SAN FRANCISCO BAY SUBTIDAL HABITAT GOALS REPORT

Appendix 2-1: Habitat Stressor Narrative Descriptions

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Contents

Sources and Impacts of Chemical Contaminants in San Francisco Bay

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This paper discusses some of the major chemical contaminants in San Francisco Bay, including aspects of their sources, loading and pathways, and their impacts on organisms and human health. Inputs of nutrients into the Bay, considered a pollutant in some contexts, are addressed in the accompanying paper on "Sources, Mechanisms and Impacts of Changes in Nutrient Inputs to San Francisco Bay" (Cohen 2008). Exotic species, which are classified as a biological pollutant under the Clean Water Act and have been confirmed as such by recent federal court decisions, are addressed under the stressor "Release Exotic Organisms."

A wide variety of contaminants have been the subject of regulation, monitoring or research in the Bay. With limited resources, this paper has relied substantially on recent review articles, which were available or more complete for some contaminants than for others. Accordingly, there is less text on some currently important contaminants than on certain contaminants of lesser or declining regulatory significance. Still, the stories of these "declining" contaminants—why they were once of greater concern, and why the perception of their significance has changed—provide important context for our current understanding of contaminants in the Bay.

In a recent review, Davis et al. (2007b) offered an assessment of the current state of contaminants in the Bay. They sorted contaminants and contaminant issues into several categories based on their status and prognosis (Table 1).[1] They classified mercury, polychlorinated biphenyl compounds (PCBs) and dioxins to be the most serious current problems among the chemical contaminants because water quality in all parts of the Bay is impaired by them, because the concentrations of these contaminants in water or biota are well above established levels of concern, and because their abundance and persistence in sediment and their slow rates of depletion mean that they are likely to remain a problem for decades. Other contaminants of substantial concern include selenium, polycyclic aromatic hydrocarbons (PAHs), and the banned organochlorine pesticides DDT, chlordane and dieldrin.

The significance of some contaminants may also be inferred from their listing as causes of impairment for various portions of the tidal waters of San Francisco Bay, in the most recent Clean Water Act 303(d) list (see Appendix A), and by recent regulatory activities (Table 2, modified from Mumley 2007).

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[1] Organic waste or enrichment, nutrients and exotic species are included as pollutants both in Davis et al. (2007b)'s assessments and on the 303(d) list, and exotic species qualify as biological pollutants under the Clean Water Act, as confirmed by recent federal court decisions. These three types of pollutants are listed in Table 1 and Appendix A, but as noted above are not otherwise treated in this paper on chemical contaminants since they are classified as distinct stressors.

Table 1. Current status and prognosis for main San Francisco Bay contaminant issues. Based on Davis et al. (2007b).

Contaminant Issue	Status	Prognosis
Mercury, Exotic Species	Biggest impacts	Further deterioration is likely unless effective management actions can be implemented
PCBs, Dioxins	Biggest impacts	Trend is toward slow reduction, but unlikely to fall below risk thresholds within 20 years
Selenium, PAHs	Significant threats remain	Trend is unclear, could fall below risk thresholds within 20 years
Legacy Pesticides (DDTs, Chlordane, Dieldrin)	Significant threats remain	Trend is toward steady reduction, likely to fall below risk thresholds within 20 years
Organic Waste, Nutrients, Silver	Problems largely solved	Trend is toward further recovery
Nickel, Copper	Problems largely solved	Will be carefully watched
PBDEs, Pyrethroids, Sediment Toxicity, Pollutant Mixtures	Emerging issues	Concerns growing due to increases in inputs or knowledge

Table 2. Some recent regulatory activities regarding contaminants in San Francisco Bay. Modified from Mumley (2007).

Contaminant	Regulatory Activity
Copper	Removed from 303(d) list in 2002, site-specific objectives adopted for South Bay in 2002 and for rest of Bay in June 2007.
Cyanide	Site-specific objectives adopted in December 2006.
Diazinon	Removed from 303(d) list in 2006.
Dioxins/Furans	On 303(d) list, TMDL project plan being developed.
DDTs, Chlordane, Dieldrin	On 303(d) list, TMDL project plan being developed.
Mercury	On 303(d) list, initial TMDL in 2004, revised TMDL and site-specific objectives in 2006.
Nickel	In South S.F. Bay, removed from 303(d) list and site-specific objectives adopted in 2002. Attained California Toxics Rule objectives in rest of Bay, Water Board to consider delisting in early 2008.
PCBs	Water Board to consider adopting a TMDL by end of 2007.
Selenium	TMDL project started in 2007, Water Board consideration in 2008/2009.

Background

By the late 19th and early 20th centuries, observers were commenting on the polluted state of parts of San Francisco Bay. Untreated domestic and industrial wastes led to contamination of shoreline areas by fecal bacteria and created anaerobic conditions near sewage outfalls and in the waters at the southern end of the Bay (Krieger et al. 2007), which promoted the growth of bacteria that caused avian botulism and cholera. Oily discharges were also common from the many refineries on the Bay shore, starting with the construction of the Union Oil Refinery in 1896. In the 1940s, synthetic organic pesticides were developed and applied to Central Valley farms and began washing into the Bay (Davis et al. 1991). Even as cities took the first steps toward cleaning up their waste streams with primary treatment in the late 1940s and 1950s (Davis et al. 1991; Buck et al. 2007; Krieger et al. 2007; Table 3), continued urbanization and population growth increased the volume of waste discharges and added further pollutants in urban runoff. In these decades, San Francisco Bay suffered from periodic incidents of oxygen depletion, which were accompanied by foul (hydrogen-sulfide) odors and sometimes by fish kills (e.g. Nichols 1979; Luoma and Cloern 1982; Cloern and Oremland 1983; Krieger et al. 2007).

Table 3. Some of the early primary treatment plants that handled wastewater discharged to San Francisco Bay. From Krieger et al. (2007).

Year	Community or Agency
1934	Palo Atlo
1938	Petaluma
1948	Central Contra Costa Sanitary District
1950	Ora Loma Sanitary District
1951	San Francisco (North Point Treatment Plant) East Bay Municipal Utility District Mountain View
1952	San Francisco (Southeast Treatment Plant)
1954	Hayward
1956	San Jose/Santa Clara Sunnyvale
1957	Los Altos

Things began to change in the late 1960s and 1970s with the passage of state and federal water pollution laws, rising public concern about the state of the environment, and other developments (Table 4). The federal Clean Water Act provided over a billion dollars to upgrade Bay Area wastewater plants; the federal government banned the manufacture and use of PCBs, DDT, dieldrin and chlordane; many Bay Area military bases and industrial plants have closed; and a great deal of regulatory attention, research, and public agency and industry effort have been applied to reducing contaminant discharges. As a result, several water quality problems that were of great concern in the 1970s have largely been resolved (Mumley 2007; Davis et al. 2007b). With the construction of secondary treatment facilities for municipal wastewater in the 1970s and 1980s, the discharge of suspended solids and biological oxygen demand (BOD) in wastewater effluent dropped sharply (Davis et al. 1991; Krieger et al. 2007), and hypoxic events have become rare in San Francisco Bay (Nichols et al. 1986; Cloern et al. 2003) The concentrations of several important contaminants in effluents, Bay water or sediments have also declined substantially, including a more than 95% reduction in total trace metal loadings into the Bay from municipal wastewater plants, from 943 tons/year in 1960 to 46 tons/year in 1999 (Davis et al. 1991; Van Geen and Luoma 1999; Squire et al. 2002; Davis et al. 2007b; Krieger et al. 2007). Copper and the organophosphate pesticide diazinon, which were listed as causes of impairment of the Bay's water quality in the 1990s, have been delisted;[2] nickel has been delisted for southern San Francisco Bay and delisting is under consideration for other segments.

Despite notable progress in understanding and managing many important contaminants in the Bay, there are reasons to be concerned for the future. Freshwater diversions have reduced the Bay's flushing capacity by about 30%, and the regional consequences of global climate change may reduce it further (Flegal et al. 2007). Large-scale tidal marsh restoration that is now underway may alter aspects of biogeochemical cycling, which could potentially include making some important contaminants more available to the biota (Davis et al. 2007b). Other concerns include an unexplained pattern of sediment toxicity in the Bay, the potential for increased erosion of Bay sediments to expose legacy contaminants, a dearth of information about possible synergistic effects between contaminants, and the problem of emerging contaminants. These issues are discussed briefly below, followed by discussions of a few of the current contaminants of concern in the Bay (mercury, PCBs, DDT, chlordane, dieldrin and PAHs), and two contaminants for which the level of concern has subsided (copper and nickel).

Sediment Toxicity

Sediment toxicity has been an unresolved puzzle in the Bay since it was first documented in the mid-1980s (Anderson et al. 2007). Different studies have found evidence of toxicity in Bay sediments, sediment elutriates, and sediment pore waters, using a variety of tests involving amphipods and the embryos of mussels and sea urchins (Anderson et al. 2007). Initial screenings at 127 sites by the Bay Protection and Toxic Cleanup Program (BPTCP) found that 21% of sediment samples were toxic to amphipod and 31% of pore water samples were toxic to sea urchin embryos (Anderson et al. 2007). In every seasonal sampling period since RMP testing began in 1993, at least one-third of the sediment samples were toxic to one or more test species, and in 1997-2001 at least 63% of the sediment samples were toxic to at least one test organism. A few stations at the north and south ends of the Bay were consistently toxic to bivalve embryos, while some but not all of these plus other stations were consistently toxic to amphipods (Anderson et al. 2007).

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[2] Site-specific water quality objectives for copper were adopted in 2007 in segments north of the Dumbarton Bridge (Mumley 2007); see discussion of copper below.

Table 4. Some key events in the management of contaminants in San Francisco Bay. Sources: Leatherbarrow et al. 2006; Buck et al. 2007; Davis et al. 2007a; Yee et al. 2007; Mumley 2007.

1934	A primary wastewater treatment plant was constructed at Palo Alto, probably the first in the Bay.
1948	The first Federal Water Pollution Control Act authorized the Surgeon General to prepare plans to reduce pollution.
1949	Dickey Water Pollution Act created the state and regional water boards.
1950s	Construction of primary wastewater treatment plants by most communities discharging into the Bay.
1960s	Construction of the first secondary wastewater treatment plants in the Bay.
1969	Porter-Cologne Water Quality Control Act.
1972	Clean Water Act (also known as the Federal Water Pollution Control Act) started a discharge permit system, required secondary treatment for municipal discharges, and provided grant funding to defray the costs of building and operating treatment plants. Federal ban on DDT.
1975	The first complete S.F. Bay Basin Plan designated the beneficial uses of the Bay; primarily addressed "conventional" pollutants (suspended solids, biological oxygen demand (BOD) and bacteria); and included a narrative water quality objective for toxic contaminants.
1979	Federal ban on sale and production of PCBs.
1986	S.F. Bay Basin Plan included water quality objectives and effluent limits for some metals in some parts of the Bay.
1988	Federal ban on chlordane.
1989	Bay Protection and Toxic Cleanup Program established. Federal ban on dieldrin.
1992	Regional Monitoring Program established.
1994	An interim fish consumption advisory was issued for all of San Francisco Bay, based primarily on high PCB and mercury concentrations in fish.
1995	The S.F. Bay Basin Plan's narrative water quality objective was amended to specify that wildlife and human health be protected against the bioaccumulation of toxic contaminants.
1996	The Bay was listed under Clean Water Act §303(d) as impaired by copper, mercury, nickel and selenium (modifying earlier listings of impairment by metals).
1998	The Bay was listed as impaired (under §303(d)) because of a Bay-wide fish consumption advisory based on elevated tissue concentrations of PCBs, dioxins, furans, chlordane, DDT and dieldrin; and listed as impaired by the organophosphate pesticide diazinon due to episodic toxicity in the Bay after storm runoff.
2000	The California Toxics Rule was adopted by the US EPA, which set water quality objectives for the Bay based on federal criteria, including a numerical objective for copper.
2002	Copper was removed from the Bay's 303(d) list, and nickel removed from the list for southern San Francisco Bay.
2006	Diazinon was removed from the Bay's 303(d) list. A revised TMDL for mercury was completed.
2007	A selenium TMDL project was started.

In the RMP samples, spatial patterns of toxicity to sediment and sediment elutriates did not match. The magnitude and frequency of sediment toxicity was greater during the winter wet season, and toxic sites were associated with Delta inflow in the northern Estuary and with urban creeks, suggesting that the toxicity may be derived from storm water inflows (Anderson et al. 2007). While sediment toxicity has been correlated to some degree with some trace metals, PAHs and organochlorine pesticides, the contaminant or contaminants causing the toxicity have not been identified. Since sediment quality objectives are now being developed for California (Anderson et al. 2007), these observations of unexplained sediment toxicity may end up driving regulatory decisions (Davis et al. 2007b).

Legacy Contaminants and Increased Erosion of Bay Sediments

"Legacy contaminants," whose concentrations in the Bay are due more to historic than to current inputs, include several trace metals (mercury, copper, nickel, silver) as well as some organochlorine compounds (PCBs, DDTs, chlordanes and dieldrin) that have not been commercially available for use in the Bay watershed for decades, but are still present at high enough levels to contribute to official warnings about the consumption of certain fish (Flegal et al. 2007; Davis et. al 2007a; Connor et al. 2007).

Bay sediments can be divided conceptually into two categories: active sediments that are at or near the surface and exchange with the water column by diffusion or by physical or biological mixing, and buried sediments that lie below the active layer and are isolated from the water column and organisms (Davis et. al 2007a). The depth of the active layer varies from place to place, and the general range is suggested by Bay organisms that turn over or irrigate sediments to depths of up to a few centimeters (snails, sea slugs, juvenile clams), up to 10-30 cm (lugworms, deeper burrowing clams), or up to 50-75 cm (bat rays, various polychaetes, and ghost shrimp) (Rubin and McCulloch 1979; Peterson 1979; Haderlie and Abbott 1980; Haig and Abbott 1980), and physical turnover due to wind waves or tidal currents that ranges from 2-5 cm depth in mud-clay sediments in Central and South Bays, to 40-100 cm depth in sand in the western part of the Central Bay (Rubin and McCulloch 1979; Hammond and Fuller 1979).[3]

Sediment flows to the Bay appear to be declining due to reduced water flows and the impoundment and retention of sediments in reservoirs (McKee et al. 2006; Jaffe et al. 2007), and recent assessments of bathymetric data suggest that the Bay as whole may now be losing sediment (Jaffe et al. 1998; Cappiella et al. 1999; Foxgrover et al. 2004; Schoellhamer et al. 2005). Planned floodplain and wetland restoration projects are likely to trap sediment and further reduce the sediment supply to the Bay (Davis et. al 2007a). All this suggests an increased and increasing rate of erosion of Bay bottom sediments, which can expose legacy contaminants in the buried layer that have been effectively isolated from the Bay. It is important to note, however, that the phenomenon of eroding bottom sediments is not new in the Bay; bathymetric analyses show that even in Bay

[3] See the accompanying paper, "Impacts of the Removal or Disturbance of Sediments, Shells or Bedrock in San Francisco Bay", for further description of the turnover of Bay bottom sediments.

segments that are accumulating large amounts of sediment, substantial erosion may be occurring over parts of those segments during the same period (e.g. Cappiella et al. 1999).

Synergisms

Every Bay organism is exposed over its lifetime to hundreds of chemicals at various concentrations and in different, dynamically shifting combinations. However, almost nothing is known about the potential effects of such multi-chemical exposures, since nearly all studies of contaminant impacts on aquatic organisms have focused on one chemical at a time (Davis et al. 2007b). Especially where a contaminant is present in concentrations that are near threshold effect levels, there is concern that it could work in combination with other stresses (including other contaminants) to impair sensitive life-history processes such as reproduction, development, sexual differentiation, etc. (Davis et. al 2007a). The subtlety and potential complexity of such interactions means that the impacts of some contaminants could, in combination with others, be substantially greater than has been recognized. Currently unexplained phenomena — the widespread sediment toxicity in the Bay, pelagic organism decline, the relative scarcity of native oysters in the Bay — could possibly be the result of synergisms between contaminants (Davis et al. 2007b).

Emerging Contaminants

There are more than 7 million chemical compounds that are commercially available in the U.S. (Hoenicke et al. 2007), and the potential effects in the aquatic environment of most of these are largely or entirely unknown. Recent analyses have detected a wide variety of anthropogenic compounds in the water, sediments or tissue samples from San Francisco Bay, including flame retardants, pesticides and insecticide synergists, insect repellents, pharmaceuticals, ingredients of personal care product, plasticizers and non-ionic surfactants (Hoenicke et al 2007). Two groups of compounds that have recently been garnering attention are PBDEs and pyrethroid insecticides.

Polybrominated diphenyl ethers (PBDEs) are a type of flame retardant that has become very common in commercial goods since the early 1990s. PBDE concentrations in Bay Area wildlife and humans are among the highest reported in the world, and there is growing laboratory evidence of toxic effects from these chemicals. California banned the use of two types of PBDEs in 2006, and the US EPA is expected to establish a threshold for concern for PBDEs soon (Davis et al. 2007b).

As organophosphate pesticides have been phased out, the use of pyrethroid insecticides has increased in agriculture and for pest control around homes. Fish and aquatic invertebrates are sensitive to pyrethroids. They have often been linked to sediment toxicity and are suspected as a factor in pelagic organism decline (Davis et al. 2007b).

Mercury

Davis et al. (2007b) described mercury as "Bay water quality enemy number one." Recent studies have concluded that elevated mercury concentrations in fish tissue in the Bay pose a human health risk (Flegal et al. 2007), and this is a primary reason for the Bay-wide fish consumption advisory. Mercury may also be harming wildlife populations, including the endangered California clapper rail (Schwarzbach et al. 2006; Davis et al. 2007b).

Mercury contamination in the Bay is mainly derived from historic mercury and gold mining (Flegal et al. 2007). An estimated 12,000 metric tons of mercury was used in California, mainly in the Sierra Nevada, to separate gold from ore, with about 40% of this lost to the environment by placer mining operations (Conaway et al. 2007). Although Davis et al. (2006) reported that there has been no general decline in mercury concentrations in Estuary fish over the past 30 years, Conaway et al. (2007) found that RMP data from 1993-2001 showed a modest decrease in surface sediment concentrations of total mercury, which they attributed to burial of older, more contaminated sediments by relatively cleaner recent sediment the Central Valley watershed.

Mercury is present in the Bay in multiple chemical forms. Typically only about one percent of total mercury is present as methylmercury, but since methylmercury is biomagnified through the food web and is a neurotoxin that is especially harmful to early stages of human and animal development, it is the form of mercury in the Bay that is of greatest concern (Conaway et al. 2003; Davis et al. 2007b; Flegal et al. 2007).

Methylmercury concentrations often change substantially over short distances and short times, and do not correlate closely with concentrations of total mercury (Davis et al. 2007b; Flegal et al. 2007). The factors that control methylmercury concentrations in the Bay are not well understood (Davis et al. 2007b). Since wetlands appear to be sites of methylmercury production, the ongoing restoration of Bay wetlands raises concerns(Davis et al. 2006; Greenfield et al. 2006; Davis et al. 2007b). On the other hand, some recent studies suggest that some wetlands can trap methylmercury and render it unavailable for biotic uptake (Davis et al. 2007b).

Several studies have found elevated levels of mercury in Bay biota. Greenfield et al. (2006) reported that 40% of small fish sampled, representing seven species from eight locations in the Bay, had mercury concentrations that were higher than the 0.03 µg/g (wet-weight) TMDL target for prey fish tissues.

Most of the mercury in the Bay is bound to sediment particles and is distributed so widely in the Bay and its watershed that it will take many decades for the Bay's total mercury concentrations to decline significantly (Davis et al. 2007b). Any more rapid improvement in the status of mercury in the Bay will depend on identifying and implementing effective management actions to control methylmercury (Davis et al. 2007b).

PCBs

PCBs bind to sediment particles and are highly persistent in the environment. In the Bay they are distributed primarily around the shallow margins of the shore, associated primarily with industrial areas and with the mouths of creeks draining industrial areas (Davis et. al 2007a; Davis et al. 2007b). They bioaccumulate and can suppress growth, cause developmental abnormalities, disrupt endocrine pathways, harm immune systems and cause cancer in humans and in wildlife at the top of the food chain. PCBs may also harm young fish (Davis et al. 2007b).

The commercial use of PCBs was phased out during the 1970s, and the federal government banned their sale and production in 1979. Since then there has been a gradual decline in PCB concentrations in the Bay (Davis et. al 2007a). However, 25 years later, concentrations in some Bay sport fish are still more than 10 times the thresholds of concern for human health.

PCBs are, along with mercury, a primary cause for the Bay-wide fish consumption advisory and for classifying the Bay as an impaired water body. To meet human health targets and protect Bay wildlife will require a greater than 90% reduction in contamination levels in Bay organisms. This is likely to take many decades since the concentrations of PCBs are currently far above the threshold for concern, they are distributed widely in sediments in the Bay and in the watershed, and concentrations in sediments decline slowly (Davis et. al 2007a; Davis et al. 2007b). Erosion of Bay bottom sediments may also expose older, more contaminated sediments lying underneath (Davis et. al 2007a).

Organochlorine Pesticides

Organochlorine pesticides (which include DDTs, chlordanes and dieldrin) came into use in the 1940s and 1950s as insecticides on farms and for pest control and mosquito abatement in urban areas (Leatherbarrow et al. 2006; Connor et al. 2007). They were used abundantly in California, and by the 1980s, the San Joaquin River had some of the highest concentrations of DDT of any U.S. river system (Gilliom and Clifton, 1990). Human health and environmental concerns led to restrictions on DDT use in California starting in 1963, most agricultural use was banned in the state by 1970, and DDT was banned by the federal government in 1972. Similarly, chlordane came into use in the 1940s to control termites and other insects, but agricultural use was banned in 1975 and in the U.S. in 1978, and using it to protect structures was banned in 1988. Dieldrin began to be used around 1950 for termite protection, for moth-proofing and to protect cotton, corn and citrus crops; most agricultural use was banned in 1974, most other uses in 1985, and use for termite control in 1987 (Leatherbarrow et al. 2006; Connor et al. 2007).

The organochlorine pesticides are neurotoxicants, they can also affect reproductive development, they are classified as probable carcinogens by the US EPA, and they may also function as endocrine disruptors (Leatherbarrow et al. 2006). They persist in the environment and biomagnify in aquatic biota, accumulating in tissues that are high in lipids. Among fish, shiner surfperch and white croaker, and some white sturgeon, which have high lipid content, tend to have the highest tissue concentrations of organochlorine pesticides. For decades after the use of these pesticides was restricted and banned, concentrations in the Bay's water and fish tissues continued to exceed threshold concentrations of concern for human health (Gunther et al. 1999; Greenfield et al. 2002; Leatherbarrow et al. 2006). They continued to arrive in the Bay in flows from the Central Valley, some local tributaries and storm drains—presumably derived from deposits in soils—along with small amounts in atmospheric deposition and very minor quantities in wastewater discharges (Bergamaschi et al. 2001; McKee et al. 2004; Connor et al. 2007). Inflow of DDT to the Bay is estimated to total about 60 kg/yr, with about 71% coming from local watersheds and about 27% coming from the Central Valley. A substantial amount (9 kg) is also estimated to erode out of burial in Bay sediments each year. Inflow of chlordane is estimated at 30 kg/yr, with over 90% coming from local watersheds, and of dieldrin at 10 kg/yr, with 55% from the Central Valley and 33% from local watersheds (Connor et al. 2007). These pesticides can also disperse out of historic "hot spots" including a former pesticide packaging plant on the Lauritzen Canal in Richmond Harbor (Leatherbarrow et al. 2006). They are only weakly soluble in water, and most of the biologically accessible organochlorine pesticide in the Bay (≈97-99%) is in the upper, active layer of sediment rather than the water column (Connor et al. 2007). The presence of elevated concentrations of organochlorine pesticides in Bay fish tissues was one of the reasons for issuing a fish consumption advisory in 1994, and they are included as a cause of Bay-wide impairment in the Bay's current 303(d) list (Leatherbarrow et al. 2006; Davis et al. 2007b).

However, concentrations of these pesticides have continued to decline in sediment cores, bivalves and sport fish (Leatherbarrow et al. 2006). In recent sampling, very few sport fish samples exceeded concern thresholds for these pesticides, and as their concentrations are continuing to fall it is likely that within 20 years no sport fish tissue will be above thresholds (Davis et al. 2007b). The decline in bioavailable organochlorine pesticides in the Bay are mainly a result degradation of the compounds in the sediments, followed by ouflow in tidal currents through the Golden Gate and volatilization.

Polycyclic Aromatic Hydrocarbons (PAHs)

The concentrations of PAH in Bay water and sediments have remained relatively constant over the past 20 years (Oros et al. 2007; Davis et al. 2007b). The concentrations in Bay sediments may pose a risk to early life stages of fish (Davis et al.

2007b), and PAH concentration thresholds that have been recommended by NOAA to protect estuarine fish are frequently exceeded (Oros et al. 2007). PAHs are included on the 303(d) list as causing impairment at several Bay locations (Davis et al. 2007b).

In urban areas, PAHs are emitted into the atmosphere by a variety of processes including vehicle emissions and the burning of heating oils, wood and other biomass. From the air, PAHs can enter the water column by gaseous exchange across the air–water interface, by dry deposition of particulate matter, or by wet deposition in rainfall, and may enter the Bay directly or by runoff from streets and storm drains. Unburned fossil fuels can introduce PAHs into the Bay via street runoff or spills (Oros et al. 2007). The current maximum annual loading into the Bay is estimated at 10,700 kg/year, from the following sources: storm water runoff (51%), inflow from tributaries (28%), wastewater plant discharges (10%), atmospheric deposition (8%) and the disposal of dredge material (2%).

Concentrations of PAHs in Bay waters are highest in the Lower South Bay (120 ng/L), followed by the South Bay (49 ng/L), San Pablo and Suisun bays (29 ng/L), and lowest in the Central Bay (12 ng/L) (Ross and Oros 2004; Oros et al. 2007). Higher concentrations in the South Bay and especially the Lower South Bay may be due to a higher density of inputs from storm water discharge and atmospheric deposition, and perhaps longer residence times (Oros et al. 2007).

Although atmospheric concentrations of PAHs have declined in the Bay Area over the past ten years, increased motor vehicle use in the future could raise PAH levels in the Bay (Davis et al. 2007b). If, however, PAH emissions and inputs to the Bay can continue to be reduced, then concentrations in the Bay should drop relatively quickly (Davis et al. 2007b).

Copper

Copper is a micronutrient needed for phytoplankton growth, but when free copper ions are present in high concentrations they block the uptake of other micronutrients (manganese and zinc) (Buck et al. 2007). At free ion concentrations above 10^{-11} M (≈0.0006 µg/L), copper can be toxic to aquatic microorganisms (Brand et al. 1986; Buck et al. 2007). Copper in water has typically been measured in three categories, as *total dissolved copper*, as *exchangeable copper*, which is the copper associated with suspended sediments that could leach into the dissolved phase, and as *total dissolvable copper*, the sum of dissolved copper and exchangeable copper. In practice, dissolved copper is measured in a water sample that has been filtered through typically a 0.20- or 0.45-micron filter (over this range, the different filter sizes produce only minor differences in the results), and exchangeable copper is the amount of copper leached from filtered particles with a weak acid over a specified time (Buck et al. 2007).

Gordon (1980) made the first reliable measurements of dissolved copper concentrations in the Bay, using rigorous trace metal clean techniques (Buck et al. 2007). He measured copper concentrations in the Bay ranging from 0.4-2.8 μg/L, which was considerably higher than the 0.07-0.4 μg/L range that he measured in coastal waters (Buck et al. 2007). In the portion of the Bay between Suisun Bay and the Central Bay where he took his samples, Gordon (1980) found that copper concentrations were related to salinity levels during periods of high Delta outflow, but that copper concentrations rose higher relative to salinity during low Delta outflows. Gordon (1980) suggested this could be due to anthropogenic inputs of copper into South Bay waters, which then infiltrated into and raised copper concentrations in the Central and northern Bay when freshwater inflows were low (Buck et al. 2007).

Kuwabara et al. (1989) measured total dissolved copper concentrations in the South Bay that that ranged from 1.8-4.2 μg/L and regularly exceeded 3.1 μg/L (=49 nM), a concentration guideline that was implemented by the US EPA as a national criterion continuous concentration in 1995 (Buck et al. 2007). Flegal et al. (1991) found that copper concentrations in the South Bay exceeded concentrations in the northern Bay when adjusted for salinity (Buck et al. 2007). Data from the Regional Monitoring Program for 1993-2001 showed total dissolved copper concentrations that were highest in the South Bay (2.0 μg/L) and the sloughs at the southern end of the South Bay (2.4 μg/L), lower in Suisun and San Pablo bays (1.8 μg/L), and lowest in the Central Bay (1.0 μg/L) (Buck et al. 2007). Along with nickel, mercury and selenium, copper was determined to be impairing beneficial uses in the lower South Bay, resulting in this section of the Bay being included on the Clean Water Act 303(d) list of impaired water bodies in 1998 (Yee et al. 2007).

These findings focused attention on the sources of copper in South Bay waters. Loadings from the largest municipal wastewater treatment plants in the area had been 343 kg/day in 1975, but were reduced to 52 kg/day by 1985 (Davis et al. 1991); total wastewater loadings of copper into the South Bay have since been reduced further, to 28-46 kg/day (Buck et al. 2007). The 1993-2001 RMP data, when adjusted for dissolved organic carbon (DOC) and Delta outflow, similarly showed that over this period copper concentrations in the Bay dropped by 44% in South Bay sloughs, by 29% in the South Bay, and by 17% in Suisun and San Pablo bays (Buck et al. 2007). Meanwhile, Gee and Bruland (2002) found that desorption of copper from sediment particles that were resuspended in the water column was probably a larger source of copper in the South Bay than wastewater discharges (Buck et al. 2007).

In the absence of organic material, at typical pH values for seawater, about 7% of total dissolved copper occurs as free copper ions (Buck et al. 2007). At dissolved copper concentrations of 2 μg/L (typical for the South Bay), the concentration of copper ions in the absence of organic material would be around 0.14 μg/L, which is a couple of orders of magnitude above the probable threshold level for toxic effects of about 0.0006 μg/L (10^{-11} nM) (Brand et al. 1986). Since large scale copper toxicity events have not been observed in San Francisco Bay in recent years, a large part of the dissolved copper in the Bay is probably bound up with organic molecules (Buck et al. 2007). This is consistent with Kuwabara et al.'s (1989) finding that dissolved copper concentrations in the South Bay correlated with concentrations of dissolved organic carbon (DOC), and with the 1993-2001 RMP data, in which dissolved copper concentrations correlate positively with DOC and negatively with Delta outflow (Buck et al. 2007)

Studies of copper speciation in San Francisco Bay undertaken between 1994 and 2005 generally confirmed earlier studies in northeast Pacific coastal waters (Coale and Bruland 1988), which had found that >99% of total dissolved copper was tightly bound to organic molecules and unavailable for uptake by biota, and that the concentrations of the potentially toxic forms of copper were not consistently related to the concentrations of total dissolved copper (Buck et al. 2007). In the first of these studies in the Bay, 80-92% of dissolved copper at a South Bay site near the Dumbarton Bridge was bound to organic molecules, with the strongest-binding type of molecules (called L_1) present at concentrations sufficient to bind only 27% of the copper, the rest complexing with weaker binding molecules (L_2) (Donat et al. 1994). However, several studies conducted since 2000 consistently found that over 99.9% of the dissolved copper was bound to the L_1 or strongest-binding molecules, with a large excess of the weaker L_2-type molecules available to "back up" the L_1 molecules (Buck et al. 2007). In addition, it was found that most of the dissolved copper that enters the South Bay in wastewater effluent is probably already strongly bound to organic molecules in the discharge (Sedlak et al. 1997).

Since the analytical methods used in the earlier (1994) and later (2000-2005) studies of copper speciation in the Bay were similar (Buck et al. 2007), it's unclear why the results were different. In any event, organic molecules capable of strongly binding copper are now present in the Bay in sufficient concentrations to keep the concentration of free copper ions below 10^{-13} nM throughout the Bay, well below the probable threshold for aquatic toxicity of around 10^{-11} nM (Buck et al. 2007). The sources of these binding molecules, which have not been structurally characterized, are not known, but strongly binding molecules have been observed in the exudates from phytoplankton, especially cyanobacteria (which a recent studies have found to be flourishing in the South Bay), and weaker binder molecules in wastewater and runoff from soils (Buck et al. 2007).

In summary, copper was considered a major problem in San Francisco Bay when some of the early measurements found that total dissolved copper concentrations regularly approached or exceeded state and federal criteria (Flegal et al. 1991; Flegal et al. 2007; Buck et al. 2007). However, copper toxicity is due to the concentration of copper ions, rather than to total copper or total dissolved copper (Buck et al.2007). Subsequent studies of copper speciation showed that >99.9% of the copper in the Bay is tightly bound in organic molecules and is not readily available for biotic uptake (Flegal et al. 2007; Buck et al. 2007). Copper ion concentrations appear to be well below the threshold levels for aquatic toxicity, and there is a substantial excess of organic molecules that are capable of at least weakly binding copper, which should buffer any further additions of copper (Buck et al. 2007). As a result of these findings, copper was removed from the Bay's 303(d) listing in 2002, and new site-specific water quality objectives for copper were dadopted in June 2007 for Bay segments north of the Dumbarton Bridge (Davis et al. 2007b; Buck et al. 2007; Mumley 2007).

Nickel

The Coast Range rock formations that surround San Francisco Bay include serpentine and ultramafic rocks that contain high concentrations of nickel, and it is likely that the geologic composition of this watershed is the cause of relatively high nickel concentrations in the Bay (Yee et al. 2007). Nickel is also common in some industrial products, and enters the Bay through wastewater discharges and urban runoff (Yee et al. 2007), though the nickel in wastewater discharges may be relatively unavailable for biological uptake (Yee et al. 2007). Certain estuarine species, including some mollusks and crustaceans, are notably sensitive to nickel (Yee et al. 2007).

From measurements made by the RMP in 1993-2003 and the City of San Jose in 1997-2007, the Lower South Bay had the highest concentrations in the water column of both total nickel (13.7 µg/L) and dissolved nickel (4.0 µg/L). Suisun and San Pablo bays had the next highest concentrations (≈8 µg/L of total nickel), and Central Bay had the lowest (2.6 µg/L total and 1.3 µg/L dissolved nickel) (Yee et al. 2007). Concentrations in the sediments generally paralleled this, with higher concentrations in the South Bay, Suisun Bay and San Pablo Bay, and lower concentrations the Central Bay and the Delta — that is, with the higher concentrations in those parts of the system that receive a larger portion of their drainage from the Coast Range (Yee et al. 2007). In the northern part of the Estuary, the concentration of dissolved nickel increases downstream, presumably due to mobilization from the resuspension of these sediments (Yee et al. 2007).

In the mid-1990s, nickel concentrations in the water of the lower South Bay often exceeded water quality objectives; in contrast, concentrations measured in fish and bivalve tissues remained well below the recommended maximum tissue residue levels (Yee et al. 2007). Because of the high concentrations in water, nickel (along with copper, mercury and selenium) was determined to be impairing beneficial uses in the lower South Bay (Yee et al. 2007). As a result, this part of the Bay was included on the Clean Water Act 303(d) list of impaired water bodies in 1998 (Yee et al. 2007).

The primary sources of total nickel to the Bay as a whole are resuspended sediments (estimated at 67%) and Delta inflow (28%); for the lower South Bay, the main sources are resuspended sediments (≈80%), the inflow from local rivers (≈15%) and the effluent from wastewater treatment plants (4%) (Table 5). Total nickel is strongly correlated with suspended sediment concentrations, dissolved nickel less so (Yee et al. 2007). The primary sources of dissolved nickel in the Bay water are desorption from resuspended sediments, benthic diffusive fluxes, and wastewater discharges (Yee et al. 2007). Concentrations of total and dissolved nickel in the northern part of the Bay are higher in the wet season, reflecting the importance of the rivers as a source of nickel in that part of the Bay. Concentrations of dissolved nickel in the South Bay are higher in the dry season, reflecting the relatively greater importance of wastewater effluent, which dominates freshwater inflows in this part of the Bay during the dry season, as a source of dissolved nickel (Yee et al. 2007).

Table 5. Estimated Sources of Nickel in San Francisco Bay water. From Yee et al. (2007), citing Davis et al. (2000) for San Francisco Bay and Tetra Tech (1999) for Lower South Bay.

Source	San Francisco Bay kg/yr	San Francisco Bay % of total	Lower South Bay kg/yr	Lower South Bay % of total
Net particulate flux	975,000	67%	31,000-34,000	79-80%
Delta inflow	410,000	28%	0	0%
Non-delta tributaries	49,000	3%	6,040	14-15%
Benthic diffusion	21,600	1%	720	2%
Wastewater effluent	4,800	0.3%	1,740	4%
Atmospheric deposition	580	0.04%	30	0.07-0.08%
Total	1,460,980	100%	39,530-42,530	100%

With substantial regulatory focus on trace metals and the implementation of tertiary treatment, the overall trace metal loadings into the Bay from municipal wastewater treatment plants is estimated to have been reduced by >95% since 1960 (Squire et al. 2002). The estimated nickel loading into the lower South Bay from wastewater treatment plants has steadily declined, from around 12,000 kg/year around 1979, to 5,400 kg/year around 1989, to 1,700 kg/year around 1999 (Tetra Tech 1999, cited by Yee et al. 2007). Dissolved nickel is a higher fraction of total nickel in the South Bay than in the Central and northern Bay (Yee et al. 2007). The dissolved nickel delivered to the Bay is sometimes strongly bound up in stable molecules derived from soil or from industrial processes, though the amount of such complexing is highly variable (Yee et al. 2007).

The concentration of nickel in San Francisco Bay sediments (≈90 µg/L) is among the highest reported in U.S. estuaries (Yee et al. 2007). Concentrations did not vary significantly with depth in 2-meter deep sediment cores from the Central Bay and San Pablo Bay, indicating that historic anthropogenic changes have not caused significant changes in overall nickel inputs and confirming that natural occurrence in the watershed is the primary source of high nickel concentrations in the Bay (Hornberger et al. 1999). This is corroborated by the similarly high concentrations of nickel measured in the sediments of Tomales Bay, which has a similar surrounding geology but lacks anthropogenic sources of trace metals (Yee et al. 2007); and also by the lack of a detectable decline in sediment concentrations in the South Bay between the 1970s and 1998 despite dramatic reductions over that period in the total quantity of nickel in the wastewater discharged into the South Bay (≈86% reduction) (Hornberger et al. 2000).

There is evidence from laboratory studies and from observations of a 25% reduction in nickel concentration in the water column during a South Bay phytoplankton bloom (Luoma et al. 1998) that phytoplankton may take up significant quantities of nickel from the Bay, potentially making it available to higher trophic level organisms. Nickel concentrations in bivalve tissues were higher in the South Bay than in San Pablo Bay;

however, concentrations in both bivalves and fish throughout the Bay were far below harmful concentrations for the organisms or human consumers. Three species of bivalves used in RMP bioaccumulation studies in 1993-2003 had tissue concentrations that ranged from 0.9-113 µg/L dry weight (equivalent to about 0.2-28 µg/L wet weight).[4] Two species of fish collected by NOAA in 1984-88 had tissue concentrations that averaged below 1 µg/L dry weight (equivalent to ≈0.15 µg/L wet weight). These are all far below both the recommended maximum tissue residue levels of 220 µg/L wet weight and the concentrations (330-1460 µg/L dry weight) at which reduced survival rates have been reported in bivalves (Yee et al. 2007).

In summary, nickel concentrations in Bay sediments and water are quite high compared to concentrations in most other estuaries. Within the Bay, concentrations are highest in the South Bay, especially the Lower South Bay, followed by San Pablo and Suisun bays, and lowest in the Central Bay. Concern over these concentrations contributed to various sections of the Bay being listed as impaired on the Clean Water Act 303(d) list in the 1990s. The primary source of nickel in the Bay is sediments washed in from the Coast Range, whose rock formations are naturally high in nickel. Other important sources are Delta inflow (low concentrations but high volume), and in the South Bay, other tributaries. Wastewater inflows are a more significant contributor in the lower South Bay, but even there are responsible for only 4% of the total input. Consequently, substantial reductions in nickel inputs in wastewater discharges, on the order of 85%, have not resulted in detectable reductions in concentrations in the Bay. While there is evidence of uptake by phytoplankton, tissue concentrations in bivalves and fish have remained well below levels that would have impacts on these organisms or on people eating them. Accordingly, nickel has been de-listed as a cause of impairment in the South Bay and will be considered for de-listing elsewhere in the Bay (Krieger 2007).

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[4] The highest recorded value of 113 µg/L, from a Suisun Bay sample, is a statistical outlier that probably resulted from sediment contamination of the tissue. Typical concentrations averaged 10 µg/L dry weight (≈2.5 µg/L wet weight) or less in the different segments of the Bay (Yee et al. 2007).

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Sources and Impacts of Sediment Inputs into the Water Column of San Francisco Bay

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This paper discusses sediment budgets and changes in sediment inputs to San Francisco Bay; the impacts of increased turbidity and sedimentation on organisms; and some potential effects of reduced sediment inputs. Dredging, the disposal of dredged materials, and in-Bay mining activities are discussed in terms of their effect on sediment budgets (by removing sediment from the Bay) and their injection of sediment into the water column. Except for the material that is injected into the water column during the disposal process (forming temporary sediment plumes), the deposit of dredged materials on the bottom at disposal sites in the Bay is addressed under the Stressor "Deposit Sediments or Shell." The direct effects on habitats and organisms of removing sediment (changing bottom topography, entraining and removing organisms, etc.) are addressed under the Stressor "Remove or Disturb Sediments, Shell or Bedrock." The injection of contaminants or nutrients into the water column by these activities, and the importing, exporting, deposition, bioavailability and impacts of sediment-associated contaminants or nutrients are addressed under the Stressors "Change Contaminant Inputs" and "Change Nutrient Inputs."

Overview

Over the years, there have been a substantial number of studies in San Francisco Bay that have estimated sediment inputs and outflows, estimated erosion and sedimentation rates and associated changes in Bay bathymetry, and in some cases organized these data into sediment budgets. The estimates were made by a variety of methods and usually have substantial uncertainties associated with them, making it difficult to compare the results of different studies, and to assess the extent to which changes in the results among studies over time reflect true temporal trends rather than methodological differences. Nevertheless, the overall results point to an initial large increase in sediment inputs to the Bay from the Sacramento-San Joaquin river systems in the last half of the 19th century, followed by a long-term, continuing decline, with concomitant changes in sedimentation and erosion patterns in the Bay.

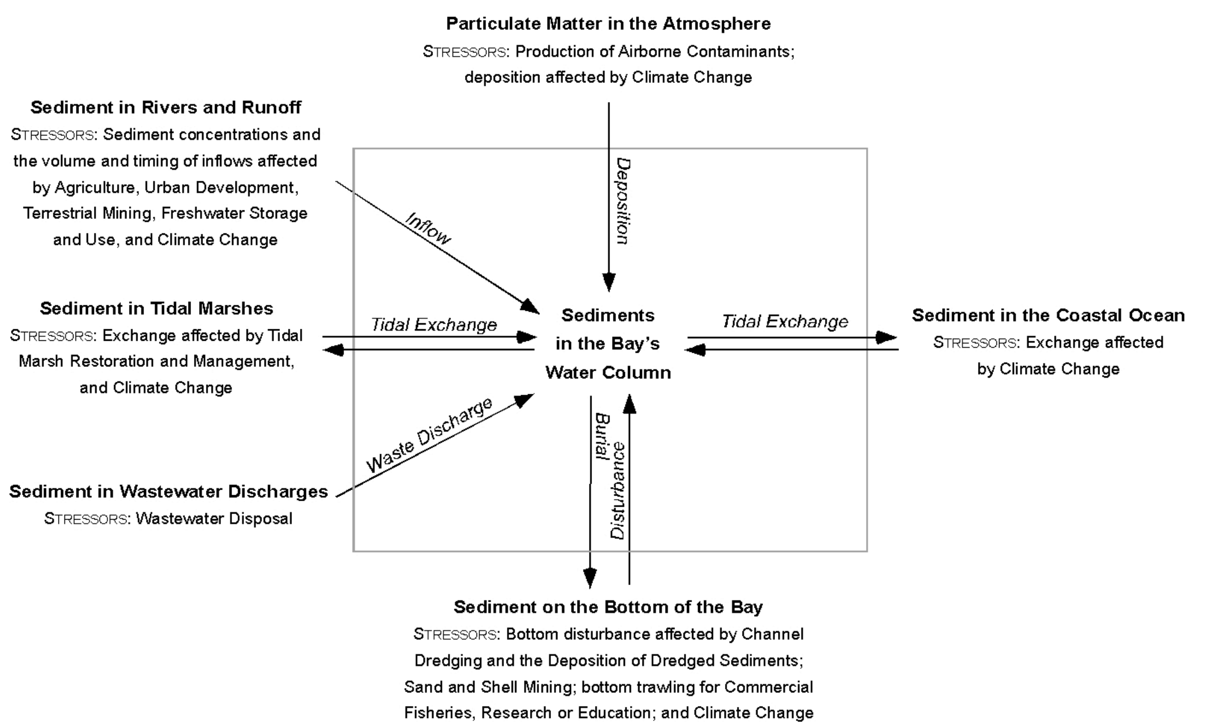
Numerous studies have documented the potential for impacts on organisms from increases in sediment concentrations in the water column, aside from any effects of sediment-associated contaminants. These impacts include clogging or damaging the gills of fish and invertebrates, especially filter feeders; repelling or attracting adult fish and changing their behavior; providing cover for prey species, and reducing predation rates of predatory species; and reducing light penetration, photosynthesis and the productivity and growth of eelgrass, seaweeds and phytoplankton (ABP Research 1999; Levine-Fricke 2004). These effects become evident only at high sediment concentrations, so the assessment of activities that inject sediment into the water column pivots on the question of whether the sediment concentrations are raised high enough, for long enough, over a large enough area to be a significant concern.

Increased inputs of sediment from external sources promotes higher deposition rates, shoaling, marsh accretion, more rapid burial of contaminants and nutrients, and an increased need for channel dredging. Conversely, reductions in sediment inputs promote the erosion of sediments, loss of shoal areas, marsh retreat, exposure and release into the water column of buried contaminants and nutrients, and a reduced need for channel dredging.

Sediment Pathways and Budgets

Sediments from various sources can be carried into the Bay with freshwater inflows in rivers or runoff, or enter the Bay directly in minor amounts as sediment in waste streams or as particulate matter deposited from the atmosphere. These latter two pathways are insignificant relative to riverine inputs, and are not treated further in this paper. Sediment settled on the bottom can be resuspended by currents or wind waves or by human activities that disturb the bottom. Tidal waters moving between the Bay and the ocean and between the Bay and its tidal marshes can carry sediment in or out of the Bay (Fig. 1).

Figure 1. Sediment Pathways and Stressors in San Francisco Bay



Sediment budgets are constructed to better understand the flows of sediment into and out of an ecosystem and the accumulation or loss of sediment from a system (Schoellhamer et al. 2005).

Studies that have contributed to our understanding of San Francisco Bay's sediment budget have typically taken the Bay and its bottom as the "system." Storage in the system is usually estimated by examining changes in the Bay's bathymetry between the beginning and end of a period, calculating the amount of sediment that would need to have been added or removed to make the changes, and dividing by the length of the period to yield an annual rate of accumulation or loss. Sediment inflows are usually estimated from data on water flows and sediment concentrations in tributary rivers. The net flux of sediment in or out of the Golden Gate is usually calculated as inflows minus anthropogenic outflows (i.e. sediment removed from the Bay by mining or dredging) minus change in storage.

San Francisco Bay Sediment Budgets

While investigating the impact of hydraulic mining debris, Gilbert (1917) constructed a sediment budget to assess the fate of material washed out of the Sierra Nevada region by mining and other activities including farming, grazing and road construction, over the period from 1849 to 1914 (Table 1). The "system" for this budget was not the Bay itself, but included both the Bay and those parts of its tributary rivers and watershed where quantities of mining debris and other sediments were deposited. The number that Gilbert used for sediment input to this system was an estimate of the volume of soil and debris washed from the lands that are tributary to the Delta. He began with prior estimates of the hydraulic mining debris produced in several parts of the Sierra Nevada. These had been made by multiplying the amount of water used to wash away the overburden, measured in miner's inches, by a "duty", the approximate amount of material removed per miner's inch. The duty varied greatly depending on "the quantity of water used, the pressure, the character of the material washed, and the grade and size of the sluices," with a reported range between 1 and 28 cubic meters of debris per inch, producing estimates of substantial uncertainty. Gilbert checked these by surveying the excavations that were left behind by mining activities at various sites in the Yuba River watershed, imagining what the original slopes had been prior to mining, surveys whose aggregate accuracy Gilbert estimated at ±10%. The estimate he produced by this method was 51% larger than the earlier estimates based on water usage made at the same sites. He then adjusted upward the corresponding estimates made over larger areas of the Sierra Nevada, extrapolated these to areas where estimates had not been made, and added estimates for other, non-hydraulic types of mining (placer mining, quartz mining and drifting), and ended up with an overall estimate of mining debris that was "nearly eight times as great as the volume moved in making the Panama Canal."

Table 1. Sediment Budget for 1849-1914 from Gilbert 1917

Source (+) or Fate (-) of Sediments	10^6 m^3	10^6 m^3/yr	Method of Estimation
Wasted from the land surface tributary to Suisun Bay	1,816	27.5	Volume of mining debris calculated from amount of water used in hydraulic mining and surveys of mining excavations, along with estimates of erosion from agriculture, roads and trails, overgrazing and the natural degradation of the land surface.
Deposited in the Sierra Nevada, the piedmont or the channels of valley rivers	-677	-10.3	Estimated from surveys of deposits and various extrapolations.
Deposited in the Bay	-876	-13.3	Estimated from changes in bathymetry between USC&GS charts.
Outflow to the ocean	-38	-0.6	Estimated "arbitrarily".
Deposited on "inundated lands"	-225	-3.4	Remainder of above.

consisting of Central Valley flood basins and Bay and Delta marshes			

To this estimate Gilbert added what were possibly rougher estimates for loss of soil from farming (multiplying estimates of farm area by 2 inches lost for active farms and 4 inches lost for abandoned farms), from road and trail construction (multiplying the total length of mapped public roads by an average width of 10 feet and a loss of one foot of depth with 85% of this reaching the streams, and adding allotments for unmapped and private roads, abandoned roads, and trails), from overgrazing and from natural erosion, which altogether added nearly 42% to his mining debris estimate. From this total he subtracted estimates of the volume of debris lodged in mining dumps, in canyons and in and along stream and river courses in the mountains and in the piedmont lands immediately below them, and in the beds of the Sacramento, Feather and San Joaquin Rivers in the Central Valley, estimates made as above by a combination of measurements, extrapolations and educated guesses, which totaled 37% of the sediment input. He made no direct estimate of the amount of sediment captured in the lateral flood basins of the Central Valley rivers or in the Delta marshes, which "would be difficult to measure" (leaving these quantities to be included in the balance of the budget equation, the deposits on "inundated lands"), and so nowhere does he actually provide an estimate of the amount of sediment entering the Bay through the Delta. Nor does he estimate or include in his budget the sediment contributed by the local creeks and rivers draining into San Francisco Bay.

Gilbert estimated the volume of sediment deposited in the Bay by comparing the water depths on successive charts produced by the U.S. Coast and Geodetic Survey (USC&GS), which spanned periods of 20 to 41 years in different parts of the Bay, and extrapolating the results to the 66-year period of his analysis. The total amounted to 48% of the sediment input. He made no mention of correcting for sea level rise, noted significant inaccuracies in the methods of the earliest surveys and raised questions about the plane of reference used, and noted that the precision of the surveys was generally inadequate for calculating changes of contour in irregular channels, where he instead made rough guesses. While his overall approach to determining changes in sediment deposition was similar to that used in later studies, the specific methods of analysis and the precision of surveys have improved.

Gilbert's estimate of the amount of sediment carried out through the Golden Gate was, as he stated, "necessarily arbitrary," based largely on the observation that "the outflowing stream is distinguished from the water it invades by a yellowish tinge." This estimate accounted for just over 2% of the sediment input. Subtracting the sediment deposited in the mountains, along and in the rivers and in the Bay, and sediment carried out to sea, from his estimate of the debris produced by mining and the soil washed from the land, resulted in a quantity that Gilbert called deposits on inundated lands, including in these the Central Valley flood basins and the Bay and Delta marshes. These account for the remaining 12% of the sediment input.

Unlike later sediment budgets for the Bay, Gilbert estimated sediment inputs as the volume of mining debris and sediment washed from the land, rather than as an estimate based on the concentration and mass of sediment carried by the rivers. Thus he was able to make all of his initial measurements and estimates in volume units with no need to convert between mass of sediment and the volume of sediment deposits (which is a critical step in the budgets discussed below). However, he made no mention of and no correction for the differing densities of different types of deposits. This could be a significant oversight because, for example, the rock and dirt that occupied a cubic meter of space before it was excavated or water-blasted from a hillside, may occupy a significantly larger volume when it is deposited in a mining dump, lodged in a canyon, or accumulated in a river bed or on the bottom of the Bay, and these different types of deposited material may themselves differ considerably in the amount of sediment material contained in a given volume due to variations in particle size and shape, entrained organic material, degree of compaction, etc. For example, if a given amount of material occupied on average 50% more volume when deposited in the watershed or Bay than it did when it was part of the original undisturbed sediment and rock, then the 1,816 million cubic meters of sediment input in Gilbert's budget would have produced 2,720 million cubic meters of deposited sediment, and the deposits on inundated lands, calculated as the remainder from the budget, would have been 1,159 rather than 225 million cubic meters, the latter then being in error by 81%.

Smith (1965) did not explicitly construct a sediment budget, but provided most of the estimates needed to assemble a rough one for the Estuary (Bay plus Delta) or the Bay (Table 2). Estimates of sediment inputs were derived from measurements of suspended sediment concentrations in tributary waters over a relatively short period (1957-59). These were used to determine the relationships between water discharge and sediment discharge in different water courses,[5] which were then used to estimate sediment discharge over a longer period (1909-1959) using long-term estimates of water discharge. In this case, the water flows were modified by assuming 1960 levels of water withdrawals, so it's really an estimate for 1960 water system conditions with water inputs assumed to be those of the preceding 50 years. Suspended sediment estimates were then adjusted to include bed load, estimated by a combination of measurements and modeling. This work was conducted by Porterfield et al. (1961), producing estimates of the average annual mass of sediments carried into the Bay and Delta by their tributary rivers.

Table 2. Partial San Francisco Estuary Sediment Budget from data in Smith 1965

Source (+) or Fate (-) of Sediments	10^6 MT/yr	10^6 m^3/yr	Method of Estimation
Inflow to Delta	4.57	5.38	Based on Porterfield et al.'s (1961) suspended sediment measurements of 1957-59 and estimates of total sediment, adjusted to 1909-59 water flows with 1960-level withdrawals. Converted to a volume of Bay and Delta sediment deposits by assuming a bulk dry density of 0.801 MT/m^3 for suspended load and 1.44 MT/m^3 for bed load.
Inflow to Bay from local rivers	0.76	0.91	As above.
Removed with Delta water withdrawals	-0.20	-0.23	Assumes water withdrawals at 1960 levels of 4,500 cfs, carrying suspended sediment only.
Deposited in the Delta		-1.2	Assumes average maintenance dredging equals 85% of deposition, the minor sloughs not being dredged.
Deposited in the Bay		-4.6	Based on calculations of bathymetric change from comparisons of successive USC&GS charts, extrapolated to the 1855-1956 period

Smith converted these mass estimates into volume estimates by assuming a bulk dry density when deposited of 0.801 MT/m^3 (=50 lb/ft^3) for suspended load and 1.44 MT/m^3 (=90 lb/ft^3) for bed load. (The appropriateness of these conversion factors will be discussed below with those used by other studies.) He estimated the amount of sediment removed from the Delta in water withdrawals based on the relative volume of water withdrawn, and the amount of sediment deposited in the Delta based on maintenance dredging records. Subtracting these from his

5 For example, the sediment discharge-water discharge relationship determined for the Sacramento River is shown in graph form in Smith's (1965) Fig. 4.

estimated input to the Delta of 5.4 million cubic meters/year, yields an estimated input to the Bay from the Delta of 3.9 million cubic meters/year; adding the input from local rivers produces an overall estimate of sediment input to the Bay of 4.8 million cubic meters/year.

Like Gilbert (1917), Smith estimated the size of deposits in the Bay by comparing the charts produced by a series of bathymetric surveys conducted in different parts of the Bay, determining the average rates of sediment deposition or erosion between surveys, and extrapolating to a common time period for the entire Bay. His resulting estimate, an average rate of deposition of 4.6 million cubic meters/year from 1855 to 1956, is close to his estimate of sediment inputs to the Bay, suggesting that the summed losses from the Bay (e.g. from sand mining, ocean export, and net deposition on tidal marshes) should be small. However, several cautions are in order. Since the large sediment inputs and presumably large sediment deposits of the hydraulic mining era occurred during the early part of the 1855-1956 period, the rate of sediment deposition in the later part of this period — corresponding to Smith's sediment input estimates based on 1960 water system conditions and 1909-59 water flows — should be significantly below the average rate. On the other hand, though Smith considered USGS data on ground subsidence in Santa Clara County and found the impact on sediment deposition rates to be minor and not worth including in his estimates, he apparently did not consider the impact of sea level rise, which could significantly raise the estimates. In addition, the several other sources of uncertainty in these estimates should be kept in mind.

Krone (1979) described a generally similar sediment budget for the Bay (Table 3), though some of the quantities were incorrectly shown in the illustration at the end of his paper, which led to erroneous citings of these quantities by later authors (including himself).[6] Like Smith (1965), Krone used Porterfield et al.'s (1961) mass estimates of suspended sediment loads from local rivers derived from 1957-59 suspended sediment concentration measurements, but did not adjust them to a longer period of water discharge data. To estimate suspended sediment inputs into the Delta and from the Delta into the Bay, Krone used a longer period of suspended sediment concentration measurements (1957-65) and a later 50-year period of water flows (1921-71) than Smith (1965). To estimate total sediment loads he added bed load equal to 0.065 of the total load by weight. He then converted these to volume estimates by assuming a bulk dry density when deposited of 0.529 MT/m^3 (=33 lb/ft^3).

Table 3. San Francisco Bay Sediment Budget for 1960 from Krone 1979

Source (+) or Fate (-) of Sediments	10^6 MT/yr	10^6 m^3/yr	Method of Estimation
Inflow to Bay from Delta	3.25	6.1	Estimated from the relationship of measured suspended sediment to river discharge for 1957-65, and 1921-71 water flows adjusted to 1960 water system facilities and withdrawals, with bed load assumed to be 0.065 of the total load. Converted to a volume of Bay sediment deposits by assuming a bulk dry density of 0.529 MT/m^3.
Inflow to Bay from local rivers	1.0	1.9	Based on Porterfield et al.'s (1961) suspended sediment measurements of 1957-59, not adjusted to a longer flow period, with bed load and volume conversion as above.

6 Krone's (1979) Figure 6 shows the average annual sediment deposition in the Bay as "New Annual Deposit 5.5 M", where M = million cubic yards. However, the figure inexplicably leaves out sediment erosion from the South Bay of 0.91 million cubic yards, for a net annual deposit of ≈4.6 million cubic yards (=3.5 million cubic meters). The figure also confusingly reports "4.0 M net outflow to ocean plus 0.9 M from erosion of So. SF. Bay," when the "net outflow to ocean" is actually 4.9 million cubic yards (=3.7 million cubic meters). This was apparently sufficient to confuse later authors, including Krone himself, who in 1996 summarized the earlier paper as finding "a total of...about 5.5 Mcy accumulated in the bays, and 4.0 Mcy exited the Golden Gate" annually.

Upland disposal of dredge sediments		-0.76	Not stated.
Deposited in the Bay		-3.5	Based on interpolations to 1923-50 of Smith's (1965) calculations of bathymetric change based on USC&GS charts, corrected for sea level rise.
Outflow to the ocean		-3.75	Remainder of above.

Krone's (1979) sediment budget figure shows annual land disposal of about 750,000 cubic meters of dredged sediments, with in-Bay disposal (which doesn't affect the sediment budget) being seven times that; the source of these numbers is not explained, but they are presumably derived from dredging records. Deposition rates for 1923-1950 are extrapolated from Smith's (1965) calculations based on USC&GS charts, and adjusted for sea level rise of 2 millimeters per year, as measured over the long-term (1860-1970) at the Golden Gate. Net sediment lost to the ocean was calculated as the balance after upland disposal and Bay deposition were subtracted from the sediment inputs.

Ogden Beeman Associates produced a sediment budget for the period 1955-1990 for the Long Term Management Strategy (LTMS) project (Ogden Beeman 1992). Estimates of the sediment input to the Delta from the Central Valley were based on sediment concentration measurements made on the Sacramento and San Joaquin rivers for periods between 1957 and 1988, adjusted to Delta inflows for the 1955-1990 period. Sediment input from the Delta to the Bay was then estimated by assuming that there was no deposition within the Delta, and that the sediment inflow was apportioned between Delta withdrawals and Delta outflows according to the size of these flows. Sediment input to the Bay from local rivers was taken from Porterfield's (1980) estimate of average sediment inflows for 1909-1966. Porterfield (1980) used suspended sediment concentrations measured during 1957-1967 and adjusted to water flows for 1909-1966, plus various models to calculate bed load. As did Krone (1979), Ogden Beeman converted the sediment mass estimates for both Delta and local inputs to volume estimates by assuming a bulk dry density when deposited of 0.529 MT/m^3 (=33 lb/ft^3).

Table 4. San Francisco Bay Sediment Budget for 1955-1990 from Ogden Beeman 1992

Source (+) or Fate (-) of Sediments	10^6 MT/yr	10^6 m^3/yr	Method of Estimation
Inflow to Bay from Delta	2.4	4.5	Estimated from the relationship between river discharge and daily measurements of suspended sediment at Sacramento (1957-66 and 1980-88) and Vernalis (1957-88), and river and Delta outflow records for 1955-90, assuming no deposition in the Delta (and ignoring bed load?). Converted to volume by assuming a bulk dry density of 0.529 MT/m^3.
Inflow to Bay from local rivers	0.81	1.5	Based on Porterfield's (1980) estimate of sediment production for 1909-66, with volume conversion as above.
Upland disposal of dredge sediments		-0.31	Based on dredging records.
Deposited in the Bay		-3.1	Based on National Ocean Service bathymetric surveys conducted around 1955 and 1990, corrected for South Bay subsidence and sea level rise.
Outflow to the ocean		-2.6	Remainder of above.

Ogden Beeman compiled dredging records from the U.S. Army Corps of Engineers and the U.S. Navy, and calculated the average rate of dredging during 1955-1990 at 4.5 million cubic meters/year, with upland disposal of about 300,000 cubic meters/year. Estimates of sediment deposition in the Bay were based on National Ocean Service (NOS, formerly the U.S. Coast and Geodetic Survey) charts, with corrections for sea level (estimated at 50 mm (0.16 feet) of rise over the 35-year period)[7] and for subsidence in the southern part of the South Bay.[8] Ogden Beeman, arguing that the Bay's tidal marshes are probably not accumulating sediment fast enough to keep up with sea level rise, estimated an upper bound on the amount of sediment deposited in these marshes of 130,000 cubic meters/year, which they considered too small to be worth including in the sediment budget.[9] As in Krone (1979), sediment lost to the ocean was calculated as the balance after dredge disposal on land and sediment deposition were subtracted from the sediment inflows.

Schoellhamer et al. (2005) constructed a different sediment budget for 1955-1990, and produced two budgets for the 1995-2002 period, one using all the years' data and one that deleted two years with very high water flows to produce a "normal water year" estimate for the period (Table 5). Sediment inflow to the Bay was estimated from new measurements of suspended sediment concentrations and estimates of suspended sediment loads for 1995-2003 made at Mallard Island at the head of the Bay (McKee et al. 2002, 2006), and adjusted for Delta outflows for the relevant periods. Schoellhamer et al. (2005) do not state whether bed load was accounted for in these estimates. Sediment input from local rivers was based on Porterfield's (1980) estimate of total sediment loads for 1909-1966.

Table 5. San Francisco Bay Sediment Budgets for 1955-1990, 1995-2002, and 1995-2002 normal water years, from Schoellhamer et al. 2005

Source (+) or Fate (-) of Sediments	10^6 MT/yr 1955-1990	1995-2002	1995-2002 normal WY	Method of Estimation
Inflow to Bay from Delta	1.1	1.3	0.8	Estimated from the relationship between river discharge and suspended sediment measurements at Mallard Island in 1995-2003, and Delta outflow records for 1955-90.
Inflow to Bay from local rivers	0.81	1.5	0.9	Based on Porterfield's (1980) estimate of sediment production for 1909-66.
Eroded from the Bay bottom	1.4	1.8	2.4	For 1955-90, extrapolated from USGS and Ogden Beeman analyses of bathymetric changes. Though not stated, probably converted to mass by assuming a bulk dry density of 0.529 MT/m^3. For 1995-2002, based on a model of sediment movement.
Inflow of sand from ocean	2.9	2.9	2.9	Derived from the change in bathymetry in the Central Bay plus the quantity removed by sand mining. Mass conversion probably as above.
Upland disposal of dredge sediments	-0.1	-1.3	-1.0	Based on USACE dredging records.
Sand mining	-0.88	-1.8	-1.8	Estimated from Hanson et al. 2004.
Deposited in tidal marshes	-0.19	-0.19	-0.2	Estimated by assuming marshes have maintained their elevations relative to sea level rise.
Suspended sediment outflow to the ocean	-5.0	-4.2	-4.0	Remainder of above.

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7 This is in fact the change in chart datums between the 1955 and 1990 charts, which were based on two different tidal epochs. The actual measured sea level rise at the Golden Gate from 1955-1990 was nearly 100 millimeters (0.32 feet). The correction is significant; applied over the area of the Bay it amounts to a change of 0.55 million cubic meters/year, or about 18% of the sediment deposition estimated in this study.

8 The subsidence correction is also significant, amounting to 0.71 million cubic meters/year, or about 23% of the sediment deposition estimated in this study.

9 This estimate was based on sea level rise over the 35-year period of 0.16 feet, but elsewhere (Appendix F) they note that the actual measured rise over this period was 0.32 feet.

Estimates of changes in sediment storage in the Bay for the 1955-1990 budget were taken from Ogden Beeman (1992) for the Central Bay, and based on recent re-analyses by USGS scientists of the USC&GS and NOS charts for the rest of the Bay (Suisun Bay: Capiella et al. 1999; San Pablo Bay: Jaffe et al. 1998; South Bay: Foxgrover et al. 2004). There are no bathymetry charts that are recent enough to allow an estimate of the change in sediment storage in the Bay between 1995 and 2002. Instead Schoellhamer et al. (2005) used a salinity model of the Bay that had been modified to incorporate sediment transport, deposition and erosion (Lionberger 2003). Schoellhamer et al. argued that there is a large flow of sand from the ocean into the Bay along the bottom of the Golden Gate. They estimated this inflow as the sediment accumulated landward of the Golden Gate as revealed by the analysis of bathymetric change in the Central Bay (Ogden Beeman 1992), plus the quantity removed by sand mining. Dredge sediments disposed on land and sediments removed by sand mining were estimated from available records. Sediment deposited on tidal marshes was estimated by assuming that the rate of accumulation was sufficient to keep pace with sea level rise, estimated at 2.17 millimeters per year. Sediment export to the ocean was calculated as inflows (from the Delta and local rivers, and sand from the ocean) minus other outflows (upland dredge disposal, sand mining, net export to marshes) plus change in storage. This calculation was done in mass units, so quantities initially estimated as volumes — the change in sediment storage in the Bay, the influx of ocean sand at the Golden Gate, the sediment deposited on tidal marshes, and possibly the material removed by upland dredge disposal and sand mining — had to be converted to mass units. Schoellhamer et al. (2005) do not say what sediment densities were used for these conversions.

There are some striking differences between the two sediment budgets constructed for 1955-1990 (Tables 4 and 5). Schoellhamer et al. (2005) estimated annual sediment inflow from the Delta at 1.1 million metric tons, which is less than half of Ogden Beeman's (1992) estimate of 2.4 million metric tons. The general approach used was similar: establish the relationship between sediment concentrations and water flows by measuring suspended sediment concentrations over a short period, and use this to estimate sediment flows based on water flow data over a longer period. The period and locations for the sediment concentration measurements differed between the two studies so some difference may be expected, but the size of the difference is surprising. Other large differences — in the change in stored sediment and in sediment exported to the ocean — are partly explained by Schoellhamer et al.'s treatment of ocean-derived sand that accumulated in the Central Bay as a distinct category rather than as a change in stored sediment.

The Role of Sediment Density

All studies since the 1960s have estimated sediment inflows to the Bay based on measurements of suspended sediment concentrations in the rivers, with the inflows calculated initially on a mass basis; and have estimated sedimentation or erosion in the subtidal Bay and deposition in tidal marshes based on changes in bathymetry and surface level, which are calculated initially on a volume basis. Constructing a sediment budget thus requires conversion from one of these units to the other. A change in the volume of bottom sediment has three components, sediment derived from outside the system, organic material derived from within the Bay or marsh (that is, material derived from phytoplankton, benthic algae, eelgrass or marsh plants), and pore space that is filled with water or air depending on the sediment's position relative to the tide. The desired conversion factor relates the mass of the externally-derived sediment to the volume it and its associated pore space would occupy if there were no organic material derived from within the Bay or marsh. At least in non-marsh sediments, the amount of organic matter is usually small and can be ignored, so the conversion factor is approximately the dry bulk density of *in situ* sediment.

Table 6 shows densities that have been used as volume-to-mass conversion factors for sediment quantities in different San Francisco Bay studies; and Table 7 shows dry bulk densities that have been measured or estimated for Bay sediments. The values reported for Bay sediments range from 0.400-1.409 MT/m^3, while the values used in Bay studies as conversion factors range from 0.529 to 0.852 MT/m^3. The use of one or another conversion factor can lead to wildly different values derived from the same data and analysis. For example, Ogden Beeman (1992) estimated the average annual sediment flow from the Delta into Suisun Bay in 1950-1990 to be 2.62 million tons (=2.4 million MT), and used a conversion factor of 33 lb/ft^3 (=0.529 MT/m^3) to report this in volume terms as 5.88 million yd^3 (Ogden Beeman 1992, Table 5) (=4.5 million m^3). However, Jaffe et al. (2007) used a conversion factor of 0.85 MT/m^3 to report Ogden Beeman's estimate in volume terms as 2.79 million m^3, which is less than 2/3 of the volume that Ogden Beeman had reported. Used in a volume-based comparison of sediment flows or in a sediment budget, these two numbers — 4.5 million m^3 and 2.8 million m^3 — which are essentially different translations of a single analysis of sediment flow, would produce very different results.

Table 6. Densities used for Volume-to-Mass Conversion

Purpose	lb/ft^3	MT/m^3	Source
For a sediment budget	33	0.529	Krone 1979, Ogden Beeman 1992
To compare estimates of sediment flow	33	0.529	McKee et al. 2002, 2006
To compare potential sediment accumulation to sea level rise	33	0.529	Van Geen & Luoma 1999
For model calibration	33	0.529	Ganju et al. 2008
For marsh soils in a sediment budget	35	0.561	Ogden Beeman 1992
For suspended load in sediment budget	50	0.801	Porterfield et al. 1961
To convert and compare sediment flows	52.3	0.852	Porterfield 1980, Jaffe et al. 2007
For bed load in a sediment budget	90	1.443	Porterfield et al. 1961

Table 7. Dry Bulk Densities Reported for Bay Sediments

Material	ib/ft^3	MT/m^3	Source
in situ SF Bay bottom sediment	25-50	0.400-0.801	Ogden Beeman 1992
bottom sediment in San Pablo Bay	31	0.496	Smith 1965] reporting USACE analysis of dredge spoil samples
mineral portion of South SF Bay marsh soils	35	0.561	Ogden Beeman 1992citing Krone 1987, using Pestrong 1972's data
bottom sediment in South Bay	45	0.721	Smith 1965] reporting USACE analysis of dredge spoil samples
sediment delivered within the Bay	45.05	0.722	Smith 1965[a] based on model
area-weighted mean of Bay bottom sediment	47.42	0.759	Based on Smith 1965] data
sediment delivered to the Bay/Delta system	48.84	0.782	Smith 1965[a] based on model
sediment delivered to the Bay/Delta system	49-50	0.785-0.801	Smith 1965] reporting USGS and DWR estimates based on model
bottom sediment in Central Bay	49	0.785	Smith 1965] reporting USACE analysis of dredge spoil samples
bottom sediment in Suisun Bay	88	1.409	Smith 1965] reporting USACE analysis of dredge spoil samples

[a] Derived from measurements of particle sizes carried by tributaries in 1959-60 ("delivered to the Bay/Delta system"), or in suspended sediment within the Bay ("delivered within the Bay") using a model for density of submerged sediments in a reservoir, with no compaction time. Calculated densities tend to decrease downstream through the system as larger particles (sand) are ground down to smaller ones (silt and clay). However, the calculated densities also increase significantly with time, due to compaction, as follows (in lb/ft^3):

	delivered to Bay/Delta	within Bay
initially	48.84	45.05*
after 5 years	57.25	54.05
after 10 years	60.10	57.10
after 20 years	63.13	60.35

* Note: Due to a math error, Smith 1965 incorrectly reported this as 47.05 lb/ft^3.

The inconsistent use of these conversion factors can also lead to results that are not just inconsistent, but clearly erroneous. For example, Porterfield (1980) estimated the average sediment flow into the Bay and Delta in 1909-1966 to be 16,993 tons/day (Porterfield 1980, Table 30) (=5.6 million MT/yr); and used a conversion factor of 53.2 lb/ft^3 (=0.851 MT/m^3) to report this as 23,598 yd^3/day (Porterfield 1980, Table 31) (=6.6 million m^3/yr). Ganju et al. (2008) then used a *different* conversion factor of 0.529 MT/m^3 to convert this back into a mass estimate of 3.48 million MT/yr (Ganju et al. 2008 at p. 520). However, this is 2.1 million MT/yr less than Porterfield's original mass estimate and is clearly incorrect, a result of converting with one density value and then back-converting with another. Ganju et al. (2008) went on to use this erroneous value for sediment flows to calibrate a historic time series of daily sediment loads, which is intended for use in modeling simulations.

Density values reported for sediments in different parts of the Bay show a wide range, from about 0.5 MT/m^3 in San Pablo Bay to nearly three times that, 1.4 MT/m^3, in Suisun Bay (Table 7). Sediment densities in San Francisco Bay marshes may be lower than these numbers, though perhaps not as low as in more densely-vegetated Atlantic and Gulf Coast marshes: Greenbaum and Giblin (2000) reported sediment densities of 0.22-0.37 MT/m^3 in a *Spartina patens* marsh in Massachusetts, and Wheelock (2003) reported densities of 0.06-0.21 MT/m^3 in a Louisiana *S. patens* marsh. On the other hand, sand deposits in the Central Bay may exhibit higher sediment densities. The area-weighted mean of the average results for the main embayments in the Bay, based on U.S. Army Corps analysis of dredge spoil samples, is 0.76 MT/m^3,[10] and a set of modeling studies in the 1960s produced similar sediment density estimates that ranged from about 0.72 to 0.80 MT/m^3.[11] However, the mean sediment density value used in most Bay studies has been 0.53 MT/m^3.[12] Thus, the frequent use of this possibly low sediment density value to represent average bay sediments may have resulted in considerable underestimates of the mass of sediment accumulated in the Bay, as well as possible overestimates of the amount of sediment deposited in tidal marshes and

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[10] Of course, the spoil samples may not fairly represent the distribution of sediment densities in the embayments. If the spoils were primarily taken from marina or back harbor sites where finer than average material accumulates, the use of spoils could produce an underestimate of mean sediment density; if primarily taken from channels with flowing water or tidal currents, where the bottom is of coarser than average material, they could produce an overestimate of mean density.

[11] These modeling studies produced estimates of the density of newly-deposited sediment. As deposits age, they compact and grow denser as shown in Footnote a in Table 7, with typical density increases of 15-20% in the first five years. Thus these studies probably underestimate mean sediment density.

[12] It's not clear where this number came from or what mensurative support it has. Its relatively wide use by researchers was probably initiated by Krone's 1979 chapter in the AAAS volume on San Francisco Bay. Ogden Beeman (1992) reported that it was "proposed by Schultz as a representative figure for the system. Actual values vary from around 25 pounds per cubic foot in areas of rapid deposition, to over 50 pounds per cubic foot at the mouth of Carquinez Strait."

underestimates of the mass of sand (which typically has a higher density than sediments containing clay or silt—Smith 1965) that is removed by sand mining or carried into the Bay through the Golden Gate.

Sediment Flows and Storage

A simple sediment model for the Bay below the high tide line exclusive of tidal marshes (the "subtidal Bay" as defined in these papers) is shown in Figure 2, and estimates of the quantities in the mode as given by various studies (with inflows reported both as suspended load and total load) are compiled in Table 8. Inspection of Table 8 reveals numerous inconsistencies in these data, including different numbers reported for the same flow or storage over the same time period, and authors citing numbers from earlier papers that are in fact different from or absent from the earlier papers. Some of this is explained by the use of different density conversion factors that may not been selected with care, some by failing to convert properly between short tons and metric tons or between cubic yards and cubic meters, and some by typographic errors, but the largest share is probably due to authors incorrectly reporting or using numbers that were produced by earlier authors, and to not clearly explaining the derivation and significance of the numbers they use. Among the most common errors are not properly distinguishing between numbers for the following: suspended load vs. total load; inflow to the Bay vs. inflow to the Estuary (the Bay and Delta combined); Delta inflow vs. Delta outflow; Delta inflow vs. total inflow to the Estuary; and Delta outflow vs. total inflow to the Bay. As discussed further below, the estimates from Gilbert's (1917) study are almost always misrepresented when cited by later authors.

Figure 2. A Simple Sediment Budget Model for San Francisco Bay (Bay defined as exclusive of tidal marshes)

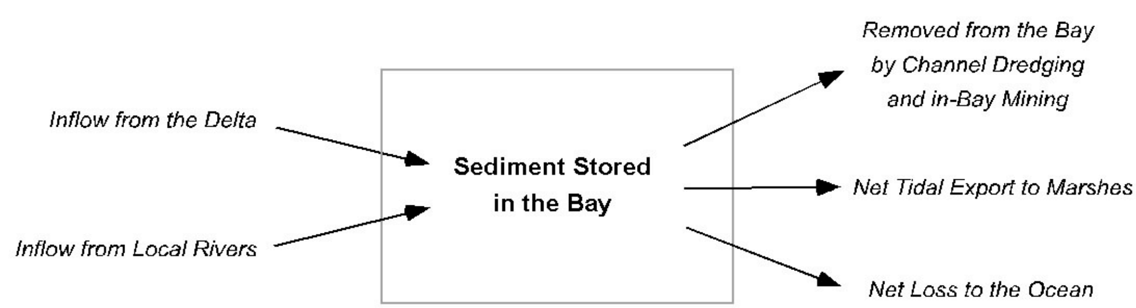


Table 8. Estimates of Sediment Inflows, Outflows and Storage Changes in San Francisco Bay
In mass or volume units, as given by the listed sources. In some publications it is not entirely clear what some of the quantities refer to, for example whether a number for sediment inflow refers to inflow to the Delta or flow from the Delta to the Bay, or whether it includes the inflow from local rivers, or whether it refers to suspended sediment or to total sediment (suspended sediment plus bed load). These have been interpreted based on the context. As can be seen, quantities appear to frequently misquoted. If the original source of an estimate is listed as a source, accurate citations of it are not. Where quantities were given in different units, these are shown in the parentheses in the Source column. Conversion factors used were 1 short ton = 0.9072 metric tons, and 1 cubic yard = 0.7646 cubic meters.

Period	10^6 MT/yr	10^6 m^3/yr	Source (quantities in original units; tons = short tons)
INFLOWS			
Inflow from the Delta - Suspended Sediment			
1909-1966	4.1		McKee et al. 2006 citing Porterfield 1980
1957-1959	3.3		Conomos & Peterson 1977 citing Porterfield et al. 1961
1960 conditions	3.0	5.7	Krone 1979 (3.35 x 10^6 tons/yr)[a]
1955-1990	2.8	5.4	McKee et al. 2002 (7.0 x 10^6 yd^3/yr) citing Ogden Beeman 1992
1955-1990	2.4		McKee et al. 2006 citing Ogden Beeman 1992
1990 prediction	1.6	3.1	Krone 1979 (1.79 x 10^6 tons/yr)[a]
1995-1998	2.1	4.0	McKee et al. 2002 (5.2 x 10^6 yd^3/yr)
1995-2003	1.2		McKee et al. 2006
2020 prediction	1.1	2.1	Krone 1979 (1.22 x 10^6 tons/yr)[a]
Inflow from the Delta - Total Sediment			
pre-1849		1.5	Gilbert 1917 (2 x 10^6 yd^3/yr)
pre-1850	0.8		Wright & Schoellhamer 2004 citing Gilbert 1917
1849-1914[b]		13.9-17.3	Gilbert 1917 (1.196-1.49 x 10^9 yd^3 in 66 yr)
1850-1914		17.5	Smith 1965 (1.49 x 10^9 yd^3 in 65 yr) citing Gilbert 1917
1852-1914		14	Van Geen & Luoma 1999 citing Gilbert 1917
1849-1914		14.1	Porterfield 1980 (18.4 x 10^6 yd^3/yr) and Schoellhamer et al. 2003 citing Gilbert 1917
peak mining yield	7.3		Wright & Schoellhamer 2004 citing Gilbert 1917
1915-64 prediction		12.2	Gilbert 1917 (800 x 10^6 yd^3 in 50 yr)
1909-1959[g]		3.9	Smith 1965 (5.133 x 10^{96} yd^3/yr)
1909-1966	3.6		Schoellhamer et al. 2003] and Wright & Schoellhamer 2004[c] citing Porterfield 1980
1931		4.4	Porterfield 1980 citing Grimm 1931 (5.75 x 10^6 yd^3/yr)[h]
1954		2.6	Porterfield 1980 citing USACE 1954 (3.36 x 10^6 yd^3/yr)[h]]
1955		3.1	Porterfield 1980 citing DWR 1955a,b (4 x 10^6 yd^3/yr)[h]]
1960 conditions	3.3	6.1	Krone 1979[a,d]
1960 conditions	4.4	8.3	Van Geen & Luoma 1999 citing Krone 1979
post-1964 prediction		6.1	Gilbert 1917 (8 x 10^6 yd^3/yr)
1955-1990[a]	2.4	4.5	Ogden Beeman 1992 (2.622 x 10^6 tons/yr, 5.88 x 10^6 yd^3/yr)
1955-1990	2.8		Schoellhamer et al. 2003] and Wright & Schoellhamer 2004[c] citing Ogden Beeman 1992
1955-1990	1.1		Schoellhamer et al. 2005
1990 prediction	1.7	3.3	Krone 1979[a,d]
1990 conditions[a]	1.6	3.0	Ogden Beeman 1992 (1.75 x 10^6 tons/yr, 3.93 x 10^6 yd^3/yr)
1995-1998		4.0	Schoellhamer et al. 2003 citing Mckee et al. 2002 (5.2 x 10^6 yd^3/yr)
1995-2001		2.8	Schoellhamer et al. 2003 (3.6 x 10^6 yd^3/yr)
1995-2002	1.3		Schoellhamer et al. 2005
1995-2002 normal[e]	0.8		Schoellhamer et al. 2005
2020 prediction	1.2	2.2	Krone 1979[a,d]
2035 prediction[a]	1.4	2.7	Ogden Beeman 1992 (1.57 x 10^6 tons/yr, 3.52 x 10^6 yd^3/yr)
"future"		1.5	Porterfield 1980 citing USACE 1954 (1.97 x 10^6 yd^3/yr)
"future"		2.3	Porterfield 1980 citing DWR 1955a,b (3 x 10^6 yd^3/yr)

Table 5 Continued. Estimates of Sediment Inflows, Outflows and Storage Changes in SF Bay

Period	10^6 MT/yr	10^6 m^3/yr	Source (quantities in original units; tons = short tons)
Inflow from Local Rivers - Suspended Sediment			
1909-1959	0.76		Porterfield 1980 (2,296 tons/d)
1909-1966	0.75		Porterfield 1980 (2,250 tons/d)
1957-1959	0.93	1.8	Smith 1965, Krone 1979 and Porterfield 1980 (2,830 tons/d) citing Porterfield et al. 1961[a]
1957-1959	1.1		Porterfield 1980 (3,297 tons/d)
1957-1966	0.81		Porterfield 1980 (2,458 tons/d)
"current"	0.75		McKee et al. 2002 (0.83 x 10^6 tons/yr) citing Krone 1979
Inflow from Local Rivers - Total Sediment			
1909-1959[g]	0.76	0.91	Smith 1965 and Porterfield 1980 (2,300 tons/d, 1.195 x 10^6 yd^3/yr)[f] citing Porterfield et al. 1961
1909-1959	0.83	0.99	Porterfield 1980 (2,514 tons/d, 3,548 yd^3/d)[i]
1909-1966	0.81	0.97	Porterfield 1980 (2,452 tons/d, 3,467 yd^3/d)[i]
1957-1959	1.0	1.9	Smith 1965, Krone 1979 and Porterfield 1980 (3,100 tons/d)[f] citing Porterfield et al. 1961[a,d]
1957-1959	1.2	1.4	Porterfield 1980 (3,560 tons/d, 5,100 yd^3/d)[i]
1957-1966	0.87	0.96	Porterfield 1980 (2,625 tons/d, 3,438 yd^3/d)[i]
1955-1990	0.81		Schoellhamer et al. 2005
?	0.75		McKee et al. 2002 citing Abu-Saba & Tang 2000
"current"	0.81		McKee et al. 2002 (0.89 x 10^6 tons/yr) citing Krone 1979
1995-2002	1.5		Schoellhamer et al. 2005
1995-2002 normal[e]	0.9		Schoellhamer et al. 2005
Total Inflow (from the Delta and Local Rivers Combined) - Suspended Sediment			
1957-1959	4.2		Conomos & Peterson 1977 citing Porterfield et al. 1961
1960 conditions	4.0	7.5	Krone 1979 (4.38 x 10^6 tons/yr)[a]
1990 prediction	2.6	4.8	Krone 1979 (2.82 x 10^6 tons/yr)[a]
2020 prediction	2.0	3.9	Krone 1979 (2.25 x 10^6 tons/yr)[a]
Total Inflow (from the Delta and Local Rivers Combined) - Total Sediment			
1849-1914	7.1	13.5	Ganju et al. 2008] citing Gilbert 1917[a]
1909-1959[g]		4.8	Smith 1965 (6.328 x 10^6 yd^3/yr)
1909-1966	3.5	6.6	Ogden Beeman 1992, McKee et al. 2002 and Ganju et al. 2008] (8.63 x 10^6 yd^3/yr) citing Porterfield 1980[a,j]
1924-1960	4.5	8.5	Ogden Beeman 1992, Krone 1979 and McKee et al. 2002, 2006 (11.1 x 10^6 yd^3/yr) citing Schultz 1965
1960 conditions	3.3	6.3	Ogden Beeman 1992 and McKee et al. 2002, 2006 (8.23 x 10^6 yd^3/yr) citing Smith 1965[j]
1960 conditions	4.2	8.0	Krone 1979 (10.5 x 10^6 yd^3/yr)[a,d]
?	4.0	7.6	Ogden Beeman 1992, McKee et al. 2002, 2006 and Levine-Fricke 2004 (10.0 x 10^6 yd^3/yr) citing USACE 1967
1955-1990[a]	3.2	6.0	Ogden Beeman 1992, Krone 1996 (3.51 x 10^6 tons/yr, 7.88 x 10^6 yd^3/yr)
1990 prediction	2.7	5.2	Krone 1979[a,d]
1990 conditions[a]	2.0	4.5	Ogden Beeman 1992, Krone 1996 (2.64 x 10^6 tons/yr, 5.93 x 10^6 yd^3/yr)
1990-2006	2.2		Ganju et al. 2008
2020 prediction	2.2	4.1	Krone 1979[a,d]
2035 prediction[a]	2.2	4.2	Ogden Beeman 1992, Krone 1996 (2.46 x 10^6 tons/yr, 5.52 x 10^6 yd^3/yr)

Table 5 Continued. Estimates of Sediment Inflows, Outflows and Storage Changes in SF Bay

Period	10^6 MT/yr	10^6 m^3/yr	Source (quantities in original units; tons = short tons)
OUTFLOWS			
Removed by Channel Dredging and in-Bay Mining			
1960 conditions		0.8	Krone 1979 (1.0 x 10^6 yd^3/yr)
1955-1990		0.36	Ogden Beeman 1992 (16.6 x 10^6 yd^3 in 35 yr)
1955-1990		0.31	Ogden Beeman 1992, Krone 1996 (0.41 x 10^6 yd^3/yr)
1955-1990	0.98		Schoellhamer et al. 2005
1995-2002	3.1		Schoellhamer et al. 2005
1995-2002 normal[e]	2.8		Schoellhamer et al. 2005
Net Tidal Export to Marshes			
1955-1990		≤0.13	Ogden Beeman 1992 (≤0.17 x 10^6 yd^3/yr)
1955-2002	0.19		Schoellhamer et al. 2005
Net Loss to Ocean			
1849-1914		0.6	Gilbert 1917 (50 x 10^6 yd^3 in 66 yr)
1915-64 prediction		0.6	Gilbert 1917 (40 x 10^6 yd^3 in 50 yr)
1924-1960		2.5	Conomos & Peterson 1977, Ogden Beeman 1992 and Levine-Fricke 2004 (30% of 11.1 x 10^6 yd^3/yr) citing Schultz 1965
?		3.2	Levine-Fricke 2004 (42% of 10 x 10^6 yd^3/yr) citing USACE 1967
1957-1959	0.25		Conomos & Peterson 1977 (6% of 4.2 x 10^6 MT/yr)
1960 conditions		3.7	Krone 1979 (4.9 x 10^6 yd^3/yr)[l]
1960 conditions		3.1	Krone 1996 (4.0 x 10^6 yd^3/yr)[l] citing Krone 1979
1960 conditions		4.0	Levine-Fricke 2004 (50% of 10.4 x 10^6 yd^3/yr) citing Krone 1979
1955-1990		2.6	Ogden Beeman 1992, Krone 1996 (3.37 x 10^6 yd^3/yr)
1955-1990	2.1		Schoellhamer et al. 2005
1995-2002	1.3		Schoellhamer et al. 2005
1995-2002 normal[e]	1.1		Schoellhamer et al. 2005
CHANGE IN STORAGE			
1849-1914		13.3	Gilbert 1917 (1.146 x 10^9 yd^3 in 66 yr)
1915-64 prediction		11.6	Gilbert 1917 (760 x 10^6 yd^3 in 50 yr)
1855-1956[k]		4.6	Smith 1965 (6.06 x 10^6 yd^3/yr)
1870-1896[k]		6.0	Krone 1979 (210.2 x 10^6 yd^3 in 27 yr)
1897-1922		3.5	Krone 1979 (4.62 x 10^6 yd^3/yr)
1923-1950		3.5	Krone 1979 (4.63 x 10^6 yd^3/yr)
1960 conditions		3.5	Krone 1979 (4.6 x 10^6 yd^3/yr)[l]
1960 conditions		4.2	Krone 1996 (5.5 x 10^6 yd^3/yr)[l] citing Krone 1979
1955-1990	–1.4		Schoellhamer et al. 2005
1955-1990		3.1	Ogden Beeman 1992, Krone 1996 (4.1 x 10^6 yd^3/yr)[m]
1995-2002	–1.8		Schoellhamer et al. 2005
1995-2002 normal[e]	–2.4		Schoellhamer et al. 2005

[a] Volume estimates based on conversion factor of 33 lb/ft^3 = 0.529 MT/m^3.

[b] The lower figure excludes the sediment deposited on tidal marshes in the Bay; the higher figure includes these plus the sediments deposited in Sacramento Valley basins and Delta marshes.

[c] Estimated from Schoellhamer et al.'s (2003) Figure 6 and Wright & Schoellhamer's (2004) Figure 2.

[d] Based on suspended sediment data and bed load = 0.065 of total.

[e] "Normal year" conditions, that is, the period with 2 unusually wet years deleted.

[f] These quantities are from Smith's (1965) Table 5 and Table 12; quantities calculated from his numbers on page 677 are different due to round-off and apparent transcription error.

[g] Estimate based on 1957-59 sediment flows adjusted to 1909-59 water flows, with 1960 levels of water withdrawals from the Delta.

[h] Might include inflows to Delta, not flows from Delta into Bay.

[i] Volume estimates apparently were intended to be based on a conversion factor of 53.2 lb/ft^3 = 0.852 MT/m^3 (Porterfield 1980 at pages 5 and 88). However, back-calculating from Porterfield's Tables 30 and 31 yields conversion factors of 51.2-53.9 lb/ft^3, and in one case (presumably a calculation error), 64.7 lb/ft^3.

[j] The original authors calculated these quantities based on sediment inflows *to* the Delta, and thus they include export from and deposition within the Delta.

[k] Based on changes in water depth with no adjustment for sea level rise, according to Krone (1979).

[l] As discussed in footnote 2 of this report, Krone's (1979) Figure 6 shows erroneous or at least confusing numbers for the annual deposition rate and the net outflow to the ocean, which led to later mis-citings by Krone and others.

[m] Includes net deposition in tidal marshes of ≤0.13 x 10^6 m^3/yr (Ogden Beeman 1992 at page 20).

[n] Volume estimates based on conversion factor of 0.85 MT/m^3 (=53.2 lb/ft^3).

Discussion of Sediment Budgets and Sediment Load Estimates

Gilbert's (1917) sediment budget is the starting point for our understanding of changes in sediment flows and sedimentation in the Bay system, and it is cited by nearly all researchers in this area, but rarely is it cited accurately. Contrary to most citations, Gilbert did not develop estimates of the sediment flow into the Delta, the Bay or the Estuary. Because of the central role of Gilbert's work in our concept of sediment flows in this system, however, it is worth taking a few minutes to understand what he did do.

Gilbert considered sediment volumes in several categories (note: the terms used here to identify these categories were not specifically used by Gilbert):

• Waste: Material removed from its original undisturbed placement in the Sierra Nevada by mining, other human activities, or natural wastage of the land surface between 1849 and 1914, which we will call Waste. Seventy percent of this, by Gilbert's estimate, was mining debris, with the rest derived from agriculture, overgrazing, road building, etc.

• Upland Deposits: That portion of Waste that had not yet reached the Delta by 1914, consisting of Mountain Deposits, Piedmont Deposits, River Bed Deposits, and Flood Basin Deposits, with the exception that deposits in the beds of rivers within the Delta were included in River Bed Deposits. The last component, Flood Basin Deposits, was not estimated by Gilbert.

• Local River Sediment: Gilbert neither mentioned nor considered this component, sediment that is carried in by local rivers and streams that are tributary to the Bay, but it is an important part of the Bay's sediment budget. Local River Sediment was probably larger during the period covered by Gilbert's work than it is today, as the local watersheds were also subject to the erosive developments of agriculture, overgrazing, road and trail building, and some mining activities, as well as urbanization, and there were fewer dams in them, but probably accounted for a smaller fraction of the sediment delivered to the Bay.

• Estuarine Deposits: That portion of Waste that could be found within the boundaries of the Estuary in 1914, consisting of Bay Deposits, Delta Marsh Deposits and Bay Marsh Deposits. Gilbert estimated Bay Deposits from changes in bathymetry shown on successive USC&GS charts, and did not estimate the volume of marsh deposits.

• Outflow to Ocean: Gilbert's estimate of this was essentially a guess unsupported by any evidence. Even today, we have no way of directly estimating this quantity, but calculate it as what remains in a sediment budget after making our best estimates of all other components.

Gilbert's sediment budget thus differs substantially in form from the budgets constructed later, and sediment inflows at the margins of the Bay or Estuary cannot be determined from the estimates that Gilbert provided. For example, sediment carried into the Bay in Delta Outflow should equal Waste minus Upland Deposits and Delta Marsh Deposits, but out of these Gilbert did not estimate Flood Basin Deposits or Delta Marsh Deposits. The sediment in Delta Outflow should also equal the sum of Bay Deposits, Bay Marsh Deposits and Outflow to Ocean minus Local River Sediment, but Gilbert did not estimate Bay Marsh Deposits or Local River sediment, and only guessed at Outflow to Ocean. At best then, we can only specify a rough upper and lower bound for the quantities that we are interested in.

In addition, the basic accuracy of Gilbert's estimates is probably much lower than it is for more modern estimates. Gilbert's estimates of the components of Waste consist of combinations, in various proportions, of rough approximations, measurements of uncertain accuracy, comparisons, extrapolations, and possibly shrewd but unverifiable guesses. The confidence intervals on these results should be quite large. Also the differences between the bulk density of undisturbed sediments in the Sierra Nevada before they were dug out by miners (typically 1-2 MT/m^3), and the bulk density of those same sediments after they've been deposited in the Bay (typically = 0.5-1.0 MT/m^3), are unaccounted for in Gilbert's budget, and if included could modify some of the residual quantities several-fold.

Gilbert's estimates of sediment deposition in different parts of the Bay, based on changes in charted bathymetry, are shown in Figure 3. The pattern he found of substantial net deposition in all parts of the Bay including the southern Bay, is contrary to the finding of later analyses based on the same chart data. These are shown in Figures 4-6, which found strong deposition in the northern part of the Bay, little deposition in the Central, and no deposition or net erosion in the South Bay. Nevertheless, the most recent estimates for the overall deposition rate in the Bay in the late 19th century are similar to Gilbert's estimates (table 9), so updating these would not greatly change his sediment budget.

Table 9. Overall Annual Deposition Rates in San Francisco Bay in ca. 1860-1890

Study	**Period**	**Deposition 10.6 m^3/yr**
Gilbert 1917	1857-1897	13.5
Smith 1965	1955-1898	10.1
Krone 1979	1870-1896	7.7
Jaffe et al. 1998, Capiella et al. 1999, Foxgrover et al. 2004. Does not include the Central Bay.	1986-1898	12.2

Figure 3. Sediment Deposited in San Francisco Bay 1856-1897, from Gilbert 1917

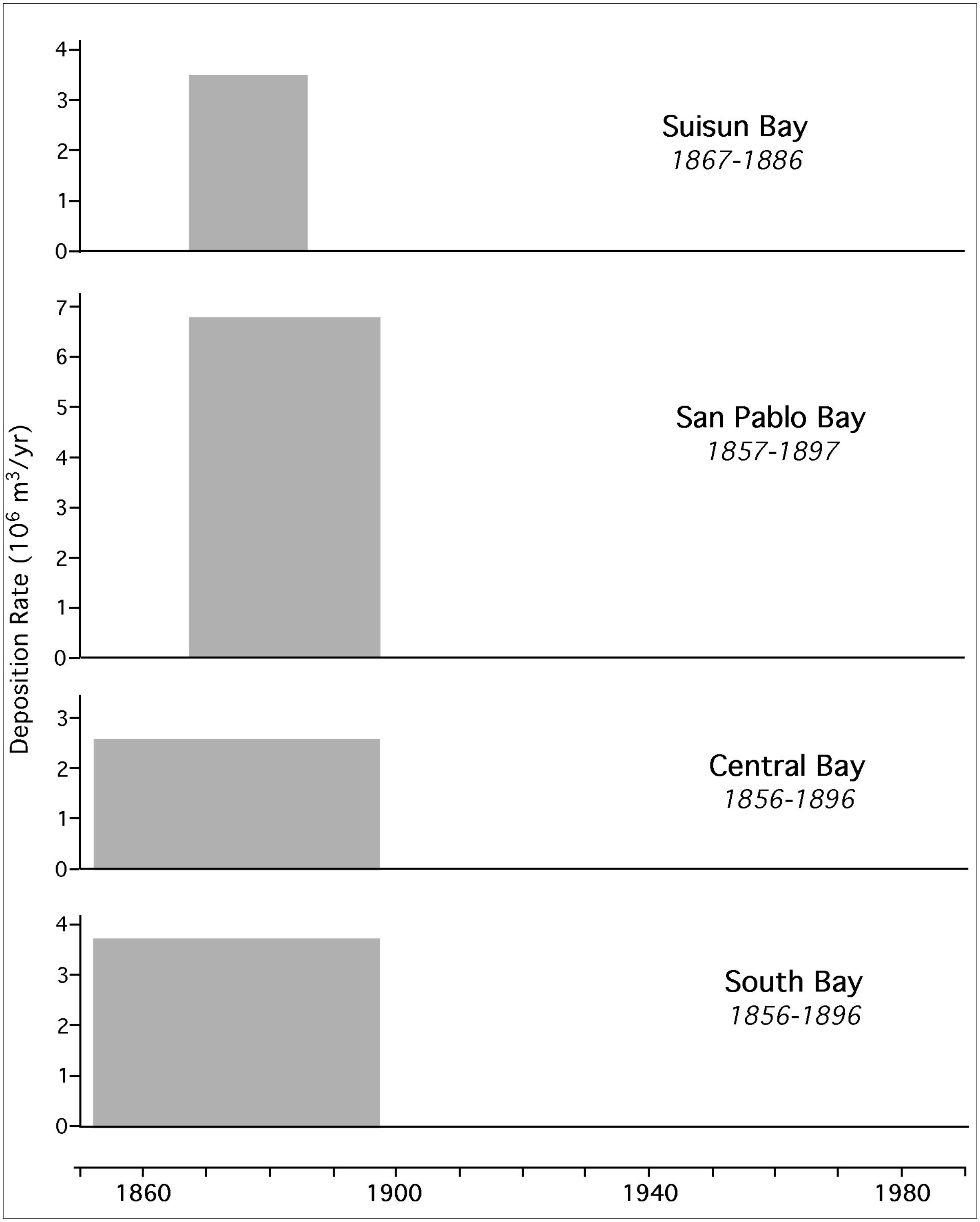


Figure 4a. Sediment Deposited in San Francisco Bay 1855-1956, from Smith 1965

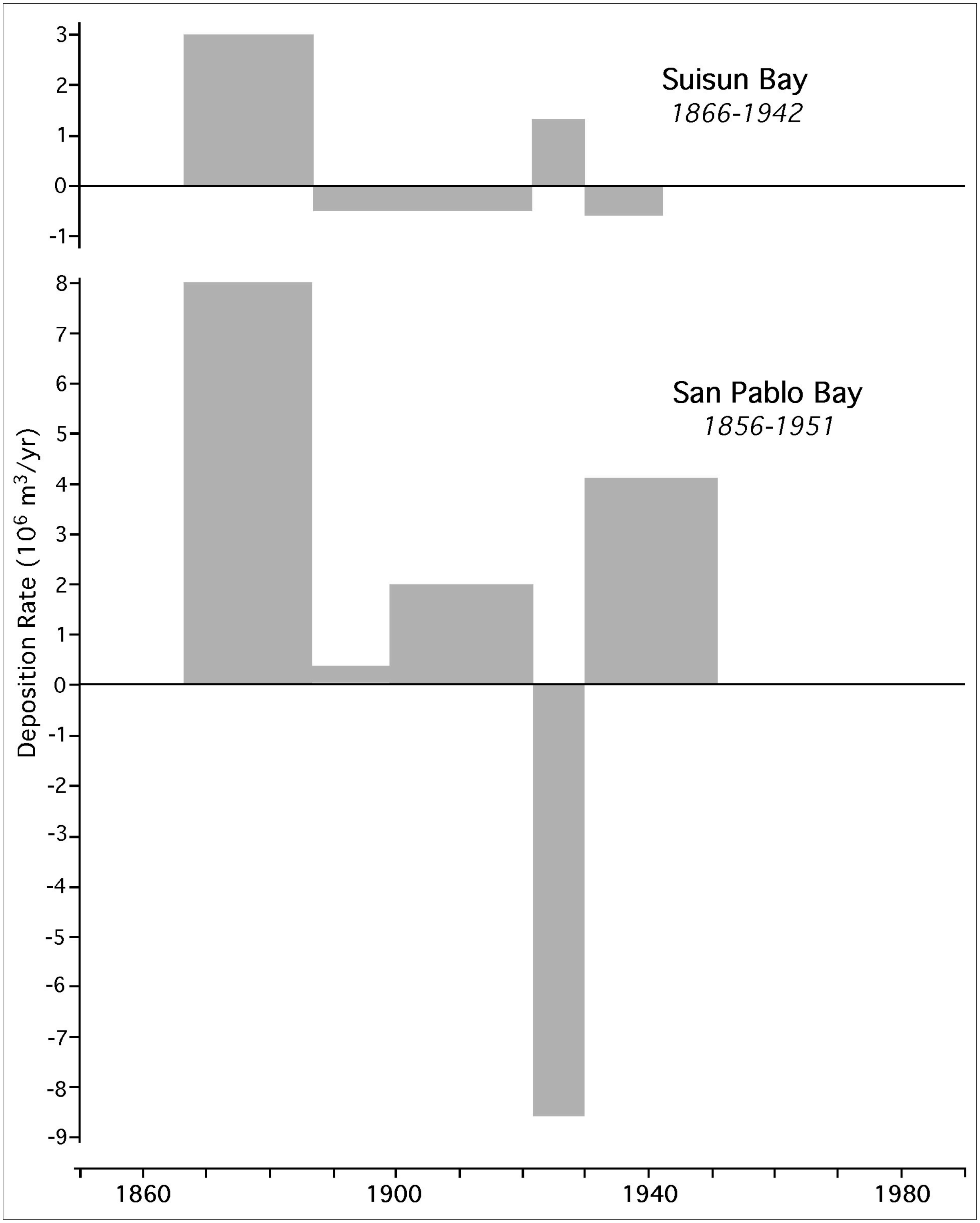


Figure 4b. Sediment Deposited in San Francisco Bay 1855-1956, from Smith 1965

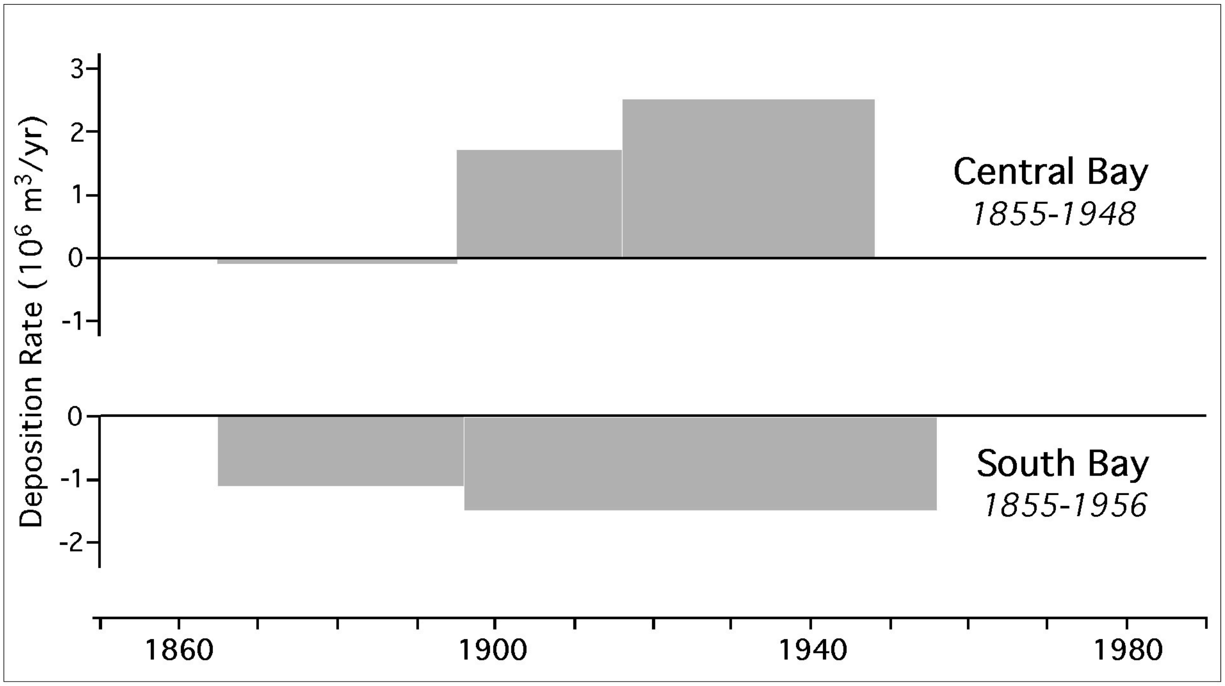


Figure 5. Sediment Deposited in San Francisco Bay 1870-1990, from Krone 1979 and Krone 1996

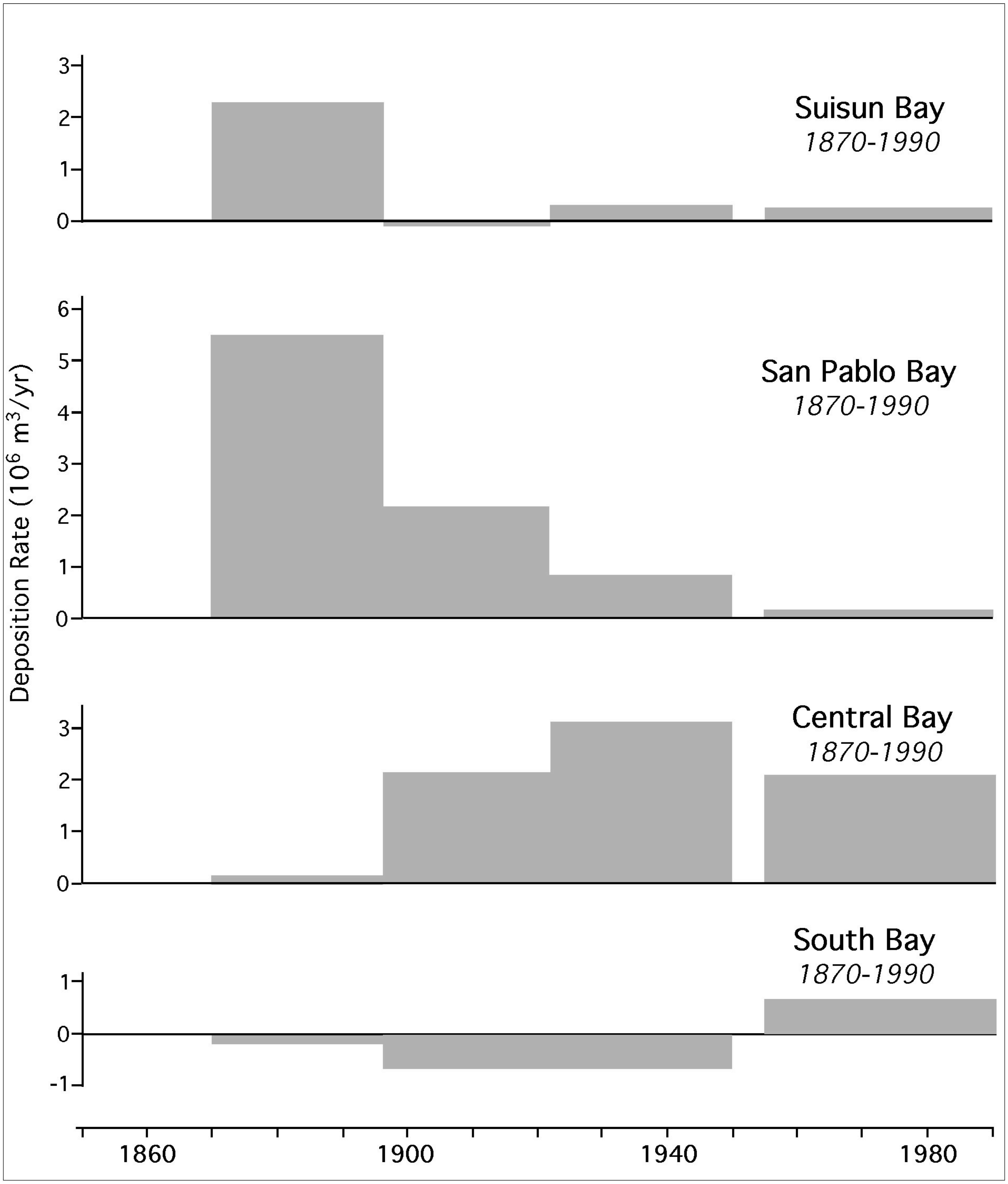
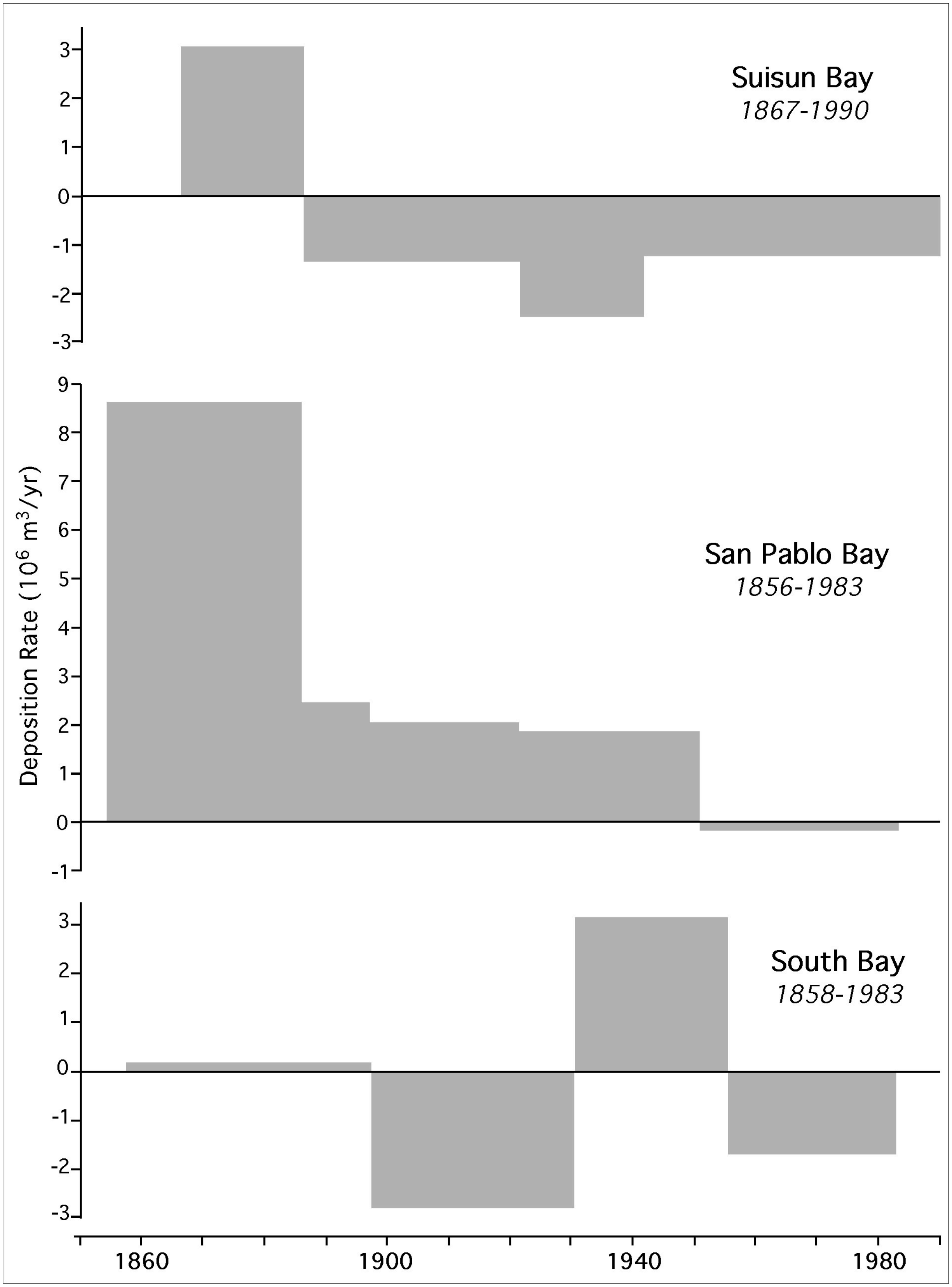


Figure 6. Sediment Deposited in San Francisco Bay 1856-1990, from Capiella et al. 1999 (Suisun Bay), Jaffe et al. 1998 (San Pablo Bay) and Foxgrover et al. 2004 (South Bay)



Despite the limitations of the Bay's sediment budgets and problems with the reporting and comparing of sediment flow estimates, some very general patterns are clear. In the late 19th century, sediment flows to the Bay were greatly increased over prior, natural levels due to mining, agricultural development and other activities. With changes in these activities, and with the construction of dams and impoundments that serve as traps to retain sediment, the delivery of sediment to the Bay has declined. The quantity delivered may be near or approaching the natural background levels that Gilbert (1917) estimated for the pre-mining period (see Table 8). Sediment deposition in the Bay also declined rapidly at first, but evidence of subsequent decline is lacking. Changing sea levels may increase the need for sediment, if marshes and mudflats are to keep pace with the rising sea. However, calculations suggest that despite the declining inflows there is still adequate sediment delivered to the Bay to meet this need (Van Geen and Luoma 1999).

One of the largest influences on the quantity of sediment carried into the Bay is the size of water flows. Changes in average flows and peak flows are caused by freshwater storage and use, by increases in the portions of the watershed that are covered by hardened surfaces due to urban development, by alterations in watercourses, by dam operations, and by climate change. In the last half of the 19th century, mining, land clearing and other activities increased both runoff rates and sediment load (Gilbert 19179). Flood control levees constructed along major watercourses reduced over-bank flooding and the deposition of sediment on floodplains, and this further increased the delivery of suspended sediments downstream. Starting in the 1940s, extensive dam construction caused the settling and retention of sediment in impoundments, which reduced the transport of suspended sediment downstream of the dams (Krone 1979). Water diversions, also increasing more rapidly since the early 1940s, also divert sediments and reduce the loadings to the Bay (Krone 1979). Most observers believe that water storage and use has substantially decreased the flow of water into the Bay relative to pre-1850 conditions (e.g. Nichols et al. 1986). Peak flows have mostly been reduced by dams and impoundments, although hardened surfaces and watercourse channelization may have increased peak flows in some local watersheds. In some areas, summer flows have been increased by the storage and delivery of water for agricultural, golf course and domestic irrigation, and by mandatory minimum releases of water to sustain fisheries or improve fish habitat.

Relevant climate change effects include changes in the timing, amount and type of precipitation, the amount of snow pack, the timing of snow melt and possibly the rate of evapotranspiration. Changes in the watershed over the past several decades have included increases in the frequency and intensity of extreme rainfall events and a shift toward earlier snow melt and earlier runoff peaks (Dettinger et al. 1995; Lund et al. 2007). Anthropogenic climate change is expected to continue these trends and to increase the year-to-year variability in precipitation, increase the frequency of large winter storms, and advance and compress the period of snowmelt, increasing the frequency and strength of peak winter runoff events (Lund et al. 2007). The overall net effect on the amount of sediment delivered to the Bay is unclear.

Impacts of Suspended Sediment

Besides activities and effects in the watershed that increase the amount of sediment delivered to the Bay in tributary waters, several activities locally inject substantial quantities of sediment into the water column and raise sediment concentrations to relatively high levels for a time. These include dredging and the in-Bay disposal of dredge materials, shell and sand mining, bottom trawling for fisheries or research purposes, and boat movements, particularly of large commercial vessels when they are maneuvering near wharfside and their keels are close to the sediment surface. The *potential* for impacts on organisms from large increases in sediment concentrations in the water column is well-documented, and include clogging the gills of fish and invertebrates; changing the behavior of adult fish; providing cover for prey species and reducing predation; and reducing light penetration, photosynthesis and the productivity and growth of eelgrass, seaweeds and phytoplankton (O'Connor 1991; ABP Research 1999; Levine-Fricke 2004). The assessment of activities that inject sediment into the water column thus hinges on the question of whether the sediment concentrations are elevated high enough for long enough to have an effect. In San Francisco Bay, the injection of sediment occurs in the context of a shallow bay with naturally turbid waters due to frequent wind and current stirring of bottom sediments into the water column, as well as the periodic discharge of sediment-laden runoff, especially in the spring and during and after storm events. The volume of bottom sediment resuspended in the water column of San Francisco Bay each year has been estimated at approximately 75 million cubic meters (Krone 1974, cited in LTMS 1998), 130 million cubic meters (Segar 1990) and 220 million cubic meters (San Francisco Estuary Project 1992, cited in LTMS 1998), quantities that dwarf the estimated 4-8 million cubic meters of sediment delivered by rivers each year (Table 8).

When sediment is injected into the water column it spreads out and downstream from the source in a sediment plume. The larger and heavier particles quickly settle to the bottom near the source, but fine material may remain suspended for some time and travel some distance before settling. A "worst case" suspended sediment field around a dredge or other source could have suspended sediment concentrations of up to 500 mg/L at up to 500 meters from the source (LaSalle 1990). Concentrations are generally much lower than this, and maximum concentrations are generally restricted to the lower part of the water column within 50-100 meters of the source. Such turbidity plumes are short-lived once the activity generating the sediment has stopped. Maximum levels of up to a few hundred mg/L are expected during major dredging operations in San Francisco Bay (Hirsch et al. 1978). In comparison, total suspended solids of up to 1,000 mg/L have been measured at turbidity maxima in northern San Francisco Bay (O'Connor 1991).

Suspended sediment concentrations were measured at 58-743 mg/L along the bottom at 50 meters downstream of an operating hopper dredge in San Francisco Bay, and were generally in the range of 70-130 mg/L 50 meters from a bucket dredge (Hanson & Walton 1990). Wakeman et al. (1975) measured suspended sediment concentrations around dredging operations in San Francisco Bay and compared these to levels needed to produce toxic effects in a few Bay species of fish, shrimp and mussel, and found that none of the organisms were sensitive to the typical dredge-produced turbidity conditions. Other tests on mussels, shrimp, a polychaete, an amphipod, an isopod and fish from San Francisco Bay found them tolerant of sediment loads "much in excess of" a few hundred mg/L for periods of up to 10 days (Hirsch et al. 1978). Concentrations measured at and near the Alcatraz dump site ranged from 10-50 mg/L (Segar 1990), well below the levels at which effects on organisms were observed (Wakeman et al. 1975). A workshop review found no evidence for any harmful effects on anadromous fish from coming into contact with dredge-associated sediment plumes (Simenstad 1990). Overall there appears to be no evidence that, chemical contaminants aside, the generation of sediment plumes by various dredging, mining or other activities poses a significant risk to organisms in a naturally turbid estuary like San Francisco Bay.

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Impacts of the Removal or Disturbance of Sediments, Shells or Bedrock in San Francisco Bay

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This paper discusses the direct impacts on organisms and habitats from the removal or disturbance of sediments by dredging, sand and shell mining, bottom trawling or ship movements and activities, and from the lowering of rock reefs or islands to eliminate navigational hazards. Impacts from these activities caused by the injection of sediments into the water column and their subsequent deposition, and from activities in the watershed that have altered sediment inputs to the Bay system (such as hydraulic mining, agriculture, grazing, road building, urban development), are treated under the Stressor "Change Sediment Inputs to the Water Column. Impacts caused by the injection of contaminants or nutrients associated with sediment are treated under the Stressors "Increase Contaminant Inputs" and "Change Nutrient Inputs."

Background Rate of Sediment Disturbance in the Bay

"Background" sources of bottom disturbance include the natural physical causes of sediment disturbance as well as activities by vertebrate and invertebrate animals (bioturbation).

<u>Physical disturbance</u>
Rubin and McCulloch (1979; see also Chin et al. 2004) used side-scan sonar to investigate changes in bedforms in the Central Bay. They found that over most of the Central Bay the principal physical process reworking the bottom is the migration of current ripples caused by tidal currents, which typically turned over only the upper 2-5 cm of sediment. However, in channels with sandy bottoms where current velocities were high the migration of sand waves resulted in turnover up to 1 m in depth. Hammond and Fuller (1979) found the surface of South Bay sediments to be fairly cohesive, and estimated that physical stirring affected only the upper 2 cm of the sediment or less. However, they found a high rate of radon flux through the sediment in the Central Bay, which if due to physical stirring of sandy sediments implies a turnover depth of about 40 cm.

<u>Bioturbation</u>
Several common gastropods In San Francisco Bay (e.g. *Ilyanassa obsoleta*, *Philine* spp., *Haminoea japonica*) and clams (juvenile *Venerupis philippinarum*) plow through the mud just under the surface, turning it over down to a few centimeters, while disturbance from the Bay's largest but much less common snail, the channeled whelk *Busycotypus canaliculatus*, might reach to around 10 cm. Lugworms (family Arenicolidae) belonging to at least two species are common in some areas, where they can rework the sediment down to around 20 cm. During the fall migration foraging water birds, especially scaup and scoter, can turn over substantial amounts of sediment in

parts of the shallow subtidal and intertidal in the South Bay and San Pablo Bay (Poulton et al. 2002, 2004; Richman and Lovvorn 2004; Thompson et al. in press; Jan Thompson pers. comm.). Large wintering congregations of foraging shorebirds must also cause some significant disturbance of intertidal sediments (personal observations). A few common clams in the Bay (*Macoma nasuta*, *Mya arenaria*) typically burrow to depths of 10-25 cm (Haderlie and Abbott 1980), and the burrowing anemone *Flosmaris grandis* may possibly reach to 50 cm (Fautin 2007). Feeding pits up to 30 cm deep dug by California bat rays (*Myliobatis californica*) are very abundant in some intertidal areas (Nichols 1979; Thompson et al. in press; personal observations), while pits up to 50 cm deep recorded by side-scan sonar in a shoal area of the Central Bay have also been interpreted as bat ray feeding pits (Rubin and McCulloch 1979; Nichols 1979). These pits are so dense in some areas that they cover virtually the entire surface. Two abundant subtidal polychaetes (*Sabaco elongatus* and *Heteromastis filiformis*) may dig burrows that are up to 40-50 cm deep (Hammond and Fuller 1979; Hammond et al. 1985). The ghost shrimp *Neotrypaea* is common in parts of the Bay (probably *Neotrypaea gigas*—J. Chapman pers. comm.), where it lives in impermanent, branching burrows that reportedly can extend to depths of 75 cm (Haig and Abbott 1980). Note that of the above species, only *Macoma nasuta*, bat rays and ghost shrimp are clearly native, so the background rate of bioturbation may differ from the natural rate.

Effects of Sediment Removal

Sediment or shell is deliberately removed from parts of the Bay by channel dredging, sand mining and shell mining. This has several potential consequences: the removal or killing of organisms living in or on the sediments; the short-term or long-term alteration of bottom habitat; hydrodynamic changes; the release of buried organic matter, nutrients or contaminants; short-term increases in suspended sediment concentrations; and the subsequent settlement of suspended sediments (LTMS 1998]; ABP Research 1999). The first three of these—the removal or killing of organisms, the alteration of bottom habitat and hydrodynamic changes—are discussed here.

An immediate impact of dredging or bottom mining is the loss of organisms that cannot escape removal by mechanical or hydraulic (suction) dredges.[13] Benthic infauna are most vulnerable though epibenthic and demersal species may also be vulnerable (Nightingale and Simenstad 2001; Levine-Fricke 2004). Depending on the depth of dredging, some infaunal organisms may escape by deep burrowing, but probably most

[13] Dredging results in at least a local depletion of these organisms. One study reported 99% mortality of fish entrained in pipeline dredges (Levine-Fricke 2004), while the mortality of Dungeness crab (*Cancer magister*) entrained by dredges ranged from 5%-100% depending on the type of dredging operation and the size of the crab (Wainwright et al. 1992; Nightingale and Simenstad 2001). Some invertebrate or algal species may fare better. If some organisms do survive the dredging, transport and disposal process, then the initial net impact of channel dredging on these organisms would be to remove them from the dredge site and transfer them to the disposal site, rather than to kill them. Whether they then survive and reproduce would depend on their condition and their response to their new environment. Note that the survival of these organisms is not necessarily a desirable outcome (depending in part on the distance between dredge and disposal sites), as it could faciliate the spread of non-native species or exotic genetic material between dredge and disposal areas.

are removed (ABP Research 1999). Some fish species may move away from the area of disturbance and sediment suspension caused by active dredging and avoid entrainment (ABP Research 1999; Levine-Fricke 2004), though demersal species are less likely to avoid elevated concentrations of suspended sediments than are surface species (Hanson Environmental 2004). Larval and juvenile fish are more vulnerable than adults (LTMS 1998; Levine-Fricke 2004), and fish that dwell in burrows in the sediment or that flee into burrows in response to disturbance (such as the Arrow Goby *Clevelandia ios*) could be entrained in large numbers. In San Francisco Bay, Dungeness crab (*Cancer magister*), bay shrimp (*Crangon* spp.) and demersal fish are vulnerable due their residence in or on bottom substrates and behaviors of burrowing or hiding in bottom substrates, and white sturgeon (*Acipenser transmontanus*) are vulnerable due to their bottom-orienting behavior and limited swimming ability (Nightingale and Simenstad 2001; Levine-Fricke 2004; Hanson Environmental 2004). In the Columbia River mouth, at least 14, mostly demersal, fish species were observed in hopper dredges (Levine-Fricke 2004). Dredge entrainment rates have been determined for 36 Pacific Coast estuarine or marine fish species in studies in Gray's Harbor (Washington) and the Columbia River estuary (Larson and Moehl 1990; McGraw and Armstrong 1990; Nightingale and Simenstad 2001). Pacific sand lance (*Ammodytes hexapterus*) were by far the most frequently entrained species, followed by other demersal fish including flatfish species and Pacific staghorn sculpin (*Leptocottus armatus*). A few pelagic fish were also entrained, including herring and anchovies (Larson and Moehl 1990; McGraw and Armstrong 1990). Longfin smelt (*Spirinchus thaleichthys*) and salmonids have been entrained by dredging in rivers, and longfin smelt and American shad (*Alosa sapidissima*) in Gray's Harbor, but are unlikely to be entrained in large numbers in estuaries (Larson and Moehl 1990; Levine-Fricke 2004).

Sites defaunated by the removal of sediments are subsequently colonized primarily by the lateral movement of organisms and by settlement of planktonic (larval) forms. The initial colonizers are often opportunistic species (e.g. characterized by relatively short generation times, small size, and high frequency and abundance of larvae in the water) that differ from those that were present prior to sediment removal; however, over time, the new biotic community often comes to resemble the pre-removal community. Studies on different types of substrate in different parts of the world have estimated the recovery time to range from around a month up to 10 years, with the time typically being shorter and recovery being more complete on unstable substrates or in disturbed areas in estuaries, in shallow inshore waters, in harbors, etc. (including sites subjected to periodic maintenance dredging), where the pre-removal community typically includes opportunistic, colonizing species (Oliver et al. 1977; Hirsch et al. 1978; LTMS 1998; ABP Research 1999; Levine-Fricke 2004). Evidence from bottom disturbance studies suggest that the most vulnerable and least resilient sites would include biogenic substrates, such as mussel beds, seagrass beds and beds of structurally significant worm tubes (Collie et al. 2000).

Longer term changes may result from modifications to the habitat or topography. Natural sediment deposits may have a complex structure, including vertical variation in particle size; bacterial or algal mats stabilizing the surface; tubes, burrows or pits created by various organisms; and accumulations of fecal pellets (Dernie et al. 2003). It can take some time to rebuild this structural complexity after disturbance or removal of the surface sediment. A permanent change in habitat may result if the area refills with sediment of a different grain size and composition than was present before the dredging or mining activity; if significant biogenic structures do not re-establish; or if the area does not refill to its pre-existing elevation (Hirsch et al. 1978]; Chin et al. 2004). Once a depression is formed, it may be maintained by tidal currents that inhibit sedimentation or cause erosion (Chin et al. 2004). It is not known how long the depressions caused by dredging or bottom mining last (Chin et al. 2004). One study reported that intertidal pits 1 m x 4 m x 0.1 m deep filled in completely within about 100 days if dug in sand but had not filled in after more than 200 days if dug in muddy sand or mud; the rate at which the pits refilled with sediment declined linearly with the increase in silt and clay content (Dernie et al. 2003).

Chin et al. (2004, Fig. 12) provide a 1985 bathymetric profile of a borrow pit near the east shore of San Francisco Bay that was a source of construction fill for Bay Farm Island. The pit covers nearly 5 km^2, and its bottom lies 6-10 m below the pre-existing and surrounding surface, which is about 2-3 m below MLLW,[14] and is also much rougher (i.e. has much greater variation in depth) than the pre-existing surface. It seems likely that environmental conditions also differ, at least in the frequency and degree of disturbance by waves, currents or passing vessels and in the amount of available light, and possibly in sedimentation rate, sediment characteristics, etc. If the water over the pit ever stratifies (which because of eddy currents or other factors might be extremely rare—Jan Thompson pers. comm.), there could also be differences in salinity and temperature (for example, Conomos (1979) indicates a difference of about 5 ppt and 1°C over 10 m depth in this part of the Bay during wet winters).

The part of the Bay where this borrow pit is located is mapped as habitat for eelgrass (*Zostera marina*; Cosentino-Manning et al. 2007, Fig. 12), which grows down to around 3 m below MLLW in parts of the Bay (Cosentino-Manning et al. 2007). Marine algae have low light requirements and may grow to considerable depths in clear water, but can be restricted to depths of 2 m or less in turbid waters (Silva 1979). Thus eelgrass or algae (such as *Gracilaria*) might have occurred on the shallow, pre-existing surface (at 2-3 m below MLLW), but would not occur or would be very unlikely to occur on the deeper, dredged surface (at 9-13 m below MLLW). This borrow pit shows up on NOAA navigation charts as an obvious hole in the bottom of the Bay, along with other holes off Emeryville, alongside Treasure Island and Hunters Point and at San Bruno Shoal, at least some of which are apparently also the result of bottom mining (Table 1). Other large borrow areas in the Bay include the Presidio Shoal Borrow Area and the Point Knox Shoal Borrow Area in the western part of the Central Bay, from which 15-22 million m^3 of sediment was dredged in 1936-38 to create Treasure Island. USGS multibeam sonar imagery shows a topographic depression still evident at the Point Knox Shoal site in 1997, with an estimated volume of missing sediment of at least 2.4 million m^3 in an area with sandy bottom (Chin et al. 2004, Fig. 10). Numerous smaller scale

[14] Chin et al (2004, Fig. 12) show this as 3-4 m below Mean Sea Level (MSL). At the Alameda Tide Station (Station #9414750), MSL is 1.05 m above MLLW (NOS records).

alterations of topography can also have a substantial cumulative effect. At the western end of Point Knox Shoal, the pits and channels caused by sand mining "are so numerous as to literally obliterate the fabric of the bay floor" (Chin et al. 2004).

Foxgrover et al. (2004) identified four large borrow pits in the South Bay, that showed up as anomalies in the patterns of erosion and deposition revealed by hydrographic surveys. Together these depressions cover approximately 31 km^2, and represent the removal of at least 39 million m^3 of sediment. Between 1956 and 1983, the sediment removed from two of these pits accounted for at least 37% of the net loss of sediment from the South Bay. These pits were apparently created either by dredging for fill material or by shell mining. However the history of these and the other large borrow pits in the bottom of the Bay is poorly known or unknown (Foxgrover et al. 2004).

Table 1. Some Large Borrow Pits in San Francisco Bay

Location	Approximate surface area (km^2)	Approximate sediment volume removed (10^6 m^3)	Period of Activity	Reference
West of Bay Farm Island	5	30-50		Chin et al. 2004
Presidio Shoal and Point Knox Shoal Borrow Areas		15-22	1936-38	Chin et al. 2004
South Bay Borrow Pit 1 (north of San Mateo Bridge)	2	≥3	1931-56	Foxgrover et al. 2004
South Bay Borrow Pit 2 (south of San Mateo Bridge)	11	≥10	1931-56	Foxgrover et al. 2004
South Bay Borrow Pit 3 (San Bruno Shoal)	9	≥9	1956-83	Foxgrover et al. 2004
South Bay Borrow Pit 4 (north of San Mateo Bridge)	9	≥17	1956-83	Foxgrover et al. 2004

Reductions in bottom elevation caused by dredging or mining can cause changes in the hydrodynamic regime which can in turn affect areas that are outside of the sediment removal zone. These hydrologic changes include the intrusion of salty bottom water further upstream; alterations in tidal ranges, tidal prisms or tidal currents; and changes in erosion patterns and consequent suspended sediment loads (ABP Research 1999). Such effects are likeliest when the size of the excavation is significant relative to the overall size of the system. Upstream salt intrusion has been noted as a potential or actual consequence of channel dredging in the Bay's northern reach and Delta.

Activities Removing Sediment from the Bay

<u>Channel Dredging</u>

Dredging removes sediments that are either in their natural condition (called "new work construction") or in a recently deposited condition ("maintenance dredging"), using either mechanical or hydraulic equipment, and then transports the sediments to a disposal site either on the dredge, on barges or scows, or in pipelines (LTMS 1998). Mechanical dredging can be used for either maintenance or new-work dredging. It removes either loose- or hard-compacted materials by applying direct mechanical force to the sediment, removing it in almost in situ densities with backhoe, bucket dredge (e.g. clamshell, orange-peel, dragline), bucket-ladder, bucket-wheel or dipper dredge. Hydraulic dredging is used mainly for maintenance projects. It removes loosely compacted sediment using cutterheads, dustpans, plain suction or sidecasters and transports the sediment in a liquid slurry through pipes (6-48 inches in diameter) either to the disposal site or to a hopper (LTMS 1998; Levine-Fricke 2004).Over the next 40 years an estimated average of 2.6 to 4.5 million m^3/yr of sediments will be dredged from the Bay, with 84-93% of this being maintenance dredging (LTMS 1998).

<u>Sand Mining</u>

Over 1-1.5 million m^3 of sand and gravel was dredged in 1912-1915 from Presidio Shoal to create San Francisco's Marina District (Chin et al. 2004). Sand mining with hydraulic suction pumps began in the Northern channels of the Bay in the 1930s, and in the Central Bay in the 1950s (Hanson et al. 2004). Currently, around 1.2 million m^3 of sand is mined from the Bay each year. About 90% is taken from the shoal areas of the West Central Bay at depths of 10-30 m, and about 10% from the main Suisun Bay channel between Benicia and Chipps Island at depths of 5-15 m (Hanson et al. 2004).

<u>Shell Mining</u>

Oyster shell has been mined commercially in South San Francisco Bay since 1924, primarily for use in the manufacture of cement, as a supplement in poultry feed, and as a soil amendment. The main mining sites were north and south of the San Mateo Bridge east of the shipping channel but in the western half of the Bay. Some shell was also mined off Bay Farm Island and south of the Dumbarton Bridge. The shell harvested is 2,300-2,500 year-old native oyster shell (*Ostrea conchaphila*) that occurs as lenses in the upper 10 m of sediment (within the "younger bay mud" deposit of Treasher 1963). The lenses are usually 1-5 m thick, and are typically overlaid by 0.6-2.5 m of fine mud. About 25-35 million tons of shell were removed between 1924 and the mid-1960s, with an estimated 75 million tons then remaining (Hanson Environmental 2004).

Shell is currently harvested at only one site in the Bay, on California State Lands Lease PRC 5534.1, a rectangular area covering 6 km^2 just north of the San Mateo Bridge on the east side of the channel, where the bottom is 2-4 m below MLLW. About 30,000 tons/year have been taken from this lease since 1999. The lease was recently renewed through December 2016 with a 10-year renewal option, and allows the removal of 40,000 tons of shell a year. Shell is harvested by burying the suction head of the dredge 0.3-1 m deep in the mud and then slowly trolling it; burying the suction head may reduce the entrainment of near-surface organisms. From the suction head a slurry consisting of approximately 50% shell, 45% water and 5% silt is carried through pipes into a barge. The shell is retained on the barge, with the water and most of the silt discharged back to the Bay (Hanson Environmental 2004). Averaged over the area of the lease, the removal of 40,000 tons of shell per year (≈60,000 cubic meters) corresponds to lowering the surface by about 1 cm per year.

Effects of Sediment Disturbance

Bottom trawling catches demersal fish or invertebrates for sale, research or educational purposes, and in the process churns up and turns over sediments. In addition to removing target species and by-catch, trawling crushes, buries or exposes organisms, which attracts predators and scavengers (Thrush et al. 1998; Watling and Norse 1998). As noted above, structural complexity in the sediment can be disrupted (Dernie et al. 2003; Watling and Norse 1988). While trawling can smooth ripples, mounds and other small-scale structures, plowing by trawl doors can create large furrows, potentially replacing "widespread, small-scale, low relief features...with a rather smoother landscape, interspersed with higher relief, but less frequent features" (Kaiser et al. 2002). The small-scale structural features destroyed by trawling can be of great importance to bottom biota and demersal fish (Watling and Norse 1998). The collapse of burrows and sediment voids, and damage to bioturbating infauna, could in turn affect biogeochemical exchange processes between sediments and the water column (Kaiser et al. 2002).

Different studies of the impacts of bottom fishing gear on biota often yield different results, in part because of the variety of gear, bottom types and environmental conditions. A common conclusion, however, is that bottom disturbance from fishing reduces large, long-lived epifauna and favors small organisms and juvenile stages (Thrush et al. 1998; Collie et al. 2000; Kaiser et al. 2002). An analysis of sites with varying degrees of fishing pressure found that greater bottom fishing reduces the density of echinoderms, large species and long-lived species; reduces the total number of species and individuals; reduces diversity as measured by the Shannon-Weiner diversity index; and increases the density of deposit feeders and small, opportunistic species (Thrush et al. 1998). A meta-analysis of 39 published studies found that all major taxonomic groups decline following bottom fishing, but concluded that anthozoa (anemones) and malacostraca (a type of crustacean including crabs, lobsters, shrimp, amphipods and isopods) are the hardest hit (Collie et al. 2000). This meta-analysis also found that otter and beam trawling has less impact than harvest methods that involve digging, raking or dredging that remove sediments as well as organisms from the seabed or that disturb sediments to a greater depth. Similarly, other studies have found that disturbance from otter trawls is largely restricted to the trawl boards (Kaiser et al. 2002). In general, mud or muddy-sand is affected more and takes longer to recover than sand (Collie et al. 2000; Kaiser et al. 2002; Dernie et al. 2003; but also see contrary results in Collie et al. 2000). One study found that otter trawl boards typically penetrate muddy sand 2-4 times deeper than fine or coarse sand, with the furrows remaining for at least a year (Churchill 1989; Kaiser et al. 2002). Collie et al. (2000) concluded that since sandy sites affected by bottom fishing gear recover in around 100 days, they can be fished 2-3 times per year without markedly changing their character, but that other types of bottom require longer recovery periods of up to 500 days. As with the impacts of dredging and bottom mining, shallow, turbid and naturally-disturbed sites are less likely to be significantly affected than deeper, undisturbed sites (Watling and Norse 1998; Kaiser et al. 2002).

Overall, these studies suggest that bottom fishing has not had a large impact on bottom habitat in San Francisco Bay, at least in recent decades when commercial trawling has been limited to a small bait shrimp fishery (see below). While most of the Bay bottom is mud, and thus more sensitive to the effects of trawling than sand bottom, the Bay is shallow and turbid with a high frequency of natural bottom disturbance from wind waves and tidal currents (e.g. Krone 1979; Conomos et al. 1979; Nichols 1979; but see Hammond and Fuller 1979, and Rubin and McCullough 1979, suggesting that natural disturbance affects only the upper 2 cm or 2-5 cm on mud bottom), and most of the commercial bottom fishing in the Bay has used gear types that have relatively smaller physical impacts on the bottom. Two caveats, however, should be borne in mind. First, there has been no quantification of the historic or current levels of fishing impacts on the bottom in terms of the distribution, acreage and frequency of trawling in the Bay. Second, impacts from trawling are believed to be substantially greater on biogenic substrates (Collie et al. 2000; Kaiser et al. 2002). In San Francisco Bay these include eelgrass, algae and oyster beds, and we have very little information on the initial extent and distribution of these beds or on their later historic or current distribution and extent relative to trawling activities.[15] Trawling also removes fauna and flora that are important sediment stabilizers, including tube-building amphipods (such as *Ampelisca abdita*) and polychaetes (such as *Sabaco elongatus*).[16] *Ampelisca* are removed in such numbers that the Department of Water Resources (research trawling) and Marine Science Institute (educational trawling) have moved transects to avoid beds of *Ampelisca*, which can completely clog nets (Jan Thompson pers. comm.).

Activities Disturbing Sediment in the Bay

Commercial Fisheries

Commercial fishing began in San Francisco Bay around 1848, initially with hand lines, beach seines and gill nets. By 1870 there were commercial fisheries for a few demersal species, including sole and flounder from the South Bay to southern San Pablo Bay, and sturgeon (*Acipenser* spp., caught on hook-and-line, or incidentally in nets deployed for other species) in northern San Pablo Bay. About this time Italian fishermen began seining for shrimp (*Crangon* spp.), but in 1871 the Chinese started catching shrimp with set or "bag" nets (nets held to the bottom by stakes or poles driven into the sediment which were emptied and then reset in the opposite direction with the change of tide) and the competition drove the Italian seine netters out of shrimping. In 1876 the paranzella or Mediterranean drag net was introduced to the Bay Area, and in 1885 the first steam tug for trawling (Skinner 1962; Smith and Kato 1979).

In 1895 the hook-and-line fishery for sturgeon in the Bay, practiced by Chinese fishermen, was prohibited. Commercial sturgeon fishing was prohibited in the Bay in

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[15] One indication that there may have been some significant overlap between trawling sites and biogenic substrates comes from Ganssle (1966) who noted that in 1963-64 the tunicate *Molgula manhattensis* (reported as *M. verrucifera*) was "so abundant in San Pablo Bay bottom tows that it was impossible to haul the trawl aboard by hand." *Molgula* attaches to hard surfaces or vegetation and does not live on sediment, and the most likely substrate for the *Molgula* filling the trawl nets in San Pablo Bay was the seaweed *Gracilaria* (personal observations). Reserach trawling may thus have had some impact on *Gracilaria* beds.

[16] These are both exotic species, as are some of the other common sediment-stabilizing species in the Bay.

most years between 1901 and 1916 and banned permanently in 1917 (Skinner 1962; Smith and Kato 1979). The Bay's shark fishery, which began in the South Bay in the 1890s, peaked in the late 1930s to the early 1940s with annual landings of around 900 tons. These had dropped to under 50 tons by the1950s, before the commercial fishery left the Bay (Skinner 1962; Smith and Kato 1979). A series of restrictions were placed on the Chinese shrimp fishery starting in 1901, and in 1911 set nets were prohibited only to be allowed again in the South Bay in 1915. Beam trawling for shrimp started in 1914-1921, mainly in San Pablo Bay, and steadily grew in volume while set net shrimping continued for a time in the South Bay. By the late 1920s, San Pablo Bay trawlers were catching nearly 800 tons of shrimp, compared to a South Bay set net catch of only 200 tons. Shrimp landings remained at around 1,000 tons/year through the 1930s, dropped to around 400 tons in the 1950s, and have been under 100 tons, sold mainly for bait for striped bass and sturgeon sport-fishing, since the mid-1960s. There were 19 boats trawling for shrimp in the Bay in 1930, and 15 boats in the late 1970s; by the mid-1990s, however, there were only seven licensed shrimp boats in San Pablo and Suisun bays and two in the South Bay (Clark 1930; Skinner 1962; Smith and Kato 1979; CDFG license data).

<u>Research/Educational Trawling</u>
Bottom trawling and beach seines are often used for research and education in the Bay. For example, the California Department of Fish and Game Bay-Delta Monitoring Program has used an otter trawl to conduct monthly sampling at 35-52 sites in the Bay and western Delta since 1980, and used beach seines at 27 shoreline sites in the Bay each month from mid-1980 through 1986 (CDFG 2007). The Marine Science Institute, an educational organization, has trawled in the South Bay for 35 years, conducting typically 200-400 otter trawls per year (MSI undated). Many other research, monitoring and education programs drag nets along the bottom of the Bay.

<u>Shipping</u>
Vessel movement, docking, anchoring and propeller wash can also cause some disturbance or alteration of bottom sediments and even of bedrock. Studies conducted at the Richmond Longwharf found that docking ships and barges stirred up large plumes of sediment (USACE 2005). During the geophysical investigation of Arch Rock conducted in 2000, deep gouges were noted that were thought to be possible anchor scars (Sea Surveyor 2001). Around 3,000-4,000 cargo vessels entered the Bay each year in 1977-1996 (Marine Exchange 1997). Although the cargo handled at San Francisco Bay ports is projected to more than double between 2000 and 2020 from less than 20 to over 40 million tons (exclusive of oil and oil products, bulk sugar and Hawaiian molasses), the number of ship calls will decline as the average ship size increases (BCDC 2003). Other things being equal, bottom disturbance by ships may become less frequent (fewer ships) but produce greater disturbance per event (larger, deeper-draft ships).

Bedrock Removal

Four bedrock features in the western part of the Central Bay have repeatedly been lowered by blasting to reduce the navigational risk they pose to shipping (Chin et al. 2004). There are several records (not all of which may be accurate) of ships striking or grounding on these rocks in the 1800s, including the following: In 1832, the East Indiaman *Seringapatan* struck Blossom Rock without causing any harm to the vessel. In 1853 the pilot boat *Sea Witch* was wrecked on Arch Rock. In 1855 the *Lenore* grounded on Arch Rock. In 1856, the clipper ship *Goddess* grounded on Blossom Rock while heading out of the Bay. In 1862 the clipper ship *Flying Dragon* entered San Francisco Bay after a record-fast passage from Newcastle in Australia, was caught by a squall, wrecked on Arch Rock, and sank. In 1868 the *Autocrat* got stuck on Arch Rock and was wrecked. In 1877, a few years after the initial lowering of Blossom Rock, the *Highland Light* struck it and the *Blanchard* grounded on it while they were under tow.

Blossom Rock, which lies about 1 km north of the San Francisco wharves and 2 km southeast of Alcatraz Island, is a subsurface ledge that originally reached to within 2 m of Mean Lower Low Water (MLLW). It was reduced by mining and blasting to a depth of 7 m below MLLW in 1870, to 9 m in 1903 and to 12 m in 1932. Arch Rock, Shag Rocks and Harding Rock lie in an arc about 1 to 2 km west and northwest of Alcatraz Island. The southernmost, Arch Rock (also known at one time as Bird Rock), reportedly stood about 10 m above the water (above low water, presumably), and was about 15 m long and 3-6 m wide. There was an arched opening in its center, large enough to pull a boat through with difficulty. In 1900-1903 Arch Rock was lowered to 9 m below MLLW, and to 11 m in 1932. Shag Rocks (once also known as Barrel Rock) consisted of two rock knobs, the taller of which stood about a meter above the highest tides and was about 3 m long. The two rocks were lowered to a depth of 9 m below MLLW in 1900 and 1901 and to 11 m in 1931-32. Harding Rock, the northernmost of these rock features, was not discovered until 1917. It is a pinnacle that originally reached to about 9 m below MLLW, and was lowered to 11 m below MLLW in 1932 (Sea Surveyor 2001; Allan 2001; Chin et al. 2004). It has been proposed that these four rock features should now be further lowered to 17 m below MLLW, to reduce risks to modern deep-draft cargo vessels (Carlson et al. 2000; Chin et al. 2004). However, the U.S. Army Corps concluded that the benefits of lowering the rocks would not be worth the costs because "current navigational practices make an oil spill resulting from a tanker or other vessel grounding on one of the knobs very unlikely" (Chin et al. 2004).

Some of the names that have been used for the two rocks that originally projected above the water surface (Bird Rock, Shag Rock) suggest that they were heavily frequented by sea birds, though it's not known if they were used as breeding sites. All four rocks now have the form of submerged rock masses with flattened tops that rise 12-15 m above the surrounding sea floor and whose highest points are 11-12 m below MLLW (Sea Surveyor 2001). The upper surfaces of these rocks down to depths of 20-25 m below MLLW consist of rock reef strewn with blocky rubble that ranges from cobbles to boulders several meters long, apparently left from the blasting (USACE 2003; Chin et al. 2004). Below this depth the rock masses are separated from each other and isolated from other exposed bedrock features in the Bay by a cover of unconsolidated bottom sediment (coarse sand, gravel and shell hash) that is around 2 m to 8 m or more thick (Sea Surveyor 2001), and thus the rocks form small habitat "islands." They are generally similar to other hard-bottom habitats at similar depths along the Central California coast, but are uncommon habitat types within the San Francisco Bay region (Garcia and Associates 2001). A benthic video survey of the rocks conducted by ROV in 2001 found three species of sea stars, rock crabs (*Cancer* spp.) and turf organisms (hydroids, bryozoans, anemones, sponges, etc.) to be common on the rocks and tabulated a low species richness, though this result was influenced by the extremely poor visibility (due to high turbidity) and the limited taxonomic resolution of the survey (Garcia and Associates 2001). No reference was made to the presence or absence of macroalgae. Turf organisms covered 25% to nearly 100% of the bottom, and were more common above 18 m depth. On the three northernmost rocks there were abundant hard-bottom sea stars—species that are not capable of migrating across the intervening sediment-covered bottom and must have arrived on the rocks as planktonic larvae—consistent with the view that the rocks function as habitat islands (Garcia and Associates 2001).

In addition to the loss of sea bird resting habitat—and possibly the loss of sea bird breeding sites or pinniped haulout sites—the initial reduction of projecting rocks to depths below the surface is likely to have eliminated some supralittoral, intertidal and near-surface subtidal species. While further lowerings to progressively greater depths that have been done or are proposed are less likely to eliminate species, the decrease in cover by turf organisms with depth noted in the benthic survey suggests that there would be impacts on the abundance of some organisms and possibly on species composition. If species of algae are present, removal of the shallowest portions of the rocks would reduce and might eliminate them. Reductions in the total surface area of these habitat islands could also result in the loss of some species, and their isolation could hamper recolonization. Both from fishing reports (USACE 2003) and from observations (e.g. schools of fish obscuring side-scan images—Sea Survey 2001), it appears that fish are abundant and fishing is good around these rocks, and lowering them further could change that.

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Impacts of Artificial Structures Placed in San Francisco Bay

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Many types of structures have been built or placed in or over the waters of San Francisco Bay, including wharves, piers, pilings, jetties, breakwaters, floating docks, buoys and moorings that service shipping and boating activities; sea walls and riprap that armor shores and protect them from erosion; nine major bay-crossing bridges and at least eleven smaller bridges over marginal arms of the bay that carry auto roads or railroads, and an unknown number of foot bridges; transmission towers and power lines; cooling water intakes for power plants and outfalls for power plants and water treatment plants; and across the floor of the bay, pipes and cables. There has been no general assessment of the effects of these structures on Bay organisms, and so an overview must be pieced together from generally unpublished sources of Bay information and inferences from studies on the impacts of structures that have been conducted elsewhere. Impacts from these structures include eliminating existing bottom habitat, creating hard substrate, shading, changing water circulation, altering adjacent habitat, changing fish behavior, creating resting or nesting sites for birds or pinnipeds, and probably facilitating the establishment of some exotic species.

There is no overall summary of the amount and distribution of artificial structures in the Bay, the portion of the shoreline that has been hardened, etc. There are six public cargo ports, several proprietary cargo terminals (including oil terminals and automobile importing terminals, primarily in Contra Costa and Solano counties) and several current or former military terminals (Mare Island Naval Shipyard, Alameda Naval Air Station and Hunters Point Shipyard, all closed, and Concord Naval Weapons Station, still active) in the Bay, along with over 200 marinas providing slips for over 33,000 boats in the Bay and Delta combined (Marine Exchange 1994; LTMS 1998). The San Francisco Bay Area Seaport Plans (BCDC 1996, 2003) report that there were a total of over 57,000 linear feet of cargo berths in the Bay in 1994, with a projected 62 effective berths in 2020 (Table 1). A recent boating guide (Dinelli and Dinelli 2003), lists 65 marinas and yacht clubs with nearly 19,000 berths distributed around the Bay (Table 2).

Table 1. Berth Length and Number at Cargo Ports in San Francisco Bay (based on BCDC 1996, 2003)

Port	Length of Berths (ft) in 1994	Projected Effective Number of Berths in 2020
Port of Redwood City	1,805	5
Encinal Terminals	1,313	0
Port of Oakland	21,110	21
Hunters Point Naval Shipyard	0	2
Port of San Francisco	25,373	14
Port of Richmond	4,409	12
Selby	0	5
Port of Benicia	3,200	3

Total	57,210	62

Table 2. Berths at Marinas and Yacht Clubs in San Francisco Bay (listed in Dinelli and Dinelli 2003)

	Number of Marinas	Number of Berths
Suisun Bay	6	1,555
San Pablo Bay	9	2,391
Central Bay	28	8,361
South Bay	22	6,560
Total	65	18,867

Eliminating Existing Habitat

Jetties, breakwaters and similar structures eliminate the habitat they are placed on, usually shallow subtidal and intertidal mud or sand bottom. Although no figures are available, the fraction of such habitat that has been eliminated in the Bay by building structures is probably small. The impacts of placing structures on rarer habitats, such as hard substrate, eelgrass beds or shellfish beds, would be more significant.

Increasing Hard Substrate

Natural hard substrate (mainly bedrock outcrops and associated boulders and cobbles) is rare throughout most of the Bay, except for the western part of the Central Bay. All of the structures listed above provide additional hard substrate in the Bay, however these artificial substrates generally do not closely resemble natural hard substrate. Floating substrates (docks, buoys, moorings) and pier pilings provide habitat conditions that differ greatly from natural substrates (Glasby and Connell 1999; Connell 2000; Holloway and Connell 2002), while rock jetties and breakwaters and unshaded concrete structures are probably most similar to natural substrates (e.g. Connell and Glasby 1999). The physical differences vary. Floating structures maintain organisms at a near-constant, mainly shallow water depth, which differs from any fixed natural substrate on which shallow water organisms are affected by the rise and fall of the tides. Floating structures to a greater or lesser degree may also isolate the organisms growing on them from benthic predators and other benthic organisms. Floating substrates also affect the exposure of organisms to surface lenses of fresh water and to floating oil and other contaminants. The texture, rigidity, temperature response and surface chemistry of materials found on artificial structures including wood (chemically treated and untreated), plastic, styrofoam, concrete, rubber and metals can differ greatly from the characteristics of natural structures. The surface orientation and the degree of shading of artificial surfaces can also depart significantly from that of most natural surfaces, with near vertical and horizontal overhanging surfaces being far more common on artificial structures. Published studies have found that natural and artificial hard substrates located near each other tend to be dominated by different suites of species (Connell and Glasby 1999; Glasby 1999b). Some studies have found that artificial substrates are more dominated by exotic species (Lambert 2002; Glasby et al. 2007). Connell and Glasby (1999) concluded that "artificial structures may increase the abundance and diversity of subtidal epibiota in the shallow areas of an estuary, but are not surrogate surfaces for epibiotic assemblages that occur on nearby natural rock."

There are no published studies quantitatively comparing species composition on artificial and natural hard substrates in the Bay. Researchers' qualitative perceptions have differed, on the one hand finding greater dominance by exotic species on artificial substrates, most notably on docks and other floating substrates (personal observations); and on the other hand finding little difference between the biotas of submerged rocks and artificial marina substrates in the Central Bay (Chris Brown, pers. comm.).

Certain fish species are commonly found in association with artificial structures (Clynick 2008), possibly because of food or cover provided by the epibiota on the structure or a preference for shade or the shadow line. In San Francisco Bay, fish commonly found near or in the fouling growth on floating docks and pilings include Bay Pipefish (*Syngnathus leptorhynchus*), Shiner Surfperch (*Cymatogaster aggregata*) (both naturally occurring in eelgrass) and the non-native Chameleon Gobies (*Tridentiger trigonocephalus or T. bifasciatus*) (personal observations).

Altering Adjacent Habitat

Structures built in the water can alter water flows and patterns of sediment erosion and deposition. Depending on the circumstances, sediments can be scoured around the base of structures and/or deposited in the lee of structures (Whitehouse 1998; Sumer 2001; Sumer and Fredsoe 2003). The long jetty at the south end of Mare Island contributed to the substantial accretion of sediment along the western shore of the island during the 20th century (Atwater et al 1979). At the Point Isabel Regional Shoreline in Richmond, sediment built up between a detached breakwater and the shore has developed into a salt marsh (personal observations).

Bay mussels, including both a native (*Mytilus trossulus*) and an exotic species (*M. galloprovincialis*) and/or hybrids between them, are common or abundant on many of the structures in the Bay. Over time, the accumulation of dead shells from these structures can change the adjacent bottom type to shell hash (Pentilla and Doty 1990; Nightingale and Simenstad 2001).

Floating docks that ground on low tides can eliminate eelgrass below them (Nightingale and Simenstad 2001), and probably affect other benthic organisms as well. Chains used to anchor mooring buoys, barges, rafts, booms, etc. can damage bottom vegetation by dragging on the bottom. Buoys moored with rope lines, especially with mid-line floats, cause less damage than buoys attached by chains (Nightingale and Simenstad 2001).

Shading

Studies in the Pacific Northwest and elsewhere have found that shading by overwater structures reduces or eliminates eelgrass and seaweeds beneath them (Pentilla and Doty 1990; Fresh et al. 1995, 2001; Burdick and Short 1999; Nightingale and Simenstad 2001). On hard substrate, shading decreases algal cover and in some studies reduces the abundance of spirorbid worms and grazing snails, and increases the abundance of attached Invertebrates, including sponges, serpulid worms, barnacles, bryozoans and tunicates (Glasby 1999a,b; Blockley 2007), but no consistent effects were observed on soft sediments under structures (Lindegarth 2001). Shading can also impair prey capture by fish, which are primarily visual feeders, and possibly affect their spatial orientation, schooling or predator avoidance behaviors, all of which are partly sight-dependent (Nightingale and Simenstad 2001). In the Hudson River Estuary, Able et al. (1998) compared fish distributions underneath the center of large commercial, piling-supported piers, in piling fields consisting of an array of pilings where the pier or deck had been removed, and in adjacent open water. Relative to the other sites, the pier sites had much lower light levels (<0.12 $\mu E/m^2$-s throughout the water column at the pier sites compared to >566 $\mu E/m^2$-s at 0.5 m depth and >9 $\mu E/m^2$-s on the bottom at the piling field and open water sites); typically lower fish abundance and species richness; greatly reduced abundance of young-of-the year fish; and increased abundance of eels. Another study found that caged fish under piers showed periods of starvation compared to caged fish at pier edges and in open water, and that this was likely due to shading impacts on prey capture (Duffy-Anderson and Able 1999; Nightingale and Simenstad 2001).

The effects of shading by overwater structures can be reduced by reducing the width or raising the height of the structure (that is, by increasing the distance between the water surface and the underside of the structure) or orienting the structure in a north-south direction, and possibly by incorporating gratings or glass blocks in the structure to transmit light, increasing the space between pilings, or using materials that reflect light (e.g. concrete rather than wood pilings) or reflective paint on the underside of docks (Burdick and Short 1999; Nightingale and Simenstad 2001; Fresh et al. 2001; but see Loflin 1993). Covered moorages, boathouses, houseboats, and other vessels moored alongside can enlarge the shade footprint of piers and floating docks and extend their impact (Nightingale and Simenstad 2001). Impacts from shading appear to be less under floating docks that are attached by chains that allow some movement rather than fixed in position by pilings (Pentilla and Doty 1990; Nightingale and Simenstad 2001).

Altering Fish Behavior

Various studies have reported fish behaviors that appear to be responses to encountering artificial structures, including reluctance to pass under docks and piers, pausing and going around docks, schools breaking up on encountering docks (Weitkamp 1982; Nightingale and Simenstad 2001). Juvenile salmonids, for example tend to remain along the line of shadow and avoid areas of deep shadow (Nightingale and Simenstad 2001). Impacts from these responses could include migration delays due to disorientation and increased predation risk due to breaking up of schools or deflection into deeper waters (Nightingale and Simenstad 2001).

Creating Resting and Nesting Sites

High relief natural landscape features such as cliffs and trees are rare in or near the shore of much of San Francisco Bay, and high relief artificial structures may provide sites for bird resting or nesting in areas where they are otherwise absent or rare. Low artificial structures near the water may also provide resting sites for birds or pinnipeds, especially if they are not connected to the mainland. The following is an incomplete description of the use of such structures in San Francisco Bay.

Raptors have been observed on artificial structures in or near San Francisco Bay salt marshes, including White-tailed Kite (*Elanus leucurus*) and American Kestrel (*Falco sparverius*) on low perches (posts), and Peregrine Falcon (*Falco peregrinus*) on transmission towers (personal observations). Since the 1990s, Peregrine Falcon have occasionally attempted to nest on the Oakland Bay Bridge and hunted from the Golden Gate Bridge (Bell 1994; Granholm 2007). Peregrine Falcon nesting on the Coronado Bridge over San Diego Bay periodically took California Least Tern (*Sternula antillarum brownii*) from a nearby colony, and there have been concerns about similar interactions in San Francisco Bay (Bell 1994).

Many birds use jetties and breakwaters as resting sites, including a colony of California Least Terns that uses the detached breakwater off the former Alameda Naval Air Station. California Least Terns have also been observed resting on the abandoned western end of the Berkeley Pier (Granholm 2007). Gulls (*Larus* spp.) frequently rest on pier railings and other structures; pelicans, herons and egrets patronize certain docks; and Turkey Vultures (*Cathartes aura*) sometimes rest on a small abandoned pier near the mouth of Meeker Slough in Richmond (personal observations). Small numbers of Western Gulls (*Larus occidentalis*) nest on the Richmond Bridge and the Oakland Bay Bridge (Granholm 2007; SFSU 2007). Double-crested Cormorants (*Phalacrocorax auritus*) nest on the Richmond Bridge (about 500 nests currently), the Oakland Bay Bridge (about 800 nests currently), the cable-crossing structure near the Oakland Bay Bridge (2 nests observed in 2007) and the transmission towers just south of the western span of the San Mateo Bridge and on Redwood Creek (Stenzel et al. 1995; Strong 2005; Granholm 2007; SFSU 2007; personal observations). In 2007, four Brandt's Cormorant (*Phalacrocorax penicillatus*) nests were seen on the cable-crossing structure near the Oakland Bay Bridge (Granholm 2007). Cormorants frequently rest on buoys in the Bay, and their common presence around piers and docks could raise the rate of predation on fish near those structures (Nightingale and Simenstad 2001). Decades ago, large numbers of Double-crested Cormorants roosted on a two-mile-long transmission line over the Richmond Channel, which was constructed in 1923 and removed some time after the early 1940s. In the early 1940s there were around 2,000-2,500 cormorants roosting on the line each night in the winter, and about 500 each night in the spring when many cormorants had departed for coastal breeding colonies (Bartholomew 1942, 1943).

Harbor Seals (*Phoca vitulina*) often haul out on breakwaters. Since 1989, several hundred California Sea Lions (*Zalophus californianus*) (with a maximum of around 1100 animals) have congregated in the winter on docks at Pier 39, on rare occasion joined by a Harbor Seal or Steller Sea Lion (*Eumetopias jubatus*). California Sea Lions often haul out on buoys and occasionally on other docks in the Bay (MMC 2007; personal observations).

Facilitating Invasions

It has frequently been observed that exotic organisms are more common on various artificial hard substrates than on natural hard substrates (Lambert 2002; Glasby et al. 2007; personal observations), though not all observers have found this when the substrates are exposed to similar physical parameters (Chris Brown, pers. comm.). Glasby et al. (2007) found that exotic species were more abundant and native species less abundant on floating structures and pilings compared to rocky reefs and sandstone seawalls, and that exotic species, especially colonial tunicates, recruited better to floating structures. They argue that artificial structures may thus facilitate the establishment or dispersal of exotic organisms in estuaries. In the Gulf of Maine, several exotic species that are common foulers of artificial structures apparently became established first in bays and estuaries where such structures are common, and subsequently spread to rocky reefs in the open waters of the Gulf (Harris and Mathieson 2000; Harris and Tyrell 2001; Bullard et al. 2007).

In the Bay, there are many exotic species that are dominant foulers of hard substrates. Thus the proliferation of artificial hard substrates in the Bay, especially in parts of the Bay where natural hard substrates are rare (*i.e.* most of the Bay outside of the western Central Bay), provides additional settlement opportunities for these exotic organisms, facilitating their spread and increasing their abundance within the Bay, and probably facilitating their eventual spread to other bays and estuaries along the coast (Chris Brown, pers. comm.). This is true regardless of whether or not artificial hard substrates favor exotic species compared to natural hard substrates.

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Sources, Mechanisms and Impacts of Changes in Nutrient Inputs to San Francisco Bay

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Background on the Effects of Nutrient Inputs

Nutrients are elements that organisms use for metabolism and growth. Macronutrients (carbon, oxygen, hydrogen, nitrogen and phosphorous, each typically constituting more than 1% of the dry weight of tissues; and sulfur, chlorine, potassium, sodium, calcium, magnesium, iron and copper, each typically constituting 0.2-1% of the dry weight of tissues) are the main components, while micronutrients (including aluminum, boron, bromine, chromium, cobalt, fluorine, gallium, iodine, manganese, molybdenum, selenium, strontium, tin, titanium, vanadium and zinc) are present in living tissues in smaller amounts (Pidwirny 2006). Silicon, which is a micronutrient for most organisms, is a macronutrient for diatoms. Nutrients occur in living organisms, in the wastes and dead organic matter derived from them, and as molecules in the environment. Chemically, they occur in both organic molecules (molecules with a carbon skeleton and oxygen and hydrogen atoms) and inorganic molecules. Inorganic nutrients are taken up by autotrophs (producer organisms, primarily algae and plants), and incorporated into living tissue, which may then be consumed by heterotrophs (animals). Nutrients are released from organisms as wastes or as dead tissue, broken down into detritus and transformed into one or more types of inorganic molecules by various bacteria or other decomposers. These inorganic molecules are available to be taken up again by autotrophs.

Concerns can arise when anthropogenic changes either deplete nutrient availability, restricting productivity, or increase nutrient supply, causing excessive growth of autotrophs. The latter has frequently been the case with nitrogen and phosphorous in aquatic ecosystems. Several human activities—including land clearing, the use of fertilizer, the discharge of human and animal wastes, and the burning of forests and fossil fuels—increased the flow of these nutrients into lakes, rivers and coastal waters (Cooper and Brush 1991). In many freshwater systems, loadings of nitrogen or phosphorous stimulated algal growth and increased the amount of organic matter settling to the bottom. Consequent increases in microbial decomposition sometimes depleted the oxygen in bottom waters, especially in stratified water bodies. This process of eutrophication in freshwater ecosystems became a major focus of limnological research, management and regulation starting in the 1960s (Cloern 2001; Howarth and Marino 2006).

Increased loadings of these nutrients into coastal waters has similarly sparked algal blooms, decomposition and oxygen depletion in bottom waters and sediments (Howarth 1988; Nixon 1995). Other effects can include reduced water transparency; declines in perennial seaweeds and sea grasses and the promotion of fast-growing, ephemeral seaweeds; increases in blooms of toxic dinoflagellates; changes in the diversity and abundance of benthic invertebrates; a shift to anaerobic metabolism, stimulation of sulfate reduction and production of metal-sulfides and hydrogen sulfide in the sediments; seasonal shifts in the timing of phytoplankton growth; and possibly a shift to smaller demersal fish species (Cloern 2001). Changes in the relative concentrations of nitrogen, phosphorus and silicon (a nutrient important in the growth of diatoms) can also change phytoplankton community composition or toxicity. For example, an increase in the ratio of nitrogen to silicon can favor flagellates and dinoflagellates over diatoms, and favor armored over naked silicoflagellates (Paerl 1997; Cloern 2001; Howarth and Marino 2006). An increase in the ratio of nitrogen to phosphorous may contribute to higher levels of toxicity in prymnesiophytes and *Pseudo-nitzschia* diatoms (Paerl 1997), while a decrease in the ratio of nitrogen to phosphorous can support noxious blooms of the flagellate *Phaeocystis* (Cloern 2001).

Overview of Nutrient Input Effects in San Francisco Bay

In San Francisco Bay, there have been occasional incidents of nuisance algal blooms, oxygen depletion, foul (hydrogen-sulfide) smells and/or fish kills (e.g. Horne and McCormick 1978; Nichols 1979; Luoma and Cloern 1982; Cloern and Oremland 1983; Josselyn and West 1985). Jassby (1992) noted past records of noxious accumulations of drift macroalgae in Alameda, decaying mats of the red drift alga *Polysiphonia* smothering benthic communities in the South Bay, dense accumulations of the green macroalgae *Ulva* and *Enteromorpha* in the Central Bay, and a pipe-clogging bloom of *Cladophora* in San Pablo Bay. Periodic *Ulva* and *Enteromorpha* blooms and decaying accumulations of washed-up *Polysiphonia* continue to occur on parts of the Bay shore (personal observations). Nutrient loadings from human activities may have caused or contributed to these incidents of rapid algal growth and high algal densities, though other environmental factors that affect nutrient availability or algal growth might also be responsible. Some incidents of oxygen depletion in the Bay may have resulted from the microbial decomposition of algal blooms stimulated by anthropogenic nutrient loadings, but the discharge of oxygen-demanding wastes (including both organic matter whose decomposition uses up oxygen, and reduced inorganic compounds that consume oxygen) may have caused or contributed to most incidents of hypoxia and ensuing nuisance odors and fish mortality.

Most of the time, light availability or benthic grazing appears to control algal growth in the Bay (Cloern 1979; Alpine and Cloern 1988; Cloern 1982; Nichols 1985; Jassby et al. 2002; Cloern et al. 2007). On most occasions when low nutrients do limit growth, nitrogen appears to be the limiting factor (Cloern 1979; Jassby et al. 2003), as it commonly is in most temperate zone estuaries (Ryther and Dunstan 1971; Howarth 1988; Oviatt et al. 1995; Howarth and Marino 2006). During phytoplankton blooms in the South Bay, silicon is sometimes depleted to levels that limit diatom growth (Hager and Schemel 1996).

Since the construction of secondary treatment facilities for municipal wastewater in the 1970s and 1980s, hypoxic occurrences have become rare in San Francisco Bay, even though nutrient levels in the Bay have generally remained high (Nichols et al. 1996). Unlike many temperate-zone estuaries, management concerns in the Bay have focused on the issue of low primary productivity and its impact on food webs, rather than on the stimulation of excessive primary productivity (Cloern 2001). There has thus been relatively little research on nutrient loadings and their impacts.

Two recent lines of inquiry have begun to change or at least modify this view of the Bay. Records of increasing phytoplankton densities in South, Central and San Pablo bays since the late 1990s (Cloern et al. 2006) have led to consideration of conditions under which the Bay's "eutrophication resistance" could be reduced and the Bay might begin to respond to nutrient inputs (Cloern et al. 2007). Meanwhile, other researchers argued that ammonia, normally considered a nutrient, also has an inhibitory effect that limits productivity in the Bay by limiting the uptake of nitrate; and that changes in wastewater treatment processes have affected ammonia inputs and productivity in the Bay (Wilkerson et al. 2006; Dugdale et al. 2007).

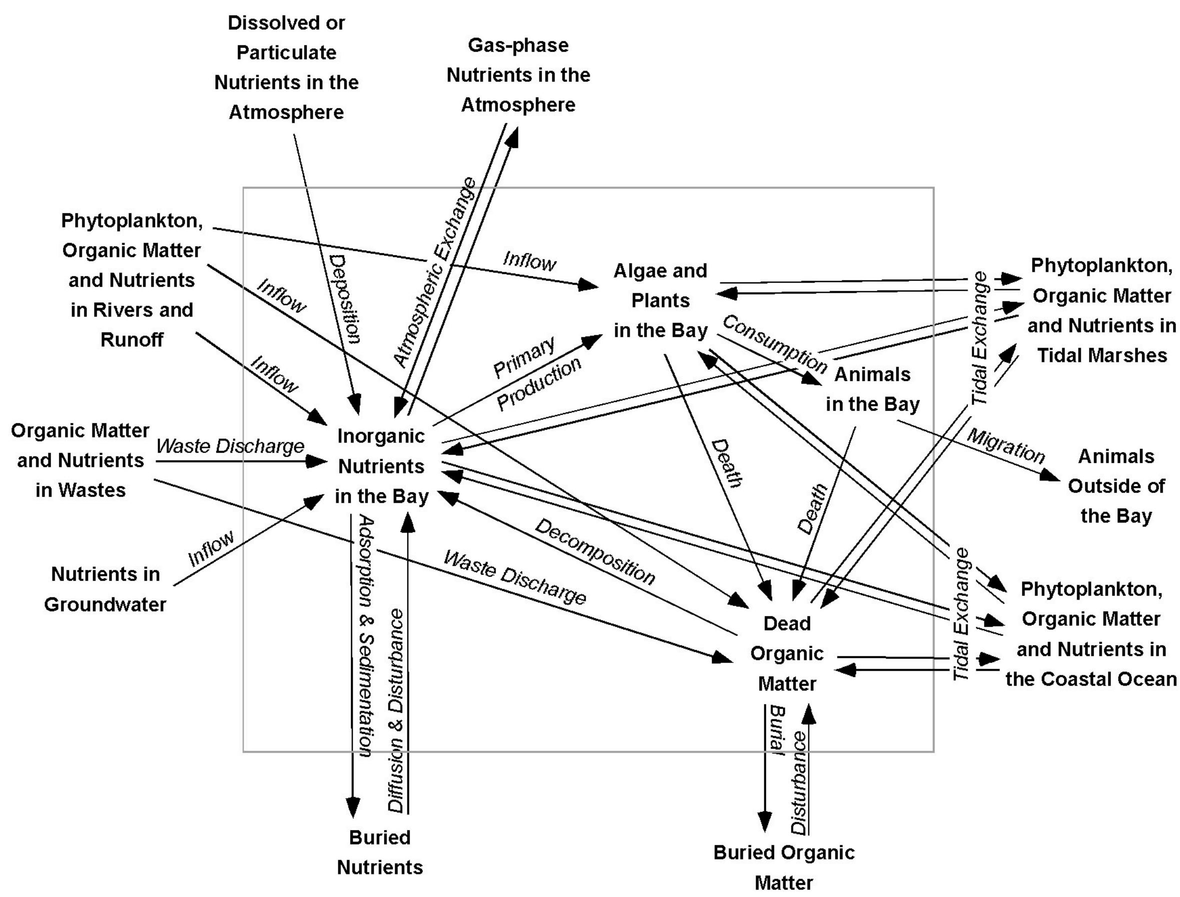
Nutrient Pathways

Nutrients can move in and out of San Francisco Bay by a variety of pathways (Fig. 1). Nutrients can be carried into the Bay with freshwater in rivers or runoff, or in groundwater; enter the Bay directly in waste streams; be deposited from the atmosphere in dissolved or particulate form; be exchanged with the atmosphere in gaseous form; leave the Bay by burial in sediments, and return to the Bay by upward diffusion or by disturbance of the sediments; be carried between the Bay and the ocean or marshes in tidal currents; and enter or leave with migrating animals. These pathways are discussed below. Conomos et al. (1979) identified the main sources of nutrients in the Bay as Delta outflow, the ocean, sewage discharge and drainage from tidal marshes, and the main sinks as the ocean, the bottom and possibly the marshes.

<u>Rivers and Runoff</u>
Nutrients can be carried into the Bay in flowing water as dissolved organic or inorganic nutrients, as inorganic nutrients adsorbed to sediment particles, as living organisms (primarily phytoplankton) and as dead organic matter. Changes in these inputs can come about through several mechanisms, including changes in the volume of flows into the Bay, changes in the concentration of suspended sediment and associated nutrients in the flows, changes in the production of phytoplankton in the Delta and other tributary waters, and changes in nutrient inputs to tributary waters in the form of treated sewage, applied fertilizers, other soil amendments, etc. For example, nearly 500,000 tons of nitrogen were applied to the land as fertilizer in California in 1975 (Peterson 1979) and the Bay watershed includes about 40% of California, suggesting that fertilizer could be the source of substantial nitrogen inputs to the Bay. Conomos et al. (1979) concluded, on the basis of Bay-wide concentration patterns, that Delta outflows and the ocean are the main sources of silica[17] and nitrate/nitrite for the northern reach of the Bay, and that loadings of all nutrients from the Delta are at least ten times higher in the winter than in

[17] Silicon doxide, the main form of inorganic silicon.

Figure 1. Nutrient Pathways in San Francisco Bay



the summer. Peterson (1979) estimated that outflow from the Delta provides 10,000 tons/yr each of dissolved inorganic nitrogen and particulate nitrogen and 100,000 tons of dissolved silica. Russell et al. (1982) estimated that inputs of total nitrogen from rivers and runoff had decreased from around 23,000 tons/yr in 1800 to 15,000 tons/yr in 1978, while inputs of total phosphorous decreased from around 4,000 to 3,000 tons/yr. Jassby and Cloern (2000) estimated the inflow to the Bay from the Delta of total organic nitrogen at 6,200 tons/yr. However, recent data show that nitrogen loading from the Sacramento River is now about three times what it was in the mid-1980s, due to population increases (David Dugdale pers. comm.).

Average water flows and peak flows are altered by freshwater storage and use, by increases in the portions of the watershed that are covered by hardened surfaces due to urban development, and by climate changes including changes in the timing, amount and type of precipitation, the amount of snow pack, the timing of snow melt and possibly the rate of evapotranspiration. Most observers believe that water storage and use has substantially decreased the flow of water into the Bay relative to pre-1850 conditions (e.g. Nichols et al. 1986). Peak flows have mostly been reduced by water storage,

although hardened surfaces may have increased peak flows in some local watersheds. In some areas, summer flows have been increased by the storage and release or the delivery of water for agricultural or domestic irrigation. Climate changes in the Bay/Delta watershed over the past several decades have included increases in the frequency and intensity of extreme rainfall events and a shift toward earlier snow melt and earlier runoff peaks (Dettinger et al. 1995; Lund et al. 2007). Over the coming decades anthropogenic climate change is projected to continue these trends and to increase the interannual variability in precipitation, increase the number of large winter storms, and hasten and compress the period of snowmelt, with associated increases in peak winter runoff events (Lund et al. 2007). These changes may make it harder for dams to retain and store as large a fraction of the runoff as they do currently, due to constraints imposed by flood control operational requirements. Any increase in the evapotranspiration rate in the watershed, due to increased temperatures (projected to increase by 1-3° C by 2030 and by 2-10° C by 2100—Lund et al 2007) and higher plant growth rates in a hotter and more CO_2-enriched environment, would tend to decrease the amount of inflow to the Bay. The net effect of all this, as mediated by human responses, on the timing and amount of inflows to the Bay is unclear.
In the last half of the 19th century, sediment loadings and suspended sediment concentrations in Bay tributaries were increased by land clearing and mining activities (Gilbert 1917; Krone 1979). Flood control levees constructed along these watercourses reduced overbank flooding and the deposition of sediment on floodplains, which further augmented the delivery of suspended sediments downstream. In the 20th century, especially after the early 1940s, extensive dam construction on these tributaries lead to the settling and retention of sediment in impoundments behind the dams, reducing the concentrations of suspended sediment downstream of the dams (Krone 1979). Water diversions, also increasing more rapidly since the early 1940s, divert sediments and associated nutrients, decreasing the total loadings to the Bay (Krone 1979).

Phytoplankton growth in the Delta decreases the amount of inorganic nutrients and increases the amount of phytoplankton in Delta outflows. Like the Bay, the Delta is nutrient-rich and light-limited (Jassby and Cloern 2000; Jassby et al. 2002), with phytoplankton productivity also limited by benthic grazing (Jassby et al. 2002; Lund et al. 2007) and possibly herbicides (Edmunds 1999; Jassby et al. 2003), though the evidence for herbicide limitation is weak (Alan Jassby pers. comm.). On the other hand, total suspended solids has declined and water transparency has been increasing, probably because of dams built upstream (Jassby et al. 2002). Nutrients are normally present in substantial excess, because of wastewater effluent from Sacramento (Davis Dugdale, Alan Jassby pers. comm.) and agricultural drainage (Jassby et al. 2002, 2003). In a review of nutrient concentrations in 1968-1998, only 0.1% of dissolved inorganic nitrogen measurements, 0.15% of phosphate measurements, and none of the silica measurements in the Delta were at apparently limiting levels (with n > 8,000, 6,000 and 8,000, respectively—Jassby et al. 2002). Despite abundant nutrients and increasing light penetration, phytoplankton productivity in the Delta declined since the 1970s (Jassby et al. 2002; Cloern et al. 2006), but recently has more-or-less recovered (Alan Jassby pers. comm.). Changes in water diversions and storage, in precipitation and runoff patterns (resulting from anthropogenic climate change), in the topography of the Delta, in agricultural practices in tributary areas (including the types and application rates of fertilizers and herbicides[18]; irrigation and drainage methods; crop types; and the amount and location of land in production), and in other land use practices (the extent of urbanization) could affect phytoplankton growth in the Delta.

Jassby and Cloern (2000) concluded that river inflow is the main source of organic matter input (and associated organic nutrients) to the Delta, followed by autochthonous phytoplankton production and agricultural drainage from Delta islands; wastewater discharges, tidal marshes and other sources were of less importance. They found that because of water project exports from the Delta, the downstream export of organic matter to the Bay is nearly always less than the riverine inputs to the Delta, especially in dry years (export to Bay ranging from 20% of riverine inputs in the summer to 55% of riverine inputs in the winter, in critically dry years). The volume of water project exports thus has a large influence on the inflow of nutrients to the Bay. They considered the potential impact of other major actions on nutrient flows to the Bay. The construction of an isolated diversion facility (such as the Peripheral Canal) to shunt water from the Sacramento River to the water project pumps, and the use of flow barriers to restrict the flow of organic-matter-rich San Joaquin River water to the project pumps, would both tend to increase the annual flow of organic nutrients into the Bay, but the effect would be weakest in spring and summer when these resources are most likely to be used by biota. Increasing shallow water habitat by either flooding Delta islands or by inundating floodplain areas for longer or more frequent periods would increase total primary productivity (including both phytoplankton and benthic vegetation) and organic matter in the Delta. For floodplain inundation at least, this would probably not significantly increase export to the Bay (Jassby and Cloern 2000), but any phytoplankton biomass produced would likely be more bioavailable than riverine inputs of organic matter (Sobczak et al. 2002).

Agricultural drainage and runoff from lawns and golf courses can carry fertilizers and soil nutrients into the Bay and its tributary waters. Waste from domestic animals is carried in runoff from feedlots, and at times this can account for much of the agricultural loading of nitrogen into the San Joaquin River (Alan Jassby pers. comm.). Nutrient loadings from Central Valley farms increased with the increasingly widespread application of fertilizers after the late 1940s and with later increases in subsurface drainage, such that nitrate concentrations in the San Joaquin River increased fivefold between 1950 and 1980 (Nichols et al. 1996). Direct discharges of municipal wastewater into the Bay are discussed below, but wastewater discharges into tributary waters can contribute substantially to the nutrient loads in Bay inflows (Alan Jassby pers. comm.).

<u>Groundwater</u>

Nitrogen inputs in groundwater can range from <10% to >30% of total nitrogen inputs into coastal waters, and in some cases may be comparable to riverine inputs (Paerl 1997). Globally, oceanic inputs from groundwater are comparable to inputs from

[18] Including a rapid shift from ammonia and nitrate fertilizers to urea fertilizers over the past decade (David Dugdale pers. comm.).

biological nitrogen fixation, and about one-third to one-sixth of inputs from rivers and runoff. There is little information on the volume of groundwater inflows to San Francisco Bay or on the nutrient concentrations in those inflows (Jassby 1992). The enrichment of nitrogen in groundwater occurs mainly in agricultural areas, due to fertilizer applications and accumulation and storage of animal wastes.

<u>Waste Discharge</u>
Wastewater disposal can discharge organic matter and organic and inorganic nutrients into the Bay. Following recurrent water quality problems in the Bay in the 1950s and 1960s, especially in parts of the South Bay, municipal treatment plants were upgraded in the 1970s and 1980s to provide at least secondary treatment (Russell et al. 1982). Secondary treatment is primarily designed to reduce suspended solids and biochemical oxygen demand (BOD) by 85-90%, but typically reduces nutrients by 50% or less; most non-particulate nutrients pass through the treatment system (Russell et al. 1982). Periodic problems with oxygen depletion in the far South Bay were resolved with further reductions in the oxygen demand of wastewater plant effluent, by adding advanced secondary treatment processes that discharge nitrate instead of ammonia (David Dugdale pers. comm.). Conomos et al. (1979) estimated that wastewater discharges to the Bay in 1975 carried 6,000 tons of organic nitrogen, 14,000 tons of inorganic nitrogen (mainly as ammonia) and 10,000 tons/yr of total phosphorous (similar estimates are provided by Russell et al. 1982; Peterson 1979 estimated nitrogen inputs to the northern reach of the Bay at 8,000 tons/yr). About 70% of these nutrients were discharged into the South Bay, with about 20-40% of the total going into the southern end of the South Bay below the Dumbarton Bridge. Conomos et al. (1979) concluded, on the basis of Bay-wide concentration patterns, that wastewater entering at the south end of the Bay was the main source of nitrogen and phosphate for the South Bay, and that wastewater was a significant source of silica and ammonia elsewhere in San Francisco Bay. Most of the current nutrient input to the North Bay is from wastewater (David Dugdale pers. comm.).

<u>Atmospheric Deposition and Exchange</u>
Gunther et al. (1987) estimated that the deposition of airborne substances directly into the Bay could be responsible for minor but not insignificant loads of certain contaminants. The burning of forests and fossil fuels (especially the production of nitrogen oxides (NO_x) in internal combustion engines), the industrial production and use of synthetic fertilizer, and animal wastes stored in open lagoons or applied as manures (which can lose up to 70-80% of their ammonia through volatilization—Paerl 1997) all release nitrogen compounds to the atmosphere that can return to the earth in precipitation or dry deposition. In addition, atmospheric nitrogen is oxidized (fixed) by the heat of lightning strikes to form nitric acid, which washes out of the atmosphere dissolved in rain. Dissolved organic nitrogen can also form a substantial fraction of the nitrogen in atmospheric deposition (Paerl 1997). Paerl (1985) reported that rainfall events in nitrogen-limited waters off North Carolina were followed by increased phytoplankton growth. The largest increases occurred after acidic rainfall derived from continental regions, with less stimulation of growth after falls of rain with near-neutral pH derived from oceanic regions, which Paerl attributed to the elevated levels of nitrogen

compounds in acid rain. Paerl (1997) estimated that overall about 0.3 to >1 g/m^2-yr of nitrogen is deposited from the atmosphere into coastal waters, accounting for 20-40% of "new" nitrogen inputs. Rainfall in the San Francisco Bay area is mostly ocean-derived and not notably acidified (about 10 µmol/L of nitrate in northern California—Peterson et al. 1985). Russell et al. (1982) estimated that in 1978 atmospheric deposition was putting 980 tons/yr of total nitrogen into the Bay (about 6% of the inputs in rivers and runoff and 5% of the inputs in wastewater), along with 120 tons/yr of total phosphorous, which works out to about 0.6 g/m^2-yr of nitrogen and 0.07 g/m^2-yr of phosphorous.[19]

Nitrogen gas (N_2) in the atmosphere is also fixed by certain bacteria and cyanobacteria (blue-green algae) to form nitrate (NO_3) and ammonium (NH_4^+). These are then taken up and utilized by plants and algae. Other groups of bacteria decompose dead plant and animal matter and animal wastes back to ammonia, nitrite (NO_2) and nitrate. Denitrifying bacteria convert nitrate into nitrogen gas or nitrous oxide gas (N_2O), which diffuse back into the atmosphere. While nitrogen-fixing bacteria are common in soil, and blooms of planktonic, nitrogen-fixing cyanobacteria are typical of nitrogen-deficient lakes (Flett et al. 1980), nitrogen-fixing is apparently less important in estuaries and coastal waters (Cooper 1937; Howarth 1988), though recent findings suggest it may be more important in the ocean generally than was previously thought (Arrigo 2005). Planktonic nitrogen-fixing cyanobacteria are uncommon or rare in most estuaries where salinities are above 10-12 ppt (Ryther and Dunstan 1971; Howarth 1988; Howarth and Marino 2006). Nitrogen-fixing by benthic cyanobacteria and cyanobacteria that are epiphytic on sea grasses may be significant in coastal waters where sufficient light penetrates to the bottom, but this excludes the major part of most temperate zone estuaries (including San Francisco Bay), except on intertidal mudflats (Howarth 1988; Howarth and Marino 2006). Denitrification rates in lakes and marine waters appear to be similar (Howarth 1988; Howarth and Marino 2006), with the released gases reaching the atmosphere through mixing and diffusion. Rates of nitrogen fixing and denitrification might be affected by changes in the abundance and composition of bacteria and cyanobacteria, or by changes in temperature or other environmental conditions that affect these organisms.

Burial, disturbance and diffusion

Dead organic matter and nutrients in the water column or in surface sediments are available for microbial transformation or uptake by biota. Organic matter and nutrients buried in sediments below the zone of biological activity are unavailable for use by biota, but the depth at which this occurs is not clear. Various organisms may turn over or irrigate the sediment to depths of a few centimeters (snails, sea slugs, juvenile clams), to 10-30 cm (lugworms, deeper burrowing clams), or up to 50-75 cm (bat rays, various polychaetes, and ghost shrimp) (Rubin and McCulloch 1979; Peterson 1979; Haderlie and Abbott 1980; Haig and Abbott 1980; Cohen 2007). In some areas, sediment turnover resulting from tidal currents or wind waves (Krone 1979; Conomos et al. 1979; Nichols 1979) may be more significant than disturbance by animals. Over most of the Central and South bays physical turnover affects only the upper 2-5 cm of the

[19] Assuming that Russell et al.'s loadings are spread over 371,000 acres estimated for the Bay at MHHW (Jassby 1992).

sediment, but might reach depths of 40-100 cm at sites in the Central Bay with sandy sediments and fast tidal currents where sand waves form on the bottom (Rubin and McCulloch 1979; Hammond and Fuller 1979). Similar beds of sand waves occur in San Pablo Bay (Nichols 1979]). Various activities that disturb the bottom may increase the background rate of physical turnover, including dredging, sand or shell mining, and bottom trawling conducted for fisheries harvest, research or educational purposes (Cohen 2007). Nutrients may also pass out of the sediments by molecular diffusion through interstitial water, but this is minor relative to the fluxes due to turnover and irrigation (Hammond and Fuller 1979), and in practice extends only a few centimeters deep. The overall fluxes of nutrients out of bay sediments were estimated at about 6 g/m^2-yr of nitrogen, 1 g/m^2-yr of phosphorous and 60 g/m^2-yr of silicon (Hammond et al. 1985). Peterson (1979) estimated fluxes from sediments in the northern reach to be about 10,000 tons/yr of nitrogen and 30,000 tons/yr of silicon.

Estimates of the portion of the organic matter supply that is lost by burial in an estuary range from about 1% to over 10% (Paerl 1997); Jassby et al. (2002) estimated that on average around 20% is lost in shallow water systems. Human activities that increase sediment inputs and sedimentation rates in the Bay can increase the rate of nutrient burial and loss from the Bay system. Alternately, activities that reduce sediment inputs and promote erosion in the Bay may expose these nutrients. Gilbert (1917), Smith (1965), Krone (1979, 1996), Atwater et al. (1979, Fig. 6), Capiella et al. (1999), Jaffe et al. (1998), Foxgrover et al. (2004) and Jaffe and Foxgrover (2006a,b) review and summarize changes in sedimentation rates in the Bay. Substantial increases in sediment production in the watershed resulted from hydraulic mining and agricultural activities in the late 1800s, with significant lags in the timing of sediment arrival in the Bay (Gilbert 1917, Krone 1979). The construction of dams and impoundments, and diversions of fresh water primarily for irrigation, subsequently reduced the delivery of sediments to the Bay (Krone 1979). In addition to overall changes in sediment inputs, dredging, sand and shell mining locally remove sediments and can expose buried nutrients, while the disposal of dredge sediments can bury nutrients locally.

The patterns of sedimentation and erosion in the Bay have been complex with some areas accumulating sediment even as nearby areas were losing it (e.g. Atwater et al. 1979; Capiella et al. 1999; Jaffe et al. 1998; Foxgrover et al. 2004). Capiella et al. (1999) reported that Suisun Bay gained 61 million m^3 of sediment between 1867 and 1887, then lost 159 million m^3 by 1990. Earlier researchers, analyzing essentially the same hydrographic survey data, came to somewhat different conclusions: Smith (1965), reported a similar pattern of sediment gain followed by greater sediment loss but involving half or one-third as much sediment; while Krone (1979, 1996) also reported that there was a large gain of sediment in the late 1800s, but that it was followed by a modest overall gain through 1990 once sea level rise was taken into account. In San Pablo Bay, both Jaffe et al. (1998) and earlier researchers (Gilbert 1917; Smith 1965; Krone 1979, 1996) reported a very large accumulation of sediment in the last half of the 19th century (range of estimates of 252-294 million m^3), followed by lesser but still substantial accumulation in the twentieth century, though the amounts reported differ, especially for the period after 1922. In Central Bay, while Gilbert (1917) reported a gain of 106 million m^3 in the late 1800s, Smith (1965) and Krone (1979) reported only a slight loss or a slight gain; for 1897-1990, Krone (1976, 1996) reported a gain of over 200 million m^3, and Smith's (1965) calculations are reasonably consistent with this. There was less agreement on sediment changes in South Bay. Gilbert (1917) reported a net gain of over 40 million m^3 in the late 1800s, Smith (1965) reported a net loss of over 40 million m^3, and Krone (1979) and Foxgrover et al. (2004) reported only small losses or gains. Authors (Smith 1965; Krone 1979; Foxgrover et al. 2004) agree that the South Bay lost sediment in the first half of the 20th century, but the range of loss estimates is from 25 to 90 million m^3. Between the 1950s and around 1990, Krone (1996) reported a gain of 24 million m^3, but Foxgrover et al. (2004) reported a loss of 71 million m^3.

Activities that erode the margins of the Bay also release sediments and nutrients into the Bay. Atwater et al. (1979, Fig. 6) summarized tidal marsh shoreline changes, depicting a complex pattern of shoreline advance and retreat that is not easily interpreted. They suggest that sites of shoreline retreat may be due to local rise in relative sea level (resulting from a combination of eustatic and tectonic effects) and/or burrowing by the exotic isopod *Sphaeroma quoiana*.

Tidal Exchange

Tidal marshes generally act as net exporters of organic matter and nutrients to the open waters of estuaries, though there are exceptions (Nixon 1980; Jassby 1992; Jassby et al. 1993). Much of the export from marshes may be in the form of detritus derived from marsh plants, while imports may occur from the trapping of sediment-associated nutrients and benthic filtering of open-water phytoplankton (Nixon 1980). It is estimated that diking and filling has reduced the Bay's tidal marsh to about one-fifth of its 1850 area (Goals Report 1999), and to about one-eighteenth of its former area for the Bay and Delta combined (Nichols et al. 1996). The Baylands Ecosystem Goals Project has recommended that tidal marsh in the Bay be increased from 40,000 acres in 1998 to 90,000-105,000 acres (Goals Report 1999). As most Bay tidal marshes will probably serve as net sources of organic matter and nutrients, the input from marshes is expected to increase substantially from current levels if these restoration plans are implemented. Marsh restoration that involves cutting through existing levees may also initially increase inputs of sediment and nutrients by eroding new channels.

The average amount of water entering and exiting the Bay on each tide cycle (the tidal prism) is about one-quarter of the Bay's volume at MLLW. Since most of this water just moves back and forth, the amount of water replaced with new water on each tide cycle is only about a quarter of the Bay's tidal prism, or about 6% of its low-tide volume (Russell et al. 1982). Still, this is about ten times the average amount of freshwater inflow during the same period, and with nearly two complete tide cycles a day adds up to a lot of water exchanged. When nutrient or organic matter concentrations or phytoplankton populations inside and outside of the Golden Gate differ significantly, the large volumes of water exchanged can have a substantial impact on Bay conditions. During spring-summer upwelling periods, northerly winds cause upwelling along the coast. Nutrients brought up with deep water stimulate blooms of large diatoms, which are carried into the Bay (Conomos et al. 1979; Cloern 1979). Conomos et al. (1979)

concluded, on the basis of Bay-wide concentration patterns, that the ocean is a moderate source of phosphate and nitrogen for the Bay. Peterson (1975) notes the difficulty of determining even the direction of net exchange at the Golden Gate. Noting that silica concentrations are generally a good deal higher at the surface than at depth at the Golden Gate, and that due to gravitational circulation the net flux of water is oceanward in the upper part of the water column and landward at the bottom, he estimates that there is net export of silica from the Bay that is large but probably less than riverine inputs. Similarly, he estimates that ammonia is exported, that the direction of nitrate flux varies, and that there is probably a net loss of nitrogen from the Bay. A recent study of nutrient gradients at the Golden Gate concluded that there is always a net export of silica, and usually a net export of nitrate (David Dugdale pers. comm. citing Martin et al. 2007). By altering sea level and changing the Bay's tidal prism, or by changing coastal upwelling patterns (through changes in ocean heating and winds), climate change would change the flux of phytoplankton, nutrients and organic matter between the Bay and the coastal ocean.

Migration

The active migration of animals can contribute to net fluxes of nutrients if the animals feed and grow in large numbers in one site and then spawn, die or deposit wastes in another. Well-known examples include Pacific species of salmon feeding and growing in the ocean then bringing nutrients back to their natal streams when they return to spawn and die; and fish-eating seabirds harvesting phosphorous from the sea and delivering it onto land sites as phosphate-rich guano deposits. For estuaries, the effect of such biotic transport is usually a net export of nutrients (Jassby 1992). Examples in the Bay would include Dungeness crab (*Cancer magister*), which enter the Bay as late-stage larvae or post-larval instars, feed and grow for about a year, and then remove nutrients from the Bay when they migrate out to the Gulf of the Farallones and coastal waters; and the approximately one million migratory shorebirds that winter on the Bay, building up fat stores by feeding on mudflat invertebrates when the tide is out, and removing nutrients from the Bay when they excrete wastes in marsh and upland resting areas when the tide is in, and migrate northward in the spring to their nesting areas. Striped bass (*Morone saxatilis*), which feed and grow in the Bay but spawn upstream, and northern anchovy (*Engraulis mordax*), which also feed and grow in the bay but may spawn primarily in coastal waters, may also result in net losses of nutrients (Jassby 1992). Pacific herring (*Clupea harengus*), which enter the Bay in the winter to spawn, may represent either a gain or loss depending on whether the consumption and loss of herring eggs within the Bay outweighs the out-migration of the surviving young nine months later. If there's a net gain it would have to less than the nutrients in the annual spawn of eggs, which is estimated to contain about 500 tons of nitrogen (Jassby 1992), assuming a C:N ratio of 4—Pilanti and Vanni 2007).

While migration overall probably results in a net export of nutrients from the Bay, there are few quantitative data. However, in developing a carbon model for the Bay Jassby (1992) judged that these were insignificant relative to other flows, and this is probably true for other nutrients as well. Thus even large anthropogenic impacts on these migrations would probably have little effect on overall nutrient flows.

The Bay's Response to Nutrient Inputs

Compared to many other estuaries, nutrient concentrations in the San Francisco Bay system are relatively high, but its productivity is low (Cloern 2001; Cloern et al. 2006 Jassby 2008). Jassby et al. (2002) reported that relative to 14 other estuaries ranging in productivity from 11 to 560 g C/m^2-yr (Underwood and Kromkamp 1999), San Francisco Bay came in at sixth lowest with productivity of 120 g C/m^2-yr (Jassby et al. 2002), with the Delta ranking even lower with productivity of 70-75 g C/m^2-yr. The Bay and Delta had similarly low rankings relative to a review of 25 river-dominated estuaries (Boynton et al. 1982; Jassby et al. 2002). Cloern (2001) reported that although San Francisco Bay has as much dissolved inorganic nitrogen and 10 times as much dissolved inorganic phosphorous as Chesapeake Bay, and higher annual loadings per square meter of both nitrogen and phosphorous, it has only one-fifth as much phytoplankton biomass and one-twentieth as much primary productivity[20], and unlike Chesapeake Bay, its bottom waters are not commonly hypoxic in the summer. Since the mid-1990s, average productivity in South, Central and San Pablo bays has increased by nearly 80%, from 120 g C/m^2-yr in 1993-96 to 215 g C/m^2-yr in 2001-2004, while nitrogen and phosphate concentrations were declining as a result of reduced loadings from improved wastewater treatment (Cloern et al. 2006; Cloern et al. 2007). San Francisco Bay overall appears to have a much weaker response to changes in nutrient inputs than many other estuaries (Nichols et al. 1996; Cloern 2001). Conomos et al. (1979) stated that San Francisco Bay is naturally nutrient rich and that this may hide the effects of added nutrients, but non-nutrient factors limiting algal growth (turbidity, benthic grazing) provide a better explanation (Alan Jassby pers. comm.).

In the 19th and 20th centuries, municipal waste discharges into the Bay increased with population growth. By 1950 anaerobic conditions were common along the eastern and southern shores of the Bay and these continued to occur, along with fish kills and other water quality problems, until the construction of secondary and tertiary treatment facilities starting in the 1970s (Russell et al. 1982; Cloern and Oremland 1983; Nichols et al. 1996). Between the 1960s and the 1970s, fish kills became rarer in the Bay; BOD declined and oxygen levels improved in the South Bay, particularly at its southern end (Luoma and Cloern 1982; Nichols et al. 1996). It's unclear whether some of the earlier low oxygen episodes may have resulted from the decomposition of algal blooms stimulated by inorganic nutrient inputs, or if all were due to the discharge of incompletely decomposed organic matter. The latter was apparently the case in 1979 when partially-treated sewage was discharged from the San Jose-Santa Clara Waste

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[20] This is based on comparing an estimate for Chesapeake Bay productivity (>400 g C/m^2-yr) to productivity in Suisun Bay in 1988 (20 g C/m^2-yr) after invasion by *Corbula amurensis*; in 1980, before *Corbula*, productivity in Suisun Bay was 100 g C/m^2-yr. Estimated productivity in Suisun Bay in 1977-1990 was 106 g C/m^2-yr when benthic grazers were scarce (pre-*Corbula* invasion) and 39 g C/m^2-yr when benthic grazers were abundant (mostly post-*Corbula* invasion) (Alpine and Cloern 1992). Cole and Cloern (1984) estimated net photic zone productivity at 93 to 150 g C/m^2-yr at six shallow and deep sites in Suisun Bay, San Pablo Bay and South Bay in 1980-81; net water column productivity was lower, or even negative (losses from respiration exceeded gains from photosynthesis), for the deep sites (-130 to 70 g C/m^2-yr) than for the shallow sites (56 to 131 g C/m^2-yr) Cloern et al. (2006).

Treatment Plant into Coyote Creek in the South Bay. During the three-week spill, dissolved oxygen was severely depressed in the creek and fish and pelagic invertebrates were absent. Phytoplankton biomass was also low. The plant effluent received only primary treatment during this time and contained twenty times its normal concentration of organic matter. The effects of the spill did not extend into the South Bay proper; "in effect, Coyote Creek operated as a sewage treatment plant." Once the real treatment plant resumed normal operations (including secondary and tertiary treatment of effluent), phytoplankton biomass increased and oxygen levels recovered (Cloern and Oremland 1983). The addition of advanced secondary treatment, which discharges nitrate instead of ammonia, has further reduced the oxygen demand from this treatment plant (David Dugdale pers. comm.).

In general, phytoplankton growth is thought to be limited in the Bay by high turbidity and low light availability[21] (Cloern 1979; Alpine and Cloern 1988; Jassby et al. 2002) mediated by the location and depth of phytoplankton stocks (the photic zone typically extends to about 10% of the water depth in the main channels and to 50-100% of the depth in the shallows—Cole and Cloern 1984), or limited by grazing by benthic organisms (primarily Asian and Atlantic species of clams) (Cloern 1982; Nichols 1985; Alpine and Cloern 1992). Most of the time, nutrient levels are more than high enough to support phytoplankton growth in all parts of the Bay. Dissolved phosphate always and silica nearly always exceeds growth-limiting concentrations (Cloern 1979; Conomos et al. 1979; Peterson 1979). Inorganic nitrogen, however, can sometimes be depleted to the point where it becomes limiting in the northern part of San Francisco Bay by late summer or fall (Peterson 1979; Cloern 1979; Peterson et al. 1985), and nitrogen sometimes becomes limiting during spring phytoplankton blooms in the South Bay (Jassby et al. 2003). Silica did drop to apparently limiting concentrations in the northern part of the Bay during a rare period of very low river flows and high air temperatures in July 1961 (Peterson 1979; Peterson et al. 1985). Thus, while light availability and grazing intensity may control the frequency, location and seasonality of bloom events, nutrient uptake rates during blooms that exceed nutrient regeneration rates may lead to nutrient depletion that controls the size of some bloom events (Cole and Cloern 1987).

Recently, Dugdale and colleagues have argued that high concentrations of ammonia in the Bay inhibit nitrate uptake, thus limiting productivity even when nitrate levels are high, and that blooms occur only when ammonia is first reduced to very low concentrations by dilution from large freshwater inflows and/or uptake by phytoplankton (Wilkerson et al. 2006; Dugdale et al. 2007). Dugdale et al. (2007) further suggest that the installation of secondary treatment systems in wastewater treatment plants in the late 1970s and early 1980s, which converted organic nitrogen to ammonia and increased ammonia loadings in wastewater discharges, increased ammonia concentrations in the Bay which suppressed nitrate uptake and contributed to a long-term decline in productivity. Conversely, advanced secondary treatment processes convert ammonia to nitrate, thereby reducing ammonia loadings, allowing nitrate uptake and increasing productivity.

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[21] While phytoplankton themselves contribute to light attenuation, the effect is generally small relative to that of other suspended particles (Cole and Cloern 1987).

Observations since the late 1990s of increases in phytoplankton biomass and changes in the timing of phytoplankton growth in South, Central and San Pablo bays suggest that the Bay may be starting to respond to its high nutrient concentrations (Cloern et al. 2006; Cloern et al. 2007). These observations include a progressive, significant increase in the baseline or minimum phytoplankton biomass, increases in the largest spring blooms, and blooms occurring during the previously bloom-less period of autumn and winter. Primary productivity increased by 75%. However, increased nutrients could not be the cause of these biomass and productivity increases, because during this time nitrogen and phosphorous concentrations were stable or weakly declining in the Bay, consistent with reductions in these nutrients in wastewater effluent (Cloern et al. 2006; Cloern et al. 2007). Rather, the increase in phytoplankton may have been caused by coastal oceanographic changes that increased the populations of some benthivore species that migrate into the Bay for parts of their life cycles, thereby triggering a top-down trophic cascade that reduced populations of filter-feeding bivalves and increased phytoplankton densities, as described in more detail below (Cloern et al. 2007).

Phytoplankton dynamics and productivity have been most extensively studied in Suisun Bay and South Bay.

<u>Suisun Bay</u>

In Suisun Bay phytoplankton densities are low in winter and spring (and dominated by freshwater diatoms—Cole and Cloern 1984) when high river flows reduce the retention time in the embayment to days or weeks, which is comparable to or shorter than the time needed for phytoplankton populations to increase (doubling time of weeks to months (Alpine and Cloern (1992), or days to weeks (David Dugdale pers. comm.)); phytoplankton are thus washed downstream as fast or faster than they can reproduce, and the population cannot build up (Alpine and Cloern 1992). In addition, insolation and water temperature are low, reducing phytoplankton growth (Conomos et al. 1979). In most years before 1987, as flows subsided phytoplankton populations slowly increased over 2-3 months to large summer peaks dominated by large coastal/brackish diatoms (Cloern 1979; Alpine and Cloern 1992; Cole and Cloern 1984), achieving densities that were typically much higher than the annual phytoplankton peaks elsewhere in the Bay. The Suisun Bay peak coincided with the development of a zone of high turbidity in Suisun Bay, thought to be controlled by gravitational circulation in the channel (Cloern 1979; Arthur and Ball 1980; Alpine and Cloern 1992).

In the summer of 1977, however, in the second year of a severe drought, there was no summer phytoplankton bloom. Two explanations were proposed. The first was that phytoplankton in the channel were trapped along with other particles in a null zone created by a gravitational circulation cell, which formed an observed zone of maximum turbidity within Suisun Bay and was often closely associated with the location of maximum netplankton (plankton >22 microns in size) and chlorophyll (Arthur and Ball 1980; Jassby et al. 1996). In most years this was located in Suisun Bay in the summer. The phytoplankton were then advected out over the broad adjoining shallows, where light penetration was sufficient for rapid phytoplankton growth. The high phytoplankton densities in Suisun Bay may also have been due in part to the trapping of exogenous phytoplankton by gravitational circulation, rather than *in situ* growth (Cole and Cloern 1984). In drought years with reduced freshwater inflows, the null zone moved upstream to the narrower and more uniformly deep waters of the Sacramento River, phytoplankton spent more time in deeper water where there was inadequate light for photosynthesis, so that growth was inhibited and the population never grew to a significant peak (Arthur and Ball 1980). The alternate explanation was that populations of filter-feeding organisms that preferred higher concentrations of salinity than were typically found in Suisun Bay, especially the Atlantic clam *Mya arenaria*, increased in Suisun Bay during the two-year drought, and by 1977 were abundant enough to consume phytoplankton as fast as they could reproduce (Nichols 1985). The relative contribution of these two mechanisms—null-zone relocation and benthic grazing—to Suisun Bay phytoplankton dynamics prior to 1987 was never disentangled, though Nichols (1985) opined that they were "certainly additive." In addition, summer measurements of currents in Suisun Bay in the 1990s often failed to show the presence of a gravitational circulation cell, and the simple picture of a particle and phytoplankton entrapment zone moving up and down the estuary in response to changes in flows, no longer seems to hold (Jassby et al. 1996).

Beginning in 1987, the filter-feeding Asian clam *Corbula amurensis* became abundant in Suisun Bay. Since then, phytoplankton densities have remained low through the summer in most years, with many observers concluding that benthic grazing is now the primary control on summer phytoplankton growth (Alpine and Cloern 1992; Jassby 2008). Annual primary productivity in Suisun Bay declined substantially (to 20 g C/m^2-yr in 1988 compared to 100 g C/m^2-yr in 1980—Alpine and Cloern 1992) along with phytoplankton biomass (Cloern et al. 2006; Dugdale et al. 2007). Dugdale et al. (2007), however, recently argued that the phytoplankton decline started in the decade before the first records of *C. amurensis* and was probably caused by increased ammonia in wastewater treatment plant discharges resulting from the adoption of secondary treatment processes. Ammonia discharged into the Sacramento and San Joaquin rivers from wastewater plant discharges increased over 1985-2005, and ammonia concentrations in the Delta and Suisun Bay rose in 1996-2005 (Jassby 2008). In Dugdale et al.'s (2007) view, the impact of *C. amurensis* was not that it ate up phytoplankton faster than the phytoplankton could reproduce, but rather that it maintained the inhibition of nitrate uptake by keeping phytoplankton populations so small that they couldn't deplete ammonia, and by excreting wastes that added ammonia to the water.

After declining for two decades (Jassby et al. 2002; Dugdale et al. 2007; Jassby 2008), there was no upward or downward trend in the (low) phytoplankton densities in Suisun Bay in 1996-2005 (Jassby 2008). During this period, nutrient concentrations (dissolved inorganic nitrogen, soluble reactive phosphorous, and silica) were high enough to not limit growth (Jassby 2008). In the Delta, phytoplankton productivity and density increased over this period, and thus phytoplankton carried into Suisun Bay from the Delta must account for a larger component of Suisun Bay's phytoplankton than they did in prior decades (Jassby 2008). Two spring blooms were recorded between 2000 and

2003, a larger one in 2000 fueled primarily by nitrate uptake, and a smaller one in 2003 fueled by ammonia uptake (Wilkerson et al. 2006; Dugdale et al. 2007).

San Pablo Bay

Cloern (1979) reported that phytoplankton peaked in San Pablo Bay in the spring, with large increases in the population of a coastal diatom, *Skeletonema costatum*. He interpreted these dynamics as resulting from *Skeletonema* proliferating in waters outside the Golden Gate as a result of nutrient enrichment due to upwelling, being advected into San Pablo Bay in the bottom layer of two-layered gravitational flow, trapped in the region of the null zone which is often located near San Pablo Bay in the spring, and then dispersed over the San Pablo Bay shallows, where enough light penetrated throughout the slight depth to promote rapid growth. Declines in the late summer or fall then resulted from reduced upwelling and decreased inputs of coastal diatoms, as well as movement of the null zone upstream and out of San Pablo Bay with declining Delta outflows (Cloern 1979). Dugdale and colleagues, however, have recently argued that spring blooms in San Pablo Bay and Central Bay result from *in situ* phytoplankton growth, sparked by nitrate uptake facilitated by low ammonia concentrations and adequate water transparency (Wilkerson et al. 2006; Dugdale et al. 2007).

The size of the baseline phytoplankton biomass from San Pablo Bay and the size of spring and fall blooms have increased significantly since the mid-to-late 1990s (Cloern 2006; Cloern et al. 2007). Cloern et al. (2007) argued that this was likely due to colder surface waters and greater upwelling along the central California coast (related to the start of an Eastern Pacific cold phase of the Pacific Decadal Oscillation), causing an increase within the Bay of some fish and crab species (Bay shrimp *Crangon* spp., Dungeness crab *Cancer magister* and English sole *Parophrys vetulus*) that prey on filter-feeding clams and mussels that live on the Bay bottom, a consequent reduction in the biomass of these filter feeders and their phytoplankton consumption rate, and thus an increase in phytoplankton density. This effect may have been augmented by advection into the Bay of coastal-produced phytoplankton, or resting stages or vegetative cells of coastal phytoplankton that could seed blooms within the Bay, whose coastal densities may have increased in response to upwelling changes (Cloern et al. 2007).

Central Bay

Similar to San Pablo Bay, phytoplankton densities in Central Bay peak between May and June and consist mainly of coastal diatoms, including *Skeletonema costatum* and other species (Cloern 1979; Jassby et al. 1996). Cloern (1979) and Jassby et al. (1996) suggested that the Central Bay phytoplankton concentrations resulted from upwelling and offshore blooms outside of the Golden Gate, which were then carried into the Central Bay in tidal currents. Dugdale and colleagues, however, have recently argued that spring blooms in Central Bay are the result of phytoplankton growth within the Central Bay, based on nitrate uptake facilitated by low ammonia concentrations and suitable water transparency (Wilkerson et al. 2006; Dugdale et al. 2007). Water transparency has decreased in the Central Bay since the mid-to-late 1990s (Cloern et al. 2006), but as noted above for San Pablo Bay, the size of the baseline phytoplankton biomass and the spring blooms increased significantly, possibly resulting from coastal changes in upwelling and surface temperature causing a top-down trophic cascade that increased benthivorous predators, reduced filter-feeding bivalves and released phytoplankton blooms, possibly augmented by an influx of coastally-produced phytoplankton (Cloern et al. 2006; Cloern et al. 2007).

South Bay

The South Bay is a brackish embayment with no large direct inflow of fresh water. Most of the nitrogen and phosphorous input is in wastewater discharges at a relatively constant rate throughout the year; large discharges at the southern end of the Bay produce a north-south gradient in nutrient concentrations (Conomos et al. 1979). The South Bay is generally less turbid than the river-dominated northern reach of the Bay and is usually well-mixed vertically (Conomos et al. 1979]; Cole & Cloern 1984). However, under certain conditions when there are adequate freshwater inflows in the winter or spring, during periods of weak tidal and wind mixing, the South can stratify with lighter, low salinity water lying over denser, saltier water on the bottom. Phytoplankton are then retained in the upper layer where there is enough sunlight for rapid growth, and populations can build up rapidly (Conomos et al. 1979; Cloern 1979; Jassby et al. 1996). When the water is stratified, the phytoplankton populations in the upper layer are also kept apart from clams and other filter-feeding invertebrates on the bottom that could consume them (Cloern 1982; Cole & Cloern 1984; Jassby et al. 1996). Jassby et al. (1996) noted that phytoplankton blooms in the South Bay may require a low biomass of benthic filter feeders in the shallows to get started and to sustain for more than 1-2 weeks, and that the typically lower benthic biomass in the spring may explain why South Bay blooms have been more frequent and stronger in the spring than in the fall.

Studies in the early 1960s found that South Bay phytoplankton blooms were dominated by large diatoms typical of coastal waters (Storrs et al. 1963), but in the late 1970s-1980s blooms were dominated by microflagellates and small centric diatoms (Cloern 1979; Cole & Cloern 1984). It's unclear whether this is a sampling artifact or a real shift in phytoplankton composition (Cloern 1979). In the summer, large diatoms are common in the northern part of the South Bay, while microflagellates and small diatoms are found in the south (Cloern 1982). Jassby et al. (1996) reported that South Bay spring blooms are dominated by diatoms and are sometimes followed by a red tide produced by a nontoxic ciliate, *Mesodinium rubrum*.

Similar to San Pablo and Central Bay, since the mid-to-late 1990s there were significant increases in the size of the baseline phytoplankton biomass in the South Bay, in the spring blooms in the northern part of the South Bay, and in fall blooms throughout the South Bay (Cloern et al. 2006; Cloern et al. 2007). As noted above for Central and San Pablo Bays, this may have resulted from coastal oceanographic changes triggering an increase in benthivorous predators and a consequent reduction in benthic filter-feeders,[22] allowing the phytoplankton to bloom, possibly augmented by an influx of larger numbers of coastal phytoplankton also related to the coastal oceanographic changes (Cloern et al. 2007).

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[22] Note, however, that the decline in the biomass of filter-feeding bivalves in the South Bay reported by Cloern et al. (2007, Fig. 3A at page 18563) began in 1999, and thus preceded the increase in benthivorous fish and crustaceans in the marine domains of the Bay (San Pablo Bay to South Bay) reported by Cloern et al. (2007, Fig. 3B), which began as a minor increase in 2000 and strengthened beginning in 2001. Furthermore, since these species of benthivores primarily feed on the newly settled or smaller bivalves, one might expect a lag of a year or more before seeing a significant imapct on bivalve biomass.

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