollutants such as mercury, copper, lead and chromium. No data exist for assessing spatial or temporal trends for polycyclic aromatic hydrocarbons (PAH); however, PAHs may be assumed to be present at their highest concentrations in the vicinity of urban and industrial centers.

Given the lack of a bay-wide, long-term monitoring program for pollutants in the sediments of the Estuary, it is difficult to determine the extent to which polluted sediments may be affecting beneficial uses. Substantial efforts are currently underway to understand the extent to which sediments from various parts of the Bay-Delta may be toxic to organisms. These studies have utilized sediment bioassays. Most have been carried out either by placing organisms on relatively undisturbed sediments (solid-phase assays), or in elutriates of sediment diluted with seawater (typically either 250 g L^{-1} or 20 g L^{-1}). Organisms are then examined for mortality or sublethal effects and compared to organisms exposed to sediments from control areas after a specified period of time.

The survival of bioassay organisms, the ability to rebury, grow and reproduce, and the formation of abnormalities are used to compare sediments from various areas and make judgements about the appropriateness of specific actions, such as dredging and dredged material disposal. Some bioassay studies have been reviewed recently by Long *et al.* (1988); however, a more recent study (Long and Buchman, 1989), and studies from several proposed dredging projects which included a large number of assays and endpoints, had not been completed at the time of the review of Long *et al.* (1988).

Chapman *et al.* (1986) tested sediments from the Bay with the *Rhepoxinius abronius* (amphipod) assay and the mussel larvae bioassay. Sediments from San Pablo Bay and from an area adjacent to Oakland Harbor were not significantly different in the amphipod bioassay than those from a control site in Washington state, while those from Islais Creek caused significantly greater mortality. Sediments from the latter area had relatively high concentrations of Pb, Hg, Ag, PCBs and PAHs as well as high concentrations of silt and total organic carbon.

Studies to examine the possible consequences of dredging in several areas of the Bay for berthing of the *USS Missouri* also included evaluation of sediments using similar bioassays (ESA, 1987). Sediments collected near Treasure Island, Alameda and Hunter's Point were assayed with several benthic species. Accumulation of pollutants was also assayed in two species of invertebrates exposed to the sediments. There were considerable mortalities in the *Rhepoxinius abronius* and sanddab (*Citharichthys stigmaeus*) assays of Bay sediment samples, and some sediments from Treasure Island and Hunter's Point caused elevated mortalities in oyster larvae. A bivalve, *Macoma nasuta*, and a polychaete worm, *Nephthys caecoides*, did not show significant mortalities in assays of Bay sediments. None of the results of the five assays

correlated well with the results of some other assays. Multiple regression analyses showed that copper and total hydrocarbon concentrations in sediments were correlated with sanddab mortalities (Spies, 1987). The bioavailability of Zn, as measured by tissue accumulation, was correlated with oyster and mussel abnormalities. Quantities of bioavailable Cd correlated with sanddab and amphipod mortality.

Another study of sediment contamination was carried out to assess the suitability of these materials for ocean disposal (EVS, 1988). Sediments were collected from Oakland Inner Harbor and the Alcatraz dredged materials disposal site and solid-phase bioassays were carried out using *Rhepoxinius abronius* , the polychaete *Nephthys caecoides* , and the mysid *Acanthomysis sculpta* . Suspended particulate-phase bioassays were carried out with *Rhepoxinius abronius* , bivalve (mussel and oyster) larvae and the sanddab *Citharichthys stigmaeus* . Results of the assays with harbor sediments were compared to those of sediments from the Alcatraz site and the proposed offshore disposal sites. Bioaccumulation tests were also carried out by placing clams, *Macoma nasuta*, and polychaete worms, *Nephthys caecoides*, in the test sediments for 20 days.

In the suspended particulate bioassays, *C. stigmaeus* and mussel larvae exposed to harbor sediments or ocean sediment showed greater mortality when exposed to some harbor sediments. There were increased mortalities among *R. abronius* and *N. caecoides* when exposed to harbor sediments. *Macoma nasuta* placed in harbor sediments had elevated concentrations of chromium, lead, zinc, and chlorinated pesticides relative to controls. *Nephthys caecoides* showed elevated silver concentrations after some sediment exposures. It should be noted that the detection limits for petroleum hydrocarbons were 5 to 15 $\mu g\ g^{-1}$ tissue (wet weight). In comparisons of the Alcatraz dump site and offshore sediments, significant mortality in the suspended particulate bioassays were seen with *A. sculpta*, *C. stigmaeus* and oyster larvae. For *M. nasuta*, elevated concentrations of lead, DDE, DDD and dieldrin were each observed in one sample out of the four samples tested.

Another study was performed to assess the potential toxicity of dredged sediments from Oakland Inner Harbor (Word *et al.*, 1988). Sediments from 14 stations in Oakland Harbor, from control sites off Point Reyes and from near Sequim Bay, Washington, were used in solid phase bioassays with four species: the clam, *Macoma nasuta*; the polychaete *Nephthys caecoides*; and the amphipods *Rhepoxinius abronius* and *Grandidierella japonica*. Suspended phase bioassays were conducted with three species: the mysid *Acanthomysis sculpta*, the speckled sanddab *Citharichthys stigmaeus*, and larvae of the oyster *Crassostrea gigas*. A large suite of metals and organic compounds were also analyzed in the sediments.

In the solid-phase bioassays, there were no significant differences in survival among controls and the 20 treatments for three of the four species. Significant effects were observed in the solid-phase bioassays for survival of *Rhepoxinius abronius; R. abronius* from control stations had lower mortality

than those exposed to sediments from four stations in Oakland Inner Harbor. In the suspended-particulate bioassays, there were small but significant decreases survival among *Acanthomysis sculpta* and *C. stigmaeus* exposed to the Harbor sediments. There were more survivors and more abnormalities in oyster larvae exposed to three Oakland Harbor sediments relative to those from control areas. There were significant elevations of lead in tissues the clam, *M. nasuta*, relative to controls for two Harbor sediments. There was one Harbor sediment treatment for which there was a significant elevation of chromium in clam tissue.

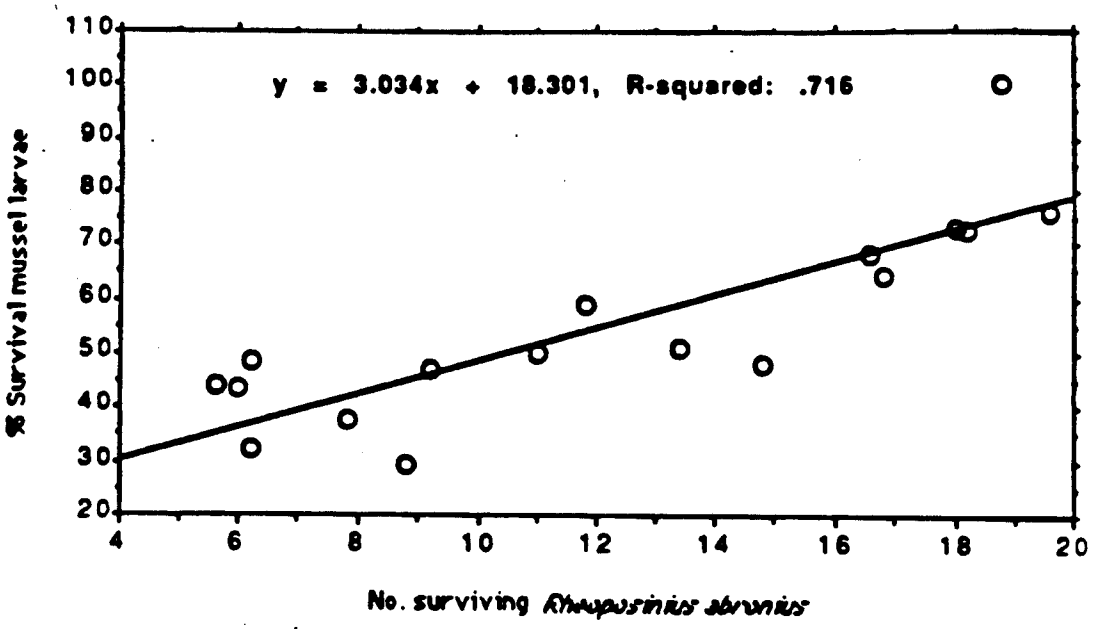
In a NOAA-sponsored study to determine the utility of several bioassays, sediments were compared from Oakland Inner Harbor, near Yerba Buena Island, near Vallejo, from San Pablo Bay and from Tomales Bay (Long and Buchman, 1989). All sites showed reduced survival of *Rhepoxinius abronius* relative to the control from the State of Washington, even those from Tomales Bay, a site selected for its low levels of anthropogenic disturbance. If the control from the State of Washington were to be eliminated, then only two areas with significant differences in amphipod mortality appear in pairwise comparisons of sites: there were more mortalities in Tomales Bay and Oakland sediments than there were in San Pablo Bay sediments. Mussel larvae bioassays with sediment elutriates showed similar results to those of the amphipod bioassays; Oakland and Tomales Bay had the lowest survival. In fact, the results of the two assays agreed well (Figure 26), a relationship that has also been noted in data from Puget Sound (Williams *et al.*, 1986; Chapman *et al.*, 1987).

Different results were obtained with the sea urchin development bioassay. Overall, site explained a significant amount of the variance ($F=3.81$; $P=0.03$), with Tomales Bay sediments causing more abnormal development than sediments from other sites. A number of other endpoints were measured in the urchin bioassays: % eggs fertilized, number of mitoses per embryo, % aberrant mitotic figures, number of cells with micronuclei per 20 embryos and the number of embryos with cytologic abnormalities per 20 embryos.

A new bioassay was also used that determines the reproductive toxicity of sediment pore water to the polychaete worm, *Dinophilus gracilis* (Carr *et al.*, 1986). Female worms growing in Tomales Bay sediment pore water had the highest number of eggs per female (9.97); females grown in pore water from sediments collected at Yerba Buena and Oakland had the lowest number, 5.43 and 6.87, respectively. While the inverse relationship between PAH and fecundity was statistically, significant it is unclear whether the statistical result is representative of a cause-effect relationship between sediment pore-water characteristics and worm fecundity. Such relationships deserve further study.

The sediment samples were also analyzed for trace metals and organic pollutants as well as grain size and total organic carbon. Oakland samples had the highest concentrations of the trace metals silver, cadmium, copper, mercury, lead and zinc. Similarly, total PAHs, DDT and PCB were also highest in the Oakland samples. Oakland and Tomales Bay samples also had the highest percentages of clay and the highest TOC concentrations. Long and Buchman

Figure 26. Relationship between the results of amphipod and mussel larvae bioassays using sediments from San Francisco and Tomales Bays (After Long *et al.*, unpublished).



(1989) reported that the number of physical and chemical variables in this study were very large relative to the number of samples, and that a quantitative interpretation of the data may be difficult. The large number of variables result in the fact that, upon examination, it can be seen that *Rhepoxinius abronius* survival, and mussel larvae survival and abnormalities were not correlated with any of the measured chemical parameters; however, they were correlated with TOC concentrations (Figures 27 and 28) and % clay content.

There are two possible ways to interpret these data. First, one may conclude that the cause of the mortalities in these sediment bioassays were anthropogenic pollutants in the sediments, and that the pollutants covaried with the total organic content of the sediments. Another interpretation might be that the fine-grained sediments, with which TOC covaries, caused the mortalities. The former interpretation does not account for the mortalities observed in the Tomales Bay sediments, but it has been suggested that some unmeasured toxic compounds, perhaps of a natural origin, were present in the samples.

The latter interpretation, that the fine-grained sediments themselves caused the mortality, must be considered as an alternative. A recent study in Puget Sound indicated that *Rhepoxinius abronius* were sensitive to fine-grained sediments, and a method for statistically separating the effects of grain size from other causes of mortality was proposed (DeWitt *et al.*, 1988). Questions such as the relative importance of physical and chemical factors as causes of sediment-related mortality are of extreme importance in the interpretation of sediment bioassay data. Since such data will be employed in drawing conclusions about the suitability of dredged sediments for in-Bay and ocean disposal, it is imperative that additional research and interpretation be applied to questions of sediment toxicity. It is likely that the alternative explanations for sediment-related mortality will vary from sediment-to-sediment at different sites in the Estuary, due in great part to the nature of the pollutant source and the physical characteristics of the sediments at the site. Unfortunately, the data that are currently available are neither numerous enough nor specific enough to lead to unambiguous conclusions, and much more work is required to understand mechanisms of toxicity in sediment bioassays. Bioassays conducted with sediments from stations in South Bay showed that toxicity was coincident with bioaccumulation of silver, cadmium and/or copper in *Macoma balthica* at these sites (Luoma and Phillips, 1988).

B. TOXICITY OF AGRICULTURAL AND URBAN RUNOFF

The potential toxicity of urban and non-urban runoff to streams and estuaries has been a growing concern. Our knowledge of the toxicity of such inputs is just developing. In one of the few studies of the effects of such sources, urban runoff in Massachusetts has been shown to be toxic to benthic invertebrates and to fish exposed to it in the laboratory (Meidros *et al.*, 1983, 1984). In these studies, diversity of benthic invertebrates on solid substrates and production of fish biomass was decreased during periods of heavy snowmelt and precipitation. It is unclear whether urban runoff in Massachusetts is comparable chemically to that in California, where roads are not treated with

Figure 27. Relationship between total organic carbon and survival of *Rhepoxinius abronius* in sediment bioassays carried out with San Francisco Bay sediments. (After Long *et al.*, unpublished).

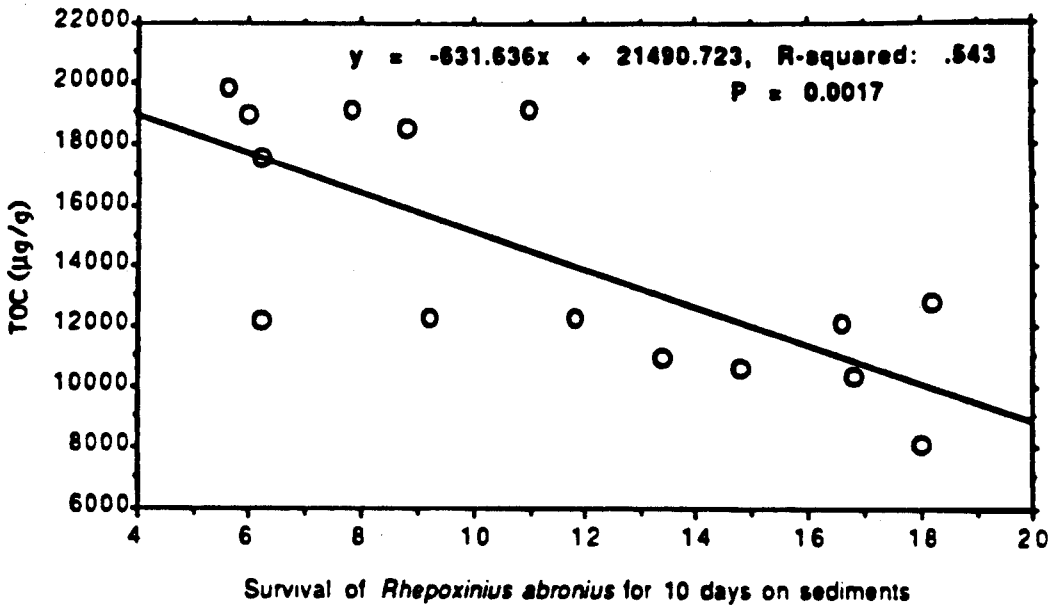
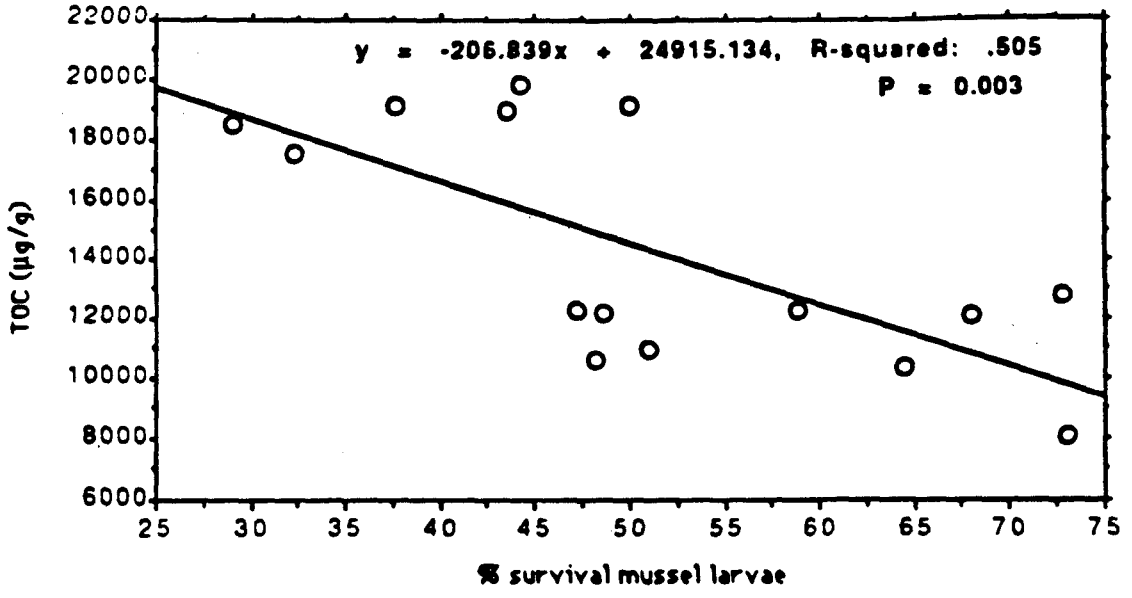


Figure 28. Relationship between total organic carbon and survival of mussel larvae in sediment bioassays carried out with San Francisco Bay sediments. (After Long *et al.*, unpublished).



salt in the winter. Nonetheless, these types of studies could supplement local efforts to understand the toxicity of urban and nonurban runoff.

The Central Valley Regional Water Quality Control Board has been assaying water quality with three species approved for use by the USEPA; the alga *Selenastrum*, the crustacean *Ceriodaphnia* and the fathead minnow *Pimephales promelas* . The work has tested a variety of inputs as well as instream toxicity over a wide geographic area of the Sacramento River, and to a lesser extent, the San Joaquin River. The emphasis has been on testing agricultural drain water, especially that returned to the Sacramento River in May and June each year from rice fields . A study was also conducted on toxicity of water in sumps that received urban runoff from the city of Sacramento and discharged it into the American River.

In the Spring of 1986, samples collected from Butte Slough, Colusa Drain and the Sacramento Slough were found to be acutely toxic to *Ceriodaphnia*. Reproduction of *Ceriodaphnia* was inhibited in some of the samples that were not acutely toxic. In the Colusa drain, detectable amounts of molinate (34 $\mu g\ L^{-1}$), thiobencarb (3.7 $\mu g\ L^{-1}$), chlorpyrifos (1.3 $\mu g\ L^{-1}$) and zinc (22 $\mu g\ L^{-1}$) were found. Significant concentrations of copper were found in many of the other water samples (approximately 5 to 10 $\mu g\ L^{-1}$, unfiltered). The absence of detectable amounts of rice-associated herbicides in drain samples other than from Colusa suggested that other sources of toxicity were also present besides the suggested herbicide toxicity (Shaner, 1986). This study was preliminary in nature, based on single grab samples and without replication of the toxicity tests.

The preliminary study undertaken during the Spring of 1986 was followed by a survey of agricultural drains and river water in a dry period, (November-December, 1986). The results showed little toxicity of water samples from the Colusa Drain, Sacramento Slough and the Feather River (Foe, 1987b). However, *Ceriodaphnia* reproduction was inhibited in the American River. None of the chemical analyses suggested the source of the toxicity.

Foe (1988a) sampled the lower Sacramento River in 1986-1987 in the months of May and June, when agricultural drain water was being released to the Sacramento River. In the Colusa drain, toxicity to fish and invertebrates coincided with the release of agricultural drainage: only the samples from May 27 were toxic. Samples from Sacramento Slough and Feather River were toxic to *Pimephales* larvae, but not to *Ceriodaphnia*.

Foe's sampling in 1988 (Foe, 1988b) was expanded and covered the area from Shasta Dam in the north to Chipps Island in the south. The Sacramento River upstream of Redding appeared to be toxic to invertebrates (data not shown). Colusa Drain water was not toxic to fathead minnows, but experiments with *Ceriodaphnia* showed toxicity that was greatest in the early spring, before the peak application of rice herbicides. Chemical fractionation suggested that the source of toxicity was methyl parathion and carbofuran.

Striped bass larvae were also exposed to the Colusa drain water, and several of the samples caused high mortality.

Similar studies were carried out in the San Joaquin River and its drains in 1987 (Connor, 1988). High mortalities and inhibition of reproduction in *Ceriodaphnia* were found in all samples collected in this survey. Included in the samples were a number of agricultural drains, as well as the main stream of the River. Several stations showed significant fathead minnow toxicity, but there were also many stations showing growth enhancement of the fish.

In an attempt to determine if the toxicity of American River samples might be due to urban runoff from Sacramento, Foe (1986) sampled three sumps that received runoff from areas with different inputs; industrial, residential and "mixed." Samples from all three drains were toxic to *Ceriodaphnia* at 100% and 40% full strength. None were toxic to fathead minnows. After a rainstorm (12 mm of precipitation), samples from the American River were toxic to fathead minnow larvae. Since the three sumps were not toxic to the fish, the source of this toxicity is not clear (Foe, 1986).

These and other results have been used to initiate major programs for the management of pollution in urban runoff and agricultural drainage. The Central Valley RWQCB has found that ambient toxicity testing, used in conjunction with other monitoring strategies, can be a very useful tool for defining toxic agents in the Delta and its major tributaries. Such studies should continue, and should be expanded, as more and more attention is paid the contribution of urban and agricultural runoff to pollution in the Estuary.

C. RECENT STUDIES OF EFFLUENT TOXICITY

The US Environmental Protection Agency (EPA) established a program in 1986 to develop a suite of effluent toxicity test protocols for use on the Pacific Coast. The use of local and more sensitive species than those used in the past (e.g., the three spined stickleback has long been used and is relatively hardy) makes these assays more relevant to current concerns about the degree of protection being provided for the Bay-Delta. The first field trial of these procedures occurred in San Francisco Bay in 1987, following the interest shown in the tests by the San Francisco Bay Regional Water Quality Control Board (SFBRWQB). The study included toxicity bioassays using both marine and freshwater test species, and marked the first field attempts to use EPA toxicity tests with sand dollars, mussels, and kelp. After preliminary screening of candidate effluents, bioassays with marine species were conducted using effluents from the Shell Refinery in Martinez, the City of San Francisco Southeast Water Pollution Control Plant (SFSE), and the East Bay Dischargers Authority combined discharge.

Shell Refinery effluent did not cause observable effects at dilutions of 10% or lower. At 32%, considerable mortality of mysids was the most significant effect, although a smaller but statistically significant reduction in sand-dollar fertilization was also found. No Bay water samples were toxic to the test

organisms. Effluents from the two municipal wastewater treatment plants were toxic; both caused significant reduction in sand dollar fertilization at 10% effluent, but no significant effect was observed at 3.2%. Mussel larvae exposed to the SFSE effluent experienced developmental problems at 32% effluent, but not at 10%. These results suggest that the municipal effluents were about three times as toxic as the Shell effluent, and that the mysid and echinoderm sperm tests were about three times as sensitive as silverside and mussel larvae tests. Tests conducted with kelp (*Laminaria saccharina*) were unsuccessful.

Toxicity tests using *Ceriodaphnia* were conducted from January to June 1987 with thirteen effluents from petroleum refineries, chemical manufacturing plants, and municipal wastewater plants to obtain information for formulating testing plans for the summer of 1987. Two urban creeks were also screened. Summer tests were performed on 10 effluents and 11 creeks that discharge to the Bay. In total, 35 tests with 17 effluents were completed. Toxicity Identification Evaluations (TIEs), investigating the chemicals responsible for the toxicity were also conducted on nine effluents as part of the summer study.

The most toxic effluent was from the city of South San Francisco, with toxicity observed at a concentration of 10% in fathead minnow subchronic assays, lower than that typically observed for municipal effluents. Effluent from Shell Refinery caused toxicity at concentrations of 30% in fathead minnows, *Ceriodaphnia*, and duckweed. Five undiluted effluent samples (from USS Posco, Sunnyvale, Palo Alto, San Jose/Santa Clara, and Dow Chemical) were not toxic to *Ceriodaphnia*. Undiluted water from several urban creeks, including Calabazas Creek, Guadalupe River, San Leandro Creek, Codornices Creek, and Elmhurst Creek caused mortality in *Ceriodaphnia* assays. Calabazas Creek water was also toxic to fathead minnows. An in-depth TIE of effluent from the Central Contra Costa Sanitary District suggested that malathion and diazinon may have been the cause of the toxicity observed for that effluent.

SFBRWQCB (1990) reported data on effluent toxicity testing for ten Estuarine discharge sites. These data showed a substantial range of variation in toxicity (No Observable Effect Concentration; NOEC) as percent of effluent initial water concentration. In addition, the results of effluent toxicity testing among different test species may vary in both magnitude and direction, making the results of toxicity testing rather difficult to interpret. Still, these studies showed that discharges were toxic even after dilution with ambient waters, and therefore support the usefulness of adequately designed bioassay procedures in pollution control strategies

4. Biological Indicators of Sublethal Effects upon Organisms

There has been increasing interest during the last 10 years in determining what sublethal changes in organisms result from pollution, and how the measurement of such changes can be usefully applied to monitoring. The greatest challenge in this pursuit is to identify changes that can be linked unequivocally to pollution. It is now known, for example, that certain enzymes in fish are induced by exposure to complex mixtures of organic pollutants, and

measurement of elevated concentrations of these enzymes usually indicates pollutant exposure. By contrast, elevated blood cortisol concentrations could result from pollutant exposure, disease, salinity stress, or poor nutrition, and therefore are not always an indicator of pollutant-specific changes. It should be recognized that determining health of organisms from the environment by measuring sublethal effects may employ methods that are still developmental. The following discussion will deal with the more promising measures of sublethal effects, and those which have been successfully used in San Francisco Bay.

One of the most widely used indices of pollutant exposure is the induction of the P-450 enzymes, particularly the monoxygenases. We hasten to point out that the P450 enzyme induction assay is only a measure of pollutant exposure, and does not, of itself, indicate the presence or potential for any particular biologic or physiologic effect. Such an interpretation would be warranted only if increased P450 induction were to be coupled, for example, with direct evidence of effects, such as reproductive failure or the presence of micronuclei in blood cells.

The P450 monooxygenase enzymes exist in invertebrates (Lee, 1981), but they are much easier to measure, and more clearly inducible in vertebrates such as fish. The P-450 enzymes are part of the electron transport system, and are involved in the metabolism of endogenous compounds (steroids, vitamins and fatty acids) and exogenous or xenobiotic compounds, such as polynuclear aromatic hydrocarbons. There are multiple forms of these enzymes, known as isozymes. The isozyme responsible for the epoxidation of PAH is called P-450E or P-450IA1, although other isozymes can also catalyze monooxygenation to some extent. This isozyme is inducible in fishes by some PAHs and some PCBs, as well as a variety of other compounds that are not currently known to be as environmentally significant in the Estuary (e.g., polybrominated biphenyls). Species-specific studies of fishes from pristine areas (using monoclonal antibodies) indicate no immunochemically detectable amounts of P-450IA1, but in urban areas, strong induction of the enzyme has been observed (Stegeman *et al.*, 1987; Spies and Stegeman, unpublished data). Increases in monooxygenase activities or certain dealkylating activities in fish tissues are also good measures of exposure to PCBs and PAHs (Spies *et al.*, 1982, 1988). There is also now a cDNA probe for the mRNA that mediates increased protein synthesis (Heilman, 1988).

The measurement of P-450 induction in San Francisco Bay has mainly been performed in studies of starry flounder (Spies *et al.*, 1988; Spies and Rice, 1988; Long and Buchman, 1989; Spies and Stegeman, unpublished data). Increased aryl hydrocarbon hydroxylase (AHH) activities (a monooxygenase activity measured by the conversion of benzo(*a*)pyrene to its 3-OH metabolite by liver microsomes) have been found in central Bay fish relative to those captured in San Pablo Bay. Increases in a dealkylating activity, ethoxyresorufin O-deethylase (EROD), have also been observed in fish captured in the Berkeley and Oakland areas compared with both those captured further up the Estuary, and with fish captured on the open coast (near the Russian River and Santa

Cruz). EROD activity has been correlated with concentrations of PCBs in the liver of starry flounder, which may indicate a causal relationship; or that other chemicals with similar characteristics (e.g., dibenzo-furans and dioxins) may be responsible.

High levels of P-450 induction, combined with exposure to PAHs, result in increased binding of certain PAH metabolites to DNA. Such chemical lesions may lead to carcinogenesis in some species. For example, English sole from the contaminated portions of Puget Sound show increased incidence of liver and bile duct carcinomas (Malins *et al.*, 1984; Becker *et al.*, 1987). Such fish also have increased amounts of DNA adducts (Varanasi *et al.*, 1988). There are no known fish species in San Francisco Bay that have high incidences of hepatomas or other neoplastic disease. Starry flounder have incidences of neoplasms in liver and pancreas less than 2% (Spies *et al.*, 1985). The external papillomas on English sole observed in San Francisco Bay (Cooper and Keller, 1969) are the result of infections by X-cells (which are protozoans) and are probably not directly linked to chemical contamination.

Certain precautions have to be taken when using P-450 enzymes as indicators of pollutant exposure as their activities are suppressed during certain portions of the reproductive cycle, particularly in females during oogenesis. In order not to draw inappropriate conclusions, comparisons should be made, wherever possible, between the same sexes and maturity groups at the same times of year.

Evidence for reproductive toxicity caused by induced P-450 enzymes has been obtained for starry flounder in San Francisco Bay. The mixed-function oxidase (MFO) activity of spawning females was inversely correlated with the proportion of viable eggs spawned, fertilization success and percent normal embryological success (Spies and Rice, 1988). Although the mechanisms of this toxicity are not known, the data strongly suggest that the presence of sufficient amounts of inducer compounds in starry flounder have a negative effect on reproduction. Thus, a subcellular biochemical response specific to organic pollutants and measurable in exposed wild populations can be related to a physiological process that has potential ecological significance.

Further evidence that reproductive impairment due to contamination can occur in wild populations of fish has been obtained in a study of female white croaker from southern California. Fish from the area of Los Angeles harbor exhibit greater pre-ovulatory oocyte degeneration, decreased egg production and lower fertilization rates than fish from less contaminated areas. Fish with greater than 3 $\mu g\ g^{-1}$ DDT were found to be reproductively impaired (SCCWRP, 1988).

Binding of PAH metabolites and other chemicals to DNA, as well as other kinds of chemically induced genetic damage, can be measured in a number of different ways. Adducts on DNA can be detected by the ^{32}P-postlabelling technique (Reddy *et al.*, 1984), or by changes in fluorescence in DNA preparations (Sanders *et al.*, 1985). Breaks in a single strand of DNA can be

measured in a simple assay, and dose response relationships have been found between PAH exposure and degree of strand breakage in two species of freshwater fish (Shugart, 1988). For example, the larvae of kelp bass exposed to microlayer samples from urban areas near Los Angeles contained higher numbers of chromosomal abnormalities, which could be correlated with PAH concentrations in the samples (Hose *et al.*, 1989).

Some chromosomal damage also results in the production of extranuclear bodies called micronuclei. These bodies are readily observed in stained blood smears. Bay-wide elevations of micronuclei occurrence have been documented in red blood cells of starry flounder relative to those from fish collected at two sites on the open coast (Hose, Cross and Spies, unpublished data). Elevated occurrences of red blood cell micronuclei have also been observed in fish near Los Angeles (Hose *et al.*, 1987). However, genetic damage is not as pollutant-specific as P-450 enzyme induction, since both metals and many organic compounds are known clastogens and mutagens. There are also spontaneous rates of genetic damage in wild populations of organisms that must be accounted for in comparative field studies.

Exposure to pollutants can also cause damage to the immune system in fishes, compromising an organism's ability to resist infectious agents. Fish have both phagocytic-based and antibody-based components in their immune system, although only the former system has been investigated in any detail relative to pollutant effects. Alterations in macrophage response have been documented in several fish species from a creosote-contaminated river in Virginia (Weeks and Warinner, 1984, 1986; Weeks *et al.*, 1986). Aggregates of liver macrophages have also been proposed as a measurable response to contamination (e.g., by Wolke *et al.*, 1985).

Histopathological abnormalities in fishes and some invertebrates from contaminated environments have probably been used as often as any other measure of sublethal impact. There is little doubt that exposure to pollutants can lead to disease, and to lesions that are either grossly or microscopically observable. It can be difficult in field populations, however, to distinguish between natural causes of various diseases and lesions, and those that are caused by anthropogenic activities and pollutants (Sindermann, 1983, 1988). In some instances, the increased prevalence of certain types of disease in heavily contaminated areas implicates pollutants as causes (Cross, 1985; Hargis and Zwerner, 1988; Malins *et al.*, 1984; Murchelano and Wolke, 1985). In moderately contaminated areas, such as portions of San Francisco Bay, it may be very difficult to assess whether increased incidences of disease are related to contamination, because there are other natural factors which may play a role. Thus, there were no simple relationships found between disease occurrence in starry flounder collected from several locations in Central San Francisco Bay and occurrence of organic pollutants or MFO activity (Spies *et al.*, 1985).

Monitoring carried out under the National Status and Trends Program by NOAA includes measurements of liver and kidney lesions in starry flounder at three sites in San Francisco Bay. Incidences of liver lesions in starry flounder

from a number of sites on the Pacific Coast are uniformly low. Lesions in flounder from sites in San Francisco Bay were no different from those at coastal sites without much urban contamination. Renal lesions, however were significantly elevated at the Southampton Shoal and Hunter's Point sites in San Francisco Bay relative to the other coastal sites in the program (Varanasi *et al.*, 1989). Earlier studies of starry flounder histopathology showed a similar trend in renal lesions in fish from the Oakland site, although there were relatively few analyses of kidney tissue in that program (Spies *et al.*, 1985).

Metallothioniens are metal-binding proteins that may be involved in regulation of metal metabolism within cells. They bind the transition metals, including cadmium, copper and zinc. Metallothioniens tend to increase in organisms exposed to high concentrations of these metals; they have been proposed as an indicator of metal exposure. One hypothesis is that as metallothionien capacity for binding metals is exceeded, increased intracellular concentrations of free metal may result in enzyme toxicity. Johansson *et al.* (1986) studied the cytosolic distribution of metals, including those bound to metallothionien-like proteins (MLP) in *Macoma balthica* from the Estuary. Increases in copper, zinc and silver on low molecular weight protein accompanied increases in these metals in the MLP. Based on data from other organisms increases in metals in the low molecular weight protein pool may indicate incipient toxicity (Sanders and Jenkins, 1984). The increased resistance of some estuarine organisms to metals (Klerks and Levinton, undated; Klerks and Weis, 1987) may be due to the expression of multiple copies of the metallothionien gene. A population of *M. balthica* near Palo Alto is 50 times more tolerant as those from other portions of the Estuary (Luoma *et al.*, 1983), indicating adaptation of this nature may be occurring within the Bay. However, adaptation or increased "tolerance" may come at a price, in the form of lost energy, placing the organism at a competetive disadvantage.

Many scientists investigating these proteins feel that not enough is known of the factors that play a role in metallothionien regulation (besides metal exposure) or the complex dynamics of metals within various species to use concentrations of metallothioneins as indicators of metal exposure and toxicity (e.g., see Engel, 1988).

5. Synoptic Approaches to Determining the Impact of Pollutants

In recent years, two approaches for determining the effects of pollutants in sediments upon benthic communities have been developed and applied to San Francisco Bay. These are the triad approach (Chapman *et al.*, 1987) and the apparent effects threshold (AET) (PTI, 1988). Both of these approaches rely on the combination of three types of data to draw inferences about contamination effects: concentrations of chemicals in sediments, results of sediment bioassays, and analyses of animal distributions in sediments.

The triad approach argues that, increasing toxicity of sediment in bioassays, increasing concentrations of pollutants, and a systematic decrease of benthic fauna from site to site, taken together, provide evidence for increasing

pollutant effects (Chapman *et al.*, 1987). The hypothesis is that these three end-points are all independent measures of sediment quality, and, when all three move in the direction of increasing toxicity, this is sufficient evidence that pollutants in sediments are having significant effects.

In the introductory paper on the triad approach, data from three sites in San Francisco Bay were used to illustrate the concept (Chapman *et al.*, 1987). Increases in mortality of mussel larvae and *Rhepoxinius abronius* exposed to sediments from these localities were observed in the order: San Pablo Bay, Oakland Outer Harbor, and Islais Creek.

In the AET, chemical analyses, bioassays and, sometimes, community analyses are performed on sediments sampled from a large number of stations in one geographic region (PTI, 1988). An empirical determination is made of effects of one substance by finding a subset of these stations for which adverse effects are always associated with concentrations above the "threshold".

Both the AET and the sediment triad are used to test hypotheses regarding the effects of contaminated sediments on marine and estuarine communities. They are being used widely to draw conclusions about the effects of pollutants in sediments. Recently, Spies (1989) has raised questions as to whether there may be alternative explanations for the interpretations made using these approaches. In particular, Spies (1989) raised the possibility that some of the toxicity observed in the bioassays, and some of community changes described, might be due to the effects of increasing silt, clay and organic carbon content of sediments, rather than the accompanying increases in pollutants. It is beyond the scope of this present review to present all of the arguments on either side of this issue, but controversy surrounds the interpretation of data from these synoptic methods as they are presently applied.

6. Review of Potential Cause and Effect Relationships, and Recommendations for Further Information Needed for Management of Pollutants

The following information suggests that pollutants may be having significant effects on San Francisco Bay and in the Delta:

1) Certain creeks and rivers in the Bay-Delta basin, and some Bay sediments, are significantly toxic in bioassays (see text above).
2) There are decreases in numbers of individuals and species of benthic invertebrates in some highly contaminated areas.
3) There is correlative evidence that reproduction of starry flounder is negatively affected by PCBs and induction of mixed function oxidase activity.
4) Extremely high concentrations of silver and copper have been found in clams near a South Bay municipal wastewater discharge.
5) There is correlative evidence that reproductive success of black-crowned night herons is negatively related to concentrations of PCBs in eggs. Concentrations of DDE in the eggs of this same

species may be high enough in some individuals to be causing reproductive failure.

In most of these cases cause and effect relationships have not been established between pollutant concentrations and the observed effects. Nonetheless, the strength of the observational data allow one to draw the inference that the effects observed are, undeniably, related to pollutants present in the environment. We review several examples below, to demonstrate where doubts may arise regarding the relationship between pollutants and effects on the ecosystem, and how complex the interpretation of environmental phenomena can be.

In the case of benthic community data it should be borne in mind that species richness and abundance can respond to the concentration of organic matter in sediment. It therefore may be difficult to conclude that metals and organic pollutants or other factors are causing decreases in benthic community parameters in areas of fine-grain, high-organic-content sediments. But it should also be remembered that sediment toxicity tests were never intended to provide unequivocal answers by themselves. Instead, sediment toxicity data are used most effectively to identify field sites where sediment contamination may be a problem. Causality will always be difficult to establish from environmental sampling simply because the natural environment is so complex. While bioassay methods may be improved and refined in the future, they will remain indicators of potential effect rather than unequivocal demonstrations of cause and effect.

In the case of correlations of PCBs with reproductive effects in fish and birds, other chemicals that may be behaving similarly but were not measured; e.g., PAH metabolites, dibenzofurans or dioxins, may be responsible for the observed effects.

The high concentrations of silver and copper found in the tissues of bivalves in parts of the South Bay are cause for concern. Bivalve molluscs in certain portions of the Estuary accumulate very high concentrations of metals, and some metals, such as copper and cadmium, may be uniquely bioavailable in the San Francisco Bay ecosystem (Luoma and Phillips, 1988). It is impossible to conclude a cause and effect relationship between metals accumulation and biological impact at this time; however, as with changes in species richness (above), and as with induction of detoxifying enzyme systems (below) accumulation of pollutants must be regarded as indicative of response to an altered and stressed environment. More detailed investigations of the physiological ecology of the benthos, particularly the molluscs, are warranted in order to determine the metals are imposing direct effects, or if some Bay-Delta biota are adapting to high concentrations of metals (Klerks and Weis, 1987).

In general we have a rudimentary understanding of the mechanisms whereby elevated concentrations of some pollutants observed in the Bay-Delta are exercising their effects. The evidence is strongest for organic pollutants.

The elevated occurrences of micronuclei in red blood cells of starry flounder from throughout the Bay suggest an effect of genotoxic chemicals, but the exact cause of this is not known (Spies *et al.*, 1990). Organochlorines are not suspected in this case as they are not generally observed to cause chromosomal damage. Studies of sublethal effects of pollutants on processes such as reproduction in field-collected animals can certainly increase our understanding as to whether the problems identified in starry flounder and black-crowned night herons are representative of other species.

Studies of pollutant effects using benthic community analysis have produced results that can be interpreted in different ways. It is unlikely that the current controversy over the interpretation of benthic community analysis and bioassay results will be resolved with a generalized, universal hypothesis. However, it can be proposed that organic enrichment of sediments and alterations in particle-size distribution in sedimentary deposits might be considered a pollution impact as important as enrichment of sediments with metallic or organic pollutants. As analytical and evaluative methods evolve further, benthic community analysis will certainly come to produce data useful in ecological analysis of all pollutant effects in the Estuary, including organic enrichment and substrate alteration. Experimental manipulations of sediments in which organisms are allowed to colonize and grow with controlled physical and geochemical properties and additions of known amounts of pollutants will, undoubtedly, come to be very useful, particularly when we develop an improved understanding of pollutant interactions with sediment chemical characteristics, including the speciation of elemental pollutants such as arsenic, selenium, and chromium.

Another line of evidence explores the implications of "hot spots" near sources and in quiet waters. Species that prefer or require relatively undisturbed bottom habitat may experience greater toxic exposures in "hot spots," leaving them at a competitive disadvantage (CBE, 1987). The sum of exposures to many pollutants in "hot spots" may select against larger, longer-lived species most useful to recreation and the economy (Luoma and Cloern, 1982).

Although they may not be immediately useful for identifying environmental effects, chemical and isotopic tracers in Bay-Delta studies could help identify sources of some pollutants or the influence of different sorts of discharges in areas with mixed sources. There are several sorts of chemical tracers in sewage, such as the fecal steroids (e.g., coprostanol) and detergent components (i.e., the linear alklybenzenes) that could be useful for tracing sewage discharges (Eganhouse *et al.*, 1988). Also spores of the bacterium *Clostridium* provide a record in the sediment of the influence of sewage discharges (Emerson and Cabelli, 1982). A potential tracer for street runoff has been identified in Bay-Delta sediments (Spies *et al.*, 1987).

E. GAPS IN KNOWLEDGE AND UNDERSTANDING

This section has summarized our current knowledge of the loading, fate, and effects of pollutants in the San Francisco Estuary. At the same time the analysis points out many deficiencies in our knowledge about pollutants as well as in our understanding of the ecology of the Estuary. This section presents a summary of our understanding of pollutants in the Estuary and highlights those areas where our knowledge and understanding are lacking.

1. Historical Trends

Drastic declines in commercial fisheries occurred in the early 1900s. The extent to which pollutants from the rapidly growing human population contributed to these declines is unknown. It is known that the biota of the Estuary were challenged by the introduction of exotic species, including, but certainly not limited to striped bass, oysters, soft-shell clams, *Corbicula*, and *Sinocalanus*. At the same time, local and exotic species may have been overfished, conversion of tidal wetlands to dry land was occurring, and diversion of fresh water from the Estuary began to occur in the mid-1900s.

The filling of wetlands in the Estuary has continued, as have the introduction of exotic species (e.g., the recent appearance of the clam *Potamocorbula*) and water diversion. Each of these factors might have significant effects on the abundance and diversity of biota in the Estuary. Such effects might be equal to, or more important than, the effects of pollutants. A case in point is the rapid growth of populations of *Potamocorbula* in the Estuary, and the remarkable filtering capacity of these organisms which, according to some investigators (S. Luoma, USGS, personal communication), could be responsible for the removal of a large fraction of the phytoplankton population in the northern reach of the Estuary.

Historical trends in pollutant loads from urban and nonurban runoff are unknown. It may be assumed, based upon historic data from other coastal cities, that pollutant loading from all sources increased in proportion to the human population, at least until the 1950s, when controls began to be imposed. These controls, which have been applied primarily to municipal and industrial effluents, have resulted in substantial reductions in certain pollutant loads in the last 30 years.

2. Abundance and Distribution of Pollutants of Concern

There exists a substantial data base on metals discharged to the Estuary in municipal and industrial effluents. These data have made possible reasonable estimates of pollutant loads to the Estuary from effluent discharges. However, the data describing the extent to which such discharges have affected the concentration of metals in the water column of the Estuary are less reliable. This is due to the fact that, until very recently, the methods for low-level detection and quantification of metals from water samples were inadequate.

Very few data are available on concentrations of organic pollutants, either in discharges or in the water column of the Estuary.

Methods are currently available that provide reliable measurements of metals and organic pollutants in natural surface waters. Given this understanding, it is critical that we increase our knowledge of pollutant abundance and distribution in the Estuary by increased application of these methods.

Studies carried out in the field and in the laboratory make it clear that the toxicity and bioaccumulation of organic and metallic pollutants are affected by chemical state; metals, in particular, behave differently in the environment depending upon pH, Eh, their valence state, their association with organic radicals, and salinity. While our understanding of the importance of chemical speciation in aquatic toxicology is adequate, our knowledge of speciation in the Estuary is lacking. The few data that are available regarding the speciation of pollutants in the Estuary (e.g., for selenium) make it clear that much additional research is needed in this area. Knowledge of chemical speciation in the Estuary is lacking for tin, copper, cadmium, mercury, and virtually all organic pollutants; methods for making the needed measurements are available and should be applied in future studies.

While metals have been well studied in the Estuary, our knowledge and understanding of organic pollutants is poor. The few data available on organic pollutant abundance and distribution from the Estuary suggest that the sources and behavior of organic pollutants should be rather predictable. That is to say, laboratory studies and monitoring studies in other estuarine systems make possible the development of reasonable predictive models for the estuarine distribution and transport of PCBs, PAHs, chlorinated pesticides and other organic pollutant classes. By gathering specific information on organic pollutants in the water, sediments, and biota of the Estuary it should be possible to apply existing models to the system, thereby making it possible to control and manage the impact of organic pollution in the system.

3. Pollutant Loads

A. MUNICIPAL AND INDUSTRIAL EFFLUENTS

Quantitative data for organic pollutants in effluent discharges are virtually non-existent. As noted above, our understanding of how these classes of pollutants should behave in estuarine systems is more than adequate; however, this understanding cannot be applied in the absence of specific data. In addition, very few analyses have been performed to determine the speciation of trace metals and organic pollutants in effluent discharges.

Trace elements are the best-quantified pollutants measured in effluents. However, the data base suffers from small numbers of samples and a low sampling frequency. The data for metals loading to the Estuary can be improved in several ways: (1) sampling frequently enough to characterize seasonal and

annual trends in loads, (2) the application of quality assurance testing and reporting, and (3) the use of analytical methods with lower detection limits relative to concentrations present in effluent. It would be particularly valuable for the largest dischargers, a small number of which contribute the majority of loads to the Estuary, to adopt these improvements.

B. URBAN AND NONURBAN RUNOFF

There is a near complete lack of information on pollutant concentrations in urban runoff to the Estuary. Data on loads of organic pollutants of concern are virtually non-existent. Our developing understanding of the importance of urban runoff as a source of organic pollutants also demonstrates that certain critical cognate data are lacking. For example, runoff coefficients for various land use types need further refinement, and empirical determinations would be informative. Studies of the influence of antecedent dry periods on runoff coefficients for open lands are also needed.

Interestingly, dry season flows of urban runoff appear to be significant. Evidence of the dry-season runoff contribution to urban runoff loads in Sacramento suggests that the magnitude of dry-season flows throughout the Estuary should be investigated, and their associated pollutant loads must be better defined before estimates of urban runoff can be considered accurate.

Pollutant concentrations in nonurban runoff have been measured only for agricultural drainage, and even these data are limited. There are almost no data on concentrations of organic pollutants in nonurban runoff. Considering the wide spectrum and large mass of pesticides used in the drainage of the Estuary, additional knowledge of pollutant concentrations is critical.

Efforts to model loads of pollutants from nonurban runoff are also hampered by a lack of data, including trace element concentrations in soils, soil moisture, and other parameters. Field verification of these models is needed if they are to be used in support of management activities.

C. RIVERINE LOADS

Very few data are available on mass transport of pollutants of concern by the Sacramento River, in spite of the fact that it is the source of 80% of the freshwater inflow to the Estuary and probably carries relatively large loads of pollutants. Loads of pesticides and other organic pollutants of concern have not been assessed in either the San Joaquin or Sacramento rivers. Ambient toxicity testing of riverine waters should be implemented in conjunction with monitoring and other management activities.

D. DREDGING AND DREDGED MATERIAL DISPOSAL

It is well known that dredging activities have the potential to mobilize pollutants, primarily as the result of the loss of particulate matter during dredging and the transport of slowly settling particles away from disposal sites.

Existing information provides an insufficient basis for quantitative analysis of pollutant mobilization due to dredging and disposal of dredged material in the Estuary. The large mass of pollutants associated with disposed dredged material in the Estuary cannot be evaluated as to real or potential impact until the extent to which these pollutants are distributed within the Estuary and their bioavailability is known.

E. ADDITIONAL INPUTS

Estimates of loads from other minor inputs, including atmospheric deposition, spills, marine vessel discharges, and waste disposal site leachates, are uncertain. Hydrocarbon loads from atmospheric deposition may be significant, and periodic releases of hydrocarbons in spills are significant on a local, and perhaps regional, scale.

4. Fate of Pollutants

A paucity of data regarding the distribution and abundance of pollutants in water, sediment, and biota limits our present understanding of the fate of pollutants in the Estuary. Generic fate models have been developed for pollutants in freshwater, estuarine, and oceanic systems. These models may be applied to the San Francisco Estuary provided sufficient data have been accumulated.

Our understanding of the importance of equilibrium partitioning of pollutants in aquatic systems is more than adequate for the purpose of applying generalized models. However, there are insufficient data from the Estuary at present even for the application of general models. The extent to which pollutant partitioning is determined by physicochemical and environmental factors specific to the Estuary and biological processes unique to the Estuary is not known and must be determined.

Complex patterns of circulation in the Estuary impede estimation of residence times of pollutants. Residence times are likely to vary significantly within each embayment. Accurate estimates of residence times will depend on an improved understanding of the effects of wind, tides, and freshwater inflows, particularly in the broad, shallow reaches of the Estuary.

Since many pollutants of concern partition to particles, the influence of processes such as flocculation, deposition, and erosion on their fate and transport must be determined. The highly dynamic nature of the Estuary and complexity of these processes constrains the development of predictive models of particle transport.

Although several pollutants accumulate in biota of the Estuary (including copper, mercury, nickel, selenium, silver, certain pesticides, and PCBs), few data on tissue concentrations are available, particularly in light of the spatial and temporal variation demonstrated by the few existing datasets. Comprehensive data regarding the abundance and distribution of pollutants in

water, sediment, and tissue are sorely needed. Accumulation of pollutants of concern in upper-trophic-level species, and the possible contribution of food-web transfer to this phenomenon require further study.

5. Pollutant Effects on Beneficial Uses

Areas of extreme contamination have not been thoroughly studied with respect to sources or effects of the pollutants in place. Studies that have established links between toxic effects and specific pollutants (e.g., between PCBs and reproductive effects in fish and birds) in the Estuary have been correlative in nature. Laboratory studies are needed to demonstrate if such relationships are indeed cause and effect.

The effect of extraneous variables on the results of sediment bioassays, bioassays of water, and benthic community analyses confounds interpretation of results. The influence of such variables needs to be better characterized. High concentrations of silver and copper found in tissues of bivalves in some areas of the South Bay are cause for concern. It is not known, however, whether these pollutants are exerting toxic effects.

Some pollutants, such as PCBs and DDT, have declined rapidly in the Estuary because their use has been restricted; i.e., trends in pollutant loading follow trends in their use, rather than trends in treatment level or populaton growth. It is reasonable to conclude that loads to the Estuary of other pollutants, including some metals, can also be reduced due to declining use. When the use of a pollutant declines it will reduce the potential discharge of the pollutant to the Estuary, independent of population growth.

V. FUTURE TRENDS, POLLUTION CONTROL, AND POLLUTION PREVENTION IN THE ESTUARY

This section begins with a discussion of factors influencing future trends in the abundance and effects of pollutants in the Estuary. The assessment is qualitative because the relationships between these factors and pollutant behavior are not known with sufficient precision to allow meaningful quantitative analysis. The second part of this section presents a summary of strategies that have been, are, or might be employed to control or reduce pollution, and its impact, in the San Francisco Estuary. Many of the strategies discussed in this section were considered by the Subcommittee on Pollutants in the San Francisco Estuary for inclusion in their preparation of Goals, Short-term Management Actions, and Long-Term Management Options. The Goals, Actions, and Options for pollutants in the Estuary are presented in Appendix IV.

A. FUTURE TRENDS

A number of factors influence pollutant abundance and effects in the Estuary. The most important factors influencing long-term trends are:

1) population growth;
2) the use and disposal of pollutant-containing products (such as pesticides, petroleum products, and household products);
3) industrial processes;
4) treatment technologies;
5) land use in the basin;
6) climate; and
7) diversion of water from the Estuary.

Future trends in pollutant loads to the Estuary will be determined in large part by the first five factors listed above. Of these, only population growth and land use show patterns that allow quantitative estimates of future trends. Such estimates are of limited utility in quantitative assessment of future trends in pollutant loads for two primary reasons: 1) the relationship between pollutant loads and population growth or land use changes is not understood with any precision; and 2) other factors listed may have an overriding influence on long-term trends.

Climate and water diversion both influence pollutant transport in the Estuary. The nature of their influence, however, is not precisely known. Although long-term projections of water export rates have been made, quantitative assessment of the influence of these projected activities on circulation and pollutant transport is constrained by a lack of knowledge of the relationship between Delta hydrology and the fate of pollutants in the Estuary. Changes in either the amount or timing of diversion could influence pollutant cycling in the Estuary. Discharges of pollutants are predicted to increase from POTWs due to a combination of increasing restrictions on land disposal of hazardous waste and the Domestic Sewage Exclusion that allows disposal of hazardous waste to public sewers (USEPA, 1988b).

1. Municipal and Industrial Discharges

Population growth affects pollutant mass by producing larger volumes of domestic sewage and household wastes. A relationship between increased population and increased municipal effluent flows (and pollutant loads) was seen in the 1950s and 1960s (Section III; Figure 6). The relationship persisted through the 1980s and will continue into the future unless changes in household water use occur. Population growth can be expected to create increases in the volumes of municipal waste water requiring management. The correlation between population and pollutant loads to the Estuary collapsed in the 1970s. The reason for this discontinuity of trend was the introduction of improved treatment technologies (e.g., pretreatment and wastewater treatment). The recent historical record shows that management techniques have replaced population growth as the major factor controlling long-term trends in pollutant loads from municipal discharges.

The central question regarding future trends in municipal loads is whether further reductions in pollution loading can counteract the effect of population growth. Future trends in municipal loads will depend primarily on the implementation of further improvements in wastewater treatment or pollution prevention (see Section V.B. below). The potential for additional, cost-effective reduction of loads is a complex topic that must be evaluated carefully as the growing population generates larger and larger volumes of waste water. In the absence of additional, new control measures, or effective pollution prevention, loads from municipal effluents will go up along with increases in wastewater flows.

Unlike municipal discharges, pollutant loads from industrial discharges are not related directly to the same factors (i.e., population growth) that create continual increases in the volumes of waste for treatment. At present, significant improvements in industrial wastewater treatment and source reduction are being attained by Bay Area refineries, some of the largest industrial dischargers in the Estuary (again, this topic is discussed in detail in Section V.B.). Adoption of these techniques will cause a significant reduction in the long-term loading from these dischargers. These efforts indicate that the potential for substantial reductions of present loads carried by industrial waste waters still exists.

2. Urban and Nonurban Runoff

Future trends in the loading of pollutants to the Estuary from urban and non-urban runoff are uncertain. These loads will be influenced by a variety of factors, including changes in land use, population, climate, agricultural practices, transportation fuels, and pollution control regulations. Estimating the importance of these factors in the future is difficult; no firm prediction of future trends for pollutant loads from urban and non-urban runoff is possible.

Land use influences the volume of runoff and the mix of pollutants scoured or leached from surfaces and carried into the Estuary. All things being equal, changing land-use patterns can be used to describe some basic future

trends in urban and nonurban loads. Substitution of urban for agricultural land will increase urban runoff volumes while decreasing agricultural runoff. In the *Status and Trends Report on Land Use* (Perkins *et al.*, 1990), projections for land use in the basin in the year 2005 have been made using a variety of assumptions regarding population growth and economic trends in the region. These projections can provide an indication of the type of runoff reaching the Bay and Delta in the future. The reader is referred to Perkins *et al.* (1990) for a discussion of the limitations of the methods used to make these projections.

As discussed in Section IV.B.2 and by Gunther *et al.* (1987), the magnitude of current pollutant loads to the Estuary from urban and nonurban runoff is very uncertain due to our lack of knowledge regarding pollutant concentrations in runoff, runoff coefficients, trace metal concentrations in soils, and sediment yield from erosion of nonurban land, pesticide usage, and other factors. Indeed, of the many parameters needed to calculate urban and nonurban loads to the Estuary, land use is among the most accurately known. It would therefore not be worthwhile to estimate the magnitude of loads for 2005 using new land-use projections until some of the other factors used in these calculations are better described.

Perkins *et al.* (1990) project a 25% increase in urban land for the Bay-Delta, and a 50% increase for the Central Valley between 1985 and 2005 (aggregate data for the entire watershed are presented in Table 18). This represents urbanization of nearly 249,000 ha. The projections indicate that about one-third of this increase will be due to the conversion of cropland and pastureland. The balance will be due to the development of rangeland and forests. About 60% of the projected increase in urban land area (145,000 ha) is expected to occur for residential land use, and an increase of about 28% is projected for industrial land uses, including utilities (Perkins *et al.*, 1990).

It might be expected that certain urban land uses (e.g., industrial sites) would result in greater loading of pollutants *via* runoff. This has been demonstrated for composite organic parameters such as oil and grease (Stenstrom *et al.*, 1984), petroleum hydrocarbons (Hoffman *et al.*, 1983), and total PAHs (Hoffman *et al.*, 1984). There is also evidence for the site specificity of urban runoff; however, attempts to make broad predictions based upon land use have yielded conflicting results. The NURP (Nationwide Urban Runoff Program) examined 81 sites around the country and found no apparent relationship between land use and pollutant loads. Hoffman *et al.* (1984) found significantly higher concentrations of PAHs in runoff from industrial sites as compared to residential sites; there was no difference between commercial and residential locations.

As discussed by Gunther *et al.* (1987), these site-specific differences reflect the broad variation among sites within the same land-use category. For example, residential sites in the the NURP study varied from 6 to 76% coverage by impervious surfaces, indicating significant potential for variation in runoff characteristics (USEPA, 1983). Some industrial sites may be mostly impervious.

Table 18. Projected land use in the catchment of the San Francisco Bay Estuary, 1985-2005. All areas in hectares, after Perkins *et al.*, (1989).

Land Use	Area in 1985	Percent of Total	Predicted Area in 2005	Change from 1985 to 2005	Percent Change 1985 to 2005
TOTAL URBAN	671,582	4.2	920,389	248,807	37.0
Residential	362,976	2.3	508,421	145,445	40.1
Commercial	97,705	0.6	131,960	34,255	35.1
Industrial	69,183	0.4	98,008	28,825	41.7
Utilities	58,251	0.4	80,056	21,805	37.4
Mixed Comm/Ind	3,578	0.0	3,710	132	3.7
Mixed Res/Comm	13,960	0.1	20,391	6,431	46.1
Other Urban	65,937	0.4	77,857	11,920	18.1
TOTAL AGR	4,034,072	25.1	3,930,170	-103,902	-2.6
Crops & Pasture	3,357,073	20.9	3,274,146	-82,927	-2.5
Orchards, etc.	657,969	4.1	637,635	-20,334	-3.1
Feedlots	9,247	0.1	8,916	- 331	-3.6
Farmsteads	9,783	0.1	9,473	-310	-3.2
TOTAL RANGE	3,280,102	20.4	3,223,066	-57,036	-1.7
TOTAL FOREST	7,467,612	46.5	7,401,194	-66,418	-0.9
Deciduous	483,949	3.0	476,434	-7,515	-1.6
Evergreen	6,346,616	39.5	6,302,113	-44,503	-0.7
Mixed	637,047	4.0	632,292	-4,755	-0.7
TOTAL BARREN	110,351	0.7	103,520	-6,831	-6.2
TOTALS	16,067,242	100 0	16,067,24	0	0 0

while others may contain significant amounts of open land (industrial "parks"). Although the variation in land-use data can be reduced by using more categories and carefully selecting and grouping study sites, any aggregation of urban land uses will undoubtedly demonstrate variability in runoff characteristics and pollutant concentrations. This suggests that extensive local surveys of pollutant concentrations in runoff are important, and predictions using land-use changes and data from other regions will be relatively uncertain.

The data suggest that the loading of petroleum-derived hydrocarbons (including PAHs) and trace metals (particularly lead and zinc) to the Estuary in urban runoff should rise as lands are converted to urban uses. The documented "first flush" effect will produce larger pulses of pollutants as greater proportions of local drainages are urbanized. There is currently significant regulatory focus upon urban runoff loads of pollutants to the Estuary, and actions to reduce loads from this currently uncontrolled input could have a significant impact upon future loads.

Nearly half the total land that will be converted to urban uses by 2005 is currently in agricultural use, mostly as "cropland and pasture." All other things being equal, reducing the amount of agricultural land should reduce the loading of certain pesticides and fertilizers to the Estuary. Furthermore, the loads of several trace elements from agricultural lands should also decline. Perkins *et al.* (1990) predict that only 3% of the agricultural land in the drainage basin will be converted to urban uses. The small change in pollutant loading from agricultural sources would probably be undetectable, especially when one considers the large year-to-year changes that can occur in precipitation, planted acreage for particular crops, and pesticide application. Furthermore, changes in agricultural practices, including more efficient chemical application and soil conservation techniques, could significantly influence loads from agricultural lands. On a local scale a much larger proportion of agricultural lands might be converted to urban uses, and this could result in detectable changes in pollutant loading to the Estuary from these altered drainages.

It is interesting to note that areas such as the western San Joaquin Valley, which contain selenium-rich soils, are not the areas where urban development is expected to undergo a large increase. As long as farming continues on this land, agricultural drainage waters containing relatively high levels of selenium can be expected to flow into the San Joaquin River drainage. There are programs in place to reduce Se discharge to the San Joaquin system.

The remaining land that is expected to be converted to urban land by the year 2005 totals about 130,000 ha and comprises rangeland (23% of the total conversion), forests (27%), and sparsely vegetated land (3%) (Perkins *et al.*, 1990). Pollutant loads to the Estuary in nonurban runoff estimated by NOAA (1988) show that rangeland and pastureland are subject to substantial erosion and can contribute significant loads of trace elements. Because the loss of rangeland represents less than 2% of the rangeland in the basin, however, it is unlikely that significant reductions in loadings from these lands will occur by the

year 2005. Land-use changes on a local scale could have a more significant impact.

Other inputs examined in this report include atmospheric deposition, vessel wastes, spills, and leaching from waste sites. Future trends for the loading of pollutants from these inputs are difficult to project and are less likely to be influenced directly by changes in land-use patterns. The relative magnitude of these inputs as compared to urban and nonurban runoff indicate their contribution to future loads will probably be minimal, except for potential local impacts.

Population growth will increase the amount of pollutants discharged to the atmosphere, although future regulatory changes, fuel mixes, and vehicle miles traveled could have a large impact on these trends. The extent to which increased emissions will result in greater atmospheric deposition of pollutants to the Estuary is unclear, because much of the predicted new development will occur farther away from the Bay and Delta (the reader should remember that atmospheric deposition refers only to direct deposition to water).

With an increase in population, the number of commercial marine vessels and pleasure craft operating in the Estuary should also increase. It is unlikely that this will be proportional to the increase in population, however, as many waterways have already been developed for boating, shipping, and sailing. San Francisco Bay has lost some commercial shipping business to Southern California in recent years, but it is not clear if this trend will continue in the future. Increasing the number of houseboats and pleasure boats in the Estuary could result in a small increase in nutrient and/or bacterial loading. It is not clear if such an increase would be large enough to be detected.

It is unknown whether the rate of marine accidents or land-based spills will rise. Previous analysis of trends in spills clearly indicates that annual loads from spills are heavily influenced by the occurrence of infrequent, large-volume events. Examples include the recent spill from the Shell Oil refinery complex in Martinez, and the collision of two oil tankers under the Golden Gate bridge in 1971. Implementation of aggressive spill-prevention policies and techniques could result in an overall decline in pollution due to spills, particularly if prevention techniques eliminate or reduce the frequency of those few, but infrequent, large-volume spills.

A ban on land disposal of untreated hazardous wastes will be in effect by 1990 (40 CFR 268). Therefore, it is anticipated that there will be no new hazardous waste disposal sites. However, hazardous waste treatment and storage facilities, as well as repositories for residuals, will be constructed in the region during this period. Operation of these facilities will probably result in accidental spills or leaks that could contribute to pollutant loading to the Estuary. While a growing population will require greater solid waste landfill capacity, more stringent regulations on solid waste disposal sites in the future will reduce the potential for leakage to estuarine waters even with an increase in the number of sites constructed.

In summary, land-use projections suggest that urban development in the basin of the Estuary will continue over the next two decades. This will cause an increase in pollutant loading from urban runoff, although the actual magnitude of future loads cannot be accurately predicted. Of particular concern in relation to urban runoff loads are petroleum hydrocarbons (including PAHs) and the trace metals zinc and lead. Current regulatory attention to this urban runoff might significantly reduce future loads from this source.

B. POLLUTION CONTROL AND POLLUTION PREVENTION

Water quality objectives are a central feature of the existing approach to regulation of pollutants in the Estuary. Key elements of this approach are 1) definition of beneficial uses to be protected, 2) establishment of limits on ambient concentrations (water quality objectives) to protect such uses, 3) the control of pollutant inputs, and 4) monitoring to determine compliance with established limits. This Section discusses the implementation of water quality objectives in the Estuary and the scientific basis for the present regulatory approach. Other options for regulating the effects of pollutants, including sediment and tissue objectives, are then briefly discussed. The section concludes with an examination of alternatives for reducing pollutant loads (pollution prevention) and the effects of pollutants on the Estuary.

It has been stated that treatment-control-management-disposal options cannot completely eliminate persistent pollutants. Indeed, such an approach may cause contamination of air or land and transfer pollutants from effluents to runoff sources. The Congressional Office of Technology Assessment (1986) and others have described the process of pollution prevention as a management strategy. Pollution prevention is discussed in detail later in this section. In general, management options should direct control efforts toward those pollutant inputs that, once controlled, would have the greatest impact on reducing pollutant effects in the Estuary.

1. Regulatory Tools

A. THE PRESENT APPROACH

Background

The Federal *Clean Water Act* and State *Porter-Cologne Water Quality Control Act* require that the Regional Boards establish water quality objectives to protect beneficial uses in California waters. The Basin Plans developed by each Regional Board (SFBRWQCB, 1986; CVRWQCB, 1989) assign beneficial uses (e.g., "municipal and domestic supply," "water contact recreation," or "wildlife habitat") to all waters within each basin. The Basin Plans also specify water quality objectives intended to protect designated beneficial uses. These objectives are subject to approval by USEPA.

Water quality objectives can either apply to specific parameters (numeric objectives) or to general characteristics of the water body (narrative objectives). Numeric objectives specify concentrations of pollutants that are not to be exceeded in ambient waters. Narrative objectives are stated in more general terms. An example of a narrative objective is a requirement present in both the Central Valley Basin Plan and the San Francisco Bay Basin Plan that all waters must remain free of toxic substances in concentrations producing detrimental effects upon aquatic organisms. Site-specific numeric objectives may also be developed for segments of the Estuary. Such objectives would be based on evidence of unusual water chemistry or sensitivity in local populations, or a demonstration that all inputs in a segment are controlled to reasonable levels.

Water quality objectives are achieved primarily through the establishment and enforcement of effluent limitations. Since the inception of the NPDES program in 1972, effluent limitations have been based primarily on pollutant concentrations that can be achieved using available treatment technologies. Application of such "technology-based" effluent limitations has resulted in dramatic reductions in loads of conventional and toxic pollutants from effluent discharges, as discussed in Section III. Technology-based effluent limitations do not, however, eliminate the possibility that discharges exert adverse effects in receiving waters (this topic is discussed further below).

Implementation of Water Quality Objectives in the Estuary

1. Existing Objectives

Numeric objectives for toxic pollutants in waters of the Estuary are listed in Table 3 (Section IV.A.). Generally, the Regional Boards adopt criteria developed by USEPA that are based upon comprehensive compilations of toxicity testing results. Criteria are derived for acute (1-hr) and chronic (96-hr) exposures in marine and freshwater environments. The most important narrative objective relating to the pollutants considered in this report is the requirement, mentioned above, that waters of the Estuary remain free of effects due to toxicants.

Numeric objectives exist for only some of the pollutants thought to be of greatest concern in the Estuary (listed in Table 3, Section IV.A.). Notably absent from Table 3 are objectives for several pollutants of particular concern, including tributyltin and selenium. The SFBRWQCB has adopted a criterion for PAHs that applies to waters downstream of the Carquinez Strait. Too few data exist, however, for the establishment of a freshwater criterion (SFBRWQCB, 1986). At present there are no objectives for other toxic organics, although many of these compounds are thought to be of particular concern in the Estuary (including chlordane and its metabolites, DDT and its metabolites, PCBs, and toxaphene) because of their occurrence at potentially damaging concentrations in sediment and biota, as discussed in Section IV.D.

Numeric objectives for many of the USEPA-designated priority pollutants (a classification that includes most of the toxic organics considered in this

report) are currently being developed by the State and Regional Boards under provisions of Section 303 of the *Clean Water Act*. This work is expected to be completed by summer of 1990. No reliable data exist on the precise concentrations, or even the general abundance, of most toxic organics in waters of the Estuary. The situation is only slightly better regarding concentrations of these compounds in effluents and other pollutant inputs. Attainment of objectives for organic pollutants cannot be assessed until such data are collected.

Site-specific numeric objectives may be appropriate for segments of the Estuary. Many factors should be evaluated in setting site-specific water quality objectives. Among the most important are:

1) the concentrations of bioavailable forms of pollutants in receiving waters,
2) biogeochemical processes governing these concentrations,
3) the sensitivity of resident organisms to acute and chronic exposures of pollutants, and
4) the relationships of body burdens to toxic effects.

Section IV discussed the considerable gaps in our present understanding of these topics.

2. Assessing Attainment of Water Quality Objectives

Analysis of pollutant concentrations in waters of the Estuary is an essential component in a regulatory scheme that employs numeric objectives for specific pollutants. The few reliable data collected to date have been summarized in Table 3 (Section IV.A.). The San Francisco Bay Regional Water Quality Control Board recently initiated a pilot study that will address the need for additional reliable data (T. Mumley, SFBRWQCB, personal communication). Ultra-clean methods are being employed to analyze total and dissolved forms of the nine metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc) for which water quality objectives exist. Samples are being collected at 26 locations in the Bay on four occasions. Conducting this type of monitoring on a routine basis would provide information critical to the management of pollutants through the use of water quality objectives. However, it must be realized that application of "ultra-clean" methods to all samples may be impractical, particularly from an economic point of view. In the long run, and for most samples, it will probably be sufficient to apply updated, clean methods of analysis, with a high degree of quality assurance, to obtain data that are adequate and reliable.

The San Francisco Bay Regional Board classifies the South Bay south of the Dumbarton Bridge as "water quality limited," meaning that water quality objectives are not met after application of typical technology-based effluent limitations on municipal and industrial dischargers. Other segments of the Estuary in addition to South Bay are suspected to be water quality limited relative to toxic pollutants (SFBRWQCB, 1986).

Once water quality limited segments of the Estuary are identified, accurate information on loads from major inputs of pollutants is essential to the cost-effective attainment of numeric objectives in these areas. As discussed in Section IV.B., however, there are significant deficiencies in our understanding of pollutant loads from all of the major input categories. Present estimates of loads from effluent discharges (the best characterized input category) are constrained by the use of insensitive analytical methods to measure pollutant concentrations, a lack of emphasis on quality assurance, and infrequent sampling. There is a lack of a coordinated, long-term effort to assess the chemistry of the Estuary, especially with regard to the effects of chemical pollution on the biota. Until recently, pollutant loads from urban and nonurban runoff could only be estimated using a very general, and highly uncertain, approach. The San Francisco Bay Regional Board has recently addressed some of these deficiencies by requiring more frequent monitoring by municipal and industrial dischargers, and by directing the local agencies in Santa Clara and Alameda counties to characterize urban runoff loads into the South Bay.

Successful application of the narrative objective stipulating that waters of the Estuary remain free of toxic effects due to pollutants requires an understanding of pollutant inputs, the fate of pollutants derived from different sources, spatial and temporal patterns of enrichment, and the effects of pollutants on resident species. As discussed in Section IV.E., investigators around the Bay region are well aware of the data that are needed to assess pollutant loads, but there are considerable deficiencies in our knowledge regarding each of these topics.

An understanding of the abundance, distribution, and bioavailability of pollutants is critical to management of the toxic effects of pollutants in the Estuary. Phillips (1988) reviewed toxic pollutant monitoring in the Estuary, concluding that existing programs provide inadequate representations of temporal and spatial trends in pollutant abundance and bioavailability. An alternative monitoring scheme was proposed, emphasizing the analysis of pollutant levels in "biomonitors" (bivalves) as the central feature of a comprehensive program assessing pollutant abundance on regional and local scales. The use of biomonitors is favored (Phillips, 1988) over sampling of the water column or sediments because the pollutants in animal tissues provide time-integrated measures of bioavailable pollutants in the environment. Biomonitoring also overcomes the difficulties encountered in the analysis of trace organics in water. In recognition of the advantages of biomonitoring, the State Board has sponsored two major biomonitoring programs (the State Mussel Watch and Toxic Substances Monitoring Program). Phillips (1988), however, proposed changes to these programs that would provide improved information on spatial and temporal trends of pollutant levels in the Estuary.

The scheme proposed by Phillips (1988) would also provide valuable information on the relative significance of different pollutant inputs. The exposure of biomonitors to waste streams was argued to offer several advantages over measurement of aqueous pollutant concentrations. Among these are

1) pollutant levels (especially for trace organics) in the biomonitors would be higher, and therefore easier to measure, than aqueous concentrations;
2) bioavailability would be measured directly; and
3) biomonitors would integrate pollutant concentrations over time.

The principal difficulty in interpreting results of biomonitoring programs relates to interspecific variation in bioaccumulation of pollutants. No single species of bivalve tolerates the wide range of salinity found in the Estuary; consequently it is necessary to use more than one species in a system-wide biomonitoring program. In fact, more species than have been tested may be useful for biomonitoring purposes; for example, the recently established exotic clam species, *Potamocorbula*. Whatever species are used, varying bioaccumulation among the species used will constrain comparison of pollutant abundance and distribution in different portions of the Estuary. Other resident species will also bioaccumulate toxicants at different rates than biomonitors.

Section IV.D. described the difficulties in obtaining certain knowledge of significant effects of pollutants upon Bay-Delta organisms, and the weak scientific bases for enforcing the ambient toxicity prohibition present in the Basin Plans. Methods of assessment of pollutant effects were also reviewed. Bioassays of ambient waters and sediment have recently been employed to evaluate the presence of toxic conditions in the Estuary. Sediment bioassays, in their current state of development, may not provide unambiguous measures of pollutant-induced toxicity because of the sensitivity of test organisms to physical characteristics of sediment, such as organic content and grain size. Attempts are being made, however, to overcome these difficulties. Application of synoptic approaches to determining pollutant impacts on benthic organisms, including the "triad" approach and the AET, are being implemented in studies of pollutants in the Estuary. It should be noted, however, that to demand specific cause-and-effect relationships between pollutants and effects in the environment may be unrealistic; as with most studies carried out in a field context, or with samples containing a mixture of pollutants, the best data that may ever be obtained may be inferential rather than conclusive.

Bioassays exposing several species to ambient waters and effluents have been employed by both Regional Boards and have yielded valuable information. As discussed in Section IV.D., the Central Valley Regional Board has studied the toxicity of agricultural drain water and urban runoff. Results from these tests suggest that the waters entering some of the receiving waters of the Central Valley are toxic. The San Francisco Bay Regional Board has performed similar multi-species toxicity testing on effluents released into the Bay. The current Basin Plan for the Bay Region includes effluent limits for acute toxicity, and requires dischargers to conduct studies to determine appropriate chronic bioassay monitoring programs and protocols. These programs promise to provide useful data regarding the toxicities of ambient waters and effluents.

As discussed in Section IV.D., however, laboratory bioassays provide only indirect evidence of potential effects upon resident Bay-Delta species. Pollutants that cause measurable toxicity in bioassays may be diluted or transformed to the extent that toxicity does not occur in the Estuary. In addition, resident species may have different sensitivities than organisms used in the bioassays. It is also impossible to use bioassay results to predict effects due to continuous low-level exposure in the Estuary. The measurement of sub-lethal effects on organisms taken from the Bay-Delta, combined with the analysis of pollutant concentrations in the same organisms, may provide the clearest indication of significant biological effects due to toxic contamination. To date, studies suggesting links between specific pollutants and toxic effects in the Estuary have been correlative in nature. Laboratory studies would be helpful in demonstrating whether such relationships are direct.

3. Limitations of Water Quality Objectives

The principal limitations of the present regulatory approach relate to the extrapolation of laboratory bioassay results to local conditions in the Estuary, and to the potential for gradual pollutant accumulation to harmful levels in aquatic biota. Water quality objectives provide incomplete assurance that the general requirement (present in both Basin Plans) prohibiting the presence in the Estuary of toxic substances at levels that produce detrimental effects is met. As noted above, either a short-term or a long-term program to assess the status of the biota of the Estuary would be of substantial value in evaluating water quality objectives.

The prediction of pollutant effects in the environment based upon data from laboratory bioassays is uncertain for a number of reasons (see Section D.Q). Perhaps the most significant is the possibility of unusual sensitivity (either greater or lesser) of local organisms to toxic effects or unusual local water chemistry. One obvious limitation of the objectives developed by USEPA is that they are derived for marine and freshwater, not estuarine, conditions. Since the toxicity of many pollutants increases as salinity decreases, saltwater objectives (adopted for more saline portions of the Estuary) may not adequately protect some species in the Estuary (SWRCB, 1988b).

Differences exist between the chemical forms of pollutants that are used in laboratory assays and the forms measured for comparison to water quality objectives. Such differences complicate the application of these objectives. Laboratory tests typically expose organisms to simple salts of trace elements, and may add organic chemicals in carrier solvents to encourage dissolution. Pollutant concentrations measured in the field for comparison to water quality objectives, on the other hand, typically are "total" concentrations. A significant fraction of the total pollutant mass in water samples may be sorbed to particles or complexed with organic or inorganic ligands. Complexed forms may not be, and frequently are not, bioavailable; consequently, total pollutant concentrations measured in the field are not directly comparable to concentrations measured in bioassays. Analytical procedures that measure bioavailable forms would be more directly comparable to the data used to

derive water quality objectives; such procedures, however, are not yet available for routine use.

Most significantly, water quality objectives based on laboratory studies of direct toxicity to aquatic organisms do not fully account for gradual pollutant accumulation to potentially damaging levels in aquatic organisms. Many of the pollutants of greatest concern listed in Table 2 (Section IV.A.) are known to bioaccumulate to elevated concentrations in portions of the Estuary, especially the trace elements cadmium, copper, mercury, selenium, and silver and the organochlorines chlordane, DDT, PCBs, and toxaphene. It is possible that a pollutant such as silver, which is found in waters of the Estuary at concentrations below established water quality objectives, may still be exerting toxic effects on aquatic organisms. The present regulatory approach, featuring water quality objectives, provides only partial assurance that toxic substances do not accumulate in the Estuary to levels that produce detrimental effects.

B. OTHER APPROACHES UNDER CONSIDERATION

Mass Emissions Strategy

In 1987 the SWRCB commenced proceedings (known as the "Bay-Delta Hearings") to develop water quality objectives to provide reasonable protection of beneficial uses of the Estuary and consider alternate allocations of water rights to achieve such objectives. In October 1988 the SWRCB staff produced a Draft Pollutant Policy Document (SWRCB, 1988b) that evaluated existing data on pollutants and their effects on beneficial uses and recommended policies for water quality control based on those data. One of the recommendations presented was the pursuit of a "mass emissions strategy" to regulate loads of several pollutants that accumulate to potentially damaging levels in sediments and organisms in the Estuary.

Key elements of the proposed mass emissions strategy include:

1) identification of locations, based on available data, where pollutant concentrations in tissue and sediment are elevated;
2) identification of inputs of pollutants at these locations;
3) imposition of an annual limit on mass emissions from all inputs equal to current loads;
4) development and implementation of a program to reduce loads, including total maximum daily loads that will protect beneficial uses;
5) monitoring of progress; and
6) development of tissue and sediment objectives.

The mass emissions strategy is aimed specifically at regulating the accumulation of cadmium, copper, mercury, silver, selenium, and PAHs in tissue and sediment (SWRCB, 1988b).

Information required for implementation of several of these components is similar to that required for enforcement of present water quality objectives. Two important features of the mass emissions strategy are already being

implemented in the South Bay. The San Francisco Bay Regional Board recently issued NPDES permits for municipal discharges to Lower South Bay that limited mass emissions to current levels (the context of these actions is described in more detail below under "Alternative Methods of Disposal"). Also, the San Francisco Bay Regional Board has initiated studies to better quantify inputs to Lower South Bay as the first step in adopting measures for their control (SFBRWQCB, 1986). The most significant departure from the present regulatory approach included in the proposed mass emissions strategy is the development of criteria for concentrations in sediment and biota.

Sediment quality criteria are receiving considerable attention at the national level (Shea, 1988). Objectives derived from such criteria could prevent impacts due to direct contact of organisms with contaminated sediments and are considered more appropriate than water quality criteria for hydrophobic trace organics. However, no sediment quality criteria exist at present. An intended use of such criteria is in the evaluation of the potential transfer of pollutants to both the water column and the tissues of biota. Sediment quality criteria might be of greatest value in an assessment of the potential for toxic effects due to dredging and disposal of dredged material (see Gunther *et al.* [1990] for a full discussion).

Two approaches to development of sediment quality criteria that have received the greatest attention are based on 1) modeling of equilibrium partitioning of pollutants between sediment, water, and biota; and 2) synoptic approaches relating pollutant concentrations in sediment to biological effects measured using bioassays or benthic population studies. As discussed in Section IV.C.2., the Draft Pollutant Policy Document recommends that apparent effects thresholds (AETs) provide an initial basis for sediment quality objectives. In fact, initial development of AETs for the Bay is already underway.

Objectives that define acceptable levels of contamination in the tissues of Bay-Delta organisms, both in terms of protection of estuarine biota and human consumers of fish, shellfish, and waterfowl, could be more beneficial than sediment quality objectives. Analysis of pollutant concentrations in tissues offers the same advantages as analysis of sediments, but provides information that is significantly more relevant to ultimate toxic effects. Existing U.S. guidelines and a summary of international standards are listed in Table 19, although it should be realized that these guidelines have their limitations. The limits for public health (USFDA, 1984) do not include data for trace elements and are not necessarily protective of aquatic organisms. The limits for predator protection (NAS, 1973) concern only organochlorines and mercury and were developed at a time when the toxicological literature was in its infancy; these guidelines require revision based on more recent work. In addition, limits for predator protection may not protect prey or other species lower in the food chain. The Draft Pollutant Policy Document recommended that tissue alert levels protective of human health be developed in cooperation with the California Department of Health Services. Development of tissue criteria protective of aquatic organisms was also recommended.

Table 19. Concentrations of toxic pollutants in tissues of organisms prescribed as recommended guidelines for predator protection (NAS, 1973) or as action levels for the protection of public health (USFDA, 1984). Additional values for trace elements are also shown, and are derived from median international standards for the protection of human health (Phillips [P.T.], 1988). All data are shown as $\mu g\ g^{-1}$ wet weight of tissues.

Contaminant	Predator protection	Median International Health standard	FDA Action Level
Antimony	-	1.0	-
Arsenic	-	1.4[a]	-
Cadmium	-	0.3	-
Chromium	-	1.0	-
Copper	-	20[b]	-
Lead	-	2.0	-
Mercury	0.5	0.5	1.0[c]
Selenium	-	2.0	-
Zinc	-	45[b]	-
DDT[d]	1.0	-	5.0
PCBs[e]	0.5	-	2.0
Aldrin	0.1[f]	-	0.3
Dieldrin	0.1[f]	-	0.3
Endrin	0.1[f]	-	0.3
Heptachlor	0.1[f]	-	0.3[f]
Heptachlor epoxide	0.1[f]	-	0.1[f]
Hexachlorobenzene	0.1[f]	-	-
Chlordane	0.1[f]	-	0.3
Lindane (γ-HCH)	0.1[f]	-	-
Hexachlorocyclohexane	0.1[f]	-	-
Endosulfan	0.1[f]	-	-
Toxaphene	0.1[f]	-	5.0

[a]Refers to arsenic trioxide (equivalent to 1.0 $\mu g\ g^{-1}$ wet weight as the element)
[b]Will be exceeded by samples of Crassostrea gigas, which bio-accumulate copper and zinc heavily
[c]As methylmercury. CDHS employs a health advisory level of 0.5 $\mu g\ g^{-1}$ wet weight in edible tissues
[d]Total of all isomers present
[e]Termed a tolerance, rather than an action level
[f]Individually or in combination

Wasteload Allocation

A promising approach to water quality regulation, and a goal specified in the Basin Plan for the Bay Region (SFBRWQCB, 1986), is the development of site-specific water quality objectives and corresponding wasteload allocations for receiving waters. In the scenario for wasteload allocation, site-specific objectives would be based upon the unique physical, chemical, and biological characteristics of each segment of the Estuary. Usually water quality objectives derive from the desire to protect the most sensitive species in the receiving water body. Under a wasteload allocation scheme each discharge into a specific segment would be allowed to discharge (i.e., "allocated") a portion of the total load of regulated pollutants calculated as appropriate for discharge. These wasteload allocations would be based upon mathematical models designed to predict concentrations of pollutants in ambient waters under varying environmental conditions (Ambrose *et al.*, 1989).

Wasteload allocation for South Bay is in progress (T. Mumley, SFBRWQCB, personal communication) according to guidelines provided by USEPA (Ambrose *et al.*, 1989). The models upon which wasteload allocation is based require the same information necessary for adequate evaluation of the overall effects of pollutants on biota in the bay; reliable concentration data, chemical speciation data, sediment transport data and additional information on the partitioning of pollutants among compartments of the Estuary.

2. Effluent Discharge Control Strategies

Options for reducing pollutant loads from effluent discharges are divided into three categories for the purpose of this discussion 1) improvements in wastewater treatment, 2) pollution prevention, and 3) alternative methods of disposal.

A. IMPROVEMENTS IN WASTEWATER TREATMENT

Improved Treatment of Final Effluents

As discussed in Section III, tremendous advances have been made in the treatment of effluents discharged into the Estuary since the inception of the NPDES program in the early 1970s. Municipal treatment plants in the Bay-Delta region provide "secondary treatment," a process primarily designed to reduce the high concentrations of biochemical oxygen demand (BOD) and suspended solids in domestic sewage. Several municipal discharges to the Bay (38% of the total volume released) receive additional treatment aimed at reducing the stimulatory effect of nutrients in receiving waters. The physical and biological processes employed in municipal wastewater treatment also reduce loads of pollutants carried by these effluents, as evidenced by the decline in municipal trace element loads from 1975 to 1985 (see Section III). These processes, however, are not designed to provide optimal removal of persistent pollutants (a category that includes many of the pollutants of concern).

Enhanced treatment of trace elements and organics in municipal wastewater will require processes designed specifically to remove these substances. Lime precipitation followed by filtration is a treatment process that could be employed to achieve additional trace element removal (BADA, 1987). Recalcination would allow recovery of the lime, and would greatly reduce the quantities of sludge requiring disposal. The removal efficiency of this process would vary for each element and at each plant. BADA (1987) estimates that lime precipitation could reduce copper loads from major POTWs by 88%, nickel loads by 78%, lead loads by 68%, and zinc loads by only 8% (potential reductions were not estimated for other elements). Trace organics are degraded to some extent in secondary treatment; further reductions in concentrations (particularly for the highly recalcitrant organochlorines such as certain PCB congeners) could be attained by employing carbon adsorption with carbon regeneration at existing plants (BADA, 1987). Concentrations of individual organics in the range of 1 ug L^{-1} are considered achievable. Considerable costs are associated with implementing these options for trace pollutant removal (BADA 1987). In addition, once removed from the wastewater, the toxicants would be transferred to another environmental medium and would still require management and further expense. A careful evaluation of the costs and effectiveness of this strategy relative to other alternatives for reducing trace pollutant loads to the Estuary would be prudent.

Some Bay Area industrial dischargers are developing sophisticated treatment systems, primarily to meet toxicity standards for their effluents established under the current Basin Plan (SFBRWQCB, 1986). Unocal's petroleum refinery in Rodeo, for example, has installed a system with expanded treatment and storage capacity and a state-of-the-art process including the use and regeneration of powdered activated carbon (Unocal, 1989). Other petroleum refineries have also enhanced their treatment systems (T. Mumley, SFBRWQCB, personal communication).

Pretreatment Programs

Pretreatment programs aimed at controlling industrial discharges to municipal treatment facilities began during the mid-1970s. Since the 1970s, USEPA, through the NPDES permit program, has required pretreatment programs to be adopted by most POTWs. These programs are intended to regulate the discharge of toxic pollutants that can interfere with POTW operation, or pass through the treatment system and into receiving waters, and that can limit opportunities for recycling wastewaters and sludges.

According to Federal regulations, two types of standards are imposed on industrial discharges to sewers. "Prohibited discharge standards" are developed by POTWs and apply to all non-domestic indirect discharges. "Categorical pretreatment standards" are developed by USEPA and apply to specific industrial categories. Pretreatment programs are enforced by the POTWs and supervised by the Regional Boards. As discussed in Section III, these programs contributed to large decreases in trace element loads to POTWs in the last 15 yr. Critics of the present program (CBE, 1988) argue that

existing standards are higher than concentrations that can be achieved in a cost-effective manner, and that some significant sources of pollutants may not be regulated.

Toxicity Reduction Evaluations

The current Basin Plan for the Bay Region established the Regional Board's authority to require a "toxicity reduction evaluation" (TRE) in cases where effluent toxicity limits are exceeded. TREs consist of three major components (Hake, 1989):

1. identification of chemicals causing toxicity, or "toxicity identification evaluation" (TIE);
2. identification and development of treatment or control options; and
3. development of an indicator limit (a limit on a particular pollutant or some other parameter that accurately reflects toxic effects).

TREs are a new technique used at a small number of municipal and industrial facilities in the United States. TIEs attempt to determine specific chemicals or chemical groups causing effluent toxicity and to locate the influent waste streams contributing most to the observed toxicity. Development of control options can include performance of pilot scale treatments of the effluent or influent waste streams or implementation of "best management practices" (to optimize existing treatment or reduce volumes of waste generated). Best management practices offer the additional potential benefit of decreasing operating costs. Indicator limits apparently have not been developed in TREs conducted to date.

Hake (1989) evaluated results of TREs conducted in several states, including California, North Carolina, Florida, Virginia, and Maryland. Most of the TREs performed so far have focused on identification of chemicals causing toxicity. Approximately half of the dischargers conducting TREs succeeded in identifying substances causing toxicity and eliminating the problem. Industrial facilities have had far greater success than municipal facilities in identifying and eliminating toxicity. It is still rather early to decide whether TREs will be effective in identifying the sources of toxicity in POTW effluents. The diffuse sources of pollutants within municipal systems could hamper their effectiveness. Municipal plants in industrialized regions must consider a multitude of potential sources. Well-designed pretreatment programs can expedite the location of sources of toxicity in these cases.

B. POLLUTION PREVENTION

Introduction

Over the past two decades, significant progress has been made in controlling pollution. However there are limits to the degree of environmental protection that can be achieved by current regulatory programs that emphasize management after pollutants have been generated. These programs focus on treatment, control, and disposal, and can sometimes result in the transfer of pollutants from one environmental medium to another where they may continue to present a hazard.

Further decreases in pollutant loads can only be achieved by reducing or eliminating discharges or implementing source reduction and environmentally sound recycling practices. The USEPA and several State and local government agencies have adopted pollution prevention policies and have initiated programs that encourage industry to minimize the generation of wastes.

Pollution prevention is any practice or activity that reduces, avoids, or eliminates the generation of wastes. Techniques focus on source reduction or recycling activities that reduce either the volume or toxicity of generated wastes. Actions that are not associated with the waste-generating activity, including waste recycling or treatment of wastes after they are generated, are not considered to be pollution prevention measures. Industry is encouraged to first reduce wastes through input material substitution, process modification, and good housekeeping. Secondarily, pollution prevention encourages the practice of in-process recycling and reuse before treating and disposing of wastes. Specifically, pollution prevention includes the following five classes of activities (OTA, 1986; CBE, 1989):

1) Redesign or reformulation of products,
2) Substitution of raw materials that introduce smaller quantities of hazardous substances into production processes,
3) Improvement of process technology and equipment to alter the primary source of waste generation,
4) Improvement of plant operations (housekeeping), and
5) Recycling of polluted substances at the site of its generation (closed-loop recycling).

Pollution prevention does not include any form of treatment, pretreatment, incineration, managed disposal, or recycling outside of the waste-generating process.

Pollution prevention measures also offer potential means of reducing costs associated with both production and regulation of waste-generating activities. By reducing the generation of waste, industry can realize savings associated with using materials more efficiently. Other economic benefits of pollution prevention include lower costs incurred by both regulatory agencies and regulated parties associated with compliance with environmental regulations. The current level of national spending for pollution control is about $70 billion and increasing; two-thirds of this amount is spent by industry.

Pollution prevention is a practical way to complement this costly pollution control regulatory system (OTA, 1986).

Governmental Programs

1. Federal Programs

USEPA issued a "Pollution Prevention Policy Statement" that calls for a strategy to incorporate the pollution prevention philosophy within every feasible aspect of program planning and decision making. An Office of Pollution Prevention and Planning (OPPP) was established to serve as a focal point for the Agency's multimedia pollution prevention efforts. The goals of USEPA's pollution prevention program are to support State and local pollution prevention programs; develop outreach targeted at State and local governments, industry and the public; create incentives and eliminate barriers to pollution prevention; develop a multimedia clearinghouse to provide educational and technical information; and to collect, disseminate, and analyze data for the purpose of evaluating national progress in multimedia pollution prevention. To encourage innovative pollution prevention activities, USEPA set aside nearly $26 million of its 1991 and 1992 budgets for new, multimedia pollution prevention projects within USEPA's Offices, Regions, and States. Funding has been approved for 26 projects across the nation. USEPA has also allocated $7 million to support State multimedia pollution prevention programs, and additional State funding will be awarded in fiscal year 1990.

The USEPA Office of Research and Development (ORD) has established a technology evaluation program in cooperation with the private sector and with facilities from the Departments of Defense and Energy. ORD has created the Waste Minimization Opportunity Assessment Manual (USEPA, 1988b) and has conducted Waste Minimization Workshops throughout the nation. USEPA also wished to evaluate progress in preventing pollution and to target opportunities for assistance programs by developing a comprehensive data collection strategy. Currently, the Toxic Release Inventory serves as a basis for measuring releases to the environment. USEPA will use other existing data bases, but will weigh the additional burden imposed by data collection with the need for data on pollution prevention activities and programs.

In USEPA's San Francisco office (Region 9), an interdivisional steering committee has been established to promote cross-media approaches to pollution prevention within ongoing programs. A Pollution Prevention Team was also created to implement specific pilot projects within the Bay Area to encourage pollution prevention programs. These projects serve to provide assistance and direction to Contra Costa County and Santa Clara County. The goals of the pilot projects are to ensure that a multimedia focus is developed and maintained and to increase the effectiveness of the Counties' waste minimization programs.

2. State Programs

The SFBRWQCB and CVRWQCB (Regional Boards) have significant potential to promote waste minimization activities in the Bay-Delta region. The Regional Boards can require that industrial dischargers implement waste minimization measures through the NPDES permitting process. Permit applications can specify a schedule for conducting waste assessment evaluations and for identifying waste streams and points of origin and can specify measures for reducing those waste streams. Adoption of additional or more stringent effluent standards may also provide incentives to industrial dischargers to improve effluent quality through innovative waste reduction measures and to identify recycling opportunities. Permits written for municipal discharges can require improvements in pretreatment programs. For example, the San Francisco Bay Regional Board recently reissued permits for Sunnyvale, Palo Alto, and San Jose/Santa Clara requiring waste minimization studies through the pretreatment program. The results of such studies will aid in developing effective programs for waste reduction.

SB-14, a State law recently signed by the Governor of California, requires that generators of hazardous waste prepare a hazardous waste reduction evaluation review and plan, a hazardous waste management performance report, and a report summary documenting hazardous waste management approaches. These requirements must be met by 1 September 1991. This is significant because it is intended to improve coordination among various state agencies regulating specific media (air, water, and solid waste) and to promote a multimedia approach to hazardous waste management issues. SB-14 requirements should be adopted by local governments, not only for hazardous waste generators, but for all industries and for all waste discharges.

3. Local Governmental Programs

Local governments can promote and encourage hazardous waste minimization in several ways, including educational outreach, technical assistance, and regulation. By promoting waste minimization local governments reduce chemical exposure in the community, encourage industrial efficiency, and reduce the need for offsite hazardous waste facilities. The Local Government Commission, a nonprofit organization based in Sacramento, has outlined types of programs that local governments can adopt.

Although educational and technical assistance programs encourage waste minimization, regulatory programs can provide the most significant direct and indirect incentives. Direct incentives include hazardous waste minimization planning requirements, recycling requirements, and employee training requirements. Many local governments already have mechanisms available to adopt and enforce these requirements. Hazardous waste minimization plans could become a component of business plans and the risk management and prevention program already required under the State Law AB 2185/2187 and AB 3777. Other means for requiring waste minimization plans include land

use/building permits, business permits, industrial waste water permits, and air pollution control permits. Certainly, it makes sense to adopt such requirements for new firms coming into the area or firms that are expanding. Other regulatory options include aggressive enforcement of existing pollution control laws, development of more stringent local pollution control laws, and application of pollution control laws to mid- and small-size businesses. Indirect regulatory incentives include modified fee structures, reduced procedural requirements, and reduced fines and penalties.

Pollution prevention can also be achieved through transportation planning. Such planning measures could include 1) elimination of subsidies to single passenger commuter vehicles by incorporating environmental costs (e.g., urban runoff, air quality degradation, habitat destruction, and oil spills) into the cost of this form of travel through tolls, user fees, or taxes, 2) planning residential communities with access to mass transportation, and 3) improving the safety, reliability, and efficiency of mass transportation.

In the San Francisco Bay Area, the Bay Area Hazardous Waste Reduction Committee (BAHWRC) was established to facilitate hazardous waste reduction and to support the development and implementation of local government programs in the Bay Area. Participants include County Health Departments, representatives of POTWs, City Fire Departments, local agencies responsible for implementing hazardous waste and minimization programs, the California Department of Health Services (CDHS), and USEPA, Region 9. The BAHWRC serves as an information exchange forum and has adopted a multimedia philosophy toward waste minimization. BAHWRC is the first step in adopting a plan to implement pollution prevention on a regional level. Planning must continue to include all media programs and to adopt laws, regulations, and programs that eliminate barriers and encourage waste minimization.

Incorporation of the pollution prevention philosophy into decision making and planning will produce ways in which regulatory agencies can induce waste minimization, identify incentives, and help to reduce barriers to waste minimization. In order to avoid the movement of waste from one medium to a less-resistant route, local governments must act jointly and uniformly to encourage waste minimization. This means the air quality districts, waste water programs, and hazardous waste programs must adopt regional plans to identify waste sources and prioritize waste minimization activities. Government agencies can also serve as an example to industry by practicing pollution prevention and improving management of their own wastes. Decision making in future growth and the building of infrastructure must include a serious consideration of pollution prevention to sustain and improve the Bay Area's quality of life.

Municipal treatment plants, as local government agencies, can adopt many of the same educational, technical assistance, and regulatory programs described above. Municipal treatment plants are well suited for development of educational and technical assistance programs. Personnel at these facilities have extensive contact with their regulated industrial community and thorough

understanding of industrial process operations. Municipal treatment plants have the authority to regulate the disposal of industrial wastes into the sanitary sewage system in order to maintain the sewage treatment plant. The regulations and ordinances that regulate industrial waste disposal are meant to maintain the treatment system and to comply with Federal and State regulations. Through the pretreatment program POTWs have the flexibility to adopt regulatory programs that induce industries to minimize their wastes. For example, municipal facilities may adopt mass-based discharge limits in conjunction with concentration-based limits. Mass-based limits encourage industry to reduce wastewater flows, preventing the use of dilution as a compliance mechanism. Mass-based limits also allow firms to better quantify pollutant loads and help to provide incentive to set reduction goals.

A combination of population growth and increased regulatory pressure to reduce toxic pollutant discharges will force municipal treatment plants to establish policies and goals that promote pollution prevention. For example, 304 (l) of the *Clean Water Act* requires that USEPA identify water bodies that will not achieve water quality standards for toxic pollutants. Municipal and industrial dischargers that impact the water bodies are required to develop Individual Control Strategies (ICS). The South Bay is one such listed water body, and the Palo Alto, Sunnyvale, and San Jose/Santa Clara treatment plants must develop ICS. Additionally, because of the upcoming land disposal restrictions, municipalities can expect to see increases in legal and illegal toxic pollutant discharges to their treatment plants. Over 10 million tonnes of hazardous waste are generated each year in California. California's system for managing these wastes for many years relied on land disposal of untreated hazardous wastes, which led to extensive groundwater and surface contamination. Land disposal restrictions will be enforced in May 1990, forcing many industries to treat, and dispose through sewage conveyances, their hazardous waste. USEPA is also setting limits for pollutants in sewage sludge and setting standards for disposal of the sewage sludge.

Commercial and small industries currently not subject to wastewater discharge permits contribute significant quantities of pollutants to the total mass loading of the treatment plants. By regulating these small firms, municipal plants can reduce pollutant loadings and can ensure that unregulated discharges are minimized.

East Bay Municipal Utility District provided an example of how municipal treatment plants can encourage pollution prevention. EBMUD found that radiator repair shops continuously discharged flushing water to the sewer and periodically discharged batch wastes from test tanks and boilout tanks. EBMUD adopted an Order that gave two options to the radiator repair shops in their jurisdiction: the first option allowed the facilities to discharge if they applied for a wastewater permit, met current local limits, and installed treatment equipment; the second option required a shop to comply by not discharging to the sewer. As a result, 10 shops installed recycling equipment, 2 shops discontinued their radiator repair work, and 1 shop now hauls waste offsite. Installation of treatment equipment was considered too costly and therefore shops sought

alternative solutions. EBMUD is considering similar actions for dry cleaners, photoprocessors, and the automobile repair industry in their service area.

Waste Minimization Assessments (Waste Audits)

Many industries and producers of waste do not manage their hazardous materials or waste in an efficient manner. A waste audit can identify opportunities for waste reduction and waste recycling and can provide options for managing chemicals and waste. The first step to waste minimization is assessing waste streams and chemical usage. Government programs can provide incentives or require that industry develop waste minimization plans with waste assessment audits as a component. Pollution prevention audits consist of several components (CBE, 1989):

1) Identify pollutants of concern used or emitted by the source;
2) Track and quantify their movement and emissions from specific process, storage, transport, and treatment activities;
3) Target , quantify, and account for their use and emissions from these activities;
4) Analyze and implement measures for reducing their use;
5) Analyze and implement measures for reducing their waste if use reduction does not eliminate them; and
6) Maintain a current evaluation of audit effectiveness.

The effectiveness of waste audits has been demonstrated at Chevron USA, the largest petroleum refinery in the region (CBE, 1989; M. Bowen, Chevron, personal communication). In 1987, faced with stricter limits on trace element loads in a revised NPDES permit, Chevron instituted a program to identify and reduce sources of trace elements to its wastewater treatment plant. More than 90% of the chromium and nickel in Chevron's effluent was found to originate from specific locations in the refinery. The major source of chromium entering the waste stream was determined to be zinc chromate, a compound added to cooling tower waters to control accumulation of scale and slime. In early 1989, Chevron began to use phosphate-based chemicals instead of zinc chromate in two of the refinery's large cooling tower batteries.

Nickel in another waste stream was traced to a catalyst manufacturing plant in the refinery. Nickel could not be eliminated from the catalyst; improved treatment and management at the catalyst manufacturing site, however, substantially reduced the quantity of nickel released as waste. Measures were taken to reduce leakage, to segregate waste streams with high concentrations, and to recycle some of the nickel formerly discharged. The efficiency of nickel removal was improved by targeting the concentrated waste stream flowing from the catalyst plant. Initial results of effluent monitoring after implementation of these measures in early 1989 indicate 70 to 90% reductions in chromium, nickel, and zinc loads from a starting point below the federal "best available technology" standard for treatment alone (CBE, 1989). The success of Chevron's source reduction initiative was due to adequate monitoring to determine significant sources of pollutants, assessment and implementation of

options to reduce loads, and monitoring to ensure the effectiveness of measures taken to reduce loads.

C. ALTERNATIVE METHODS OF DISPOSAL

Alternative methods of effluent disposal include consolidation or relocation of outfalls, and reclamation or recycling of wastes. Consolidation and relocation of outfalls has received a great deal of emphasis (especially from the San Francisco Bay Regional Board) during the past 20 yr. Many smaller municipalities have formed regional discharge authorities, which collect and treat sewage from member communities for combined discharge to deep waters. An estimated 63% of POTW effluent discharged to the Bay is released into deep water (Condit, 1987). Of the remaining 37%, some 28% is discharged into shallow waters of the Lower South Bay by San Jose/Santa Clara, Palo Alto, and Sunnyvale. Few opportunities remain for further consolidation and relocation of outfalls.

Relocation of the San Jose/Santa Clara, Palo Alto, and Sunnyvale outfalls has received detailed consideration by the SFBRWQCB since 1975, when the Basin Plan for the Bay Region prohibited effluent discharges into the shallow, water quality-limited reaches of the Estuary south of the Dumbarton Bridge, unless it could be shown that such a discharge resulted in a net "environmental benefit" to receiving waters. Such benefits are hard to ascertain because deep-water discharge does not reduce mass loads of pollutants to the Estuary.

The SFBRWQCB granted San Jose/Santa Clara, Palo Alto, and Sunnyvale a deferral from this prohibition under the condition that they perform a long-term, multi-faceted study to determine the effects of their discharge on South Bay receiving waters and to evaluate the existence of net environmental benefit. The study was eventually conducted from 1981 to 1986 (SBDA, 1987). Based on results of this study, the SFBRWQCB concluded that construction of a deep-water outfall north of the Dumbarton Bridge would only enhance water quality by a small amount, and might not improve water quality at all.

This 5-yr study suggested that these discharges did appear to cause significant alterations to salt marsh habitat and cause outbreaks of avian botulism. NPDES permits were issued to San Jose/Santa Clara, Palo Alto, and Sunnyvale in 1989, requiring either measures to mitigate the impacts of their discharges, or to move their outfalls to deeper waters north of the Dumbarton Bridge. They were also required to perform studies of trace element behavior and toxicity in the South Bay. Such studies are designed to assist in gathering data for development of site-specific water quality objectives and effluent limitations, and to evaluate the potential for reductions in loadings to these plants through enhancement of their pretreatment programs or application of other "waste minimization" technologies. Effluent limitations in these permits include limits on mass loading of toxic trace elements, based on current loads flowing from these plants. The South Bay has thus become a testing ground for

some of the more advanced water quality management techniques being employed in the Estuary.

Reclamation or recycling of wastewater is another alternative to effluent disposal into the Estuary. Many municipal dischargers currently reclaim waste water (i.e., apply it to land), especially during summer months (Figure 29). The current Basin Plan for the Bay Region indicates that 21 municipalities in the Region reclaim some of their waste water (SFBRWQCB, 1986). In addition to supplementing the assimilative capacity of receiving waters, reclamation reduces the demand for water supply.

Recently, the SFBRWQCB has also supported the direct recycling of municipal wastewater for use in industrial processes. Two projects of this nature are being planned at present (T. Mumley, SFBRWQCB, personal communication). One of these projects would supply effluent from Central Contra Costa Sanitation District to the Tosco and Shell Oil refineries. The other would transfer effluent from West County Agency (comprised of Richmond Wastewater Treatment Plant and West Contra Costa Sanitary District Water Pollution Control Plant) to the Chevron refinery. The major benefit of these recycling projects would be to reduce requirements for water supply; these arrangements do, however, offer potential benefits associated with consolidation of outfalls.

Disposal of dredged material also contributes pollutant loads to waters of the Estuary, as discussed in Section IV.B. Alternative methods of dredged material disposal are discussed by Gunther *et al.* (1990). Presently, most material dredged from channels in the Estuary is disposed of at aquatic sites near Alcatraz, in San Pablo Bay, and in Carquinez Strait. Alternatives under consideration are disposal at other in-Bay aquatic sites, in the ocean, and on land.

3. Urban and Nonurban Runoff Control Strategies

Runoff from urban areas and drainage from agricultural regions contribute significant loads of pollutants to the Estuary. Various management practices can reduce loads from urban and nonurban runoff, although questions remain about the feasibility and cost-effectiveness of many of these options.

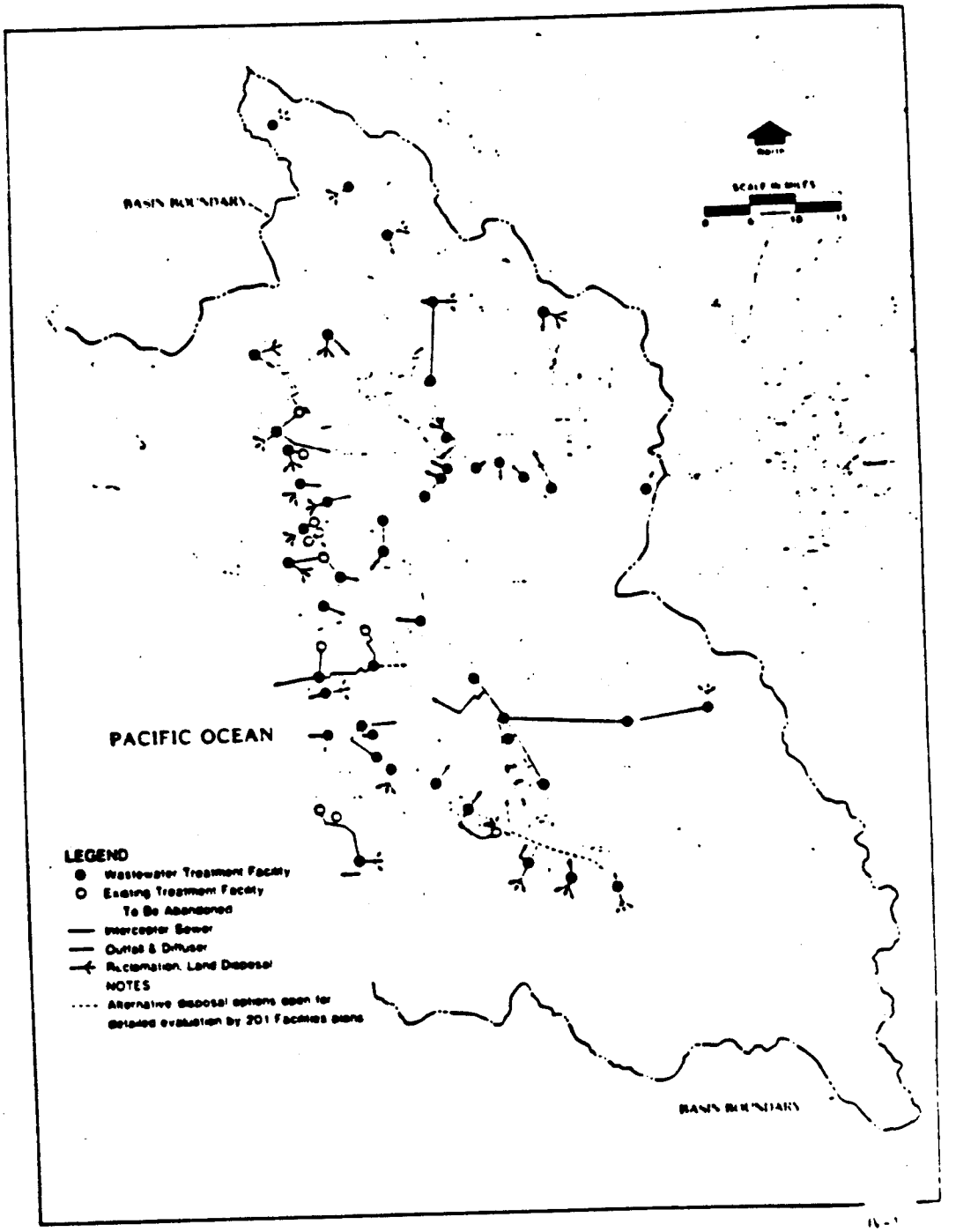
Technological and regulatory options exist for the reduction of pollutant loads to the Estuary by urban and nonurban discharges. The following discussion focuses on the most practical and interesting of these options.

A. TECHNOLOGICAL OPTIONS

Urban Runoff

Techniques for reducing loads from urban runoff include the use of retention basins, street sweeping, and wetlands. The effectiveness and cost of these options will vary among locations such that site-specific analysis will be

Figure 29. Municipalities that reclaim waste water (i.e., apply it to land) in the Bay Region (SFBRWQCB, 1986).



required to assess the load reductions available in a particular region. The USEPA's NURP studies identified treatment options available for urban runoff (USEPA, 1983); those applicable to the Bay-Delta region are briefly described below. It should be noted that if discharge standards to be established for urban runoff are comparable to those for municipal and industrial effluents, then the treatment options described below will not be adequate to bring these waste streams into compliance (C. Batts, CCCSD, personal communication).

1. Detention/Retention Devices

Retention basins can be used to treat or store urban runoff. Runoff collects in such basins, and pollutant loads are reduced through the settling of suspended particles prior to discharge of the collected runoff. The predominance of particle-associated pollutants (see Section IV.B.2.) in urban runoff makes retention basins an effective tool for treating urban runoff. The NURP studies noted that retention devices generally provide a highly effective approach to control urban runoff; however, design details can significantly affect performance (USEPA, 1983). Several types of retention basins exist, including dry detention basins, wet detention basins, dual purpose basins, and various catch basins designed to allow runoff to percolate into soils. Detention basins are a class of retention basin that provides temporary storage of urban runoff.

Dry detention basins are currently in use in various parts of the country. They are used mainly to control flooding and erosion in areas downstream of new developments by limiting the maximum flow rate of runoff to a designated rate. High flows during larger storms fill the basin temporarily, while runoff from smaller storms is discharged with minimal restriction. A detention basin that holds storm water for an extended period of time due to a designed slow release is classified as a dual-purpose basin (USEPA, 1983). Although performance data for dry detention basins as pollution control devices are limited, their performance in pollutant removal is generally considered to be poor (USEPA, 1983). The longer retention time characteristic of dual-purpose basins, however, may reduce the high pollutant loads that may occur with peak runoff flows.

Unlike dry or dual-purpose basins, a permanent pool of water is maintained in a wet detention basin. Designs of wet detention basins range from natural ponds or small lakes to enlarged sections in constructed drain systems. Runoff from an individual storm displaces part or all of the prior volume, and the residual is retained until the next storm event (USEPA, 1983).

The NURP determined that wet detention basins can be highly effective for urban runoff treatment. Performance of individual basins depended upon the sizing of the basin relative to the catchment area and local storm characteristics. Wet basins generally performed better than dual-purpose basins, and much better than dry basins. Pollutant removal was most efficient when the mean storm displaced about 10% (or less) of available basin volume, and when overflow rates (the ratio of mean runoff rate to basin surface area) were only a fraction of the median settling velocity of particles in a basin (USEPA, 1983). In

addition to removal of particulate forms of pollutants by sedimentation, some wet basins exhibited substantial reduction in soluble nutrients, which was attributed to biological uptake occurring in the permanent pool of water found in these structures (USEPA, 1983).

- Catch basins are chambers or wells built at a street curb line and admit surface water to a subdrain (usually a sewer). A sediment sump or settling basin at the base of the drain collects solid matter, preventing it from entering the drain. In this sump the heavier solids settle to the bottom while the lighter solids float on top. Water drains from the basin to a sewer through an outlet structure below the water surface. Catch basins are considered to be poor sedimentation devices, due to their tendency to resuspend collected solids, even at moderate inflow rates (Aronson *et al.*, 1980).

Detention basins can be designed as recharge devices to prevent direct runoff to surface waters by enhancing percolation and infiltration of runoff into soils. These devices range widely in size and include infiltration pits, trenches, ponds, open-bottom galleries, percolating catch basins, and porous pavements. Overall performance depends on the size of the recharge device relative to the catchment area, the surface area provided for sub-surface percolation, and storage volume of the device. Pollutants are removed in direct proportion to the runoff volume that is intercepted and recharged (USEPA, 1983).

Porous asphalt pavements may be a practical means of modifying pavement surfaces to reduce pollutant loading. Rainfall is retained in the base and pavement materials, providing an opportunity for pollutant adsorption and degradation. There are few performance data on the effectiveness of porous pavements (Silverman and Stenstrom, 1985). Greenbelts can be used as recharge devices around parking lots. These grassy areas can be designed to catch runoff and allow it to percolate through the soil. Recharge devices may be inappropriate in areas of steep slopes, impermeable soil conditions, shallow ground water, and proximity of water supply wells. Although such local conditions affect the performance of recharge devices, NURP findings indicate that if properly applied these methods would have a minimum impact on groundwater quality (USEPA, 1983).

Retention basins require dedicated land; this generally constrains their use in urban regions. In many portions of the Estuary, the costs of such basins may be too high relative to their expected performance in load reduction. Some flood control systems in the Bay Area may contain existing basins that could be retrofitted to be used as pollution control devices. More data are needed regarding the performance of the retention basins in reduction of pollutant loads from urban runoff.

2. Street Sweeping

Street sweeping may reduce pollutant concentrations in urban runoff by removing contaminated particles from street surfaces (ABAG, 1979). Street sweeping is the primary public works practice that has been employed for

reduction of pollutant loads from urban runoff, but sidewalk cleaning, catch basin cleaning, and litter reduction are other measures that have been employed in urban runoff pollution control (USEPA, 1983). The NURP studied ten sites (none of these were in California) to determine the benefits of street sweeping. Using two basins to gather test and control data, the study measured concentrations of pollutants in water under both swept and unswept conditions. No statistically significant reductions (of more than 50%) in Event Mean Concentrations for any pollutants tested (total suspended solids, total Kjeldahl nitrogen, chemical oxygen demand, lead, and total particulates) were realized by street sweeping at any of the study sites (USEPA, 1983). The study suggested that street sweeping might be effective, but only in certain areas at particular times of year; further study would have to done to verify this hypothesis.

Catch basins receive large amounts of street dirt and debris, which can be easily removed and prevented from reaching receiving waters. Cleaning programs can be established to remove much of this material before storms. In one study, which focused on 20 residential storm drain inlets (draining about 80 acres of the Castro Valley area), only about 2% of the annual yield of total solids was captured with the usual basin cleaning frequency of once per year (Pitt *et al.*, 1981). With an increase to three or four times per year, basin cleaning could reduce loads by up to 10% (Pitt *et al.*, 1981). In comparison to other control measures, basin cleaning is considered one of the least costly methods of reducing loads of pollutants from urban runoff.

Techniques for reducing the quantity of petroleum hydrocarbons entering the Estuary include establishing readily accessible used-motor-oil collection stations and recycling centers, and installation of grease traps for runoff from parking lots and gasoline stations.

3. Wetlands

Wetlands can also be used to reduce pollutant loads carried in urban runoff. Wetlands have the capability to remove pollutants from surface waters (Wolverton *et al.*, 1976; Richardson and Davis, 1987). Reduced surface runoff velocities and increased channel width in wetlands allow suspended sediment deposition (Small, 1976). Other removal mechanisms include loss to the atmosphere, incorporation into sediments or biota, and biochemical decomposition (Reed and Bastian,1980).

Few studies have addressed the ability of natural wetlands to treat urban storm waters, but in those cases, reduction of suspended solids and BOD concentrations has been better than 50% (Hickok *et al.*, 1977). As indicated in Table 20, however, reduction in nutrient and heavy metal concentrations can be quite variable.

The Demonstration Urban Stormwater Treatment (DUST) Marsh at Coyote Hills Regional Park in Alameda County was designed by ABAG and the

Table 20. Wetland removal efficiencies (%) for water pollutants. Blank spaces indicate that data were not available.

Wetland Removal Efficiencies for Water Pollutants, %

Pollutant	Water Source Applied to Wetlands			
	Primary-treated sewage (constructed wetlands)	Secondary-treated Sewage		Urban runoff (natural wetlands)
		Natural wetlands	Constructed wetlands	
Total Solids		40 - 75		
Dissolved Solids		5 - 20		
Suspended Solids		29 - 90	0 - 92	87 - 99
BOD_5	59 - 90	70 - 96	37 - 92	54 - 97
COD	50 - 90	50 - 80		
Nitrogen (as N)	30 - 98	40 - 97	60 - 86	0 - 95
Phosphorus (as P)	20 - 90	10 - 97	77 - 97	37 - 99
Heavy Metals		20 - 100	23 - 94	25 - 99

Source: Chan et al. 1982

East Bay Regional Park District to examine the ability of a constructed wetland to treat urban storm waters. Measurements during 1985-86 indicated that the DUST Marsh reduced influent loads of total dissolved solids by 64%, oil and grease by 11%, nitrate nitrogen by 15%, ortho-phosphate by 56%, chromium by 68%, copper by 31%, lead by 88%, and zinc by 33% (Meiorin, 1986).

Constructed wetlands also offer much more flexibility in maintenance than do natural wetlands (Meiorin, 1986). The study suggested that until further research was conducted to ascertain the ultimate fate of pollutants captured in wetlands, their use in urban runoff treatment should be limited to constructed wetlands. If constructed wetlands are effective in removing trace elements and persistent toxic organic compounds from urban runoff, these substances will accumulate in sediment or biota of these wetlands. Wildlife may be exposed to accumulated pollutants while migrating through the wetland or feeding in it. For further information on wetlands, including the sensitivity of wetlands to pollutants, the reader is referred to the *Status and Trends Report on Wetlands* (Meiorin *et al.*, 1990).

Agricultural Runoff

Agricultural water conservation can reduce contaminated subsurface drainage and irrigation return flows. Water conservation can also reduce other problems associated with excess water application, such as nutrient leaching, deficiencies in soil aeration, and yield reductions. Conservation techniques include reuse of agricultural drainage water and alternative irrigation practices and equipment.

Long-term field investigations are being conducted to examine the effects of reusing agricultural drainage water for irrigation. This practice could reduce the salt and mineral load discharged into receiving waters, with the added benefit of extending the available water supply. Two strategies of drainage water reuse have been considered. The first is a cyclic strategy, in which water of high quality is alternated with drainage water. Switching to poorer water quality generally occurs after a crop becomes established. Blending of drainage water with high quality water is another approach. This method works best on salt-tolerant crops. Increasing salt buildup in the root zone occurs after many years (without the use of additional water for leaching), limiting the long-term usefulness of any reuse technique (Coppock, 1988).

An irrigation/cropping management strategy has been developed to facilitate the use of blended irrigation waters. A 4-yr field study of this strategy in the Imperial Valley of California showed that drainage water can be reused in an effective manner if it is monitored for potential build-up of salts (Rhoades *et al.*, 1988; UCCC, 1988). The reduction of pollutant loads that may be achieved using these techniques has not been studied.

Many workers believe that improved irrigation is the key to management of agricultural drainage in the Central Valley (Coppock, 1988). Volumes of agricultural drainage can be reduced through more efficient irrigation. Furrow

irrigation systems are used throughout the Central Valley. Differences in infiltration rates among portions of a furrow-irrigated field can result in varying depth of water infiltration. Thus in order to fully irrigate a given field a farmer often must over-saturate one portion, resulting in excess subsurface drainage. One approach to reducing subsurface drainage caused by differences in infiltration rates is surge irrigation, in which water is applied in pulses instead of continuously. One study of use and effectiveness of surge irrigation in Kern and Fresno Counties of California indicated that 50% less water was needed for complete wetting across a sandy loam soil; consequently deep percolation of applied water and subsurface drainage were greatly diminished (Goldhamer *et al.*, 1987). Drainage from fields with surface irrigation can also be reduced by shortening furrow lengths and reducing the running time of water.

Efficient irrigation is easier to achieve with pressurized water delivery systems. Drip irrigation is an alternative to furrow irrigation that is being used in selected locations in California. Drip systems can be installed at or below the soil surface, with the latter being particularly efficient as no evaporative losses occur. Deep percolation and subsurface drainage are minimized using this technique. Drip irrigation problems include diminishing water pressure within the delivery system and occasional plugging of flow holes.

The CVRWQCB has been working to reduce peak concentrations and mass residue discharges of rice field pesticides, particularly Ordram and Bolero. The CVRWQCB is also reducing the volume of irrigation return flows by requiring tail-water recycling and effluent spreading on fallow fields, primarily in the Colusa Basin Drainage.

The storage of drain waters in evaporation ponds has also received consideration. Depending on the characteristics of the influent waters, ponds of this type may pose a threat to wildlife (Ohlendorf *et al.*, 1986a; Rhoades *et al.*, 1988). Taking contaminated or poorly drained fields out of production is another option for reducing the generation of contaminated agricultural drainage.

B. REGULATORY STRATEGIES

The State and Regional boards have the ultimate responsibility for urban and nonurban pollution management in California. The SWRCB recently identified three general management options to implement Best Management Practices (BMPs) for urban and nonurban pollution (SWRCB, 1988a). Federal regulations (40 CFR 130.2) define BMPs as methods, measures, or practices selected by an agency for pollutant source control, and can be applied before, during, and after pollution-producing activities to reduce or eliminate the introduction of pollutants into receiving waters.

The first strategy outlined by the State Board is voluntary implementation of BMPs by property owners or managers who may implement BMPs for economic or environmental reasons. Voluntary implementation can be encouraged through education, training, financial assistance, technical assistance, and demonstration projects.

Second, use of BMPs can also be encouraged by regulation. Although the *Porter-Cologne Act* constrains Regional Boards from specifying the manner of compliance with water quality standards, they can use their regulatory authority to provide incentives for implementation of BMPs. An example would be waiving adoption of waste discharge requirements on the condition that discharges comply with BMPs. Finally, through adoption and enforcement of effluent limitations for any proposed or existing discharge of urban and nonurban runoff, Regional Boards can set limits that, in practice, require implementation of BMPs (SWRCB, 1988a).

The federal government has proposed regulations requiring NPDES stormwater discharge permits for some municipalities and industrial facilities. This approach is being actively pursued in the Bay Area (SFBRWQCB, 1986). Studies being conducted on urban and nonurban loads in the Santa Clara Valley watershed are expected to provide a basis for one of the first NPDES municipal stormwater discharge permits in the nation (T. Mumley, SFBRWQCB, personal communication).

Construction sites are a major source of surface runoff contamination in the Bay Area (ABAG, 1981). Basic measures to control construction-related erosion can be relatively inexpensive but quite effective (ABAG, 1980). Cities and counties can improve grading ordinances to require practices to reduce sediment loss to surface waters (ABAG, 1980). Erosion control ordinances can prohibit poor landscaping, drainage, and grading practices at construction sites. In agricultural areas, plowing and grazing techniques can be modified to reduce erosion. Other techniques include stabilization of stream banks and channels (ABAG, 1979). Finally, controls on new construction are possible if other measures fail. For information on sediment control measures, see ABAG (1981).

Perhaps the most cost-effective means for local agencies to reduce oil and grease loading to the Estuary is to control commercial land use. Undeveloped buffer strips can be acquired along water bodies, and performance standards can be established for developments within water supply catchments (ABAG, 1979). Although land-use control is not particularly feasible in the Bay Area due to the level of development that has already taken place, it might be favorable in undeveloped areas. Controls that require establishment of desirable residences near employment areas and convenient mass transportation alternatives to automobile use could dramatically reduce runoff pollution attributable to vehicle operation.

Other regulatory options include screening for illegal discharges, chemical use control, and enforcement of existing law (T. Mumley, SFBRWQCB, personal communication). The quantities of pesticides and fertilizers that flush into surface waters can be reduced by restricting the use of these chemicals (ABAG, 1979). Pollutant loads from residential areas may also be curtailed by decreases in pesticide and fertilizer application. Management of urban runoff loads can be enhanced by enforcement of the regulations prohibiting

household and garden wastes (ABAG, 1979). Strict enforcement of anti-littering ordinances can reduce loads carried by surface runoff (ABAG, 1979).

Large-scale technical solutions to drainage water problems in the Central Valley appear unlikely. Most of the drainage reduction efforts must be made by individual farms. Long-term water contracts at below-market prices, however, currently provide little economic incentive to farmers to improve irrigation efficiency. Institutional changes that could supplement these conservation efforts include allowing more flexible delivery schedules from surface canals, a establishing tiered pricing based upon a crop's calculated water needs, and taking into account any lateral sub-surface drainage movement when determining water needs in certain areas (Coppock, 1988). The costs of changes in irrigation practices will be critical in determining whether they are adopted. Other options that may succeed with the proper institutional incentives include planting different crops, retiring of land, and public subsidy of capital costs. There is a need to provide technical assistance in the area of irrigation technologies (Coppock, 1988).

Education and public information programs can have a positive effect on technical or political pollution control techniques. For example, the establishment of waste oil recycling centers and the restriction of the use of harmful pesticides would benefit from increased public awareness.

Pollution prevention techniques, discussed in Section V.B.2.B. above, can also be employed to achieve reductions in pollutant loads in urban and nonurban runoff.

4. Summary of Pollution Control and Pollution Prevention Strategies

This section has focused on the scientific information needed for successful implementation of management techniques presently in use and has presented some alternatives to present techniques. The following list summarizes the alternatives that were discussed.

A. REGULATORY STRATEGIES

The Regional boards could adopt a mass emissions strategy to regulate loads of pollutants that accumulate to potentially harmful concentrations in sediments and organisms of the Estuary.

Establish sediment quality criteria for evaluation of polluted sediment.

Establish a more comprehensive set of criteria for contamination in tissues of Bay-Delta organisms to protect both estuarine biota and their human consumers.

B. EFFLUENT DISCHARGE MANAGEMENT STRATEGIES

Implement enhanced treatment of municipal and industrial wastewater aimed specifically at removal of trace elements and toxic organic pollutants.

Review adequacy of pretreatment programs, including discharge standards and coverage of industrial sources.

Conduct toxicity reduction evaluations of additional municipal and/or industrial discharges.

Increase reclamation and recycling of wastewater.

C. URBAN AND NONURBAN RUNOFF CONTROL

Employ detention or retention basins to treat or store urban runoff.

Increased street sweeping to remove contaminated particles from urban surfaces.

Divert runoff through constructed wetlands to promote deposition of suspended sediment and biological processing of nutrient loads.

Improve water conservation on agricultural lands, including water reuse and more efficient irrigation.

Implement best management practices to reduce or eliminate the transport of pollutants by runoff.

D. POLLUTION PREVENTION

Implement pollution prevention measures for reducing pollutant loads to the Estuary from all input categories, including municipal and industrial effluent discharges, urban and nonurban runoff, riverine loads, and other inputs. Waste audits to identify opportunities for pollution prevention would be a first step in this process. Pollution prevention measures could include

1) reformulation of products,
2) substitution of raw materials that introduce smaller quantities of hazardous substances into production processes,
3) improvement of technology to alter the primary sources of waste generation,
4) improved housekeeping in operations that produce waste, and
5) recycling of polluted substances at the site of waste generation (closed-loop recycling).

VI. CONCLUSIONS AND RECOMMENDATIONS

A summary of the findings of this report is presented below, followed by recommendations for reducing important scientific uncertainties relating to management of pollutants in the Estuary.

A. CONCLUSIONS

1. Historical Trends

Although only indirect evidence is available, population growth in the Bay-Delta region since the mid-1800s surely resulted in progressively increasing loads of pollutants to the Estuary. Advances in the management of effluent discharges have reduced the discharge of BOD and suspended solids to the Estuary in the last 20 yr. Although reductions in loads of toxic pollutants have accompanied those for conventional pollutants, the relatively poor historical data base for toxic pollutants makes accurate quantification of these reductions difficult. Similarly, there are few historical data regarding the abundance, distribution, or effects of pollutants of concern in the Estuary. This lack of knowledge is due in part to the fact that methods for measuring pollutants in solution were unavailable. Although evidence indicates that loading rates of toxic pollutants have declined in the last 20 yr, present rates of mobilization of toxicants in the Estuary due to human activities remain high relative to natural rates of mobilization and continue to pose a potential hazard to biota.

2. Pollutants of Concern

Pollutants known to be of particular concern in the Estuary include cadmium, copper, mercury, nickel, selenium, silver, tin, chlordane and its metabolites, DDT and its metabolites, PCBs, toxaphene, and PAHs. As more data become available, more pollutants will certainly be added to the list. The addition of more pollutants of concern to the list will not only be due to the introduction of new chemicals, but will also be due to the development of appropriate analytical procedures for many pollutants.

Existing data on ambient concentrations of pollutants in the Estuary indicate the enrichment of several trace elements, organochlorines, and PAHs; but the data provide a limited basis for assessment of spatial or temporal trends of the abundance or distribution of these substances in water, tissue, or sediment. Very few reliable studies of pollutant concentrations in water have been performed. More data are available on concentrations in sediment, but these primarily pertain to trace elements, are of varying quality, and are not collected in a systematic manner. Even the substantial data base for pollutant concentrations in tissues does not include data for PAHs or PAH metabolites.

3. Pollutant Loads

Estimates of pollutant loads from effluent discharges, although more accurate than estimates of other inputs, still are uncertain. This is due to the use of insensitive analytical methods such as those prescribed in the existing USEPA manuals for the analysis of pollutants in environmental samples. The use of insensitive methods has yielded many "below detection limit" analytical results for many pollutants. Other factors contributing to uncertainty in the available data include infrequent sampling and the absence of quality control test results in data reports.

Although concentrations of many classes of pollutants are analyzed in effluents, only trace element loads can be quantified, and even these calculations are based upon relatively infrequent sampling in many cases. Although the spatial distribution of these loads can be assessed, the fragmented, heterogeneous nature of the data base makes it difficult to establish meaningful temporal trends in loading. The largest effluent loads of many of the trace elements were released into Upper South Bay (cadmium, lead, mercury, silver, and zinc) and Lower South Bay (arsenic, copper, and nickel) by large municipal dischargers in those regions. A high proportion of trace element loads from effluent discharges consists of dissolved forms.

Urban and nonurban runoff clearly contribute significant loads of pollutants of concern to the Estuary, although the magnitude of these loads cannot be accurately determined. Limited local data, in combination with data from other regions, suggests that urban runoff carries substantial loads of hydrocarbons, zinc, and lead. Data from Sacramento indicate that the volume of urban runoff during the dry season is significant, but pollutant concentrations are lower than wet season runoff. It is unknown whether the dry-flow loading determined for Sacramento is indicative of an Estuary-wide pattern. Significantly more local data regarding event-mean concentrations of pollutants and runoff coefficients are needed to characterize urban runoff loads to the Estuary.

Estimates of pollutant loads in nonurban runoff are even more poorly characterized. These estimates, derived using models of sediment erosion (for trace elements) and estimates for losses of applied pesticides could be very large relative to other pollutant inputs. Erosion of range and pasture land could comprise a significant portion of nonurban pollutant loads. Considerably more data are needed regarding key parameters such as trace element concentrations in soils, pesticide application rates, and soil moisture levels. Recent estimates of trace metal loads in the Sacramento Valley indicate that agricultural drainage carries large quantities of chromium, nickel, and arsenic in that region.

The Sacramento and San Joaquin rivers carry significant loads of pollutants to the Estuary. With the exception of Se, the data for the San Joaquin are of better quality. The San Joaquin River contributes trace element loads (based on concentrations in unfiltered samples) that exceed those from all

municipal and industrial discharges combined. Most of the mass transport to the Estuary occurs during periods of maximum flow in the rainy season, especially for elements that tend to associate with particles. Pollutant loads carried by the Sacramento River are probably substantial, but available data do not allow their quantification.

The significance of pollutant mobilization from dredging and dredged material disposal activities has received little study to date. The susceptibility of pollutants to remobilization will ultimately depend upon the manner in which they are bound to dredged sediment and the physicochemical characteristics of the disposal site. The ultimate fate of dredged material (and its associated pollutant load) depends on complex patterns of water and sediment movement in the Estuary.

Other inputs, including atmospheric deposition, spills, marine vessel discharges, and leachates from waste disposal sites, contribute minor quantities of most toxic pollutants to the Estuary. Atmospheric deposition may contribute significant loads of PAHs and total hydrocarbons. Spills intermittently contribute large quantities of petroleum hydrocarbons that are significant on a local, and perhaps regional, scale.

An important subject of future research will be determining the origins, or primary sources, of toxic materials present in municipal and industrial discharges, urban and nonurban runoff, tributary rivers and streams, and other modes of pollutant transport to the Estuary. Detailed investigation of primary sources will be necessary for pollutants found to be entering the Estuary in excessive amounts.

4. Fate of Pollutants

Little is known regarding the specific fate of pollutants in the Estuary. This is due to both a lack of fate modeling for the system and a paucity of data regarding the abundance and distribution of pollutants in water, sediments, and biota. Pollutants exist in a variety of chemical forms, change forms in the ecosystem, and move throughout the Estuary in response to complex hydrodynamic forces derived from tides, freshwater inflows, and winds. The data requirements for models demand site-specific information for pollutants as well as a knowledge of ambient environmental conditions. In the absence of such data, predictions of the partitioning of pollutants in the environment cannot be made. The dynamics of the Estuary cause variation in environmental conditions over relatively small spatial and temporal scales; such variations frustrate attempts to predict the behavior of pollutants in the Estuary.

It is likely, however, that particular processes, especially those relevant to sorption, dominate the fate of certain pollutants. Efforts to further elucidate these key processes for pollutants of concern in the San Francisco Estuary could, therefore, significantly reduce uncertainties concerning pollutant fates. These key processes include:

(1) the speciation of pollutant loads to the Estuary,

(2) the fractions to which a given pollutant sorbs,
(3) the kinetics of sorption/desorption reactions, and
(4) the influence of key environmental parameters such as salinity and biological cycles.

Understanding pollutant transport is also vital in determining the effects attributable to specific pollutant inputs. Complex currents in the Estuary, produced by the interaction of tides, freshwater inflows, and winds, will transport dissolved and particle-associated pollutants far from their original point of introduction. Gravitational circulation appears important in the northern reach, while wind-driven currents play an important role in circulation in the South Bay. Circulation and mixing in both portions of the Estuary are significantly influenced by freshwater inflows.

An improved understanding of flocculation, deposition, and erosion is also needed to assess the transport of particles. The highly dynamic nature of the Estuary and the complexity of these processes makes development of predictive models of particle transport especially difficult.

Some inferences regarding the transport of dissolved species and non-settleable particles can be made based upon what is known of circulation patterns in the Estuary. Knowledge of these patterns, however, is based almost exclusively on measurements of currents in the deeper channels. Our lack of knowledge for the shoal areas that make up the majority of the Estuary constrains efforts to regulate the loading of pollutants using comprehensive modeling schemes such as wasteload allocations. Chemical and isotopic tracers in Bay-Delta studies could help identify sources and fates of some pollutants, particularly in areas with mixed sources. Tracers (either existing or introduced) are available for sewage, urban runoff, and detergents. New tracing techniques using bacteriophages (bacterial viruses) are available to trace movement of particles.

Understanding residence times for pollutants is also important for the successful application of a comprehensive modeling scheme. The complexity of circulation and mixing in the Estuary makes estimates of residence times uncertain. Estimates of residence times for entire embayments almost certainly overlook the significant variations in residence times within embayments, particularly with respect to the broad, shallow areas. An improved understanding is needed of the currents in broad, shallow areas of each embayment and the influence of winds, tidal fluctuations, and freshwater inflows on these currents and the resultant exchange of water between the shallows and the channels. More detailed understanding is also needed of ocean-Bay exchange at the Golden Gate, and the effect of local morphometry and bathymetry upon mixing regimes. Only with this information will more accurate estimates of residence times be available for use in predicting the fate of pollutants in the Estuary.

Several pollutants accumulate in biota of the Estuary, including mercury, selenium, copper, nickel, silver, certain pesticides, PCBs, and petroleum

hydrocarbons. Relatively few data are available, however, particularly in light of the spatial and temporal variations demonstrated by the existing datasets. Comprehensive data regarding the abundance and distribution of pollutants in the Estuary are sorely needed. Furthermore, the concentrations of pollutants of concern in the tissues of upper-trophic-level organisms in San Francisco Bay, and the possible contribution of food-web transfer to this phenomenon, have not been studied. Current knowledge of pollutant fluxes among sediments, waters, and biological membranes is relatively poor and is an area where further research is needed.

It remains extremely difficult to estimate potential bioavailability of pollutants solely from analyses of water, sediments, or suspended particles. Tissue analysis provides a direct measure of bioavailability. A greater understanding of the interplay among physiological, biochemical, geochemical, and ecological factors is required before predictions of bioavailability, based upon knowledge of water and sediment chemistry, can become a reality.

5. Effects of Pollutants

Previous review of pollutant effects in the Estuary and environs (Phillips, 1987) reached several relevant conclusions, including:

1) areas of extreme pollution have not been well studied with respect to sources or effects of the pollutants in place;
2) the amounts of silver in the South Bay are significant and may be having effects upon biota;
3) copper and cadmium appear to be unusually bioavailable in the Bay;
4) further studies are needed regarding forms of arsenic in the Estuary and the biological effects of tin (especially TBT); and,
5) more congener-specific studies of PCBs are needed, since the toxicological effects of PCBs may be related to only a small number of co-planar forms of these 209 compounds.

The following observations suggest that pollutants may be having significant effects in the Estuary:

1) certain creeks and rivers in the Bay-Delta catchment, and some Bay sediments, are significantly toxic in bioassays;
2) there are decreases in numbers of individuals and species of benthic invertebrates in some highly polluted areas;
3) there is statistical evidence that reproduction of starry flounder is negatively affected by PCBs;
4) extremely high concentrations of silver and copper are found in South Bay clams;
5) embryo size and eggshell thickness in black-crowned night herons have been negatively correlated to concentrations of PCBs and DDE, respectively, in eggs;
6) high concentrations of Se in agricultural wastewater are associated with deformities in waterbirds in the Kesterson area; and

7) elevated occurrences of micronuclei in red blood cells of starry flounder from throughout the Bay suggest an effect of genotoxic chemicals.

Such observations have led many workers to conclude that pollutants are exerting deleterious effects on the Estuary and its biota. While the available data suggest apparent pollutant-related effects, cause-and-effect relationships between pollutants and effects will only be established from laboratory investigations. Whether cause-and-effect relationships can be established from field studies is unclear.

Conclusions based upon bioassays with *Ceriodaphnia* (as used in the creeks and rivers of the catchment) may not be applicable to the Estuary as a whole; they may be confounded by effects of factors such as salinity. Factors such as grain size, organic carbon content, and the presence of hydrogen sulfide can affect the outcomes of sediment bioassays. Species richness and abundance of benthos respond strongly to amounts of organic matter in sediment. In the case of correlations of PCBs with reproductive effects in fish and birds, other chemicals that may be behaving similarly but were not measured (e.g., PAH metabolites, dibenzofurans, or dioxins) may be responsible for the observed effects. Extremely high concentrations of silver and copper found in the tissues of bivalves in some areas of the South Bay are cause for concern, but effects have not yet been demonstrated. The possibility that some part of the Bay-Delta estuarine fauna and flora is adapted to high concentrations of metals needs to be investigated.

Studies of pollutant effects using communities of benthic organisms are valuable, but subject to alternative interpretations. Experimental manipulations of sediments in which organisms are allowed to colonize and grow with controlled physical and geochemical properties and additions of known amounts of pollutants may be very helpful. In such cases something must be known of the chemical and physical form of the pollutant under the experimental conditions if the results are to be fully useful.

In general, although effects are occurring due to the elevated concentrations of some toxic pollutants found through much of the Bay-Delta, these relationships are difficult to demonstrate in the field. Many species have been observed to bioaccumulate toxicants, but this does not necessarily imply that the toxicants are reducing the viability of populations.

The evidence for toxic effects on the biota is strongest for organic pollutants, especially for organochlorine compounds. Studies of sublethal effects of pollutants on processes such as reproduction in field-collected animals can certainly increase our understanding as to whether the problems identified in starry flounder and black-crowned night herons are representative of other species.

6. Future Trends

Population growth and ensuing land-use changes will create a continuing need for management of increasing loads of pollutants to the Estuary. Future trends in pollutant loads will depend primarily on the implementation of improvements in the management of these loads. Potential areas for improved management include the application of pollution prevention techniques to reduce the generation of polluted wastes and implementation of enhanced treatment of effluent discharges and urban and nonurban runoff.

7. Pollution Control and Pollution Prevention Strategies

A. REGULATORY TOOLS

Water quality objectives are a central feature of the present approach to regulation of pollutants in the Estuary. Numeric water quality objectives have been adopted for only some of the pollutants of concern in the Estuary. Routine, reliable measurement of pollutant concentrations in receiving waters is essential to assessing attainment of these objectives. Sampling efforts in receiving waters to date have consisted of separate, uncoordinated studies; whether objectives are being attained is currently unclear. The principal shortcoming of water quality objectives is that they provide only partial control of the gradual accumulation of pollutants to potentially damaging levels in organisms and sediment. Successful application of narrative objectives prohibiting toxicity in the Estuary are made difficult by deficiencies in our understanding of pollutant loads, fate, abundance, and effects.

Documented high concentrations of pollutants in sediments are of concern, and regulatory agencies are considering adopting sediment quality criteria to identify sites where toxic effects may be occurring and to assess the risk of toxicity for disposal of sediments. There is significant interest in the use of equilibrium partitioning techniques to define toxic levels of pollutants in sediments, but the complexities of the sorptive behavior of pollutants in sediments (particularly trace metals) has prevented the development of such criteria to date. A key uncertainty is the consistency with which a compound partitions in different sediments. Further studies of sediment-water partitioning of pollutants will help the Regional Boards to establish reasonable sediment quality criteria.

The SWRCB is currently considering implementation of a "mass emissions strategy" intended to address undesirable pollutant accumulation in organisms and sediment. This strategy would limit mass emissions from all sources to current levels, implement a program to reduce loads, and would include eventual development of tissue and sediment objectives. Difficulties associated with the prediction of effects due to pollutant concentrations in sediment (see above) constrain the use of sediment objectives. Tissue objectives would more clearly relate to ultimate toxic effects.

The most advanced form of water quality regulation would be the establishment of site-specific water quality objectives and corresponding wasteload allocations for receiving waters. Deficiencies in the present understanding of the sources, fate, distribution, and effects of pollutants in the Estuary constrain development of a wasteload allocation based on models of the ecosystem. Nevertheless, limits on mass loads, like other regulatory policies, can be established with existing data. The discussion in this report can be used to aid in determining the necessary safety factors for assuring protection of beneficial uses until the scientific basis for wasteload allocation is improved.

B. EFFLUENT DISCHARGE CONTROL STRATEGIES

Enhanced treatment of trace elements and organics in municipal wastewater will require processes designed specifically to remove these substances. The expense of these techniques requires a careful evaluation of the cost-effectiveness of this option for load reduction relative to other methods. Industrial plants are better suited to treatment of trace pollutants, and many major industrial dischargers are currently developing sophisticated treatment systems.

Activities that reduce loads of pollutants entering waste streams are another option for reducing inputs from effluent discharges. Pretreatment programs have contributed to decreases in trace element loads to municipal plants in the past 15 yr; further reductions through this program may be possible. "Toxicity reduction evaluations" have been successfully employed by industrial dischargers in the Region. An improved understanding of the sources of pollutants entering municipal plants is needed for effective reduction of these inputs. Pollution prevention techniques, including waste reduction audits, represent a promising option for achieving reductions of pollutant loads carried by municipal and industrial effluents.

Reclamation and recycling of wastewater are additional means of reducing pollutant loads to the Estuary from municipal and industrial discharges.

C. URBAN AND NONURBAN RUNOFF CONTROL STRATEGIES

Technological methods for reducing loads from urban runoff include the use of retention/detention devices, street sweeping, and wetlands. Several types of retention/detention devices exist, including dry detention basins, wet detention basins, dual purpose basins, and various catch basins designed to allow runoff to percolate into soils. Land requirements for these devices constrain their use in urban regions. Data on the effectiveness of street sweeping are inconclusive. Catch basin cleaning is the most cost-effective method of reducing loads of pollutants from urban runoff. Constructed wetlands may be employed to reduce pollutant loads in urban runoff; potential exposure of wildlife to accumulations of persistent toxicants in these systems, however, may limit the usefulness of this technique.

Agricultural water conservation can reduce volumes of polluted agricultural drainage and salt and mineral loads to receiving waters. Reuse of agricultural drainage may be effective for salt-tolerant crops. Increasing salt accumulation in the root zone limits the long-term utility of reuse techniques. Improved irrigation can also reduce pollutant loads carried by agricultural drainage. Surge irrigation and drip irrigation are techniques that may increase irrigation efficiency.

Regulatory strategies for reducing loads from urban and nonurban runoff include requiring the adoption of "best management practices", permitting of discharges of urban runoff, imposing erosion controls (especially at construction sites and on agricultural lands), establishing appropriate land use controls, screening for illegal discharges into storm drains, controlling chemical use, establishing public information programs aimed at reducing use of household toxicants and safe disposal of household toxic waste, and restructured transportation and planning to reduce vehicular traffic and the associated waste emissions. Pollution prevention techniques can also be applied to the reduction of loads carried by urban and nonurban runoff.

B. RECOMMENDATIONS

This report has discussed many topics where improved understanding would enhance the scientific basis for management of pollutants in the San Francisco Estuary. Specific recommendations are presented in this section for advancing our understanding of the abundance and effects of pollutants.

Management of pollutants in the Estuary would benefit from an improved understanding of the overall ecological context in which pollutant-related effects occur. This context is complex because the Estuary is a highly dynamic environment where environmental conditions vary considerably on daily, seasonal, and annual bases. Furthermore, the system has been subjected to several types of human influence other than pollution, including freshwater diversion, alteration of habitats, the introduction of exotic species, and exploitation of fish and wildlife populations. Characterization studies would improve our understanding of how the effects of pollution and other human influences are interrelated. Improved understanding of long-term trends in the ecological effects of pollution will rely, to a large extent, on improved understanding of long-term "natural" variability, and long-term trends in effects of these other human influences. Stable, long-term funding would be required to gain such an improved understanding.

Establishing priorities for addressing these multi-disciplinary research needs is a complicated task. Ideally, a systematic strategy should be developed for progressively attacking the most important broad questions. Such a strategy must include characterization studies, application of existing process knowledge to specific problems in the Estuary, and improved understanding of processes. Development of such a strategy would require a cooperative effort of those involved in managing and studying the ecology of the Estuary.

Much remains to be learned about the Estuary. Studies must be carried out in the future to provide answers to the most pressing questions that surround the health and well being of the Estuary as a system. Precisely which studies should be carried out is a policy decision that is outside the scope of this Status and Trends Report on pollutants. The conclusions listed above, however, can help to shape certain aspects of recommendations for further studies.

Despite the mass of data available on pollutants in discharges, pollutants in sediments and pollutants in biota, no systematic, long-term studies of the Estuary as a whole have been conducted. Such basic ecological studies are critical to management of the Estuary and should be implemented. The major recommendation of this report is that there should be established a series of long-term studies of basic Estuarine function at the physical, chemical, and biological level. Such a program should be provided with continued funding and should consist of a comprehensive monitoring program that can serve as a framework for more specific studies of pollutants and their effects at particular locations throughout the Estuary.

Specific recommendations for improving the state of knowledge regarding the ecological effects of pollutants are presented below.

1. Trends in Pollutant Abundance and Distribution

Existing monitoring programs provide inadequate representations of temporal and spatial trends in the abundance and distribution of pollutants in the Estuary. It is strongly recommended that comprehensive, system-wide, long-term monitoring be conducted to provide a sound basis for assessment of the effectiveness of regulatory activities.

At the request of the SWRCB, the Aquatic Habitat Institute provided a recommended design for such a program that would cover the entire Estuary (Phillips,1988). This design emphasized the importance of biomonitors, but also recommended water column analyses (to ascertain attainment of water quality objectives) and sediment analysis on a local scale. The SFBRWQCB has announced its intention to implement a comprehensive monitoring program aimed at providing information needed for regulation of effluent discharges and urban and nonurban runoff. Other State and Federal agencies have also expressed interest in a comprehensive pollutant monitoring effort.

2. Pollutant Loads

An accurate understanding of the relative significance of pollutant loads is essential to the cost-effective attainment of water quality objectives. The following recommendations relate to improving upon present knowledge of loads from the most manageable inputs: effluent discharges and urban and nonurban runoff.

A. MUNICIPAL AND INDUSTRIAL EFFLUENTS

Although loading data for effluent discharges are more comprehensive than those for any other input, they are still deficient in several respects. Improvements in the data base are needed to support more sophisticated management techniques, such as wasteload allocation, and to document the effectiveness of management techniques. Efforts to improve this data base should focus on the largest discharges, relatively few of which contribute the majority of these loads to the Estuary. The following efforts would improve our understanding of these loads:

1) Greater emphasis should be placed on quality assurance of monitoring data. Quality control test results should accompany results of toxic pollutant monitoring to affirm their accuracy, and to enhance their use in historical assessments of pollutant loading.
2) Sampling frequency of some major discharges should be increased. Sampling programs should be capable of detecting seasonal and annual trends in pollutant loads.
3) Concern over the potential effects of particular pollutants may warrant the use of analytical methods with lower detection limits, or methods

capable of determining the chemical species introduced to the Estuary from particular inputs.

4) Efforts to reduce loads from major inputs may benefit from pollution prevention audits, which would identify the primary sources of pollutants present in effluents and examine methods of reducing or eliminating them.

B. URBAN AND NONURBAN RUNOFF

The limited data available clearly indicate that urban and nonurban runoff carry significant loads of pollutants to the Estuary. These data, however, provide an inadequate basis for reliable estimates of the relative magnitude of these loads. Further study of pollutants from urban and nonurban runoff is needed to address the following issues:

1) Estimates are needed of the critical environmental parameters related to the loading of pollutants to the Estuary *via* stormwater runoff in urban areas. These include the volume-weighted concentration of pollutants in runoff from different urban regions and land uses, empirical estimates of runoff coefficients, and the fate of particulate pollutants in runoff in stormwater conveyances and urban streams. Efforts currently underway in the Santa Clara and Alameda counties should be expanded to other regions.
2) Analyses are needed of the critical environmental parameters related to the loading of pollutants to the Estuary *via* nonurban runoff. Careful examination of models used to predict nonurban runoff should be conducted, and further studies of the magnitude and temporal variation of pollutant concentrations in agricultural drainage should be conducted. More data on agricultural drainage in the San Joaquin Valley are needed.
3) The speciation of pollutant loads from urban and nonurban runoff, especially trace element species and metabolites/transformation products of a variety of organic compounds (PCBs, PAHs, dibenzofurans, dibenzodioxins) should be investigated.
4) Information from (1), (2), and (3) should be employed to establish spatial and temporal trends in the loading of all pollutants to the Estuary from urban and nonurban runoff, especially on the seasonal and subembayment scales.
5) Efforts to reduce loads from urban and nonurban runoff may also benefit from pollution prevention audits, which would identify the primary sources of pollutants present and examine methods of reducing or eliminating them.

C. RIVERINE INPUTS

1) A lack of information on the pollutant loads carried by the Sacramento River, the Estuary's principal source of freshwater, leaves a significant gap in our understanding of loads to the Estuary as a whole. Sampling should be sufficiently frequent to characterize seasonal trends and loads during high-flow periods.

D. CONTROL OF EFFLUENT DISCHARGES AND URBAN AND NONURBAN RUNOFF

Pilot studies of the relative load reductions which would result from application of promising prevention or control techniques to various sources would assist in fashioning management strategies. Sampling programs should be established to accurately determine the effectiveness of control measures that are instituted.

3. Pollutant Fates

The well-being of the Estuary and its living resources cannot be maintained (or, in certain locations, reclaimed) with only a knowledge of the masses of pollutants in the system or entering the system. A thorough understanding of the fate (i.e., the direction, rate, and magnitude of pollutant movements) within and among the compartments of the Estuary would be useful in managing pollutant abundance and effects. All pollutants have physical and chemical characteristics that cause them to accumulate in specific compartments in the Estuary. For metals and lipophilic organic pollutants the major sinks should be the deposited sediments. However, in a high-energy environment such as the San Francisco Estuary, deposited sediments are, in most locations, a seasonally transitory phenomenon. The status of our understanding of pollutant fates in the Estuary could be improved by the following efforts:

A. POLLUTANT CHEMISTRY

1) Studies should be conducted to determine the speciation of pollutants of concern in the Estuary and the influence of speciation on solubility, vapor pressure, settling, and, most importantly, bioavailability.
(2) A generalized fate assessment model for the Estuary is needed to estimate fates for pollutants of concern, with emphasis placed on the development of mathematical models (fugacity models) of pollutant fate for the major constituents.
(3) The relative distribution (partitioning) of pollutants from the water column and suspended solids compartment of the Estuary to the deposited sediments requires further study.

B. POLLUTANT TRANSPORT

1) Studies on the circulation patterns and mixing processes within the Estuary should be continued, with attention given to transport mechanisms for both dissolved and particle-associated forms of pollutants. These studies should include the use of tracers to track particle movements under different hydrologic regimes.
2) An improved understanding of the dynamics and kinetics of sediment resuspension in the subembayments of the Estuary is

needed, with particular reference to the seasonal resuspension of deposited sediments containing substantial loads of polluted materials.

3) Sediment transport (bedload and suspended) among the subembayments of the Estuary, as well as the residence times for water, sediment, suspended sediments, and pollutants in definable compartments of the Estuary should be better characterized.
4) The sedimentary cycle of the Estuary traps, transports, and releases pollutants. Improved understanding of these processes will require further analysis and evaluation of partitioning of pollutants, studies of adsorption-desorption phenomena that occur in the sediments, and the transport of sedimentary-derived pollutants among the subembayments of the Estuary.

4. Pollutant Effects

Water quality objectives are intended to limit the potential for pollutants in the Estuary to affect beneficial uses of the system. Information available from the Estuary does not at present provide a clear understanding of pollutant interactions with the biota; indeed, questions still remain with regard to such fundamental questions as adsorption/desorption kinetics of pollutants on sediments, bioavailability of pollutants to biota, biotransformation of pollutants in the Estuary, and the biological effects of the pollutants of concern on resident species. Some of the most important questions regarding pollutant accumulation and pollutant effects are addressed below.

A. POLLUTANT ACCUMULATION

Each of the pollutants of concern listed in this report have the potential to alter significantly the biological communities of the Estuary through chronic effects on organisms, or by affecting such basic functions as reproduction, growth, resistance to disease, and resistance to predation. Such chronic and sub-chronic effects require that organisms accumulate pollutants over time, either from the water, from the sediments, or through ingestion of a polluted diet. It is recommended that research and monitoring efforts be continued, or accelerated, as indicated below:

1) Rates and routes of pollutant accumulation by the biota in the Estuary require further study, with particular emphasis placed upon establishing, through biomonitoring studies, the bioavailability of pollutants in the water column.
2) For estuarine sediments, the bioavailability of pollutants of concern from deposited and suspended sediments under a variety of conditions should be determined.
3) An improved understanding of the nature and extent of food-chain transport (trophic transport and food-web amplification) of pollutants of concern in the Estuary is needed, with particular reference to metals not physiologically regulated (cadmium, lead, and silver), recalcitrant organics (PCBs, toxaphene, DDT/DDE), and those

organics that are readily biotransformed, but may exert their effects by the activity of their intermediate metabolites (MAHs, PAHs, dioxins).

B. POLLUTANT EFFECTS

As with virtually every other estuarine/marine system currently under study, clear evidence linking pollution with specific biological effects is lacking in the San Francisco Estuary. Although such evidence is not absolutely required to manage pollutants in the Estuary, establishing such relationships would represent a valuable improvement in the scientific basis for management. It is recommended that current studies aimed at determining the direct effects of pollutants of concern in the Estuary continue, and that additional efforts be undertaken, or strengthened, as follows:

1) Water column bioassays in all parts of the Estuary should be continued, with an increasing emphasis on determining the identity of toxicants in specific discharges (urban runoff, agricultural drainage) that are responsible for observed toxicity, and that may threaten resident flora and fauna.
2) The occurrence of chronic or sub-chronic impacts in the biota of the Estuary as the result of exposure to pollutants should be studied. In this regard, particular emphasis should be placed upon developing, or assisting in the development, of realistic biological indicators of pollutant effects, such as genotoxic effects, physiological effects, and effects at the level of the immune system.
3) Correlative evidence of pollutant effects (e.g., MAHs in striped bass and PCBs in starry flounder and black-crowned night-herons) require confirmation through additional field and laboratory investigations.
4) Sediment bioassays, and synoptic approaches such as the triad concept and apparent effects thresholds, employed in determining the toxicity of polluted sediments to organisms, require refinement. In particular, the effects of extraneous factors such as sediment grain size, organic carbon content, and presence of hydrogen sulfide need to be carefully assessed.
5) The relationship between the accumulation of complex mixtures of pollutants and their effects on the biota of the Estuary should be further investigated.

These recommendations for additional study are aimed at reducing the considerable uncertainty that presently exists regarding the status and trends of pollutant abundance and effects in the Estuary. They are in no way intended to delay or confound the development of policies aimed at protecting estuarine resources. Rather, it is hoped that improved information provided through the filling of important data gaps will enhance the effectiveness of regulatory decisions, and the vitality of this complex estuary.

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APPENDIX I

GENERATION OF THE CONTAMINANT MATRIX

A. CRITERIA EMPLOYED

(1) General significance of contaminants: This category is included because several of the contaminants listed are not considered "priority pollutants", either in the EPA sense or otherwise. It seems to make no sense to commit SFEP funds to the review of iron or manganese in the Estuary, for example, if these are not of significance (i.e. found to be toxic, or to affect beneficial uses) in estuarine ecosystems in general. General significance has been graded as H (high), M (moderate) or L (low).

(2) Extent of the database on the San Francisco Estuary: This category is self-explanatory. Where no (or very minor) data exist on particular contaminants, it is proposed that these areas be listed by the report as "datagaps" and their importance defined and discussed. The extent of the database for particular contaminants has been graded as E (extensive), S (significant), M (minor) or P (poor or non-existent).

(3) Whether the contaminant is widespread in the Estuary: Also self-explanatory. Responses to this category are graded Y (widespread) or N (not widespread).

(4) Whether the contaminant exhibits localized contamination ("hot-spots") in the Estuary: This is to some extent judgemental, depending on one's definition of a hot-spot. However, in most cases, responses are fairly clear-cut. Responses are graded Y (localized contamination exists) or N (no evidence for hot-spot existence).

(5) Whether the contaminant exerts detrimental effects on biological resources in the Estuary: Few definitive answers are possible in this category, and judgement calls are necessary in several cases. Responses are graded Y (adverse effects exist), P (effects may exist), or N (no evidence for effects, and the existence of such seems unlikely).

(6) Whether the contaminant affects other beneficial uses in the Estuary: In a few cases, contaminants probably affect beneficial uses other than biological resources; public health effects might be one example. Responses are graded Y (effects exist, or are likely to) or N (no evidence exists for effects and these seem unlikely).

B. GENERAL COMMENTS ON GROUPS OF CONTAMINANTS INCLUDED IN THE MATRIX

Metals

Most of the metals listed which are on the EPA Priority Pollutants list may be considered to be of potential toxicity in estuarine ecosystems. Other metals listed which do not appear on the EPA Priority Pollutants list are often rare in occurrence in estuaries, or of low toxicity.

Pesticides

Most of the pesticides listed are organochlorines and are therefore persistent in aquatic environments and potentially important contaminants. Only some of these are thought to be widespread or of significant localized contamination in the Estuary, however. A few additional compounds have been added to the EPA list here, these being compounds which are thought to be of local significance because of their use on crops in the Delta or the upstream catchment. Organotins are included elsewhere (under metals).

Organics- General

"Oil and grease" is suggested to be such a non-specific term that its consideration should be dropped from the STR. PAHs as a group are thought to be of importance in the Estuary, and several of these are listed individually elsewhere. "Persistent organohalogens" and "petroleum hydrocarbons" are generic terms also thought to be covered by the inclusion of specific contaminants elsewhere in the list.

Low and High Molecular Weight Aromatic Hydrocarbons

All the contaminants listed are PAHs; all are thought of potential significance in the Estuary, although the existing database is sparse. The EPA list has been slightly amended to produce a more concise listing of individual compounds in certain cases, but nothing has been dropped.

CONTAMINANT MATRIX

Criteria	1 Gen. Sig.	2 Data	3 Wide.	4 Local	5 Eff.1	6 Eff.2	Remarks
Metals							
Al	L	S	Y	?	N	N	
Sb	M	P	?	?	?	?	Datagap
As	M	S	I	Y	N	N	Include
Be	L	P	?	?	?	?	Datagap
Cd	H	E	Y	Y	P	Y	Include
Ca	L	S	Y	N	N	N	
Cr	M	S	Y	Y	?	?	Include
Cu	H	E	Y	Y	P	?	Include
Fe	L	S	Y	N	N	N	
Pb	M	S	Y	Y	N	N	Include
Mg	L	S	Y	N	N	N	
Mn	L	S	Y	N	N	N	
Hg	H	E	Y	Y	Y	Y	Include
Ni	M	S	Y	Y	?	?	Include
Se	H	E	Y	Y	Y	?	Include
Si	L	S	Y	N	N	N	
Ag	M	S	Y	Y	?	?	Include
Na	L	M	Y	N	N	N	
Th	M	P	?	?	?	?	Datagap
Sn	H	M	N	Y	Y	?	Include
Ti	M	P	?	?	?	?	Datagap
V	M	P	?	?	?	?	Datagap
Zn	M	E	Y	Y	N	N	Include

CONTAMINANT MATRIX (contd.)

Criteria	1 Gen. Sig.	2 Data	3 Wide.	4 Local	5 Eff.1	6 Eff.2	Remarks
Pesticides							
Aldrin	M	S	N	Y	?	?	Include
Dieldrin	M	S	N	Y	?	?	Include
Chlordane + derivatives	M	S	Y	Y	?	?	Include
Demeton	M	P	?	?	?	?	
DDT + derivatives	H	E	Y	Y	P	?	Include
Endosulfan + derivatives	M	S	N	Y	N	N	Include
Endrin + derivatives	M	S	N	Y	N	N	Include
Guthion	M	P	N	N	N	N	
Heptachlor + derivatives	M	S	N	Y	N	N	Include
HCH + derivatives	M	S	N	Y	N	N	Include
Malathion	M	S	N	Y	P	N	Include
Methoxychlor	M	S	N	N	N	N	
Mirex	M	M	N	N	N	N	
Parathion + derivatives	M	S	N	Y	N	N	Include
Toxaphene	M	S	N	Y	N	N	Include
HCB	M	S	N	Y	N	N	Include
Chlorbenside	M	S	N	Y	N	N	Include
Dacthal	M	S	N	Y	N	N	Include
Kepone	M	P	N	N	N	N	

CONTAMINANT MATRIX (contd.)

Criteria	1 Gen. Sig.	2 Data	3 Wide.	4 Local	5 Eff.1	6 Eff.2	Remarks
PAHs (Low and High Molecular Weight)							
Acenaphthene	M	M	Y	Y	P	?	Include
Acenaphthylene	M	M	Y	Y	P	?	Include
Anthracene	M	M	Y	Y	P	?	Include
Fluorene	M	M	Y	Y	P	?	Include
Naphthalene + derivatives	M	M	Y	Y	P	?	Include
Phenantherene + derivatives	M	M	Y	Y	P	?	Include
Benzo (a) anthracene	M	M	Y	Y	P	?	Include
Benzo (a) pyrene	M	M	Y	Y	P	?	Include
Benzofluoranthenes	M	M	Y	Y	P	?	Include
Benzo (g,h,i) perylene	M	M	Y	Y	P	?	Include
Chrysene	M	M	Y	Y	P	?	Include
Dibenzo (a,h) anthracene	M	M	Y	Y	P	?	Include
Fluoranthene	M	M	Y	Y	P	?	Include
Indeno (1,2,3-c,d) pyrene	M	M	Y	Y	P	?	Include
Pyrene	M	M	Y	Y	P	?	Include
Benzo (e) pyrene	M	M	Y	Y	P	?	Include
Miscellaneous							
Asbestos	L	P	N	N	N	N	
Cyanide	M	S	N	N	N	N	

C. OTHER GROUPS OF CONTAMINANTS IN EPA LIST

Phenols

All compounds listed generally only significant in conditions of direct exposure to high concentrations; all poorly characterized to date in the Estuary with the possible exception of effluents; none widespread; none likely to accumulate in sediments or biota significantly. **Conclude: Exclude from STR.**

Chlorinated Aromatic Hydrocarbons

Hexachlorobenzene (HCB) listed under pesticides above. All other compounds included in EPA list as noted for phenols above. **Conclude: Exclude from STR.**

Phthalates

All compounds listed poorly characterized to date in the Estuary; none likely to be toxicologically significant with respect to effects on beneficial uses. **Conclude: Exclude from STR.**

Polychlorinated Biphenyls

All mixtures of commercial products listed in EPA list are of extreme environmental concern. Most PCB profiles in the Estuary sediments and biota characteristic of Aroclor 1254, but a few exceptions exist. Database is extensive; PCBs are both widespread and present at high levels in particular hot-spots in the Estuary; adverse effects on biota are probable. **Conclude: Include in STR.**

Organonitrogen Compounds

All compounds listed poorly characterized to date in the Estuary; none likely to be toxicologically significant with respect to effects on beneficial uses. **Conclude: Exclude from STR.**

Chlorinated Aliphatic Hydrocarbons

All compounds listed poorly characterized to date in the Estuary; none likely to be toxicologically significant with respect to effects on beneficial uses. **Conclude: Exclude from STR.**

Halogenated Ethers

All compounds listed poorly characterized to date in the Estuary; none likely to be toxicologically significant with respect to effects on beneficial uses. **Conclude: Exclude from STR.**

Miscellaneous Oxygenated Compounds

Isophorene unlikely to be of toxicological significance in the Estuary. Data on dioxins are sparse for the Estuary, although these compounds are of extreme persistence and toxicity in the environment. **Conclude: Dioxins to be considered as a datagap.**

Volatile Halogenated Alkanes

All compounds listed generally only significant in conditions of direct exposure to high concentrations; all poorly characterized to date in the Estuary with the possible exception of effluents; none widespread; none likely to accumulate in sediments or biota significantly. **Conclude: Exclude from STR.**

Volatile Halogenated Alkenes

As noted above for alkanes.

Volatile Aromatic Hydrocarbons

These compounds (benzene, ethylbenzene, toluene, xylene, styrene) are thought to be of moderate significance generally as environmental contaminants. The database on these compounds in the Estuary is significant (at least in some respects). The first four of these compounds have been suggested to be implicated in the decline of striped bass populations locally, although the evidence for this is controversial. **Conclude: Include in STR. Local styrene levels may be considered as a datagap.**

Volatile Chlorinated Aromatic Hydrocarbons, Volatile Unsaturated Carbonyl Compounds, Volatile Ethers

All compounds listed generally only significant in conditions of direct exposure to high concentrations; all poorly characterized to date in the Estuary with the possible exception of effluents; none widespread; none likely to accumulate in sediments or biota significantly. **Conclude: Exclude from STR.**

Miscellaneous Extractables

Compounds of significant volatility are covered by comments above. Coprostanol thought to be a useful indicator of sewage-derived pollution, although relatively few studies have been performed in the Estuary on this compound. Phenanthrene, naphthylene and perylene derivatives included in list of PAHs above. **Conclude: Exclude from STR, with exception of those compounds elsewhere listed as significant.**

Nutrients

These have been added here, not being shown in the EPA list. These contaminants may be considered to be of general significance because of their bio-stimulation of primary productivity in aquatic ecosystems. A considerable body of data exists on nutrients and related parameters in the Estuary, and it is thought that nutrient enrichment of the Estuary was a contributing factor to the dissolved oxygen sags experienced (particularly in the South Bay) in the 1960s and early 1970s. However, subsequent to the improvement of sewage treatment facilities at that time, there has been little evidence that nutrient enrichment leading to eutrophication is a significant problem, either on a

regional or local scale. **AHI recommends that these contaminants not be included in the synthesis of data for the STR.**

Microbial Agents

Although not classical chemical contaminants *per se*, these have also been added to the EPA list. There is justified concern related to the microbial contamination of shellfisheries in the Estuary, and its possible impacts on public health. The database is moderate in quality only (including much data relating to total coliforms and other indicators not now thought to be of particular use), but we recommend that some resources be expended on this issue in the STR. We do not recommend that massive expenditure be incurred on computing loads/effects, however; rather, **we recommend that the present status of the microbial/shellfishery concerns be reviewed in a specific section of the STR, with management options for the future to be included.**

APPENDIX II

Table 1. Frequencies of detection for point sources.

POLYNUCLEAR AROMATICS (EPA METHOD 610)			
	TOTAL	DETECTED	FREQUENCY OF DETECTION
Acenaphthene	77	0	0.00
Acenaphthylene	77	0	0.00
Anthracene	77	1	0.01
Benzo (a) anthracene	77	0	0.00
Benzo (a) pyrene	77	0	0.00
Benzo (b) fluoranthene	77	0	0.00
Benzo (g,h,i) perylene	77	0	0.00
Benzo (k) fluoranthene	77	0	0.00
Chrysene	77	0	0.00
Dibenzo (a,h) anthracene	77	0	0.00
Fluoranthene	77	2	0.03
Fluorene	77	0	0.00
Indeno (1,2,3-cd) pyrene	77	0	0.00
Naphthalene	77	2	0.03
Phenanthrene	77	2	0.03
Pyrene	77	3	0.04
TOTALS	1232	10	0.01

Table 1. Frequencies of detection for point sources.

ORGANOCHLORINES (EPA METHOD 608)			
			FREQUENCY OF
	TOTAL	DETECTED	DETECTION
a-BHC	107	3	0.03
Aldrin	120	1	0.01
b-BHC	120	0	0.00
Chlordane	107	0	0.00
d-BHC	114	1	0.01
Dieldrin	122	1	0.01
Endosulfan I	92	1	0.01
Endosulfan II	96	0	0.00
Endosulfan Sulfate	109	1	0.01
Endrin	109	0	0.00
Endrin Aldehyde	111	0	0.00
g-BHC	107	7	0.07
Heptachlor	119	0	0.00
Heptachlor Epoxide	121	0	0.00
PCB-1016	93	0	0.00
PCB-1221	94	1	0.01
PCB-1232	94	0	0.00
PCB-1242	104	0	0.00
PCB-1248	92	0	0.00
PCB-1254	100	0	0.00
PCB-1260	100	0	0.00
Toxaphene	108	0	0.00
4,4'-DDD	119	1	0.01
4,4'-DDE	130	0	0.00
4,4'-DDT	125	0	0.00
TOTALS	2713	17	0.01

Table 1. Frequencies of detection for point sources.

SEMI-VOLATILE ORGANICS (EPA METHOD 625)			
			FREQUENCY OF
	TOTAL	DETECTED	DETECTION
Acenaphthene	338	4	0.01
Acenaphthylene	334	0	0.00
Anthracene	338	2	0.01
Benzidine	357	6	0.02
Benzo (a) anthracene	341	2	0.01
Benzo (a) pyrene	346	1	0.00
Benzo (b) fluoranthene	337	2	0.01
Benzo (g,h,i) perylene	340	0	0.00
Benzo (k) fluoranthene	341	0	0.00
Benzyl butyl phthalate	319	15	0.05
Bis (2-chloroethoxy) methane	339	0	0.00
Bis (2-chloroethyl) ether	339	0	0.00
Bis (2-chloroisopropyl) ether	340	4	0.01
Bis (2-ethylhexyl) phthalate	358	44	0.12
Chrysene	341	3	0.01
Di-n-butylphthalate	356	32	0.09
Dibenzo (a,h) anthracene	330	0	0.00
Diethyl phthalate	337	20	0.06
Dimethyl phthalate	359	5	0.01
Dioctylphthalate	339	7	0.02
Fluoranthene	337	3	0.01
Fluorene	344	2	0.01
Hexachlorobenzene	339	1	0.00
Hexachlorobutadiene	337	1	0.00
Hexachlorocyclopentadiene	339	6	0.02
Hexachloroethane	339	0	0.00
Indeno (1,2,3-cd) pyrene	336	1	0.00
Isophorone	343	11	0.03
N-Nitrosodi-n-propylamine	346	5	0.01
N-Nitrosodimethylamine	321	5	0.02
N-Nitrosodiphenylamine	330	14	0.04
Naphthalene	350	26	0.07
Nitrobenzene	337	0	0.00
p-Chloro-m-cresol	241	3	0.01
Pentachlorophenol	239	13	0.05
Phenanthrene	342	8	0.02
Phenol	344	36	0.10
Pyrene	337	3	0.01
Trichlorofluoromethane	117	1	0.01
1,2-Dichlorobenzene	203	36	0.18
1,2-Diphenylhydrazine	313	6	0.02
1,2,4-Trichlorobenzene	333	0	0.00
1,3-Dichlorobenzene	204	6	0.03
1,4-Dichlorobenzene	340	34	0.10
2-Chloronaphthalene	338	1	0.00
2-Chlorophenol	335	2	0.01

Table 1. Frequencies of detection for point sources.

VOLATILE ORGANICS (EPA METHOD 624)	TOTAL	DETECTED	FREQUENCY OF DETECTION
Acrolein	332	29	0.09
Acrylonitrile	332	29	0.09
Benzene	390	22	0.06
Bromodichloromethane	251	120	0.48
Bromoform	368	41	0.11
Bromomethane	358	4	0.01
Carbon Tetrachloride	367	1	0.00
Chlorobenzene	371	3	0.01
Chloroethane	359	2	0.01
Chloroform	392	271	0.69
Chloromethane	397	19	0.05
Dibromochloromethane	374	96	0.26
Dichloromethane	300	134	0.45
Ethyl Benzene	377	21	0.06
Tetrachloroethene	310	120	0.39
Tetrachloroethylene	76	13	0.17
Toluene	385	84	0.22
Trans-1,2-dichloroethene	302	22	0.07
Trichloroethene	311	47	0.15
Trichloroethylene	68	5	0.07
Vinyl Chloride	364	7	0.02
1,1-Dichloroethane	333	0	0.00
1,1-Dichloroethene	319	9	0.03
1,1-Dichloromethane	106	7	0.07
1,1,1-Trichloroethane	389	85	0.22
1,1,2-Trichloroethane	364	0	0.00
1,1,2,2-Tetrachloroethane	368	3	0.01
1,2-Dichloroethane	368	8	0.02
1,2-Dichloropropane	368	4	0.01
1,3-Dichloropropene	342	1	0.00
2-Chloroethylvinyl Ether	363	2	0.01
TOTALS	10104	1209	0.12

Table 1. Frequencies of detection for point sources.

SEMI-VOLATILE ORGANICS (EPA METHOD 625)			
	TOTAL	DETECTED	FREQUENCY OF DETECTION
2-Nitrophenol	337	2	0.01
2,3,7,8-Tetrachlorodibenzo-p-dioxin	70	0	0.00
2,4-Dichlorophenol	332	3	0.01
2,4-Dimethylphenol	347	6	0.02
2,4-Dinitrophenol	355	0	0.00
2,4-Dinitrotoluene	335	0	0.00
2,4,6-Trichorophenol	330	6	0.02
2,6-Dinitrotoluene	332	2	0.01
3,3-Dichlorobenzidine	358	0	0.00
4-Bromophenyl phenyl ether	338	3	0.01
4-Chlorophenyl Phenyl Ether	329	0	0.00
4-Nitrophenol	357	1	0.00
4,6-Dinitro-o-cresol	250	0	0.00
TOTALS	18983	396	0.02

APPENDIX 3

GOALS AND MANAGEMENT ACTIONS TO ADDRESS POLLUTION IN THE SAN FRANCISCO ESTUARY

APPENDIX III

GOALS AND MANAGEMENT ACTIONS TO ADDRESS POLLUTION IN THE SAN FRANCISCO ESTUARY

The attached Goals and Management Actions are intended to address activities that cause pollution within the catchment of the San Francisco Estuary; and to establish a program of monitoring, data management, and research that will improve the scientific basis for managing the Estuary. The recommendations were developed by the Subcommittee on Pollutants and Quality Assurance of the San Francisco Estuary Project (SFEP). Subcommittee members represent a diverse array of organizations including environmental groups, wastewater dischargers, government agencies, and research institutions.

In March 1990, the Subcommittee began developing their recommendations to capitalize upon the findings of the Status and Trends Report on Pollutants in the San Francisco Estuary (Pollutants STR). Before approving the Goals and Management Actions by consensus in January 1991, the Subcommittee met often to scrutinize and revise their recommendations. While much work has been devoted to developing the Goals and Management Actions, the language contained in the attached package should not be construed as final nor all inclusive. Instead, the document is offered to stimulate public dialogue and participation.

The package is divided into four key management areas: Environmental Protection, Monitoring, Data Management, and Research. It is then subdivided into categories of Immediate and Potential Actions. Immediate Actions are defined as tasks that the Subcommittee agrees should be initiated before the Comprehensive Conservation and Management Plan (CCMP) is completed in November 1992. A number of the recommended Immediate Actions are already underway; they are acknowledged here to provide a reference point for new regulatory initiatives. Potential Actions are defined as recommendations that will be evaluated for possible inclusion into the CCMP. Also, the Potential Actions category includes recommendations that were proposed as Immediate Actions, but did not receive the Subcommittee's consensus approval.

Following public workshops on the STR, the Subcommittee and SFEP staff plan to begin identifying funding sources and developing implementation strategies that will translate the Goals and Management Actions into environmental programs that will outlive SFEP's five-year planning period.

I. ENVIRONMENTAL PROTECTION

A. Goal

Adopt an "ecosystem approach" for protecting water and sediment quality in the Estuary from degradation. This includes preventing pollutant discharges, preventing the transfer of pollutants from one medium to

another (e.g. from water to land), and ensuring that all food resources harvested from the Estuary are safe to eat.

B. Immediate Actions (environmental protection)

Pollution Prevention

1. Establish an Estuary-wide pollution prevention program.

   a. As part of a regional pollution prevention program, the San Francisco Estuary Project (SFEP) should recommend ways for public and private facilities to routinely incorporate the following strategies:

      i. Redesign or reformulation of products;

      ii. Substitution of raw materials that introduce smaller quantities of hazardous substances into production processes;

      iii. Improved process technology and equipment to alter the primary source of waste generation;

      iv. Improved plant operations (housekeeping); and

      v. Recycling of polluted substances at the site of its generation (closed loop recycling).

   b. SFEP should ensure that all levels of government incorporate pollution prevention measures into their planning and enforcement programs.

   c. The California Regional Water Quality Control Boards (Central Valley and San Francisco Bay Regions [RWQCBs]) should make pollution prevention audits mandatory for all industrial facilities that discharge toxic pollutants into the Estuary directly (under National Pollutant Discharge Elimination System [NPDES] permits), or indirectly (through Publicly Owned Treatment Works [POTWs]).

2. The RWQCBs should pursue a mass emissions strategy to reduce pollutant discharges into the Estuary from point and nonpoint sources, and to address the accumulation of pollutants in estuarine organisms and sediments. This strategy should include the following recommendations:

   a. All point source dischargers should participate in a pollution prevention program.

b. POTWs should require all dischargers (industrial, commercial, and residential) to implement pollution prevention measures, and to include mass-emission limitations in their industrial pretreatment programs.

c. SFEP should help fund, and the RWQCBs should implement, wasteload allocation projects for all water bodies in the Estuary that do not meet water quality standards for pollutants.

3. The U.S. Environmental Protection Agency (EPA), the State Water Resources Control Board (SWRCB), and the RWQCBs should implement all existing regulatory mechanisms needed for permit and enforcement actions including the following:

   a. Adopt appropriate water quality standards for the Estuary to protect the early life stages of sensitive aquatic species (see Potential Action "C.1." that did not receive Subcommittee consensus).

   b. Impose numerical effluent limitations and toxicity control requirements to assure compliance with adopted standards.

4. SFEP, in conjunction with the RWQCBs and SWRCB, should recommend the institutional and financial changes needed to place more focus on pollution prevention, including the redirection and/or augmentation of existing staff resources. This should include strategies for increasing staff resources for pollution prevention programs.

Urban Runoff

5. The California Legislature should enact a state law that requires all new urban developments and significant redevelopments, both commercial and residential, to incorporate pollution control measures which meet set performance standards for preventing and controlling the discharge of toxics from urban runoff to the maximum extent practicable.

6. The RWQCBs should work with local management agencies (counties and cities) to develop, implement, and enforce the following programs (pursuant to existing state and federal requirements) to control pollutants in urban runoff:

   a. Baseline control programs with a focus on prevention throughout all watersheds of the Estuary. The programs should include the following elements:

      i. Improve the operation and maintenance of existing and planned storm drain systems.

ii. Educate public and private entities regarding the proper use of polluting substances, the proper disposal of waste, and the correct practices for controlling urban runoff.

iii. Eliminate illicit connections and illegal dumping into storm drain systems.

iv. Implement controls on new development before, during, and after construction.

b. Comprehensive control programs with a focus on prevention and remediation in the urbanized areas of the Estuary. In addition to baseline control elements, these programs should include the implementation of stormwater management programs designed to reduce pollutants in runoff. NPDES permits should be issued requiring the implementation of these comprehensive programs, and should include transportation entities as responsible parties.

7. SFEP, in conjunction with the RWQCBs and SWRCB, should recommend the institutional and financial changes needed to place more focus on urban runoff, including the redirection and/or augmentation of existing staff resources. This should include strategies for increasing staff resources for urban runoff programs.

Nonurban Runoff

8. The Central Valley RWQCB, in conjunction with the California Department of Food and Agriculture, the California Department of Fish and Game, the U.S. Soil Conservation Service, and other related agencies should work to eliminate toxic pesticide discharges into the Estuary from commercial agriculture.

9. The Central Valley RWQCB should identify and control the major nonurban pollutant sources contributing to riverine loads.

10. SFEP, in conjunction with the RWQCBs and SWRCB, should recommend the institutional and financial changes needed to place more focus on nonurban runoff, including the redirection and/or augmentation of existing staff resources. This should include strategies for increasing staff resources for nonurban runoff programs.

C. Potential Actions (environmental protection)

1. EPA, SWRCB, and the RWQCBs should immediately adopt water quality standards for the Estuary based on appropriate California Ocean Standards and/or EPA criteria to protect the early life stages of sensitive aquatic species.

2. The State should establish a program to provide small and medium-sized businesses with the technical assistance needed to help them conduct pollution prevention audits and implement waste reduction measures.

3. Water supply agencies should identify and evaluate alternatives to the use of zinc orthophosphate and copper sulfate in water supplies and reservoirs.

4. The RWQCBs, in conjunction with SWRCB, should analyze potential incentive programs to encourage the reduction of pollutants in urban runoff. Incentives could include a watershed "utility tax", a waste oil law (see New York State's Environmental Conservation Law), or a "deposit" system for motor oil. Implement programs deemed most effective.

5. Evaluate alternatives to eliminating government subsidies to "one passenger commuter cars" by pricing automobile travel (e.g., increased tolls or gas taxes) to reflect the estimated costs of automobile-related environmental damage and corresponding mitigation costs. Collected revenues should be used to support mitigation projects (e.g. public transportation, pollution prevention, and habitat restoration).

6. Integrate Estuary-wide land-use planning with water quality planning. Specifically, promote and support local ordinances, zoning measures, and permit restrictions aimed at: (a) reducing urban sprawl, (b) focusing development near public transit corridors, and (c) preventing construction activities from degrading estuarine water quality, and (d) protecting sensitive aquatic sites.

7. Inform residents about commercial products known to contain potential "pollutants of concern", and support expansion of household hazardous waste collection programs.

8. Evaluate the merits of instituting Estuary-wide controls on the sale and application of chemicals that adversely affect the estuarine ecosystem.

9. Evaluate incentives and compensation provisions of the Food Security Act and Agricultural Credit Act that could be used to conserve soils on erosion prone uplands and wetlands currently used for range, pasture, and agricultural purposes. Also, evaluate economic incentives for promoting water conservation and reclamation, and reducing the discharge of polluted farm drainage into the Estuary.

10. Request that the proper authorities annually evaluate the effectiveness of current "spill-prevention" plans for chemicals being refined, stored, transported, used, and disposed of Estuary-wide. Recommendations should be made to the proper authorities to enhance consistency and enforcement of the various plans.

11. Investigate whether tightening restrictions on the refining of high-selenium crude oil in the Bay Area would significantly reduce the amount of selenium entering the Estuary. Implement restrictions if appropriate.

II. MONITORING

A. Goal

Establish multimedia monitoring programs for the Estuary that help characterize its critical processes, provide a better understanding of the Estuary as an ecosystem, and assess the effectiveness of regulatory and enforcement measures.

B. Immediate Actions (monitoring)

1. SFEP should form a Committee for Research, Monitoring, and Data Management to design and implement Estuary-wide multimedia monitoring programs (that serve as an "early warning system" for environmental problems); and to oversee the implementation of an Estuary-wide Quality Assurance/Quality Control (QA/QC) program. Implementation of the programs would involve the following tasks:

   a. Develop rigorous QA/QC protocols for collecting and analyzing samples, and reporting monitoring data.

   b. Ensure that laboratories participate in interlaboratory quality assurance (intercalibration studies), and select appropriate methodologies to enhance analytical precision and accuracy consistent with agreed-upon monitoring goals and data quality objectives resulting therefrom.

   c. Support the development and validation of models that predict the fate and effects of pollutants, including the validation of important parameters of pollutant fate and transport through appropriate laboratory and field measurements (i.e., independent of the models).

   d. Characterize the type, abundance, temporal and geographical distribution, and biotoxicity of pollutants contributed by Estuary-wide sources; if possible, estimate the effects on the estuarine ecosystem.

   e. Assess compliance with, and evaluate the effectiveness of, relevant water quality standards, federal and state regulatory and enforcement measures, the Committee's monitoring and QA/QC programs, and other SFEP Management Actions.

   f. Contribute technical information for setting new standards for water,

sediment, and tissues.

g. Prepare a regular, descriptive report on the results of the monitoring programs. This report should include a comparison of the Estuary's pollutant levels with (i) national and international environmental standards and criteria, and (ii) pollutant levels found in other estuaries.

2. The RWQCBs should require commercial agriculture to monitor discharges of tailwater to provide better characterization of pollutant loads.

3. When issuing or reissuing NPDES permits, the RWQCBs should assure that all point sources are characterized for all pollutants of concern identified in SFEP's Status and Trends Report on Pollutants in the San Francisco Estuary (Pollutants STR).

4. The California Department of Health Services (DHS) should continue to assess the potential human health effects resulting from consumption of the estuary's fish, shellfish, and wildlife. Also, DHS should identify and study exposed human subpopulations.

5. EPA (pursuant to Title III of the Superfund Amendments and Reauthorization Act), in conjunction with DHS, should attain and maintain awareness of existing and emerging pollutant sources and newly marketed, potentially toxic chemicals.

6. SWRCB should develop a stable funding base for an Estuary-wide monitoring program.

C. Potential Actions (monitoring)

III. DATA MANAGEMENT

A. Goal

Establish an accessible data management system for the Estuary that ensures the proper documentation and maintenance of information and technical literature, and the archiving of data sets that would not be accessible otherwise.

B. Immediate Actions (data management)

1. SFEP should form a Committee for Research, Monitoring, and Data Management to develop a plan for a data management system that is integrated with compliance monitoring programs, and acts as a repository for research data. The Committee should accomplish the following tasks:

a. Develop methods to (i) collect, format, and distribute "new" information; (ii) extract, format, and provide access to existing ("old") data bases maintained by industries, municipalities, and agencies; and (iii) input data into a Geographic Information System (GIS).

b. Establish criteria to ensure data in various locations are properly maintained and documented for future use in trends analysis. Important data sets that are not currently being maintained and documented in accordance with these criteria should be obtained and placed in a consolidated data base.

c. Acquire pertinent data sets and scientific abstracts that are presented in the standardized format, and compensate researchers as appropriate.

d. Identify appropriate State or Federal agencies, or non-governmental organizations to operate the data management system.

e. Establish an Advisory and Review Committee to periodically evaluate the effectiveness of the data management system.

2. RWQCBs should adopt a standardized computer reporting format, and use permit conditions to require all dischargers (direct and indirect) to submit Estuary-wide monitoring data in the standardized format.

IV. RESEARCH

A. Goals (research)

Prioritize and fill critical "gaps in knowledge" that will increase understanding of the Estuary's ecological processes and functions, and improve the scientific basis for protecting its resources.

B. Immediate Actions (research)

1. SFEP should form a Committee for Research, Monitoring, and Data Management to improve coordination and communication among research projects that increase understanding about the sources and fate of estuarine pollutants, effects on receiving waters and estuarine biota, and the potential human health risks associated with these pollutants.

2. The Committee should identify needed research projects and potential funding sources to support the work. The research should focus on four key elements of the pollutants issue: sources, fate, effects, and control strategies.

a. **Sources**

i. Identify source activities within the Estuary's catchment that produce and mobilize toxic pollutants of concern.

ii. Identify the specific pollutants in urban and non-urban runoff water that cause observed toxicity in stormwater conveyances, streams, and estuarine receiving waters.

iii. Investigate atmospheric deposition of particulates generated by mobile and stationary sources (e.g., automobile transportation, the spraying and burning of agricultural fields, industrial activities, and airport emissions).

iv. Identify and recommend analytical methods that measure petroleum hydrocarbons in permitted discharges and water quality studies.

b. **Fate**

i. Develop an understanding of processes affecting the toxicity and bioavailability of pollutants in the Estuary including hydrodynamics, chemical transformation, and transport through various habitat types.

c. **Effects**

i. Evaluate existing bioassay procedures, and, if necessary, develop new approaches to relating bioassay results to sublethal effects caused by pollutant concentrations in tissues, sediments, and the water column.

ii. Assess pollutant effects on biotic functions in this Estuary (e.g., reproduction, growth, immunity, and energy transfer) by focusing on those pollutants, and sensitive organisms (or estuarine communities) that occur in the Bay-Delta.

iii. Conduct mesocosm exposures of "representative communities" to appropriate mixtures of the relevant pollutants of concern to determine whether observed field effects can be duplicated.

iv. Improve the basis for developing water quality and toxicity standards applicable to the Estuary based on estuarine water chemistry, sediment properties, human health criteria, and toxicity to aquatic organisms.

v. Establish criteria for sediment quality by identifying an analytical method that accurately reveals a "threshold"

concentration above which organisms are negatively affected (e.g., measuring concentrations in pore water, bulk sediment, and acid extracts).

d. Control Strategies

i. Identify and evaluate available control strategies (including best management practices, and best available technologies) that help reduce pollutant loading into the Estuary.

ii. Study the feasibility of implementing alternative technologies and strategies to prevent pollutant discharges at their source.

iii. Recommend administrative changes necessary to enforce implementation of existing control strategies, and encourage implementation of alternative technologies and strategies.

iv. Explore the feasibility of implementing regulatory standards based on measures of sublethal effects (i.e., biochemical, physiological, genetic, and behavioral) that can be shown to be early indicators of impairments to estuarine populations or communities.

v. Explore the feasibility of obtaining the lowest possible detection limits when conducting water quality studies, and permitting discharges.

C. Potential Actions (research)

1. EPA, SWRCB, and RWQCBs should initiate the development of site-specific water quality objectives for the South Bay. These objectives should be based on estuarine water chemistry, sediment properties, human health criteria, and chronic aquatic toxicity.

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[1] Page numbers in bold type denote passages where subjects are discussed in detail