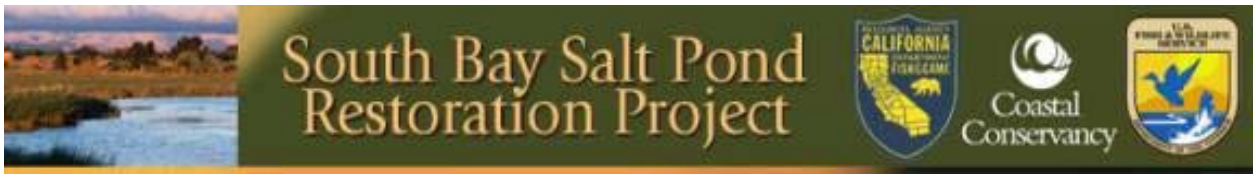

M E M O R A N D U M

TO: Members of the South Bay Salt Ponds Restoration Project Management Team

FROM: Marc Beutel, Brown and Caldwell; Khalil Abu-Saba, Larry Walker and Associates

DATE: August 2, 2004

RE: **Mercury Technical Memorandum – Final Draft**

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1. EXECUTIVE SUMMARY

The South Bay Salt Pond (SBSP) Restoration Project (Project) provides a unique opportunity to restore habitat for a wide variety of biota. However, coupled with this opportunity is a concern that the restoration could enhance the cycling of mercury into biota by increasing the rate of mercury methylation, the microbial conversion of inorganic mercury to methylmercury (MeHg). MeHg can accumulate in biota, particularly at the top of aquatic food webs. While the factors that control mercury methylation are not fully understood, wetlands are known to be significant sites of microbial methylation and important areas for MeHg production and transfer to aquatic food webs.

The initial objective of mercury assessment for the Project has been to inform the alternatives formulation and selection process to minimize mercury impacts on biota and to establish a framework for adaptive management. However, the state of the science regarding mercury cycling in the San Francisco Bay currently limits the ability to make definitive recommendations at this time. Two primary mechanisms that need to be better understood to manage mercury in the SBSP Restoration Project are MeHg production in sediments and the subsequent accumulation of MeHg into the base of the food web. In addition, the effects of two key factors on mercury methylation and bioaccumulation need to be further evaluated including salinity levels and habitat types (e.g., tidal marsh, tidal flats, and managed ponds). Finally, the threshold level of methylmercury that negatively impacts biota is still in question for many wildlife species of interest. The SBSP Initial Stewardship Plan (ISP) presents an opportunity to learn more about mercury cycling through monitoring and adaptive management of the Project area, as do other ongoing, planned or future studies and projects in the San Francisco Bay-Delta.

The text below summarizes the results of pertinent reports on existing mercury levels in the Protect area, presents a Conceptual Model of mercury cycling and a Mercury Management Matrix developed specifically for the SBSP Project, and makes recommendations regarding the Project relative to the management of mercury. The Project Team would like to acknowledge the numerous stakeholders who attended the two mercury meetings held as part of this project, and those stakeholders who provided guidance and input on this document.

Based on the findings presented in this memorandum, mercury is not seen as a "fatal flaw" for the restoration of the SBSP. While the science of mercury cycling is still under development, it is likely that ongoing and future studies of mercury cycling in the San Francisco Bay-Delta will provide managers with the information needed to design and manage the restoration to minimize the impacts of mercury on biota, while obtaining important habitat benefits for wildlife.

Mercury in Sediments. Three recent studies on mercury in pond sediments were reviewed including a site assessment which examined total mercury (THg) in sediment in six Alviso Ponds (Maurer and Adelsbach 2002), THg and MeHg monitoring of sediments in Pond 8A (Light Air and Space Construction (LA&S) 2004), and recent THg and MeHg monitoring in sediment from 16 ponds throughout the Project area evaluated as part of the ISP.

Average levels of THg in sediments in the Alviso Ponds range from 0.2 to 2.0 parts per million (ppm) dry weight, and commonly exceed ambient levels in the San Francisco Bay (0.3 to 0.5 ppm). Peak levels of THg ranging from 1.6 to 4.5 ppm were observed in individual sediment samples from ponds near the Alviso Slough including A7, A8, A11, A12, and A13. In contrast, THg levels in sediments from the Baumberg/Eden Landing and Redwood/West Bay Ponds range from 0.05 to 0.15 ppm and are consistently below ambient levels for the Bay. Existing data show that ponds that are under the influence of the Alviso Slough, the discharge point for the Guadalupe River, have elevated THg levels in sediment that are, on average, two to five times ambient levels.

Average MeHg levels in surface sediments in 10 of the 16 monitored ponds are below approximately 2 parts per billion (ppb) dry weight. Sediments from three ponds (A3N, A7 and B11) have relatively high levels of MeHg ranging from around 5 to 11 ppb. These levels are higher than MeHg concentrations found in sediments in the open Bay which are typically less than 1 ppb. Levels in Pond sediment are more typical of MeHg concentrations observed at the Bay Margins and in impounded sediments in the Guadalupe River watershed (1 to 30 ppb) (Heim and others 2003; Thomas and others 2002).

Based on the entire dataset, there is little correlation between THg and MeHg, thus THg in sediment does not appear to be a key factor controlling MeHg levels in sediments. Without complementary data on MeHg levels in biota, it is not possible to assess whether the high MeHg levels in pond sediment correlate with high levels in biota.

Mercury in Biota. Two recent studies on mercury in pond biota were reviewed including an environmental site assessment which examined mercury levels in snails in four Alviso Ponds and in fish in six Alviso Ponds (Maurer and Adelsbach 2002), and recent bird egg monitoring in four ponds in the Project area as part of the CALFED Bay-Delta Mercury Project (Schwarzbach and Adelsbach 2003a).

Existing data show that mercury levels (wet weight) in biota in the Alviso Ponds generally range from 0.1 to 0.3 ppm in snails, 0.1 to 0.5 ppm in fish, 0.3 to 0.4 ppm in non-fish eating bird eggs, and 1.2 to 1.6 ppm in fish eating bird eggs. Mercury concentrations in snails were similar to invertebrates elsewhere in the

South Bay, but high enough to promote biomagnification into higher trophic levels. The majority of fish in some ponds (A3N, A9, and A10) exceeded the EPA fish tissue criterion of 0.3 ppm for the protection of human health, which in some cases may not be stringent enough to protect the health of wildlife. However, fish tissue levels in ponds, excluding A9, are comparable to levels in fish throughout the Bay-Delta, suggesting that fish in Pond A9 are accumulating mercury at a relatively elevated rate.

Mercury levels in bird eggs in the Project area are consistently elevated compared to samples collected from other sites in the Bay-Delta. Thus, birds in the ponds appear to be accumulating mercury at relatively elevated rates. In addition, levels in upper trophic level bird eggs from Ponds A7 and A16 may exceed levels which could affect reproduction success (~0.5 ppm). It is difficult to link ISP sediment monitoring with biota monitoring since monitoring sites differed. In addition, seasonal and annual patterns of mercury cycling may make the comparison of data from different studies of limited value. It is worth noting that some ponds with elevated levels of mercury in fish (A3N, A10) and birds (A7, A16) monitored by the United States Fish and Wildlife Service and the United States Geological Survey did not consistently exhibit high MeHg levels in sediment monitored during the ISP (A3N, 6.8 ppb; A10, 1.5 ppb; A7, 4.8 ppb; A16, 1.4 ppb).

Conceptual Model and Management Matrix. This memorandum presents a Conceptual Model for mercury cycling in the SBSPs. The model incorporates the framework presented in the San Francisco Bay mercury TMDL in which mercury in upper trophic level biota is a function of three factors: mercury loads and the resulting mercury concentrations in sediment, net mercury methylation rates, and MeHg bioaccumulation. Within the context of those three key factors, the SBSP Conceptual Model evaluates mercury cycling in three primary restoration habitats (tidal flats, tidal marsh, and managed ponds) under various environmental conditions (e.g., low versus high tide in marshes and flats; low versus high salinity in ponds). The model also includes linkages between mercury cycling and bioaccumulation/ biomagnification in biota specific to each habitat type.

In contrast to the other habitats, mercury cycling in tidal flats is strongly impacted by tides via turbulent advection and cycles of drying and wetting which affect redox levels in surficial sediments. Tidal marsh is unique in that this habitat supports a diverse community of marsh plants that can greatly effect the cycling of compounds in sediments, including mercury. Ponds are much less hydrodynamically active than flats and marsh, and mercury cycling is more dependent on processes within the water column. Ponds can also have simpler food weds, particularly at higher salinities where fish are excluded. These characteristics result in dramatically different rates of and controls on mercury methylation and accumulation between habitats.

The memorandum also presents a Mercury Management Matrix which is a preliminary attempt to link mercury cycling with potential management strategies to minimize mercury contamination in biota. The Matrix is made up of three tables (mercury loading/sediment concentration, net methylation, and bioaccumulation/ biomagnification) that include science information related to mercury cycling processes, controls and mechanisms, and outline potential design/construction and operations strategies to manage mercury impacts. As researchers develop a greater understanding of mercury cycling and the

Matrix is updated, it could act as a tool to optimize the design and management of tidal marshes in the Bay-Delta.

Recommendations. Based on an evaluation of existing mercury data for the SBSP, and on the SBSP Conceptual Model for mercury cycling, and the Mercury Management Matrix presented in this memorandum, the following recommendations are made regarding the minimization of mercury impacts from the SBSP Restoration Project. These recommendations are discussed in detail in Section 5 of this memorandum.

1. Do not consider mercury a "fatal flaw" of the restoration Project.
An examination of existing mercury data suggest that mercury levels in sediments and biota in the Project area are elevated as a result of historic and ongoing mercury pollution to the San Francisco Bay. Mercury levels observed in the Project area are, in general, similar to levels observed at other mercury-impacted sites in the San Francisco Bay-Delta. While the scientific understanding of mercury cycling is still under development, adaptive management of the Project will help to mitigate mercury impacts and minimize the potential for accumulation of mercury in biota over the long term.

2. Use results of ISP and other monitoring efforts to inform mercury data gaps and adaptive management process.
The ISP and other studies present opportunities to learn more about mercury cycling through monitoring and adaptive management. Recommendations for future monitoring include:

- *Perform baseline and long-term monitoring of mercury in biota.*
- *Monitor mercury methylation across salinity gradient in managed ponds.*
- *Monitor mercury methylation across restoration habitat types.*
- *Monitor "bioavailability" of inorganic mercury in sediment.*
- *Monitor upcoming near-term changes in pond operations.*
- *Monitor existing and recently created marsh habitat to predict future outcomes.*

3. Develop and prioritize testable hypotheses concerning mercury cycling.
To allow for effective adaptive management decisions as the Project is implemented, key testable hypotheses regarding mercury cycling need to be further evaluated and prioritized as part of the Project's Science Plan. This ranking of testable hypotheses will act as a framework for the implementation of monitoring and experimental studies that will successfully inform adaptive management decisions.

4. Pending further study, consider potential design components to mitigate mercury impacts.
Pending further study, a few design strategies may have the potential to limit mercury exposure to biota. Strategies that should be considered and evaluated include: covering sediment in ponds with highest methylation potential using natural sedimentation processes or dredged material, and maintaining substantial areas of managed ponds in the SBSP Project area. These preliminary design strategies are discussed in more detail in Section 6 of this memorandum.

5. Coordinate with comparable pilot projects in the San Francisco Bay-Delta.
Mercury studies and monitoring are being conducted in other habitats around the San Francisco Bay-Delta by a number of researchers and agencies. The Project Team should track the results of these monitoring efforts to inform the adaptive management of the Project, and hold dialogue with investigators during the development of study goals to help implement studies that will address data gaps specific to the Project.

6. Refine SBSP Conceptual Model for mercury and the Mercury Management Matrix.
The Conceptual Model and Mercury Management Matrix proposed in this memorandum are based on available scientific information on mercury cycling. Future studies and monitoring efforts should be used to revise and refine the Model and Matrix, and modify and/or strengthen applicable recommendations.

7. Refine Sediment Quality Guidelines for the beneficial re-use of sediments, focusing on mercury.
The draft Long-Term Management Strategy for dredged sediments (LTMS) provides guidelines for the reuse of dredge sediments, but the LTMS guidelines were not developed with a sophisticated conceptual model for mercury bioaccumulation. The Project Team should track the development of any sediment guidelines that could impact the Project, and provide input to regulators as appropriate to update the draft LTMS guidelines (San Francisco Bay Regional Water Quality Control Board 2000).

2. INTRODUCTION

Over the past century, human activities have resulted in a dramatic loss of tidal wetlands and associated habitats in the South San Francisco Bay. Less than 20 percent of historical tidal marsh in the South Bay remains intact (Goals Project 1999). The South Bay Salt Pond (SBSP) Restoration Project (Project) will restore over 15,000 acres of salt production ponds to a mixture of habitats including tidal marsh, tidal flats, and managed ponds. The Project provides an unparalleled opportunity to restore vast areas of habitat for a wide variety of biota including native special status species, such as the California Clapper Rail and the Salt Marsh Harvest Mouse. However, coupled with the opportunity is the concern that the restoration could enhance the cycling of mercury into biota.

Human activity throughout the San Francisco Bay-Delta and its tributaries over the past two hundred years has resulted in dramatic disruptions to the ecology of the Bay (Trulio and others 2004; Wiener and others 2003). Historic mining for gold and mercury has led to the prolonged release of large amounts of mercury to the Bay-Delta. The Sacramento River Watershed is the primary source of mercury to the Bay-Delta (Wiener and others 2003). The Guadalupe River watershed, which includes the historic New Almaden mercury mine, is a major source of mercury-laden sediments to the South Bay (Tetra Tech Inc. 2004; Trulio and others 2004).

A major concern with mercury pollution to the Bay-Delta is the accumulation of methylmercury in biota, particularly at the top of aquatic food webs. Mercury occurs in many forms, but methylmercury is the form which poses the highest bioaccumulation risk. Throughout this report, mercury concentrations in different media (water, sediments, and biota) will be distinguished as methylmercury (MeHg) or total mercury (THg), which is the sum of both inorganic and methylated mercury.

Elevated levels of MeHg can adversely affect the health and fitness of fish and birds. Some studies suggest that elevated MeHg levels are negatively impacting reproductive success of aquatic birds in the Bay-Delta (Schwarzbach and Adelsbach 2003a). Elevated MeHg levels in fish can also result in mercury exposure in humans who consume contaminated fish (National Research Council Committee on the Toxicological Effects of Methylmercury 2000). MeHg is produced in aquatic ecosystems via the methylation of inorganic mercury by microorganisms (Benoit and others 2003). The rate of methylation is a complex function of an array of variables including mercury levels, mercury speciation, microbial activity, sulfate levels, salinity, dissolved oxygen, oxidation reduction potential, organic carbon, turbidity, solar radiation, and vegetation type. While the interaction of these variables are not fully understood, wetlands are known to be significant sites of microbial methylation (Marvin DiPasquale and others 2003) and potentially important sources of MeHg to aquatic food webs (Wiener and others 2003). For this reason, there is concern that the restoration of tidal marsh in the South Bay could enhance the exposure of biota to mercury. The cycling of mercury in the aquatic habitats of the South Bay is discussed in more detail in Sections 4 and 5.

This memorandum first presents a summary of existing mercury data collected to date in the Project area. Data include mercury levels in birds, bird eggs, and fish compiled by the United States Geological Survey

(USGS), the United States Fish and Wildlife Service (USFWS), and the San Francisco Bay Bird Observatory (SFBBO). Other data also include mercury in pond sediment collected as part of the environmental site assessment conducted in 2002 and, more recently, as part of the ISP monitoring effort. This memorandum presents a Conceptual Model for mercury cycling in the Project area, taking into account the type of habitat planned for the restoration, including tidal marsh, tidal flats, low-salinity managed ponds, and moderate-salinity managed ponds. Finally, coupled with the Conceptual Model is the Mercury Management Matrix. This matrix attempts to link mercury cycling processes with potential pond and wetland design and operational strategies that could minimize net methylation of mercury, thereby reducing the potential for the incorporation of mercury into biota.

3. EXISTING MERCURY DATA FOR THE SBSP

This section summarizes existing data on mercury levels in the SBSPs from the four sources listed below. Note that it was beyond the scope of this memorandum to fully access the quality of this data, but the data was judged to be of sufficient quality for the planning level discussions presented in this technical memorandum. If interested, readers are urged to consult the primary sources of the data in order to evaluate sampling methods and quality control and assessment.

- Recent THg and MeHg levels in pond sediment evaluated as part of the ISP by Lisa Stallings of Life Sciences! and Keith Miles of the USGS.
- Recent THg and MeHg levels in Pond 8A sediment evaluated by the Santa Clara Valley Water District (SCVWD) (Light Air and Space Construction (LA&S) 2004).
- Mercury levels in bird eggs reported as part of the CALFED Bay-Delta Mercury Project (Schwarzbach and Adelsbach 2003a).
- Mercury levels in sediment, snails and fish evaluated as part of the environmental site assessment performed by the USFWS prior to final acquisition of the ponds (Maurer and Adelsbach 2002).

3.1 Sediment Mercury Levels in Ponds

The section presents an overview of THg and MeHg levels in pond sediments. Existing data show that ponds near the Alviso Slough (e.g., A7, A8, A11, A12, and A13), the discharge point for the Guadalupe River, have elevated THg levels in sediment that are, on average, two to five times ambient levels in the San Francisco Bay. Average MeHg levels in surface sediments in three ponds (A3N, A7 and B11) are relatively high, but MeHg levels do not appear to correlate strongly with THg Levels.

Regarding regulatory guidelines for mercury in sediment, the State legislature directed the State Water Resources Control Board to develop Sediment Quality Objectives (SQOs) for pollutants in 1989, through the Bay Protection and Toxic Cleanup Program (BPTCP). Monitoring programs funded through BPTCP have provided data for development of SQOs, but a workplan for SQO development was only recently implemented in 2002. Guidance for SQOs for bioaccumulative pollutants is not expected in the first phase of implementation (see water quality link at www.swrcb.ca.gov). At present, the only regulatory objective for mercury in sediments is the Basin Plan narrative objective for bioaccumulation:

> *Controllable Water Quality Factors shall not cause a detrimental increase in the concentrations of toxic substances found in bottom sediments or aquatic life. Effects on humans, wildlife, and aquatic organisms shall be considered.*

As part of its role in the Long-Term Management Strategy for dredged sediments (LTMS), the San Francisco Bay Regional Water Quality Control Board developed draft guidelines for placement of dredged sediments in wetlands. The draft guidelines currently call for 0.43 ppm or less in sediments that are placed in surface of restored wetland (“cover” material). This draft guideline is based on the ambient concentration of mercury in Bay sediments. The “noncover” guidelines call for 0.71 ppm mercury or less

in sediments placed at the foundation of wetlands ("noncover" material). This draft guideline is based on the National Oceanic and Atmospheric Administration's median effects concentration for toxic effects in sediments.

3.1.1 USFWS Site Assessment Results.

In July 2002, the USFWS sampled sediment cores collected from Ponds A1, AB1 (identified as Pond B1 in the Maurer Report), A5, A9, A10 and A16 within the Alviso ponds system (Maurer and Adelsbach 2002). For each pond, three sediment samples were collected and each of these samples was a composite of three individual sediment samples. The sediment samples represented the upper 10 to 15 centimeters (cm) of surficial sediment. Each composite sample was analyzed for several metals, including THg. THg concentrations ranged from 0.2 to 1.2 parts per million (ppm) dry weight (all sediment mercury concentrations presented in this memorandum are in dry weight), with a mean concentration of 0.5 ppm. The mean mercury concentrations for each pond in ascending order are: Pond A1, 0.31 ppm; Pond A5, 0.37 ppm; Pond A9, 0.48 ppm; Pond A16, 0.53 ppm; Pond AB1, 0.56 ppm; and Pond A10, 0.92 ppm.

3.1.2 ISP Monitoring Results.

In October 2003, as part of the ISP, the USGS collected sediment samples from 16 salt ponds within the three pond complexes: Alviso Ponds, A2E, A3N, A7, A8, A10, A11, A12, A13, A14, A16; Baumberg/Eden Landing Ponds, B2, B6A, B11, B12 (note that some reports and maps use the designation "E" in identifying these ponds); and Redwood Ponds/West Bay, R2 and R4. Sediment cores were collected from three locations within each pond, except for Pond A8 where five locations were sampled. More intensive monitoring was conducted in Pond 8 because the pond may have elevated levels of mercury since it is periodically flooded with stormwater and sediments from the Guadalupe River roughly once every ten years. In addition, the pond has an upland region with a different redox regime than the rest of the pond (Stallings 2004). For each location, three subsamples were collected within a ten meter radius and composited. The sediment cores collected represent two depth ranges, from the sediment surface to 5 cm deep (surface samples) and from 15 to 20 cm deep (subsurface samples). Each composited sediment core was analyzed for THg and MeHg.

The ISP study results for THg are shown on Figure 3-1 for the Eden Landing/Baumberg and West Bay/Redwood Ponds and Figure 3-2 for the Alviso Ponds. For comparison purposes, Figures 3-1 and 3-2 include the long-term sediment objective of the draft San Francisco Bay Mercury Total Daily Maximum Load (TMDL) of 0.20 ppm and the current ambient level of THg in sediments in the San Francisco Bay of 0.43 ppm. The ambient level is based on the 85th percentile value for fine-grained sediments from deeper portions of the San Francisco Bay. The San Francisco Bay Regional Water Quality Control Board also suggests that 0.43 ppm, the ambient level of THg in Bay sediments, be used as the "noncover" guideline for the beneficial reuse of dredge materials as wetland surface material (San Francisco Bay Regional Water Quality Control Board 2000). Under these draft guidelines, sediment with less than 0.43 ppm THg could be reused as cover material in restored wetlands, and sediment with less than 0.7 ppm THg could be reused as base material in restored wetlands.

The National Oceanic and Atmospheric Administration (NOAA) has also developed screening guidelines for the reuse of sediment for wetland creation based on an extensive review of sediment toxicity studies (Long and others 1995). They report an effects range low (ERL), the 10^{th} percentile of the effects data and a value below which adverse effects were rarely observed, and an effects range median (ERM), the 50^{th} percentile of the effects data and a value above which adverse effects were frequently observed. The reported ERL and ERM for mercury is 0.15 and 0.71 ppm, and these ranges can be used to evaluate risk associated with sediments in the SBSP Project area.

MeHg results are shown on Figure 3-3 for the Eden Landing/Baumberg and West Bay/Redwood Ponds and Figure 3-4 for the Alviso Ponds. These figures are based on preliminary data compiled by Ms. Lisa Stallings (Life Science!) and Dr. Keith Miles (USGS), who are performing a more comprehensive evaluation of the data. Bars on these figures represent the mean mercury concentrations for each pond. The error bars represent one standard deviation, and while estimated from a relatively small data set (n = 5 for Pond A8, n = 2 for Ponds B11 and B12, and n = 3 for all other sites), the size of the error bars are a rough indicator of the spatial variation in THg in a given pond.

Total Mercury. Based on the ISP data, sediments in Baumberg/Eden Landing and Redwood/West Bay Ponds have average THg levels ranging from 0.05 to 0.15 ppm. These levels are below ambient levels in the Bay (0.43 ppm) and the draft San Francisco Bay TMDL long-term objective (0.2 ppm). They are also below NOAA ERL and ERM screening guidelines (0.15 and 0.71 ppm). Most ponds in these areas have slightly higher levels of mercury in the subsurface sediments except for Pond B11, which appears to have significantly higher THg in surface sediments (0.16 ppm surface versus 0.05 ppm subsurface). Error bars for all ponds are fairly small indicating little spatial variability in THg levels within the ponds.

In contrast to the Baumberg/Eden Landing and Redwood/West Bay Ponds, the ISP data indicate that sediments from Alviso ponds are generally elevated above ambient levels in the Bay of around 0.4 ppm. Average THg levels range from 0.2 to 2.0 ppm in the ponds. There is no obvious trend with respect to mercury in the surface samples versus the subsurface samples. The highest THg levels, as well as the highest level of spatial variability within the ponds, are in Ponds A7, A8, A12 and A13. Peak levels of THg in individual samples were 3.2 and 4.4 ppm in Pond A8, 4.5 ppm in Pond A12 and 3.1 ppm in Pond A13. Ponds that are under the influence of the Alviso Slough, the discharge point for the Guadalupe River, appear to have elevated THg levels in sediment that are on average two to five times ambient levels in the San Francisco Bay (0.3 to 0.5 ppm).

Aside from Ponds A2E, A3N and A14, all ponds had sediment samples that exceeded the NOAA ERL screening guideline of 0.71 ppm. However, since THg and MeHg levels in sediment do not correlate with each other (see MeHg sub-section below), and since MeHg levels are a better indicator of the potential to contaminate biota, the elevated THg levels in some Alviso Ponds are not necessarily a concern from the standpoint of mercury levels in biota. In addition, the elevated THg levels are not dramatically different from other South Bay sediments. Previous studies in the South Bay report sediment mercury concentrations in the lower Guadalupe River ranging from 1 to 10 ppm, with a median of 2.5 ppm, and in the Alviso Slough ranging up to 1.1 ppm, with a median of 0.8 ppm (Maurer and Adelsbach 2002).

Methylmercury. Sediments from the Baumberg/Eden Landing and Redwood/West Bay Ponds generally have average MeHg levels below 2 ppb, with the notable exception in Pond B11, which has MeHg levels in surface sediment of 10.7 ppb. Average MeHg levels in the Alviso Ponds range from 0.8 to 6.8 ppb in surface sediment and from 0.1 to 1.5 ppb in subsurface sediment. In all but one sample, MeHg levels are higher in biologically active surface sediment compared to the subsurface sediment, which is indicative of the biological origins of MeHg. The highest average MeHg levels in surface sediment, as well as the highest level of spatial variability within the ponds, are in Ponds A3N, A7, A11, A12 and A13. Peak levels of MeHg in individual surface sediment were 11.6 ppb in Pond A7, 10.9 and 10.5 ppb in Pond B11, 6.1 ppb in Pond A8, and 6.0 ppb in Pond A12. The MeHg levels in sediments in the Project area are typical of those observed in other studies of the Bay ecosystems (Marvin-DiPasquale unpublished, 2004).

Based on the dataset presented here, there is little correlation between THg and MeHg, thus THg in sediment does not appears to be a key factor controlling MeHg levels in sediments. In fact, the two ponds with the highest averages of MeHg, B11 (10.7 ppb) and A3N (6.8 ppb), had relatively low levels of THg (< 0.2 ppm in B11 and <0.5 ppm in A3N) when compared to other Alviso ponds which have THg levels above 1 ppm (e.g., Ponds A7, A8, A12, and A13). This lack of correlation is typically the case for moderately contaminated areas (Henry and others 1993) and points to the fact that other environmental factors control mercury methylation. Without complementary data on MeHg levels in biota, it is not possible to assess whether the high MeHg levels in pond sediment correlate with high levels in biota. Limited biota data from past studies are discussed in detail below.

Figure 3-1. Total Mercury in Baumberg/ Eden Landing (B) and Redwood/West Bay (R) Ponds

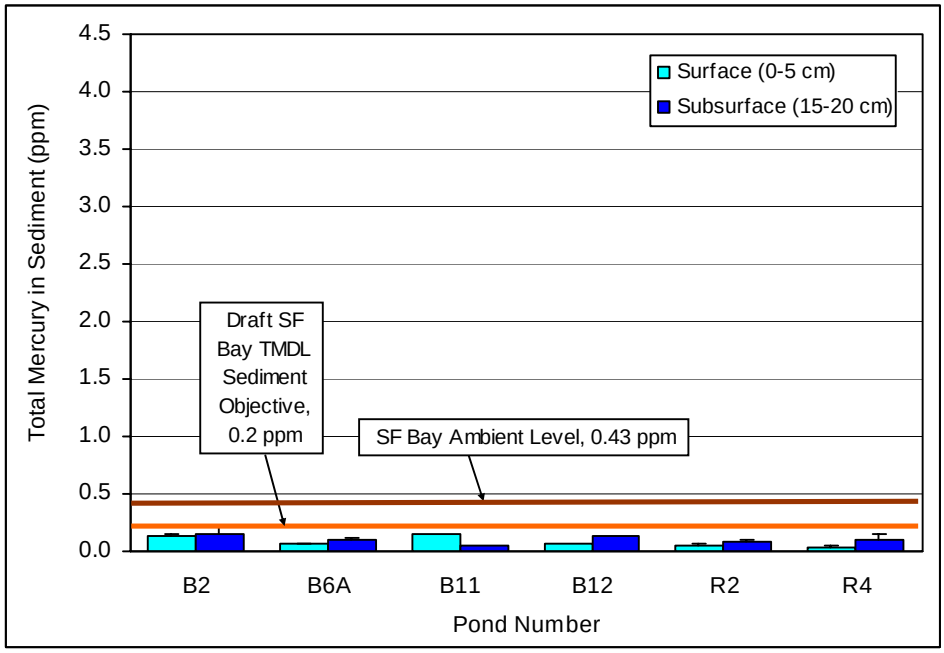
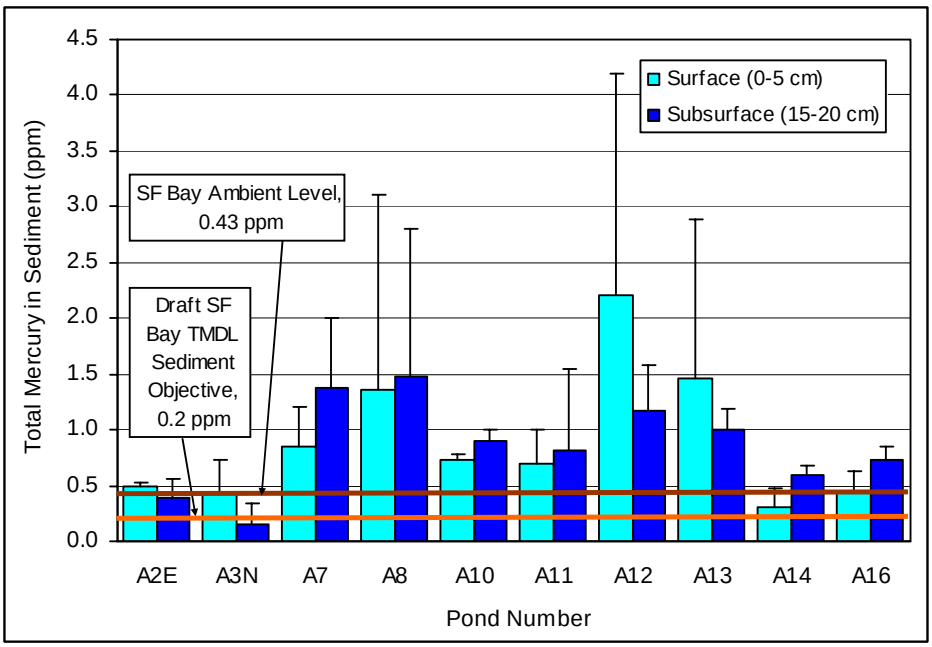


Figure 3-2. Total Mercury in Alviso Ponds



Note: Figures are based on preliminary data collected in October 2003 for the ISP and compiled by Stallings (Life Science!) and Miles (USGS). Bars represent the mean mercury concentrations for each pond and error bars represent one standard deviation (n = 5 for Pond A8, 2 for Ponds B11 and B12, and 3 for all other ponds). Error bars not included for ponds where n < 3.

Figure 3-3. Methylmercury in Baumberg/ Eden Landing (B) and Redwood/West Bay (R) Ponds

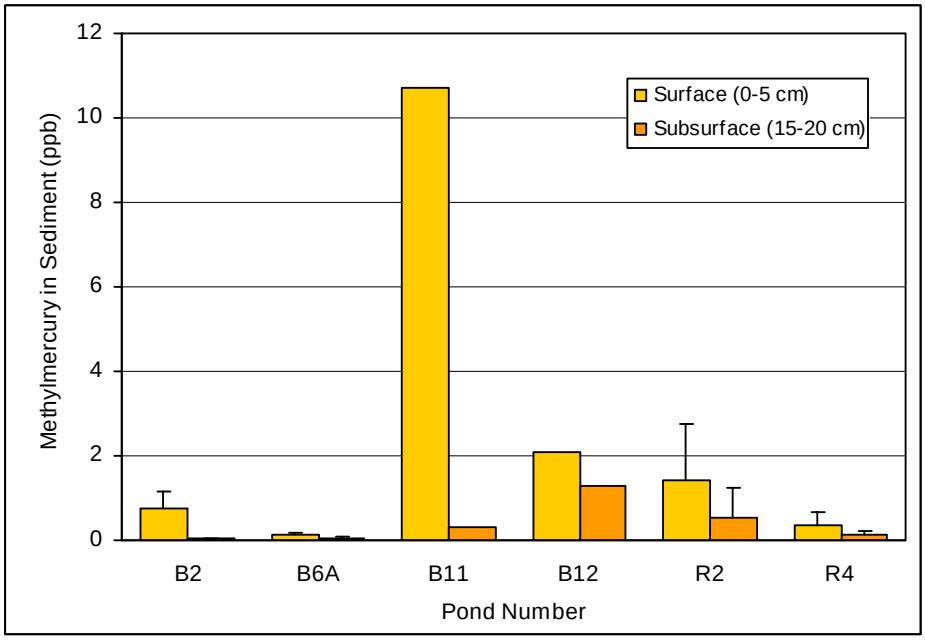
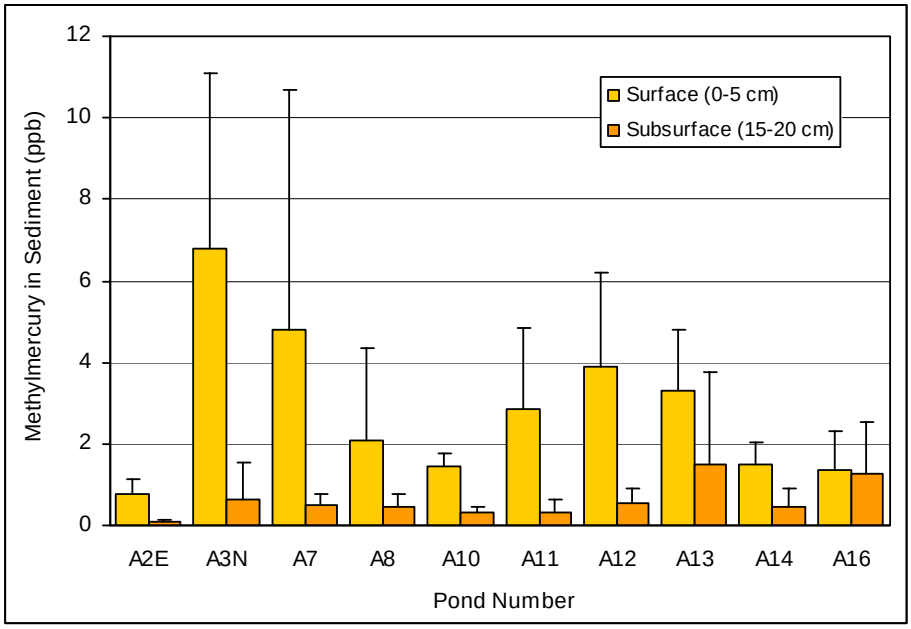


Figure 3-4. Methylmercury in Alviso Ponds



Note: Figures are based on preliminary data collected in October 2003 for the ISP and compiled by Stallings (Life Science!) and Miles (USGS). Bars represent the mean mercury concentrations for each pond and error bars represent one standard deviation (n = 5 for Pond A8, 2 for Ponds B11 and B12, and 3 for all other ponds). Error bars not included for ponds where n < 3.

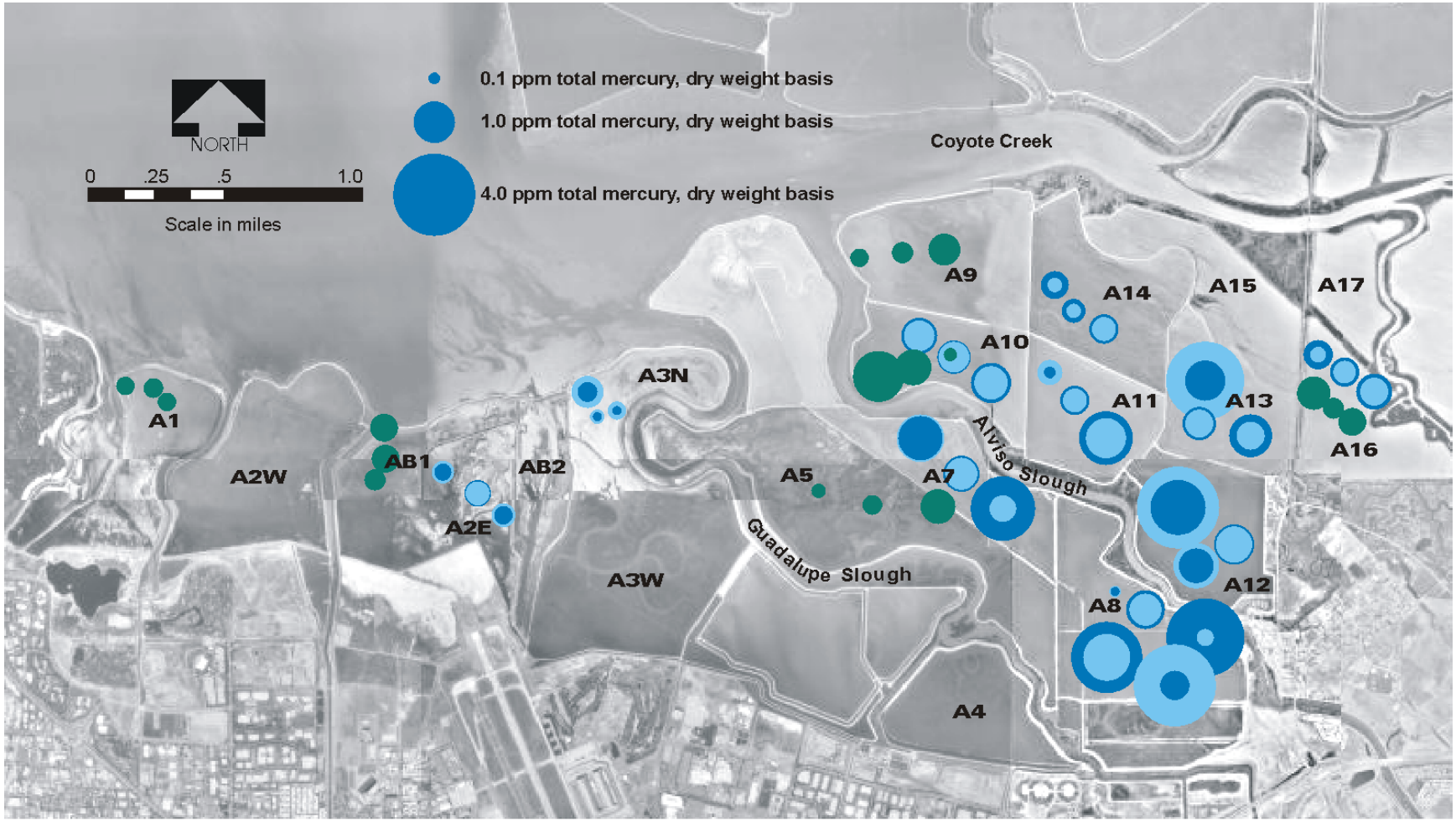
Spatial Distribution of Mercury in Alviso Ponds. Figures 3-5 and 3-6 show the spatial distribution of THg and MeHg in sediments in the Alviso Ponds collected for the ISP. Figure 3-5 (THg) also includes data from the Maurer report. Note that the ISP data points in the Figures do not correspond exactly with the sample location within each pond. As can be seen from the figure, there is a clear spatial pattern with the highest THg sediment concentrations (Ponds A7, A8, A12 and A13) located adjacent to the Alviso Slough, the current discharge point for the Guadalupe River. Note that THg levels collected in the 2002 Maurer study and the 2003 ISP monitoring in Ponds A10 and A16 are relatively similar in magnitude, an observation that supports the validity of the data.

An important factor that affects the spatial distribution of mercury in sediments of ponds adjacent to Alviso Slough is the mode of sediment transport from the Guadalupe River: bed load or wash load. In "bed load" transport, sediment and associated pollutants are transported along the bottom of a channel via scouring during high flow events. In "wash load" transport, sediment and associated pollutants are transported via the advection of suspended sediments. With respect to mercury in sediments in the project area, bed load transport should produce a gradient in sediment mercury levels in impacted ponds radiating from the stormwater point of entry into the pond. In contrast, wash load transport should result in a broader distribution of mercury-laden sediments throughout impacted ponds. Further data analysis, beyond the scope of this memorandum, and perhaps additional monitoring, is required to fully evaluate the relative significance of bed versus wash load as a transport mechanism of mercury to the project area.

A second factor affecting mercury distribution in ponds adjacent to Alviso Slough is the timing of the diking of the Baylands with respect to mercury mining in the Guadalupe River watershed. Ponds diked later would be expected to have higher mercury levels in sediments since these areas were exposed longer and more directly to deposition of mercury-laden sediment. The historical ecology of the Baylands is a major study area of the San Francisco Estuary Institute, and a preliminary assessment suggests that there is a rough correlation between diking history and THg levels in sediment (Grossinger and others 2003). Hydromodification as a result of urban development may have also impacted mercury distribution in the ponds. The original entry point to the Bay for the Guadalupe River was a flood plain between Guadalupe Slough and Alviso Slough, with most of the year-round flow entering through Guadalupe Slough. After installation of significant flood control infrastructure, most likely in the 1950s, the Guadalupe River was channeled into Alviso Slough.

Figure 3-6 shows the spatial distribution of MeHg in the Alviso Ponds. Elevated MeHg levels occur in surface samples from Ponds A3N and A7, and moderate levels occur in Ponds A8, A11, A12 and A13. Ponds A2E, A10, and A14 consistently have low levels of MeHg (< 1 ppb). Some ponds, such as A7 and A8, exhibit high spatial variability. Surface sediments contain higher levels of MeHg than subsurface samples, and there is little correlation between THg and MeHg levels in sediment. Because there are striking spatial differences in MeHg levels in sediments between some ponds, further study of environmental conditions in the ponds during MeHg sampling may indicate what environmental parameters control or enhance mercury methylation. Note that Figures 3-5 and 3-6 do not include West Bay or Baumberg Ponds, nor Pond B11, which also has elevated levels of MeHg in surface sediment (10.9 and 10.5 ppb).

Figure 3-5. Spatial Pattern of Sediment Total Mercury in Alviso Ponds



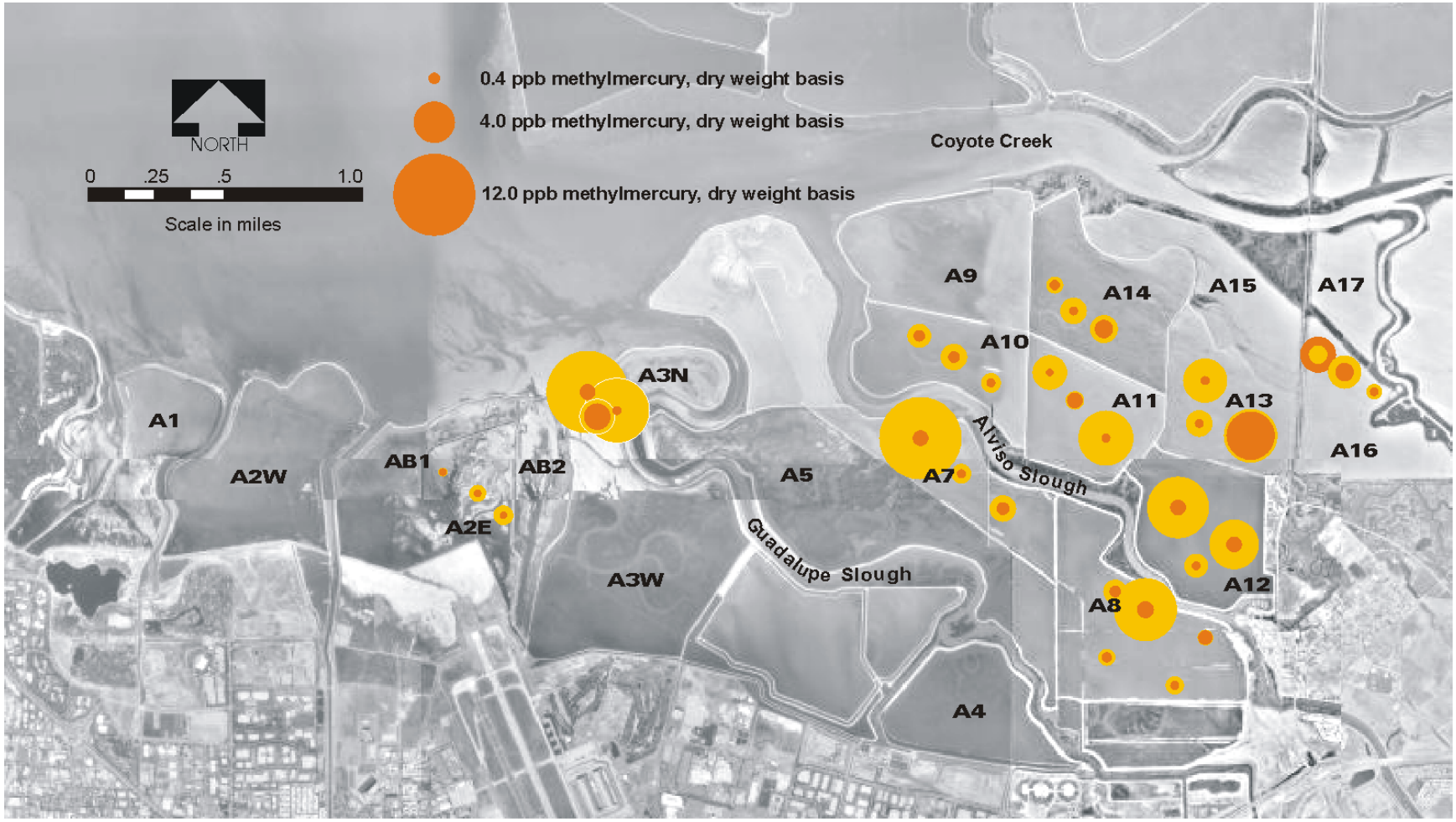
Surface sediment samples (0 to 5 cm) collected during 2003 ISP monitoring. Location of circle does not correspond to location of sample.

Subsurface sediment samples (15 to 20 cm) collected during 2003 ISP monitoring. Location of circle does not correspond to location of sample.

Sediment sediment samples (0 to ~10 cm) reported by Maurer and Adlesbach (2002). Location of circle corresponds to location of sample.

Figure modified from Maurer and Adlesbach (2002)

Figure 3-6. Spatial Pattern of Sediment Methylmercury in Alviso Ponds

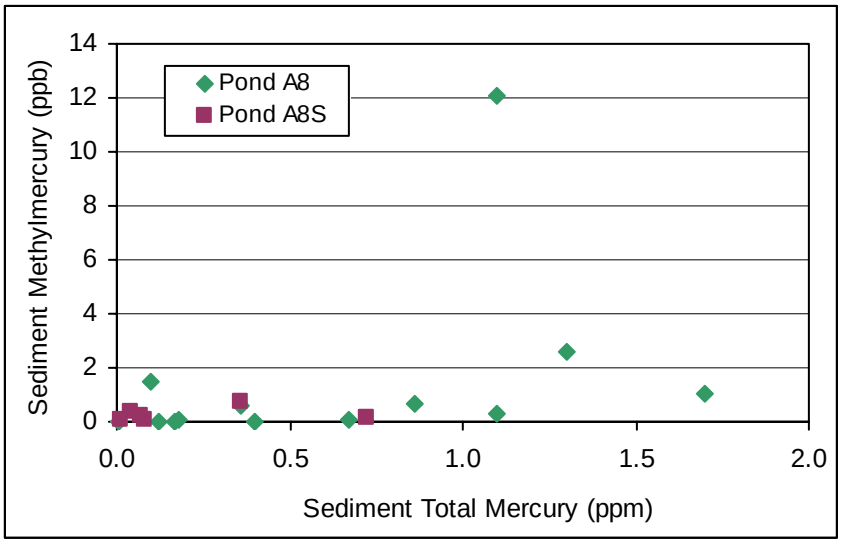


Surface sediment samples (0 to 5 cm) collected during 2003 ISP monitoring. Location of circle does not correspond to location of sample.

Subsurface sediment samples (15 to 20 cm) collected during 2003 ISP monitoring. Location of circle does not correspond to location of sample.

Santa Clara Valley Water District Monitoring. The SCVWD is considering using Pond 8 to assist with flood control along the Alviso Slough, and as a mechanism to capture sediment and limit mercury loading to the South Bay. In order to define baseline conditions, the District collected sediment samples from Pond A8 and Pond A8S, a small pond to the south of Pond A8, and analyzed the samples for THg, MeHg, sulfate, pH, and total organic carbon (Light Air and Space Construction (LA&S) 2004). A total of thirteen samples were collected in Pond A8 (referred to as A8W in the LS&A report) and six samples in Pond A8S (referred to as A8d in the LS&A report). Samples reportedly consisted of the upper six inches of pond sediments. Mercury results are summarized in Figure 3-7. THg ranged from non-detect (0.02 ppm) to 1.7 ppm in Pond A8 and from non-detect (0.02 ppm) to 0.72 ppm in Pond A8S. MeHg ranged from 0.02 to 12.1 ppb in Pond A8 and from 0.045 to 0.76 ppb in Pond A8S. Note that there is no clear correlation between total and MeHg levels in the sediments. THg levels observed in Pond A8 in the SCVWD study (0.62 ± 0.54 ppm, average plus/minus one standard deviation, n = 13) appear to be less than half the levels observed in the ISP study (surface: 1.36 ± 1.74 ppm, n = 5; subsurface: 1.48 ± 1.31 ppm, n = 5). This may be an artifact of high spatial variability within the pond. MeHg levels reported in Pond A8 in the two studies were comparable averaging around 1 to 2 ppb and generally ranging from 0.1 to 2 ppb. Peak MeHg levels ranged from 6 to 12 ppb.

Figure 3-7. Mercury in Pond A8 and A8S



Note: Figure is based on data collected in February 2004 for the SCVWD by Light, Air & Space Construction (LS&A).

3.2 Mercury Levels in Biota

The following section presents an overview of mercury levels in pond biota including snail, fish and bird eggs. Existing data show that mercury levels in snails are similar to invertebrate elsewhere in South Bay.

Mercury levels in fish commonly exceed EPA criterion for the protection of human health. However, excluding A9 which has elevated levels, mercury levels in fish tissue are generally comparable to levels reported in the Bay-Delta. Mercury levels in bird eggs, particularly eggs from fish eating birds, are consistently higher than samples collected from other areas of the Bay-Delta. Thus, birds in the ponds appear to be accumulating mercury at rates faster than birds in other parts of the Bay-Delta.

Regulatory Guidelines for Mercury in Fish. The U.S. EPA has established a water quality criterion for mercury in fish tissue of 0.3 ppm wet weight. This criterion is based on an assumed human consumption rate of 17 g fish per day. Higher consumption rates result in lower target concentrations of mercury in fish to be protective of human health. The SFRWQCB has proposed a fish tissue target of 0.2 ppm based on regional consumption surveys. With respect to the protection of wildlife, the United States Fish and Wildlife Services has proposed a risk-assessment approach to determining targets for mercury concentrations in fish. There is insufficient food web information to apply the USFWS guidance to the project area. The SFRWQCB has asserted in its proposed mercury TMDL that attainment of the human health fish tissue target of 0.2 ppm will also be protective of wildlife, and that it is committed to reviewing this assertion through adaptive implementation of the TMDL.

Mercury Speciation in Biota. Both inorganic mercury and MeHg are accumulated at the base of the food chain by phytoplankton and invertebrates, but MeHg is preferentially assimilated by predators at successive trophic levels (Back and Watras 1995; Baudrimont and others 1997; Gagnon and Fisher 1997; Inza and others 1997). Thus, essentially all of the mercury in the tissue of higher trophic level organisms is present as MeHg (Bloom 1992). Mercury deposited in bird eggs is also primarily MeHg (Wolfe and others 1998). Practically, this means when planning biological monitoring projects or reviewing biological monitoring data, THg measurements can be substituted for MeHg measurements in higher trophic level fish and bird eggs. However, monitoring of lower trophic level biota (e.g., clams and other invertebrates) should evaluate both THg and MeHg since these biota can assimilate both inorganic and methylated mercury.

Interactions with Other Pollutants. Mercury assessments need to consider the presence and effects of other pollutants, which may act additively, synergistically, or antagonistically with mercury. Selenium can reduce acute and chronic mercury toxicity to organisms (Das and others 1985; Lucu and Skreblin 1981; Wang and others 2001). However, more recent studies have shown that while selenium and mercury are antagonists in mature birds, they have additive or synergistic effects on bird embryos (Bischoff and others 2002; Heinz and Hoffman 1998). In other words, selenium in the diet may protect adult organisms from mercury toxicity, but methylmercury impairment of reproductive success will be compounded by selenium. The presence of other pollutants can also increase the toxic effects of methylmercury on biota. Chlorpyrifos, an organophosphorous insecticide commonly found in urban runoff, can increase the overall accumulation of methylmercury by some benthic organisms (Steevens and Benson 2001). Polychlorinated biphenyl compounds (PCBs) can potentially exacerbate developmental impairment caused by methylmercury (Muckle and others 2001).

Mercury Levels in Snails. During the summers of 2000 and 2001, the USFWS evaluated THg levels in snails in Ponds A1, A5, A9 and A10 (Maurer and Adelsbach 2002). Mercury levels in invertebrates, such as snails, are critical since these animals are eaten by fish and birds, and as a result are an important source of mercury into upper trophic level biota. THg levels in snails were highest in Ponds A9, A10, and A5. Mean mercury concentrations in snails in these ponds ranged from 0.24 to 0.27 ppm (all biota mercury concentrations presented in this memorandum are in wet weight). Pond A1 snails had the lowest mean mercury concentration of 0.1 ppm. Mercury concentrations appear to be an order of magnitude below levels that reportedly cause genetic damage in snails (Benton and others 2002). But levels appear to be in the range (0.2 to 0.4 ppm) that can promote subsequent biomagnification of mercury into upper trophic levels and, in the case of some birds, potentially impact breeding success (Barr 1986). However, as noted above, invertebrates can accumulate both inorganic and methylated mercury. Since THg does not differentiate between methyl and inorganic mercury, it is not an ideal indicator of an invertebrate's potential to be a source of contamination to upper trophic level predators. Future studies in the Project area should quantify both THg and MeHg in lower trophic level biota.

Mercury concentrations in SBSP snails were similar to invertebrate concentrations previously detected in South Bay sloughs which ranged from roughly 0.14 to 0.3 ppm (Maurer and Adelsbach 2002). Thus, snails in the SBSP Project area appear to be accumulating mercury at similar rates as snails elsewhere in the South Bay.

The recent ISP sediment mercury monitoring was only conducted in one of the ponds where snails were accessed, thus sediment mercury levels can not be compared to snail levels. In addition, seasonal and annual patterns of mercury cycling may make the comparison of data from different studies of limited value. However, it is interesting to note that the ISP showed that sediments in Pond A10 had somewhat elevated THg levels (~0.8 ppm) but low MeHg levels (~1.5 ppb). Low levels of MeHg in sediments suggest that the pond would not act as a substantial source of mercury to biota.

Mercury Levels in Fish. In the summer of 2001, the USFWS also collected fish samples from within Ponds A1, AB1 (identified as Pond B1 in the Maurer Report), A3N, A9 and A10 (Maurer and Adelsbach 2002). Several species of fish were sampled to represent a wide variety of trophic levels. These species included Longjaw Mudsucker, Northern Anchovy, Jacksmelt, Shiner Perch, Pacific Staghorn Sculpin, and Yellowfin. The mean mercury concentration for all fish sampled from these five ponds was 0.21 ppm, with a range of below 0.02 to 1.4 ppm. The mean mercury concentrations reported for fish from each pond in ascending order were: Pond AB1, 0.06 ppm; Pond A1, 0.11 ppm; Pond A3N, 0.26 ppm; Pond A10, 0.28 ppm; and Pond A9, 0.44 ppm. Note that the highest mercury levels correspond to the ponds located closest to the Alviso Slough where the Guadalupe River discharges to the South Bay. Of all the fish species, Jacksmelt appeared to accumulate mercury at the highest rate, having a mean concentration ranging from 0.3 to 0.7 ppm in Ponds A3N, A9 and A10. Levels in all ponds, excluding Pond A9, are comparable to levels in Bay-Delta fish reported by the San Francisco Estuary Institute Regional Monitoring Program (RMP) (Maurer and Adelsbach 2002).

Like invertebrates, mercury levels in prey fish are important since they are a source of mercury into upper trophic level biota. Levels in fish tissue in the Project area appear to be in the range (0.2 to 0.4 ppm) that can promote biomagnification of mercury into upper trophic levels (Barr 1986). Mercury levels in fish from many ponds exceed the United States Environmental Protection Agency fish tissue criterion for the protection of human health of 0.3 ppm. The percentages of fish exceeding the 0.3 ppm criterion by pond in ascending order are: Pond AB1, 5 percent; Pond A1, 7 percent; Pond A3N, 45 percent; Pond A10, 50 percent; and Pond A9, 77 percent. However, it is important to note that this criterion is not necessarily protective of wildlife health (Russell 2003). Animals differ sufficiently in size, metabolic rate and metabolic pathways, trophic status, and other physiological and life history features that safety thresholds protective of humans may not be entirely protective of wildlife. Mercury levels in fish from the ponds are below reported ranges at which mercury is directly toxic to the fish (5 to 20 ppm) (Schwarzbach and Adelsbach 2003a).

The early life stages of fish are particularly sensitive to methylmercury. Survivorship of fish embryos can be significantly reduced by small amounts of mercury in fish eggs, on the order of less than 10 percent of the level associated with adult toxicity (Wiener and Spry 1996). Since only adult fish tissue has been monitored to date, it is difficult to evaluate impacts on early life stages of fish in the Project area as a result of existing mercury levels.

Note that Ponds A9 and A10, the ponds with the highest levels of mercury in fish, also exhibited relatively high levels of mercury in snails (see previous sub-section above). However, as noted above, results of the recent ISP sediment monitoring shows that Pond A10 has low levels of MeHg (~1.5 ppb), which suggests the pond would not act as a substantial source of mercury to biota. Based on these results, fish throughout the SBSP Project area are accumulating mercury, and in some ponds (e.g., A9) may be accumulating mercury at a faster rate than generally observed in the Bay-Delta.

Mercury Levels in Bird Eggs. Limited data exist concerning mercury levels in birds in the South Bay. In conjunction with the USFWS, SFBBO has monitored mercury concentrations in bird eggs throughout the San Francisco Bay (Adelsbach and Strong unpublished, 2004). Additionally, the USGS and the USFWS have conducted bird egg studies for the CALFED Bay-Delta Mercury Project, which included sampling throughout the Bay-Delta ecosystem (Schwarzbach and Adelsbach 2003a). Bird eggs are a broad indicator of mercury exposure and uptake since birds act as ecological auto-samplers, collecting a range of samples that vary spatially and temporarily throughout their nesting grounds. The area sampled varies according to species. For example, clapper rails are territorial benthic omnivores that will integrate the benthic food chain over a limited area on the order of two acres. In contrast, terns can be more wide ranging and forage over entire embayments. Larger predators such as egrets may sample food chains spanning a distance of tens of miles or more.

In addition, mercury uptake by birds affects the overall reproductive success of birds at far lower levels than it affects overt toxicity in adult birds. For the studies discussed below, various species of bird eggs were sampled to represent different trophic levels. Birds studied in the Project area included non-fish

eating birds including the Black-Necked Stilt, the American Avocet and the Western Snowy Plover, and fish eating birds including the Caspian Tern and the Forster's Tern, were evaluated.

For each species sampled, bird egg mercury concentrations were consistently highest in the samples collected from nests in the South Bay versus samples collected from nests in the general Bay-Delta ecosystem. The mean mercury concentration in eggs of non-fish eating birds was 0.31 ppm for American Avocets from Pond A16, 0.41 ppm for Black-Neck Stilts from Baumberg/Eden Ponds (specific pond not identified in report), and 0.45 ppm for Western Snowy Plovers from Pond A22. In comparison, the mean bird egg mercury concentrations for two of these species across the entire Bay-Delta are 0.19 ppm for American Avocets and 0.31 ppm Black-Neck Stilts. The mean mercury concentration in eggs of fish eating birds was 1.62 ppm for Forster's Terns from Pond A16 and 1.18 ppm for Caspian Terns collected from Pond A7. These levels are the highest mercury levels measured during the study throughout the Bay-Delta ecosystem. In comparison, the mean bird egg mercury concentrations for these species across the entire Bay-Delta are 0.83 ppm for Forster's Tern eggs and 0.93 ppm or Caspian Tern eggs.

Based on limited existing data, mercury levels in bird eggs in the SBSPs exceed average levels in the Bay-Delta and mercury levels are higher in eggs from upper trophic level birds. Thus, birds in the ponds appear to be accumulating mercury at rates faster than birds in other parts of the Bay-Delta, and upper trophic level birds may exceed levels which could affect reproduction success. In addition, while embryotoxic thresholds are variable among bird species, levels of mercury in bird eggs for fish eating birds in Ponds A7 and A16 exceed or approach reported embryotoxic thresholds reported for other bird species such as Mallards (0.79 to 0.86 ppm) (Heinz 1979) and Ring-neck Pheasants (0.5 to 1.5 ppm) (Fimreite 1971). In contrast to these bird species, reproduction success of the common loon was reportedly not affected by egg levels of around 1 ppm (Barr 1986). Levels of mercury in bird eggs for fish eating birds in Ponds A7 and A16 also exceed the Lowest Observed Adverse Effect Concentrations or LOAECs for avian eggs of 0.5 ppm cited in the Maurer report (Maurer and Adelsbach 2002).

It is interesting to note that based on the ISP sediment monitoring, Pond A7 exhibited elevated levels of methylmercury in surface sediments (average of 6.8 ppb) relative to Pond A16 (average of 1.4 ppb), yet mercury levels were higher in avian eggs in Pond A16. This discrepancy points out the complex nature of mercury accumulation in biota.

4. SBSP Conceptual Model for Mercury Cycling

This section presents the SBSP Conceptual Model for mercury which is based on the conceptual model developed in the CALFED Mercury Science Strategy (Wiener and others 2003) and references cited therein. The section opens with a general discussion of the mechanisms that control mercury cycling. The mechanisms are organized according to the scheme presented in the draft San Francisco Bay mercury TMDL. In the TMDL, mercury in top level predator fish is assumed to be a function of three factors: mercury loads and the resulting mercury concentrations in sediment, net mercury methylation rates, and MeHg bioaccumulation. Within the context of those three key factors, the SBSP Conceptual Model evaluates mercury cycling in three primary SBSP restoration habitats: tidal marsh, tidal flats, and managed ponds. In addition to physical and geochemical factors, the SBSP Conceptual Model also emphasizes differences in ecosystem structure between habitat types that may be important in understanding mercury risk to biota.

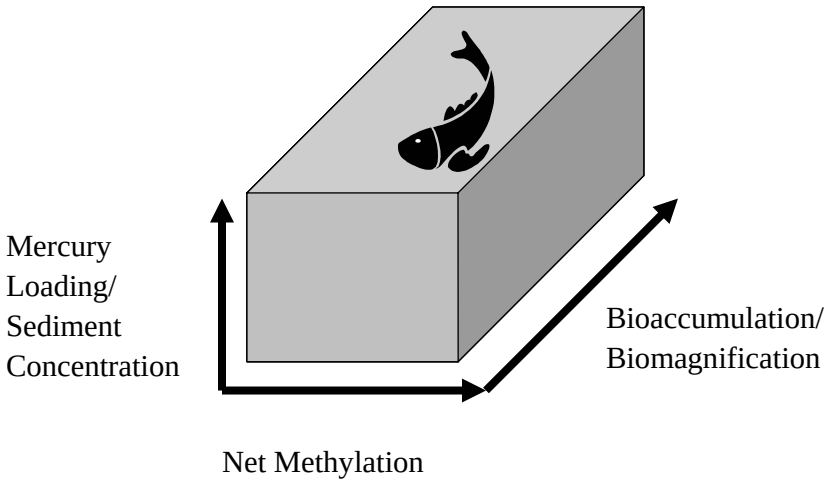
Figure 4-1 shows the fundamental conceptual model for mercury developed in the draft San Francisco Bay Mercury TMDL (San Francisco Bay Regional Water Quality Control Board 2003). This simple model is composed of a box that includes three axes or factors: mercury loading/sediment concentration, net methylation, and bioaccumulation/biomagnification. These three factors interact to determine the amount of mercury in fish or wildlife, which is represented by the volume of the box in Figure 4-1.

Limiting the magnitude of any of these three factors causes the axes of the box to shorten which causes the volume of the box (the level of mercury in fish or wildlife) to decrease. In the context of the draft San Francisco Bay mercury TMDL, efforts are focused on controlling mercury loading. For the SBSP Restoration Project, there is potential to affect all three factors through Project design and adaptive management. The following sections describe what is generally known about these three factors, and then describe in more detail what is known and what data gaps need to be resolved about mercury loads, methylation, and bioaccumulation across the continuum of habitat types in the Project area.

4.1 Factors Controlling Mercury Cycling

Mercury cycling in the aquatic environment is a complex process that is influenced by a number of environmental variables that drive the three primary processes or steps in the conceptual model. Variables of interest include mercury concentrations, mercury speciation, microbial community composition and metabolic rates, temperature, sulfate and nutrient levels, sulfide concentration, salinity, dissolved oxygen, oxidation reduction potential, organic carbon quantity and quality, turbidity, solar radiation, and vegetation type. Note that three dominant mercury species can exist in the aquatic environment: elemental gaseous mercury (Hg(0)(g)), dissolved or particulate inorganic mercury (Hg(II)), and dissolved or particulate methylmercury (MeHg). Total mercury (THg) is the sum of all mercury species.

Figure 4-1. Fundamental Conceptual Model of Mercury Cycling in San Francisco Bay



4.1.1 Mercury Loading

Mercury loads enter the Bay from a variety of sources. Mercury loading to the San Francisco Bay-Delta comes predominantly from historic gold and mercury mines in the Sacramento River Watershed (Hunerlach and others 1999; Wiener and others 2003). The Guadalupe River watershed, which includes the historic New Almaden mercury mine, is a major source of mercury loading to the South Bay (Tetra Tech Inc. 2004; Trulio and others 2004). The transport of pollutants from the Guadalupe watershed, through the project area, and into the Bay is extremely complex. In the upper watershed, pollutants move by both bed load, the transport of pollutants along the bottom of a channel via scouring of sediment, and wash load, the advective transport of suspended sediments. Knowing the relative significance of bed load versus wash load is important since it will help define how to manage flows and minimize recontamination potential. For example, bed load could readily be contained in a pond by overflowing flood flows into the pond at a location close to maximum velocity. Wash load would be harder to control, and would require control of more frequent, moderate flows.

An additional source of mercury loading, particularly in the South Bay, is urban runoff (San Francisco Bay Regional Water Quality Control Board 2004). Urban surfaces can accumulate mercury from atmospheric deposition and the mercury can be washed into receiving waters during storm events. Mercury from atmospheric deposition may be more readily methylated than mercury associated with sediments (Steding and Flegal 2002), so an important question is to what extent do urban surfaces transmit atmospheric mercury into methylating areas. Feasibility studies funded by the Clean Estuary Partnership and State Proposition-13 are currently under way to evaluate what landscape and infrastructural Best Management Practices (BMPs) are most effective at reducing the transmission of urban runoff mercury loads into the aquatic ecosystem. Although that work is outside of the Project area, it could have implications for storm water management in urban areas upstream of the SBSP Project, as well as the routing and configuration of flood control conveyances through the Project area.

Because of mercury's strong tendency to adsorb to particles on long time scales, the equilibrium concentration of mercury in Bay sediments are expected to decrease in proportion to mercury loads. This is partly the basis for proposing a sediment target for mercury in the draft San Francisco Bay Mercury TMDL, a 50 percent reduction in the Bay wide average mercury concentration in sediments would equate to a 50 percent reduction in the mercury inventory within the Bay. The draft San Francisco Bay Mercury TMDL has the long-term objective of decreasing ambient levels of THg in Bay sediments and fish from approximately 0.4 to 0.2 ppm by reducing mercury loading to the Bay. The 50 percent reduction target is expected to take over a century to be realized. The adaptive management hypothesis, implied by the linkage between sediment concentrations and beneficial uses, is that reducing mercury concentrations in sediments will reduce the transport of mercury into methylating areas such as the interstitial waters of sediments, thereby reducing the overall production of MeHg.

The TMDL adaptive management hypothesis linking mercury in fish to mercury in sediments may or may not be true. Order of magnitude increases in mercury concentrations in sediments are correlated with order of magnitude increases in MeHg production and accumulation (Marvin-DiPasquale and others 2000). Recent research suggests strong linkages between sediments and biota in marine and estuarine systems (Bloom and others 2004). However, other studies can be readily found that show no correlation between bulk mercury concentrations in sediments and MeHg in interstitial waters (Henry and others 1993). Other factors to consider include the bioavailability of mercury from different sources and site specific factors that affect net methylation rates. Some of these factors are discussed below.

4.1.2 Mercury Cycling: Methylation and Demethylation

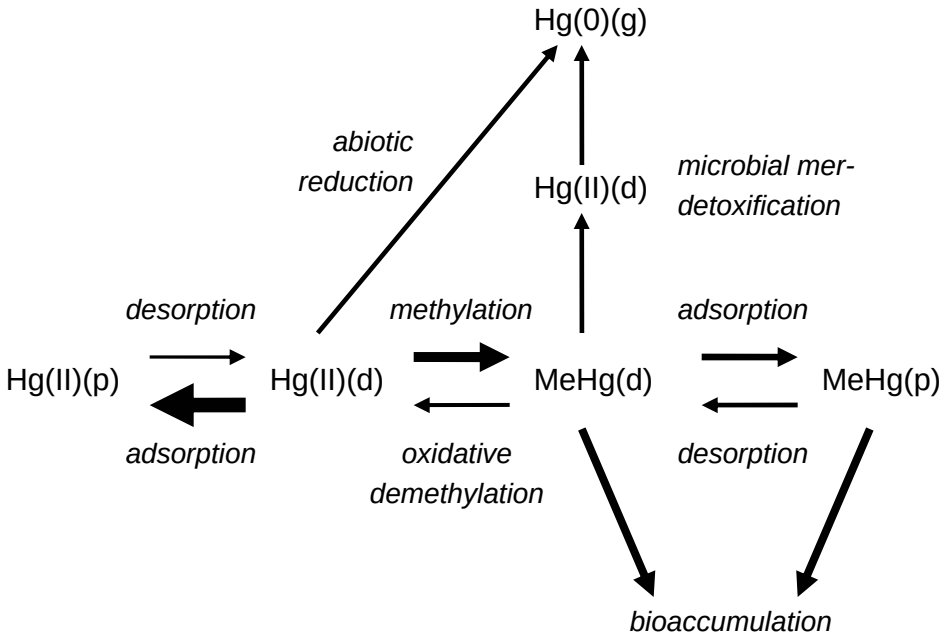
MeHg concentration in water and sediment are key indicators of a system's potential to contaminate aquatic biota. Ambient MeHg concentrations result from the interplay of several basic chemical transformations (Figure 4-2). Note that the size of the arrows in Figure 4-2 indicates the relative rate and significance of each transformation. The dominant form of mercury in the Bay and its margins is inorganic particulate mercury, Hg(II)(p). Adsorption rates of Hg(II) are much faster than desorption, so dissolved inorganic mercury, Hg(II)(d), forms a small but important pool of the THg in the Bay.

Methylation. MeHg is produced when bacteria, particularly sulfate-reducing bacteria, convert Hg(II)(d) to MeHg(d). Dissolved constituents can cross bacterial cell walls, while particulate constituents cannot. As discussed below, the chemical form of dissolved mercury, or its speciation, also plays an important role. Mercury can form complexes with inorganic ligands such as sulfide and chloride, as well as organic ligands. The complexes formed can increase or decrease the availability of Hg(II). MeHg also has an affinity for particles, though not nearly as strongly as Hg(II).

Demethylation. MeHg concentrations in water and sediment result from the balance between methylation rates and demethylation rates. In contrast to methylation, which is primarily a biological process, demethylation can occur both with and without microbial mediation. Sunlight on water can break the mercury carbon bond either directly by photo-degradation, or indirectly by production of free radicals

that attack the mercury carbon bond (Krabbenhoft and others 1998b). Photochemical degradation of MeHg yields reactive Hg(II)(d), which can readily be recycled to MeHg.

Figure 4-2. The Mercury Cycle



Microbial degradation of MeHg can proceed by one of two pathways (Marvin-DiPasquale and others 2000). In oxidative demethylation, the microbes are thought to oxidize the methyl group of MeHg via a co-metabolic pathway associated with the heterotrophic oxidation of single-carbon compounds. The resulting product is Hg(II)(d). Unlike mercury methylation, which is carried out largely by one bacterial group, the sulfate reducers, oxidative demethylation can be carried out by many types of heterotrophic bacteria, including aerobes, denitrifiers, sulfate reducers and methanogens (Marvin-DiPasquale and Oremland 1998; Oremland and others 1991; Oremland and others 1995). When bacteria are exposed to toxic levels of mercury, the "mer-operon" pathway switches on. The resulting product is elemental mercury, Hg(0)(g). Knowing which abiotic or biochemical pathway is degrading MeHg is important because the end-products are different. Hg(II)(d) can be recycled to MeHg, or adsorb to particles, and therefore remains in the system. Hg(0)(g) can be released into the atmosphere as a gas, providing a net loss of mercury from the system.

Dissolved Oxygen. Since the sulfate-reducing bacteria that methylate mercury thrive under anaerobic conditions, methylation rates are generally higher under low-oxygen conditions. More precisely, the oxidation-reduction (redox) potential, a measure of the net sum of all oxidized and reduced dissolved species, must be low enough to promote the activity of sulfate-reducing bacteria. Since dissolved oxygen

and redox potential are affected by a wide range of parameters (e.g., algal activity and decay, organic loading, sediment oxygen demand, thermal stratification, wet versus dry conditions), rates of methylation are indirectly impacted by a composite of environmental variables.

Sulfide. Sulfide can both enhance and diminish the availability of dissolved inorganic mercury, Hg(II)(d), to methylating bacteria. When sulfide is absent or in very low concentrations below 0.1 micromolar (µM), adsorption reactions may limit the concentration of Hg(II)(d), reducing the amount of available mercury (Benoit and others 1999a). At low to moderate sulfide concentrations (<10 µM), a neutrally charged dissolved sulfide-mercury complex is formed, HgS^0. The neutral character of this complex increases its the lipid solubility, allowing it to more easily pass through the lipid cell wall by passive diffusion (Benoit and others 2001a). At higher sulfide concentrations (>100 µM), the availability of mercury decreases, either by the shifting of the mercury adsorption equilibrium or by the formation of charged species which do not easily diffuse into cells (Benoit and others 2001a). The optimum window of sulfide concentration, between 5 to 20 µM, favoring mercury methylation has been referred to as the "Goldilocks Window," because there is a sulfide concentration that is "just right" for microbial uptake and methylation (Gilmour and others 1998).

The opposing processes of MeHg production stimulated by increasing sulfate reduction rates, yet inhibited by high levels of reduced sulfur end-product, can affect trends in Hg-cycling. In a recent study in the Bay-Delta, the potential for net MeHg production was greatest during late winter compared to spring and fall. During late winter gross MeHg degradation rates were low, sulfate reduction rates were low to moderate, and dissolved and solid phase reduced sulfur concentrations were low (Marvin DiPasquale and others 2003). Sulfide can also enhance the dissolution of the mineral cinnabar (HgS) in natural waters (Benoit and others 1998). This has direct implications for the restoration area since this is a dominant form of mercury entering South Bay from the New Almaden mining region.

Salinity. Like sulfide, dissolved salts may bind with mercury to form dissolved neutral mercury species which may more easily cross the non-polar cell wall ($HgCl_2^0$) (Mason and others 1996). At extreme salinities, charged species may form that do not easily diffuse across the cell wall ($HgCl_4^{2-}$). Thus, there may be a "Goldilocks Window" for salinity as well as sulfide. In addition, saline waters commonly contain sulfate (SO_4^{2-}), which provide sulfate-reducing bacteria with a needed substrate. Note that a by-product of sulfate-reducing bacterial activity is sulfide which, as noted above, also forms dissolved neutral mercury species. In these multiple ways, moderate salinity levels (i.e., fresh-brackish waters) may promote methylation, whereas higher salinities may decrease methylation rates. The effect of high salinities on mercury methylation is unknown, but needs to be determined.

Dissolved Organics. Dissolved organic matter (DOM) quantity and quality impacts mercury cycling in multiple ways. Mercury forms very strong bonds with natural DOM (Haitzer and others 2002). This may limit its availability to microbes for methylation. Conversely, heterotrophic microbial metabolism generally increases as suitable low molecular weight organic substrates increase, which can have a stimulatory effect on microbial mercury cycling. Like sulfide, certain fractions of dissolved organic carbon can facilitate the dissolution of cinnabar (HgS) (Ravishandran and others 1999).

In addition to passive diffusion, facilitated transport is another method by which bacteria can uptake Hg(II) (Golding and others 2002; Kelly and others 2003; Laporte and others 2002). In this pathway, Hg(II) is complexed by organic ligands either at the cell surface or secreted extracellularly. Facilitated transport may result from a biochemical strategy to sequester required micro-nutrient divalent cations, such as Zn^{2+} or Cu^{2+}. Since facilitated transport requires chelation by a microbially produced ligand, it will be inhibited by competitive chelation with other naturally occurring ligands (Hintelmann and others 2002). These competing ligands may be dissolved, colloidal, or on the surface of particles. Complexation reactions of naturally occurring ligands with mercury can compete with both the formation of neutrally charged inorganic species and complexation with microbial sequestering ligands. So complexation by naturally occurring organic ligands can interfere with both facilitated transport and passive diffusion pathways, reducing mercury bioavailability to methylating bacteria.

4.1.3 Bioaccumulation/Biomagnification

Bioaccumulation/Biomagnification includes both the accumulation of MeHg from the water column to phytoplankton and other biota low on the food web (bioaccumulation), and the magnification of MeHg up the food web from prey biota to predator biota (biomagnification). The bioaccumulation factors (ratio of MeHg mercury in biota to mercury in water) are generally on the order of 1,000 to 10,000 for phytoplankton and zooplankton (Hudson and others 1994). In contrast, biomagnification factors (ratio of mercury in predator biota to mercury in prey biota) are generally on the order of 2 to 5 at successively higher trophic levels. Therefore, entry into the food web is a critical step in the biomagnification process. Shifts in the abundance, diversity, or feeding habits of phytoplankton and zooplankton communities can have a profound influence on mercury concentrations at the top of the food web (Sanders and Gilmour 1992; Watras and others 1998).

It is important to discern food web and metabolic effects that drive variation in the mercury concentration of indicator fish from perturbations in mercury loads or methylation. For example, mercury naturally increases in many fish with increasing age, so fish assessments often plot mercury concentration in fish as a function of length or age. Similarly, to understand how spatial or temporal shifts in the mercury concentration of fish relate to food web changes, a marker for trophic position is needed. Light nitrogen isotopes have recently proved useful in relating mercury concentrations in fish to trophic position (Atwell and others 1998).

4.2 SBSP Conceptual Model of Mercury Cycling

The SBSP Conceptual Model for mercury cycling is shown in Figure 4-3. The model is organized around the continuum of habitat types that will form the mosaic of the Project area including tidal flats (upper left schematic in Figure 4-3), tidal marsh (upper right schematic), and managed ponds (lower right schematic) Within each habitat type there are a variety of conditions (e.g., tidal flats and tidal marsh at high and low tide; low and high salinity ponds). Using the general principles of mercury cycling and accumulation described above, the effect of the different physical, hydrological, and ecological properties of each

habitat on mercury methylation and bioaccumulation/biomagnification are discussed. In Figure 4-3, particulate mercury species are designated with a (p), dissolved species with a (d), and gaseous species with a (g). Black, blue, and orange arrows are used to designate transport, transformation and bioaccumulation/biomagnification. The thickness of the arrow is indicative of the level of significance of the process.

Tidal Flats. The schematic in Figure 4-3 shows mercury cycling at high (upper figure) and low (lower figure) tide in tidal flats. At high tide, tidal, and wind mixing exchanges suspended particulate matter from deep water areas with suspended particles from the shallows, and the cycle of deposition and resuspension equilibrates the mercury concentration in active layer sediments (Hg(II)(p)) with that of suspended particles. Mercury in the active sediment layer equilibrates with dissolved mercury (Hg(II)(d)) in the interstitial waters of sediments (pore waters). Methylating bacteria acquire dissolved mercury, not particulate mercury, so sediment pore water mercury concentration is an important variable affecting methylation rates (Krabbenhoft and others 1998a).

Dissolved MeHg (MeHg(d)) in sediment pore waters can be either adsorbed onto particle surfaces (MeHg(p)) where it can be remobilized through resuspension, or it can be released as MeHg(d) to overlying waters through wind and tidal mixing, bioturbation, or diffusion. Demethylation of mercury can produce either gaseous elemental mercury (Hg(0)(g)), which can escape to overlying waters and the atmosphere via evasion, or Hg(II)(d) which can be recycled to either MeHg or Hg(II)(p).

Within the active sediment layer, benthic invertebrates can accumulate mercury directly via ingestion of particles and detritus, or indirectly, by consuming algae that have accumulated dissolved MeHg from pore waters. In overlying waters, bioaccumulation is primarily initiated by phytoplankton acquiring dissolved MeHg from the water column, although filter feeders may also acquire suspended particulate MeHg.

The primary difference between tidal flats at high and low tide is that at low tide, replacement of overlying water with air facilitates oxygenation penetration into the sediments, which moves the depth of the redox discontinuity, the horizontal interface between aerobic and anaerobic conditions, vertically downward into the sediment column. This may also move the zone of maximum net methylation downward. This is shown schematically in the lower right side of the tidal flat schematics. Wetting and drying cycles experienced by sediments in a tidal flat environment may stimulate biological activity and result in relatively high levels of methylmercury production during some times of the year. In addition, low tide represents the period when tidal flat foraging birds are acquiring their mercury from benthic invertebrates.

Tidal Marshes. The physical layout, hydrology, and ecosystems of tidal marsh (upper right schematic of Figure 4-3) are more complex than tidal flats. As with tidal flats, maximum mercury methylation rates occur just below the sediment redox potential discontinuity, with pore water Hg(II) supplying the raw material for bacteria to produce MeHg. As in tidal flats, MeHg(d) in pore water can be demethylated to either Hg(0)(g) or Hg(II)(d), with different escape pathways.

It has been observed in the sloughs of the northern reach of San Francisco Bay that MeHg concentrations in slough sediments increase moving from the wider, deeper, tertiary slough channels to the narrower, secondary and primary slough channels (Schwarzbach 2000). This likely reflects the importance of sediments as methylation zones and the proportionally higher sediment-water contact area in secondary and primary sloughs. Tidal mixing moves MeHg produced in primary, secondary, and tertiary sloughs to the open Bay.

In the sediments of tidal marshes, plants and algae can acquire dissolved MeHg, which is biomagnified in consumers such as benthic invertebrates and their consumers such as small fish and shore birds. The presence of small fish in tidal marshes also provides a link to predatory birds such as terns. Tidal marshes have both benthic and pelagic food webs. Phytoplankton can acquire dissolved MeHg released from bottom sediments, providing a MeHg source to zooplankton and higher consumers. Tidal marsh food webs can also involve terrestrial animals, which can be exposed to mercury through consumption of plants.

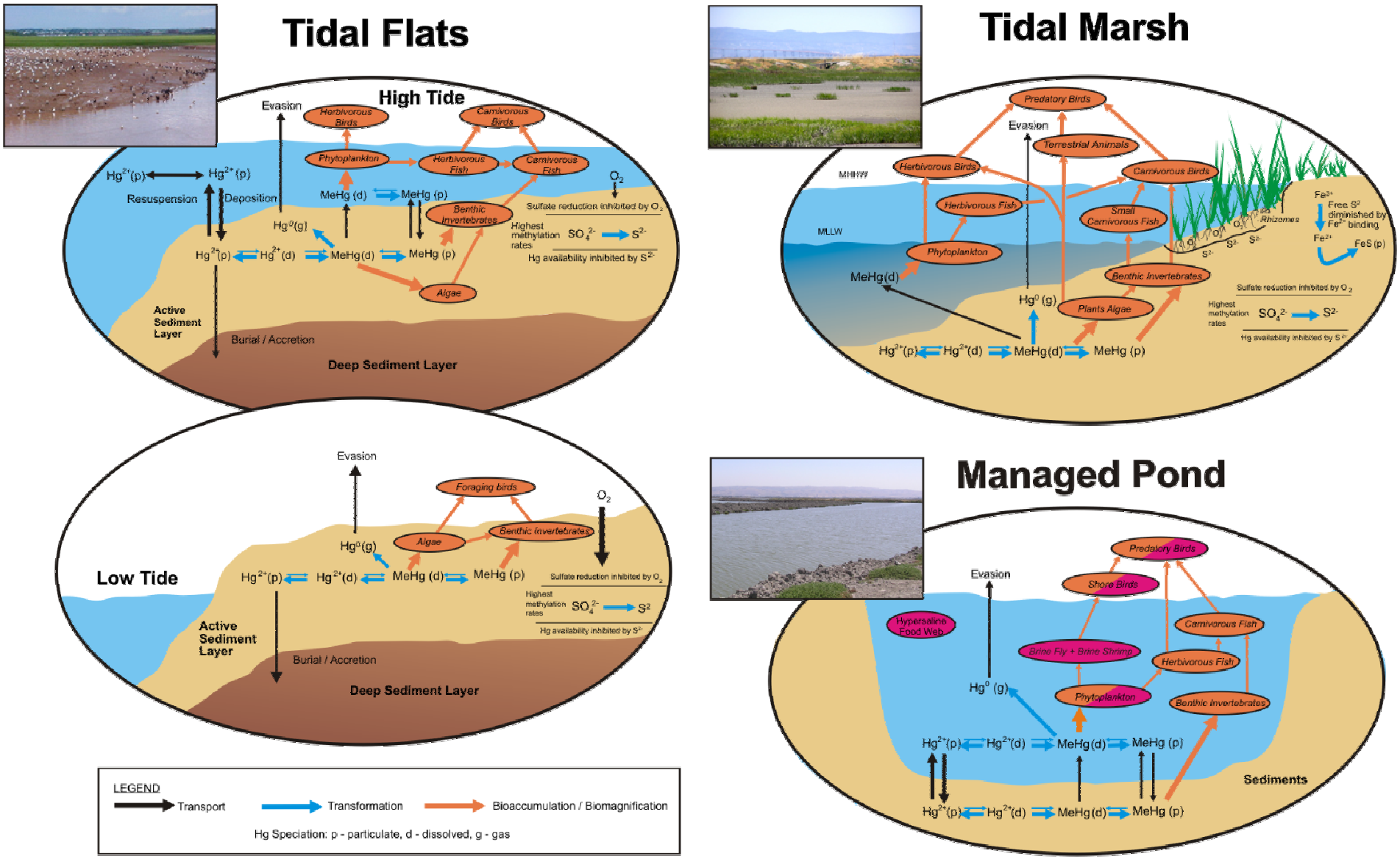
Another feature that sets apart tidal marsh habitat is the presence of a well developed rhizosphere (root zone). Plant species such as *Scirpus maritimus* and *Spartina foliosa* transport oxygen to their roots for respiration and to prevent sulfide toxicity, moving the redox potential discontinuity, the horizontal interface between aerobic and anaerobic conditions, downward into the sediment column. The microenvironments established by rhizomes can have unexpected effects on mercury methylation rates. For example, in a recent study, rapid iron and sulfur cycling near the oxic/anoxic interface may have increased net methylation rates by shifting sulfide concentrations from levels typically inhibitory for Hg-methylation (> 500 uM) to levels that are more stimulatory to Hg-methylation (< 1 uM) (Marvin-DiPasquale unpublished, 2004). Plants can also take up mercury into their biomass and temporarily store it in labile (leaves) or recalcitrant (stem) biomass (Windham and others 2001), which can be recycled at various rates. *Spartina* can also emit Hg(0)(g) from their leaves sediment and to the atmosphere.

Managed Ponds. In contrast to tidal flats and marshes, managed ponds move water relatively slowly between cells via discreet human-controlled inlets and outlets. The original purpose of the ponds was to make salt, and because hypersaline ponds do provide wildlife habitat, managed salt ponds will remain an important feature in the mosaic of habitat types found in the SBSP.

Because the hydraulic residence time of ponds is longer than tidal wetlands or flats, mercury cycling in the water column may be significant when compared to benthic processes. In both sediment and water mercury cycling, salinity is an important variable affecting methylation rates and uptake rates into the food web. At extremely high salinities, Hg(II) speciation shifts from neutrally charged and more bioavailable species such as HgClOH or $HgCl_2$ to less bioavailable charged species such as $HgCl_4^{2+}$. Unusual microbial communities may also exist in hypersaline environments that can either directly or indirectly affect mercury cycling and net methylation rates, although specific information on mercury transformation processes in hypersaline systems is limited.

Food webs in managed ponds are also different from tidal flats and marshes, and will vary according to the salinity. Both the moderate salinity and the hypersaline ponds will have basic food webs that include plankton, benthic invertebrates, and shore birds. Moderate salinity ponds will include small fish and larger fish that feed predatory birds, whereas higher salinity ponds will have brine flies and brine shrimp that feed shore birds. Note that the food web for a high salinity pond is colored in crimson in Figure 4-3.

Figure 4-3. SBSP Conceptual Model for Mercury Cycling



5. MERCURY MANAGEMENT MATRIX

A Mercury Management Matrix has been developed in a preliminary effort to link mercury cycling processes with potential management strategies to minimize mercury contamination in biota. Note that the understanding of mercury cycling in the natural environment is limited at this time, and the processes and mechanisms outlined in the matrix are, in many cases, still not completely understood. In its current form, the matrix should not be interpreted as an authoritative guide for the design of ponds or marsh habitat to minimize mercury contamination into biota.

The matrix is made up of three tables which parallel the overarching processes or factors controlling mercury cycling as outlined in the draft San Francisco Bay Mercury TMDL (San Francisco Bay Regional Water Quality Control Board 2000) which include: mercury loading/sediment concentration (Table 5-1), net methylation (Table 5-2), and bioaccumulation/biomagnification (Table 5-3). Each table includes science information related to mercury cycling processes, controls and mechanisms, and adaptive management strategies for design/construction and operations. Some of the listed strategies may not ultimately prove desirable, based on the other goals of the restoration Project. However, for the purpose of this evaluation, theoretically feasible strategies were included to provide the full suite of management options for further consideration.

Mercury Loading/Sediment Concentration. Table 5-1 evaluates issues related to mercury loading. As mercury loading increases to methylating areas, methylation rates can increase. Mercury loading is a function of both sediment quality and quantity. A major source of sediment and mercury loading to the South Bay is the Guadalupe River watershed, which includes the historic New Almaden mercury mine. An additional source of mercury loading to the South Bay is atmospheric deposition and subsequent urban runoff (San Francisco Bay Regional Water Quality Control Board 2004). Urban surfaces accumulate mercury from atmospheric deposition, and this highly bioavailable mercury is washed into receiving waters during storm events (Steding and Flegal 2002). Methods and the effects of controlling the quality and quantity of sediment and runoff to the Project area need to be evaluated.

Net Methylation. Table 5-2 summarizes issues related to mercury net methylation. Because of the role of dissolved oxygen in promoting mercury methylation, simple design features to promote well-oxygenated waters is desirable. An example is designing ponds to be shallow, with a large fetch, to promote wind mixing and aeration. This passive approach is preferable to more active management alternatives, such as mechanical aeration/oxygenation. In addition, if there is an optimal sulfide concentration that promotes mercury methylation, an active management approach could be to remove sulfide by adding a sulfide scavenging compound (e.g., ferrous iron).

Because of the formation of neutrally charged mercury species that are readily acquired by methylating bacteria, fresh-brackish salinities are known to favor mercury methylation. If this is proved to be the case in the SBSP Project area, ponds and wetlands can be designed with hydraulic configurations that minimize the amount of water in the window of intermediate salinities that promote methylation, the so-

called “Goldilocks Window.” An operational approach to avoiding sensitive salinities would be to flush wetlands with fresh water, or with high-salinity pond effluent to keep them sufficiently salty.

An investigation led by the Association of Bay Area Governments and funded by State Proposition-13 has been recently been approved. The study will evaluate, among other things, whether some types of vegetation are more conducive to mercury methylation than others. A similar investigation is being conducted by the U.S. Army Corps of Engineers at Hamilton Airfield. Microbial communities and oxygen transport dynamics in the rhizome layer may differ among vegetation types, thereby affecting mercury methylation. If vegetation is a significant factor, unfavorable species may be avoided either through design (e.g., elevation) or operation (e.g., removal). A particularly important issue is the potential for non-native invasive species to promote mercury methylation. If studies were to show that these species were effective MeHg producers relative to native species, efforts already under way to control invasive species could be accelerated.

Sediment quality, especially the nature and amount of organic carbon, may affect methylation rates in sediments. In addition to developing mercury guidelines for placement of sediment, any other sediment quality factors promoting mercury methylation, such as grain size, mineralogy, and organic carbon, should be investigated.

It is important to understand the mechanism of demethylation, because the two different microbial demethylation pathways (oxidative demethylation and mer-degradation) lead to different end products. Mer-degradation could enhance mercury transport out of the Project area via gaseous evasion of elemental mercury. While it may be possible, through design and operational controls, to promote the mer-degradation microbial pathway, gaseous evasion from the Project could increase rates of atmospheric deposition of mercury outside of the Project area.

Bioaccumulation/Biomagnification. Table 5-3 summarizes some of the issues related to mercury bioaccumulation and biomagnification. Because biomagnification increases with increasing food web complexity, restoration goals to create more well-developed food webs may need to be balanced against the risk of enhanced mercury methylation. For example, moderate salinity ponds can support fish that feed predatory birds, but the resulting food webs may be more complex than high salinity ponds, potentially increasing the mercury exposure risk to upper trophic level biota.

Another, relatively simple way to manage biomagnification risk would be to match mitigating factors with sensitive food webs. For example, if moderate salinity ponds have higher biomagnification (a deeper box in Figure 4-1), then those ponds should also have the lowest possible mercury concentrations in sediments (a shorter box), and may also need passive or active aeration and oxygenation to reduce mercury methylation (a narrower box).

Invasive species can sometimes perturb food webs, and can cause significant biomagnification impacts if the invasive species is a hyperaccumulator of mercury. For example, the incursion of *Potamocorbula amurensis*, the Asian Clam, had a marked effect on one branch of the food web in the northern reach of

the Bay, because the clam is a selenium hyperaccumulator (Linville and others 2002). Managers need to vigilantly identify invasive species, and rapidly assess their potential for enhanced food web transfer.

In many lakes, mercury biomagnification at the top of the food web is exacerbated by low phytoplankton abundance in a phenomenon called bio-dilution. Zooplankton presumably need to consume a certain amount of phytoplankton to survive. When phytoplankton abundance is low, phytoplankton can accumulate more MeHg per organism. Conversely, when phytoplankton densities are high there is less mercury per cell available for transfer to primary consumers (Pickhardt and others 2002). Since much of the overall food web biomagnification occurs at the base of the food web, phytoplankton abundance may be a critical parameter. A design and operational approach to use this information would be to avoid oligotrophic waters, and promote high phytoplankton abundance, while maintaining well-oxygenated conditions.

Table 5-1. Mercury Management Matrix – Mercury Loading/Sediment Concentration

Science Information				Potential Adaptive Management Strategies	
Process	**Control**	**Mechanism**	**References**	**Pond and Wetland Design/Construction**	**Operation**
Mercury Loading Increased loading of mercury to methylating areas may increase methylation rates and subsequent incorporation of mercury into the food web	Sediment and runoff quality	MeHg production may increase with increasing concentrations of mercury in sediments; Urban runoff may be important source of bioavailable mercury as a result of atmospheric deposition	(Krabbenhoft and others 1999; Mason and Sullivan 1998; Thomas and others 2002)	• Define and implement sediment quality guidelines for reuse of fill material during restoration • Avoid development of habitats with high methylation potential in areas with existing mercury-rich sediments • Site pond breaches to minimize potential for re-mobilization of mercury laden sediments as a result of scouring • Provide water control features that can divert polluted flows away from Project area to avoid re-contamination of restoration area	• Divert "first flush" flow events or other critical flow events away from sensitive areas in the restoration area • Control flows out of restoration area to minimize re-mobilization of mercury laden sediments • Implement upstream BMPs to avoid and minimize inputs of bioavailable mercury to the Project area
	Sediment loading	MeHg production may increase with increased loading of mercury-rich sediments; Guadalupe river is major source of mercury-enriched sediments from the New Almaden mines	(Krabbenhoft and others 1999; Mason and Sullivan 1998; Thomas and others 2002)	• Avoid deposition of mercury-rich sediments in habitats with high methylation potential • Provide water control features that can divert polluted flows away from Project area to avoid re-contamination of restoration area • Provide pretreatment areas where mercury in polluted inflows can be contained and/or managed	• Divert "first flush" flow events or other critical flow events away from sensitive areas in the restoration area • Control flows out of restoration area to minimize re-mobilization of mercury laden sediments • Implement upstream BMPs to avoid and minimize inputs of bioavailable mercury to the Project area

Table 5-2. Mercury Management Matrix – Net Methylation

Science Information				Potential Adaptive Management Strategies	
Process	**Control**	**Mechanism**	**References**	**Pond and Wetland Design/Construction**	**Operation**
Net Methylation Methylation and demethylation are microbially mediated processes	Sulfide	a. Low to moderate sulfide concentrations may form bioavailable neutrally charged mercury species b. High sulfide concentrations may form charged mercury species that are less bioavailable	(Benoit and others 1999b; Benoit and others 2001b; Benoit and others 1999a; Mehrotra and others 2003)	• Minimize pond depth and maximize fetch to promote wind mixing and maintenance of aerobic conditions	• Add compound (e.g., iron) to sediment to bind with sulfides and/or mercury • Maximize sulfidic conditions to form charged Hg-S complexes
	Salinity	a. Brackish salinities may favor formation of neutrally charged sulfide-mercury species. b. Hypersaline conditions may form charged mercury species less easily methylated	(Barkay and others 1997; Compeau and Bartha 1984; Gilmour and others 1992; Laporte and others 1996; Laporte and others 1997)	• Incorporate ability to flush ponds at a wide range of flow rates • Provide initial mixing zone so that the level of dilution of incoming salt water can be controlled • Maximize tidal habitat in the saline reaches of the project area.	• Provide option for wetland areas to be fed by pond effluent • Keep managed ponds at high salinity to promote the formation of charged Hg-Cl complexes
	Vegetation type	Different types of vegetation (e.g., *Salicornia* vs. *Spartina*) support different microbial communities in the rhizome layer, potentially with differing methylation capacities	(Bagwell and others 2001; Windham and others 2001)	• Provide design features that promote plant species with low mercury methylation rates	• Remove unfavorable species that promote mercury methylation in the root zone
	Sediment quality	High organic matter and/or sediment oxygen demand may yield higher rates of mercury methylation	(Krabbenhoft and others 1999)	• Provide water control features that can divert flows heavy in organic matter away from Project area • Define and implement sediment quality guidelines for reuse of fill material during restoration • See comments above in "Sulfide" sub-section to maintain and enhance aerobic conditions	• Divert flows heavy in organic matter away from Project area
	Redox potential	The optimum window for mercury methylation is roughly zero to -100 mV	(McFarland and Lee 2002)	• See comments above in "Sulfide" section regarding design features to promote aerobic conditions • Provide water control structures to control cycles of wetting and drying to minimize conditions that promote methylation	• See comments above in "Sulfide" section regarding operations to promote aerobic conditions or maximize sulfidic conditions • Operate water control structures to control cycles of wetting and drying to minimize conditions that promote methylation
	Microbial de-methylation pathway	Two different pathways lead to different end products – Hg(II) or Hg(0); Production of Hg(0) may result in increased mercury removal from Project area by gaseous evasion	(Marvin-DiPasquale and others 2000)	• Determine whether design features exist that could promote "mer-operon" pathway, favoring production of gaseous mercury, Hg(0)	• Measure Hg(0) fluxes from Project area to refine mass balance • Optimize habitat to promote mer-detoxification and Hg(0) evasion

Table 5-3. Mercury Management Matrix – Bioaccumulation/Biomagnification

Science Information				Potential Adaptive Management Strategies	
Process	**Control**	**Mechanism**	**References**	**Pond and Wetland Design/Construction**	**Pond and Wetland Operations**
Bioaccumulation/ Biomagnification MeHg bioaccumulates from the water column to phytoplankton, then biomagnifies up successive trophic levels resulting in potentially high levels in biota at the top of the food web	Food web complexity	Mercury in upper trophic level biota tends to increase with the number of links in the food web	(Atwell and others 1998)	• Promote habitats that support short food webs (e.g., moderate-salinity managed ponds of 60 to 180 ppt) • Promote habitats that exclude hyperaccumulating biota at base of food • Promote habitats that support non-accumulation biota at the base of the food web	• Manage food web complexity through the management of salinity in ponds and wetlands – food web complexity tends to decrease as salinity increase • Identify and selectively harvest biota that are hyperaccumulators or keystone species with respect to bioaccumulation
	Biodilution	High phytoplankton levels may decrease bio-accumulation since MeHg is diluted into large pool of phytoplankton biomass	(Gill and others 2002; Kirkwood and others 1999)	• Provide operational flexibility through the use of appropriate water control structures to manage flow rates and schemes through pond and wetland systems to affect primary productivity	• Add limiting nutrients to ponds to enhance productivity while avoiding promotion of anaerobic conditions • Add reclaimed water to ponds
	Vegetation type	a. Different types of vegetation may have different MeHg and Hg(II) accumulation rates b. Various plants can take up mercury into either their labile (leaves) or refractory (stems) biomass c. Some plant species release Hg(0) from their leaves	(Vasconcelos and Leal 2001)	• Provide design features that promote the exclusion of hyperaccumulating vegetation • Provide design features that promote hyperaccumulating vegetation in targeted locations for intensive management/removal	• Identify and harvest vegetation that are hyperaccumulators • Optimize habitat for plants that take up mercury into stems as opposed to leaves • Optimize habitat for plants that facilitate the transpiration of Hg(0) to the atmosphere

6. MERCURY MANAGEMENT RECOMMENDATIONS

Given the scope and high profile nature of the overall SBSP restoration effort, unraveling the habitat-specific details of key mercury transformations and bioaccumulation pathways is extremely challenging and will require a significant long-term financial commitment to fully resolve. The Project provides an important opportunity to explore design and management alternatives that will mitigate the transfer of mercury into the food web based in sound science. The result of these efforts will not only benefit restoration efforts in the South Bay, but future wetland restoration activities throughout the Bay-Delta. Based on a synthesis of the data and information presented in this memorandum, the following recommendations are provided regarding the management of mercury in the context of the SBSP Restoration Project.

1. Do not consider mercury a "fatal flaw" of the restoration Project.
The Project provides an unparalleled opportunity to restore vast areas of habitat for a wide variety of biota including native special status species. An examination of existing mercury data suggest that, while mercury levels are elevated in sediment and biota in some ponds, mercury levels in the Project area are, in general, similar to levels observed at other mercury-impacted sites in the Bay-Delta. While the science of mercury cycling is under development, results of a robust monitoring program coupled with results of ongoing, proposed and future mercury studies, can be used to inform the adaptive management plan for the Project. Proper adaptive management of the restoration Project will mitigate mercury impacts and minimize the potential for accumulation of mercury in biota over the long term.

2. Use results of ISP and other monitoring efforts to inform mercury data gaps and adaptive management process.
The ISP and other studies present opportunities to learn more about mercury cycling through monitoring and adaptive management. Monitoring should include THg and MeHg content in sediments, water and biota, methylation rates in sediment and the water column, and mercury bioavailability in sediments. Adequate monitoring should be performed to capture spatial and temporal patterns. Recommendations for future monitoring to address specific data gaps include:

- *Perform baseline and long-term monitoring of mercury in biota.* One of the objectives for the Project is to "protect or improve existing levels of water and sediment quality in the South Bay, and take into account ecological risks caused by restoration." In order to assess the potential impacts of the Project, baseline monitoring of mercury in biosentinel species within the Project area should be performed, and long-term monitoring should continue. Biosentinel organisms include small prey fish and microinvertebrates that are sensitive and consistent indicators of MeHg (Wiener and others 2003). Bird eggs are also ideal bioindicators of mercury exposure since bird foraging varies spatially and temporarily throughout their nesting grounds (Schwarzbach and Adelsbach 2003a). In the case of the SBSP, birds with foraging ranges that are more spatially limited (e.g., Clapper Rails, Black-necked Stilts), as opposed to those that forage more widely (e.g., Snowy Egrets, Forster's Terns), would be more useful in determining the location and extent of mercury related problems in the Project area. For mammals and birds, fur and feathers are also easily obtainable and non-invasive matrices for monitoring mercury levels (Wolfe and others In Press). Biota mercury data will be critical to evaluate the relative changes in

mercury cycling resulting from the restoration Project, and to inform and guide adaptive management of the Project. As noted elsewhere in this document, relative levels of mercury in fish may be an ideal indicator of which existing ponds pose a risk of mercury accumulation in biota, and these ponds could be managed and/or modified appropriately. Mercury levels in fish can also be used in the future to access the mercury risk posed by restored marsh and pond components of the Project area.

- *Monitor mercury methylation across salinity gradient in managed ponds.* Salinity is a primary factor that managers can manipulate within managed ponds, thus it is critical to understand how salinity levels impact mercury methylation. As noted in this memorandum, salinity levels may affect mercury methylation in a number of ways. To avoid confounding factors such as mercury sediment content and spatial differences in habitat type, the ideal monitoring setup would include the monitoring of the same pond with altering levels of salinity.
- *Monitor mercury methylation across restoration habitat types.* The restoration of the SBSP will result in a mixture of habitat types including tidal marsh, tidal flats and managed ponds. The relative amount and location of each type of habitat is a design variable under the control of restoration designers and managers. A clearer understanding of the relative potential of various habitats to act as sources of MeHg would help to better inform the restoration design. To achieve this, it is strongly recommended that initial studies be implemented that focus on clearly defining the dominant mercury transformations in the various sub-habitats at a number of existing wetland reference sites in the South Bay.
- *Monitor "bioavailability" of inorganic mercury in sediment.* Some researchers have proposed that a simple analytical method in which mercury is extracted from a sediment sample may be a good measurement of mercury "bioavailability" for methylation (Marvin-DiPasquale unpublished, 2004). There are a number of simple analyses that could be performed on sediment samples including a water extraction, a weak acid extraction, and/or a tin chloride reduction/extraction. The third method may be preferable since it would yield a measure of the chemically reducible inorganic mercury dissolved in porewater as well as that weakly bound to the sediments. Dissolved mercury that is strongly complexed with organic carbon or tightly bound to the sediments, and therefore unavailable for methylation, would not be extracted. The USGS is reportedly starting to use this approach as a way to assess mercury bioavailability at among sites in a number of existing study sites. This analytical method could provide additional valuable information for understanding the spatial and temporal variation of MeHg production among ponds of different salinities or of different habitat types (e.g., tidal marsh versus tidal flats).
- *Monitor upcoming near-term changes in pond operations.* Operational changes are being implemented, and more are planned, through the Interim Stewardship Plan (ISP). These include the pending breaching of the "Island Ponds" (Ponds A19, A20 and A21) and the use of Pond A8 for flood control. In addition, groups of ponds are being hydraulically connected to the South Bay as part of the ISP. These operational changes should be coupled with monitoring of appropriate parameters related to mercury cycling. This is an ideal, near-term opportunity to test some of the key uncertainties related to mercury discussed in the memorandum. The suitable mechanism for monitoring during implementation of the ISP is the Plan's Waste Discharge Requirements and associated monitoring provisions.

- *Monitor existing and recently created marsh habitat to predict future outcomes.* A handful of current or recently formed marsh areas exist near the Project area. In addition, restored marsh areas exist within the greater San Francisco Bay (e.g. Cooley Landing, 115-acre salt pond breached in 2000; Warms Springs, 220-acre formerly diked, subsided marsh breached in 1986). These marsh areas can act as an analog to the marsh habitat that will be restored as part of the project. Monitoring of appropriate parameters related to mercury cycling should be performed in these areas in order to predict the fate of mercury in future restored marsh.

3. Develop and prioritize testable hypotheses concerning mercury cycling.

This memorandum identifies a number of areas where uncertainties exist regarding mercury cycling in the Project area. To allow for effective adaptive management decisions as the Project is implemented, these uncertainties need to be further evaluated and prioritized. As part of the Project's Science Plan, key testable hypotheses regarding mercury cycling should be developed and ranked. This ranking of testable hypotheses will act as a framework for the implementation of monitoring and experimental studies that will successfully inform adaptive management decisions.

4. Pending further study, consider potential design components to mitigate mercury impacts.

As noted in this memorandum, the level of understanding of mercury cycling in aquatic systems is still under development, and is not currently at the level which supports the recommendation of specific design components. However, a few obvious potential strategies that should be considered pending further study are outlined below.

- *Cover sediment in ponds with highest methylation potential.* Further study of mercury methylation dynamics may provide an indication as to which ponds have the highest potential to impact biota. A potentially simple factor by which ponds could be evaluated for capping is the existing level of mercury in biota, particularly fish. Ponds with high levels of mercury in fish relative to other ponds should be considered for capping. Restoration designers should consider the feasibility of capping these sediments with clean sediments either via natural sedimentation processes or using dredged material, and keeping the areas as shallow managed ponds, thereby isolating contaminated sediments from the aquatic ecosystem. These areas could also be restored tidal marsh if contaminated sediments are deep enough to avoid scouring and remobilization within the wetland.
- *Maintain substantial areas of managed ponds in the SBSP Project area.* Mercury cycling data suggest that rates of MeHg production can be an order of magnitude higher in marsh versus open water (Marvin DiPasquale and others 2003), though this has not been confirmed. In addition, factors that affect MeHg production including mercury loading, oxygen levels, and salinity are more easily managed within a pond when compared to a tidal marsh since a pond has a discrete inlet and outlet. Ponds could also be utilized to capture and immobilize sediments in stormwater flows in the Alviso Slough/Guadalupe River, thereby limiting mercury loading to the greater South Bay. Maintaining managed ponds in the project area must be balanced with the need to provide restored tidal marsh habitat for critical species, including special status species such as the California Clapper Rail and the Salt Marsh Harvest Mouse.

5. Coordinate with comparable pilot projects in the San Francisco Bay-Delta.
Mercury studies and monitoring are being conducted in other habitats around the San Francisco Bay-Delta by a number of researchers and agencies (e.g., Association of Bay Area Government's State Proposition-13 proposal, San Francisco Estuary Institute's CALFED project, the United States Army Corps of Engineers' study at Hamilton Airfield). As discussed at the Regional Monitoring Program's annual meeting, there is a need for regional coordination of these various projects. The Project Team should track the results of these monitoring efforts to inform the adaptive management of the Project, as well as dialogue with investigators during the development of study goals to help implement studies that will address data gaps specific to the Project.

6. Refine SBSP Model Conceptual for mercury and the Mercury Management Matrix.
The Conceptual Model and Mercury Management Matrix proposed in this memorandum are based on available scientific information on mercury cycling. Studies and monitoring efforts performed after the development of the Conceptual Model and Matrix should be used to revise and refine the Model and Matrix, and modify and/or strengthen applicable recommendations. The Conceptual Model and Matrix should ultimately be used to inform the development of more predictive numeric fate and transport mercury models, linked to sediment models, for both tidal wetland and managed pond environments in the South Bay.

7. Refine Sediment Quality Guidelines for the beneficial re-use of sediments, focusing on mercury.
The LTMS draft guidelines, discussed in Section 3, were not developed with a sophisticated conceptual model for mercury bioaccumulation at hand, and these guidelines may need to be revisited. It has been recently proposed that water quality planners consider the feasibility of surgical, or even widespread, dredging to remove pollutants from the sediment reservoir of the Bay (Mcgrath 2004). To reuse these sediments as base material for restored wetlands, it is important to identify sediment quality guidelines that will avoid impacts to beneficial uses of the wetlands. Because of its tendency to bioaccumulates in biota, mercury is an important component of any sediment reuse/quality guidelines. The Project Team should track the development of any sediment guidelines that could impact the Project, and provide input to regulators as appropriate to update the LTMS guidelines.

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