Restoring Areas of Suisun Marsh to Tidal Wetlands: Potential Effects on Mercury Geochemical Interactions and Implications for the Suisun Marsh TMDL

January 25, 2013

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Executive Summary

This work provides a general overview of methylmercury (MeHg) production and flux from tidal wetland environments from the Sacramento San Joaquin Delta and other regions. This information can be applied to estimate potential loads that may result from conversion of certain regions of Suisun Marsh from managed wetlands to fully tidal wetlands. Existing managed wetlands in Suisun Marsh, the major fraction of the land area, may be considered to be muted tidal wetlands, with some water exchange with surrounding sloughs, as compared to fully tidal wetlands with a daily inflow and outflow of water over the tidal cycle. Tidal wetlands can be both sources and sinks of total (THg) and MeHg as they tend to retain particulates and associated mercury (Hg) but export dissolved materials. Season, elevation, tidal volume, vegetation, organic carbon, and Hg availability all play key roles in affecting THg and MeHg dynamics in tidal wetlands and should be taken into account when collecting data.

Current data from muted tidal managed wetlands and tidal wetlands in the Delta suggest similar MeHg loading rates across sites on the order of 10^{-9} and 10^{-8} g m^{-2} d^{-1} range. Our review suggests that for Suisun Marsh, MeHg exports from tidal wetlands may range between $3x10^{-9}$ to $1x10^{-8}$ m^{-2} d^{-1} and in higher intertidal regions with high organic matter soils, export values may reflect those estimated at Blacklock marsh, a managed wetland in Suisun Marsh that was converted to tidal wetland, ranging well above 10^{-8} m^{-2} d^{-1}. As only a few studies exist that provide MeHg flux data from tidal wetlands and an even more limited amount of research regarding the effects of tidal conversion of managed wetlands, flux estimates are rudimentary and should be viewed with caution. Clearly, managed wetlands and tidal wetlands are very different with regard to marsh footprint, hydrology (i.e., magnitude, inundation frequency) and vegetation. All these factors greatly affect wetland biogeochemistry and thus MeHg production. However, not enough is known to fully understand the effects on MeHg in Suisun Marsh if large wetland tracts currently managed for duck hunting and other anthropogenic activities are restored to tidal conditions.

Besides the rate estimates, our assessment of the biogeochemical drivers suggests the tidal restoration results in many factors that could reduce MeHg production (and the likely associated low DO events currently associated with managed wetlands in Suisun Marsh). Yet the available data suggest overall MeHg supply to the sloughs could potentially increase because of greater hydrologic exchange. However, other results from restoration should help. First, conversion to tidal wetlands will minimize anthropogenic management and resulting disturbance which have been associated with low DO events in Suisun and episodic MeHg load export events elsewhere. Second, hydrology will be greatly changed. Tidal wetlands act as tidal pumps. Converting from a managed to full tidal marsh will greatly increase tidal prism and flows in sloughs (likely on an order of magnitude) and this effect will profoundly affect MeHg concentrations, regardless of the MeHg loads from the tidal wetlands. We hypothesize that these two results will decrease spatial and temporal MeHg hotspots in Suisun Marsh. While this hypothesis suggests that conversion of managed wetlands to tidal marsh may be beneficial from the standpoint of MeHg, ideally, further research, both with modeling efforts and through data collection at higher density across Suisun Marsh, should be considered before embarking on a regional scale tidal restoration.

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Abbreviations and Acronyms

DOM	Dissolved Organic Matter (OM)
DOC	Dissolved Organic Carbon (DOC)
Fe	Iron
Fe^{3+}	Ferric Iron
FeRB	Iron Reducing Bacteria
H_2SO_4	Hydrogen sulfate
Hg	Mercury
Hg^0	Elemental mercury
Hg^{2+}	Inorganic mercury
MeHg	Methyl Mercury
OM	Organic Matter
POM	Particulate Organic Matter (OM)
S^{2-}	Sulfide
SO_4^{2-}	Sulfate
SRB	Sulfate Reducing Bacteria
µg/L	micrograms per liter, parts per billion (ppb)
THg	Total mercury
TMDL	Total Maximum Daily Load

1. INTRODUCTION

Suisun Marsh is the largest contiguous brackish marsh on the west coast of North America encompassing 52,000 acres of diked managed wetlands, 8,000 acres of tidal wetlands, 16,000 acres of upland grasslands, and 26,000 acres of open water bays and sloughs (Figure 1; DWR, 2001). Although Suisun Marsh originally contained 68,000 acres of tidal wetland, 90% was drained or reclaimed for agriculture or converted to managed wetlands for duck hunting and limited livestock grazing from the mid-1800's to early-1900's. Areas managed for waterfowl are seasonally flooded and mosquito control of these areas requires controlled flooding regimes in the fall, winter and spring (DWR, 2001). Existing tidal marsh areas make up only about 8% of the total Suisun Marsh area (Figure 1, Siegel et al., 2011).

The restoration of tidal marsh throughout Suisun Marsh is becoming a growing priority. Several regional ecosystem planning efforts, including the Delta Plan (in preparation), the Bay Delta Conservation Plan (in preparation), Ecosystem Restoration Program Stage 2 Conservation Strategy (DFG 2010), Suisun Marsh Plan (USBR and other agencies, 2011), Tidal Marsh Recovery Plan (USFWS 2010), Delta Vision (Delta Vision Blue Ribbon Task Force 2008) and the Baylands Habitat Goals Project (Goals Project 1999), call for extensive additional restoration throughout the San Francisco Bay Estuary system, suggesting as much as an additional 26,000–40,000 ha (65,000-100,000 acres) of tidal restoration should take place in the Delta and Suisun Marsh alone.

Methylmercury (MeHg) production and subsequent bioaccumulation and biomagnification throughout the food web is a primary environmental consideration with regard to tidal wetland restoration in the estuary. Wetlands provide opportunities for MeHg production because wet/dry cycling, potential to elevate water temperatures, sources of labile carbon, and low redox conditions that enable sulfate and iron reducing bacteria in anaerobic environments (St. Louis et al. 1994; Hurley et al. 1995; Rudd 1995; Gilmour et al. 1992; Gilmour et al. 1996; Yu et al. 2011). The estuary has mercury (Hg) contamination of fish, sediment, and water, as compared to other North American estuaries, due to the history of Hg mining in the Coast Range mountains and the use of Hg for gold extraction in the Sierra Nevada mountains in the 19th century (Churchill 2000; Heim et al. 2003; Wiener et al. 2003; Heim et al. 2007; Davis et al. 2008). Many have found that MeHg is produced from inorganic Hg primarily by activities of sulfate- and iron-reducing bacteria in anaerobic environments (Gilmour, Henry et al. 1992; Gilmour, Riedel et al. 1996; Yu, Flanders et al. 2011).

Conditions in Suisun Marsh have resulted in efforts to develop the Suisun Marsh Total Maximum Daily Load (TMDL). Following implementation, the TMDL is expected to meet various water quality objectives pertinent criteria from the San Francisco Bay Basin Water Quality Plan, including 2.1 µg/L hourly MeHg average and site-specific fish tissue concentrations (Baginska, 2012).

However, the landscape of Suisun Marsh may change as a consequence of the restoration plans noted above. Two questions relating to mercury are relevant to the proposed TMDL: (1) If large-scale tidal wetland conversion occurs in Suisun Marsh how will MeHg production, loads and concentrations change? (2) Will restoration result in conditions more difficult to meet TMDL water quality criteria? This work addresses these questions through a review of the scientific literature on geochemical interactions affecting mercury (Hg) and methyl mercury (MeHg) cycling in tidal wetlands.

2. Wetland Mercury and Methylmercury Cycling and Production

2.1. Mercury Sources and Ambient Concentrations

Mercury sources entering a wetland system include direct and indirect atmospheric deposition, river flows and sediment, and in the case of tidal wetlands, estuarine input (Conaway et al., 2003; Das et al., 2012). In Northern California, a majority of Hg originates from riverine runoff and sediment due to legacy anthropogenic inputs from gold and mercury mining, present day municipal and industrial discharges, and wet/dry atmospheric deposition (Conaway et al., 2003; Heim et al., 2007). Current inputs to Suisun Marsh are largely controlled by the Sacramento-San Joaquin Delta (Delta), which is in turn affected by historical gold and mercury mining, and by tidal inflows, urban, agricultural, industrial, and local watershed runoff (Baginska, 2012). External Hg loading is a constraint on Suisun Marsh and management within the marsh does not alter these external loads.

Hg transport is primarily facilitated by particulate and dissolved organic matter (POM and DOM[1]) and fine grained sediment, by adsorption and by the formation of complexes (Domagalski et al., 2004; Brigham et al., 2009). Thus, high organic matter (OM) soils such as peat and solutions high in DOM tend to contain higher concentrations of total Hg (THg, the sum of inorganic Hg and MeHg) (Davis et al., 2003; Marvin-Di Pasquale et al., 2009; Grenier et al., 2010). In the Delta, Stephenson et al. (2008) found MeHg in sediments correlated positively with percent organic matter content. As wetlands contain high organic content soils and are sources of DOC, they are prime regions for MeHg production and potentially export both THg and MeHg.

In both rivers and wetlands of the Delta region, MeHg is highly correlated with total suspended sediments (Foe et al., 2008; Stephenson and Bonnema, 2008). Sediment loading has been found to be a major MeHg source to the Delta, with 54-74% of MeHg bound to particles (Foe et al., 2008) and average particulate MeHg sedimentation rates of 4.9 g/d, with higher rates typically found in winter months when flows are higher (Stephenson and Bonnema, 2008). THg and MeHg concentrations in sediment, pore water, and surface water vary in the Delta depending on the major water source and location, but both sediment and surface water THg and MeHg were generally found to be highest in marsh habitats and lowest in open water habitats, with higher concentrations found in winter months (Heim et al., 2008; Stephenson, Foe, et al., 2008).

To compare concentrations in Suisun Marsh to similar environments where mercury cycling has been studied, concentrations for several locations in the Delta and elsewhere are summarized in Tables 1, 2, and 3 for sediment, pore water, and surface water concentrations. Suisun Marsh

[1] The terms POC for particulate organic carbon and DOC for dissolved organic carbon are also used in this report and refer to the same constituents as POM and DOM, respectively. The terms OM for organic matter and OC for organic carbon are also used to refer generically to both dissolved and particulate fractions.

surface water and sediment concentrations in relation to concentrations in other study areas are shown in Figure 2. Although sediment and surface water THg and MeHg concentrations in Suisun Marsh fall within median values found in the Delta (110 and 2.5 ng g^{-1} THg and MeHg concentrations in sediment, respectively; 7-37 and 0.08-0.4 ng l^{-1} for THg and MeHg solution concentrations, respectively, Figure 2), concentrations are higher in comparison to other tidal wetland and estuarine regions.

2.2. MEHG BIOGEOCHEMICAL PRODUCTION AND CYCLING

The biogeochemical production and cycling of MeHg depends upon a large number of oftentimes interacting factors including dissolved oxygen, pH, sulfate, organic carbon, temperature and salinity (**Error! Reference source not found.**). Figure 3 and Figure 4 are conceptual diagrams describing these factors and their interactions in non-tidal and tidal wetland environments. The transformations summarized in these figures are based on the literature described below.

Dominant Hg species entering a wetland include inorganic mercury (Hg^{2+}), elemental Hg (Hg^0), and MeHg. MeHg is of primary concern to human health and wildlife as it is a neurotoxin to humans and bioaccumulates up the food web to much higher concentrations than present in the water column (Davis et al., 2003).

Predominant factors influencing sediment and dissolved phase MeHg formation are Hg-methylating microbes and Hg speciation (King et al., 2002; Marvin-Di Pasquale et al., 2009). Under reducing conditions, sulfate reducing bacteria (SRB) and iron reducing bacteria (FeRB) use SO_4^{2-} or ferric iron (Fe^{3+}) respectively as electron acceptors to mineralize OM, subsequently methylating selected species of inorganic mercury that can diffuse into the cell membrane (King et al., 2002). Thus, anoxic conditions that promote higher concentrations of SO_4^{2-}, and in some cases Fe^{3+}, could have relatively higher MeHg concentrations and production (Ulrich et al., 2001; Hollweg et al., 2009). Best et al. (2009) estimated highest methylation rates to occur at redox values between -100 and +100 mV. This environment can be found in the oxic/anoxic interface of wetland sediments, typically existing within the top 10 cm and has been shown to exhibit relatively higher MeHg concentrations (Choe et al., 2004; Hall et al., 2008; Rothenberg et al., 2008a). Decreasing MeHg concentrations with sediment depth have been noted and attributed to more reducing conditions, reduced concentrations of SO_4^{2-}/Fe^{3+}, complexation of Hg^{2+} into biologically inaccessible compounds, and increased demethylation (Ulrich et al., 2001; Best et al., 2005; Holmes and Lean, 2006; Mitchell and Gilmour, 2008; Yang et al., 2010).

The bioavailability of Hg^{2+}, the species primarily associated with MeHg formation, governs its susceptibility to methylation. Factors affecting Hg^{2+} bioavailability include presence of SO_4^{2-}, Fe, appropriate redox conditions, DOM, temperature, pH, Cl, and turbidity. Although SO_4 can increase Hg methylation by promoting microbial activity, high concentrations of sulfide (S^{2-}) resulting from SO_4^{2-} reduction can complex with Hg^{2+} to form aqueous or solid phase mercury sulfides (e.g., metacinnabar or cinnabar (HgS)). These complexes are too large to diffuse through cell membranes (King et al., 2002; Drott et al., 2007), limiting their potential methylation. In addition, under reducing conditions and high S^{2-} concentrations, Fe^{2+} can form pyrite (FeS), which can also scavenge Hg^{2+} (Ulrich and Sedlak, 2010). Ideal methylating environments have SO_4^{2-} concentrations ranging between 200-500 µmol l^{-1} (19-48 mg/l) and S^{2-} concentrations <10µmol l^{-1} (0.6 mg/l)(Rothenberg et al., 2008b; Hollweg et al., 2009). Within these concentration ranges, Hg^{2+} can form small, dissolved, uncharged complexes with S^{2-}, which are available for microbial methylation (Benoit et al., 2001; Mehrotra and Sedlak, 2005). Brackish to saline waters high in SO_4^{2-} therefore, have the potential to decrease MeHg formation due to HgS and FeS accumulation.

Organic matter, particularly DOM, is an energy source for microbes and can enhance SRB and FeRB activity as well as increase the biochemical oxygen demand (BOD), decreasing redox conditions and promoting MeHg production. OM quality is important, however, and the presence of OM alone does not necessarily result in increased MeHg production. Grenier et al. (2010) found higher methylation rates in wetlands receiving labile algal OM compared to wetlands receiving recalcitrant, terrestrially-derived OM. Larger molecular weight, aromatic DOM with high UV absorption have also been strongly correlated with both THg and MeHg (Schuster et al., 2008; Henneberry et al., 2011). Reduced sulfur sites on these DOM species form strong bonds with Hg^{2+}, maintaining its solubility even under reducing conditions. In addition, DOM has been found to dissolve cinnabar, subsequently releasing Hg^{2+} (Ravichandran et al., 1998). DOM can also inhibit Hg^{2+} bioavailability by forming complexes that are too large to diffuse through the cell membrane, and inducing photochemical reduction of Hg^{2+} into the relatively unreactive and insoluble Hg^0 (Ravichandran, 2004).

Both temperature and pH can have varying effects on Hg methylation. Higher temperatures tend to enhance microbial activity and thus are thought to promote Hg methylation (Matilainen, 1995; Canário et al., 2007; Mitchell and Gilmour, 2008). Studies have also found MeHg production to increase in mildly acidic environments (>pH 5), but decrease in anoxic soils with a pH <5 due to increased demethylation (Ulrich et al., 2001 and references therein). Thus, in high OM environments, MeHg concentrations can be high due to the supply of labile C and acidic conditions. High S^{2-} concentrations can also affect pH; S^{2-} can convert to sulfuric acid upon exposure to aerated conditions resulting in a subsequent drop in pH and increased MeHg production. Other factors including chloride concentrations and turbidity also can influence the bioavailability of Hg^{2+}. Chloride can combine with Hg^{2+} to form more stable complexes that remain in solution (Du Laing et al., 2009) and high turbidity can decrease the rates of photochemical reduction of Hg^{2+} to Hg^0.

Although the effect of vegetation on Hg cycling has not been researched extensively, recent studies have shown plants and their roots may have significant impact on Hg biogeochemistry (Canário et al., 2007; Yee et al., 2008). Plant roots introduce oxygen into saturated wetland soils, decreasing S^{2-} concentrations and creating increased anoxic/oxic gradients, which are ideal environments for SRB and FeRB (Marvin-Di Pasquale et al., 2003). The root zone also provides labile OM, enhancing microbial activity. Wetland vegetation has been found to sequester Hg in plant tissue and reallocate Hg within the soil and solution profile (Lindberg et al., 2002; Best et al., 2007).

Taken together, this literature underscores the complexity of MeHg formation and cycling in aquatic environments and the challenge of predicting MeHg levels when one or more of the underlying drivers are changed, as might occur with large-scale transformation of managed wetlands to tidal wetlands.

2.3. Quantification of MeHg Export from Wetlands

As part of this literature review, area-normalized exports (or yields) of MeHg, reported in units of mass per unit area per unit time were summarized from prior work in similar environments. It is envisioned that the quantification of the MeHg exports from related work will provide a starting point for evaluating the contributions of MeHg from different types of wetlands in Suisun Marsh.

Error! Reference source not found. is a graphical representation of the range of MeHg yields from select tidal and non-tidal wetlands and other water bodies. These data are shown in tabular form in Table 5. The overall range of MeHg exports from both tidal and fresh water wetlands vary quite widely, from 10^{-11} to 10^{-8} g m^{-2} d^{-1}. Freshwater wetlands seem to have the widest variation however, spanning multiple orders of magnitude (10^{-11} to 10^{-8} g m^{-2} d^{-1}), while tidally influenced regions as a whole are confined between 10^{-9} and 10^{-8} g m^{-2} d^{-1}.

The narrow value range for tidally influenced regions may be due to the relatively frequent wetting and drying cycle, providing a steady-state MeHg production environment in contrast to many of the fresh water wetlands and flood plains that experience drastic changes in wetting and drying cycles throughout the year. Although there are seasonal differences in MeHg yield (i.e., Brown's Island, Figure 5), the range is usually still within an order of magnitude (4.6×10^{-9} to 9.9×10^{-9} g m^{-2} d^{-1}). In comparison to most freshwater wetlands, MeHg yield from tidal wetlands are generally higher (Figure 5); however, if just Delta wetlands are taken into account, both systems fall within the 10^{-9} and 10^{-8} g m^{-2} d^{-1} range, with some fresh water Delta wetlands producing much higher MeHg fluxes than the tidal wetlands. When considering total system flux that considers the area of the wetland, larger wetlands export more (Table 5). In the Delta alone, total flux values span orders of magnitude from a few ng d^{-1} to a few ug d^{-1}.

2.4. QUANTIFICATION OF METHYLATION RATES

In addition to the export of MeHg from wetlands as described above, it is also important to compare the first-order methylation rate, where the rate (in units of 1/time, typically 1/day) estimates the formation of MeHg for a given concentration of inorganic Hg. The rates are a way to explore the effect of methylation independent of mercury concentrations. This allows consideration of a broader range of studies because several studies that report export rates do not report methylation rates and vice versa.

Table 6 presents a summary of the methylation rates across wetlands and also the method used compute the rate. A regression of methylation rates with MeHg concentration in sediments is shown in Figure 6 and results in a poor correlation (r^2=0.1). Unlike export rates, methylation rates vary over more than two orders of magnitude, and exhibit significant variability even within a single wetland. The wide range in methylation rates may also reflect the variable methodology used to estimate the values; methylation rates are often measured as potential rates using radio-labeled mercury isotopes, as shown in Table 6. As the isotopes are more bioavailable than in situ Hg, rates are estimates only and should be used for comparison between locations. Some investigators adjust the measured rates using sediment inorganic mercury concentrations. Other methods that have been used include measuring methylmercury flux with DGT[2] strips in sediment pore water and the overlying water column. In addition to the use of potential methylation using radioisotopes, other factors not considered could confound the data. For example, many pore-water to surface water diffusion rates may be grossly under- or over-estimated as they do not take advection from wetland plants into account but instead assume diffusion to be the main driver behind MeHg transport from sediment to surface water (Bachand et al, 2013b). Sampling methodology can also affect flux data; although continuous seasonal

[2] From diffusive gradients in thin films.

sampling over multiple years is ideal, many studies can only take discrete samples with a few time points due to high costs associated with instrumentation and sampling procedures.

The evaluation of the methylation rate literature thus provides minimal guidance for methylmercury production in Suisun Marsh because of the vast differences in observed rates that may be attributed at least in part to differences in analysis and sampling methodology employed

2.5. Role of Marsh Vegetation in MeHg Cycling

Studies have shown plant roots and transpiration may have a substantial effect on internal wetland hydrology and diel/seasonal Hg cycling (Windham et al., 2001; Canário et al., 2007; Bachand et al., 2013b). Transpiration in wetlands accounts for 50-90% of the evapotranspiration in shallow flooded soils, thus controlling a large portion of downward water flux into the sediment profile (Herbst and Kappen, 1999; Bachand et al., 2013b). A majority of transpiration has been noted to occur during the growing season and reversed in the winter senescing period, resulting in exfiltration of water from the root zone to surface water (Bachand et al., 2013b). This plant-mediated infiltration and exfiltration may thus be responsible for some seasonal concentration fluxes observed in the field. Many researchers have indeed noted increased Hg concentrations in winter months for various aquatic environments (Brigham et al., 2001; Pato et al., 2008; Siegel et al,. 2011) and some have correlated this phenomenon to plant senescence (Best et al., 2008, Bachand et al., 2013b).

Tidal wetlands in particular contain a unique array of plants that may further impact wetland hydrology and material cycling. Tidal wetland vegetation have adapted to tolerate high salinity sediments and solution as well as frequently inundated environments. Common forms of salt-tolerant plants, or halophytes, include *Salicornia virginica* (pickleweed) and *Spartina foliosa* (California cordgrass) and *Phragmites australis*. Flood tolerant plants include *Scirpus* (Tules), *Typha* (cattail), and *Juncus* species (rushes).

Halophytes and metal-tolerant plants selectively uptake ions in solution and store unwanted ions or metals such as Na and Hg in the root system, later excreting them through salt glands located above the sediment surface (Shabala and Mackay, 2011). Translocation of ions from root to salt glands is an active process and is thought to be a major pathway for Hg uptake and excretion rather than transpiration (Windham et al., 2001; Best et al., 2008). This redistribution of Hg from environments with high S^{2-} to relatively more oxidized regions can promote Hg bioavailability and contribute a substantial amount of Hg to the reactive sediment pool. In San Pablo Bay, CA, Yee et al., (2008) found Hg excreted from *Distichlis spicata* (spikegrass) leaf tips contributed 3-5% of the total reactive Hg^{2+} in surface sediments.

Translocation may also influence the partitioning and timing of Hg release into the water column; Canário et al., (2007) found below ground halophyte biomass contained 9 and 44 times the amount of THg and MeHg concentrations, respectively, than the surrounding sediment and plant biomass contributed 70 times more MeHg to sediment compared to non-vegetated areas. Best et al., (2008) found dead leaves of *S. virginica* in San Pablo Bay contained 5-10 times THg compared to live leaves (102-191 ng g^{-1} and 7-33 ng g^{-1} for dead and live leaves, respectively) and that both *S. virginica* and *S. foliosa* lost 50% of their biomass per year (Best et al., 2004). The authors hypothesized Hg associated in this plant tissue is returned to the aquatic and sediment environment through fragmentation and degradation, as high solution Hg concentrations coincided with the senescing season.

The storage of Hg in root biomass may also restrict Hg flux from pore waters contributing to lower than expected surface water concentrations of MeHg compared to the pore water (Hall et al., 2008). Plant matter release during the senescing period could not only contribute to increased solution concentrations and export of THg, but could also play a large role in Hg bioaccumulation in herbivores and detritovores. Other halophytes that are sources of Hg through salt gland secretion include *Avicennia marina* (mangrove), *Spartina alterniflora*, and *P. australia* (Weis and Weis, 2004, and references therein). However, other studies show that certain halophyte species may aid in storing and accumulating Hg from solution and sediment. Phytoremediation using halophytes has been applied to various contaminated saline environments to sequester heavy metals including Hg with successful results (De Souza et al., 1999; Khondaker and Caldwell, 2003). Marques et al. (2011) found *Juncus maritimus* and *Scirpus maritimus* species sequestered Hg through phytostabilization (via complexation in the rhizosphere) and phytaccumulation of Hg in the below ground biomass. Using other metals, Castro et al. (2009) noted *S. maritimus* stored 90% of the metals below ground and only a small percentage was translocated above ground.

Vegetation is also a large source of labile OM, stimulating microbial activity and promoting Hg bioavailability. Flood tolerant plants distribute oxygen to the root region, which alters redox conditions in soil, affecting SO_4 and Fe cycling, Hg bioavailability, and microbial populations (Lillebø et al., 2010; O'Driscoll et al., 2011). Marvin-Di Pasquale et al. (2003) noted methylation rates in vegetated marshes in San Pablo Bay, CA were 25 times higher than open water areas (Table 6). Removal of vegetation in wetlands in a San Francisco Bay salt marsh resulted in a 36 and 38% decrease in methylation rates and microbial activity, respectively (Windham-Myers et al., 2009). Studies have also shown higher concentrations of MeHg at deeper sediment profiles in the presence of vegetation, most likely due to OM from root exudation and O_2 effects on surrounding redox (Canario et al., 2007).

Limited research is available on vegetation effects on Hg emissions from wetlands; however, Lindberg et al. (2002) found that Hg vapor emission from cattail and saw grass species exceeded that of water evasion in the Florida Everglades, as can be seen in Table 7. Atmospheric emissions of Hg could potentially return to the aquatic system via particle deposition; however, understanding the importance of this factor in Suisun Marsh Hg cycling will require further studies.

3. Restoring Muted Tidal Wetlands to Tidal Conditions

Restoring managed muted tidal or non-tidal wetlands to tidal conditions in Suisun Marsh will dramatically affect local and sometimes regional biogeochemistry and hydrology. This section benefits from analysis conducted on Blacklock Marsh, an area within Suisun Marsh that was converted from managed wetland to tidal wetland following an accidental levee breach, thus representing conditions that are similar to what might be encountered at other restoration sites in the region. Bachand et al. (2013a) describe an order of magnitude increase in flow and tidal exchange and greatly lower MeHg concentrations in the surface waters at the Blacklock marsh after levee breaching restored the system to full tidal exchange. Heim et al. (2013) record lower MeHg sediment concentrations and concentrations in inland silversides for the same system when comparing post-breach to pre-breach conditions. In this section, we describe the changes to hydrology, wetland area and vegetation management based upon available information and assess how those changes might affect the drivers of MeHg production and cycling (**Error! Reference source not found.**).

3.1. Changes to Hydrology, Wetland Area and Vegetation Management

Non-tidal and muted-tidal wetland water levels are controlled either seasonally or anthropogenically (i.e., salinity gates). These controls can result in relatively long periods of sediment inundation or exposure (days to months). Reduced compounds in sediment can become oxidized during these drying periods, releasing complexed Hg^{2+}, increasing Hg bioavailability and potentially increasing SO_4 concentrations (Gilmour et al., 2004). Upon flooding, the combination of available Hg^{2+}, SO_4, and OM (mainly from plant deposition) may result in a spike of MeHg production and concentration lasting up to months (Gilmour et al., 2004, Siegel et al., 2010). Siegel et al. (2010) report large dissolved oxygen (DO) drops and elevated DOC exports in the fall season in Suisun Marsh when managed wetlands are initially flooded to prepare for duck season. These conditions are expected to promote MeHg production (Bachand et al, 2013b; Alpers et al 2013). Soils that are allowed to dry completely seem to show the highest concentrations of MeHg upon re-flooding (Gustin et al., 2006; Alpers et al., 2008; Yee et al., 2008). Therefore, seasonal wetting/drying cycles conducted in Suisun Marsh where soil is exposed for the entire summer period provide ideal conditions for high MeHg concentrations upon re-flooding in the fall and winter.

Tidal wetlands experience daily fluxes in water level, salinity, temperature, and constituent concentration (Davis et al., 2003). Restoring a managed wetland to fully tidal wetland will profoundly hydrology with regard to source water, tide frequency and magnitude, flow, water depths and inundation frequency. Bachand et al (2013a) assessed the Blacklock marsh in Suisun Marsh pre- and post-levee breach. Before the levee breach, wetland stage remained relatively constant, with a twice-daily tidal cycle range of about 0.2 – 0.4 ft and about 4% of the total volume within Blacklock marsh exchanged with the slough. These water exchange rates are similar to those found at other managed wetlands within Suisun Marsh (Siegel et al., 2010). After the restoration, a new equilibrium condition began to occur. Under that equilibrium condition, tidal exchange between the wetlands and the slough increased by about 25 times; 80% of the water volume in the wetland exchanged each tidal cycle; tidal stage varied on average by about 4 ft each tidal cycle; high tide stage increased by over one foot from under 5 ft-NAVD under muted tidal conditions to over 6 ft-NAVD under full tidal conditions; and the active marsh area increased because of the occasional inundation of high marsh areas (Bachand et al, 2013a).

Error! Reference source not found. provides a summary of expected hydrologic, area and management differences between tidal and muted tidal systems. Restoring a managed or muted tidal wetland to a full tidal wetland is expected to affect the hydrology through the following processes:

- Increasing tidal exchange on an order of magnitude;
- Tidal exchange resulting in most of the water within the wetland being exchanged during each tidal cycle;
- Higher high tide elevations and much lower low tide elevations leading to daily to twice daily flooding and draining for wetland below average high tide elevations and less frequent flood/draining cycles for wetland areas above high average high tide elevations;
- Increasing flow rates through channels into wetlands, resulting in channel widening and enlargement to accommodate increased tidal prisms, reversing long-term trends of channel sizes decreasing as a result of reduced flows and tidal prisms under muted tidal conditions.
- Wetland areas will effectively increase with higher high tides resulting in higher elevations being occasionally wetted. Within Suisun Marsh, vegetation management goals will also change from managing to optimize duck hunting to managing for ecological goals (e.g., support for endangered species, reduction of invasive species).

Although several studies have noted the importance of soil wetting and drying cycles on MeHg cycling and production in wetlands (Gilmour et al., 2004; Gustin et al., 2006; Miles and Ricca, 2010), few of these have considered tidal wetlands and even fewer have observed the effects of restoring a wetland from muted/seasonal to fully tidal wetland.

3.2. Impacts of Tidal Restoration on Drivers of MeHg Production and Cycling

As discussed in Section 0, there are several key drivers of MeHg biogeochemical production and cycling: SO_4^{2-} or Fe(II) reducing bacteria associated with mercury methylation, available or labile forms of organic carbon, temperature, pH, and salinity. The effects of these drivers in the context of tidal restoration are discussed below.

3.2.1 Redox and DOC

A restoration of a muted tidal wetland to a full tidal wetland will affect redox conditions both spatially and temporally.

Spatially, we expect a redox gradient from more oxidized upper marsh conditions to more reduced lower marsh conditions. In higher elevations or regions with low tidal reach, soils may undergo frequent exposure, resulting in an almost continuous oxic/anoxic gradient. Krabbenhoft et al. (2005) found this almost continuous gradient to favor MeHg production. Many studies noted a drop in MeHg concentrations after some time (up to two years) if the wetland remained flooded or the sediments did not undergo complete drying (St Louis et al., 2004; Gustin et al., 2006; Marvin-Di Pasquale et al., 2009; Siegel et al., 2011). Long-term flooded and open water regions also seem to have lower concentrations of MeHg (Marvin-Di Pasquale and Cox, 2007; Alpers et al., 2008; Stephenson, Foe, et al., 2008). This may partly be attributed to lower DOC levels over areas that remain inundated and anoxic for longer periods of time.

Although the redox gradient may spatially increase in tidal wetlands, MeHg concentrations in sediment and water may not necessarily increase. Lower tidal regions that are more frequently inundated or permanently inundated as well as high elevation tidal regions that do not completely dry out when exposed may not experience substantial redox changes, which negatively affects MeHg production. Moreover, the temporal redox gradient is expected to decrease. For instance, Yee et al., (2011), found sediment MeHg concentrations did not respond to short term redox changes induced by tidal exchange in some San Francisco Bay tidal wetlands. In the case of more permanently flooded or saturated wetlands, MeHg concentrations and production may decrease for a number of reasons: a decrease in readily available labile organic matter; redox conditions not enabling Fe^{3+} or SO_4^{2-} reduction and subsequent MeHg production; complexation of S^{2-} with Hg^{2+} reducing its bioavailability. In contrast, rice fields show increases in MeHg loading when reflooded after periods of harvesting (Bachand et al, 2013b; Alpers et al, 2013), restoration of Blacklock Marsh showed a substantial increase in MeHg export when first breached (Bachand et al, 2013a) and managed marshes in Suisun have caused severe DO drops in channels when flooded in the fall after a season of agricultural management (Siegel et al, 2011). These studies indicate that after extended dry periods in which organic materials have grown and then senesced naturally or through harvest, reflooding establishes redox conditions and opportunity for elevated MeHg production and export. These monthly or seasonal changes are more characteristic of the anthropogenically managed wetlands, and are expected to be lower in restored tidal marshes.

3.2.2 Salinity

Estuarine input can increase SO_4^{2-} concentrations and the ionic strength in the water column, which in turn affects Hg cycling. In some regions such as the Florida Everglades, sulfur chemistry is the main driving factor behind Hg methylation (Mitchell and Gilmour, 2008) and presence of S^{2-} in pore waters of some tidal wetlands has been found to be more influential than the presence of THg in determining MeHg production (Merritt and Amirbahman, 2009). Salinity has been found to both enhance and inhibit Hg methylation. Chloride ions form strong complexes with metals in water, increasing their solubility and desorption from soil (Du Laing et al., 2009). Thus, increasing salinity may enhance Hg^{2+} bioavailability by forming neutral dissolved species that can easily diffuse into cell membranes. Hollweg et al. (2009) found increased salinity produced MeHg within deeper sediment profiles. An influx of saline water can also result in Hg removal, however, through increased ionic strength and subsequent precipitation of constituents (Babiarz et al., 2001). In addition, Hg demethylation has been found to increase in saline environments (Compeau and Bartha, 1987).

The topography and geography of wetlands have a large effect on sediment and solution salinity. Although upland tidal wetlands are inundated less frequently and receive more fresh water inputs from rivers, increased drying periods may cause salt accumulation in sediment (Mitsch and Gosselink, 2000). Wetlands receiving water primarily from estuaries or saline groundwater will also exhibit more saline water and sediment. Soil type also influences salinity; clays and silts tend to retain more salts compared to sand, while more porous sediments can be flushed and will be less saline (Mitsch and Gosselink, 2000).

Sediments in tidal environments maintain relatively consistent salt levels due to frequent flushing of water (DWR, 2001). In Suisun Marsh, dike construction, which isolates flooded areas in conjunction with various flooding regimes and salinity gate control has resulted in sediments high in salt. As the area was once a brackish tidal wetland, the soils of the region were already saline. Upon conversion for agricultural use, sediments were leached to remove salts in order to plant crops (DWR, 2001). Transition of the land for waterfowl use altered the flooding regime consisting of fall and spring flooding, water drainage following water fowl hunting season, and maintaining drained conditions during the summer. During the dry season, salt accumulates through water evaporation and saline water drawn up from lower areas of the soil profile. Although flooding in fall and spring is partly intended to leach salts from soil, over time, sediment salinity has increased to toxic concentrations (Baginska, 2012). This is especially the case for lowland areas that cannot drain completely (Siegel et al., 2011). Converting the area to a tidal marsh may have significant effects on salt concentrations in the region, affecting plant and animal populations, sediment and aquatic biogeochemistry.

3.2.3 Marsh Area, Tidal Flows and Tidal Prism

Hydrology differs greatly between tidal and non-tidal wetlands, having implications on the magnitude and frequency of constituent transport, sediment drying and wetting frequencies and duration, and subsequent effects on redox conditions, influencing metal speciation, microbial activity, and Hg bioavailability. Wetland area is also important; MeHg flux values vary quite significantly depending on the wetland footprint with larger wetlands exporting greater total loads. In the Delta alone, flux values span orders of magnitude from a few ng d^{-1} to a few ug d^{-1} depending on wetland acreage.

Many managed wetlands and sloughs are isolated from large water bodies and can form concentrated regions of MeHg due to minimal mixing and dilution (Bachand et al., 2013a). Flooding these areas may result in an initial release of large amounts of MeHg accumulated in sediment and subsequent export downstream, as discussed above. Tidal wetlands, in contrast, experience daily fluctuations in water level which shift on a bi-monthly seasonal basis due to lunar activity, as well as random fluctuations caused by storms and wind (Siegel et al., 2010; Bergamaschi et al., 2012). This increased tidal prism and substantial exchange with adjacent water bodies can cause tidal wetlands to become significant sources or sinks of material (Davis et al., 2003; Merritt and Amirbahman, 2009; Das et al., 2012). To date, a small number of studies have been conducted where material flux from tidal wetlands was measured, particularly for THg and MeHg (Pato et al., 2008; Hollweg et al., 2009; Yang et al., 2010; Bergamaschi, Fleck, et al., 2012). These studies suggest both POC and DOC significantly influence THg and MeHg transport and tidal wetlands are sinks for POC but sources of DOC and associated Hg fractions (Pato et al., 2008; Mitchell et al., 2012). Particulate-associated Hg (PHg) was affected more by storms and wind while dissolved Hg (DHg) was mostly tidally driven (Pato et al., 2008; Bergamaschi et al., 2011). Season, wind, and storms are major factors affecting Hg fluxes from tidal wetlands to surrounding water bodies; mangrove swamps in the Florida Everglades were a net source of both THg and MeHg but more so during the wet season due to increased water levels (Bergamaschi et al. 2012). In other wetland areas, however, the combined seasonal effects of wind and tide resulted in net exports of zero (Bergamaschi, Krabbenhoft, et al., 2012), which indicate continuous seasonal sampling is necessary to obtain accurate flux data.

Frequent flushing of a wetland system could also promote Hg methylation. Input of fresh water can re-mobilize nutrients and Hg associated with particulate matter, increasing their dissolution and bioavailability. Constant flushing of material can also remove constituents such as S^{2-} from sediment pore waters and maintain redox conditions that favor Hg methylation. Marvin-Di Pasquale et al., (2007) noted an increase in bioavailable Hg^{2+} when anaerobic marsh sediment was flushed with fresh water and hypothesized the introduction of oxygenated waters created varying redox conditions conducive to microbial activity.

Finally, continuous flushing may also decrease MeHg in a wetland over time, especially regions that have been converted from a managed to tidal system through dilution. After conversion of the Blacklock marsh from a muted tidal system to a full tidal system, net export of MeHg increased from 78 – 139 ng m^{-2} d^{-1} ($8x10^{-8}$ to $1.4x10^{-7}$ g m^{-2} d^{-1})(Bachand et al., 2013a). Thus the export rates were generally higher than reported in the literature (Table 5). However, water exchange with the adjacent system increased by 25 times, lowering MeHg concentrations in the adjacent sloughs. The intermediate transition stage from a muted tidal to non-tidal system experience much higher fluxes of MeHg and the authors attributed this to a greater flux of MeHg from the sediments (via diffusion) because of lower water column concentrations in the water and because of the opening up of additional acreage in the higher marsh area with its available organic carbon, previously not undergoing wetting and drying conditions. However, once equilibrium conditions re-established and excess produced MeHg was flushed from the system, hydrology greatly controlled MeHg concentrations through greatly diluting its concentration in the water column.

3.2.4 Effects of Vegetation

The creation of more open water areas with longer inundation in portions of restored tidal marshes are expected to reduce DOC supply and methylation, similar to what was observed in San Francisco Bay salt marshes by Windham-Myers et al. (2009). The effects of plant population shifts and density, however, are quite complex. Tidal conversion of Suisun Marsh and other areas will most likely alter THg and MeHg distributions both laterally and vertically in the sediment profile. Many native halophyte plant species in Suisun Marsh include species that store Hg in their leaves and shoots. In San Pablo Bay, Best et al. (2008) found *S. foliosa* took up $3.0x10^{-4}$ and $5.7x10^{-6}$ g d^{-1} of THg and MeHg, respectively, while *S. virginica* took up $9.9x10^{-5}$ and $1.9x10^{-7}$ g d^{-1} of THg and MeHg, respectively. However, halophyte species distribution is influenced by salinity and inundation period, thus upon transition to a tidal environment, native species such as *Scirpus maritimus*, which sequester Hg, could potentially re-colonize the area. Siegel et al., (2011) found tidal water input into northern Suisun Marsh was higher than transpiration; however, increases in halophytes and population distribution may alter this pattern. Research should be conducted to assess different plant species effects on Hg distribution and how shifts in population densities during wetland transitions may affect MeHg concentrations.

4. RECOMMENDATIONS AND DATA GAPS

The data presented in this review suggest that the MeHg loadings from tidal marshes may be similar to those from non-tidal systems. In Suisun Marsh, the Blacklock data set (Bachand et al, 2013a; Heim et al, 2013) suggests that even though water column MeHg concentrations declined following conversion of a uted tidal system to a tidal wetland, loading could increase to small extent. This increase is largely attributable to significantly higher hydrologic exchange between the marsh and the surrounding slough waters.

Although the Blacklock study provides a scenario similar to the tidal conversion of Suisun Marsh, the flux and yield values of the region before tidal conversion was already much higher compared to other tidal marshes (8x 10^{-8} g m^{-2} d^{-1}, Table 5). Various factors may play into the high values exhibited from Blacklock. Sediments in this region are most likely highly organic Joice muck (with organic matter between 30-50 %) and clays (Figure 7), which may promote more vegetation and MeHg production and accumulation. In addition, the area is located in one of the few intertidal zones, which would experience more pronounced wet and dry cycles due to higher and lower water levels, although this would be more influential upon tidal conversion.

Given their sediment type and elevation, Grizzly Island and Brown's Island are better representatives of Suisun Marsh conditions. Much of Suisun Marsh, including Grizzly Island, is composed of clay loams and 43% of the region is sub tidal, or up to 1.8m below MLLW, while the other half is intertidal (Siegel et al., 2010, Figure 8 and 9). Thus, most areas in Suisun Marsh upon tidal conversion would be completely inundated, with tides influencing overlying water levels. Sub tidal areas would experience constant saturation of soil, while the intertidal region would most likely experience drying periods too short to affect significant soil oxidation and thus maintain redox conditions less suited for high MeHg production, as discussed above. In these inundated regions, vegetation types and redox states will probably be similar resulting in constant MeHg production rates. Using yield values taken from studies conducted in these regions, estimates of MeHg exports from Suisun Marsh tidal wetlands may range between $3x10^{-9}$ to $1x10^{-8}$ m^{-2} d^{-1}. In higher intertidal regions with high organic matter soils, however, yield values may reflect those of Blacklock, ranging higher than 10^{-8} m^{-2} d^{-1}. Further research should be conducted, however, to assess sediment accumulation rates in Suisun Marsh and potential changes in seasonal tidal ranges upon tidal conversion to estimate periods of sediment wetting and drying.

Although Blacklock MeHg exports are higher than what may be expected from the rest of Suisun Marsh, evidence from Blacklock suggests tidal conversion will result in a period of heightened Hg and MeHg transport and concentrations due to flushing of accumulated material in pore water and sediment due to current minimal mixing conditions (Bachand et al., 2011). This increase may span an order of magnitude in MeHg loads; however, over time (one or more years), concentrations should decrease due to equilibrium and dilution effects from tidal flushing and plant population shifts. Although post-breach MeHg loads remained higher than initial values, these values remain within an order of magnitude. Data from Blacklock as well as other studies should be viewed with caution given the small sample size and predominantly discrete sampling methodology. Small-scale studies to monitor changes in MeHg dynamics upon tidal restoration should be continued before embarking on a region-wide conversion. Finally, much higher flows due to tidal conditions would be expected to greatly reduce MeHg concentrations within the marsh and in the adjoining channels.

The following are the important additional considerations for restoration. First, converting managed wetlands to tidal systems is likely to reduce episodic events. These episodic events have been shown to occur in wetland type systems when flooded after prolonged dry periods in which vegetation management has occurred (Bachand et al, 2013a, 2013b; Siegel et al, 2011) and after periods of disturbance (Alpers et al, 2013; Bachand et al, 2013a). These events will be decreased with tidal wetland systems undergoing minimal anthropogenic management and disturbance. Additionally, the tidal systems experience much greater flow, essentially acting as tidal pumps. This pumping action draws water up and down the downstream and adjacent sloughs and within the wetland itself, greatly decreasing constituent concentrations when production rates are similar. These effects cannot be understated as they reduce spatial or temporal hotspots, and thus the opportunity for bioaccumulation and biomagnification through the food web.

With regard to the two questions raised at the outset of this review, the following recommendations are made:

(1) If large-scale tidal wetland conversion occurs in Suisun Marsh how will MeHg production, loads and concentrations change? Given current knowledge and data, there are expected to be decreases in MeHg production and concentrations; however, loads from wetlands to connected sloughs may either remain at similar levels, or increase slightly because of the greater hydrologic exchange in tidal versus managed wetland systems.

(2) Will restoration result in conditions more difficult to meet TMDL water quality criteria? Restoration, by reducing the episodic discharges of high MeHg and low DO water, will make it easier to meet water quality criteria in sloughs. This will happen because concentrations in wetland discharges will decrease and because the amelioration of low DO conditions is expected to decrease MeHg formation in slough sediments.

With regard to the processes discussed in this section, a number of data gaps exist:

- Loading and production data are based upon very limited data sets. However, as shown during a number of studies (Bachand et al, 2013a; 2013b; Alpers et al 2013; Heim et al 2013; Siegel et al 2011), anthropogenic disturbance (e.g., vegetation management; flooding) and transitional periods can suppress DO and elevate DOC, two drivers of MeHg production, and elevate MeHg exports. These effects are not easily captured using regular sampling intervals. This effect is exacerbated with Hg sampling and analyses where the expense associated with sample analyses restrict experimental designs and spatial and temporal sampling densities. Improved land or water management may be able to help manage MeHg exports from wetland systems (Siegel et al, 2011) yet higher density efforts or perhaps the use of in situ methods as indicators of MeHg production and transport (Bergamaschi et al 2011; 2012a; 2012b) are needed to fully understand the implications and potential of these efforts.

- Hydrology has generally been ignored in MeHg analyses. Most importantly for this issue is how will hydrology changes through Suisun affect MeHg concentrations. Bachand et al (2013b) analyses strongly suggests the much greater tidal pumping occurring with the restoration of wetlands to tidal wetlands will greatly decrease MeHg concentrations within the wetlands themselves as well as in the adjacent sloughs, greatly reducing temporal and spatial hotspots for mercury bioaccumulation and biomagnification. Understanding how to leverage these profound hydrologic changes to minimize MeHg challenges will be important and will require both modeling and data collection efforts.

5. Conclusions

Current data from muted tidal managed wetlands and tidal wetlands in the Delta suggest similar MeHg loading rates are on the order of the 10^{-9} and 10^{-8} g m^{-2} d^{-1} range. Our review suggests that for Suisun Marsh, MeHg exports from tidal wetlands may range between $3x10^{-9}$ to $1x10^{-8}$ m^{-2} d^{-1} and in higher intertidal regions with high organic matter soils, export values may reflect those estimated at Blacklock marsh, ranging above 10^{-8} m^{-2} d^{-1}. Clearly, managed muted tidal wetlands and tidal wetlands are very different with regard to marsh footprint, hydrology (i.e. magnitude, inundation frequency) and vegetation. All these factors greatly affect wetland biogeochemistry and thus MeHg production. Not enough is known to fully understand the effects on MeHg in Suisun if large wetland tracts currently managed for duck hunting and other anthropogenic activities are restored to tidal conditions. Our assessment of the biogeochemical drivers suggests that the tidal restoration results in many factors that could reduce episodic MeHg production (and the likely associated low DO events currently associated with managed wetlands in Suisun). Yet the available data suggests MeHg loading over the long term could potentially increase. However, other results from restoration should help. First, conversion to tidal wetlands will minimize anthropogenic management and resulting disturbance which have been associated with low DO events in Suisun Marsh and episodic MeHg load export events elsewhere. Second, hydrology will be greatly changed. Tidal wetlands act as tidal pumps. Converting from a managed to full tidal marsh will greatly increase tidal prism and flows, likely on an order of magnitude, and this effect will profoundly affect MeHg concentrations. We hypothesize that these two results will decrease spatial and temporal MeHg hotspots in Suisun Marsh. While this hypothesis suggests that conversion of managed wetlands to tidal marsh will be beneficial, ideally, further research, both with modeling efforts and through data collection at higher density across Suisun Marsh, should be considered before embarking on a regional scale tidal restoration.

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7. FIGURES

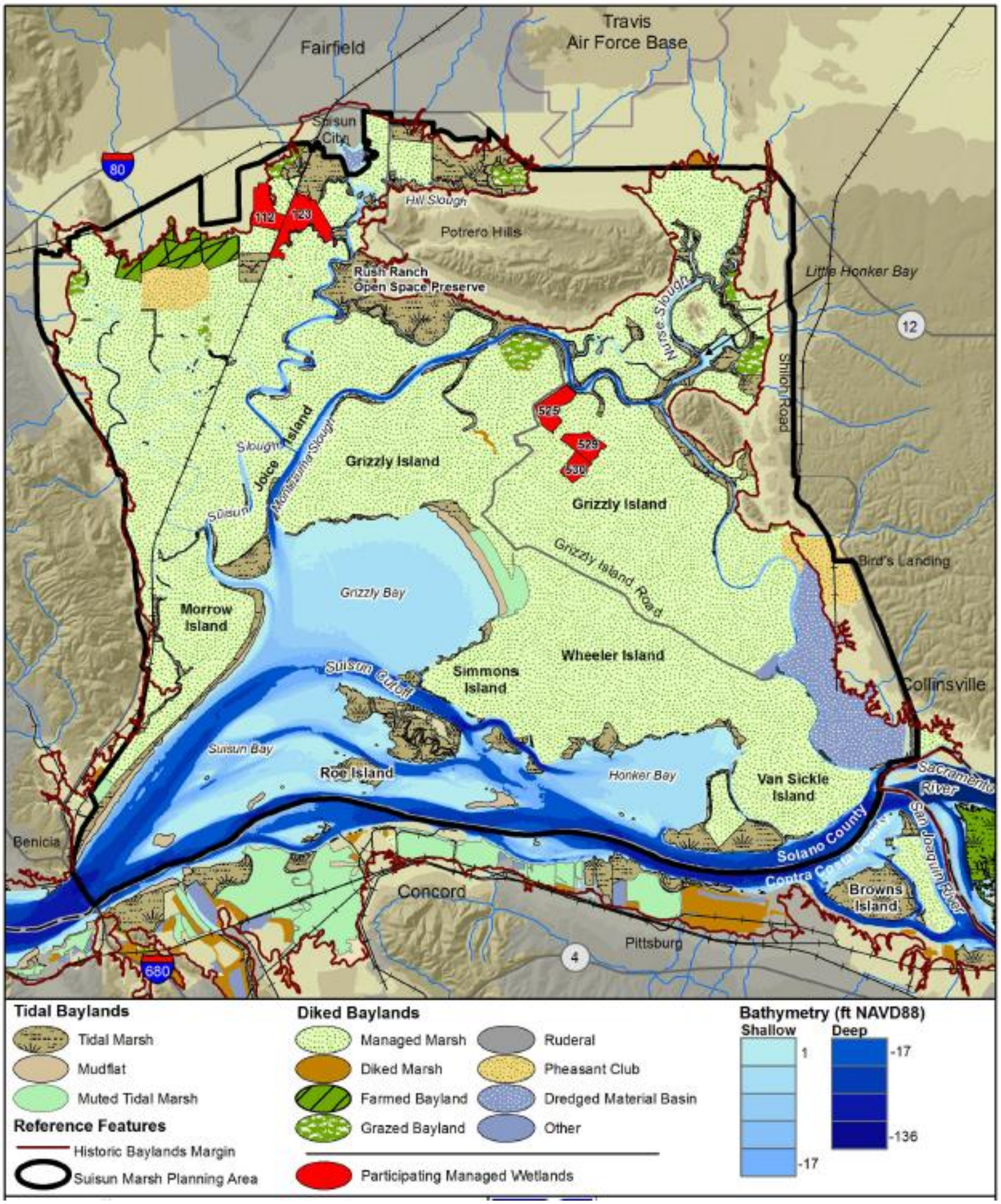
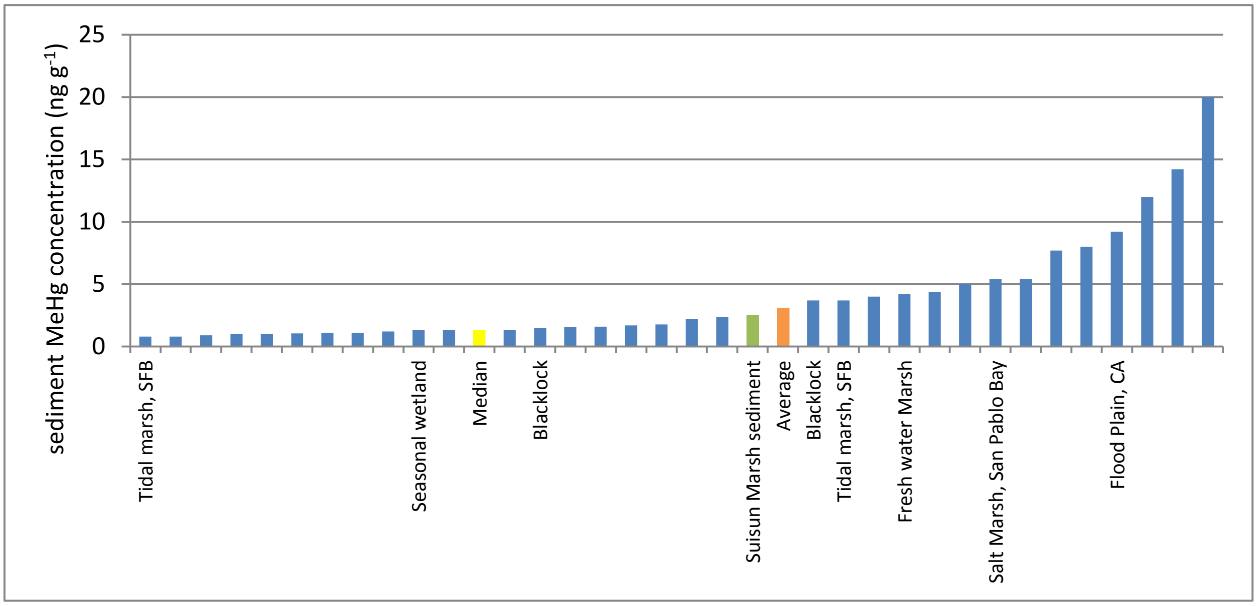


Figure 1. Existing Tidal Marsh and Managed Marsh Areas in Suisun Marsh (Siegel et al, 2011).

A.



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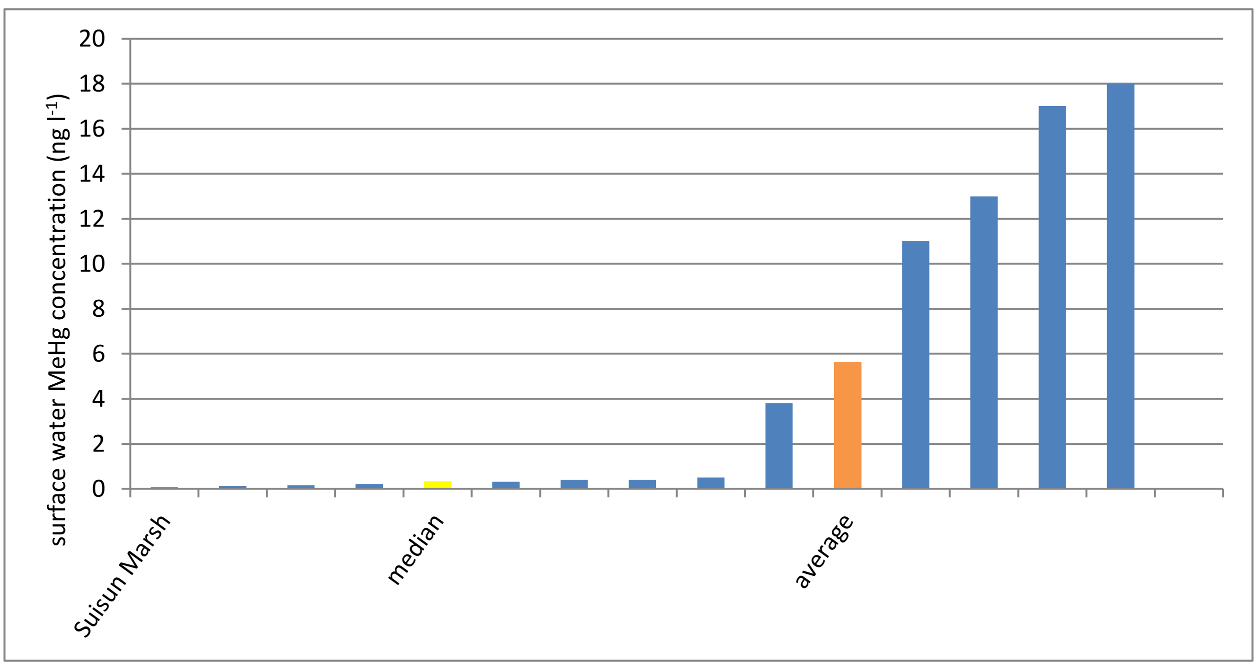

Figure 2. **Sediment (A) and surface water (B) MeHg distributions for various tidal and non-tidal regions. Select locations are identified on x-axis; mean and average values are highlighted in a separate color.**

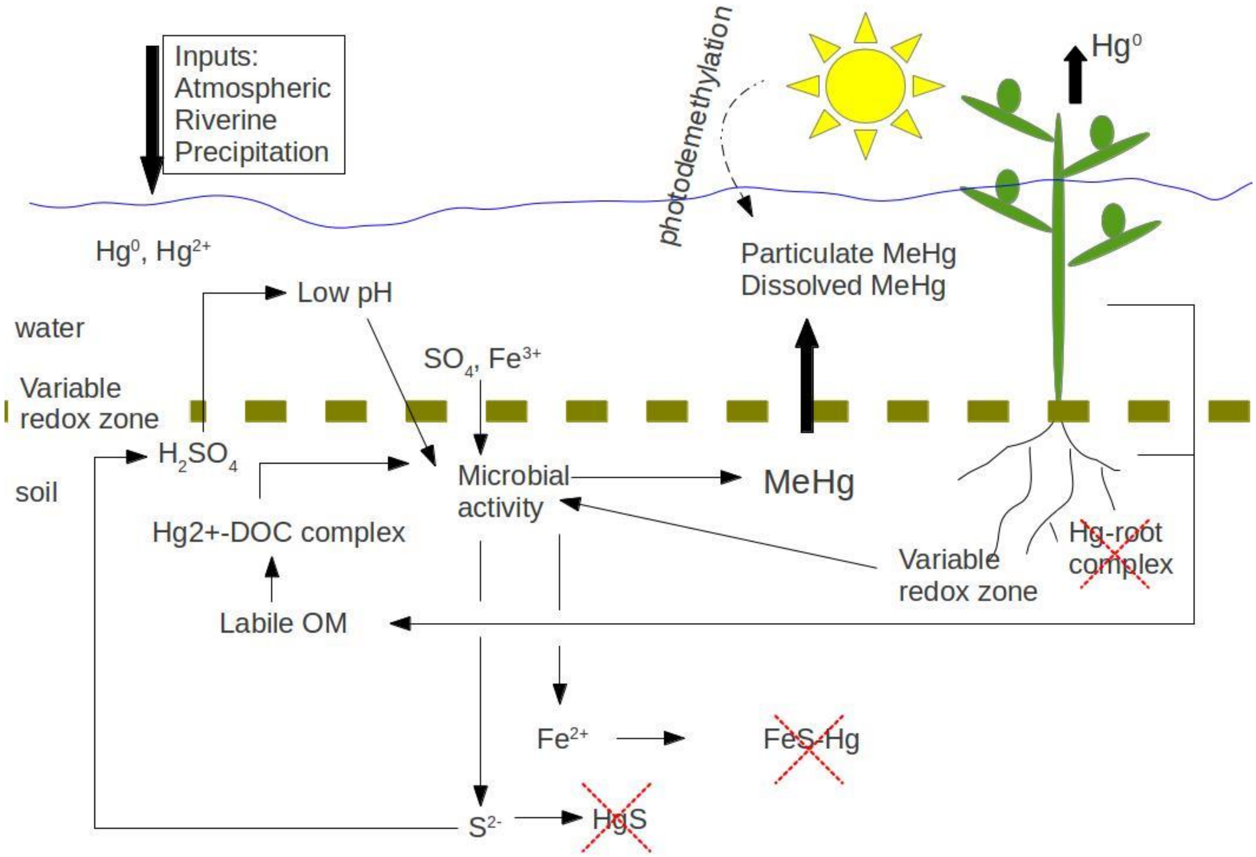

Figure 3. **Conceptual model of Hg cycling in a general wetland; crossed out species are considered non-reactive from the standpoint of methylation, dotted arrows are factors that inhibit production and reduce methylmercury concentrations.**

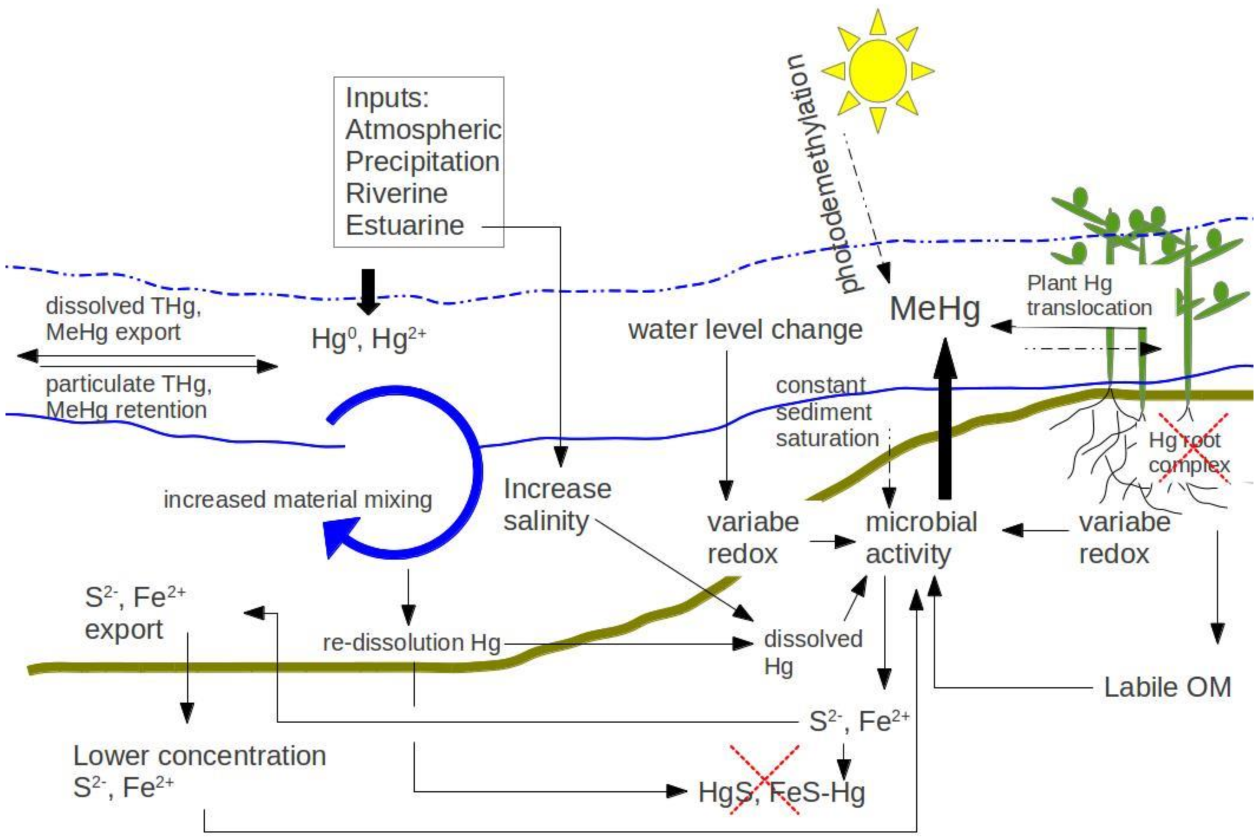

Figure 4. **Conceptual model of biogeochemical processes in tidal wetlands; crossed out species are considered non-reactive from the standpoint of methylation, dotted arrows are factors that inhibit production and reduce methylmercury concentrations.**

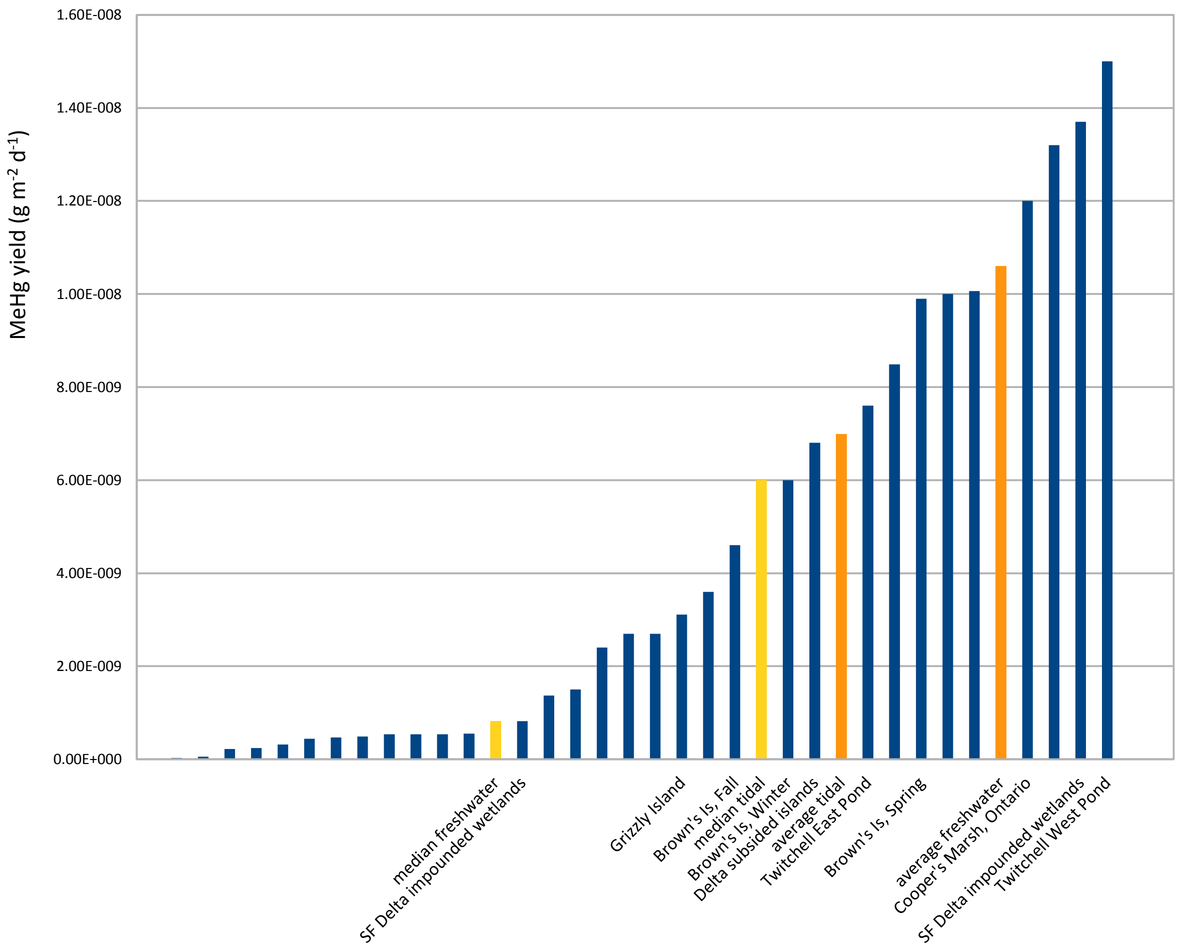

Figure 5. Distribution of MeHg yield for various tidal and non-tidal systems. Select locations are identified on x-axis, mean and averages are highlighted with a separate color.

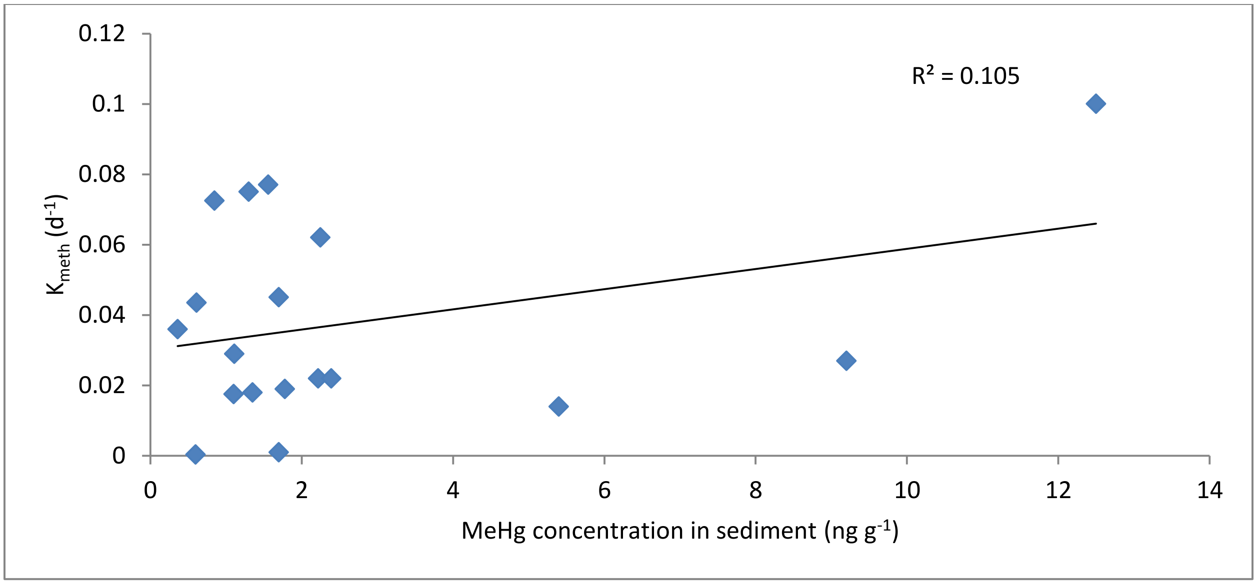

Figure 6. Correlation between MeHg sediment concentration and methylation rate (Kmeth).

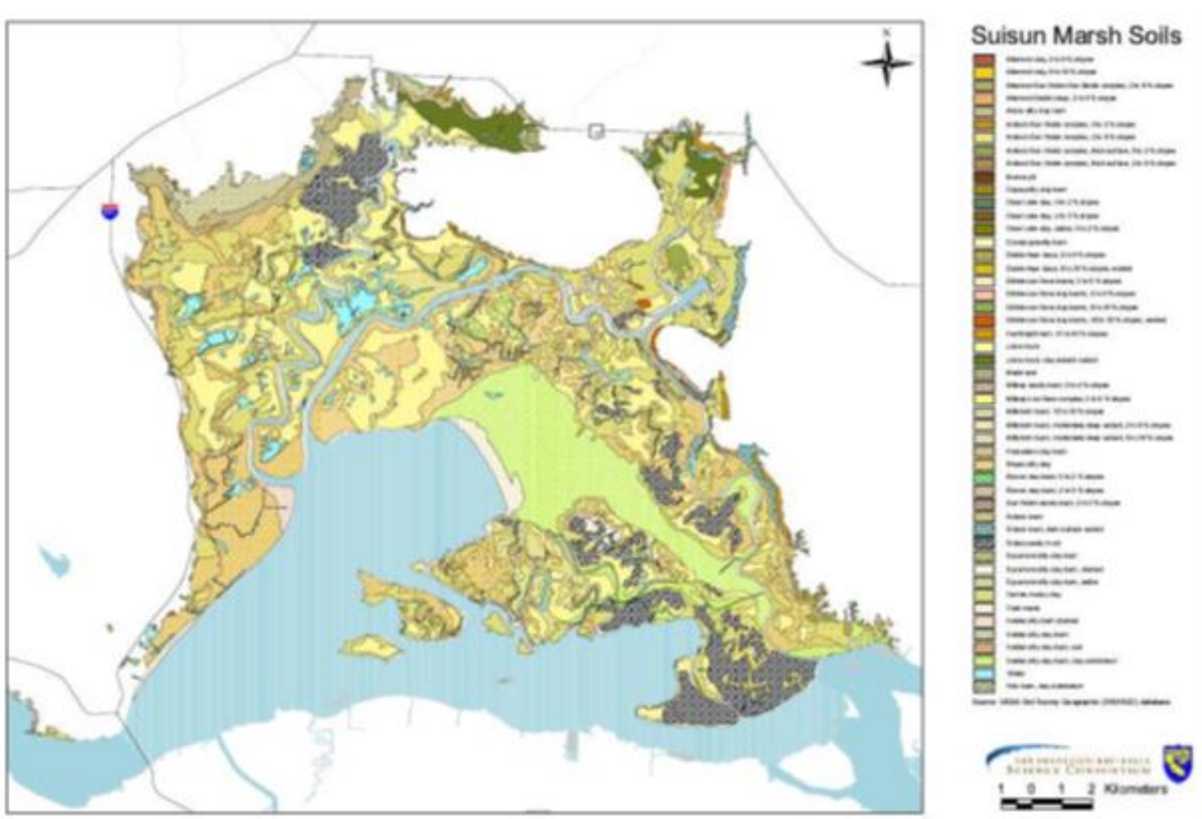

Figure 7. Sediment map of Suisun Marsh region (Department of Fish and Game)

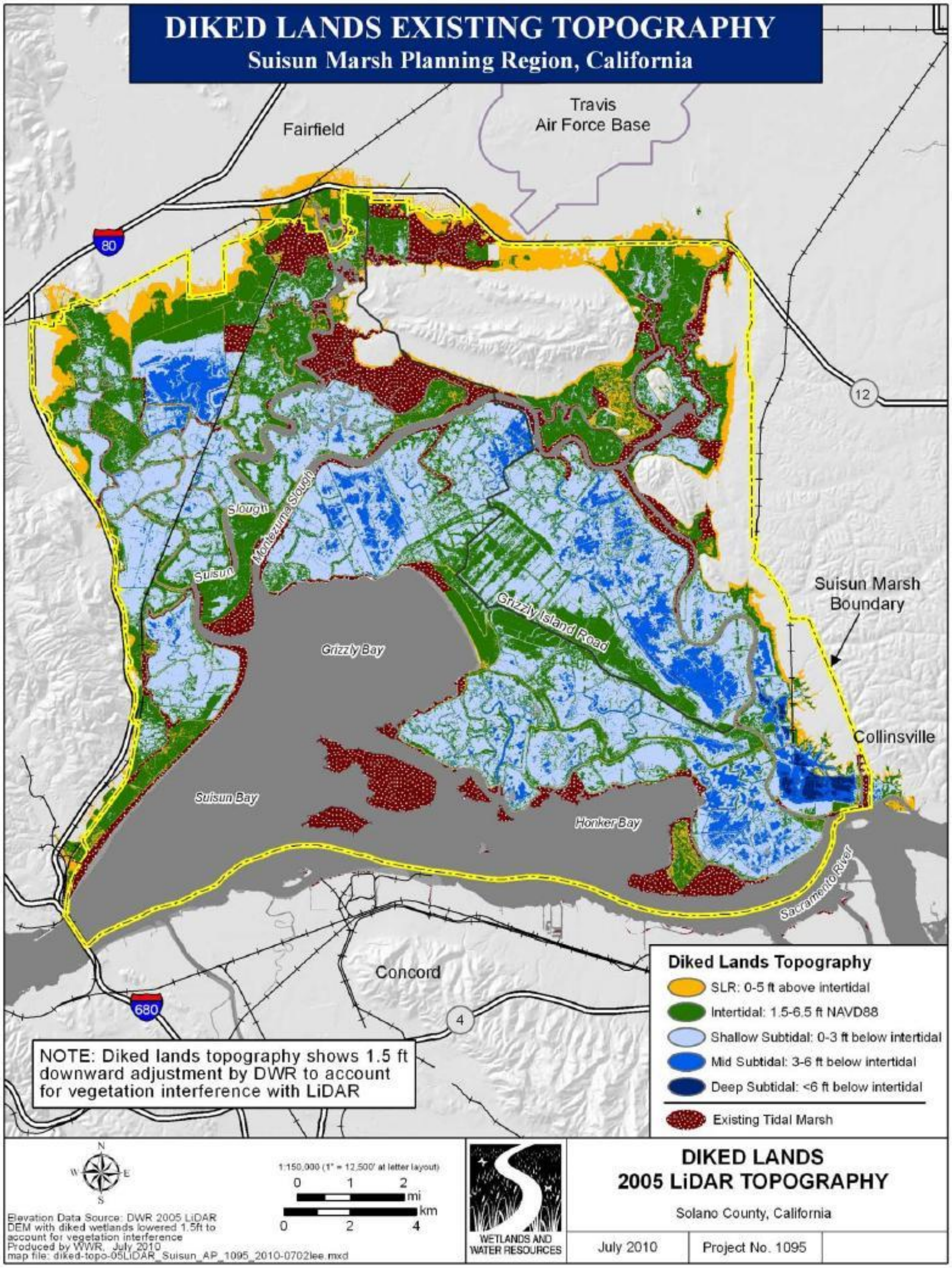

Figure 8. Topographical map of Suisun Marsh (from Siegel et al., 2010)

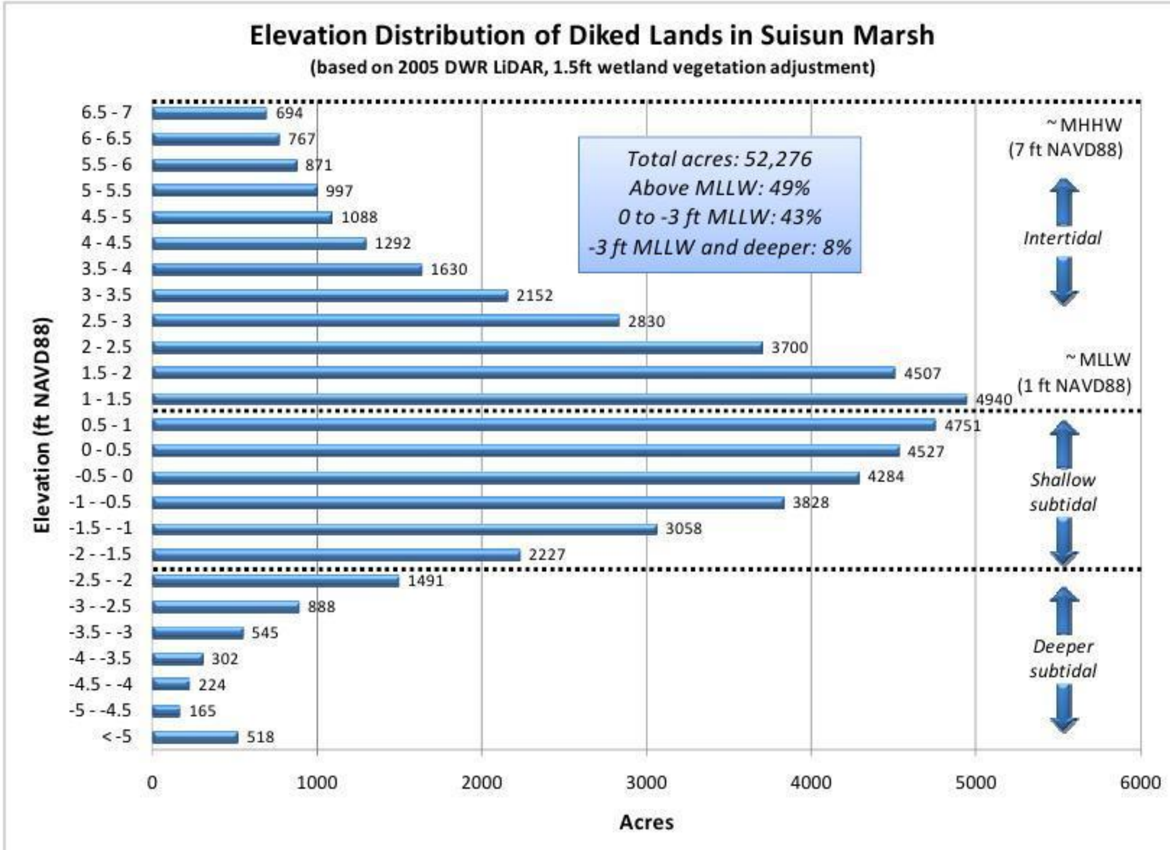

Figure 9. Distributions of tidal elevation in Suisun Marsh (from Siegel et al., 2010)

8. TABLES

Table 1
Sediment concentrations in select aquatic regions receiving tidal influence.
THg = total mercury, MeHg = methylmercury.

Location	Sample depth (cm)	THg Concentration (ng g^{-1} dw)	MeHg Concentration (ng g^{-1} dw)	Reference
Suisun Marsh, CA	5-10	110	2.5	*Siegel et al., 2011*
Blacklock (Suisun Marsh), CA	0-2	162-228	1.48-3.69	*Bachand et al., 2012*
Salt marsh and open water sites in San Pablo Bay, CA	0-4	marsh 300 open water 300-350	marsh 5.4 open water 0.45-0.75	*Marvin-Di Pasquale, et al 2003*
Tidal marshes, rice fields and seasonal wetlands around San Francisco Bay Delta, CA	0-2	Tidal marshes 256-333 Non veg tidal marsh area 280-559 Rice fields 298-411 Rice fields fallow 135-204 Seasonal wetland 169 Flood plain 109 Non vegetated 189	Tidal marshes 0.8-3.7 Non veg tidal marsh 0.6-1.6 Rice fields/fallow 0.7-1.0 Seasonal wetland 1.3 Flood plain 9.2 Non vegetated 1.7	*Windham-Myers, L., et al 2009*
Tidal wetlands along Petaluma River, CA	0-2	200-380	5-20	*Yee et al., 2008*
Various sites, San Francisco Estuary, CA			0.1-1.1	*Conaway et al., 2003*
Hypersaline marsh, South San Francisco Bay, CA	0-2	260-711	7.7	*Grenier et al., 2010*
San Francisco Bay Delta, CA	0-0.5	100-350	<1-8	*Heim et al., 2007*
Cache Creek and Yolo Bypass, CA	0-2	0.3 – 959	0.4-1.2	*Marvin-Di Pasquale et al., 2009*
Salt marsh and open water sites in San Pablo Bay, CA	marsh 0-4 open water 0-50	300-600	marsh 5.4 open water 0.45-0.75	*Marvin-Di Pasquale et al.,* 2003
Mugu Lagoon, CA	0-30	18.1	0.91	*Rothenberg et al., 2008*
Alviso Slough, CA		*(Hg^{2+})*0.3		*Marvin-Di Pasquale et al., 2007*
Tributary streams, waterways, open water sites in San Francisco Bay, CA	0-10	2	0.1-14.2	*Choe et al., 2007*

Location	**Sample depth (cm)**	**THg Concentration (ng g^{-1} dw)**	**MeHg Concentration (ng g^{-1} dw)**	**Reference**
Wetlands along San Pablo Bay, CA		*non vegetated* HAAF 378 CC 327 *S.foliosa* HAAF 407 CC 371 *S. virginica* HAAF 314 CC 304	*non vegetated* HAAF 1.78 CC 1.56 *S.foliosa* HAAF 1.35 CC 2.22 *S. virginica* HAAF 1.11 CC 2.39	Best et al., 2005
Tidal wetlands along San Pablo Bay, CA	0-25	300	vegetated 0.72 non-vegetated 4.4	*Best et al., 2007*
Kirkpatrick Marsh, MD		125-153	0.36	*Mitchell and Gilmour, 2008*
Cabretta Island salt marsh, GA	0-2	3.4-15.7		*Das et al., 2012*
Fresh water and salt water marshes in Barataria Basin, LA	0-50	fresh water marsh 140 salt water marsh 80	fresh water marsh 4.2 salt water marsh 1.3	*Kongchum et al., 2006*
Salt marshes in Bay of Fundy, New Brunswick, Canada	15-25	7 – 79		*Hung and Chmura, 2006*
Tidal mudflat and coastal wetland in Minas Basin, Bay of Fundy, Nova Scotia	0-30	coastal wetland 17.4 mudflat 0.5-23.7	coastal wetland 0.25 mudflat 2.7×10^{-3}	*O'Driscoll et al., 2011*
Cooper's Marsh, Ontario	0-2	66-223	0.8-12	*Holmes and Lean, 2006*
Marshes in Tagus, Sado, and Guadiana estruaries, Portugal	2-5	*vegetated* Tagus 500-2370 Sado 700- 2790 Guadiana 270-1070 *non-vegetated* Targus 770-1920 Sado 580-1070 Guadiana 160-1290	*vegetated* <6% of THg *non-vegetate*d up to 18% of THg	*Canario et al., 2007*
Sediment shelf, Chesapeake Bay, MD		50.1 to 170.5**	0.17 to 1.05**	Hollweg et al., 2009
Ria de Aveiro estuary, Portugal	0-1	9400 – 1020		*Lillebo et al., 2010*

Table 2
Surface water concentrations in select aquatic regions receiving tidal influence. THg = total mercury, MeHg = methylmercury.

Location	concentration THg (ng l^{-1})	concentration MeHg (ng l^{-1})	Reference
Suisun Marsh, CA	7-37	0.08-0.4	*Siegel et al., 2011*
Blacklock (Suisun Marsh), CA	*Unfiltered* Pre-breach 5.18 Post-breach 5.65	*Unfiltered* Pre-breach 1.03-1.47 Post-breach 0.29	*Bachand et al., 2012*
Brown's Island, CA		*Unfiltered* Fall 0.03-0.14 Winter 0.11-0.22 Spring 0.09-0.26 *Filtered* Fall 0-0.05 Winter 0.06-0.16 Spring 0.04-0.13	*Bergamaschi et al., 2011*
Hypersaline Pond, channel along Aviso Slough, fringing marsh, CA	*Unfiltered* Pond 60 Marsh 21 Slough 23	*Unfiltered* Pond 2.88 Marsh 0.52 Slough 0.38	*Grenier et al., 2010*
Cache Creek and Yolo Bypass, CA	*Unfiltered* 5.1 – 76.6 Filtered 0.6-2.4	*Unfiltered* 0.2-29.2 *Filtered* <3.8 – 41.9	*Marvin-Di Pasquale et al., 2009*
Mugu Lagoon, CA	*Unfiltered* 4.41 *Filtered* 1.08	*Unfiltered* 0.258 *Filtered* 0.047	*Rothenberg et al., 2008*
Malibu Lagoon, CA	2.0-1.4	0.02-0.4	*Ganguli et al., 2012*
San Francisco Estuary, CA	*Unfiltered* 0.146-88	*Unfiltered* 0.01-0.494	*Conaway et al., 2003*
Marine/brackish/freshwater wetlands, LA		Freshwater wetland 0.31 Brackish lakes <0.1 Brackish wetlands 0.5	*Hall et al., 2008*
Ria de Aveiro estuary, Portugal		*Salicornia perenis* mats 18-17 Non vegetated 11-13	Lillebo et al., 2010

Table 3
Pore water concentrations for select aquatic regions receiving tidal influence. THg = total mercury, MeHg = methylmercury. All samples are filtered unless otherwise stated.

Location	THg Concentration (ng l^{-1})	MeHg Concentration (ng l^{-1})	Reference
Tributary streams, waterways, open water sites in San Francisco Bay, CA	2-52	4.3	*Choe et al., 2007*
Marine/brackish/freshwater wetlands, LA	*Unfiltered* Marine wetland 1.9 Brackish wetland 4 Freshwater wetland 0.8 *Filtered* Marine wetland 0.7 Brackish wetland 1.6 Fresh water wetland 1.6	*Unfiltered* Marine wetland 0.1 Brackish wetland 0.2 Freshwater wetland 0.1 *Filtered* Marine wetland 0.06 Brackish wetland 0.3 Fresh water wetland 0.3	*Hall et al., 2008*
Tivoli South Bay tidal mudflat, NY	190-1040	0.4-2.9	*Zelewski et al., 2001*
Cooper's Marsh, Ontario	1.5-2.6	0.1-0.7	*Holmes and Lean, 2006*

Table 4
Factors affecting Hg cycling in tidal wetlands (Davis et al., 2003)

Variable	Relationship with methylmercury production	References
Oxygen	Sulfate reducing bacteria are the primary methylators and require anaerobic conditions	Compeau and Bartha 1985; Regnell and others 1996; Gilmour and others 1998; Choi and Bartha 1994
pH	Low pH associated with increased methylation	Xun and others 1987; Westcott and Kalff 1996, Rose and others 1999
Sulfate	In low sulfate waters, increased sulfate increases methylation.	Chen and others 1997
Sulfate	In high sulfate waters (e.g., estuaries) increased sulfate increases demethylation. Increased sulfide decreases methylation.	Oremland and others 1995; Benoit and others 1998
Dissolved organic carbon	High DOC in the water column may indicate high organic loading leading to high bacterial activity and anoxic sediments.	Watras and others 1995; Krabbenhoft and others 1995; Driscoll and others 1995
Dissolved organic carbon	Complexation of Hg species by DOC may increase dissolved concentrations without appreciably increasing biological uptake.	Barkay and others 1997
Temperature	In general, higher temperatures (up to 35 to 40 °C) result in higher bacterial activity.	Choi and Bartha 1994
Salinity	Bacterial and algal mercury uptake related to concentration of neutral species ($HgCl_2$, MeHgCl)	Barkay and others 1997; Mason and others 1995

Table 5
Summary of fluxes and yields from various wetland and aquatic systems.

Location	Area (ha)	Salinity (psu)	Tidal Range (cm)	Daily flux (g d^{-1}) THg	Daily flux (g d^{-1}) MeHg	Yield (g m^{-2} d^{-1}) THg	Yield (g m^{-2} d^{-1}) MeHg	Reference	Methods
Flux and yield from site surface water to adjacent water body (slough/bay/estuary)									
Suisun Marsh (SM), California, USA	21,000	0.5 – 24	variable					*DWR, 2001*	
Pond 17, Grizzly Is (SM)	127				3.95×10^{-3}		3.11×10^{-9}	*Stephenson et al., 2008*	once a month collection of MeHg
Nurse Slough (SM)				5.7×10^{-2} (sink)				*Heim et al., 2008*	measured concentrations over full tidal cyle (25 hours), water velocity and depth measured using Acoustic Doppler Current Profiler
Boyton Slough (SM)				1.6×10^{-3} (source)				*Heim et al., 2008*	
Blacklock (SM), CA	28				Pre-breach 2.2×10^{-6}* *Post-breach* 3.9×10^{-6}*		*Pre-breach* 7.8×10^{-8} Post-breach 1.4×10^{-7}	*Bachand et al., 2012*	five sampling events pre and post breach, mass-balance
Brown's Island, California, USA	275	<2	spring: 1800 neap: 200	*Unfiltered* Winter/Fall 2.3×10^{-3}(sink) Spring 3.3×10^{-3}**	*Unfiltered* Fall 1.2×10^{-3} Winter 3.4×10^{-3} Spring 1.6×10^{-3}		*Total* Fall 4.6×10^{-9} Winter 6.0×10^{-9} Spring 9.9×10^{-9}‡	*Bergamaschi et al., 2011/ 2012*	used continuous flow-through fluorescent DOM (FDOM) fluorometer
					Filtered Fall 6.6×10^{-4} (sink) Winter 2.3×10^{-4} Spring 1.1×10^{-3}**			*Bergamaschi et al., 2011/ 2012*	used continuous flow-through fluorescent DOM (FDOM) fluorometer

Location	Area (ha)	Salinity (psu)	Tidal Range (cm)	Daily flux ($g\ d^{-1}$) THg	Daily flux ($g\ d^{-1}$) MeHg	Yield ($g\ m^{-2}\ d^{-1}$) THg	Yield ($g\ m^{-2}\ d^{-1}$) MeHg	Reference	Methods
Gunboat Island, FL		April 31 Sept 9	100	April 0.17 Sept *1.5*	April 0.01 Sept 0.17	7.67×10^{-8}‡	8.49×10^{-9}‡	*Bergamaschi et al., 2012*	used continuous flow-through fluorescent DOM (FDOM) fluorometer
Barn Island salt marsh, CT	140		80	*Particulate* 0.0048 *Filtered* 0.018 *Total MeHg* (sediment to water) Total MeHg 0.11 *			*Particulate* 3.45×10^{-9} (sink) *Filtered* 1.32×10^{-8} *Total MeHg* 8.84×10^{-9} (sediment to water) Total MeHg 8.41×10^{-8}	*Langer et al., 2001*	calculated using concentration data and hydrologic data
Kirkpatrick salt marsh, MD	3ha or 19ha marsh	*Spring* 0 *Fall* 18	30	*Unfiltered* 6.2×10^{-3} (sink) *Filtered* 1.3×10^{-3}	*Unfiltered* 9.0×10^{-5} (sink) *Filtered* 1.1×10^{-4}	*Unfiltered* 2.0×10^{-7} (sink) *Filtered* 4.0×10^{-8}	*Unfiltered* 2.9×10^{-9} (sink) *Filtered* 3.6×10^{-9}	*Mitchell et al., 2012*	mass balance
From pore water to surface water									
Wetlands in San Pablo Bay, CA	45	25-32	frequently inundated		(from vegetated sediment to surface water) 2.05×10^{-4}		4.1×10^{-9} to 4.1×10^{-8}‡	*Best et al., 2007*	Fick's law, concentration gradient of pore/sediment MeHg concentration and surface water, diffusive flux estimate calculation
Tidal wetlands and marsh, in San Pablo Bay and upstream site on Petaluma River, CA	HAAF 203 CC 45	7-32					2.77×10^{-11} Tidal wetlands net flux zero‡	*Best et al., 2009*	Fick's law, concentration gradient of pore/sediment MeHg concentration and surface water, diffusive flux estimate calculation

Location	Area (ha)	Salinity (psu)	Tidal Range (cm)	Daily flux (g d^{-1})		Yield (g m^{-2} d^{-1})		Reference	Methods
				THg	MeHg	THg	MeHg		
San Francisco Bay including tributaries and open water sites		near zero		-26 (for entire Delta region)	-1.2 (for entire Delta)	*Estimate* $2.6x10^{-10}$ to $4.41x10^{-8}$ *Benthic chamber* $3.8x10^{-7}$ to $5.21x 10^{-7}$	*Estimate* $9.48x10^{-10}$ (sink) to $948x10^{-8}$ (source) *Benthic chamber* $1.98x10^{-8}$ (sink) to $1.8x10^{-7}$ (source)	*Choe et al., 2004*	Fick's law, concentration gradient of pore/sediment MeHg concentration and surface water, diffusive flux estimate calculation and also in-situ measurements using benthic chambers
Tidal creeks and mudflats in Mugu Lagoon, CA	600			*Hg(II) 1.3x10-7 (sink) to 2.1x10-5 (source)*	2.7x10-8 (sink) to 4.4x10-6 (source)	*Hg(II)* Tidal creeks $3.1x10^{-9}$ to $1.5x10^{-8}$ Mudflats $4.0x10^{-10}$ (sink) to $1.9x10^{-9}$ (sink)	Tidal creeks $2.0x10^{-9}$ Mudflats $8.8x10^{-10}$	*Rothenberg et al., 2008*	Fick's law, concentration gradient of pore/sediment MeHg concentration and surface water, diffusive flux estimate calculation
Various sites, Florida Everglades, FL					*Unfiltered* $1.7x10^{-8}$ to $5.8x10^{-7}$ (sediment to surface water)			*Li et al., 2012*	mass balance
Sediment shelf, Chesapeake Bay, MD		4.3-34.9					2.16x10-10	*Hollweg et al., 2009*	Fick's law, concentration gradient of pore/sediment MeHg concentration and surface water, diffusive flux estimate calculation

Location	Area (ha)	Salinity (psu)	Tidal Range (cm)	Daily flux ($g\ d^{-1}$)		Yield ($g\ m^{-2}\ d^{-1}$)		Reference	Methods
				THg	MeHg	THg	MeHg		
Tivoli South Bay, a tidal mudflat, NY	113		120			*Historical THg deposition in sediment 1930's* 5.48×10^{-6} *1960's* 8.2×10^{-6} *1970's* 2.2×10^{-6}‡		*Zelewski et al., 2001*	deposition rates calculated fro m ^{137}Cs profiles
Fresh water, from site to adjacent waterbody									
Twitchell Island								*Sassone et al., 2008*	mass balance
East Pond	2.7			2.0×10^{-4}			7.6×10^{-9}‡		
West Pond	2.7			4.0×10^{-4}			1.5×10^{-8}‡		
Cooper's Marsh, Ontario	218						1.6×10^{-9} (sink) to 1.0×10^{-8} (source)	*Holmes and Lean, 2006*	Fick's law, concentration gradient of pore/sediment MeHg concentration and surface water, diffusive flux estimate calculation
Various rivers in Wisconsin							1.2×10^{-8} to 1.95×10^{-7}	*Babiarz et al., 1998*	calculation using flow, concentration, and area of site
Yolo Bypass wildlife area, CA								*Bachand et al., 2012b*	
Wild rice fields, summer							5.48x10-10 to 2.74x10-9		mass balance
Wild rice fields, winter							2.4x10-10 to 2.4x10-9		mass balance
San Francisco Delta, CA								*Heim et al., 2009*	
Corn field							5.4x10-10 to 6.85x10-9		

Location	Area (ha)	Salinity (psu)	Tidal Range (cm)	Daily flux ($g\ d^{-1}$)		Yield ($g\ m^{-2}\ d^{-1}$)		Reference	Methods
				THg	MeHg	THg	MeHg		
Impounded wetlands							1.37x10-9 to 1.37x10-8	*Heim et al., 2009*	
Tomato fields							8.21x10-10 to 2.74x10-9		
Rice fields							1.09x10-10 (sink) to5.48x10-11		
Subsided delta islands							5.4x10-10 (sink) to 5.4x10-10		
From Driscoll 1995									
Watershed, Southern Sweden						6.3x10-9 to 9.58x10-9	3.29x10-10	*Lee and Hultberg 1990*	
Watershed, Northern Sweden						3.27x10-9 to 4.93x10-9	2.2x10-10 to 4.38x10-10	*Lee et al., 1995*	
Watershed, WI						1.37x10-9	1.64x10-10 to 4.11x 10-10	*Krabbehhoft et al., 1995*	
Upland, Ontario						8.2x10-10 to 6.3x10-9	1.91x10-11 to 2.68x10-11	*St. Louis 1994*	
Wetland, Ontario							4.9x10-10 to 1.5x10-9	*St. Louis 1994*	
Adirondack wetland, NY						6.03x10-9	4.66x10-10	*Driscoll et al., 1995*	

*calculated by multiplying yield by area

†calculated by dividing flux by area

‡original data per year, converted to per day

**converted from per season to day

Table 6
Methylation rates and summary of how rates were derived.

Location	Methylation Rates	*Reference*	Method
K_{meth}			
Salt marsh and open water sites in San Pablo Bay, CA	Marsh 0.014 Open water <0.0003	*Marvin-DiPasquale, et al 2003*	measured using radio-tracer derived 203Hg(II) methylation rate constant and in-situ Hg(II)R measurement.
Tidal marshes, rice fields and seasonal wetlands around San Francisco Bay Delta, CA	Tidal marshes 0.002-0.122 Non veg 0.005-0.03 Rice fields 0.015-0.130 Seasonal 0.075 Flood plain 0.027 Non veg 0.001	*Windham-Myers, L., et al 2009*	measured using radio-tracer derived 203Hg(II) methylation rate constant and in-situ Hg(II)R measurement.
Wetlands along San Pablo Bay, CA	*non vegetated* HAAF 0.019 CC 0.077 *S.foliosa* HAAF 0.018 CC 0.022 *S. virginica* HAAF 0.029 CC 0.022	*Best et al., 2005*	measured using isotope derived 199Hg methylation rate constant]
Tidal wetlands along Petaluma River, CA	0.02 to 0.18	*Yee, D.,et al., 2008*	measured using radio-tracer derived 203Hg methylation rate constant and in-situ Hg(II)R measurement.
Wetlands along San Pablo Bay and upstream Petaluma River, CA	HAAF 0.7 CC 1.3 PR 3.3	*Best, et al 2009*	measured using DGT strips for in-situ MeHg in sediment porewater; used isotope 199Hg and 202Hg to measure methylation
Kirkpatrick Marsh, MD	0.002 to 0.07	*Mitchell, C.P.J. and C.C. Gilmour, 2008.*	measured using isotope derived 201Hg methylation rate constant]
Sediment shelf, Chesapeake Bay, MD	0.007 – 0.08	*Hollweg et al., 2009*	measured using isotope derived 201Hg methylation rate constant
Salt marsh, CT	2.37-241.5*	*Langer et al., 2001*	
Salt marsh, CT	0.001 – 0.2	*Langer et al., 2001*	measured using radio-tracer derived 203Hg(II) methylation rate constant and in-situ Hg(II)R measurement.
Wetlands along Everglades, FL	Soil 0.03 Floc 0.03 Periphyton 0.01	*Li, Y., et al., 2012*	measured using isotope derived 199Hg methylation rate constant

Location	Methylation Rates	***Reference***	Method
Long Island Sound	0.01-0.08	*Hammerschmidt and Fitzgerald, 2004*	
Potential/Estimated			
Salt marsh and open water sites in San Pablo Bay, CA	*(potential methylation rate) Marsh* methylation 3.1 Demetylation 3.1 *Open water* methyltaion <0.06 demethylation 1.9 to 2.8	*Marvin-DiPasquale, et al 2003*	measured from Kmeth values and in individually measured Hg(II)R
	(in situ methylation rate) Marsh Methylation 0.5 *Open Water Demethylation 0.02-0.04*	*Marvin-DiPasquale, et al 2003*	
Wetlands along Everglades, FL	(potential methylation rate) Soil 4.52 Floc 3.04 Periphyton 0.54	*Li, Y., et al., 2012*	
San Francisco Bay-Delta areas	potential methylation rate (pg/g/d) Tidal marshes 9-83 Non veg 3-7 Rice fields 66-105 Seasonal 8 Flood plain 39 Non veg 1	*Windham-Myers, L., et al 2009*	measured from Kmeth values and in individually measured Hg(II)R

Table 7
Mercury emission from plants (adopted from Lindberg et al., 2002).

Location	Season	Emission (ng m^{-2} h^{-1})	Reference
Everglades Nutrient Removal Project Site, FL			*Lindberg et al., 2002*
Cell 2	Spring	38	
	Summer	87	
Cell 3	Winter	16.9	
Buffer Cell	Fall	49	
WCA 2a	Summer	17	
Deciduous forest	Summer, Fall	37	*Lindberg et al., 1998*
Salt marsh, *Spartina*	Spring, Summer	3.6	*Lee et al., 2000*
Phragmites communis	Summer	90	*Kozuchovski and Johnson, 1978*
Wetland canopy	Winter, Summer	30	*Marsik, 2002*

Table 8
List of drivers affected by tidal conversion of wetland (left most column) and resulting effects on various factors (top row).

Summary	Redox	DOC	BOD	Flow Rates	Salinity	Tidal Prism
Hydrology						
Source Water						
Tide Frequency	Increased spatial gradient; from frequently to infrequently saturated conditions	Marsh discharges more similar to source waters due to increased tidal flows			Marsh discharges more similar to source waters due to increased tidal flows	
Tidal Exchange Magnitude	Increased spatial gradient; from frequently to infrequently saturated conditions	Marsh discharges more similar to source waters due to increased tidal flows	Lesser; More saturated conditions and less pulsing of higher BOD organic material due to less seasonal variations	Greater	Marsh discharges more similar to source waters due to increased tidal flows	Greater, estimated order of magnitude
Flow		Marsh discharges more similar to source waters due to increased tidal flows	Lesser; More saturated conditions and less pulsing of higher BOD organic material	Greater	Marsh discharges more similar to source waters due to increased tidal flows	Greater, estimated order of magnitude
Water Depth	Increased spatial gradient; from frequently to infrequently saturated conditions	More frequently inundated marsh plain could lead to lower DOC due to slower decomposition in anaerobic conditions.	same as DOC			
Inundation Frequency	Increased spatial gradient; from frequently to infrequently saturated conditions	see comment under depth	same as DOC			

Summary	Redox	DOC	BOD	Flow Rates	Salinity	Tidal Prism
Marsh Areas						
Footprint	Increased spatial gradient; from frequently to infrequently saturated conditions					
Vegetation Management						
		Reduced DOC due to less decomposition of mowed vegetation from habitat management	Reduced BOD discharges due to shift from agr to managed wetland conditions			

Table 9
Comparison table of tidal vs. non-tidal systems.

		Tidal	Muted Tidal/Managed Wetlands
Summary characteristic		Full tidal Exchange in which water flows are governed by tidal gradients and slough capacities	Water levels seasonally managed for different goals (e.g. agriculture, hunting) and thus potentially exposed to seasonal wet and dry periods. Daily flows constrained by gate structure capacity and management.
Hydrology			
Source Water		Primarily from tidal exchange. Seasonal inflow from storms.	Primarily from tidal exchanges. Other inputs (e.g. storms, wastewater discharges) can be relatively higher and more important than for tidal systems due to more limited tidal exchange.
Tide Frequency		2 tides per day, synchronous with source waters.	Dependent on goal of management (e.g. hydrology/habitat). More "pulse-like" flows possible. If gates open, muted tidal exchange is possible (2 tides a day, but with decreased range).
Tidal Exchange Magnitude		Magnitude similar to source tidal waters.	Magnitude reduced (muted).
	Mean	Full tidal exchange. Prism primarily depends upon bathymetry.	Fraction of total water in system. Prism primarily depends upon gate management. Mean exchange estimated as an order of magnitude less than for full tidal systems
	Driver	External tidal range	Gate management
	Standard Deviation	Wide Standard Deviation driven by variations in the tide	Narrow standard deviation with slight affect from outside tidal variations
Flow			
	Flow Direction	Tides ebb and flow through marsh system relying upon established and legacy channels. Likely bi-directional within marsh.	Flows can be controlled through gate structures to enable bi-directional or uni-directional flow, depending upon adjacent sloughs and land configurations
	Rates	Rate depends upon tide gradients across marsh and between sloughs and marsh, and upon slough capacity	Gate structures restrict flows; control flow rates.
Water Depth			
	Mean	Higher mean water depth	Depends on management goal. May be higher but if gates are open, mean depth will generally be lower.
	Driver	Outside tidal range	Gate management and seasonal goals
	Standard Deviation	Wide standard deviation throughout the day, twice per day.	Narrow standard deviation with slight affect from outside tidal variations
	Maximum	Driven by high tides, varies daily and seasonally	Depends on management goal. If gates open, exchange is restricted resulting in muted tidal regime.
	Minimum	Driven by low tides, varies daily and seasonally	Depends on management goal. If gates open, exchange is restricted resulting in muted tidal regime.
Inundation Frequency			
	Daily	Gradient inundation frequency decreasing from low to high marsh.	Large areas of marsh constantly inundated during periods managed as marsh. Inundation gradient changes across narrow elevation range near mean water levels. High marsh never inundated.

		Tidal	Muted Tidal/Managed Wetlands
	Seasonal	Governed by seasonal tidal cycles.	Management for vegetation results in long periods of time during which marsh either inundated or exposed.
Marsh Areas			
Footprint		For same marsh region, greater magnitude in water levels will lead to occasional inundation of high marsh, resulting in greater effective marsh footprint.	Wetted footprint determined primarily by gate management and seasonally dependent. Foot print is smaller if management goal maintains water at lower/muted levels, resulting in high marsh never being inundated.
Vegetation Management			
		Limited. May manage for endangered or invasive species.	Seasonal management for different goals (e.g. agriculture, ducks)