Water Quality in South San Francisco Bay, California: Current Condition and Potential Issues for the South Bay Salt Pond Restoration Project

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1 Introduction

Reengineering of the natural world is a hallmark of the human species. Along with this reengineering comes a need to sometimes reverse previous modifications. Management of wetlands in the USA is one example of this cycle of modification and restoration. Loss of wetlands across the USA during European colonization and industrialization has been followed decades and even centuries later by efforts to restore many of these habitats. Habitats can never be restored to their original, pristine form and function, and complete restoration is even more difficult in highly modified landscapes that have large human populations. In this chapter, we address

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water quality concerns that derive from chemical contamination that is related to the restoration of a large area of former tidal wetlands in the highly urbanized San Francisco Estuary in California, USA. Other water quality concerns are also briefly addressed. We begin by describing the San Francisco Estuary and cogent background associated with the South Bay Salt Pond Restoration Project.

1.1 San Francisco Estuary

An unprecedented extent of tidal wetland restoration is ongoing and planned in San Francisco Bay and the greater Estuary, henceforth referred to as the "Estuary" (www.californiawetlands.net/tracker/). The Estuary is a series of large embayments fed by the Sacramento and San Joaquin rivers, which drain California's Central Valley and the surrounding Sierra Nevada mountains. Smaller, local watersheds also contribute to the waters that exit the Estuary through the Golden Gate at the mouth of San Francisco Bay (Fig. 1). In total, the Golden Gate watershed comprises nearly 40% of the area of California and includes a vast variety of land uses. Principal among these uses are agriculture, urban and industrial areas, and cattle ranching (Goals Project 1999). Freshwater inputs to the Estuary from the major rivers are reduced by diversions for agriculture and urban use. These diversions vary in wet and dry years from 10% to nearly 75% of the volume of water that flows into the Bay (URS 2007).

Water quality in the Estuary is affected not only by how much water is diverted from entering it, but also by the quality of the waters that do enter the Estuary from the surrounding watershed. The quality of the incoming waters depends on land use and water quality management in the surrounding watersheds. The local watershed of the Estuary is home to the cities of San Francisco, San Jose, and Oakland as well as high-tech industry in Silicon Valley. Therefore, the local watershed is characterized by urban development, especially toward the south. In 2005, more than 7 million people resided in the Bay Area, and by 2035, a population of 9 million is expected (ABAG 2007). Although the Bay Area has produced advances in many areas of human endeavor, including technology, social values, and environmental consciousness, it has also produced a legacy of contamination in its waterways. Mercury, polychlorinated biphenyls (PCBs), and a host of other chemical contaminants are prevalent in the Bay and in the surrounding wetlands. Many of these contaminants have local urban sources, and others have distant sources, such as mines in the Sierra Nevada or coal-fired power plants in China.

Although the human population of the Bay Area has become urbanized and less directly dependent on the local landscape in recent decades, abundant biological resources and ecological services were important attractions for the original settlement and industrialization of the region. Early accounts of the Estuary describe an ecological jewel characterized by thriving fisheries, oak woodlands teeming with wildlife, and vast marshes stretching from Bay to upland (Goals Project 1999). The Estuary has a plethora of biological diversity and retains many wildlife taxa found

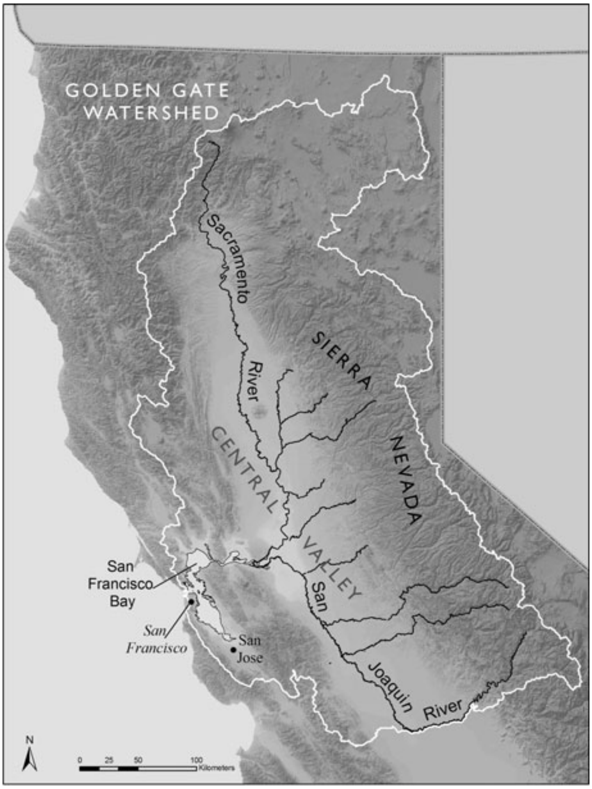

Fig. 1 San Francisco Bay and its watershed, which comprises much of northern California, USA. The Bay is connected to the Pacific Ocean at the Golden Gate, a narrow opening about mid-way down the western side of the Bay (see Fig. 5 for location). The San Francisco Estuary includes an inland delta ("the Delta") where the Sacramento and San Joaquin rivers meet. The maps in Fig. 5 show the approximate spatial extent of the Estuary

nowhere else. However, several species that were once abundant in the Estuary have declined to the point that they now require protected status from state and federal agencies. The tidal wetlands, in particular, are home to several special-status, endemic taxa, such as the California Clapper Rail (*Rallus longirostris obsoletus*) and saltmarsh harvest mouse (*Reithrodontomys raviventris raviventris*), both of which are endangered species (Federal Register 1970).

1.2 Salt Ponds and Wetland Restoration

More than 85% of the tidal wetland acreage (fresh, brackish, and salt marsh) of the San Francisco Estuary has been lost to human alteration of the landscape since Europeans settled the area (Fig. 2) (Goals Project 1999). Many Estuary marshes

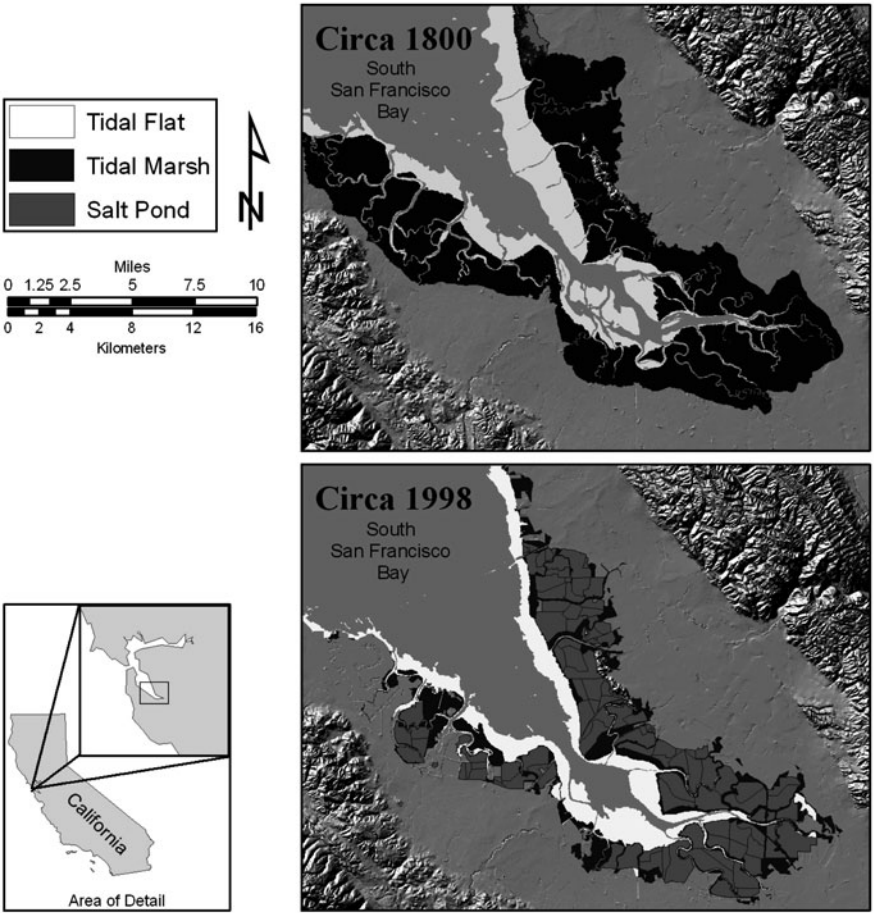

Fig. 2 Tidal marsh habitats in South San Francisco Bay circa 1998 were reduced greatly in acreage, compared to their extent in 1800. In this region of the San Francisco Estuary, most of the former marshes have been converted to salt ponds (Goals Project 1999)

were diked for agriculture, ranching, or urban and industrial development. Other marshes, in San Pablo Bay and South San Francisco Bay, in particular, were converted to salt evaporation ponds. Nearly the entire perimeter of South Bay, which was historically fringed by marshes several miles wide in some areas, was converted to salt ponds (Fig. 2). These salt ponds were developed gradually over decades, from 1857 to 1960 (Collins and Grossinger 2004). The ponds were managed to produce industrial-grade salt by trapping estuarine water in the ponds nearest the Bay and gradually concentrating the salts in the water through evaporation. Ultimately, the highly saline water would be completely evaporated in the ponds closest to land, and then the salt was harvested.

An effort to protect the Bay began in the 1960s, when inadequate treatment of sewage led to regular fish die-offs. Recognition of the importance of restoring local estuarine ecosystems followed, and, in the 1990s, culminated in a consortium of government agencies, non-governmental organizations, scientists, and private citizens working together on the Bay Area Wetlands Ecosystem Goals Project. The purpose of the "Goals Project" was to identify the quantity, type, and distribution of wetlands needed to sustain diverse communities of estuarine wildlife; the goals also included the task of performing an analysis of the pre-industrialization extent and function of Bay Area tidal wetlands (Goals Project 1999).

Wetlands restoration boomed following the publication of the Bay Area Wetlands Ecosystem Goals (Goals Project 1999); the largest single effort is the South Bay Salt Pond Restoration Project (SBSPRP). This vast project comprises 15,100 A in South San Francisco Bay (Fig. 3). The SBSPRP aims to restore and enhance a mix of wetland habitats, provide for flood management, and enhance public access and recreation opportunities (www.southbayrestoration.org). The Project came into being following public acquisition of the ponds from the Cargill Company and transfer of ownership and management to the US Fish and Wildlife Service and the California Department of Fish and Game.

The first stages of the SBSPRP are underway. The pond purchase from Cargill was finalized in 2003, and an initial stewardship plan has been in operation since

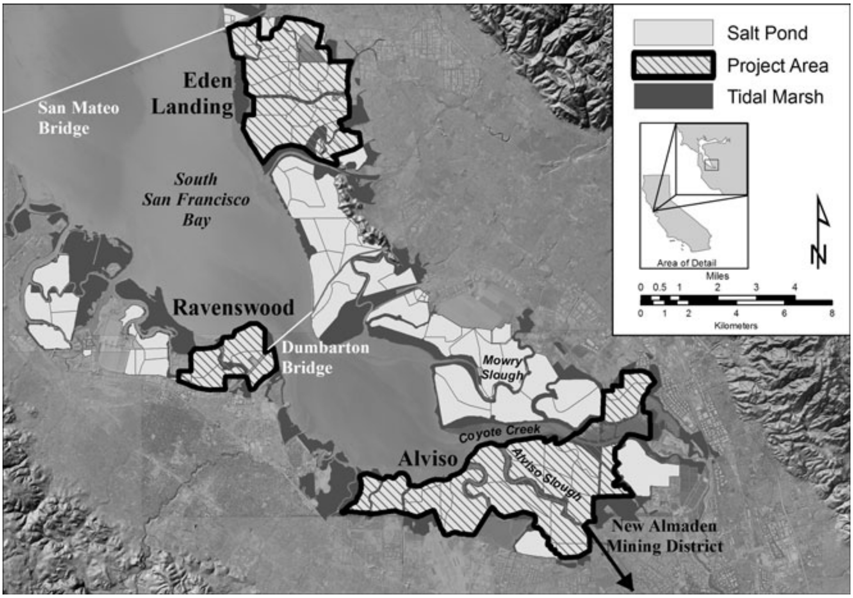

Fig. 3 The South Bay Salt Pond Restoration Project encompasses 15,100 A south of the San Mateo Bridge in San Francisco Bay. In this paper, South Bay is defined as the Bay south of the San Mateo Bridge, a subset of which is lower South Bay, defined here as the Bay south of Dumbarton Bridge

then. The first round of marsh restoration actions began with the breaching of a few ponds in 2006 and will continue with upcoming breaches planned in the next 2 years. Various alternatives for the final, restored South Bay landscape are under consideration, which differ mainly in location and in the ratio of restored marshes and managed ponds. Both habitat types are necessary to support endemic marsh wildlife and migratory water birds.

Both the importance of restoring the South Bay Salt Ponds and some of the associated challenges arise from the fact that the ponds are located in a highly urbanized Estuary. The Project is important for biological conservation, because so many tidal wetlands have been lost with the result that endemic wildlife are now endangered. One of the challenges for the Project is that, despite vast improvements in sewage treatment and water quality over the past several decades, the South Bay nevertheless faces substantial current and future pollution threats. Therefore, SBSPRP managers must consider water quality as a prime factor as they proceed to restore wetland habitats and manage the remaining ponds for wildlife. The location of the Project at the interface between land and water means that water quality in the Project will be tied to water quality management both in the South Bay watershed and in South San Francisco Bay itself. Furthermore, Project managers must be careful that restoration and associated actions do not exacerbate pollution problems in the region. In this paper, we explore the relationship between chemical contamination of the South Bay and the SBSPRP, including current conditions and how the Project and South Bay water quality may affect each other in the future.

2 Current Water Quality

Water quality in South San Francisco Bay is compromised by a variety of chemical contaminants and other types of pollutants that mainly originate in the watershed. In this paper, we focus on water quality in the Estuary, wherein most of the water quality research and monitoring has occurred, rather than on tributary water quality. Pollution in the Bay was present prior to the nascence of the SBSPRP and provides the water quality context in which the wetland restoration will proceed. Thus, the “before” restoration condition of the South Bay includes a history of legacy pollutants in the sediment and water that will affect the “after” restoration condition. In this pollution-impacted Estuary, the goal of the SBSPRP may be to maintain contaminants at or below current concentrations, rather than to restore habitats to pristine water quality.

2.1 Chemical Contaminants

Mercury, PCBs, and polybrominated diphenyl ethers (PBDEs) are the persistent contaminants of greatest concern in the South Bay. All three are present at elevated concentrations in both the abiotic environment and wildlife. Selenium, pyrethroid insecticides, polycyclic aromatic hydrocarbons (PAHs), and dioxins are also problematic. Legacy insecticides (DDTs, dieldrin, and chlordanes) have historically affected the Bay, and these remain above accepted thresholds of concern in a small proportion of samples. Brief summaries of the current state of knowledge for these contaminants in the San Francisco Estuary are given below, with an emphasis on recent data from the South Bay.

2.1.1 Mercury

Mercury is one of the primary current threats to water quality in the South Bay and tops the list of contaminant issues for the SBSPRP. In addition to the concern for present mercury concentrations, there is concern that restored wetlands could result in increased methylmercury production and bioaccumulation. This latter concern is addressed in Section 3.2.

There are several sources of total mercury contamination to the South Bay. These include legacy mercury from mercury- or gold-mining operations, respectively, in the Coast and Sierra Nevada ranges, and ongoing sources such as atmospheric deposition, runoff from urban and industrial areas, and outflow from wastewater treatment plants. The historic New Almaden mining district is situated in the hills above San Jose, in the watershed that drains through the Guadalupe River and Alviso Slough into lower South San Francisco Bay (Fig. 3). New Almaden was the largest mercury mine in North America. This mine ceased operations in 1976 when purchased by Santa Clara County. Legacy mercury from this mining district continues to enter the Bay today. Based on empirical measurements and a sediment-transport model, one load estimate for the Guadalupe River was 4–30 kg of mercury transported into lower South Bay per year (Thomas et al. 2002). More recent measurements indicate high inter-annual variation with much higher loads in peak years: 116, 15, and 8 kg in 2003, 2004, and 2005, respectively (McKee et al. 2006).

In a recent study, concentrations of total mercury and methylmercury were tracked in sediment from the SBSPRP ponds from 2003 to 2007 (Miles and Ricca 2010). Average total mercury in sediment in the Eden Landing ponds (0.11 μg/g) was below the South Bay ambient concentration (0.23–0.27 ppm) (SFEI 2009), whereas average concentrations exceeded the ambient level in the Alviso ponds (0.75–1.03 μg/g). Even within the Alviso pond complex, total mercury concentrations in sediment varied greatly among ponds. Sediment total mercury tended to be stable over time and was not correlated with methylmercury, which was more temporally variable (Miles and Ricca 2010). Understanding the variability, in space and time, of methylmercury concentrations in pond sediment was difficult, because seemingly similar ponds had different chemical and physical conditions that potentially affected mercury cycling. One notable change in methylmercury concentration was a greater than fivefold increase that occurred in two ponds after a levee breach returned them to tidal action. A third pond that was part of the same restoration action did not exhibit such a large increase in sediment methylmercury concentration (Miles and Ricca 2010).

Bioaccumulation of methylmercury also varies spatially across the SBSPRP area, but no long-term temporal trends are indicated by available data. In water birds, the highest exposure (as indicated by blood total mercury concentrations) was consistently observed in the western Alviso ponds, in both terns (Ackerman et al. 2008a) and recurvirostrids (Ackerman et al. 2007). The spatial pattern in marsh birds was completely different, however. Concentrations of total mercury in the blood of tidal marsh Song Sparrow (*Melospiza melodia pusillula*) were related to distance from the sampling site to the Bay, rather than to region within South Bay (Grenier unpublished data). Regarding time trends in methylmercury bioaccumulation, the best dataset available for the Estuary is for total mercury concentrations in striped bass muscle (*Morone saxatilis*) from 1970 to 2000. These data indicate no change over three decades (Fig. 4) (Greenfield et al. 2005). This result probably reflects the ongoing inputs to the Bay from both legacy and contemporary sources and flux from the massive reservoir of mercury in Bay sediments (Conaway et al. 2008).

Methylmercury bioaccumulates in South Bay food webs to concentrations that are sufficiently high to cause concern for adverse effects in humans and sensitive wildlife. Humans are at risk from exposure to methylmercury from eating sport fish from San Francisco Bay. The concentrations of mercury in sport fish taken from the Bay, and particularly from the South Bay, are higher than mercury residues in fish from other parts of California (Davis et al. 2007a). A consumption advisory for sport fish in the Bay was issued in 1994 and was driven by the concentrations of

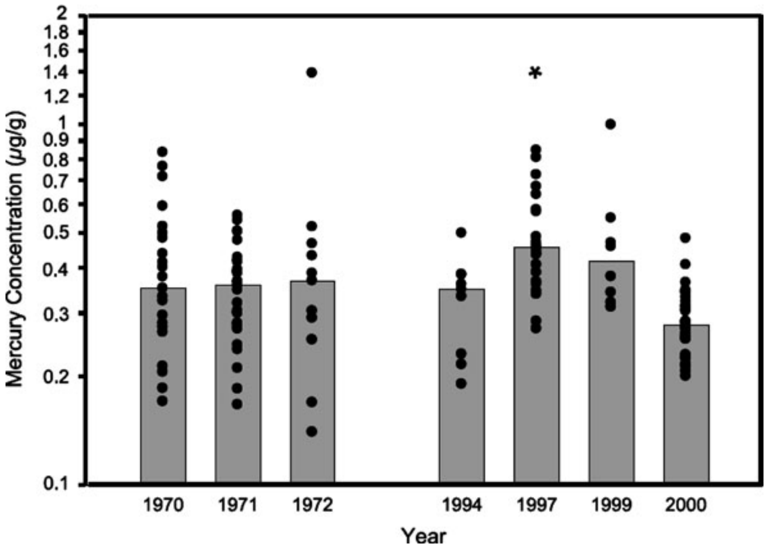

Fig. 4 Total mercury concentrations in striped bass muscle from San Francisco Bay showed no trend between 1970 and 2000. Gray bars indicate annual median concentrations. To correct for variation in fish length, all plotted data were calculated for a 55-cm fish using the residuals of a length:log (Hg) relationship. The asterisk above 1997 indicates significant difference from overall length:mercury regression (see original paper). Note log scale on the *y*-axis. From: Greenfield et al. (2005)

mercury, PCBs, organochlorine insecticides, and dioxins (OEHHA 1994). The public is advised not to eat Bay fish too frequently, and large predatory fish should not be consumed at all by certain demographic groups. Leopard shark (*Triakis semifasciata*), striped bass, and white sturgeon (*Acipenser transmontanus*) are examples of species that tend to have the highest mercury concentrations (Greenfield et al. 2005; Hunt et al. 2008).

Evidence of the exposure to and effects of methylmercury on wildlife in the Bay and its wetlands is increasing. Wildlife exposure to methylmercury is becoming better understood as the body of research has grown over the last 20 years; the effects of methylmercury on wildlife are less well studied. The effect thresholds of specific species constitute an important data gap, and linking elevated exposure to population-level effects is difficult. For example, a large proportion of Black-necked Stilt (*Himantopus mexicanus*) and Forster's Tern (*Sterna forsteri*) breeding adults had total blood mercury concentrations that placed them at moderate to high risk of reproductive impairment (Eagles-Smith et al. 2009). However, a related study of the effect of egg mercury concentrations on chick survival, in these same populations, failed to show significant effects (Ackerman et al. 2008b, c).

The overall change in habitat acreage from salt pond to tidal marsh and tidal slough that will occur as the SBSPRP progresses, and the particular locations of the ponds that are converted to marsh, will likely affect methylmercury concentrations in South Bay water birds. Where these birds forage, both in terms of region within the Bay and habitat type, is closely tied to the variability of mercury tissue concentrations of recurvirostrids, terns, and scaup (Ackerman et al. 2007, 2008a; Eagles-Smith et al. 2009). Water birds from the lower South Bay had the highest total mercury concentrations in their tissues (Eagles-Smith et al. 2009). The habitats that were associated with higher mercury concentrations in water birds were at the margin of the Bay, and these habitats (i.e., salt ponds and managed marsh) had altered hydrology (Ackerman et al. 2007, 2008a).

In contrast to the water birds discussed above, many tidal marsh bird species are endemic to San Francisco Bay. These tidal marsh birds also have elevated exposure to methylmercury and consequent health risks. In particular, the federally endangered California Clapper Rail has poor reproductive success that may be related to methylmercury. An estimated 15–30% of the observed reduction below normal hatchability in this subspecies has been attributed to contaminants, with methylmercury principal among them (Schwarzbach et al. 2006). Effects on other marsh birds in San Francisco Bay have not been studied, but information on the exposure of the Black Rail (*Laterallus jamaicensis*) (Tsao et al. 2009) and tidal marsh Song Sparrow (Grenier unpubl data) indicate that many breeding adults are above a 25% effect concentration (0.81 μg/g wet weight total mercury in blood) that is based on lab and field studies of other songbirds and above which 25% of eggs are predicted not to hatch due to methylmercury effects.

A recent egg-injection study provided information on the relative sensitivity of 26 bird species to methylmercury (Heinz et al. 2009) including representative species from the taxonomic families that have been studied in San Francisco Bay. Ducks (e.g., scaup and Mallard) had low sensitivity; terns, rails, and songbirds had medium sensitivity to mercury; and two ardeid (herons and egrets) species had high sensitivity, whereas another ardeid had medium sensitivity. This new information calls into question how the avian embryotoxic threshold – which is based on studies with captive Mallards (Heinz 1979) – should be interpreted for species that are more sensitive to mercury than Mallard. For example, wading-bird mercury exposure has not been studied for some time in South San Francisco Bay, but Black-crowned Night Heron (*Nycticorax nycticorax*) and Snowy Egret (*Egretta thula*) eggs collected between 1982 and 1990 (Ohlendorf et al. 1988; Hothem et al. 1995) had mean mercury concentrations that were less than half the Mallard threshold value. Given the difference in sensitivity of the ardeids and the Mallard, it is difficult to interpret whether the observed concentrations in South Bay wading birds are problematic or not.

Mammalian exposure and effects from methylmercury are less well characterized than they are for birds. Many mammalian wildlife populations that may have been sensitive to mercury are now extirpated from the Bay. The taxa of concern that remain in South Bay are harbor seals (*Phoca vitulina*) and small mammals in tidal marshes. Harbor seals in San Francisco Bay have elevated mercury concentrations in blood and hair (Kopec and Harvey 1995; Brookens et al. 2007), and Mowry Slough in the lower South Bay (Fig. 3) is an important breeding area for the seals. However, effect studies in harbor seals have indicated greater risk from organic contaminants than from mercury. Mercury exposure and effects in small mammals resident in tidal marshes are virtually unstudied. In the one study performed in San Francisco Bay, endangered salt marsh harvest mice were absent from marshes that had higher concentrations of mercury in other rodents, yet were present in marshes with lower rodent mercury residues (Clark et al. 1992). This broad correlation may indicate that high methylmercury bioaccumulation may be a stressor that contributes to extirpation of these mice in some marshes. A prime candidate for evaluation of mercury exposure in the tidal marsh ecosystem is the shrew. These tiny insectivores are completely carnivorous, have extremely high metabolic rates, and endemic marsh subspecies have been described for both ornate (*Sorex ornatus*) and wandering (*Sorex vagrans*) shrews in San Francisco Bay. Shrews are good bioindicators of metal pollution (Sanchez-Chardi et al. 2007), and in contaminated areas elsewhere shrews have accumulated liver mercury concentrations up to and in excess of 30 μg/g dry wt (Cocking et al. 1991; Talmage and Walton 1993).

2.1.2 PCBs

Despite the 1979 federal ban on PCB production and sale and subsequent gradual decline of PCBs in the environment, this suite of chemicals remains one of the main contaminants that affects water quality in San Francisco Bay, and the South Bay in particular. PCB residues are widely spread in Bay sediments, and they continue to bioaccumulate in the food web to a degree that poses health risks to humans and wildlife (Davis et al. 2007b). The most recent water quality monitoring data (SFEI 2009) as well as reviews of sediment PCB data for the Bay (Davis et al. 2007b; SFBRWQCB 2007) show that South Bay has relatively high concentrations, and PCB hotspots are present in the wetlands and other Bay margin habitats. The most important pathways by which PCBs enter South Bay waters are urban runoff and erosion of buried sediment. Riverine inputs from the Sacramento-San Joaquin Delta are important in other parts of San Francisco Bay but have less influence in South Bay, which is geographically and hydrologically more removed from the Delta (Conomos et al. 1979). Urban runoff is a source of PCBs that can be controlled and, as such, is a focus of the PCB total maximum daily load (TMDL) regulation. The effectiveness of different management options for reducing PCBs in stormwater is an important knowledge gap (Davis et al. 2007b). Release of PCBs through erosion of buried sediment is discussed later in this paper.

Along with mercury, PCBs are a primary driver of the advisory to limit consumption of fish taken from the Bay (OEHHA 1994). Median concentrations of PCBs in sport fish from South Bay have consistently exceeded the PCB TMDL cleanup target of 10 ng/g (Hunt et al. 2008). White croaker (*Genyonemus lineatus*) and shiner surfperch (*Cymatogaster aggregata*) are species with relatively high lipid content; these species had median PCB concentrations that exceeded the target by more than one order of magnitude. White sturgeon, anchovy (*Engraulis mordax*), and black surfperch (*Embiotoca jacksoni*) also consistently exceeded this value.

PCB concentrations in the Bay may be high enough to adversely affect wildlife, with fish-eating species at the top of the food web generally facing the greatest risks (Davis et al. 2007b). Since the early 1980s, the available data have indicated that PCBs accumulate to high concentrations in South Bay piscivorous birds and may cause adverse effects on survival. PCB concentrations in eggs well above 1.0 μg/g wet wt have been measured in Black-crowned Night Herons, Snowy Egrets, Forster's Terns, Caspian Terns, and Least Terns (*Sterna antillarum*) (Hoffman et al. 1986; Ohlendorf et al. 1988; She et al. 2008). Effects on growth and induction of cytochrome P450 were documented to have occurred in Black-crowned Night Herons and Double-crested Cormorants (Hoffman et al. 1986; Davis et al. 1997). The highest geometric mean concentrations in tern eggs were found in and near Eden Landing, one of the main restoration areas for the SBSPRP (Fig. 3; She et al. 2008). Even California Clapper Rails, a relatively low-trophic-level marsh species that consumes mainly invertebrates, were found to have some eggs with PCB concentrations sufficient to potentially cause deleterious effects (Schwarzbach et al. 2001). Authors of a recent study with a small number ($n = 4$) of fail-to-hatch Clapper Rail eggs reported a median PCB concentration in the eggs of 4,640 ng/g lipid wt (She et al. 2008). An obstacle to determining whether these concentrations may produce adverse affects is that the relative sensitivity of rails to PCBs is not known.

Harbor seals from San Francisco Bay, sampled in various studies over the past 30 years, had PCB concentrations in blubber and whole blood that exceeded those associated with impaired reproduction in a controlled feeding study (Davis et al. 2007b; Thompson et al. 2007). Total PCB concentrations in liver from five adult seals stranded in the Bay between 1989 and 1998 had a median concentration of 35.6 μg/g lipid wt (Park et al. 2009). Decreases in seal blood PCB concentrations occurred between the early 1990s and 2001–2002. Despite this reduction, some seals in the most recent time period may have experienced health effects related to organic chemical contaminants; this conclusion was based on a correlation between leukocyte counts and concentrations of PCBs, PBDEs, and DDE (Neale et al. 2005). Thus, PCBs are a significant concern in harbor seals.

Although the available data from top predators in the Bay were not collected with the intention of measuring long-term trends, there is a general pattern of slow PCB residue decline in wildlife over the past two to three decades (Davis et al. 2007b). The longer the time span of the dataset, the more apparent the decline, which indicates that the net dissipation of PCBs in the Bay food web is quite slow. The difference in analytical methods used to measure for PCBs over the years and variation in study designs and tissues sampled make this a tentative conclusion that will be clarified only with future monitoring.

2.1.3 PBDEs

Compared to mercury and PCBs, PBDEs have surfaced much more recently as contaminants of concern in San Francisco Bay. PBDEs are an example of emerging contaminants that are persistent and biomagnify, and that could affect higher-trophic-level species in restored habitats. PBDEs were virtually undetected in samples during the 1980s. However, over the course of the 1990s, PBDE residues became common in the water, sediments, and food web of the Bay. Such brominated flame retardants are relatively new contaminants of environmental concern, and the scientific understanding of their effects is limited (Birnbaum and Staskal 2004). Thresholds of concern for PBDEs have not yet been established. The two lower-brominated congener mixtures (penta-BDEs and octa-BDEs) were banned in California in 2006, but the deca-BDE mixture is still in commercial production.

Sources and pathways by which PBDEs enter the Bay are under investigation. Possible pathways include municipal and industrial discharges, stormwater, atmospheric deposition, small and large tributaries, and landfill leaching (Oram et al. 2008). Based on studies from tributaries and wastewater treatment plants in South Bay, PBDE loads are 3–10 times greater than those of PCBs (McKee et al. 2006; SFEI 2007). Monitoring data from 2002 to 2008 indicate that the lower South Bay is a hotspot for PBDEs in sediment and water, especially in certain years (Fig. 5; SFEI 2009). PBDEs are a high priority for ongoing monitoring and research to understand trends in food-web exposure and effects on sensitive species.

PBDE concentrations measured between 1989 and 2003 in harbor seal blubber, tern eggs, and human breast tissue from the San Francisco Bay Area were among the highest ever recorded (She et al. 2002, 2004). Some individual Forster's Tern eggs from the Eden Landing area of the SBSPRP (Fig. 3) had extremely high total PBDE concentrations of 62,400 and 63,300 ng/g lipid wt in 2001 and 2002, respectively (She et al. 2008). The geometric means for total PBDE concentrations in tern eggs sampled from 2000 to 2003 were lower, but still quite elevated – in the range of 3,700–4,800 ng/g lipid wt for Caspian, Forster's, and Least Terns (She et al. 2008). PBDEs have also been documented to exist in the sediment, water, bivalves, and fish of the Estuary (Holden et al. 2003; Oros et al. 2005). Mean concentrations (wet wt) in fish collected in 2006 were 56 ng/g for white croaker, 20 ng/g for white sturgeon, 13 ng/g for shiner surfperch, and 12 ng/g for northern anchovy (Hunt et al. 2008).

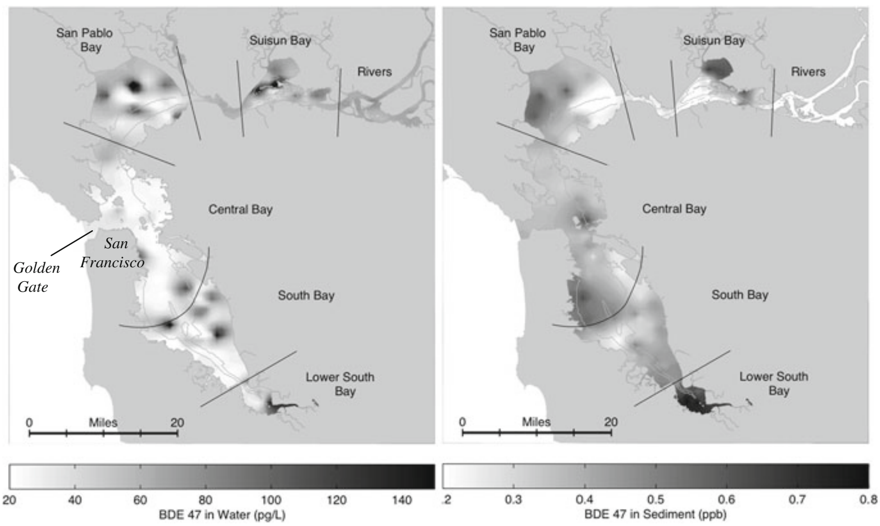

Fig. 5 Spatial trends of Brominated Diphenyl Ether (BDE) 47 in water from 2002 to 2008 (left) and in sediment from 2004 to 2008 (right) for the San Francisco Estuary. The two most southern segments of the Bay, as delineated between the black lines, correspond to the location of the South Bay Salt Pond Restoration Project (SBSPRP). These segments contain several hotspots for BDE 47, which is one of the most abundant Polybrominated Diphenyl Ethers (PBDEs) and is an index of PBDEs as a group. The highest average concentration from 2004 to 2008 of BDE 47 in sediment was 0.75 ppb in lower South Bay. From: SFEI (2009)

2.1.4 Other Chemical Contaminants

Polychlorinated dibenzodioxins and dibenzofurans are other persistent contaminants of concern in South Bay, and they are also among the pollutants in fish that prompted the fish consumption advisory for San Francisco Bay. The Regional Monitoring Program for Water Quality in the San Francisco Estuary has been analyzing residues of dioxins and dioxin-like compounds in fish (furans and co-planar PCBs) every 3 years, since 1994. The results have been relatively consistent throughout that time period, suggesting that dioxins are maintaining relatively constant concentrations in fish (Fairey et al. 1997; Davis et al. 2002; Greenfield et al. 2005; Hunt et al. 2008). Dibenzodioxins and dibenzofurans in all the fish (white croaker) sampled in 2000, 2003, and 2006, exceeded the screening value of 0.3 pg/g wet wt dioxin toxic equivalents (TEQs) established by the California Office of Environmental Health Hazard Assessment. Dioxin TEQ concentrations in 2006 were highest in the South Bay, with a maximum level of 16.3 pg/g wet wt (Hunt et al. 2008).

There is evidence that dioxins and dioxin-like compounds may be causing effects in two federally endangered bird species: the Least Tern and California Clapper Rail. Ten Least Tern eggs from the 2001 and 2002 breeding seasons had a geometric mean of TEQs that was within the effects threshold range calculated by the study authors (Adelsbach and Maurer 2007) using avian-specific toxic equivalency factors. These eggs had failed to hatch, so they represented a biased sample of the population. However, their failure to hatch could have been related to high TEQs. Clapper Rail fail-to-hatch eggs ($n = 4$) exhibited concentrations of individual dioxins above a threshold for effects in chickens (Adelsbach and Maurer 2007).

Once at the top of the list of contaminants of concern for San Francisco Bay, legacy insecticides are now declining. Historical use of dieldrin, chlordanes, and DDTs resulted in widespread contamination of the Bay and resident wildlife, and they were each banned 20 or more years ago. Legacy pesticides persist in Bay sediment and continue to enter the Bay from its tributaries (Connor et al. 2004), but the current situation is much improved. Reviews are available on the bioaccumulation behavior of these contaminants from statewide monitoring programs (Davis et al. 2007a) and San Francisco Estuary monitoring (Gunther et al. 1999; Greenfield et al. 2005). These reviews document significant residue declines of these contaminants in fish and bivalves over the past two to three decades. However, the rate of the decline may have slowed since the early 1990s, and the Bay may not be adequately cleansed of these chemicals until ongoing inputs are eliminated (Connor et al. 2004).

A small proportion of sport fish sampled from the Estuary exceed human health thresholds of concern for legacy insecticides. The most recent published data from 2006, when three fish species were sampled, show that most of the exceedances were for dieldrin (32%, 9 of 28 composites), and there were fewer for DDTs (11%, 3 of 28) and chlordanes (4%, 1 of 28) (Hunt et al. 2008). For each contaminant, some or all of the composites that exceeded the threshold of concern were from South Bay. The evidence suggests that current effects of legacy insecticides on wildlife are minimal. In some places, chlordanes may impact benthic invertebrates, but the impacts of DDTs on bird reproduction that were documented in the 1980s are no longer in evidence (Thompson et al. 2007).

A variety of other organic chemicals are cause for concern in the Bay, including pyrethroid insecticides and PAHs, but fewer data are available for these compounds than for the chemical groups discussed above. Pyrethroids are relatively new insecticides and are used as alternatives to the legacy organochlorine and organophosphate insecticides. As the use of pyrethroids as agricultural, commercial, and household insecticides increases, the potential for effects on invertebrates and fish, to which pyrethroids are highly toxic, also increases. Lowe et al. (2007) examined the toxicity of sediments in six tributary creeks around the Bay and found that only the South Bay tributaries were toxic to amphipods. The cause of this toxicity was identified as likely originating from exposure to either pyrethroids or DDT metabolites. Amweg et al. (2006) found that eight creeks along the eastern side of the Bay displayed toxicity from pyrethroids, mainly bifenthrin, on at least one of four sampling occasions. The effect of the pyrethroids on biota upon entering the Bay at the mouth of the creeks is not known.

PAHs are organic contaminants derived from carbon-based fuels, such as petroleum products. These chemicals are persistent in sediment and water, but they are metabolized by vertebrates, rather than accumulating in vertebrate tissue.

Monitoring studies suggest that PAH sediment concentrations may be above reproductive effects thresholds for fish (SFEI 2007). A new study has been funded by the Regional Monitoring Program to examine the effects of PAHs on larval fish in the Bay. PAH concentrations have been relatively constant over the past two decades (SFEI 2007). Events such as the Cosco Busan Oil Spill in November 2007 in central San Francisco Bay are a reminder that the high density of shipping traffic renders the Estuary vulnerable to PAH contamination from oil spills.

Besides mercury, other elemental contaminants that have been studied in the Bay include selenium, nickel, and copper. Selenium bioaccumulates in the San Francisco Bay food web to an extent that could cause harm to fish and to humans that consume diving ducks, but the focus of that concern is in the northern reaches of the Estuary. A consumption advisory for ducks is in place and a selenium TMDL is being developed. Nickel and copper were considered historical problems in San Francisco Bay, but samples obtained over the past decade indicate that concentrations are consistently lower than the water quality objective considered to be completely protective of aquatic life (SFEI 2007). Concentrations of arsenic, cadmium, chromium, copper, lead, nickel, selenium, and zinc, measured in the Project pond sediments close to the time of public acquisition, were generally at or below ambient concentrations in the surrounding area, although some ponds had above-ambient selenium levels (Brown and Caldwell 2005).

Despite the contaminant monitoring and research summarized above, many chemicals in the Bay may be causing unrecognized effects, and, conversely, some documented effects cannot be tied to specific contaminants. Evidence from biomarkers in fish and benthos, such as breaks in DNA and cellular abnormalities, shows that these taxa have been exposed to contaminants, yet pinning down which chemical has induced the effect is difficult in an Estuary with a complex cocktail of environmental pollutants. Toxicity of sediment to benthic invertebrates is prevalent around the margin of the Bay, yet the particular cause or causes have not been easy to identify. Contaminant mixtures may be as important, or more important, than individual contaminants (Thompson et al. 2007). A recent study in San Francisco and Tomales Bay tidal marshes documented benthic marsh fish population effects that correlated with a variety of contaminants in sediment (McGourty et al. 2009).

Emerging contaminants are a diverse group of unregulated and relatively unmonitored chemicals that have been detected in the environment and whose effects are largely unknown. Some chemicals considered to be emerging contaminants for the Estuary include perfluorinated chemicals (PFCs), pharmaceuticals, and non-PBDE flame retardants. Harbor seals recently analyzed for PFCs had concentrations in blood that were several times greater than those found in seals from other parts of the world and, seals from the South Bay had particularly high concentrations (SFEI unpubl data). Recent data from lower South Bay documented the presence of pharmaceuticals at concentrations well below available acute and chronic toxicity thresholds (SFEI unpubl data). Current use, non-PBDE flame retardant chemicals have also been detected in South Bay by the Regional Monitoring Program (unpubl data).

2.2 Other Aspects of Water Quality

2.2.1 Biological

Water quality in the Bay is also affected by biological constituents, particularly invasive species. San Francisco Bay is one of the most invaded estuaries on the planet, both in terms of the numbers of exotic organisms and their ecological dominance (Cohen and Carlton 1998). The arrival of exotic organisms in ballast water of ships and via other human activities has altered the ecology and food web of the Bay in ways so significant that water quality has also been profoundly affected. Hundreds of non-native taxa have been identified in Bay waters, and more than one hundred others exist that are of unknown origin (possibly non-native). These invaders dominate a large number of biological communities in the Bay, typically comprising 99% of the biomass of soft-bottom benthos, fouling communities, brackish-water zooplankton, and freshwater fish (Cohen and Carlton 1998). The Asian clam (*Corbula amurensis*) has radically altered the biological productivity of the northern subembayments by eliminating seasonal phytoplankton blooms that previously fueled flourishing populations of zooplankton, invertebrates, and fish in the pelagic food web (Thompson 2005). *Corbula* filtered nearly all of the phytoplankton from the water, thus suppressing the seasonal blooms. This effect of *Corbula* did not occur in South Bay, where differences in physical factors caused the phytoplankton bloom to occur earlier in the year, reducing the influence of this exotic clam (Thompson 2005). Nevertheless, the potential for exotic species to radically alter water quality clearly exists in South Bay.

Other changes in phytoplankton blooms have been occurring in San Francisco Bay and are thought to result from changing physical factors rather than invasive species. In 2004, the first dinoflagellate bloom, or red tide, observed in nearly 30 years of monitoring occurred in South San Francisco Bay (Cloern et al. 2005). These blooms can be toxic, causing die-offs of fish and other organisms, but in this case the bloom ended before any negative effects were observed. More recently, a cooling phase in the nearby Pacific Ocean was associated with various marine species moving into the Bay seeking warmer waters (Cloern et al. 2007). These new arrivals preyed on clams and reduced their numbers. This resulted in a release of phytoplankton populations that increased biomass year-round and allowed new autumn–winter blooms. These fluctuations at the base of the food web can have significant effects on fisheries and wildlife. The evidence suggests that the recent changes in phytoplankton were related to climatic events that extended well beyond the domain of the Estuary and its watershed (Cloern et al. 2007). Thus, future surprises of this kind can be expected to occur in association with climate cycles and global climate change.

2.2.2 Chemical

Chemical attributes of water quality, such as dissolved oxygen and salinity, are important to the success of the SBSPRP and the health of South Bay aquatic life.

High salinity and/or low oxygen levels can occur in the enclosed ponds in summer as water evaporates and algae decompose. Low dissolved oxygen then may cause fish and invertebrates to move upward into the oxygenated water layer near the surface, where they are more likely to be depredated by birds (Lonzarich and Smith 1997). If hypoxia becomes extreme, aquatic animals die. Shortly after the ponds were acquired for the purposes of restoration, there were some pond discharges into the Bay in 2004 that exceeded regulatory thresholds for dissolved oxygen. Those events provided important lessons for how to monitor the ponds. Monitoring practices have since changed, and the ponds are managed to maintain optimum water quality for wildlife, insofar as possible. The vast acreage of the Project makes this management and monitoring a large and costly task. The restoration of ponds to tidal marsh will eliminate both dissolved oxygen and salinity water quality concerns and the need to manage and monitor them. However, water quality will need to be managed and monitored in some ponds that will remain non-tidal and serve as foraging areas for water birds.

3 Potential Future Changes Related to Restoration

The spatial extent of the SBSPRP is so large that the Project could have regional effects on water quality of the South Bay that extend beyond the Project areas (Fig. 3). Particularly in the lower South Bay, south of the Dumbarton Bridge, the Project has the potential to strongly influence water quality at a regional scale, because water residence time is long (Conomos et al. 1979) and the acreage of the Project ponds is comparable to that of the open waters of the Bay. The effect of Project implementation could be to worsen, improve, or not affect the already impaired water quality in South Bay.

3.1 Erosion of Contaminants at Depth

Accelerated erosion of buried sediment is a potentially serious regional threat to South Bay water and sediment quality. Studies by the US Geological Survey (USGS) have shown that the South Bay (Foxgrover et al. 2004; Jaffe and Foxgrover 2006) and other parts of the Bay (Jaffe et al. 1998; Cappiella et al. 1999) experience fluctuating periods of erosion and of sedimentation, probably related to changes in sediment supply (McKee et al. 2002; Jaffe and Foxgrover 2006). Opening salt ponds to tidal action will create a new demand for sediment and will likely cause erosion of buried sediment in some areas. Such erosion could pose a risk with respect to recovery of the South Bay from legacy contamination, because the layers of sediment that would be unearthed are from earlier decades when the Bay was generally more contaminated (van Geen and Luoma 1999).

Bathymetric surveys conducted in 1931, 1956, 1983, and 2005 provided the basis for recent analyses of South Bay erosion and deposition (Foxgrover et al. 2004; Jaffe and Foxgrover 2006). From 1931 to 1956, a period with rapid urbanization, industrialization, and little wastewater treatment, the South Bay experienced widespread deposition of relatively contaminated sediment. From 1956 to 1983, a period including an era of peak contamination in the 1960s and marked improvements with the onset of wastewater treatment in the 1960s and 1970s, the South Bay experienced net erosion. In the most recent time period, net deposition has once again occurred. The erosion and deposition varies by locale, with more erosion in the northern part of South Bay and more deposition in lower South Bay. These long-term patterns are a critical piece of information needed to predict the rate of improvement of Bay water quality in future decades.

Sediment coring studies around the Bay consistently show greater concentrations of legacy contaminants at depth than at the surface. Cores taken from a South Bay tidal marsh along Coyote Creek (Fig. 3) exemplify the pattern, with mercury peaking at a depth that corresponds to deposition in the mid-twentieth century (Fig. 6) (Conaway et al. 2004). Time lines for the cores were established using radiocarbon and pollen of introduced plants. This coring study showed that pre-mining concentrations of mercury in South Bay sediment were similar to those in other parts of the Bay from the same time period. This similarity indicates that natural weathering

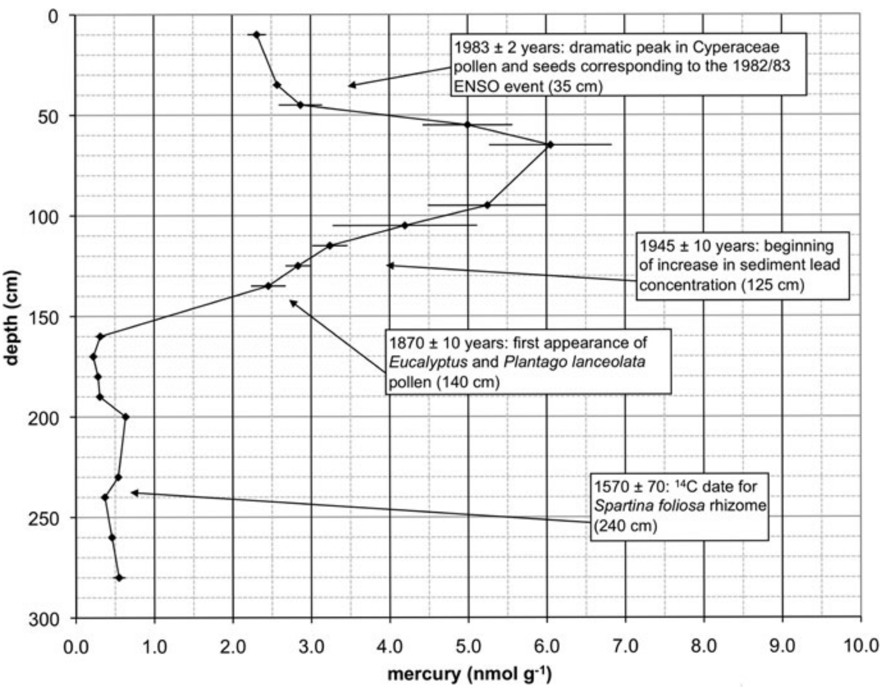

Fig. 6 In this sediment core from South Bay (Triangle Marsh), total mercury concentrations peaked at a depth corresponding to deposition in the mid-1900s. From: Conaway et al. (2004)

of mercury sources did not cause elevated total mercury in San Francisco Bay prior to mining; rather, the concentrations only increased after mining began. Pre-mining concentrations (0.08 μg/g) (Conaway et al. 2004) were approximately one-third of current concentrations, which average 0.24 μg/g (SFEI 2009).

A series of 15 sediment cores 2-m deep were taken from Alviso Slough, which currently drains the New Almaden district. These cores showed somewhat greater mercury concentrations at depth, compared to surface concentrations, and only a few cores with much higher mercury maxima below the surface (Marvin-DiPasquale and Cox 2007). Profiles of mercury by depth varied among the cores, and the sub-surface concentration maxima were generally four to five times greater than the surface concentrations. This coring study included an experiment designed to assess whether mercury from deep sediment would become bioavailable if released into the water column by erosion. When buried sediment was mixed with oxygenated overlying water, reactive mercury ($Hg(II)_R$) concentrations increased 35- or 53-fold over 1 week, depending on salinity. Reactive mercury was operationally defined in this study as the fraction of mercury that was readily reduced to elemental Hg^0 by an excess of tin chloride ($SnCl_2$) over a short exposure time. Reactive mercury is thought to be available for conversion to methylmercury by sulfate-reducing bacteria. This result, and the increase in sediment MeHg (previously discussed) at two ponds after being returned to tidal action (Miles and Ricca 2010), suggests that a pulse of bioavailable mercury may be introduced to South Bay waters during erosion events associated with restoration actions. The duration of this pulse would probably be related to the duration of the erosion, which might last from months to years, depending on the restoration event. Modeling and monitoring of sediment erosion, transport, and fate processes, in conjunction with methylmercury monitoring, will provide important information for better understanding these relationships and the consequences of erosion in South Bay.

The data available for organic contaminants at depth in South Bay sediments are for PCBs and legacy pesticides, and these data also show higher concentrations below the surface. A subset of the cores collected from Alviso Slough (Marvin-DiPasquale and Cox 2007) was analyzed for PCBs and legacy pesticides (SCVWD 2008). PCB concentrations were consistently higher in the lower half, as compared to the upper half, in these 2-m cores. PCB concentrations in the upper half (52 ± 17 ng/g, mean ± st dev) exceeded a criterion (22 ng/g) set by the San Francisco Bay Regional Water Quality Control Board for the beneficial reuse of dredged material as wetland surface fill. PCB concentrations in sediment at depth (173 ± 17 ng/g) were similar to the criterion value for wetland foundation fill (180 ng/g). These criteria were used for comparison, since much of the scoured sediment will ultimately probably settle on extant tidal marsh and sloughs or in the breached ponds, which are the future sites of restored wetlands. PCB concentrations at the surface were similar to PCB values from randomly sampled nearshore surface sediments in San Francisco Bay (areas south of San Pablo Bay) (AMS 2005). PCB concentrations at depth were similar to those from sediment cores collected downstream of storm drains known to be contaminated by PCBs elsewhere in the Bay (AMS 2004).

Patterns of DDT concentrations were similar to those described for PCBs above, with one exception: DDT concentrations from the Alviso cores near the surface (27 ± 5 ng/g) and at depth (48 ± 16 ng/g) were higher than typical Bay concentrations (SCVWD 2008). Chlordane concentrations were 3.7 ± 2.3 ng/g at the surface and 16 ± 9 ng/g at depth. Concentrations in samples from the lower half of the cores exceeded the wetland foundation criterion (5 ng/g) and were above those found under ambient Bay conditions. Dieldrin was more concentrated in the lower half of the cores (2.0 ± 0.2 ng/g), and only the deeper samples exceeded the wetland surface screening guideline (0.7 ng/g) and were higher than typical Bay concentrations.

The contaminant depth profiles in sediment cores from South Bay exhibited patterns similar to those in cores from the northern reach of San Francisco Bay that were extensively studied for many contaminants and dated by the USGS (van Geen and Luoma 1999). The authors found that metals (Hg, Pb, Cu, Zn, Ag), DDTs, PCBs, and PAHs all had low baseline concentrations prior to industrialization, from which concentrations increased during the early to mid-twentieth century (Hornberger et al. 1999; Pereira et al. 1999; Venkatesan et al. 1999). Most of the contaminants peaked at depth and then declined toward the surface, but PAHs, Cu, and Zn did not decline in more recent sediments. Similarly, concentrations of trace metals from anthropogenic sources peaked at depth in a sediment core from a North Bay tidal marsh (Hwang et al. 2009).

Remobilization of buried sediments that are more contaminated than surface sediments poses a significant issue for restoration activities. Prior research has suggested that legacy contaminants persist in the upper layers of Bay sediment for decades, because the top 30 cm stay in the active layer due to mixing (Fuller et al. 1999). Both organic chemical contaminants and mercury from the Alviso cores showed a pattern of increasing concentrations at depth, necessitating consideration of how erosion could result in increased concentrations of these contaminants in the food web.

3.2 Change of Habitat Types

The planned restoration of salt ponds to tidal marsh has raised concerns about the possibility of increased net methylmercury production and subsequent accumulation in the food web. This concern applies not only to the restored marshes, but also to the South Bay as a whole, which could be affected at a regional scale. These concerns are based on studies in freshwater wetlands conducted at different locations in the USA. Local studies are underway to assess mercury risks associated with restoring particular ponds in South San Francisco Bay. An additional contaminant issue regarding the transformation of ponds to tidal marsh is that the conversion will involve sequestration of millions of cubic meters of sediment. The key question about sediment sequestration in marshes relates to whether it will remove contaminated sediment from the active sediment layer of the Bay and whether it will create marshes that have contaminated food webs.

3.2.1 Methylmercury Production in Wetlands

Wetland biogeochemical conditions may be conducive to the production of methylmercury. Sulfate-reducing bacteria are abundant in wetlands as a result of the anaerobic conditions that prevail in these organic-rich environments, and these bacteria are the main agents of mercury methylation. Many wetlands also have peat soils, and sediment that retains a high percentage of organic matter has been correlated with high concentrations of methylmercury (Krabbenhoft et al. 1999).

The central concern is that the restored marshes may cause greater accumulation of methylmercury in the food web than is already present. This concern has arisen from several studies performed in various parts of the USA. Results from these studies have linked wetlands to relatively high net methylmercury production and export into waterways (Selvendiran et al. 2008). Local- and regional-scale studies have correlated wetland acreage in upstream watersheds with high rates of methylmercury production and export in Wisconsin rivers (Hurley et al. 1995) and have identified wetlands as methylmercury sources in boreal forest (St. Louis et al. 1996) and in riparian wetlands of Massachusetts (Waldron et al. 2000). Additionally, a national-scale study of mercury contamination along multiple gradients found that sediment methylmercury contamination was most strongly correlated with proportion of wetland in the sub-basin (Fig. 7) (Krabbenhoft et al. 1999).

More limited research on this topic has been completed in San Francisco Bay, although some studies are ongoing. The results published to date suggest that wetlands are sites of variable and sometimes high rates of net methylation. Sediment methylmercury concentrations have been observed to be higher in tidal marsh (3–5 ng/g dry wt) than in the open waters of North San Francisco Bay (0.7 ng/g), despite having similar total mercury concentrations (Marvin DiPasquale et al. 2003). Krabbenhoft et al. (1999) found that methylmercury in sediments of five

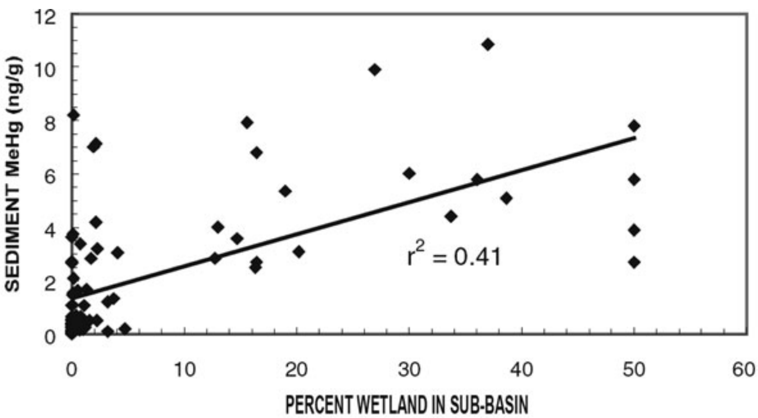

Fig. 7 Methylmercury concentrations in sediment were positively correlated to the percent of wetlands in the sub-basin in a national-scale study of 106 sites from 21 basins. From: Krabbenhoft et al. (1999)

tributaries of the Sacramento-San Joaquin Delta ranged from 0.55 to 2.84 ng/g, with methylmercury accounting for 0.1–2.2% of total mercury. The average percent of methylmercury present for these five sites was the lowest observed in 21 study basins across the USA. In contrast, in a 1994 investigation of tidal marsh sediments in the Bay, Schwarzbach et al. (2000) found methylmercury concentrations between 0.41 and 25.2 ng/g, with percent methylmercury comprising 0.1–6.6%. The maximum value of percent methylmercury to total mercury falls above the 80th percentile of data from the 21 study basins mentioned above, but the minimum value is quite low. Because free methylmercury is short-lived, percent methylmercury is considered to be an indicator of net methylmercury production (Gilmour et al. 1998). The high maximum percent of methylmercury that has been measured in Bay tidal wetlands indicates the potential for high net methylation rates in these ecosystems, while the low minimum percent methylmercury shows that not all tidal marshes are problematic. Understanding the underlying wetland characteristics and processes that lead to this variation is a key consideration for the SBSPRP and other wetland projects that seek to minimize mercury risks related to restoration actions.

Hydrology and organic matter may be the key factors that govern net methylmercury production in Bay wetlands. Research from other parts of the country indicates that hydrology is important in determining which wetlands have higher methylmercury production. St. Louis et al. (1996) found that spatial heterogeneity among wetlands in export of methylmercury was related to differences in hydrology. Newly flooded areas have been linked to spikes of methylmercury concentrations in water and the food web that can continue at elevated levels for decades (Bodaly et al. 2007). Experimental flooding of a boreal forest wetland caused it to increase in methylmercury export by a factor of nearly 40 (Kelly et al. 1997). The applicability of this result to tidal wetlands is uncertain, as some of the change after flooding the forest wetland was related to the death and decomposition of stressed vegetation (Kelly et al. 1997). In contrast, tidal marsh vegetation is adapted to tidal flooding; therefore, a flooded marsh may not be an appropriate parallel to a wetland where plants die after flooding. However, tidal wetlands in Louisiana were found to be probable sites of high methylmercury production and likely sources of methylmercury contamination in local marine food webs (Hall et al. 2008).

Results from San Francisco Bay tidal wetlands do point toward hydrology and organic matter as important factors relating to methylmercury production. A study comparing sediment methylmercury and methylmercury:total-mercury ratios between marsh interior and marsh edge showed concentrations of both to be higher at interior sites (Heim et al. 2007). Physical processes of tide and sediment transport govern marsh geomorphology, generally resulting in oxygenated, inorganic marsh edge sediments near the banks of tidal creeks and more poorly drained, peaty sediments in the marsh interior (Collins et al. 1986; Collins and Grossinger 2004; Culberson et al. 2004). Also, experimental work (Windham-Myers et al. 2009) has shown that methylmercury production in sediment decreased in some tidal wetlands of San Francisco Bay when plants were removed. The authors hypothesized that the mechanism for this reduction was a reduction in labile carbon (acetate) exuded by the plant roots that was previously fueling microbial activity and, hence, mercury methylation.

3.2.2 Sequestration of Contaminants in Marshes

Wetlands sequester and break down some contaminants, largely through processes that take place in sediment and secondarily in plants (Gambrell 1994; Reddy et al. 1999). Wetlands are such effective tools for remediation of pollutants that they are often constructed with the purpose of improving water quality, particularly for wastewater (Hammer and Bastian 1989; Reddy and d'Angelo 1997; Sheoran and Sheoran 2006). The restored South Bay tidal marshes will be created by the deposition in salt ponds of vast amounts of sediment with contamination at levels either of the ambient Bay or of local tributaries that influence particular sites. Restored marshes will begin to vegetate with cordgrass at low-marsh elevations, and they will continue to accrete sediment and associated contaminants as they age, until reaching a maximum elevation of many centimeters above mean high water (Collins et al. 1986; Collins and Grossinger 2004). Thus, the South Bay restored marshes may be sinks for persistent, sediment-bound contaminants. Although the restored wetlands may act as sinks for contaminated sediments, whether the net effect on Bay water quality will be beneficial is difficult to predict, given the potential remobilization of more contaminated sediment layers through erosion caused by restoration.

Contaminants in marsh surface sediments would likely bioaccumulate in primary producers and biota higher in the food web. Creating habitat for wildlife is one of the main goals of the SBSBRP, including habitat for endangered species that are marsh obligates. Thus, it will be important for the restored marshes to provide relatively uncontaminated food resources for species such as the California Clapper Rail. The magnitude of any contaminant effects on wildlife will depend partly on the degree of contamination in the sediment the marshes accrete and in the tidal water. Thus, the health of marsh biota will depend in part on the state of water and sediment quality in the Bay at the time the marshes begin to vegetate to become highly productive habitats capable of attracting dense wildlife populations. The salt ponds are to be restored in phases that, at any point in time, will yield restored marshes at different stages of development. Some ponds may not reach water elevations that will allow vegetation to take hold for decades, either because of low initial elevation, insufficient sediment supply, or sea level rise. Thus, any reductions in contaminant sources to the South Bay over the coming years will benefit the SBSPRP, and any increases will be detrimental to the Project.

3.2.3 The Effect of Bird Populations on Water Quality in Managed Ponds

Some of the former salt ponds will not be restored to tidal action and, instead, will be managed to support migratory water birds in winter and resident breeding birds in summer. The Pacific Flyway populations of water birds that migrate through the South Bay will encounter less managed pond acreage and more marsh acreage over time as the SBSPRP changes the landscape. The restored marshes will provide habitat for some species in the form of pannes (unvegetated tidal ponds in the marsh) and sloughs (tidal channels), yet many birds will continue to rely on the resources of the remaining managed ponds to build up fat reserves during winter and to support reproduction during the breeding season. These individuals may be crowded by necessity into dense aggregations in the remaining managed ponds.

The effect of dense bird populations on water quality is variable, with the outcome depending on the details of the situation. Bird feces can degrade water quality through eutrophication or introduction of pathogens. Some studies of eutrophication of water bodies by avian populations indicate that water birds contributed a significant proportion of total nutrients and degraded water quality (Manny et al. 1994). In one investigation of a shallow urban lake, the bird feces comprised nearly all the phosphorus loading (Scherer et al. 1995). Large bird populations can also introduce substantial quantities of bacteria to water bodies from their droppings (Valiela et al. 1991; Levesque et al. 1993; Graczyk et al. 2008), although in some studies water quality was not affected by bacteria from bird feces (Levesque et al. 2000). The body size of the bird, the density of the population, and the ability of the water body to dilute avian feces are important considerations in determining whether water quality might be impacted.

In a recent study of two managed ponds in the SBSPRP, it was found that the relationship between bird use and water quality was seasonal and related to how pond hydrology was managed. Shellenbarger et al. (2008) found that fecal indicator bacteria concentrations in managed ponds were higher in summer than in winter, despite bird abundance on the ponds being 10 times greater in winter. The researchers concluded that, in the summer, water from an adjacent slough with poor microbial water quality entered the ponds and increased concentrations of these bacteria. In the winter, bird feces probably contributed large quantities of fecal indicator bacteria. Whether this pattern holds true for other ponds in the SBSPRP will depend on factors such as hydrology, intensity of bird use, and quality of the source water to the ponds.

4 Future Changes that May Affect Water Quality

Other future changes that could affect water quality will include some changes that are beyond the control of the SBSPRP managers. Located in the transition area between the upland watersheds and the Bay, the Project has an intimate connection to the water quality of both areas. Increasing urbanization and human population in the San Francisco Bay Area and the sea level rise that is accompanying global climate change are likely to significantly impact the waters of the Bay and of the Project. The human population of the Bay Area is expected to rise by 2 million over the next 30 years, which is an increase to 140% of the current population (ABAG 2007). Given that human activities have caused nearly all the water quality impairment that is now observed in San Francisco Bay, the advent of 40% more people in the local watersheds may adversely affect water quality, despite efforts to minimize their impact. New contaminants will continue to emerge, such as pesticides, pharmaceuticals, flame retardants, and nanotechnology waste. The effects of population growth may also include greater peak loads of storm water, greater absolute quantity of contaminants in storm water, greater volume of treated wastewater, more atmospheric loading of contaminants from vehicle exhaust, and more trash. Many strategies are being developed to counteract these effects of population growth, including better management and recycling of storm water, reduction of pollutant sources, reuse of wastewater, and better trash management.

Climate change is also likely to affect the SBSPRP, and changing precipitation patterns may be the aspect of climate change that most affects water quality in the Project. Pulses of water are predicted to come from the Sierra Nevada mountains earlier in the year, in winter/spring instead of spring/summer, due to earlier snowmelt and to precipitation falling as rain instead of snow (Miller et al 2003; Dettinger et al. 2004; Vanrheenen et al. 2004). This change would probably result in greater flooding in spring and larger loads of contaminated sediment being carried to the Bay during these events. Wetter springs are predicted to be followed by drier summers, which will reduce natural dry-season inflows to the Bay. Therefore, it is likely that dry-season flows will have a greater concentration of contaminants (less total water volume) and will be more dominated by wastewater than in the past.

5 Recommendations

A general recommendation for the SBSPRP managers, and for others managing wetland restoration at a regional scale, is to practice adaptive management and ongoing monitoring for water quality, particularly bioaccumulation of contaminants in the food web. The four main questions that need to be answered with adaptive management and ongoing monitoring are as follows:

- What are the present levels of contamination in locations to be restored and in adjacent habitats?
- What is the effect of different types of restoration on contaminant exposure of wildlife in restored areas and adjacent habitats?
- What restoration approaches in terms of habitat type, location, water management, etc., can minimize the bioaccumulation of contaminants in the food web locally and regionally?
- What is the effect of restoration on the South Bay sediment budget and long-term trends in South Bay regional contamination?

We recommend an approach to address these uncertainties that includes the following elements.

1. Most importantly, a long-term regional program of monitoring and research is needed that assesses contaminants prior to, during, and after each restoration action within the larger SBSPRP. Emphasis should be placed on exposure and effects in biota, along with ongoing synthesis of the information obtained into conceptual and numeric models that describe contaminant dynamics at local and regional scales.

Long-term monitoring should be performed to ascertain the impact of restoration actions on water quality relative to ambient condition on local and regional scales. This monitoring should include sampling of concentrations in sport fish as an index of human exposure and of marsh, pond, and Bay wildlife that are appropriate indices of exposure in the food webs of each habitat. Monitoring of other food-web or ecosystem components may also be useful in establishing long-term trends and spatial patterns, and biosentinel organisms with high-site fidelity will be useful in differentiating relative mercury bioavailability at fine spatial and temporal scales. Water quality monitoring should be conducted in all of the habitat types that are part of each restoration plan, and tidal marsh, managed pond, and intertidal mudflat habitats will be of particular interest to monitor with respect to methylmercury.

Detailed surveys should precede individual pond restoration projects to document existing concentrations of mercury and other contaminants and to evaluate the potential for increased food-web accumulation. Determining the impact that a restoration project has will depend on the availability of baseline information collected prior to the project start. For restoration projects likely to mobilize or erode large quantities of sediment, preliminary studies are needed to evaluate contaminant concentrations in surface and buried sediments and in the water and sediment supply. The effect of remobilized contaminated sediment accumulating in restored areas and adjacent habitats must be monitored.

Long-term monitoring of other water quality indicators will also be needed. To ensure that contaminants do not interfere with the health of wildlife in restored habitats, the Project will need information from toxicity testing to assess the effects of current-use pesticides and other non-persistent contaminants and to ascertain general trends in contaminant concentrations in South Bay.

Studies performed in association with restoration should contribute to development of a conceptual understanding of mercury cycling in Bay wetlands that allows prediction of bioaccumulation in restored habitats, including different subhabitats within wetlands. Food-web monitoring should be coupled with strategic process studies crafted to disclose the mechanisms of variation in methylmercury bioaccumulation within and among tidal wetlands. This knowledge will provide the foundation for environmental managers and engineers to develop designs that minimize the impact of restoration activities. High priority should be given to examining the effects of restoration on bioavailability and net methylation rates, as these processes have the potential to increase methylmercury exposure in biota.

Conceptual and numeric models of contaminant fate are required on local and regional scales. These models provide a framework for organizing the current state of knowledge and for defining uncertainties, and they should continue to be updated with new information.

2. Studies are needed that provide better information on the sensitivity of species facing the greatest exposure to methylmercury and other contaminants.

More information on sensitivity to methylmercury of California Clapper Rails, terns, and harbor seals is a priority. Piscivorous species are also highly exposed to

PCBs, dioxins, PBDEs, and other persistent organic chemicals, yet the sensitivity of these species to these individual chemicals and combinations of them is not well known.

3. Development of a sediment-transport dynamics model and sediment budget is required that accurately describe sediment mixing, deposition, and erosion in South Bay.

Numerical models are needed to predict the sources and quality of sediment that are supplied to restored wetlands, and the impacts of the Project on sediment erosion and possible contaminant remobilization at a regional scale.

6 Summary

The SBSPRP is an extensive tidal wetland restoration project that is underway at the margin of South San Francisco Bay, California. The Project, which aims to restore former salt ponds to tidal marsh and manage other ponds for water bird support, is taking place in the context of a highly urbanized watershed and an Estuary already impacted by chemical contaminants. There is an intimate relationship between water quality in the watershed, the Bay, and the transitional wetland areas where the Project is located. The Project seeks to restore habitat for endangered and endemic species and to provide recreational opportunities for people. Therefore, water quality and bioaccumulation of contaminants in fish and wildlife is an important concern for the success of the Project.

Mercury, PCBs, and PBDEs are the persistent contaminants of greatest concern in the region. All of these contaminants are present at elevated concentrations both in the abiotic environment and in wildlife. Dioxins, pyrethroids, PAHs, and selenium are also problematic. Organochlorine insecticides have historically impacted the Bay, and they remain above thresholds for concern in a small proportion of samples. Emerging contaminants, such as PFCs and non-PBDE flame retardants, are also an important water quality issue. Beyond chemical pollutants, other concerns for water quality in South San Francisco Bay exist, and include biological constituents, especially invasive species, and chemical attributes, such as dissolved oxygen and salinity.

Future changes, both from within the Project and from the Bay and watershed, are likely to influence water quality in the region. Project actions to restore wetlands could worsen, improve, or not affect the already impaired water quality in South Bay. Accelerated erosion of buried sediment as a consequence of Project restoration actions is a potentially serious regional threat to South Bay water and sediment quality. Furthermore, the planned restoration of salt ponds to tidal marsh has raised concerns about possible increased net production of methylmercury and its subsequent accumulation in the food web. This concern applies not only to the restored marshes, but also to the South Bay as a whole, which could be affected on a regional scale. The ponds that are converted to tidal marsh will sequester millions of cubic meters of sediment. Sequestration of sediment in marshes could remove contaminated sediment from the active zone of the Bay but could also create marshes with contaminated food webs. Some of the ponds will not be restored to marsh but will be managed for use by water birds. Therefore, the effect of dense avian populations on eutrophication and the introduction of pathogens should be considered. Water quality in the Project also could be affected by external changes, such as human population growth and climate change.

To address these many concerns related to water quality, the SBSPRP managers, and others faced with management of wetland restoration at a regional scale, should practice adaptive management and ongoing monitoring for water quality, particularly monitoring bioaccumulation of contaminants in the food web.

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