Conceptual Model of Contaminant Fate on the Margins of San Francisco Bay

Draft Report

by

Craig Jones

Sea Engineering, Inc.

Donald Yee

Jay A. Davis

Lester J. McKee

Ben K. Greenfield

Aroon R. Melwani

Michelle A. Lent

San Francisco Estuary Institute

CONTRIBUTION
NO. 663

APRIL
2012

SAN FRANCISCO ESTUARY INSTITUTE

4911 Central Avenue, Richmond, CA 94804

p: 510-746-7334 (SFEI), f: 510-746-7300,

www.sfei.org

This report should be cited as:
Jones, C., Yee, D., Davis J. A., McKee, L. J., Greenfield, B. K., Melwani, A. R., and Lent, M. A. (2012). Conceptual Model of Contaminant Fate on the Margins of San Francisco Bay. Draft Report. Contribution No. 663. San Francisco Estuary Institute, Richmond, California.

Conceptual Model of Contaminant Fate on the Margins of San Francisco Bay

An RMP Technical Report

Craig Jones, *Sea Engineering, Inc.*

Donald Yee, Jay A. Davis, Lester J. McKee, Ben K. Greenfield, Aroon R. Melwani, and Michelle A. Lent *San Francisco Estuary Institute*

Executive Summary

Beneficial uses of San Francisco Bay are considered to be impaired by numerous contaminants. Some of the areas with the highest concentrations of these contaminants are in the shallow Bay margins, a result of historical and current discharges and disposal in various local tributaries and shoreline sites. Productive and valuable ecosystems are present in and rely upon the health of the margins. This report presents a Bay Margins Conceptual Model (MCM) to provide an assessment of our present state of understanding of the complex and interacting physical, chemical, and biological processes associated with contaminant fate on the margins, and a foundation for developing quantitative models in support of ecosystem management actions.

The primary management questions revolve around linkages between contaminant sources and pathways and their ultimate biological (human or wildlife health) impacts locally and regionally under various scenarios. Specifically for margins, the primary interests are:

1. Linking pathways to impairment: What are the sources, pathways, loadings, and processes leading to contaminant-related impacts in the Estuary?
2. Forecasting recovery: What are the projected concentrations, masses, and associated impacts of contaminants in the Estuary under various management scenarios?
3. Contribution to regional impairment: What is the contribution of contaminated Bay margin sites to regional impairment?

The report reviews ecosystem characteristics and processes to be considered in conceptual, empirical, and numerical modeling of the Bay and its margins. The major components of integrated quantitative modeling efforts generally include the following: 1) hydrodynamics, 2) sediment transport, 3) chemical transport and fate, and 4) biological processes. Sections on each of these components summarize key processes and factors to be considered and assess the adequacy of existing data and understanding to support modeling.

Our understanding and available data are richest for hydrodynamics. Hydrodynamic principles and tools applied to modeling other waterbodies can be and have been readily applied to the local ecosystem using relatively (compared to the other components) temporally and spatially extensive and detailed data sets. Sediment transport has also been modeled, albeit less robustly due to variation in parameters that are often difficult to measure at sites and also difficult to extrapolate to other sites. However, supporting data for model validation such as sedimentation trends and water column suspended sediments are generally available or can be obtained relatively inexpensively.

Principles behind various chemical transport and fate processes are well understood, but as for sediments, many of the process parameters or important cofactors are variable and difficult to

measure or extrapolate beyond sparse localized case studies. Chemical concentration data are often available but patchy and difficult to estimate for unsampled areas due to limited knowledge of sources. The modeling of biotic processes such as bioaccumulation presents formidable challenges, as organism life histories and structures of local food webs are poorly known and variable both spatial and temporally due to seasonal or interannual shifts in population and habitat usage by many species. Another key challenge will be identifying appropriate indicator species to track management progress for specific contaminants.

This review of available information and of previous modeling efforts in the San Francisco Bay and in other estuaries suggest that we need to and can move beyond our current highly simplified models of the Bay and its margins to adequately answer the priority management questions. Although no comprehensive model of processes in the margins has been developed to date, the information summarized in this MCM shows that previous modeling frameworks and datasets exist to model the Bay and its margins to a standard that will be useful to begin addressing the priority questions. Desirable attributes of a platform for modeling contaminant fate on the margins are described, and an overall conceptual strategy is outlined. The approach ensures any models developed will focus on the specific questions to be answered, and include sufficient realism, logistical feasibility, and integration with other regional modeling efforts to maximize cost-effectiveness. Model development can proceed iteratively, with sensitivity testing along the way to identify model components most in need of refinement or better data. Collection of additional empirical data will likely be essential to development of many model components, and ensuring both new and historical data are accessible and well-integrated will also be very valuable.

1 Introduction

There are a number of areas in San Francisco Bay that are on the 303(d) List due to elevated concentrations of one or more contaminants (PCBs, mercury, PAHs, selenium, and others). While there are biota with elevated tissue contaminant concentrations throughout the Bay, there are also localized zones of particularly high sediment and biota concentrations, where current or historical sources entered the Bay and deposited contaminants, or historical activity contaminated a site. The majority of these locations appear to be on the Bay margins - intertidal and shallow subtidal areas adjoining the Bay shoreline. In margin areas, landward and shoreline processes and sources play a substantial role in determining site characteristics. Productive and valuable ecosystems are present in and rely upon the health of the margins. The margins provide unique and diverse habitats that are often limited in areal extent and sensitive to both local and system-wide modifications. Additionally, urban and industrial development near the margins has often resulted in the margins acting as a reservoir for contamination, providing a potential pathway for transport of contamination into the wider Bay environment. For these reasons sediment and contaminant transport and fate on the margins are of primary interest, but understanding exchange with deeper subtidal areas is also important, as they convey sediment and associated contaminants among most margin regions. The overall goal of this document is to develop a Bay Margins Conceptual Model (MCM) to provide a framework for understanding physical, chemical, and biological processes associated with contaminant fate on the margins, providing a foundation for developing quantitative models in support of ecosystem management actions.

In general a major challenge to management is linking contaminant pathways to impairment in the Bay so that the ecosystem can be managed holistically. Sources of contamination from surrounding watersheds, historic contamination in the sediment, or transport of contaminants to the local area from other areas of the Bay are often the largest pathways to locations in the Bay and its margins. The contaminants are then introduced to the food web through environmental processes that are often site- and species-dependent. Once the significant pathways have been determined for particular locations and contaminants of concern, recovery forecasts under various management scenarios are needed in order to compare the effectiveness of alternative management strategies, such as monitored natural recovery, active watershed and wastewater management, or in the case of highly contaminated sites, active remediation of the site.

Conceptual and quantitative models are valuable tools for integrating knowledge of complex environmental processes in contaminated sites and regions. Previous monitoring and modeling strategies of the Bay developed to date for the RMP have focused on large spatial (basin scale) and temporal (multi-decadal) scale descriptions of physical, chemical, and biologic processes.

However, Bay margin areas could require modeling of various environmental processes at a finer temporal and spatial resolution, depending on the types of questions to be answered. In general, Bay margin modeling efforts will provide a crucial link between models of the watershed and the Bay as a whole. In the near-term, improved models are needed in support of management of nutrients and of mercury and PCB contamination through TMDLs and other efforts, with a continued need in the longer-term, given the immense value of the Bay ecosystem and continued need for its protection.

2 Management Questions

The primary audience for this MCM is regional environmental managers and stakeholders charged with protecting the beneficial uses of the Estuary ecosystem. The details of these management questions vary among contaminants, but generally, these questions revolve around linkages between contaminant source and loading pathways in the margins and their ultimate biological (human or wildlife health) impacts under various scenarios. It is important to note that these same management questions also apply to nutrient impacts, not just toxic pollutants.

4. Linking pathways to impairment: What are the sources, pathways, loadings, and processes leading to contaminant-related impacts in the Estuary?

Specifically for margins, the primary interest is in the nature and magnitude of various local sources and processes relative to other (e.g., regional subtidal) processes with respect to their impacts on the local environment (i.e., within the margins themselves).

5. Forecasting recovery: What are the projected concentrations, masses, and associated impacts of contaminants in the Estuary under various management scenarios?

Due to their proximity to watershed and shoreline contaminant sources, some margin areas are expected to be more impacted by contaminant loads than most deeper subtidal areas of the Bay, and thus also show more benefit from any management actions taken. Margin areas thus perhaps provide the best environments to predict and observe the outcomes of the implementation of various management alternatives.

6. Contribution to regional impairment: What is the contribution of contaminated Bay margin sites to regional impairment?

This parallels the first two management questions, but focuses on understanding and predicting the impacts of local margin processes and actions on the wider Bay ecosystem. Although the greatest benefits of management actions at the margins are expected to be seen in margins, the benefits of localized actions (either individually or in aggregate when implemented on a wide scale) may also show more subtle but wider-scale benefits.

This MCM report presents a framework for synthesizing available information on the physical, chemical, and biological processes on the margins. This document summarizes the present state of understanding that forms the basis for the MCM (Section 3) and identifies the greatest data gaps and uncertainties in our understanding. The report then discusses how this information has been incorporated into modeling efforts for both the Bay and other similar systems (Section 4), with the aim of identifying approaches with the greatest potential for addressing management questions. Lastly, the document presents recommendations for addressing knowledge gaps and uncertainties that are likely to affect local management decisions (Sections 5 and 6).

3 The Margins Conceptual Model

As a result of human activities, large masses of contaminants have been disposed of in and adjacent to surface waters throughout the Bay. Particle-associated contaminants are of greatest long-term management concern in the Bay due to their toxicity and persistence (PCBs, mercury, PAHs, PBDEs, and others). They are adsorbed to suspended sediment particles and subsequently deposited in bottom sediment where they can act as a continuing reservoir of environmental concern. There are also ongoing concerns about soluble contaminants such as nutrients and many emerging contaminants on the local ecosystem.

Figure 3-1 illustrates many of the important contaminant transport and fate processes in water bodies. In a common scenario, contaminants from upland areas were historically released into surface waters, often in margins. The contaminants were often sorbed to, or became sorbed to, particles that settled to the sediment bed. Subsequent physical (e.g., resuspension), biotic (e.g., bioturbation by benthic macroinvertebrates), and geochemical (e.g., dissolution) processes redistributed and degraded the contaminants. Through these processes, contaminants may remain available as long-term ecosystem stressors.

Simplifying Figure 3-1 and centering on the open Bay, connected margin and ocean habitats are shown linked in Figure 3-2. Inflows to margins come from upland watersheds, wetlands, and exchange with the open Bay. The interaction of processes in the margins results in storage and transport that govern the net exchange with the open Bay, and in turn Bay processes and exchange with the ocean govern the net transport of contaminants and biota risk.

It is important to be able to quantitatively predict the behavior of contaminants as a function of time, forecasting fate and exposure as much as 20 to 50 years or more into the future given the persistence of many of these contaminants. Other ecosystem stressors (e.g., nutrients) are also affected by most of the same or similar processes, so the MCM framework can be applied to management of non-hydrophobic substances as well. Conceptual models provide a foundation for the quantitative models needed for forecasting.

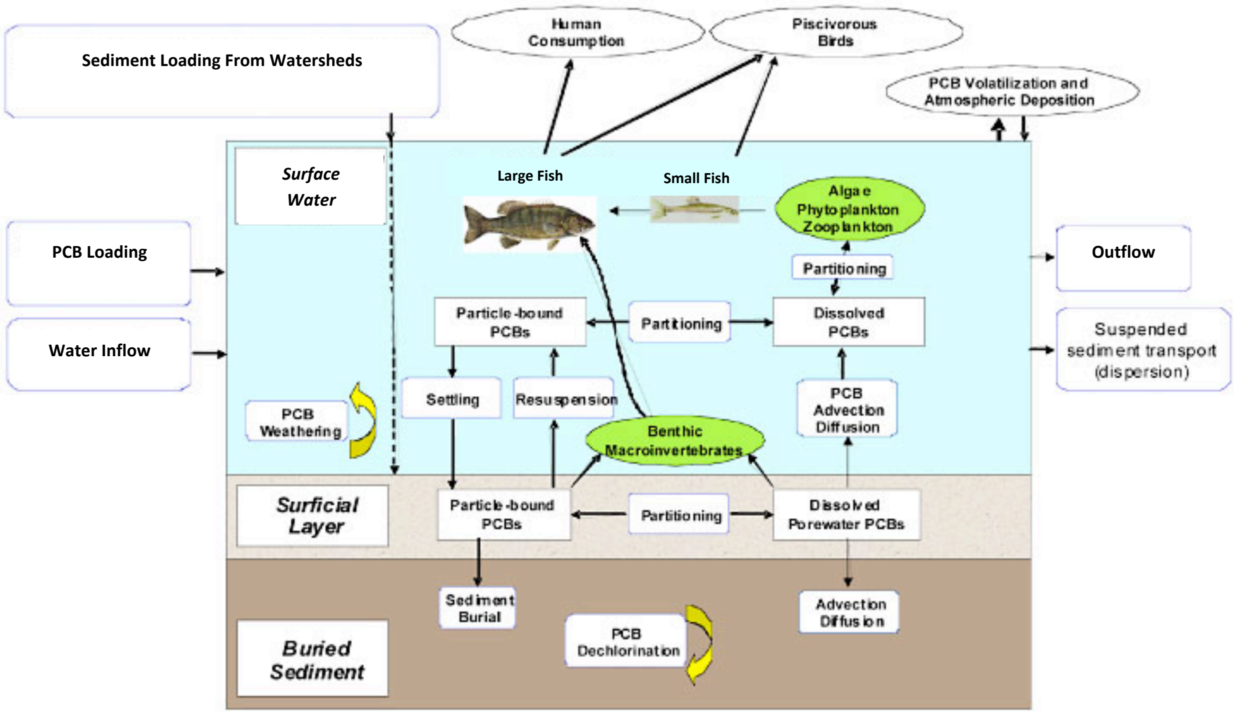

Figure 3-1. Conceptual diagram of PCB fate – adapted from a freshwater system, similar diagrams can be applied for other contaminants in Bay margins.

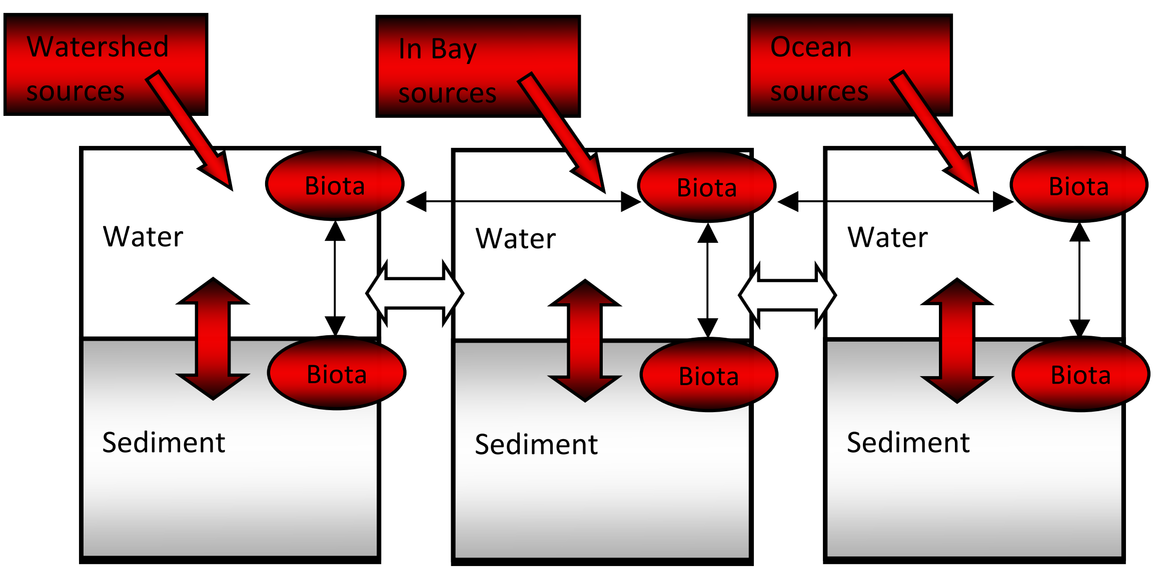

Figure 3-2. Diagram of Contaminant Transfer Within and Among Margins, Bay, and Ocean

Although there are numerous interacting processes governing contaminant fate and transport, quantitative models typically divide these processes into the following four classes.

- Hydrodynamics – Tidal circulation and tributary flows, wind-driven circulation, waves, and salinity transport.
- Sediment Transport – The loading of sediment to the system, transport of sediment, deposition of sediment, and erosion of sediment.
- Chemical Transport and Fate – The loading and partitioning of chemical constituents in water and particulate material, transformation, degradation, volatilization, and other biogeochemical processes associated with them. This may include a water quality model to fully describe processes important to contaminant fate.
- Biotic Processes – Interactions of contaminants with biota such as bioaccumulation (e.g., food web uptake), bioturbation, and any other biotic processes affecting overall contaminant fate.

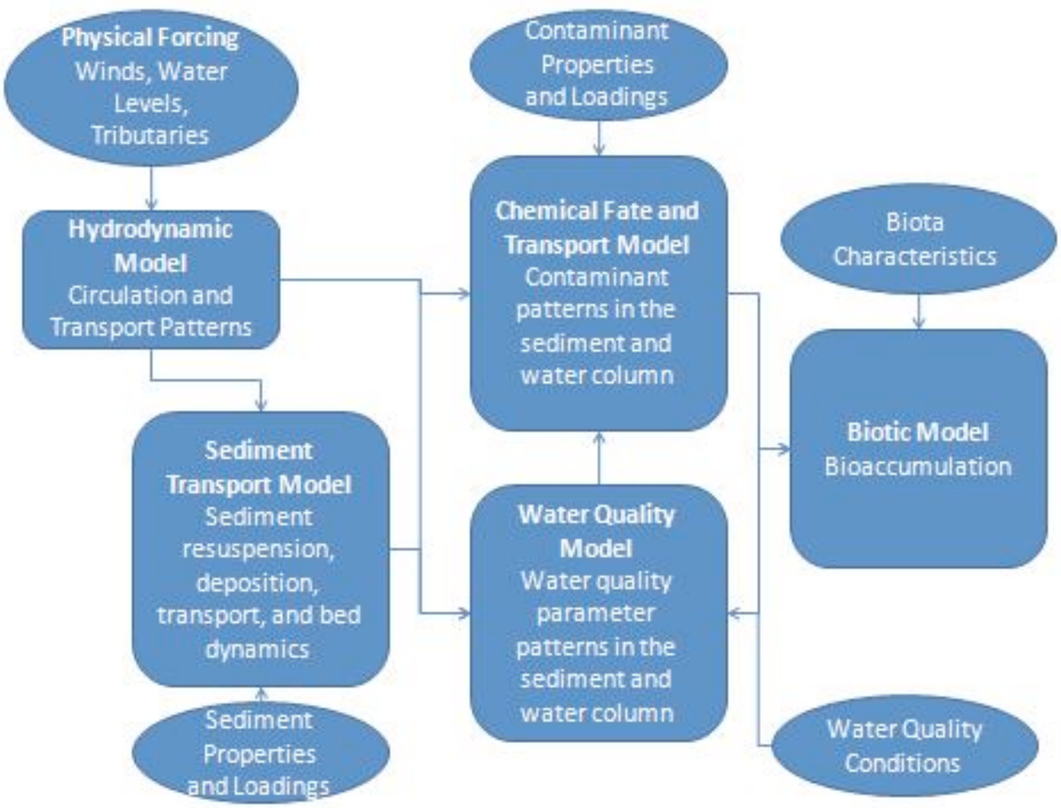

Figure 3-3. Diagram of linked ecosystem sub-models (rounded boxes) and data (ovals)

A typical contaminant fate and transport model will include sub-models for each class of processes (Figure 3-3). Each submodel simulates a subset of the relevant processes, passing along outputs to subsequent model components. These linkages may integrate dynamically, e.g., output for each time step of a hydrodynamic model driving a sediment transport model, which can then feed back and affect hydrodynamics in subsequent time steps. Alternatively, models can be linked off-line, e.g., hydrodynamics for an entire period modeled and used as inputs to a sediment transport model. Generally, dynamically linked models are either designed a priori to be used together, or ultimately require extensive adaptation to be made to integrate well. Linking models off-line provides more flexibility in model selection, but may result in less ability to capture tightly linked processes (e.g., transient effects of algal blooms on water column nutrients). The appropriateness of different modeling approaches is highly dependent on the questions being asked, so the understanding of these processes is important for both conceptual and numerical modeling.

The discussions in the following sub-sections roughly mirror the four general classes of processes, focusing on tracking the fate of contaminants from their introduction to the Bay margins to their ultimate uptake by biota.

- Water and sediment transport – Bay and margin physical characteristics, resulting hydrodynamics, sources and loadings of sediment and contaminants, sediment transport processes including erosion and deposition, and processes such as porewater diffusion, bioirrigation and bioturbation, and advection through groundwater flow or tidal pumping, which may be important to more hydrophilic contaminants.
- Biogeochemical cycling – processes affecting water and sediment chemical distributions and fate, including partitioning between air, water, and solid phases, and microbial or chemical transformations, which can convert contaminants into more or less accumulative or harmful forms.
- Bioaccumulation – concerns with bioaccumulation are primarily over net transfer to macrobiota, but understanding of this begins with understanding of the initial transfer from water and sediment matrices to the (often microscopic) base of the food web, and then transfer, accumulation, and loss processes through the food web.

These basic reviews are intended to facilitate effective development and improvement of data collection and modeling study programs. Each section includes an assessment of the adequacy of existing data and understanding to support quantitative modeling. The key processes of interest for any contaminant and/or location can be prioritized such that an efficient strategy for data collection and modeling can be developed. The final section of this report catalogs the information available to quantify the processes above, outlines an approach for identifying and prioritizing areas in which improved information is most needed, and provides a general strategy for obtaining that information.

3.1 San Francisco Bay Physical Characteristics

Typically, estuaries were formed during sea level rise since the end of the last ice age, when lower-elevation river valleys were drowned. An estuary formed in a drowned river develops a region in which ocean water is mixed landward and river water is mixed seaward (Dyer 1997). Drowned river estuaries are mixed by a combination of tides, waves, and river flows, and typically have a cross section varying from narrow river valleys near the landward head, to deeper wider sections towards the ocean. Often the width to depth ratio is large, with extensive mudflats and wetlands in the fringing areas (i.e., margins) where tidal circulation plays a key role in sediment distribution. The primary sediment source is fluvial, with the ocean as a secondary source. Sedimentation in drowned river estuaries generally keeps pace with inundation (i.e., sea level rise). Estuaries typically act as a sediment sink, or reach a dynamic equilibrium between inundation and sedimentation (Meade 1969). Anthropogenic actions, such as shoreline protection and dredging, also significantly alter these natural progressions (Perillo and Syvitski 2009).

The San Francisco Bay estuary is a prime example of such a system. The Bay developed over the past 10,000 years as rising sea level flooded the coastal outlet of the San Joaquin and Sacramento rivers, with sediment from the rivers depositing in the Bay and developing broad fringing margins along the shorelines. Sea level rise slowed and stabilized to its present rate of approximately 2 mm/yr, with extensive margin plains developing in this period of steady sea level rise (Atwater 1979).

The Bay is comprised of regions with differing characteristics: the North Bay into which the Sacramento-San Joaquin Delta empties 40% of California's watershed area; the Central Bay, which is connected with the Pacific Ocean via the Golden Gate; and the South Bay, draining a number of smaller local watersheds. The North Bay is divided into two sub-embayments by the narrow Carquinez Strait separating Suisun Bay to the east from San Pablo Bay. Similarly, South Bay is divided into South Bay and Lower South Bay by a constriction at Dumbarton Strait. The Bay overall is relatively shallow, with an average depth of approximately 6 m at MLLW, excluding the extensive intertidal mudflats and marshes above MLLW. The Golden Gate is the deepest point in the Bay, with water depths in excess of 100 m (Conomos 1977).

The following sub-sections outline the key transport processes governing the distribution of water, sediment, and contaminants in the subtidal Bay system as this is of critical importance to their transport in the Bay margins.

3.2 Physical Transport

3.2.1 Hydrodynamics

The hydrodynamics of San Francisco Bay and its margins are responsible for the dominant sediment transport patterns. Locally, wind waves and tributary flows can control erosion or deposition and movement of sediment, but large-scale tidal circulation is the dominant force behind the net movement of sediment. Extensive measurement and modeling programs have provided the foundation for an excellent understanding of hydrodynamic circulation in the Bay.

Driven by the interaction of fresh river water from the Delta and salt water from the ocean, the salinity structure of the Bay can be critical to circulation and sediment distribution. Systems such as the North Bay are classified as partially mixed (Pritchard 1955) , with freshwater from the Delta flowing over denser salt water from the Bay developing two distinct layers in the water column. This two-layer zone of estuarine circulation is typically located in the main channels of Suisun Bay during low Delta flows in the dry season (summer and early fall). In regions where the range of the tide is large compared to the water depth (e.g., the Bay margins), the turbulence in the water column is very high, mixing out any *vertical* salinity stratification into a homogenous water column. These regions often have a *horizontal* gradient from fresh to saline water that can cause differential horizontal circulation (Dyer 1997). On mudflats adjacent to the main channels (a typical component of the margins in Suisun Bay and other sub-embayments), the tidal action results in net horizontal circulation across the flats.

Generally, during dry periods there are small salinity differences between the Central Bay and South Bay, with no persistent stratification present in either embayment. In the dry season, the circulation in the Central Bay is tidally dominated, with efficient flushing and low residence times for water in that sub-embayment. In the wide and shallow South Bay, with relatively large margin areas, circulation is controlled by a combination of tides and northwesterly summer winds (Smith 1987; Schoellhamer 1996).

The zone of estuarine circulation in the northern Estuary is pushed seaward during high Delta flows during the wet season (winter and spring). The circulation in Suisun Bay and northeastern portions of San Pablo Bay are generally dominated by freshwater flow. Exchange and net transport seaward through Central Bay are substantially increased during high flow events. Low salinity water entering Central Bay may also induce gravitational circulation in South Bay, increasing exchange between the two embayments (Smith 1987). Salinity levels drop substantially in Central Bay and even into South Bay during very wet winters.

For more detailed descriptions of the hydrodynamics, extensive analytic and modeling studies have been conducted to investigate both the tidal hydrodynamics, wind, and salinity effects on circulation in San Francisco Bay at the whole Bay and sub-embayment scale (Cheng 1993; Gross

1999; Lacy 2000). These and other studies are readily available to provide more detailed information on Bay and margin circulation where needed.

The present understanding of the hydrodynamics in the Bay and its margins covers the dominant circulation processes observed, but details margin processes only coarsely or in aggregate aside from a few specific locations. The studies mentioned here are supported by extensive observational data and theoretical understanding of the data. Ongoing efforts by the USGS and SFEI continue to monitor key parameters (water level, salinity, velocity, tributary flow, and others) to document any present or future changes in the hydrodynamics of the Bay and its margins. Quantitative hydrodynamic modeling efforts have had excellent success (discussed later in Section 4) in simulating the overall circulation of the Bay. The present state of understanding is adequate to support future hydrodynamic modeling efforts at moderate scales (on the order of 100 m). Finer scale modeling efforts (on the order of 10 m) have shown success at site-specific scales, but generally are not extended to a full Bay scale due to computational and data demands.

3.2.2 Sources of sediment

Sources and characteristics of sediment are critical in the modeling of sediment and contaminant transport and fate in the Bay. Not only are historic and current loads important, but expectations for future loads are also required to model the future of Bay margins. Long-term studies monitoring suspended sediment contribute a large database for use in modeling efforts, with characterization of large infrequent events that can dominate the sediment loading to the Bay being particularly important given the potential for more extreme weather events with global climate change. In addition to sediment loads, sediment characteristics such as grain size, organic carbon content, and flocculation potential affect the transport, deposition potential, bed stability, and contaminant partitioning of sediment. In many cases these characteristics are key uncertainties in sediment and contaminant fate modeling.

Sediment generally enters the Bay directly through margins, and often carries contaminants of interest. Sedimentation events in the Bay Area are primarily seasonal, with a winter wet season of high Delta, tributary, and watershed flows and sediment loads (McKee et al. 2006), and a dry summer season with low flows and sediment input. The largest single source of sediment into the Bay (and thus its margins) is the outflow from the Sacramento-San Joaquin Delta. The Bay is essentially the endpoint for the vast watershed of the California Central Valley, and the Delta load was historically thought to be responsible for 80-90% of the annual average allochthonous sediment budget (Krone 1979). Intensive hydraulic gold mining in the Sierra Nevada range resulted in a large delivery of sediment in the late 1800s, much of it depositing in North Bay and gradually eroding away in the past century (Schoellhamer 2011). However, due to the passing of the mining sediment wave and water use and manipulation (dams and diversions) in the Central Valley, the Delta is now thought to contribute just 44% of the total fluvial load into the Bay, and is trending lower (Sabin et al. 2005; Schoellhamer 2005; McKee et al. 2006).

Delta sediment is predominantly inorganic in nature, with only 1-2% organic carbon (Schemel et al. 1996) and is mostly less than 20 microns in diameter (David Schoellhamer pers. com. 2011). A coarser-grained fraction delivered along the bed estimated to be 5% of the total load is also transported, but deposits soon after entering the Bay. The loads data available for the Delta are resolved at a daily time step for the period 1995-2010. Given that flow events from the Delta take many days to several weeks to pass through the system, this is likely adequate for modeling longer-term (e.g., multi-decadal) processes of contaminant transport and fate.

The Bay's local watersheds (with associated urban activities) contribute additional sediment in each sub-embayment. Although no individual local watersheds deliver anywhere near the quantity coming from the Delta, their combined loading is comparable to the loading from the Delta, and local tributaries are often a dominant influence on nearby Bay margins . Based on a reanalysis of data in local tributaries using a more sophisticated interpolation method combining hydrologic forcing and land use factors, one local study (Lewicki and McKee 2009) estimated local tributary fine sediment supply to the Bay to be 1.27 million metric t, 56% of the average annual load to the Bay.

These loads are relatively quickly delivered over periods of hours to a day during flood events by hundreds of small tributaries discharging to the Bay margins, where they are filtered by wetlands and settle over mud flats. Sediment loads from small tributaries have been spatially resolved for individual watersheds (Lewicki and McKee 2009), and temporally resolved down to a monthly time scale for the past 50 years with less confidence, and down to a daily time scale with much less (e.g., order of magnitude) confidence. Event timescales of hours are achievable for some tributaries but not easily developed for the whole Bay due to heterogeneity in watershed characteristics and precipitation within and between events. Most (80%) of the suspended sediment mass entering the Bay from small tributaries is less than 62.5 microns, with 65% less than 20 microns (McKee et al. 2002). The latest available (Porterfield 1980) estimate of bedload coarse sediment from the small tributaries (8% of annual load) may or may not be still valid given modern management of sediment in flood control channels.

The quantity of sediment introduced from the Pacific Ocean through the Golden Gate is less certain and is predominantly coarse sediment moving as bedload. Due to the coarse grain size of this sediment, aside from limited sandy areas (mostly in Central Bay), it is not likely a significant source in most Bay margin areas, where fine sediment prevail. The best estimate of gross input at the Golden Gate (mostly as coarse sand bedload) is approximately 2 to 3 times the Delta input to the Bay (Schoellhamer 2005), with an estimated annual suspended sediment (i.e., fine sediment) loss of approximately 30% of Delta, tributary, and watershed inputs (Conomos 1977). Coarse sediment influences sediment bed characteristics such as critical shear stress and suitability as habitat for some species, but for modeling of contaminant fate and transport, tracking fine particles is critical. Uncertainties in fine particle fate will have larger impacts on predicting long-term contaminant fate.

There are a few other smaller sources of sediment, which are generally dwarfed by Delta and local tributary loads. Municipal wastewater only supplies an average of about 0.007 million t sediment per year (McKee et al., 2008). Local production (e.g., autochthonous primary production of organic solids) and atmospheric deposition are smaller yet; they are unlikely to be important as sources of sediment except in very limited instances.

Presently, there is a good baseline understanding of the Delta and local watershed sediment loads most important to the Bay margins in many sub-embayments, yet many characteristics important to sediment fate are presently poorly known in many locations. As future modeling efforts progress, efforts should be made to better understand uncertainties and impacts of sediment loads and characteristics on model outcomes. Model sensitivity testing on test case sites can illustrate the importance of these parameters. Modeling studies themselves will ultimately help in determining whether the present understanding of sediment loading and characterization is adequate. Section 4 discusses model development considerations in more detail.

3.2.3 Sources of Contaminants

Contaminant loads are delivered to the Bay via a range of pathways (McKee et al., 2008) including the Delta, local tributaries, municipal wastewater, industrial wastewater, atmospheric deposition, and releases from contaminated sites near the Bay margins. Information on contaminant loads from these pathways is most rich for mercury and PCBs (McKee et al., 2008) mainly because of the TMDLs that have been developed for each of these substances.

3.2.3.1 Contaminated Sites on the Margins

During much of the period of industrialization in the region, the San Francisco Bay margins were treated as wastelands, rather than valued environmental resources. Numerous landfills, waste discharges, and industries were located in the Bay margins, so many sites in these areas are contaminated and have required remediation. **Error! Reference source not found.** Figure 3-4 shows a subset of the major Superfund Sites (as designated by the USEPA) and Toxic Hot Spots (designated by the California State Water Resources Control Board [SWRCB]) located in the Bay margins. Additional sites have been characterized as part of the Bay Protection and Toxic Cleanup Program by the SWRCB and collaborating agencies (Hunt et al. 1998; Hunt et al. 1999).

Contaminant concentration data from site characterization studies are not readily accessible for many sites and are often available only in hardcopy versions of site reports. Data are seldom compiled into public web-accessible databases, but reports with tables of raw or summarized data are available for some sites (e.g., Lee et al. 1994; Daum et al. 2000; Battelle et al. 2005). Data from some of these studies of contaminant concentrations in San Francisco Bay sediment, including margin locations, have been compiled in the California Sediment Quality Objectives (SQO) database (Myre et al. 2006), and also the Query Manager/MARPLOT database developed by NOAA-NMFS Office of Response and Restoration (National Oceanic and Atmospheric Administration 2003).

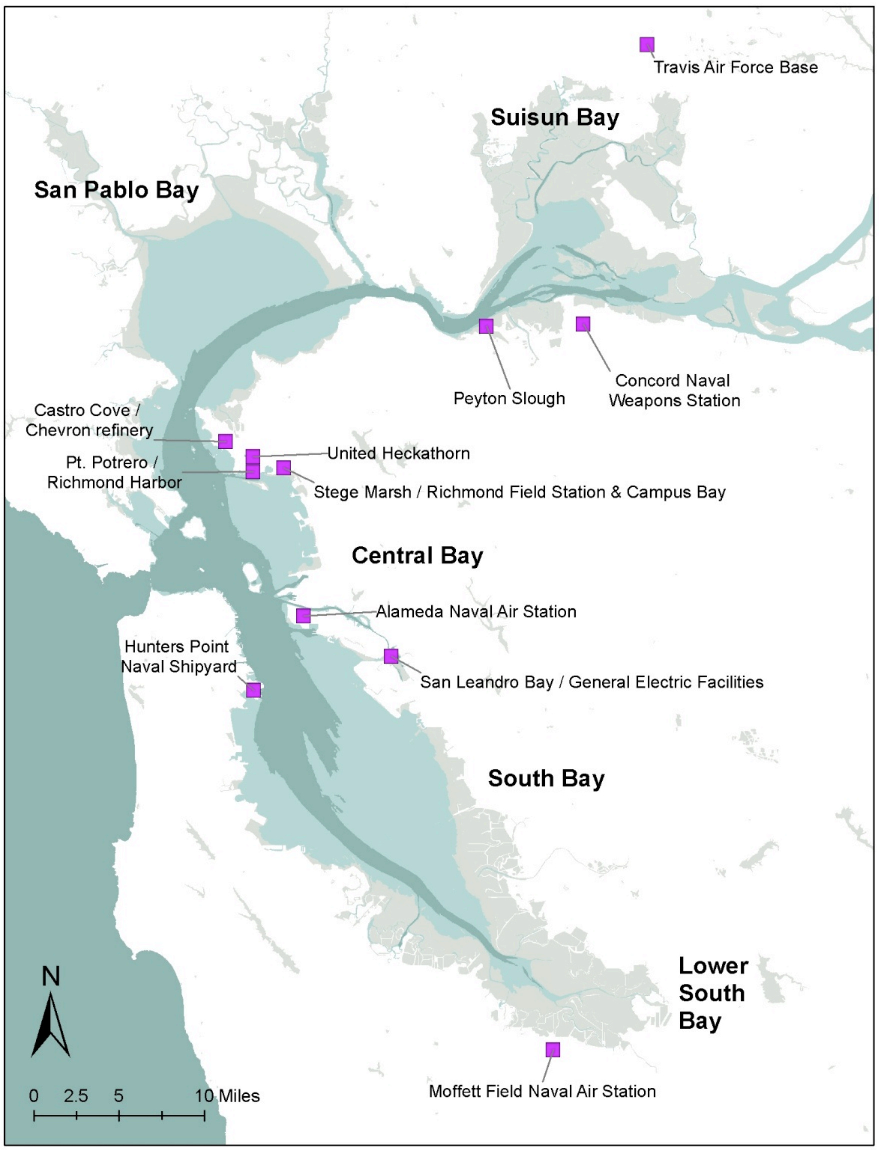

Figure 3-4 - Locations of San Francisco Bay margin contaminated sites (details in Appendix Table 1).

For sites where both areal extent and depth of contamination have been measured, it is possible to roughly estimate the volume of the contaminated sediment and the mass of the contaminant contained in the site. This allows comparison of contaminant inventories of different sites to overall Bay inventories, such as the estimate for mercury done for the Clean Estuary Partnership (URS Corporation 2002). Although legacy contaminants are generally widely dispersed throughout the Bay, some margin sites might benefit from more spatially limited, targeted remedial actions where local biota currently face relatively high risk due to local exposure. Such sites illustrate examples of scenarios likely to be widely encountered in margin modeling and are relatively likely to have sufficient data to use for model calibration or validation.

3.2.3.2 PCB Sources and Pathways

Available data suggest that the majority of *de novo* PCB loads to the Bay are delivered via local tributaries draining urban areas. Detailed data on PCB concentrations and loads are available over multiple years for three tributary locations. PCB loads are available for individual congeners, and estimates of the dissolved fraction can be modeled from concomitant data collected on suspended sediment and organic carbon concentrations (David et al. 2009). Loads have also been estimated for the Guadalupe River for WYs 2003-2006, and 2010, and in a small urban tributary in Hayward (Zone 4 Line A) for WYs 2007-2010. The Guadalupe data along with a minor data set collected on Coyote Creek at Hwy 237 during WY 2005 were used to make a regional estimate of average annual load to the Bay of 20 kg. Annual average spatially resolved loads are presently being developed by the RMP using a simple rainfall-runoff model.

Other pathways contribute much smaller PCB loads to the Bay. Data on PCB loads via atmospheric deposition are spatially and temporally limited, estimated for dry deposition in the North Bay only (Tsai et al. 2002), with small net loads to the Bay from the air compared to other pathways. Data on PCB concentrations and loads have been summarized in the PCB TMDL for nine wastewater treatment facilities and 14 industrial discharges (SFBRWQCB 2008), totaling 2.3 kg loaded to the Bay and adjacent margins each year. Redistribution from contaminated margin locations is largely uncharacterized; the frequency and severity of contaminated hotspots and their contribution to wider Bay contamination is known for few Bay margin locations. The importance of needs for better data on these smaller loading pathways, which can be the dominant loading to an adjacent margin, can be explored through sensitivity testing of loading, partitioning, and transport assumptions on model projections before engaging in further empirical measurements.

3.2.3.3 Mercury Sources and Pathways

Mercury is also of great concern to local managers due to extensive contamination resulting in bioaccumulation and risks to wildlife and human health. The Guadalupe River in Lower South

Bay is singled out as a special small tributary for total mercury loading because of the history of mining contamination adjacent to and downstream from the historic New Almaden mercury mining district. For the rest of the Bay, a larger proportion of total mercury loads enter via the Delta than for PCBs, although on a concentration basis, local urban tributaries still contribute a load of mercury disproportionate to their flow. Mercury concentration and load data were also collected for the same three tributaries as for PCBs. Annual average loads of total mercury entering the Bay from the Delta and Guadalupe River are presently estimated to be 218 kg and 115 kg respectively (SFEI 2010). The current best estimate of loads from all other small tributaries is 160 kg (Johnson and Looker 2006). However, this estimate was not spatially or temporally resolved, and mercury speciation is only now being collected.

Other loading pathways for Hg are smaller, but may have significant influence near specific Bay margin sites. Several efforts have quantified Bay Area atmospheric Hg fluxes (Tsai and Hoenicke 2001; Steding and Flegal 2002). Using these data, the annual mass loading of Hg to the Bay surface is estimated to be 27 kg, but with high uncertainty (Tsai and Hoenicke 2001). Available local data have not shown large gradients in atmospheric Hg concentration or deposition aside from in the very near-field (<1 km) of a large (cement plant) source (Rothenberg et al. 2010), so better-resolved atmospheric data are not needed for regional modeling of Hg fate, but might prove important for limited margin sites. Monitoring of municipal and industrial wastewater for Hg is routinely conducted, with estimated total annual loads of 17 kg/yr (Johnson and Looker 2006), and planned reductions to 11 kg/yr within 20 years. Spatial and temporal resolution and speciation is currently poorly developed for most of these pathways. Effects of localized sources on the Bay margins can be explored in model sensitivity testing, followed up with field monitoring if needed.

3.2.3.4 Other Contaminants

Knowledge of concentrations and loads for other contaminants such as PBDEs, organochlorine (OC) pesticides, PAHs, copper, selenium, nutrients, and dioxins is currently less advanced, with data available for few locations and years. Data on PBDEs are the best developed and were reported in a Bay mass balance (Oram et al. 2008), but not temporally or spatially highly resolved. OC pesticide data are available for Mallard Island, Guadalupe River and Zone 4 Line A, but only rough regional estimates have been developed (Connor et al. 2007). PAH and selenium have each had a regional-scale mass balance developed (Abu-Saba and Ogle 2005; Greenfield and Davis 2005), with more recent data available for Mallard Island and Zone 4 Line A in WY 2010 only. Nutrient and pyrethroid data are only available for Zone 4 Line A. Data on dioxins collected during WY 2010 in wastewater and storm water for the Delta, Guadalupe River, and Zone 4 Line A are presently being analyzed. The storm water data are congener-specific and estimates of particulate and dissolved fractions can be derived from SSC and organic carbon data that were also collected, which may help to validate and refine estimates in a previous assessment (Connor et al. 2004).

3.2.3.5 Improving Loading Estimates

RMP efforts for improving loading estimates in the near term will focus primarily on expanding the spatial coverage of tributaries measured for contaminants of greatest concern (i.e., those with existing or developing TMDLs). Widely distributed sampling of tributaries in the region will be followed up with more temporally intensive sampling of tributaries with high concentrations or discharging to biologically sensitive areas (including margins). Load estimates from the Delta can also be improved, particularly for extremely large storms, for which few samples have been measured using current highly sensitive analytical methods. The expansion in data on regional tributaries from these efforts will form a baseline for quantitative models of loading processes. Whether and how much more finer-scale data are needed for modeling is critically dependent on the very specific questions to be asked and decisions to be made using models; needs for mapping a site remediation would be vastly different from those for predicting results of distributed local control actions on margin biota at sub-embayment scales.

3.2.4 Sediment Transport

Due to the interplay of various physical forces in the Bay and its margins, the transport of sediment and associated contaminants occurs across a range of temporal and spatial scales. The following sections outline general sediment transport patterns in the Bay and characterize transport in the margins in more detail. Since the Delta and local watershed tributaries are the predominant sources of fresh water and suspended sediment, they provide a logical starting point for the characterization of transport. Many of the sediment transport processes described below first for the North Bay, also occur in other sections of the Bay.

3.2.4.1 Regional-scale sediment transport

3.2.4.1.1 North Bay

Water and sediment inputs from the Delta heavily influence seasonal patterns in sediment transport. The largest sediment loads to the North Bay occur during winter high flows from the Delta. Estuarine circulation is moved downstream and partially disrupted during these episodic events. A net seaward advection of sediment occurs in the channels, where the combined effects of tidal and riverine currents prevent the accumulation of sediment in the channels. Tidal circulation also disperses sediment into the margins. Lower currents in these regions allow sediment to deposit during receding and slack tides. The net deposition leaves a pool of fresh unconsolidated sediment in the margins.

During low flow periods, the strongest estuarine circulation is set up in the North Bay. Estuarine circulation at the confluence of the fresh and salt water transports sediment to the upstream portion of the salinity gradient. The zone of convergence between the fresh and saline water stalls the sediment in the water column, causing a zone of elevated sediment concentrations called the Estuarine Turbidity Maximum (ETM). The ETM is typically a region of high sediment accumulation and net sediment bed deposition. In the North Bay, the net flux of sediment is landward (i.e., towards Suisun Bay) from San Pablo Bay and into Carquinez Strait during low Delta flow periods (Ganju 2006). The resulting dynamic substantially reduces sediment loads to margins below the ETM during low flow periods.

In low flow conditions, peak tidal currents often resuspend sediment only in deeper channels, where there are higher peak velocities. The amount of sediment mobilized during a tidal cycle in most estuaries is typically a few millimeters (Sanford 1992). These sediments tend to settle back out in the channels, resulting in little net transport. Low flow conditions coincide with strong diurnal summer winds, which cause regular resuspension of sediment on the mudflats. Mudflats in San Pablo Bay show peak suspended solids with wind-wave induced resuspension. These combination of these processes results in a pattern of strong sediment delivery from the Delta and tributaries during high flow periods, with tidal flow in channels and wind–wave resuspension on mudflats redistributing the excess material during low flow periods (Ganju 2006).

3.2.4.1.2 Central Bay

The Central Bay is dominated by tidal circulation during low flow periods. It has the strongest tidal currents of the Bay, owing to the tidal prism of both North and South Bay being exchanged through it. Deeper waters and shorter residence times in Central Bay also cause it to have the lowest suspended sediment concentrations of any of the sub-embayments. Much of Central Bay acts as a tidal conduit to the ocean for the bulk of suspended solids entering from North and South Bay, except in its fringing tidal flats and marshes, of which Central Bay has the least areally. Similar to the North Bay, these margins generally accumulate sediment, although they experience relatively lower sediment loads, which could potentially reduce the long term fluctuations in erosion and deposition (Chin 2010).

Although most of the suspended solid transport is out to the Pacific during high Delta flow events, the landward bedload transport of coarse sediment via the Golden Gate is a significant transport mode in the Central Bay. Large sand waves have been observed in multiple surveys of Central Bay and indicate significant near bed sediment movement (Barnard 2007). Exchange through the Golden Gate has been examined by many researchers, yet there is large uncertainty in these estimates (Schoellhamer 2005). However, the coarse sediment involved seldom interacts with the shallow margins, even in Central Bay, and they generally contain low concentrations of most contaminants.

3.2.4.1.3 South Bay

In the relatively shallow South Bay with its broad mudflats, most of the area has a homogenous water column in all but the wettest years, when short-term salinity stratification may occur. Like the rest of the Bay, sediment transport in this region is seasonal, with tributary loads predominantly occurring during high flows, and redistribution by tidal and wind driven circulation among mudflats and channels in the dry season. The redistribution of sediment shows a net southeasterly flux of solids from Guadalupe River and Coyote Creek sources into the far southern reaches below Dumbarton Bridge (Lacy 1996). The same interactions takes place on a smaller scale for the other large tributaries throughout the Bay (e.g., Petaluma River (Schoellhamer et al. 2003)). Xx might be good to briefly mention that flux to LSB recently documented in the USGS fact sheet

3.2.4.2 Sediment Transport in the Margins

Processes that act upon sediment (both newly delivered from tributaries and redistributed from the sediment bed) in the Bay margins are spatially and temporally variable due to wind-wave exposure, elevation, and variation in the vegetation and sediment supply. The resulting morphology of the system reflects a dynamic balance of the relationships between all these processes.

Tidal circulation is the primary sediment transport mechanism in the margins. On flood tide, water fills into the channels, wets the mudflats, and fills into vegetated marshes. Sediment tends to deposit as the velocity slows, so a portion of sediment transported to the intertidal zone during flooding settles to the bed at high slack tide. Vertical salinity stratification is not a factor because of relatively high vertical mixing rates in shallow intertidal margins. Cohesive fine sediment can consolidate such that they cannot be resuspended on the subsequent ebb tide, resulting in net deposition. Vegetation enhances deposition due to its tendency to slow velocities and shield the sediment bed from erosion. Water velocities decrease through each developmental stage of a wetland due to increasing area and vegetative resistance. As the marsh fills in with sediment, the tidal prism exchanged decreases, further decreasing velocities.

Wind-waves are typically important to sediment transport on the outer exposed mudflats in areas such as San Pablo and South Bay. They can resuspend sediment much more effectively than tidal currents, making sediment available for subsequent transport by tidal currents. For example, the mudflats of San Pablo Bay receive loadings of sediment during the wet winter months from the Delta and smaller tributaries. Some of the sediment consolidates into the bed. Another portion is resuspended by wind waves and further dispersed by transport in tidal channels. The long-term net effects of these cycles shape the margins, and determine their overall exchange with the Bay. The highest net sediment accumulation rates in many margins are on protected mudflats and vegetated marshes closest to the channels.

The general conceptual model described here has been observed in many margin areas. However, the rates of transport on the margins are typically much slower than in the channels, so morphologic evolution predictions are generally required to determine patterns and long term trajectories of sediment transport (Leonard 1997; Friedrichs and Perry 2001). The channels in shallow margins are morphologically similar to rivers, with cut banks and point bars forming. They can migrate over time, changing the configuration of the margin. Although small local tributaries and urban outfalls act as sediment sources, they can also be responsible for local scouring during large events. Subsidence is another critical factor in margins. The high percentages of organic material generally create an environment that undergoes much more significant rates of subsidence than other sedimentary systems.

Anthropogenic modifications on the margins can either expand or reduce the tidal prism, which directly affects water velocities. Land reclamation has reduced the local marsh and mudflat areas in much of the Bay, leaving channel capacities much larger than necessary to convey the reduced tidal prism. The result is often lower tidal velocities and enhanced deposition in these regions. Margins are also sensitive to the reduced sediment loads that have been occurring in the Bay, with significant habitat loss as the system adjusts in areas of reduced supply. Over the long term, the balance of sediment sources, internal sediment budget, sea-level rise, and subsidence will determine the morphological stability of a margin (Perillo and Syvitski 2009), and all need to be considered in any quantitative assessment of transport in the margins.

3.2.4.3 Long-Term Trends

Since hydraulic gold mining in the mid-19th century, the sediment transport patterns of the Bay have been drastically altered by anthropogenic influences and natural processes. Large quantities of sediment have been added and removed from the Bay. Two major behaviors in the Bay have been observed: 1) shoaling from deposition during the hydraulic gold mining period, and 2) deepening from erosion during the latter half of the 20th century. Extensive work by the USGS has analyzed and cataloged historic trends in the Bay based on bathymetric change. The results provide unique insight into the net effects of the activities in the Bay since approximately 1850.

Bathymetric changes resulting from hydraulic mining and dredging were accompanied by broad morphological alterations to sub-embayments. Generally, increased loading during and after the Gold Rush resulted in substantial sediment accumulation in North and Central Bays. South Bay has shown a possible delayed pulse (Foxgrover et al. 2004). More recently, reduced sediment loadings have shifted the Bay into an erosional state; in addition to the cessation of the hydraulic mining sediment load, the current reduction in sediment inflow is the result of sediment captured in reservoirs, riverbank protection, and alterations in land use. The total area of tidal flats increased significantly during the Gold Rush; however, over half of these flats have eroded since the 1990s. Decreases in suspended sediment concentrations since around 1999 indicate that the available pool of erodible sediment has decreased to the point where we may no longer see continued net deposition in many areas of the Bay (Schoellhamer 2009).

3.2.4.4 Sediment Properties

Although water flows and associated sediment supplies are large forces driving the long-term trajectory of Bay margins, other factors such as sediment physical and chemical characteristics and biological activity within the margins interact with these forces and can have significant influence on sediment and contaminant fate in these areas.

Most Bay sediment is predominantly comprised of silts and clays, commonly known as Bay mud. This is particularly true for the broad shallow regions of the North and South bays. Sands are found in the deeper channels and narrow straits (e.g., Carquinez Strait), where velocities prevent deposition and accumulation of fine sediment. Sandy and silty sediment are found mixed throughout the deep higher current environments of the Central Bay, except at the Golden Gate, where gravelly sands dominate. Detailed characterization of physical properties of surface and deep Bay sediment has been carried out at some sites by the USGS (Conomos 1977; Ganju et al. 2006; Chin 2010); characteristics for sparsely sampled areas would need to be extrapolated from those of nearby sites or on the basis of similarities in physical setting (e.g., water depth, distance from main channel, etc.).

3.2.4.5 Flocculation and Settling

Particle dynamics in the water column are an important part of estuarine sediment transport. Most fine sediment in a water column exists in aggregates of individual particles known as flocs. These can include both inorganic and organic particulate material. The morphology and stability of flocs changes significantly in response to variations in water column shear, sediment load in the water column, salinity, water column chemistry, mineralogy, and other factors. Typically as riverine sediment moves downstream in a relatively high shear environment, flocs are small and often broken up. When flocs reach lower energy, higher salinity estuarine waters, this combination of physical and chemical factors promotes the formation of large dense flocs with higher settling speeds. Flocculation processes can allow for higher deposition rates of sediment in regions such as the ETM, channels, and mudflats. Increased effective sediment sizes of flocs can increase the overall settling speed and deposition of fine sediment in estuaries (Krank 1992; Winterwerp and Kesteren 2004; Lick 2009). There are few data on rates of aggregation and disaggregation of flocs specific to the Bay, but rates from the literature can be explored to evaluate their significance in quantitative model outcomes.

3.2.4.6 Bioturbation and Biostabilization

Sediment that remains relatively stable even during high energy events can still undergo active mixing due to biological activity, or *bioturbation*, by benthic macrofauna (i.e., animals) living in the surficial sediment**Error! Reference source not found.**. The most common bioturbators in marine/estuarine environments are polychaetes, crustaceans, and mollusks. These animals can have a significant effect on the sediment they inhabit, depending on their modes of feeding and other activities. Bioturbation not only affects the physical properties of the sediment (i.e., bulk density and cohesion), but can also redistribute contaminants.

Bioturbation occurs in the uppermost layers of sediment in which the animals reside, with the most intensive activity in surficial sediment (generally on the order of centimeters), and a decrease in activity with increasing depth (Clarke et al. 2001). Limited geochronology studies in the Bay have shown that virtually no mixing occurs below 33 cm (Fuller 1999). Depths and rates of bioturbation are highly variable in the Bay. The heterogeneity suggests that averaged values of biotic sediment mixing are not viable for site-scale modeling of sediment and contaminant fate.

Benthic communities in margins can have a significant and variable effect on the erosion potential of the sediment. They can increase sediment erosion by reducing its overall cohesion and can enhance deposition through bottom roughness modifications (Graf and Rosenberg 1997). Alternatively, benthic species such as algae can biostabilize the sediment and reduce erosion (de Brouwer et al. 2000). Vegetation in intertidal wetlands described previously plays a similar role, attenuating hydrologic forces and helping to retain sediment. These factors vary spatially and temporally, but can play a large role in the overall sediment transport in a wetland (Wood and Widdows 2002).

3.2.4.7 Sediment Transport Summary

Our understanding of the general patterns of sediment transport in the Bay and its margins have been developed based on extensive short-term water column measurements and evaluation of long-term morphologic change. Overall these lines of evidence agree in the description of both short-term delivery of sediment to the Bay and the longer-term cycling within the Bay. Figure 3-5 conceptually summarizes the large scale transport pathways among sub-embayments. Sediment transport in the margins is a cyclic process of sediment delivery during large events with redistribution of sediment by tides and wind waves. Water flows and sediment loads are spatially and temporally heterogeneous, dependent on climatic conditions, watershed size and characteristics, and water and land use management practices upstream. Sediment properties (particle size, critical shear stress, density, and others) are also heterogeneous and will require adequate initialization in any model. Additionally, continuing changes in the sediment loads and associated transport properties will likely need to be included in any model to describe longer-term fate scenarios. These characteristics (e.g., loadings, sediment, contaminant, and biota properties) of each embayment and associated margin are needed whether for box models or finer scale mechanistic models, differing primarily in the degree of spatial and temporal aggregation or resolution required.

Quantitative information for shorter-term (decadal or less) transport trends and at the fine spatial scale needed to characterize often heterogeneous Bay and margin areas is currently only available at select study locations. The present sampling being conducted as part of the RMP and as part of ongoing USGS sediment transport studies will continue to increase the database of information on both water column and bed sediment properties.

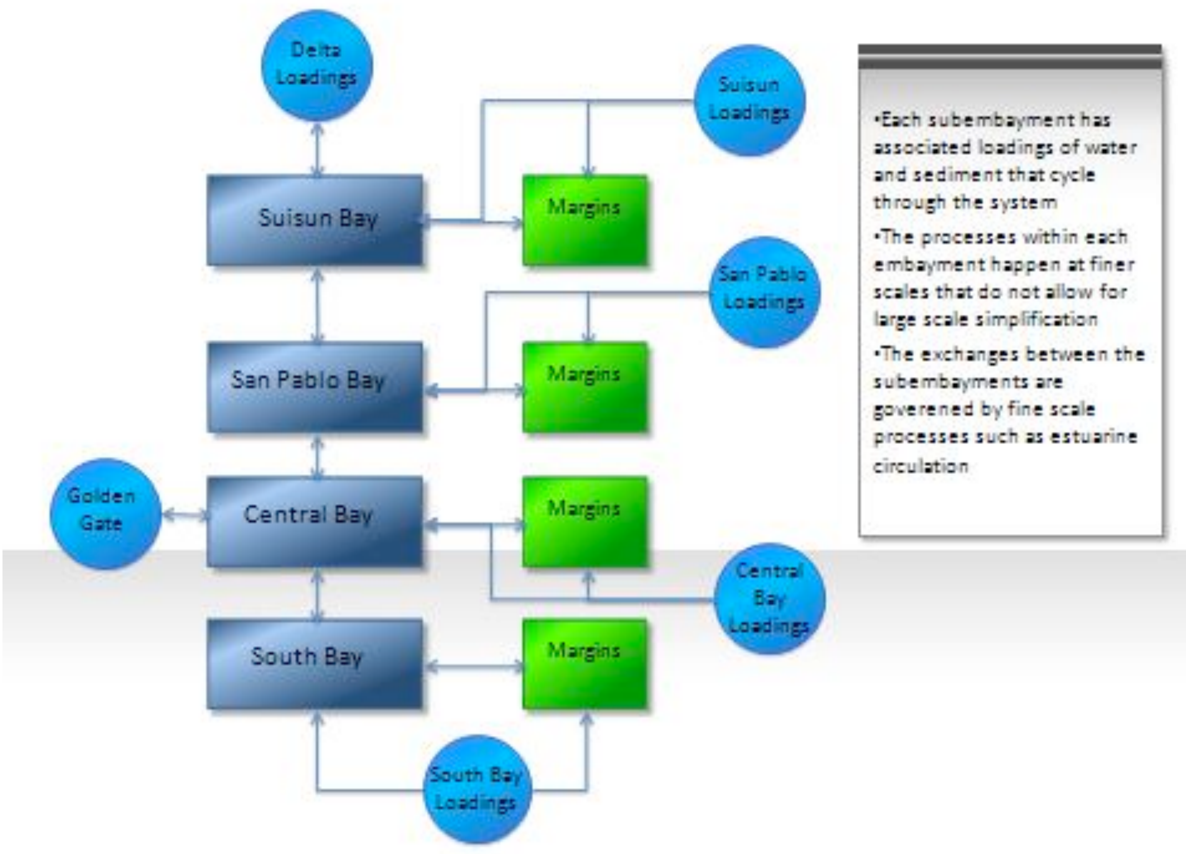

Figure 3-5. Conceptual Summary of Bay & Margin Transport

Although, quantitative hydrodynamic modeling efforts have had excellent success, the present state of sediment transport modeling in San Francisco Bay has been characterized by limited success. Full-bay modeling efforts (described in following sections) generally model trends in either suspended solids or morphology in the Bay, but have not been able to model both trends together. The inability of the present models to describe these trends calls into question the certainty of any predictive simulations. However, future mechanistic models can help to integrate the observed trends to provide a more informed picture of sediment transport as related to contaminant transport, albeit with associated uncertainties. Model sensitivity testing with existing datasets or literature values will help to identify the most critical data gaps and assess needs for future sampling.

Particularly critical parameters identified in modeling efforts can be used to direct future studies to decrease uncertainty in those parameters. By conducting sensitivity testing and validation of model behavior, the uncertainty can be better understood and constrained. Section 4 covers the overall model development process in more detail.

3.3 Biogeochemical Cycling

Abiotic and biologically-mediated chemical processes in the estuarine environment affect the transport and fate of pollutants. For conceptual and quantitative models, these processes can be broken down into two general categories, partitioning processes that transfer and distribute pollutants among phases of environmental matrices, and transforming processes that convert pollutants into other chemical species and compounds that may be more or less bioaccumulative or toxic than the original pollutant.

3.3.1 Partitioning

Distributions of chemical pollutants among matrices are functions of their physical and chemical properties, and those of the receiving matrices. Studies often simplify treatment of many complex interactions into empirically derived "partition coefficients" using (pseudo-) equilibrium assumptions. Partition coefficients often vary by several orders of magnitude between environments, dependent on site-specific characteristics such as pH, organic carbon content and quality (e.g., natural versus soot carbon), and other factors. Attempts are sometimes made to account for some of these factors by including them in equations for deriving site-specific partition coefficients.

Mass budget models developed to date for various pollutants (PCBs, PAHs, PBDEs, Hg, MeHg, Cu, and Ni) in the Estuary have generally not needed spatially explicit partition coefficients due to the large regional scales (e.g., in "one-box" models) at which the data were being integrated. Water and sediment characteristics in Bay margins may differ somewhat from those in deeper water areas of the Bay, and the finer resolution needed for transport and fate models of specific margin sites would likely benefit from finer-resolved partition coefficients to capture fine-scale variation that would be blurred by application of uniform parameters.

The appropriateness of equilibrium assumptions implicit in the use of partition coefficients depends in part on the spatial and temporal scales of interest. For modeling of annual- and longer-scale processes (much longer than most equilibration rates) in pollutant transport and fate, equilibrium assumptions are a reasonable simplification. However, for modeling of short-term sediment transport and biological systems, some with population responses (e.g., phytoplankton blooms (Luengen and Flegal 2009)) over similarly short periods, dynamics of partitioning processes may be critical, and explicit kinetic partition modeling may be needed.

Much of the RMP and other local monitoring data include some characterization of contaminant partitioning (dissolved versus particulate phase). These data can be compared to literature values to validate parameters used in modeling. Where discrepancies are found, or model sensitivity testing identifies major uncertainties, effort can be directed to resolve questions and uncertainties (e.g., deriving site specific values, incorporating co-factors such as soot carbon content (Cornelissen et al. 2005)). Limitations of existing data and assumptions can be explored through a modeling framework for comparison to other model uncertainties and used to prioritize needs for additional data collection or model refinement.

3.3.2 **Chemical Transformation**

Although chemical transformations are often categorized and considered separately from partitioning, in reality these processes are often inter-linked, as the products of chemical transformations will often have partitioning characteristics vastly different than the chemicals from which they originated. Figure 3-6 shows an example of linkages between transformation, partitioning, and transport processes for mercury cycling.

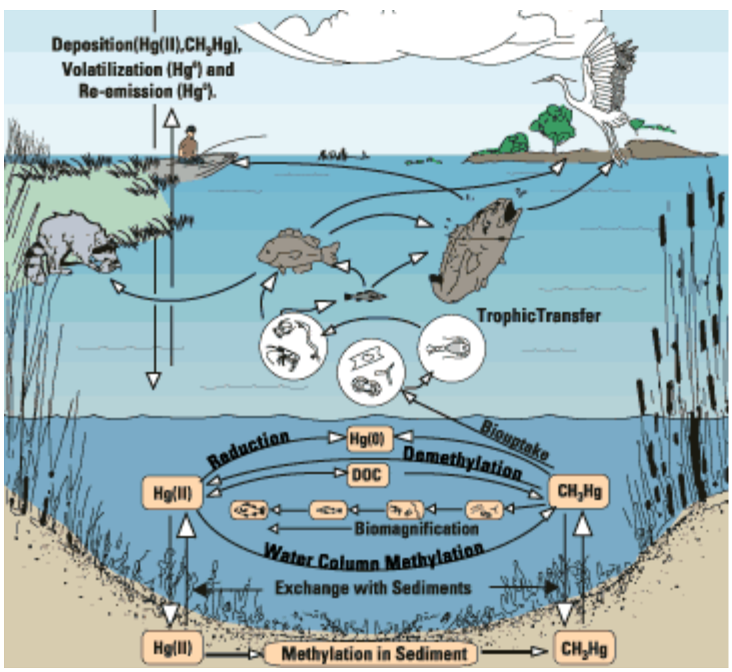

Figure 3-6. Mercury Cycle Schematic (from USGS Everglades project)

3.3.2.1 ***Speciation***

For elemental pollutants, transformations include changes in speciation, such as changes in oxidation state of ions and incorporation into inorganic or organic molecules and complexes.

Chemical speciation has impacts on partitioning, and is also important in the biological uptake and net accumulation of pollutants. Selenite, selenate, and organo-selenium species are taken up and accumulated at different rates by aquatic biota (Cutter 1989; Mason et al. 1995; Luoma and Presser 2000; Cutter and Cutter 2004). Similarly, ionic, elemental, and methylmercury also show very different accumulation rates at the base of the food web (Mason et al. 1995), and RMP data for the Bay show poor to no correlation of methylmercury to total mercury.

For example, since the majority of mercury bioaccumulated in aquatic organisms is methylmercury (Wiener et al. 2002), an understanding of transformations between these species is critical to modeling and designing management strategies for mercury in the Bay. A simple regional mass budget of methylmercury for the Bay (Yee et al. 2011) suggests that there are insufficient external sources to account for the methylmercury in Bay sediment and waters, so more finely resolved models of Bay margin processes must also include net methylmercury production as an internal source. Mercury methylation and demethylation in sediment is primarily conducted by bacteria, so methylmercury generally shows a great dependence on environmental factors (Kelly et al. 1995) influencing microbially bioavailable mercury species and microbial activity (e.g., pH, organic carbon, redox). Thus kinetically explicit models of net methylmercury production, such as that used by the Contamination Assessment and Reduction Project (CARP) for the NY/NJ Harbor Estuary (Hydroqual 2007), ultimately are tightly linked to organic carbon production and mass balance models.

Given its importance to bioavailability, speciation is needed in most Bay and margin models of trace element contaminant fate. Data on chemical speciation are generally more difficult and expensive to obtain and thus much less widely available than for partitioning, so models initially should employ the limited data that are available locally, supplemented with estimates from the literature for comparable habitats where necessary. Using literature values will generally carry large uncertainties about their applicability to local scenarios; sensitivity testing of any models developed can provide some indication of the importance of improving speciation data relative to other model uncertainties and guide study plans for reducing data gaps.

3.3.2.2 Degradation

“Degradation” is a term usually applied the transformation of organic pollutants. Although in common usage degradation connotes diminished function, for environmental pollutants, this is often not the case, with DDT and DDE for example showing comparable toxicity. Degradation rates vary widely among compounds, with half-lives for parent compounds ranging from days for simpler organic compounds such as methylmercury (Marvin-DiPasquale and Oremland 1998) and years for PAHs (Greenfield and Davis 2005), to multiple decades for some of the more persistent organic pollutants such as PCBs (Sinkkonen and Paasivirta 2000).

Degradation can occur via biological or abiotic processes. Photolysis is one pathway commonly considered, which can be used in dynamic models incorporating estimates of light penetration and water column mixing rates. Biological (primarily microbial) degradation can also be modeled, although it is often applied as a "black box" using empirically derived estimates of net macroscopic rates, rather than a detailed mechanistic modeling of the underlying processes. Microbial respiration rates and community structures are seldom measured, so more simplistic treatments may be all that is feasible, until and unless it is shown (e.g., via sensitivity testing) that more mechanistic treatment is needed.

Long-term fate models, including some previously applied to the Bay (Greenfield and Davis 2005; Yee et al. 2011), are often very sensitive to estimated degradation rates and are often highly uncertain. Similar to the case for partitioning and speciation parameters, modeling of contaminant fate can be conducted using existing local data or literature values on degradation to evaluate its importance relative to other model uncertainties.

3.3.3 Biogeochemical Cycling Summary and Information Needs

Given the long persistence of many pollutants and concerns primarily in higher trophic level organisms, (pseudo-)equilibrium partition coefficients may suffice for most pollutants in the Bay. Local data can be used to derive site-specific partition coefficients or verify appropriate ranges to use from the literature. Similarly, chemical speciation and degradation may be temporally and spatially variable, dependent on various factors such as initial speciation, transformation rates, and interacting factors such as organic carbon content, temperature, and pH. Because speciation and degradation data are quite sparse and possibly variable for many contaminants, there is considerable uncertainty associated with extrapolating limited data to modeling a large and heterogeneous system like San Francisco Bay and its margins.

In the near-term, sensitivity testing of partition coefficients, speciation distributions, and degradation rates from local and literature values applied to quantitative models including margin areas can be used to gauge the need for more dynamically interactive treatment in models or more spatially and temporally explicit characterization. The need for dynamic models of relatively short-lived pollutants such as methylmercury will depend on the specific questions being asked. The methylmercury modeling effort undertaken for CARP demonstrates such mechanistic treatment, but requires numerous biological processes (i.e., organic carbon production and fate) to be included and linked. Alternatively, time-averaged approximations (e.g., seasonal maps of average phytoplankton densities) might work in lieu of fully mechanistic models for some processes, again depending on the scales of interest for the questions asked.

3.4 Bioaccumulation

Aquatic organisms can acquire and retain chemical contaminants (e.g., PCBs and methylmercury) that they are exposed to through contact with sediment and water, dietary uptake, and transport across gills or skin. Bioaccumulation occurs when contaminant concentrations in an organism become higher than concentrations in its environment as a result of uptake from all exposure routes (Mackay and Fraser 2000). Bioaccumulation can be viewed as the net result when uptake processes are greater than loss (e.g., metabolism and excretion) processes for an organism. Biomagnification, a special case of bioaccumulation, occurs when the chemical concentration in an organism at a higher trophic level exceeds that of its diet at lower trophic levels (Mackay and Fraser 2000). Many contaminants of concern in the Bay biomagnify, including methylmercury, PCBs, dioxins, legacy pesticides, and selenium. In fact, these contaminants are of concern in large part because they biomagnify, leading to significant exposure and risk to humans and wildlife species at higher trophic levels (Davis et al. 2002; Greenfield and Jahn 2010).

3.4.1 Food Web Bioaccumulation and Trophic Transfer

Uptake by autotrophs or heterotrophic microorganisms serves as an initial route of entry of contaminants into the aquatic food web, primarily occurring through the rapid and direct exchange with sediment and water (Gobas 1993). Methylmercury uptake by phytoplankton occurs through passive diffusion into algal cytoplasm (Mason et al. 1995). Uptake of PCBs and other organics occurs primarily through the partitioning of these chemicals into the lipid of aquatic organisms (Webster et al. 1999). Some trace element contaminants (metals, selenium) may be taken up actively as essential micronutrients, but accumulate beyond dietary requirements at contaminated sites.

Uptake by microorganisms can represent the largest step-increase in concentration in the bioaccumulation process (Watras et al. 1998). For contaminants such as PCBs, methylmercury, and selenium, the increase can be many orders of magnitude over aquatic concentrations (Watras and Bloom 1992; Harding et al. 1997; Watras et al. 1998). On the Bay margins, this critical initial step in bioaccumulation may occur in areas and at times where numerous factors have caused higher concentrations due to local sources and processes.

Once incorporated into the food chain, the extent of biomagnification will depend on the contaminant, its specific properties, environmental conditions, and composition and complexity of the food web. Different contaminants are transferred through the food web via different biochemical mechanisms. Methylmercury bioaccumulates and biomagnifies due to its strong affinity for sulfhydryl groups on proteins and processes that lead to high retention (Morel 1983). Organic contaminants bioaccumulate due to fugacity (equilibrium partitioning) gradients created in the gut of predator species that favor high absorption of lipids and associated contaminants (Gobas et al. 1993). Trophic transfer is governed by dietary absorption and feeding relationships (e.g., algae – zooplankton – invertebrate – forage fish – predatory fish - mammals). The degree of contamination of each species is ultimately governed by the degree of contamination at the base of the food web, and the trophic position of the species and overall food web structure.

3.4.2 Bioaccumulation on the Margins

Concentrations of PCBs, dioxins, and other persistent legacy organic contaminants in the San Francisco Bay food web are elevated and have generally shown no clear indication of declining since RMP monitoring began in 1994 (Greenfield et al. 2005). One hypothesis that may explain this lack of declines is the presence and influence of legacy contaminated areas on the margins. Contributions of margins may be high due to patches of higher contaminant concentrations with long residence times due to reduced transport in these low energy environments. Margin contamination would contribute to human and wildlife exposure through consumption of aquatic species foraging in these areas (e.g., shiner surfperch) or of predators (larger sport fish) consuming mobile prey (i.e., small fish) foraging in these areas.

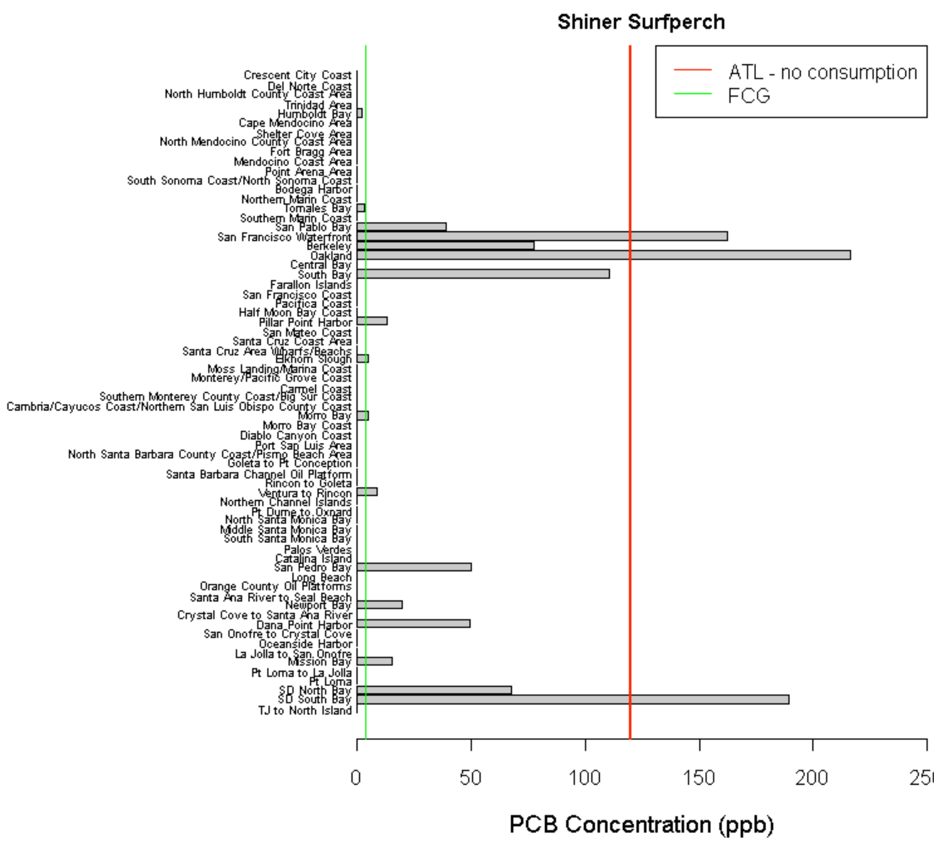

Figure 3-7 - PCB concentrations in shiner surfperch samples on the California coast, 2009-2010. From Davis et al. (2012).

High contribution of margins to food web contamination in general may explain some observed spatial patterns. Shiner surfperch forage on the margins, have high site fidelity, and thus are sensitive indicators of spatial variation (Davis et al. 2011). Average PCB concentrations in the most recent sampling (2009) were statistically unique at each of the five locations sampled (Figure 3-7), ranging from 216 at Oakland Harbor to 39 ppb in San Pablo Bay. Concentrations in San Francisco Bay were high relative to most other coastal locations. Other lower trophic level fish species have also been found to have consistently high concentrations of PCBs: in 2009 northern anchovy PCBs averaged 118 ppb, and data for small fish are discussed below.

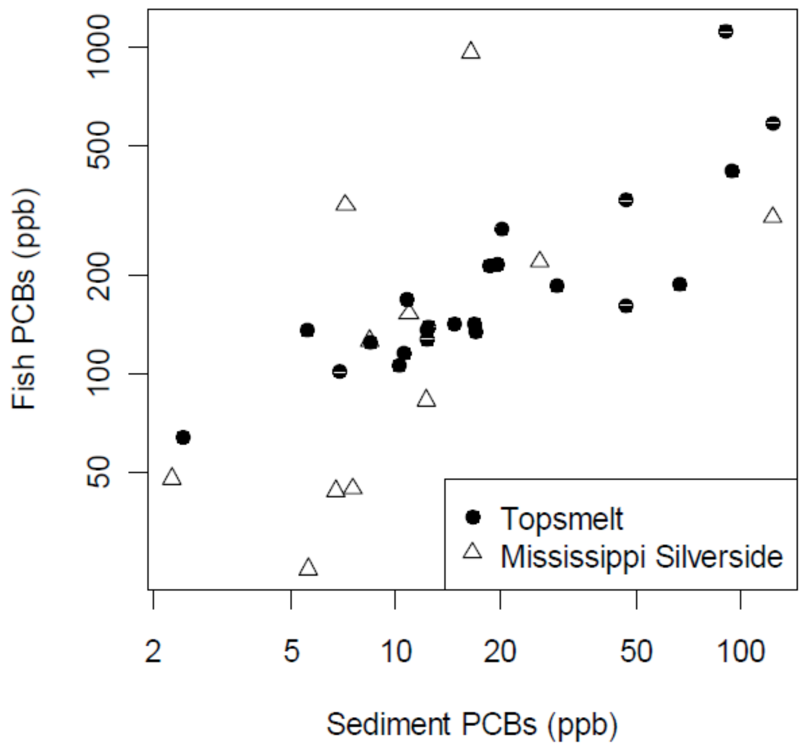

Figure 3-8 - PCB concentrations in small fish collected in San Francisco Bay in 2010 versus nearby sediment samples collected in prior years. From Greenfield et al. (2011).

If this hypothesis is correct, forage fish, benthic invertebrates, and other biota with small home ranges would exhibit concentration gradients paralleling decreases in ambient contamination away from margin sites. This pattern was recently found for PCB concentrations in small fish (Figure 3-8) sampled from margin sites in 2010, which were closely correlated (R^2=0.51) with sediment concentrations from nearby sites in a variety of previous studies. These small fish accumulated high concentrations of PCBs, frequently exceeding the highest concentrations measured in Bay sport fish. The highest PCB concentrations were observed in samples from margin sites with well-documented historic contamination (Hunter's Point, Stege Marsh, Oakland Harbor). The high PCB concentrations observed in shiner surfperch and small fish in spite of their low trophic position, along with the correlation of concentrations in fish with gradients in sediment contamination, suggest that bioaccumulation from contaminated margin sites is an important factor in the persistent exposure observed in Bay fish and wildlife.

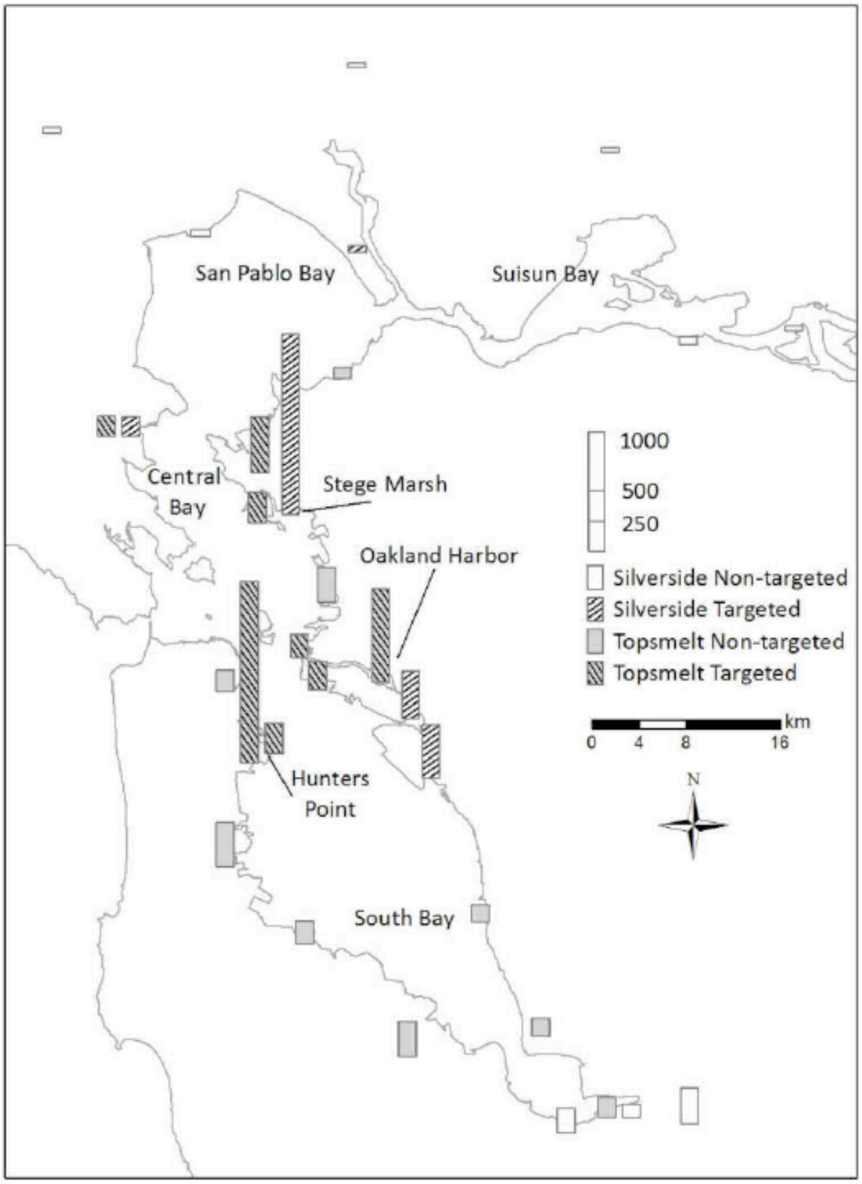

Figure 3-9 - PCB concentrations in small fish collected in San Francisco Bay in 2010. From Greenfield et al. (2011).

Greenfield et al. (2011) also examined spatial patterns in small fish mercury in relation to habitat features on the margins. Of the two main indicator species (silverside and topsmelt), only topsmelt showed an indication of higher concentrations in any habitats: subembayment (as opposed to open shoreline), and legacy contaminated sites. The strongest correlation, for both species, was with distance from the Lower South Bay (concentrations decreased with greater distance).

3.4.3 Bioaccumulation Information Needs

Quantitative mechanistic models that couple contaminant transport and fate in water and sediment with bioaccumulation in the Bay and its margins can aid in evaluating alternative management scenarios. The RMP small fish study provided a significant dataset for mercury, but modeling methylmercury biogeochemistry and bioaccumulation would be complex and challenging. Modeling uptake of PCBs and other persistent organics without in-Bay sources may be a more tractable task. The complexity of modeling nutrients may fall somewhere in between; primary concerns are impacts on primary producers and lower trophic levels and thus may require little food web modeling, but nutrients undergo numerous relatively rapid biogeochemical transformations, which can be challenging for closing mass balances.

For all of the persistent chemical pollutants, a major gap is limited understanding of the foraging behavior of key indicator species. More empirical data linking bioaccumulation to site conditions (e.g., ambient contaminant concentrations, available prey) will likely be needed to effectively model bioaccumulation whether at specific sites or across a range of sites of interest. Given the critical importance of dietary uptake for bioaccumulation of most contaminants of concern in the Bay, understanding the diets and foraging areas of indicator species may aid in understanding the linkage between contaminated water and sediment of the Bay margins and impairment in biota. Some information is available on the dietary preferences of key Bay margin indicator species such as white croaker, shiner surfperch, inland silversides, and topsmelt (Jahn 2008), but information on foraging ranges and seasonal movements is limited. These factors are large sources of uncertainty in modeling bioaccumulation that have not yet been addressed. This knowledge can also guide decisions on the appropriate scales of temporal and spatial resolution for modeling contaminant fate. Therefore, the RMP should carefully evaluate information needs related to the home ranges of key species, with an emphasis on providing quantitative information for species selected for monitoring and inclusion in models.

Where the concern is response of specific margin sites to possible management actions, modeling should focus on species known to have more restricted foraging ranges or sedentary life histories, as this reduces uncertainties about contaminant exposure locations to be considered. For impacts of contaminated margin sites on wider ranging higher trophic level organisms, these more localized species are often important prey items consumed by fish and wildlife indicator species and can help link water and sediment contamination in margin locations to biological uptake. Large gaps remain in our understanding of processes affecting contaminant uptake and bioaccumulation such as modes of transfer from sediment to aquatic plankton, temporal variation in phytoplankton and zooplankton populations, and contaminant depuration processes and rates for various indicator species of interest, both in the margins and the open Bay.

Previous modeling efforts that have addressed similar questions linking contaminant fate and biological uptake in freshwater and estuarine systems, can be adapted and expanded upon for future work on the Bay margins (e.g., Knightes et al. 2009; Sunderland et al. 2010; Yee et al. 2011). Gobas and coworkers have developed general bioaccumulation models for the Bay (Gobas and Arnot 2010) that can be adapted to site-specific applications. Where local data for species of interest are unavailable, data from the literature can be used for modeling, with uncertainties from those assumptions explored via sensitivity testing, but specific local data on contaminants in margin environments and indicator species of interest may be needed to reduce uncertainty in characterizing specific areas being simulated.

3.5 Review of Available Margin Contamination Data

A working hypothesis of the MCM is that contamination is elevated in the margins, as compared to deeper offshore portions of the Bay. Bay margins may be expected to contain depositional areas where fine particulate materials with adsorbed contaminants may settle and accumulate. Processes favoring net mercury methylation may also be more prevalent in the margins due to greater influence of tributary sources of mercury and organic carbon, and intermittent wetting and drying cycles that favor mercury deposition and methylation. For other contaminants such as legacy organochlorine compounds (e.g., PCBs and DDTs), proximity to terrestrial sources, and high productivity and utilization by biota in nearshore margins may elevate contaminant exposure relative to deeper areas of the Bay, where contaminants are more dispersed and diluted by regional-scale water and sediment sources and processes.

To evaluate the hypothesis of elevated sediment contamination in the Bay margins, several lines of evidence were explored: 1. spatial patterns of contamination; 2. temporal trends; 3. parallels in other estuaries.

Spatial graphical analyses of available local data were performed, focused on PCBs, DDTs, and mercury. These pollutants were selected for analysis because they have large datasets available for evaluation and history as priorities for management in the region, leading to relatively larger datasets. PCBs, DDTs, and Hg are environmentally persistent, largely present in particulate rather than dissolved phase, and biomagnify up the food web, similar to many other pollutants of concern. Because of these attributes, PCBs, DDTs, and Hg may be considered to be model compounds illustrative of patterns in the Bay margin contaminant processes.

Another concern for the Bay and its margins in particular is long-term trends in pollutant concentrations (Greenfield et al. 2005; Connor et al. 2007; Davis et al. 2007). Some legacy pollutant concentrations have been slowly declining, whereas others show no apparent trend, and emerging compounds may be increasing. To evaluate trends in the margins, time series for bivalves and other nearshore species were evaluated for contaminants including PCBs, mercury, selenium, copper, polybrominated diphenyl ethers (PBDEs), polycyclic aromatic hydrocarbons (PAHs), and organochlorine (OC) pesticides (Gunther et al. 1999).

3.5.1 Spatial Patterns of Ambient Contamination

Bay-wide sediment contamination data from the Sediment Quality Objectives (SQO) program and the Regional Monitoring Program (RMP) were analyzed for spatial patterns in PCBs and DDTs. The SQO program database includes data collected from regional programs (e.g., Bay Protection and Toxics Cleanup Program, RMP), site data assembled from specific Superfund and Toxic Hotspot monitoring sites (Appendix Table A-1), as well as local dredging studies (Myre et al. 2006). This dataset is an opportunistic compilation of available data, rather than a probabilistic survey design. As a result, analyses of these data include some bias due to deterministic sampling site placement (e.g., concentration statistics are biased high due to sampling sites of known or suspected sources).

Figure 3-10 and Figure 3-11 illustrate PCB data from the SQO and RMP data sets, respectively, demonstrating a general pattern of increasing PCB contamination moving southward in the Bay, and higher concentrations in margin and watershed sites compared to RMP open water data. Suisun Bay and Lower South Bays both have extensive lengths of shoreline relative to surface area of the embayment, and may therefore be expected to have a higher influence of Bay margins on PCB fate. Lower South Bay in particular has generally elevated contamination, suggesting more contaminant sources due to higher urbanization, or reduced flushing, as it lacks the large volumes of tidal and seasonal flushing that occur in the North Bay.

Sediment PCB concentrations are more variable and higher on average in shallow (near-margin) sites than deep sites (Figure 3-12). This pattern is consistent with a conceptual model of greater influence of discrete margin sources of contamination on shallow water sediment. Furthermore, shallow sites in the SQO dataset had more sites with high PCB concentrations and higher maximum concentrations than the RMP dataset, which by design excludes intertidal and subtidal locations too shallow to access with a large marine vessel and thus includes relatively fewer margin sites. Although the prevalence of highly contaminated samples in the SQO dataset in part represents the bias of deterministic sampling of or near known contaminant sources, the persistent contamination at these sites decades after expected releases suggests that other margin sites with smaller unknown sources might experience less extreme but similarly persistent contamination.

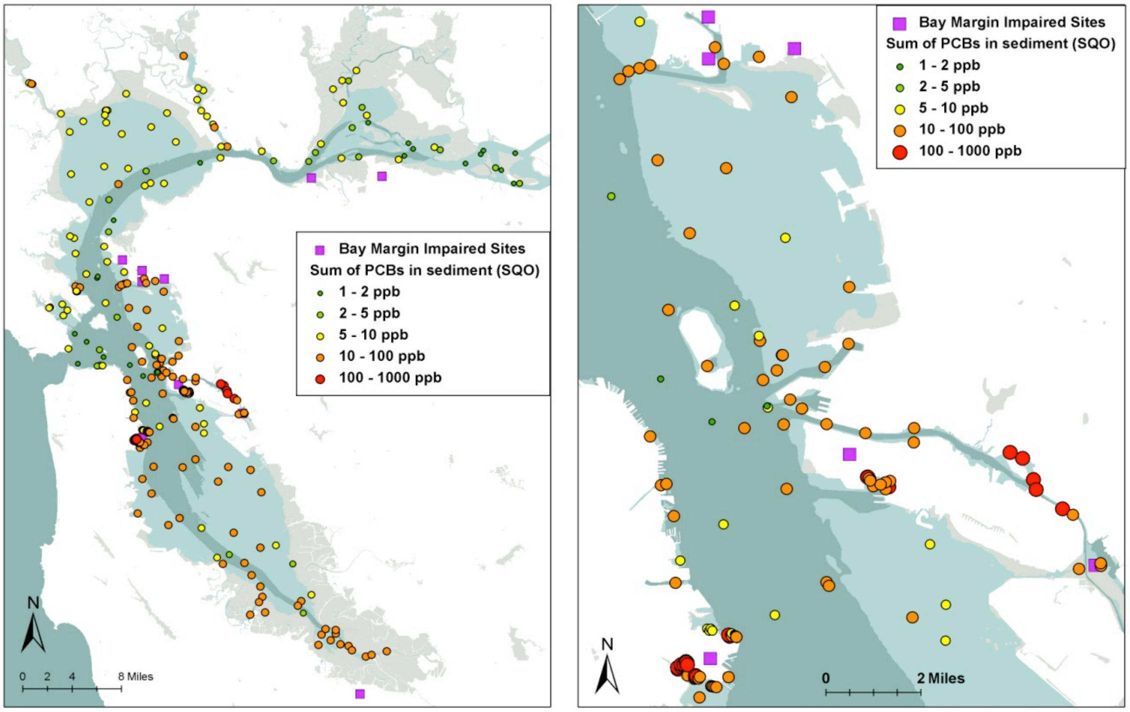

Figure 3-10 - a) Bay-wide sum of PCBs in sediment, and b) zoomed in on Central Bay. Data are from samples taken between 1990 and 2003, compiled by the Sediment Quality Objectives (SQO) program. Bay Margin Impaired Sites are the locations listed in Appendix Table A-1.

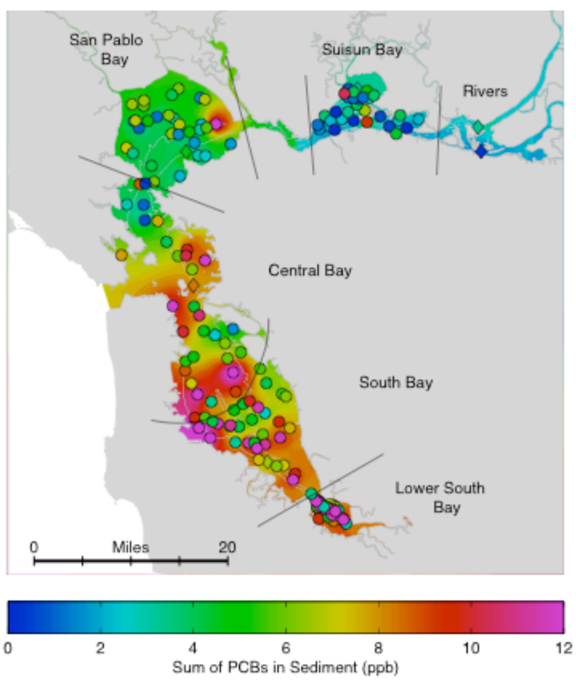

Figure 3-11 - Bay-wide sum of PCBs in sediment. Data are from samples taken between 2004 and 2008 by the Regional Monitoring Program (RMP).

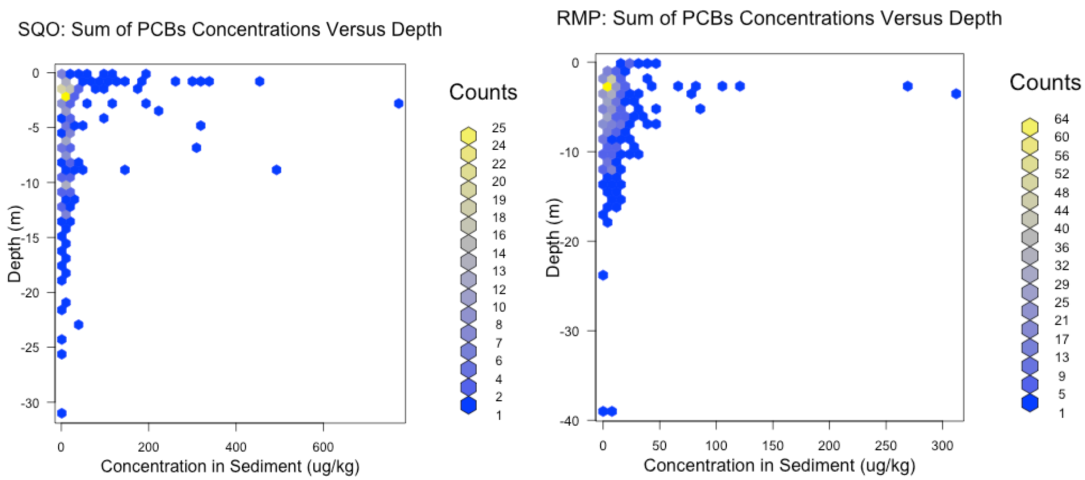

Figure 3-12 - Sum of PCB concentrations in sediment versus water depth at the sampling location. a) the Bay-wide SQO data set; b) RMP data only. Note difference in scales of x axis.

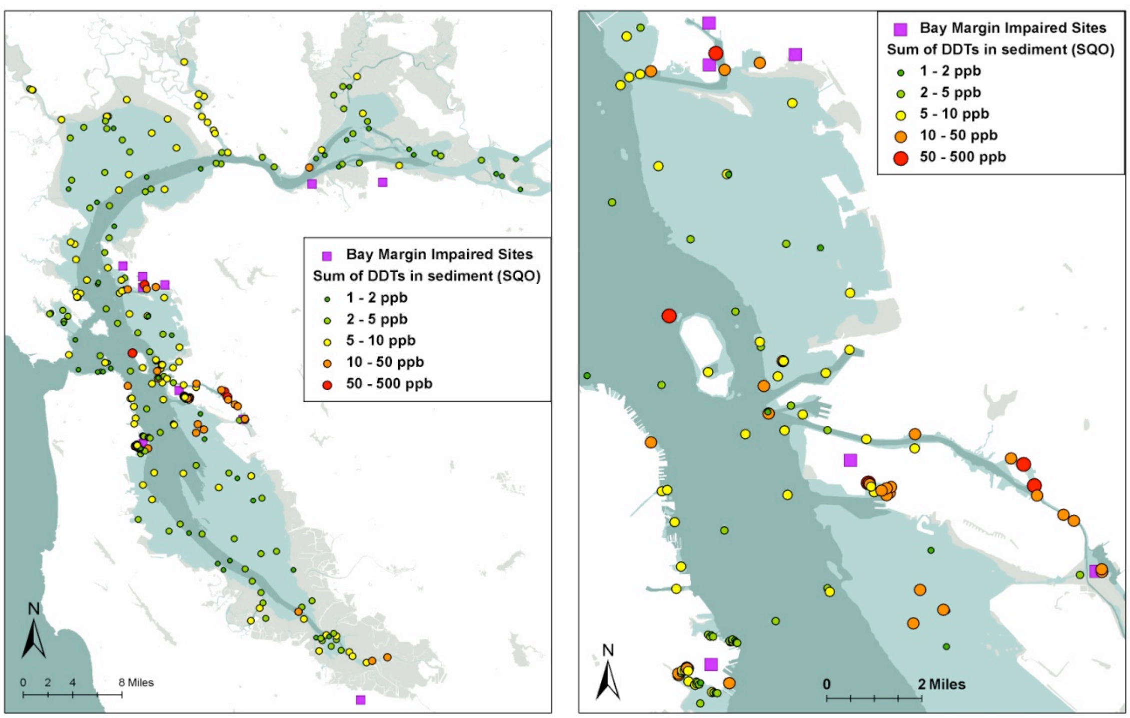

Figure 3-13 - a) Bay-wide sum of DDTs in sediment, and b) zoomed in on Central Bay. Data are from samples taken between 1990 and 2003 by the Sediment Quality Objectives (SQO) program. Bay Margin Impaired Sites are the locations listed in **Error! Reference source not found.**.

DDTs (Figure 3-13) do not exhibit a gradient increasing southward as observed for PCBs, but do exhibit similarly patchy and elevated contamination in certain Bay margins areas, particularly near points of historic industrial activity. DDT concentrations are elevated compared to offshore areas for Napa River near Mare Island, Coyote Creek, Richmond Harbor and Marina

Bay, Oakland Harbor, Alameda Naval Air Station (Seaplane Lagoon), and Hunters Point. As with PCBs, DDTs also exhibit a higher incidence of elevated sediment concentrations for shallow water sediment, most of which are in Bay margins (Figure 3-14).

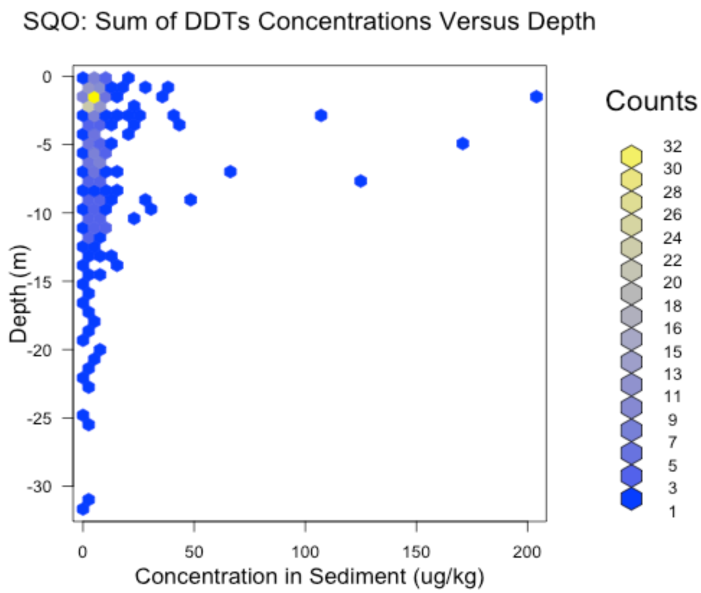

Figure 3-14 - Sum of DDTs in sediment, presented as concentrations versus sample depth for the Bay-wide SQO data set.

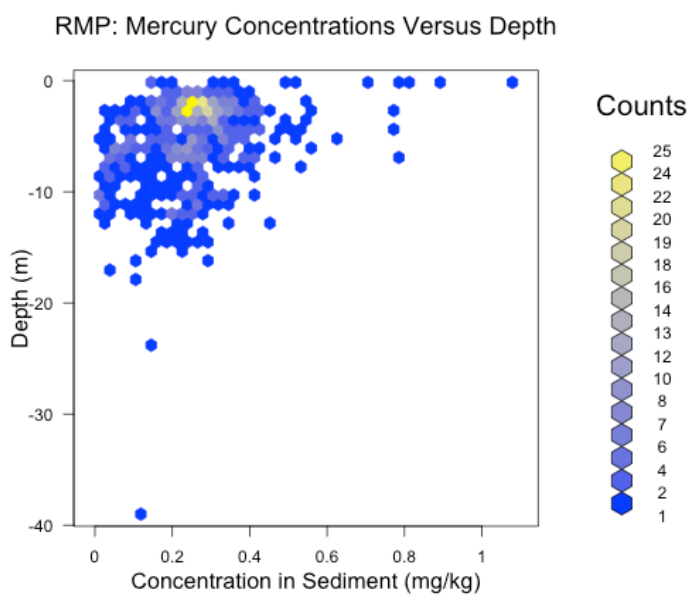

Figure 3-15. Mercury in sediment. Concentrations versus sample depth for Bay-wide RMP dataset

Mercury shows a similar but weaker pattern of higher and more variable concentrations in shallow water (near-margin) areas as seen for organic contaminants, again consistent with dominantly margin and watershed sources. The lower variability and less extreme maximum concentrations may in part be due to the natural presence of mercury (as compared to synthesized PCBs and DDTs) as well as their longer presence in the Bay (since the Gold Rush) allowing prolonged mixing and dispersion.

Available data also indicate differences in mercury processes between open water and near-shore margin sites. In a synthesis of regional mercury environmental and biota concentrations for better understanding of mercury cycling and uptake processes, sediment data analyzed for total mercury (THg), methylmercury (MeHg), % clay, and total organic carbon (TOC) as available were compiled from the RMP Status and Trends Program and other local studies (Appendix Table A-2), with five spatial parameters obtained for each site using GIS: 1. distance to the nearest shoreline; 2. percentage of land area within 2 km radius; 3. percentage of land area within 200 m; 4. water depth; and 5. latitude. Pearson correlation coefficients were examined between parameters. THg, MeHg, and TOC were positively correlated with each other (all $r > 0.5$) and also with proximity to shoreline. Of the spatial parameters examined, MeHg was most strongly correlated with percent land in a 200 m or 2 km radius (both $r \sim 0.5$), and both THg and TOC had similar but weaker positive correlations with percent land in a radius (200 m or 2 km) ($r = 0.39$). These spatial correlations with surrounding land density suggest underlying mechanisms to be considered in modeling; a stronger correlation to percent land rather than to distance to nearest shoreline suggests the importance of shoreline morphological features. A site enclosed in a small sub-embayment or creek mouth could be an equal distance to shoreline as a more open site along the Bay edge, but the latter would be more exposed to Bay tidal flows and wind wave activity, and less influenced by landward contaminant sources. Thus modeling of specific Bay margins would benefit by inclusion of site geomorphologic factors that affect contaminant processes.

3.5.2 Temporal Trends in Contamination Levels

3.5.2.1 Trends in Margin Hot Spots

Examination of temporal trends is hindered by a lack of long-term data at most Bay margin sites, although a few have been examined through studies performed as part of site assessment and remediation programs. As case studies of patterns that might be seen at known contaminated margin sites, we examined sub-decadal trends in contamination in Lauritzen Canal in Richmond (location of a pesticide formulator from 1947 to 1966) and at the South Basin area of Hunters Point Shipyard (a U.S. Navy ship repair facility until 1991).

In 1997, extensive dredging activity was conducted in an attempt to reduce DDT contamination at Lauritzen Canal. Weston et al. (2002) summarized the sediment results of four prior studies and collected additional data, spanning a period from 5 years before remediation dredging to 20 months after. The EPA performed a post-remediation study that sampled the site for six years following dredging in 1996-1997 (U. S. EPA 2004). In general, sampling results showed substantial temporal variation, with no apparent trend over time, no apparent reduction after dredging remediation, and some post-dredging concentrations even higher than those before remediation. The temporal variation may have been confounded by small scale spatial variation; in 2002 EPA sampling of the site, concentrations ranged six orders of magnitude.

Hunter's Point Shipyard also exhibited large spatial variation, with total PCBs ranging between 100 and 5,100 ug/kg for samples collected in 2001 from Area X of the South Basin Area (Battelle et al. 2005). PCB samples were collected from Area X again in December 2005, July 2006, and July 2007, with time series available at a specific subplot location within this area. At that subplot, average concentrations were 184 ug/kg in 2001 and up to 2,040 ug/kg (SD = 0.81) in 2007. The substantial variation among years may also be confounded by small-scale spatial variability at this site.

The small scale variability in sediment samples collected over time in these examples illustrates likely challenges for characterizing and modeling temporal variation in contaminant distributions in margin sites. Obtaining data to track long term concentration trends in sediment even within a small area may prove difficult, and quantitatively modeling the small scale variation found may be computationally and data intensive. However, depending on the objectives of the modeling effort, replicating temporal and spatial patterns at a fine scale may not be necessary; for estimating bioaccumulation of wider-ranging biota, simulating average concentrations rather than variability over larger spatial and temporal scales may be most important.

3.5.2.2 Trends in Bivalves at Sites Near Bay Margins

The RMP monitors contaminant accumulation in caged transplanted bivalves suspended in the water column at nine sites distributed throughout the Bay (Appendix Figure A-1**Error! Reference source not found.**). Many of these sites have been monitored since 1980 by the State Mussel Watch program and subsequently by the RMP (Gunther et al. 1999). Several more monitoring sites were added in 1993 with the inception of the RMP. Having data sets that span multiple decades affords better opportunities for detecting trends.

Trends were plotted for sum of PCBs, sum of DDTs, sum of PAHs, and sum of PBDEs at three relatively shallow sites. These sites were chosen to be representative of shallow Bay margin conditions, in <10 feet of water for the Northern (San Pablo Bay, BD20), and Southern (Coyote Creek, BA10) sites, and 10 to 20 feet for the Central Bay site (Red Rock, BC61). Of the three sites, Coyote Creek is most surrounded by features characteristic of Bay margins, such as sloughs, tributaries, wetlands, and salt ponds. To examine trends of trace organics in bivalves at these sites, linear regressions of log-transformed lipid-normalized tissue concentrations over time were plotted. Bivalve tissue concentration data normalized to lipid weight were plotted.

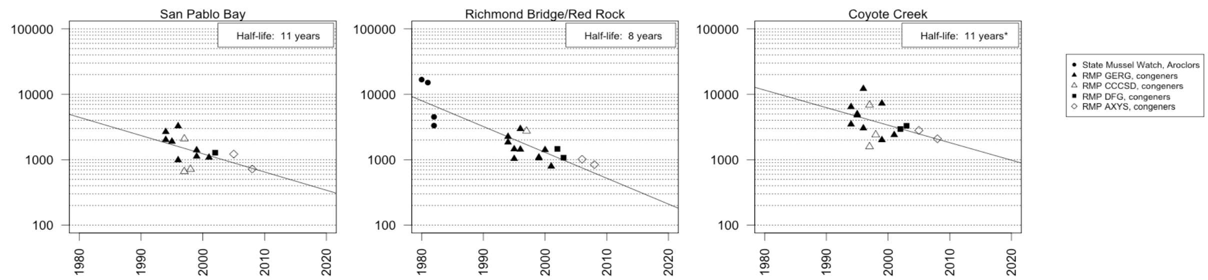

Figure 3-16 - RMP bivalve PCB trend regressions.

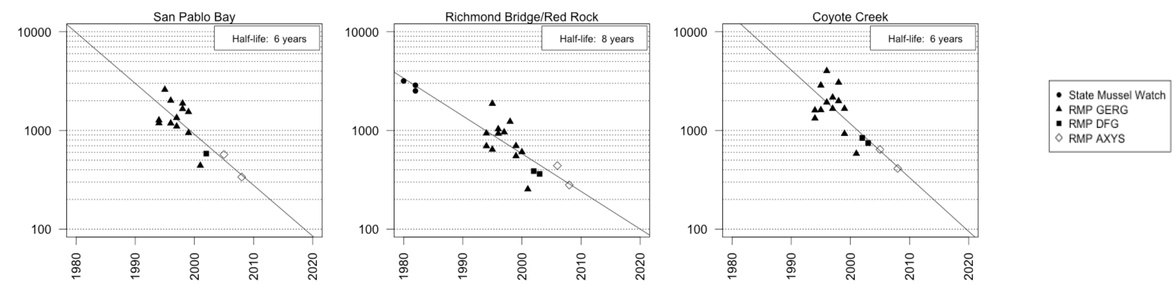

Figure 3-17 - RMP bivalve DDT trend regressions.

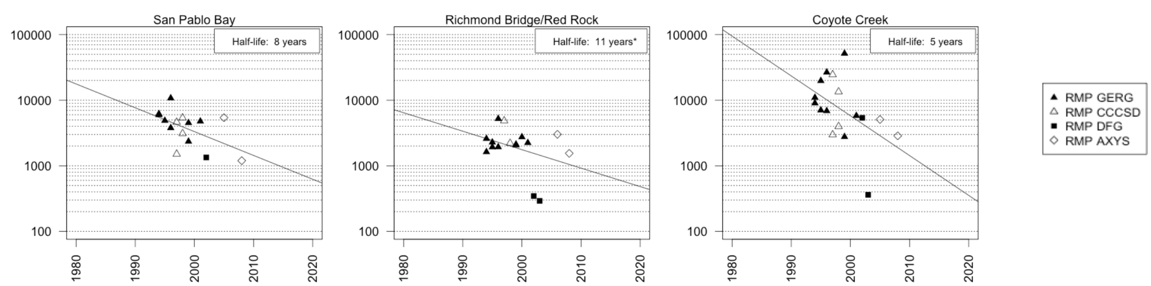

Figure 3-18 - RMP bivalve PAH trend regressions.

All sites showed a decline in bivalve PCBs (Figure 3-16), DDTs (the sum of o,p'-DDD; o,p'-DDE; o,p'-DDT; p,p'-DDD; p,p'-DDE; and p,p'-DDT) (Figure 3-17), and PAHs , with half-lives ranging from 5 to 11 years (declines not significant at a 0.05 level have half-lives marked with an asterisk). Although described here as half-lives, these rates of change describe net concentration changes which can include contaminant import and export, not just chemical degradation or conversion and other first-order decay processes more conventionally considered half-lives. For these groups of contaminants, Coyote Creek, the site most surrounded by watershed and margin sources, exhibited higher maximum organic pollutant concentrations and greater variability, compared to the other two sites nearer large channels, which likely would be more subject to waters uniformly diluted and dispersed from the open Bay. These hypotheses could be tested by more detailed monitoring of spatial and temporal variability in pollutant concentrations within other locations in the Bay margins.

The declines observed for PCBs and other legacy organics in bivalves are in contrast to the lack of trends observed in RMP monitoring of sport fish (Davis et al. 2011) and avian eggs (Grenier et al. 2012). The conceptual model for these contaminants in the Bay must account for these divergent observations. A hypothesis that could explain these patterns is that despite the proximity of some sites to margins, the suspended bivalves are more reflective of pelagic food webs in the open Bay, while sport fish and piscivorous birds may obtain a portion of the diet from benthic food webs in the margins with longer recovery curves. Decreasing trends at the three bivalve stations presented are similar to those observed at the bivalve stations in deeper water, supporting this hypothesis.

3.5.3 Margin Contaminant Data Summary and Needs

Long term monitoring by the RMP in the open water areas of the Bay, and spatially intensive but more sporadic information for specific contaminated Bay margin sites generally support our conceptual expectations of contaminant behavior in margin areas. In general, shallow near-margin areas are more contaminated and more variable than open water sites. Within contaminated margin sites such as those subject to Superfund remediation efforts, concentrations are also highly heterogeneous spatially, and apparently variable over time. Temporal trends in bivalve uptake at open Bay sites of legacy organic pollutants (PCBs, DDTs, and PAHs) indicate a general decrease over time, in contrast to the lack of trends observed in Bay sport fish and avian eggs. These divergent patterns may reflect slower recovery for some (e.g., benthic) food webs. More extensive collection of Bay margin contaminant data and analysis of existing data in a modeling framework will be needed to further evaluate these hypotheses.

The current RMP sampling design deemphasizes Bay margins due to logistical considerations. It is anticipated that more extensive empirical data will be needed to calibrate and validate quantitative models of contaminant fate the Bay margins. As mentioned previously, details of how much more information is needed will depend in large part upon the specific questions to be answered. Identifying and prioritizing the specific questions to be answered in the Bay and associated timelines for decision making are critical to deciding the appropriate type and scales of models to apply in predicting contaminant fate in the margins. Modeling efforts, as will be discussed in the next section, even when based on non-ideal data sets, can provide valuable information in developing future sampling strategies for specific management questions.

4 Modeling

Contaminant fate and transport in the Bay, and especially its margins, are dictated by complex and dynamic interactions between physical, chemical, and biological processes. Conceptual and numerical models are valuable for understanding these processes so that management questions can be effectively addressed. Models can also help fill knowledge gaps and guide the investigation of relationships and processes that are least fully understood and most influential on outcomes.

In this section, key aspects of the modeling process are presented (as described by the USEPA, 2008) and brief examples given. The examples are models used for the management of Chesapeake Bay and the New York – New Jersey Estuary. A brief summary of local modeling efforts in San Francisco Bay is given along with suggestions for areas of improvement.

4.1 Model Development

Although conceptual models are useful for describing expected ecosystem response, ultimately these responses need to be quantified in order to compare among management options. A numerical model can be as simple as a statistical regression, or as complicated as a process-based mathematical description of the physics, chemistry, and biology occurring at a specific location. Regardless of the level of complexity, all models begin with a conceptual understanding of the governing processes at a site (i.e., a conceptual model). The qualitative process description in the previous sections outlines an initial MCM that can be analytically described to form numerical models that *quantitatively* represent the Bay.

While mathematical models can improve the description of contaminant pathways and underlying processes, their primary purposes are to predict reductions in exposure and risk and to evaluate the effectiveness of management strategies, generally over extended timescales (i.e., decadal or more). They are mainly prognostic in nature (i.e., used to predict future conditions), and when developed along with a management project, models can support management decisions. They can be used to:

- direct data gathering during a management study;
- help understand post-remedy monitoring data;
- perform hypothesis testing to refine the conceptual model;
- support evaluations and selections of proposed actions; and
- support management action design.

The complexity of environmental systems makes it impossible to establish a universal algorithm for the development of numerical models. Information needs and system characteristics must be investigated on a case-by-case basis to determine the relevant scope for the modeling study. A typical numerical model for contaminant fate and transport has the four sub-models previously shown in Figure 3-3. The development, calibration, and validation of each of these sub-models is required before the model can be used as a whole, but individual sub-models can independently provide important information on critical data gaps and submodel responses (e.g., averaging out variability). The USEPA identified the elements of an modeling effort that can be used to help ensure successful modeling efforts that are briefly outlined here (USEPA 2009).

4.1.1 Data and Models

Data are a fundamental element for the development of numerical models. The previous sections outlined the breadth of data available in the Bay for model development and described some general Bay and margin characteristics that models would need to simulate or account for. Ideally, those data can be used to quantify critical processes: contaminant loads, water flows, sediment budgets, contaminant partitioning, and food web bioaccumulation. Empirical data serve as a foundational component to the formulation of empirical relationships, and the calibration and validation of developed models (USEPA 2008).

Table 4-1 - Parameters that are commonly targeted by data collection efforts.

Types of Data Often Required for Modeling Studies				
Sediment Data	**Hydrodynamic Data**	**Data on Solids, Erosion and Deposition**	**Contaminant Data**	**Biological Data**
Contaminant concentrations	Bathymetry and shoreline geometry	Water column samples of suspended solids, sampled over a range of flows	Partitioning coefficients for chemicals of concern (typically from literature or handbooks)	Identification of endpoint species, diet, and predator/prey relationships
Organic carbon content	Upstream flows	Bed load flux rates and physical properties, if bed load is present	Dissolved, colloidal, and adsorbed concentrations of contaminants in the water column, sampled over a range of flows and temperatures	Tissue contaminant concentrations, along with data on model inputs including age, size, and lipid, moisture, and solid contents of organisms
Acid volatile sulfide (AVS, when metals are present at concentrations that may pose risks)	Watershed drainage areas for important ungaged tributaries	Flume studies of erodibility at high velocities and shear stresses		Temperature and dissolved oxygen concentrations
Dry bulk density	Water levels at any downstream boundaries (*e.g.* river, lake, or tidal boundary)	Long-term measures of erosion/deposition		
Grain size distribution	Stream velocities			
	Water surface elevations			

Sampling plans in the Bay can be continually improved, with the aim of collecting the data most needed to support management discussions. Common parameters that are often targeted during data collection phases are tabulated in Table 4-1. Other data collection considerations are discussed in a USEPA document: Understanding the Use of Models in Predicting the Effectiveness of Proposed Remedial Actions at Superfund Sediment Sites (USEPA 2009).

Once a numerical model has been developed using the best available data, its performance is evaluated and improved through a process of verification, calibration, and validation using the available data for a range of likely scenarios (USEPA 2005).

4.1.2 Evaluating Uncertainty

Model predictions always come with a degree of uncertainty due to a number of reasons (USEPA 2008). Equations used and collected data will not exactly fit all the true physical, chemical, or biological processes being modeled, creating what is known as *model uncertainty*. Models at best predict average expectations, but natural variability is always present. Model uncertainty can be reduced by going through a process of model development, hypothesis testing, and refinement. With this in mind, uncertainties will always be present and should be quantified and presented with model results. There are two distinct purposes to quantifying uncertainty: 1) to determine which model fits available data the best if numerous models exist; and 2) to estimate uncertainty bounds on predictions. Unless uncertainties are presented with model results, management decisions are at risk of improper justifications.

4.2 Examples of Models Used for Management Purposes

Numerical modeling has been applied in the development of environmental management strategies for major water bodies throughout the nation. The current subsection focuses on models used in large estuarine systems: Chesapeake Bay and the New York – New Jersey Estuary, describing the modeling efforts used in their environmental management.

4.2.1 Chesapeake Bay

Chesapeake Bay is the largest and most biologically diverse estuary in the United States (USEPA 2010). With a drainage basin including the District of Columbia, and parts of six states (New York, Pennsylvania, Delaware, Maryland, Virginia, and West Virginia), it is fed by over 150 rivers and streams and empties into the Atlantic Ocean. Similar to San Francisco Bay, much of the system is relatively shallow, with an average water depth of 21 ft (USEPA 2010).

Nutrient pollution is perhaps the largest contaminant concern in Chesapeake Bay. Excessive nitrogen and phosphorous have caused eutrophication, creating anoxic conditions that are estimated to kill 75,000 tons of bottom-dwelling clams and worms each year. The excess nutrients entering the Bay are introduced by three primary sources: 1) wastewater treatment plants; 2) runoff from farmland and urban areas; and 3) air pollution from industries, vehicles, and other combustion sources (Nagle et al. 1997). Chemical contaminants including DDT, mercury, organophosphate pesticides, PCBs, and PAHs have also been widely detected at low levels throughout the Chesapeake, with three regions near major urban centers identified as having significant problems: the Patapsco River (Baltimore, MD); the Anacostia River, (Washington, D.C.); and the Elizabeth River (Hampton/Norfolk,VA)

Conservation and management efforts have fueled extensive research aimed towards understanding and forecasting hydraulic patterns, contaminant transport, and ecosystem health in Chesapeake Bay with a linked group of models called the Chesapeake Bay Program modeling suite (CBP 2010), which includes sub-models covering airshed emissions and deposition, watershed transport and loading, estuarine processes, scenario development, and land use trends.

The Chesapeake Bay modeling and monitoring program represents an example of a highly integrated approach to modeling and monitoring applied to a large and complex estuary. It applies a grid model framework with mechanistic treatment of both the physical processes that governing hydrology and contaminant fate and transport, and biotic processes governing effects of the main pollutants of concern (nutrients and BOD). However, it requires a great level of dedication and resource allocation, more than has been previously applied by the RMP to models.

4.2.2 New York-New Jersey Estuary System

The New York-New Jersey Estuary includes New York Harbor and tidally influenced portions of all rivers and streams that empty into the Harbor. The Harbor lies at the confluence of the Hudson River (the major fresh water source), the New York Bight (Atlantic Ocean), and Long Island Sound. The latter two are essentially marine, supplying saltwater and tidally influenced flow.

As with the Chesapeake, the NY-NJ Estuary contains numerous contaminated locations. The main toxics of concern are dioxins, cadmium, lead, mercury, PAHs, PCBs, and pesticides (Miller et al. In Press), mainly from legacy sources, but with some continuing loadings from various sources. High pathogen counts are also of concern, causing closures of swimming beaches and harvestable shellfish beds. One major pathway for these microorganisms is from combined sewer overflows, where untreated sewage is discharged into the Estuary during heavy rains.

Through a bi-state agreement between New York and New Jersey, conservation efforts were combined to form the Contamination Assessment and Reduction Project (CARP), with the goal of developing numerical models to help quantify present and future contamination in dredged material, and to determine the effects of changes in loadings (Miller et al. In Press) on the water, sediment, and biota contaminant levels. CARP undertook an ambitious numerical modeling effort, with intensive contaminant monitoring from 1998 to 2002.

CARP models are now used to predict contamination resulting from both historical sources and current loadings, and to forecast expected changes due to natural attenuation phenomena (burying of contaminants from settling sediment), mitigation strategies, and combinations of those. The CARP model includes sub-models for hydrodynamics, sediment transport, organic carbon production, contaminant fate and transport, and bioaccumulation (Miller et al. In Press).

In addition to the goals already achieved, the USEPA (Region 2) and the States of NY and NJ intend to apply the CARP models to analyze the effects of total maximum daily loads (TMDL) on water quality (Miller et al. In Press). The results will be used to set regulations in the TMDL, ensuring contaminant concentrations within the Estuary will not exceed acceptable levels. Management efforts will also employ CARP models to direct future Superfund and restoration projects, including stormwater management.

4.2.3 Summary

The two systems described here provide examples of multi-dimensional mechanistic models applied to better understand contaminant fate and transport in estuarine systems with similar margin features. Additionally, risk models are applied in these systems, in particular the CARP model, to quantify the outcomes of management decisions. The quantitative models applied to each of these systems differ, yet the underlying effectiveness of more detailed mechanistic modeling is apparent in the development of technically defensible quantitative goals. As no such system of models exists for addressing the priority management questions in San Francisco Bay, the examples in these systems illustrate possibility and the utility of a similar approach in the Bay.

4.3 Past San Francisco Bay Modeling Efforts

Numerous efforts to model San Francisco Bay have been and are being performed by academic, governmental, and private groups. By no means is the summary in this section meant to be all-inclusive; however a summary of some key modeling efforts provides a useful baseline when evaluating future model development. Below, a few selected local modeling studies are summarized in chronological order.

A Numerical Model of Sediment Transport Applied to San Francisco Bay, California (1997): A two-dimensional sediment transport model was developed and used to simulate suspended sediment concentrations measured during field work in northern San Francisco Bay (McDonald and Cheng 1997). The model solves modified depth-averaged shallow water equations that have additional terms for erosion and deposition. Field measurements were accurately reproduced by the model when vertical concentration gradients were small. However, only one size class of sediment was incorporated into the model. The model does not perform well when there is significant spatial variability in sediment properties.

Three-Dimensional Modeling of the Seasonal Transition of Salinity in San Francisco Bay: From Well Mixed to Stratified Conditions (2001): A three-dimensional San Francisco Bay Area numerical model (BAM3D) was developed and used to simulate hydrodynamic and salinity transport patterns (Canizares et al. 2001). The model was based off of DELFT3D, a general 3D modeling system. Accurate predictions of salinity changes in South Bay were made both on a daily and weekly time scale.

Simulating Periodic Stratification in the San Francisco Estuary (2005): TRIM3D (hydrodynamic model) was used to simulate three-dimensional circulation patterns in San Francisco Bay (Gross et al. 2005). The model was calibrated to produce observed tidal elevations and tidal currents. After calibration, it accurately predicted currents and simulated variability in salinity at both the seasonal and tidal time scales. Model results were consistent with the current understanding of stratification dynamics in San Francisco Bay.

A Model of Long-Term PCB Fate in San Francisco Bay (2008): The long-term fate of PCBs in San Francisco Bay was investigated by the RMP using a multi-box hydrodynamic and sediment transport model (Oram 2008) based on a version of the *Tidally Averaged Sediment Transport Model* below. Sediment and PCB fluxes were calculated by solving standard advection-diffusion transport equations, with special attention given to PCBs, where terms were added for volatilization and degradation rates. The model reasonably simulated observed patterns of PCB impairment.

Tidally Averaged Sediment Transport Model for San Francisco Bay, California (2009): A sediment-transport model was incorporated into a tidally averaged salinity box model (Lionberger and Schoellhamer 2009) to calculate budgets of sediment and associated contaminants. The model results were calibrated with field data to best track long-term sedimentation trends. However, spring-neap tidal suspended-sediment concentrations were consistently underestimated. The model simulated net sedimentation in the four basins that comprise San Francisco Bay, but incorrectly eroded shallows.

Decadal-Timescale Estuarine Geomorphic Change Under Future Scenarios of Climate and Sediment Supply (2010): A hydrodynamic/sediment transport model (ROMS coupled with CSTMS) was used to estimate geomorphic changes in Suisun Bay under one current and three future scenarios, including sea-level rise, and freshwater flow changes and/or decreased watershed sediment supply (Ganju and Schoellhamer 2010). In all future scenarios, net deposition in the entire estuary was slower than sea-level rise, suggesting that intertidal and wetland areas may struggle to maintain elevation.

Three-Dimensional Hydrodynamic Modeling of Sediment Transport in San Francisco Bay Using SUNTANS (2010): A three dimensional sediment and contaminant transport model for San Francisco Bay was developed (Chou 2010) as a predictive and diagnostic tool for ecological restoration, climate change, contaminant transport, and management issues. It is based on the Stanford Coastal Ocean Model (SUNTANS), which predicts hydrodynamic patterns and transport phenomena. Preliminary results fit data well, but the work is still in progress.

4.3.1 Modeling Summary

A few key conclusions can be drawn from review of modeling studies in the Bay:

1. The precedent of successful hydrodynamic modeling efforts in the Bay shows that adequate model technology and data exist to develop a hydrodynamic model for Bay and margin studies.
2. Sediment transport in the Bay has not been sufficiently studied in past modeling efforts to demonstrate success. However, the previous efforts provide a framework from which a successful effort may be built.
3. Box models have not shown success in reproducing sediment transport trends at less than sub-embayment scales. The results suggest that transport in the Bay, much less its margins, is too complex to be fully captured in box models. The TMDL efforts are the only demonstrated Bay-scale contaminant transport efforts. These are based on the sediment transport box models and share their limitations, extended to contaminant transport. The finer scale patterns of margin morphology and contaminant fate and transport again suggest the need for more highly resolved models if smaller-scale spatial processes and mechanisms are to be captured.
4. No specific modeling focused on the margins has been conducted in the Bay. The margins, as presented in the MCM, are an integral part of the Bay and have been included explicitly in previous efforts such as the DELFT3D, TRIM3D, and SUNTANS models. All of these models utilize state-of-the-art wetting and drying schemes capable of simulating transport in the intertidal margins. However, it is important to note that to date transport modeling on the Bay margins has not been rigorously validated. Multiple efforts are presently moving forward for Bay modeling that also explicitly include the margins, but are not available in peer-reviewed applications at this time.

5 Review and Conclusions

The information presented in the MCM can be reviewed with respect to our ability to answer the priority management questions.

1. Linking pathways to impairment: What are the sources, pathways, loadings, and processes leading to contaminant-related impacts in the Estuary?

We cannot presently answer this question with much precision, with the default simple assumption being that impact is directly proportional to loading. The oversimplification of this assumption is evident in the non-uniform distribution of contamination and impacts within the Bay and its margins. A goal for contaminant modeling would be to allow quantitative assessment of the linkage of specific loadings (e.g., from a particular small tributary) to impacts in the Bay, either in aggregate or at specific locations.

Existing information supports some general conclusions on this question. The available data indicate ongoing local watershed and Bay sources of contaminants to Bay margins, with legacy contamination in local margin sediment posing continued risk even without new inputs. Contaminant sources in local watersheds pose a long-term risk as continued supplies threaten to keep surface sediment concentrations elevated. Risks posed are due to unacceptable levels of contamination in surface sediment and water that are currently bioavailable, or become available after erosion or other transport processes.

Currently, co-located information on ambient (water and sediment) and biota contaminant concentrations in Bay margins is sparse in spatial and temporal coverage. Even with limited existing data, quantitative model development can help identify critical data gaps and uncertainties to better focus future data collection and modeling efforts. Given the heterogeneity and complexity of the sources and processes affecting the Bay and its margins, it is critical that the scales of priority management questions be well-defined and matched to any monitoring and modeling efforts; coarse models cannot reproduce fine scale processes, and finely resolved models tuned to specific sites cannot be extrapolated regionally with much certainty (as well as being impractically computationally and data intensive).

2. Forecasting recovery: What are the projected concentrations, masses, and associated impacts of contaminants in the Estuary under various management scenarios?

Empirical observations indicating a lack of trend in key sport fish indicator species over the past 16 years (most importantly shiner surfperch), elevated concentrations in small fish collected in margin areas, and even a lack of change in response to remediation (Lauritzen Channel) suggest that in the absence of effective management actions impairment is likely to persist for decades. Whether effective management actions can be identified to accelerate recovery is a key question.

Given that locations of highest contamination are often in margin sites, recovery will depend upon actions taken and both physical and biogeochemical processes of those sites. Physical recovery of a contaminated site occurs through stable sequestration, export, or degradation of the contaminants of interest. Stable sequestration can occur through net burial to a depth of lower erosional and biotic exposure risk, or partitioning to materials (e.g., activated carbon) with low bioavailability. Export via erosion, dissolution, and dispersion may reduce contaminants in a localized margin site, but distribute the contaminant and increase exposure risk to biota in the wider Bay. A site can also experience recovery if a contaminant degrades significantly through biogeochemical processes. However, contaminants are often problems because of slow natural degradation (e.g., PCBs half-lives of many decades), requiring intervention such as removal or in-situ bioremediation. However, if contaminant loads are too high, burial, export, and degradation of existing contamination may not be enough to reduce concentrations, and recovery can only occur if the sources are controlled.

Quantitative models are useful tools for studying the interactions of these processes to forecast outcomes under different management options, incorporating also projections of future trends in natural factors (e.g., climate, sea level). The USEPA Contaminated Sediment Remediation Guidance for Hazardous Waste Sites (USEPA 2005) typically recommends a feasibility study to develop and evaluate remedial options for a site. Even where quantitative models applied may not be highly certain or accurate, they can provide some insight on identifying factors most critical to achieving desired outcomes.

3. Contribution to regional impairment: What is the contribution of contaminated Bay margin sites to regional impairment?

As mentioned above, bioaccumulation data suggest that contamination on the margins plays a major role in impairment in the Bay at a regional scale. One mechanism for this may be bioaccumulation by small prey fish on the margins and subsequent food web transfer to predators, rather than just export of contaminated water or sediment from the margins. The contributions to regional impacts of these pathways have presently not been quantified.

Quantitative models forecasting contaminant transport and fate at margin sites will also inform managers on their impacts on the regional Bay ecosystem, as accurate site models need to address interactions with the Bay, at the least as exchanges across site boundaries. Margin 'hot spots' resulting from deposition of historical sources may represent lower regional risk via contaminant export, as historically little of the deposited contamination was transported away from the hotspots. However, those conditions could change through anthropogenic (e.g., dredging, shoreline modifications) or natural (altered climate, sediment supply) disturbances, which need to be incorporated in forecast models. Contaminated sites with ongoing inputs generally present a larger risk, as the localized contamination represents only a temporary storage of a pollutant before its wider dispersion to the rest of the Bay region. As mentioned above, contaminants can be transmitted from margin food webs to wider-ranging Bay species. This can be quantified through linkages in food web models. Scenarios relating margin contamination to regional impairment can be explored through quantitative modeling to choose appropriate management strategies and model their impacts on the Bay ecosystem.

6 Recommendations

The evaluation of priority management questions show that the interplay of many complex processes governs risk and recovery. The processes are region-specific and variable both spatially and temporally. As previously discussed, models are valuable in synthesizing understanding of fate processes in complex environments such as the Bay. Fate models can organize existing information to help fill knowledge gaps in the understanding of the processes critical to priority management questions. For many reasons, it is desirable to develop an adequate model to describe contaminant fate and transport in the Bay and on its Bay margins. Although no comprehensive model of processes in the margins has been developed to date, the information summarized in this MCM shows that previous modeling frameworks and datasets exist to develop such a model, and associated sub-models, to a standard that will be useful in addressing the priority management questions.

A modeling strategy for addressing management questions can be investigated with general recommendations made by the EPA (USEPA 2009). The first three points of the EPA modeling strategy, in bold below, can be addressed with information from the MCM.

Consider site complexity before deciding whether and how to apply a mathematical model.

Contaminant fate and transport in the Bay is dictated by a complex interplay of processes that can be difficult to quantify. However, a broad database of scientific studies on many of the basic processes is available and can be used to develop a numerical mechanistic model to aid in their quantification. The complexity of the Bay presents a modeling challenge, yet also indicates the need for numerical models to answer priority management questions. Models described previously for other large estuaries provide evidence that large systems can be modeled numerically to aid conservation and restoration efforts.

The Bay margins environment, as shown in the MCM, is affected by cycling of sediment and associated contaminants between the margins and Bay throughout the year. The nature of these processes and linkage with whole Bay transport argues for the use of integrated Bay and margin model as opposed to individual site models that likely cannot adequately capture the interactive margin connection with the larger Bay.

Develop and refine a conceptual model that identifies the key areas of uncertainty where modeling information is needed.

The conceptual model presented here outlines the processes and provides data sources on those processes where available. The MCM provides a baseline foundation for the development of higher-order quantitative models. Areas of uncertainty can be further characterized as any modeling efforts progress.

***Determine what model output data are needed to facilitate decision-making*.**

Although in the rest of this MCM, discussion of environmental processes has been presented in a mechanistically logical sequence, from physical transport, to contaminant fate, to biological exposure, to maximize the utility of modeling, it may be best to start from desired outcomes.

1) What site-specific information is needed to make the most appropriate management plan?

Ultimately the impairment listings for the Bay are based on biota exposure to contaminants. A successful suite of models must therefore adequately simulate the current condition of impairment, and any recent historical trends seen, before we can have confidence in their ability to predict outcomes under various future management scenarios. Although it is tempting to wish for an "all in one" integrated tool that can accurately model across (from site to regional) scales, this is unlikely; in analogy, it is impractical or impossible to both build a good house and carve a figurine using only a chainsaw or a carving knife. Although models addressing different scales can be nested, their linkages will not be fully interactive, and analysis of their results needs to recognize such limitations. It is therefore important that the desired output (i.e. what biological endpoint, when and where) be well defined and constrained to determine the spatial and temporal scale of most interest for modeling. The required output, appropriate scenarios, modeling approaches, and required input data for the various sub-models will then derive from that. Modeling tools and supporting data may be shared among some contaminants and questions, but others will cover very different sources and environmental processes, and thus require different resolutions for data and models.

When evaluating scale, developing a model that can be analyzed for multiple scenarios in a time frame that allows for the results to be properly evaluated is important. Sensitivity analyses of various grid and temporal resolutions often allow modelers to converge on a balance between model accuracy and computational time. Some sacrifice of resolution and physical, chemical, and biological complexity is also often made in order to achieve computational times and modeling costs acceptable to managers. A complete modeling study will investigate the implications of any such decisions.

2) What model(s) are capable of generating this information?

Again, here it may be best to work backward from the desired outcome, i.e., biological exposure; at what temporal scales are differences or trends found that managers wish to act upon or distinguish outcomes? Answers will depend on the particular contaminants of interest, and although to an extent we can share tools (e.g., model code, training data sets) across questions to minimize duplicating effort, we need to recognize the limitations and artifacts of doing so. For example, a model optimized to simulate long term sedimentation data may have lower water column SSC than actually seen, and thus overpredict phytoplankton productivity or photodegradation. Although adaptive management goals may change over time with improved information (e.g., if eliminating Bay-wide impairment appears not possible in less than 200 years, managers may instead focus on reducing top quartile "hot spot" concentrations), goals should be considered carefully to minimize repeated redevelopment of modeling tools.

3) How can the model results be used to help make these decisions?

Recovery rates of a contaminant in a region of interest provide the primary metric for determining an appropriate management strategy. A model can be used to investigate a number of different recovery, remedial, and management scenarios so that direct comparison of recovery rates to activity can be obtained. The scenario simulations will allow a cost to benefit analysis to be conducted so that effective and efficient management strategies can be developed. Even where the outcomes of various management options are not distinct, model results can provide some indications of possible reasons for the lack of differences, allowing to us identify and develop management approaches that may be more effective.

6.1 Priority Items for a Margins Model to Address

The key physical, chemical, and biologic processes that need to be modeled are described in this document. Process identification will assist in selecting a model which produces an adequate and defensible representation of the Bay system. Models not producing a complete representation of these processes can only be relied upon for preliminary investigations and not the development of management strategies.

The critical processes to model can be split into four basic classes; hydrodynamics, sediment transport, chemical transport and fate, and biotic processes. Conveniently, numerical models are typically developed with these classes, allowing for easier evaluation of modeling options.

Table 6-1 identifies the key parameters needed for each class of model and categorizes (low/medium/high) the current state of data availability and confidence level for use in the modeling of Bay margins, and brief reasoning for their current categorization.

6.2 Recommendations

A mechanistic suite of models (physical, chemical, and biological) for the Bay and its margins could incorporate all of the complex processes described in the MCM. Such tools would be useful to answer Bay management questions that require integrated understanding of linkages among these processes.

Besides the primary ability to simulate the processes described herein, the general characteristics important in model selection also include the model applicability, availability, validity, and available expertise for model application and support. The model would ideally be transportable among local user groups with a reasonable level of baseline expertise. The model must also be flexible, as it is impossible to anticipate a priori all possible model applications.

Ideally a model should allow application and testing at a fairly coarse scale initially, with simple assumptions (e.g., spatial interpolation, averaging, and correlation), and then use sensitivity analysis to identify the important data gaps. The flexibility of models is not unlimited, so there is likely not a single set of models that is optimal for addressing all the management questions of interest due to the range of sites and management actions to be considered. The relative importance of various management questions to be answered must be prioritized.

Although the spatial and temporal scale and resolution of transport modeling needed is likely to vary among contaminants to be considered, for practical logistical reasons, the use of a single (physical transport) modeling platform will likely be needed. With a sufficiently flexible transport model, development can proceed in parallel to work on bioaccumulation and contaminant fate models. Toward that end, the following are recommended:

1) A model such as DELFT3D or EFDC would be useful for the RMP. Either would allow for mechanistic modeling of physical transport in margin sites and the Bay as a whole. The following are some of the key attributes of these models.

- They are realistic enough to capture critical processes.
- It would be feasible with either to make significant progress in the next few years to inform next iterations of the mercury and PCBs TMDLs (EFDC has been used in multiple estuaries in the United States to develop contaminant TMDLs), and/or development of numeric nutrient endpoints.
- Both are established models with existing user groups (Delft3D is currently used by USGS in Bay sand provenance studies and EFDC is widely used by the EPA in TMDL studies nationwide).
- Either can be developed economically (Delft3D still has license fees on water quality modules).
- Both would provide a technically defensible foundation for science and management.
- Both have sufficient flexibility in spatial resolution to provide realism, but also allow reasonable run times.

For such a modeling approach to achieve success in the next several years, dedicated resources must be set aside to support model development and validation.

Table 6-1 - Data Availability and Confidence for Some Key Model Factors

Parameter	Data availability	Confidence	Comments
Hydrodynamic Model			
Tidal Circulation	High	High	Large volume of data and modeling on circulation in the Bay
Salinity Gradients	High	High	Large volume of data and modeling salinity gradients in the Bay
Wetting and Drying	Medium	Medium	Some site specific studies examining wetting and drying processes
Marsh, Mudflat, Channel Exchange	Low	Medium	Only studied for limited cases, but the physical processes can be modeled
Wind-Generated Waves	Medium	Medium	Studies have been conducted, and current modeling efforts are being conducted to link waves and circulation in the Bay
Storm Events	Medium	Medium	Data on storm effects on circulation in the bay available for recent decades
Sediment Transport Model			
Sediment Loads	Medium	Low - Medium (variable)	Loads available for major tributaries; but smaller tributaries uncertain for variable storm events. Future scenarios are only partially understood.
Sediment Bed Properties	Low	Medium	Few bay wide studies conducted; but general distribution sedimentunderstood.
Sediment Transport Properties	Medium	Medium	Water column measurements available throughout the Bay; greatest uncertainties in modeling of sediment flocculation parameters.
Morphology	Medium	Medium	Good historic bathymetric change data and adequate model reproduction of past trends, but the predictive capability of these models is uncertain.
Sea Level Rise	Medium	Medium	IPCC sea level rise estimates used in predicting Bay shoreline effects.
Contaminant Transport			
Contaminant Loads	Medium	Low	More data for some contaminants, highly variable temporally and spatially
Contaminant Partitioning	Medium	Medium	Some local data, can be confirmed with literature values
Biogeochemical Processes	Low	Low	Few local data, must use literature values for many rates
Empirical Contaminant Data	Low	Medium	Limited margin(mostly Superfund) sites that were fairly intensely sampled
Bioaccumulation Model			
Initial Uptake	Low	Low	Little on local margins, variable literature values, unquantified cofactors
Food Web Structure	Low	Low	Few local data, some available for similar species or other regions
Assimilation Efficiency	Low	Low	Almost no local data, need to use literature values
Metabolism and Excretion Rates	Low	Medium	Almost no local data, need to use literature values
Organism Life History	Medium	Medium	Few local data, but some for other regions/sites
Non-dietary Exposure	Low	Medium	Few local data, some mechanistic models, likely small input and variation
Empirical Bioaccumulation Data	Medium	Medium	Most abundant type of local bioaccumulation data

2) Coordination of Bay modeling and sediment transport work is needed. Multiple organizations are involved in actively developing models of water, sediment, and contaminant fate in the Bay. There is a great need for coordination of these efforts to maximize the utility of the work that is being done and to avoid wasting limited resources.

3) A repository is needed for data from local studies. Existing data related to contaminant fate on the margins are largely inaccessible. Steps should be taken to ensure that data from future studies can be accessed and incorporated into models and management decisions.

4) Additional empirical data will be needed to support model development. These needs will become clearly defined as the modeling plan is further developed. It is likely that site-specific information will be needed to model margin sites selected as priority areas for initial model development and calibration.

Although these steps represent significant commitments of effort and resources, they will be necessary to advance beyond our highly aggregated and simplified current understanding of contaminant processes in the Estuary and design more effective strategies for managing their impacts on the ecosystem.

7 References

Abu-Saba, K. and S. Ogle (2005). Selenium in San Francisco Bay Conceptual Model/Impairment Assessment, Clean Estuary Partnership.

Atwater, B. F., Conrad, S.G., Dowden, J.N., Hedel, C.W., MacDonald, R.L., Savage, W. (1979). "History, landforms, and vegetation of the estuary's tidal salt marshes." San Francisco Bay: the unbanized estuary. San Francisco (CA): Pacific Division, AAAS: 347-385.

Barnard, P. a. H., D. (2007). "Sand waves in San Francisco Bay, California." Journal of Coastal Research **23**(3).

Battelle, Blasland Bouck & Lee Inc. and Neptune & Company (2005). Final Hunters Point Shipyard Parcel F Validation Study Report. San Francisco Bay, California. San Diego, CA, U.S. Navy**:** 298 + App.

Canizares, R., E. Smith and S. Alfageme (2001). Three-Dimensional Modeling of the Seasonal Transition of Salinity in San Francisco Bay: From Well Mixed to Stratified Conditions. Estuarine and Coastal Modeling.

CBP. (2010, 11/02/2010). "Chesapeake Bay Modeling." from http://www.chesapeakebay.net/modeling.aspx.

Cheng, R. T., Casulli, V., and Gartner, J.W. (1993). "Tidal residual intertidal mudflat (TRIM) model and its applications to San Francisco Bay, California." Estuarine, Coastal, and Shelf Science **36**: 235-280.

Chin, J., Woodrow, D., Mcgann, M., Florence, W., Fregoso, T., and Jaffe, B. (2010). "Estuarine Sedimentation, Sediment Character, and Foraminiferal Distribution in Central San Francisco Bay, California." USGS Open File 2010-1130.

Chou, Y. J. (2010). Three-Dimensional Hydrodynamic Modeling of Sediment Transport in San Francisco Bay Using SUNTANS:. Stanford, California, Stanford University.

Clarke, D. G., M. R. Palermo and T. C. Sturgis (2001). Subaqueous Cap Design: Selection of Bioturbation Profiles, Depths, and Process Rates. Vicksburg, MS, U.S. Army Engineer Research and Development Center**:** 14.
Conaway, C. H., J. R. M. Ross, R. Looker, R. P. Mason and A. R. Flegal (2007). "Decadal mercury trends in San Francisco Estuary sediment." Environmental Research **105**: 53-66.
Conaway, C. H., S. Squire, R. P. Mason and A. R. Flegal (2003). "Mercury speciation in the San Francisco Bay estuary." Marine Chemistry **80**(2-3): 199-225.
Connor, M., D. Yee, J. A. Davis and C. Werme (2004). Dioxins in San Francisco Bay: Conceptual Model/Impairment Assessment. Oakland, San Francisco Estuary Institute**:** 60.
Connor, M. S., J. A. Davis, J. Leatherbarrow, B. K. Greenfield, A. Gunther, D. Hardin, T. Mumley, J. J. Oram and C. Werme (2007). "The slow recovery of San Francisco Bay from the legacy of organochlorine pesticides." Environmental Research **105**(1): 87-100.
Conomos, T. J. a. P., D.H. (1977). "Suspended-particulate transport and circulation in San Francisco Bay: an overview." Estuarine Processes, Academic, San Francisco, CA **2**: 82-97.
Cornelissen, G., Ö. Gustafsson, T. D. Bucheli, M. T. O. Jonker, A. A. Koelmans and P. C. M. van Noort (2005). "Extensive Sorption of Organic Compounds to Black Carbon, Coal, and Kerogen in Sediment and Soils:� Mechanisms and Consequences for Distribution, Bioaccumulation, and Biodegradation." Environmental Science & Technology **39**(18): 6881-6895.
Cutter, G. A. (1989). Estuarine Behaviour of Selenium in San Francisco Bay.
Cutter, G. A. and L. S. Cutter (2004). "Selenium Biogeochemistry in the San Francisco Bay Estuary: Changes in Water Column Behavior." Estuarine Coastal and Shelf Science. Vol. **61**(3).
Daum, T., S. Lowe, R. Toia, G. Bartow, R. Fairey, J. Anderson and J. Jones (2000). Sediment contamination in San Leandro Bay, CA. Richmond, CA, San Francisco Estuary Institute, San Francisco Bay Regional Water Quality Control Board, California Department of Fish and Game, and Port of Oakland**:** 52 pp.
David, N., L. J. McKee, F. J. Black, A. R. Flegal, C. H. Conaway, D. H. Schoellhamer and N. K. Ganju (2009). "Mercury concentrations and loads in a large river system tributary to San Francisco Bay, California, USA." Environmental Toxicology and Chemistry **In press**.
Davis, J. A., F. Hetzel, J. J. Oram and L. J. McKee (2007). "Polychlorinated biphenyls (PCBs) in San Francisco Bay." Environmental Research **105**: 67-86.
Davis, J. A., M. D. May, B. K. Greenfield, R. Fairey, C. Roberts, G. Ichikawa, M. S. Stoelting, J. S. Becker and R. S. Tjeerdema (2002). "Contaminant concentrations in sport fish from San Francisco Bay, 1997." Marine Pollution Bulletin **44**: 1117-1129.
de Brouwer, J., S. Bjelic, E. de Deckere and L. Stal (2000). "Interplay between biology and sedimentology in a mudflat." Continental Shelf Research **20**: 1159-1177.
Dyer, K. (1997). Estuaries - A physical introduction. New York, John Wiley & Sons.
Foxgrover, A. C., S. A. Higgins, M. K. Ingraca, B. E. Jaffe and R. E. Smith (2004). Deposition, erosion, and bathymetric change in South San Francisco Bay: 1858-1983. Open-File Report Reston, Virginia, US Geological Survey**:** 1-25.
Friedrichs, C. T. and J. E. Perry (2001). "Tidal saltmarsh morphodynamics: a synthesis." Journal of Coastal Research(SI 27): 7-37.
Fuller, C., van Green, A., Baskaran, M., and Anima, R. (1999). "Sediment chronology in San Francisco Bay, California, defined by Pb-210, Th-234, Cs-137, and Pu-239,240." Marine Chemistry **64**: 7-27.
Ganju, N. K. and D. H. Schoellhamer (2010). "Decadal-Timescale Estuarine Geomorphic Change Under Future Scenarios of Climate and Sediment Supply." Estuaries and Coasts **33**: 15-29.
Ganju, N. K., D. H. Schoellhamer, M. C. Murrell, J. W. Gartner and S. A. Wright (2006). Constancy of the relation between floc size and density in San Francisco Bay. In: Estuarine and Coastal Fine

Sediment Dynamics. INTERCOH 2003. J. P. Maa, Sanford L.H., and Schoellhamer, D.H. Eds. Amsterdam, Netherlands, Elsevier: 75-91.
Ganju, N. K. a. S., D.H. (2006). "Annual sediment flux estimates in a tidal strait using surrogate measurements." Estuarine, Coastal and Shelf Science **69**: 165-178.
Gobas, F., X. Zhang and R. Wells (1993). "Gastrointestinal Magnification - The Mechanism Of Biomagnification And Food-Chain Accumulation Of Organic-Chemicals." Environmental Science & Technology **27**(13): 2855-2863.
Gobas, F. A. P. C. (1993). "A model for predicting the bioaccumulation of hydrophobic organic chemicals in aquatic food-webs: application to Lake Ontario." Ecological Modelling **69**: 1-17.
Graf, G. and R. Rosenberg (1997). "Bioresuspension and biodeposition: A review." Journal of Marine Systems **11**: 269-278.
Greenfield, B. K. and J. A. Davis (2005). "A PAH fate model for San Francisco Bay." Chemosphere **60**: 515-530.
Greenfield, B. K., J. A. Davis, R. Fairey, C. Roberts, D. Crane and G. Ichikawa (2005). "Seasonal, interannual, and long-term variation in sport fish contamination, San Francisco Bay." Science of the Total Environment **336**: 25-43.
Greenfield, B. K. and A. Jahn (2010). "Mercury in San Francisco Bay forage fish." Environmental Pollution **158**: 2716-2724.
Gross, E. S., Koseff, J.R., and Monismith, S.G. (1999). "Three-dimensional salinity simulations of South San Francisco Bay." Journal of Hydraulic Engineering **125**(11): 1199-1209.
Gross, E. S., M. L. MacWilliams and W. Kimmerer (2005). Simulating Periodic Stratification in the San Francisco Estuary. Estuarine and Coastal Modeling.
Gunther, A. J., J. A. Davis, D. D. Hardin, J. Gold, D. Bell, J. R. Crick, G. M. Scelfo, J. Sericano and M. Stephenson (1999). "Long-term bioaccumulation monitoring with transplanted bivalves in the San Francisco Estuary." Marine Pollution Bulletin **38**(3): 170-181.
Harding, G. C., R. J. LeBlanc, W. P. Vass, R. F. Addison, B. T. Hargrave, S. P. Jr., A. Dupuis and P. F. Brodie (1997). "Bioaccumulation of polychlorinated biphenyls (PCBs) in the marine pelagic food web, based on a seasonal study in the southern Gulf of St. Lawrence, 1976-1977." Marine Chemistry **56**: 145-179.
Heim, W. A., K. H. Coale, M. Stephenson, K.-Y. Choe, G. A. Gill and C. Foe (2007). "Spatial and habitat-based variations in total and methyl mercury concentrations in surficial sediments in the San Francisco Bay-Delta." Environmental Science & Technology **41**(10): 3501-3507.
Hunt, J., B. Anderson, B. Phillips and K. Taberski (1999). Bay Protection and Toxic Cleanup Program: Studies to identify toxic hot spots in the San Francisco Bay Region. 1997 Annual Report: San Francisco Estuary Regional Monitoring Program for Trace Substances. Richmond, CA, San Francisco Estuary Institute: 132-139.
Hunt, J. W., B. S. Anderson, B. M. Phillips, J. Newman, R. S. Tjeerdema, K. Taberski, C. J. Wilson, M. Stephenson, H. M. Puckett, R. Fairey and J. Oakden (1998). Sediment quality and biological effects in San Francisco Bay: Bay Protection and Toxic Cleanup Program Final Technical Report. Oakland, CA, San Francisco Bay Regional Water Quality Control Board.
Hydroqual, I. (2007). Contamination Assessment and Reduction Project (CARP) A Model for the Evaluation and Management of Contaminants of Concern in Water, Sediment, and Biota in the NY/NJ Harbor Estuary: Contaminant Fate & Transport & Bioaccumulation Sub-models. New York, NY, Hudson River Foundation for Science and Environmental Research, Inc., The Port Authority of New York and New Jersey.
Jahn, A. (2008). RMP Food Web Analysis: Data report on gut contents of four fish species. Oakland, Ca, San Francisco Estuary Institute.

Johnson, B. and R. Looker (2006). Mercury in San Francisco Bay Proposed Basin Plan Amendment and Staff Report for Revised Total Maximum Daily Load (TMDL) and Proposed Mercury Water Quality Objectives, California Regional Water Quality Control Board: 87.

Kelly, C. A., J. W. M. Rudd, V. L. St.Louis and A. Heyes (1995). "Is total mercury concentration a good predictor of methyl mercury concentration in aquatic systems?" Water, Air, & Soil Pollution **80**(1): 715-724.

Knightes, C. D., E. M. Sunderland, M. Craig Barber, J. M. Johnston and R. B. Ambrose (2009). "Application of ecosystem-scale fate and bioaccumulation models to predict fish mercury response times to changes in atmospheric deposition." Environ Toxicol Chem **28**(4): 881-893.

Krank, K. a. M., T. (1992). "Characteristics of suspended particles at an 11-hour anchor station in San Francisco Bay, California." Journal of Geophysical Research **97**(C7): 11373-11382.

Krone, R. B. (1979). Sedimentation in the San Francisco Bay system San Francisco Bay: The Ecosystem. Further investigations into the natural history of San Francisco Bay and Delta with reference to the influence of man. T. J. Conomos, A. E. Leviton and M. Berson. San Francisco, CA, Pacific Division of the American Association for the Advancement of Science c/o California Academy of Sciences: 85-96.

Lacy, J., Schoellhamer, D., and Burau, J. (1996). "Suspended solids flux at a shallow water site in South San Francisco Bay, California." Proceedings of the North American Water and Environmental Congress '96, ASCE, New York, 1996.

Lacy, J. R. (2000). "Circulations and Transport in a Semi-Enclosed Estuarine Subembayment." Ph.D. Thesis, Stanford Universit.

Lee, H., II., A. Lincoff, B. L. Boese, F. A. Cole, S. F. Ferraro, J. O. Lamberson, R. J. Ozretich, R. C. Randall, K. R. Rukavina, D. W. Schults, K. A. Sercu, D. T. Specht, R. C. Swartz and D. R. Young (1994). Ecological Risk Assessment of the Marine Sediments at the United Heckathorn Superfund Site. Newport, OR 97365, U. S. Environmental Protection Agency Pacific Ecosystems Branch, ERL-N: 461.

Leonard, L. (1997). "Controls of sediment transport and deposition in an incised mainland marsh basin, southeastern North Carolina." Wetlands **17**(2): 263-274.

Lewicki, M. and L. J. McKee (2009). Watershed specific and regional scale suspended sediment loads for Bay Area small tributaries. A technical report for the Sources Pathways and Loading Workgroup of the Regional Monitoring Program for Water Quality. Oakland, CA, San Francisco Estuary Institute: 56.

Lick, W. (2009). Sediment and Contaminant Transport in Surface Waters. Boca Raton, FL, CRCPress.

Lionberger, M. and D. Schoellhamer (2009). A tidally averaged sediment-transport model for San Francisco Bay, California. U. S. G. Survey. U.S Geological Survey.

Lowe, S., B. Anderson and B. M. Phillips (2007). Investigations of sources and effects of pyrethroid pesticides in watersheds of the San Francisco Estuary. Oakland, CA, SFEI: 269.

Luengen, A. and A. R. Flegal (2009). "Role of phytoplankton in mercury cycling in the San Francisco Bay estuary." Limnology and Oceanography **54**(1): 23-40.

Luoma, S. N. and T. S. Presser (2000). Forecasting Selenium Discharges to the San Francisco Bay-Delta Estuary: Ecological Effects of a Proposed San Luis Drain Extension. Menlo Park, CA, U.S. Geological Survey Water Resources Division: 157 pp.

Mackay, D. and A. Fraser (2000). "Bioaccumulation of persistent organic chemicals: mechanisms and models." Environmental Pollution **110**: 375-391.

Marvin-DiPasquale, M. and R. S. Oremland (1998). "Bacterial Methylmercury Degradation in Florida Everglades Peat Sediment." Environmental Science and Technology **32**: 2556-2563.

Mason, R. P., J. R. Reinfelder and F. M. M. Morel (1995). "Bioaccumulation od mercury and methylmercury." Water Air And Soil Pollution **80**: 915-921.

Mason, R. P., J. R. Reinfelder and F. M. M. Morel (1995). "Bioaccumulation of mercury and methylmercury." Water Air and Soil Pollution **80**(1-4): 915-921.
McDonald, E. T. and R. T. Cheng (1997). "A numerical model of sediment transport applied to San Francisco Bay, California." Journal of Marine Environmental Engineering **4**(1): 1-41.
McKee, L., N. K. Ganju, D. H. Schoellhamer, J. A. Davis, D. Yee, J. Leatherbarrow and R. Hoenicke (2002). Estimates of suspended-sediment flux entering San Francisco Bay from the Sacramento and San Joaquin Delta. SFEI Contribution. Oakland, CA, San Francisco Estuary Institute**:** 1-28.
McKee, L. J., N. K. Ganju and D. H. Schoellhamer (2006). "Estimates of suspended sediment entering San Francisco Bay from the Sacramento and San Joaquin Delta, San Francisco Bay, California." Journal of Hydrology **323**: 335-352.
Meade, R. (1969). "Landward Transport of Bottom Sediments in Estuaries of the Atlantic Coastal Plain " Journal of Sedimentary Petrology **39**(1): 222-234.
Miller, R. E., K. J. Farley, J. R. Wands, R. Santore, A. D. Redman and N. B. Kim (In Press). "Fate and Transport Modeling of Sediment Contaminants in the New York/New Jersey Harbor Estuary." Urban Habitats.
Morel, F. M. M. (1983). Principles of Aquatic Chemistry. New York, NY, Wiley-Interscience.
Myre, P. L., D. E. Vidal-Dorsch and S. M. Bay (2006). California Sediment Quality Objectives Database User Guide. Draft. Port Townsend, WA, Exa Data & Mapping Services, Inc.
Nagle, N., G. Evanylo, W. Daniels, D. Beegle and G. Velva (1997). Chesapeake Bay Region Nutrient Management Training Manual. C. B. Program, Chesapeake Bay Program.
National Oceanic and Atmospheric Administration (2003). Watershed database and mapping projects/San Francisco Bay, National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration**:** 3.
Oram, J., Davis, J., and Leatherbarrow, J. (2008). "A Model of Long-Term PCB Fate in San Francisco Bay - Model Formulation, Calibration, and Uncertainty Analysis." San Francisco Estuary Institute Regional Monitoring Program for Trace Substances **2.1**.
Oram, J. J., L. J. McKee, C. E. Werme, M. S. Connor, D. R. Oros, R. Grace and F. Rodigari (2008). "A mass budget of polybrominated diphenyl ethers in San Francisco Bay, CA." Environment International **34**(8): 1137-1147.
Perillo, G. and J. Syvitski (2009). "Mechanisms of sediment retention in estuaries." Estuarine, Coastal and Shelf Science **87**(2010): 175-176.
Porterfield, G. (1980). Sediment transport of streams tributary to San Francisco, San Pablo, and Suisun Bays, California, 1909–1966, US Geological Survey.
Pritchard, D. W. (1955). "Estuarine circulation patterns." Proceedings of the American Society of Civil Engineers **81**(717): 1 - 11.
Rothenberg, S. E., L. McKee, A. Gilbreath, D. Yee, M. Connor and X. Fu (2010). "Evidence for short-range transport of atmospheric mercury to a rural, inland site." Atmospheric Environment **44**(10): 1263-1273.
Sabin, L. D., J. H. Lim, K. D. Stolzenbach and K. C. Schiff (2005). "Contribution of trace metals from atmospheric deposition to stormwater runoff in a small impervious urban catchment." Water Research **39**: 3929-3937.
San Francisco Estuary Institute (SFEI) (2007). The 2006 RMP Annual Monitoring Results. Oakland, CA, San Francisco Estuary Institute and the Regional Monitoring Program for Water Quality in the San Francisco Estuary**:** 232.
Sanford, L. (1992). "New sedimentaition, resuspension, and burial." Limnology and Oceanography **37**(6): 1164-1178.
Schemel, L. E., S. W. Hager and D. Childers Jr. (1996). The supply and carbon content of suspended sediment from the Sacramento River to San Francisco Bay. San Francisco Bay: The Ecosystem. J.

T. Hollinbaugh. San Francisco, CA, American Association for the Advancement of Science: 237-260.

Schoellhamer, D. (1996). "Factors affecting suspended solids concentrations in South San Francisco Bay, California." Journal of Geophysical Research **1010**(C5): 12087-12095.

Schoellhamer, D. (2011). "Sudden Clearing of Estuarine Waters upon Crossing the Threshold from Transport to Supply Regulation of Sediment Transport as an Erodible Sediment Pool is Depleted: San Francisco Bay, 1999." Estuaries and Coasts **34**(5): 885-899.

Schoellhamer, D., Lionberger, M., Jaffe, B., Ganju, N., Wright, S., and Shellenbarger, G. (2005). Bay Sediment Budget: Sediment Accounting 101. The Pulse of the Estuary. Oakland, CA, Regional Monitoring Program for Water Qaulity, San Francisco Estuary Institute.

Schoellhamer, D. H. (2009). Suspended Sediment in the Bay: Past a Tipping Point. The Pulse of the Estuary: Monitoring and Managing Water Quality in the San Francisco Bay. Oakland, Ca, San Francisco Estuary Institute: 56-65.

Schoellhamer, D. H., G. G. Shellenbarger, N. K. Ganju, J. A. Davis and L. J. McKee (2003). "Sediment dynamics drive contaminant dynamices." Pulse of the Estuary, SFEI, San Francisco: 21-27.

SFBRWQCB (2008). Total Maximum Daily Load for PCBs in San Francisco Bay. Oakland, CA, San Francisco Bay Regional Water Quality Control Board: 134.

SFEI (2010). Pulse of the Estuary 2010. Oakland, CA, San Francisco Estuary Institute: 96.

Sinkkonen, S. and J. Paasivirta (2000). "Degradation half-life times of PCDDs, PCDFs and PCBs for environmental fate modeling." Chemosphere **40**: 943-949.

Smith, L. (1987). A Review of Circulation and Mixing Studies of San Francisco Bay, California. Circular 1015, U.S. Geological Survey.

Steding, D. J. and A. R. Flegal (2002). "Mercury concentrations in coastal California precipitation: evidence of local and trans-Pacific fluxes of mercury to North America." Journal of Geophysical Research **107**(D24): 11-11 - 11-17.

Sunderland, E. M., J. Dalziel, A. Heyes, B. A. Branfireun, D. P. Krabbenhoft and F. Gobas (2010). "Response of a Macrotidal Estuary to Changes in Anthropogenic Mercury Loading between 1850 and 2000." Environmental Science & Technology **44**(5): 1698-1704.

Tsai, P. and R. Hoenicke (2001). San Francisco Bay atmospheric deposition pilot study Part 1: mercury. Oakland, CA, San Francisco Estuary Institute: 45p.

Tsai, P., D. Yee, H. A. Bamford, J. E. Baker and R. Hoenicke (2002). "Atmospheric Concentrations and Fluxes of Organic Compounds in the Northern San Francisco Estuary." Environmental Science & Technology **36**(22): 4741-4747.

U. S. EPA (2004). United Heckathorn Site Reinvestigation of Waterways Completed. San Francisco, CA: 6.

URS Corporation (2002). Task 5- Toxic Cleanup Sites, for the Clean Estuary Partnership.

USEPA (2005). Contaminated Sediment Remediation Guidance for Hazardous Waste Sites.

USEPA (2008). Guidance on the Development, Evaluation and Application of Environmental Models. C. f. R. E. Modeling. Washington: 89.

USEPA (2009). Understanding the Use of Models in Predicting the Effectiveness of Proposed Remedial Actions at Superfund Sediment Sites. Sediment Assessment and Monitoring Sheet.

USEPA (2010). Chesapeake Bay Compliance and Enforcement Strategy: 12.

Watras, C. J., R. C. Back, S. Halvorsen, R. J. M. Hudson, K. A. Morrison and S. P. Wente (1998). "Bioaccumulation of mercury in pelagic freshwater food webs." Science Of The Total Environment **219**: 183-208.

Watras, C. J. and N. S. Bloom (1992). "MERCURY AND METHYLMERCURY IN INDIVIDUAL ZOOPLANKTON - IMPLICATIONS FOR BIOACCUMULATION." Limnology And Oceanography **37**(6): 1313-1318.

Webster, E., D. Mackay and K. Qiang (1999). "Equilibrium lipid partitioning concentrations as a multi-media synoptic indicator of contaminant levels and trends in aquatic ecosystems." Journal of Great Lakes Research **25**: 318-329.
Weston, D. P., W. M. Jarman, G. Cabana, C. E. Bacon and L. A. Jacobson (2002). "An evaluation of the success of dredging as remediation at a DDT-contaminated site in San Francisco Bay, California, USA." Environmental Toxicology and Chemistry **21**(10): 2216-2224.
Wiener, J. G., D. P. Krabbenhoft, G. H. Heinz and A. M. Scheuhammer (2002). Ecotoxicology of Mercury. Handbook of ecotoxicology, 2nd ed. B. A. R. D.J. Hoffman, G.A. Burton, Jr., and J. Cairns, Jr. Boca Raton, FL, CRC Press**:** 407-461.
Winterwerp, J. and W. v. Kesteren (2004). Introduction to the physics of cohesive sediment in the marine environment. Amsterdam, The Netherlands, Elsevier.
Wood, R. and J. Widdows (2002). "A model of sediment transport over an intertidal transect, comparing the influences of biological and phisycal factors." Limnology and Oceanography **47**(3): 845-855.
Yee, D., L. J. McKee and J. J. Oram (2011). "A regional mass balance of methylmercury in San Francisco Bay, California, USA." Environmental Toxicology and Chemistry **30**(1): 88-96.

A. Appendix

Table A-1 - Subset of Superfund Sites (US EPA) and Toxic Hot Spots (SWRCB) in San Francisco Bay margins

Region	Site Name	Listing	Contaminants of Interest	Site History	Remediation Status
SU	Concord Naval Weapons Station	Superfund	Cu, OC pesticides	Ammunition transshipment port (1942-1999)	Ongoing
SU	Travis Air Force Base	Superfund	PAHs	Facilities for aviation fleet activities (1943-ongoing)	Ongoing
SU	Peyton Slough	Toxic Hot Spot	Cu, Se, PCBs, PAHs, OC pesticides	Copper ore smelter & pyrite roaster (~1899-1966)	Completed in 2009
SPB	Castro Cove / Chevron refinery	Toxic Hot Spot	Hg, Se, PAHs, OC pesticides	Petroleum refinery (1902-1987)	Ongoing
CB	United Heckathorn/Lauritzen Canal	Superfund	OC pesticides	Pesticide processing & distribution (1947-1966)	Ongoing
CB	Pt. Potrero / Richmond Harbor	Toxic Hot Spot	Hg, Cu, PCBs, PAHs	Ship building & scrapping (~1942-1985)	Active in 2005-2006; unclear if ongoing
CB	Stege Marsh / Richmond Field Station & Campus Bay	Toxic Hot Spot	Hg, Cu, Se, PCBs, OC pesticides	Manufactured explosives, pesticides, other chemicals (1870-1985)	Ongoing
CB	Alameda Naval Air Station	Superfund	PCBs	Facilities for aviation fleet activities (1936-1997)	Ongoing
CB	Hunters Point Naval Shipyard	Superfund	Hg, PCBs, OC pesticides	Facilities for ship building and repair (1940-1994)	Ongoing
SB	San Leandro Bay / General Electric Facilities	Toxic Hot Spot	Hg, Se, PCBs, PAHs, OC pesticides	Industrial (ongoing) incl. manufacturing transformers (G.E. 1923-1975)	Ongoing at G.E. site
LSB	Moffett Field Naval Air Station	Superfund	PCBs	Facilities for aviation fleet activities (1933-1991)	Ongoing

Region codes: SU=Suisun Bay, SPB=San Pablo Bay, CB=Central Bay, LSB=Lower South Bay

Table A-2 - Data sets evaluated for spatial patterns in mercury.

Parameter(s)	Matrix	Program [a]	Location	Sediment Habitat Types	N	Months [b]	Years	References
THg, MeHg, % clay, TOC	Sediment (0 - 5 cm)	RMP	Seven fixed trend sites	Shallow and deep bay	100	7, 8	1993 - 2007 [c]	(Conaway et al. 2003; Conaway et al. 2007; San Francisco Estuary Institute (SFEI) 2007)
THg, MeHg, TOC	Sediment (0 - 5 cm)	RMP	Probabilistic sampling sites, Bay wide	Shallow and deep bay	196	7, 8	2001-2006	(San Francisco Estuary Institute (SFEI) 2007)
THg, MeHg	Sediment (0 - 0.5 cm)	CBDA	Bay wide	Channel slough, bay mudflat, and shallow bay	111	10-12	1999	(Heim et al. 2007)
THg, MeHg, TOC	Sediment (0 - 2 or 0 - 3 cm)	RMP	East Bay shoreline (Albany)	Marsh slough, bay mudflat, and shallow bay	15	5	2007	Unpublished data
THg, MeHg, TOC	Sediment (0 - 2 cm)	PRISM	Six rivers and creeks, Bay wide	Creek	24	4,11	2005 - 2006	(Lowe et al. 2007)

a CBDA = California Bay Delta Authority Science Program; NSTP = National Status and Trends Program; PRISM = CA Pesticide Research and Identification of Source, and Mitigation Grant Program; RMP = Regional Monitoring Program for Water Quality in San Francisco Bay; USGS = U.S. Geological Survey b 1 = January, 2 = February, etc.
c MeHg analysis began in 2000 (N = 56); Spatial analyses (Tables 2 and 3) included data from 2001 through 2006 only (N = 67)

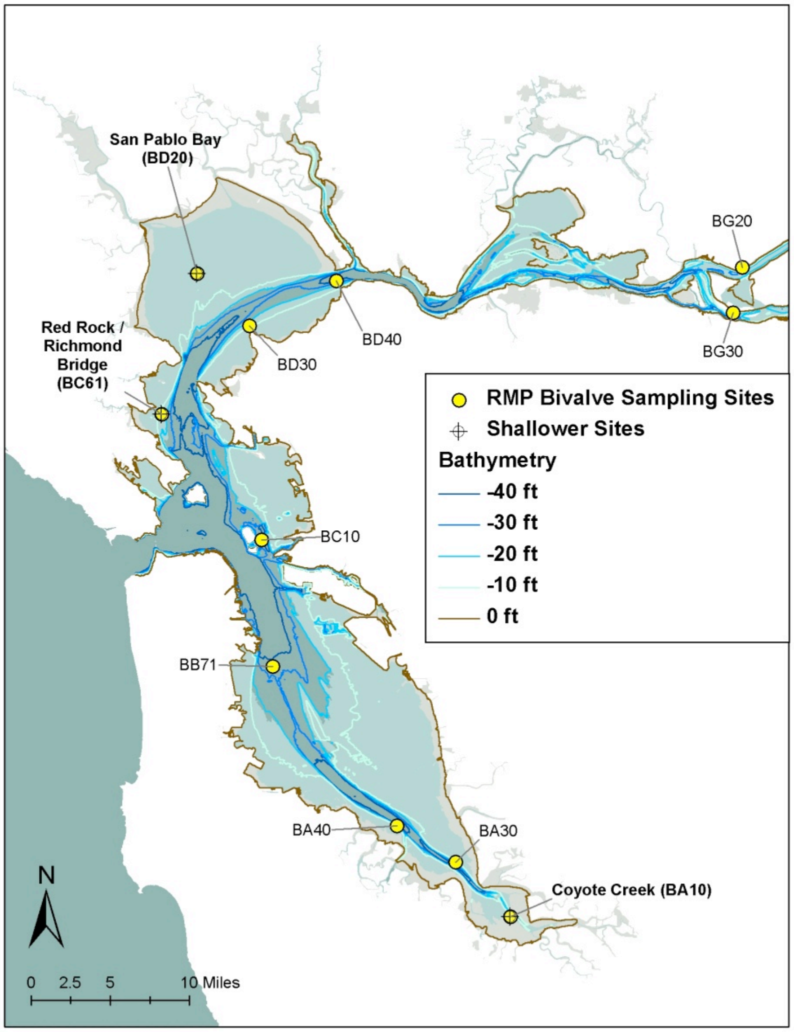

Figure A-2012749849 - RMP bivalve monitoring sites superimposed with 10-foot contours of the Bay (only to 40 feet deep).