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Biogeochemical Processes in an Urban, Restored Wetland of San Francisco Bay, California, 2007–2009: Methods and Data for Plant, Sediment, and Water Parameters

By Lisamarie Windham-Myers, Mark C. Marvin-DiPasquale, Jennifer L. Agee, Le H. Kieu, Evangelos Kakouros, Li Erikson, and Kristen Ward

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Cover: At the foot of San Francisco's historic Presidio, the restored Crissy marsh comprises subtidal, intertidal, and upland habitats. Jennifer Agee kneels in high intertidal habitat dominated by pickleweed (*Sarcocornia pacifica*). Photo by L. Windham-Myers.

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Conversion Factors

SI to Inch/Pound

Multiply	By	To obtain
Length		
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
Area		
square meter (m^2)	0.0002471	acre
square kilometer (km^2)	247.1	Acre
square centimeter (cm^2)	0.001076	square foot (ft^2)
square meter (m^2)	10.76	square foot (ft^2)
square centimeter (cm^2)	0.1550	square inch (ft^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
Volume		
liter (L)	0.2642	gallon (gal)
cubic meter (m^3)	264.2	gallon (gal)
cubic centimeter (cm^3)	0.06102	cubic inch (in^3)
cubic meter (m^3)	0.0008107	acre-foot (acre-ft)
Mass		
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound avoirdupois (lb)
Density		
gram per cubic centimeter (g/cm^3)	62.4220	pound per cubic foot (lb/ft^3)
Flow Rate		
milliliters per minute (mL/min)	0.06102	cubic inches per minute (in^3/min)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

°F=(1.8×°C)+32

Horizontal coordinate information is referenced to the insert datum name (and abbreviation) here, for instance, “North American Datum of 1983 (NAD 83)”

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L), micrograms per liter (µg/L), or nanograms per liter (ng/L). Concentrations for some chemicals are given as molarity (*M*) which is defined as moles per liter, where a mole is 6.022×10^{23} atoms of a pure compound or element. Multiply concentrations expressed as millimolar (m*M*) by 10^{-3} to obtain moles per liter, and multiply concentrations expressed as micromolar (µ*M*) by 10^{-6} to obtain moles per liter.

Biogeochemical Processes in an Urban, Restored Wetland of San Francisco Bay, California, 2007–2009: Methods and Data for Plant, Sediment, and Water Parameters

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Abstract

The restoration of 18 acres of historic tidal marsh at Crissy Field has had great success in terms of public outreach and visibility, but less success in terms of revegetated marsh sustainability. Native cordgrass (*Spartina foliosa*) has experienced dieback and has failed to recolonize following extended flooding events during unintended periodic closures of its inlet channel, which inhibits daily tidal flushing. We examined the biogeochemical impacts of these impoundment events on plant physiology and on sulfur and mercury chemistry to help the National Park Service land managers determine the relative influence of these inlet closures on marsh function. In this comparative study, we examined key pools of sulfur, mercury, and carbon compounds both during and between closure events. Further, we estimated the net hydrodynamic flux of methylmercury and total mercury to and from the marsh during a 24-hour diurnal cycle. This report documents the methods used and the data generated during the study.

Background

After decades of loss and neglect of wetlands, restoration efforts are occurring rapidly throughout the San Francisco Bay, the Sacramento-San Joaquin Delta, and elsewhere nationally. In San Francisco Bay (hereafter SF Bay), 90 percent of historic tidal wetlands have been separated from tidal connectivity with the bay. Less than 8 percent of historic native habitat exists in the coastal environment, and the primary cause of species endangerment is habitat loss (Monroe and others, 1999).

One of many restoration projects in SF Bay is the tidal marsh (hereafter Crissy marsh) at Crissy Field. As with many other restoration projects, competing land use restricted the size of the approved footprint. To promote a stable regime of erosional and depositional forces likely to maintain a continuous tidal connection to the Bay, hydrologic engineers had recommended a 30-acre tidal footprint, but only 18 acres were available for restoration. Therefore, to counteract depositional forces, the tidal prism was maximized by grading 80 percent of the acreage to subtidal elevations. Despite this initial optimization, rapid sand migration and deposition has further reduced the tidal prism by nearly 50 percent. Although the reduction in tidal prism was expected, it occurred much more rapidly than predicted, prompting a follow-up analysis of the relation between tidal prism and inlet function. Engineering calculations, based on post-restoration monitoring data collected from 2001–2002, confirmed pre-project modeling results and suggest that the marsh

would need a tidal prism of 47 acre-feet (28-acre footprint at mean higher high water) to naturally maintain the tidal inlet (PWA, unpub. data, 2004[1]).

Since 1999, hydrologic connectivity between the marsh and SF Bay has not been consistently maintained by natural processes. The inlet closed within 18 months of initial tidal restoration in 1999 and has closed at least twice per year since then. The system is flood-tide dominated, and inlet closures are primarily due to sand deposition on the beach and in the tidal marsh near the inlet entrance, coupled with insufficient sediment transport out of the channel on ebb tides. Closure periods begin with continued tidal input of water over the top of the beach barrier on flood tides, but limited or no tidal output on ebb tides. As the inlet approaches a complete closure in tidal exchange, the water level in the marsh remains near high-tide elevations and tidal flushing ceases (fig. 1). With consequent tides, the volume of water in the marsh rises, increasing the hydrostatic pressure on the sediment in the channel. This provides an opportunity for self-regulation upon a spring tide, when the impounded marsh water may be capable of forcing out the sediment barrier in the channel. On occasion, however, the pressure is not great enough, and the impoundment persists for greater than a 2- to 3-week period. National Park Service (NPS) policy is to mechanically dredge the channel to restore connection between the marsh and SF Bay, when it becomes clear that it is unlikely to re-open naturally (generally after 3–4 weeks).

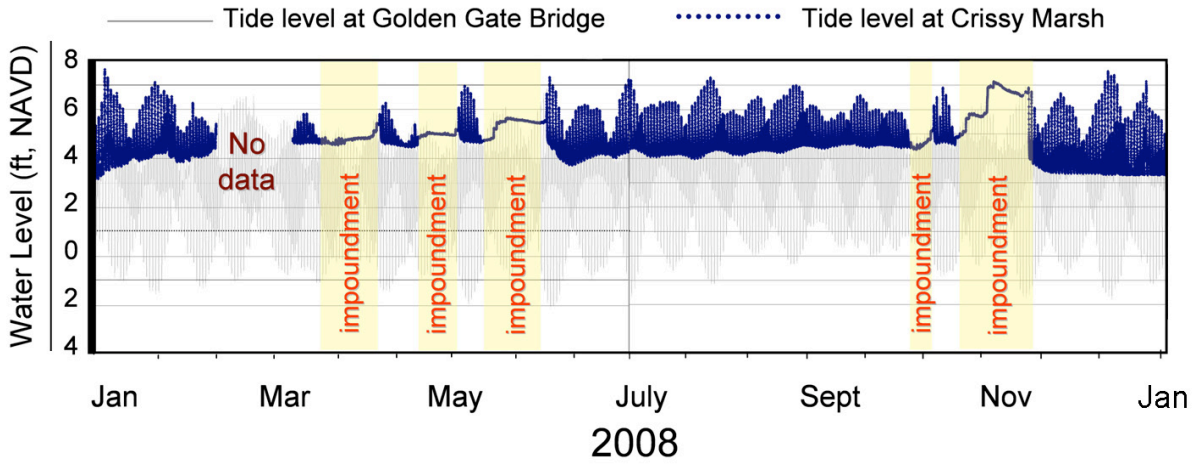

Figure 1. Calendar Year 2008 record of water level at tidal inlet in Crissy marsh vs. San Francisco Bay, California.

[Water levels represent surface elevation of water under Golden Gate Bridge (gray) (NOAA/NOS station 9414290) and surface elevation of water in tidal inlet under footbridge (blue). Data collected by pressure transducer and datalogger, retrieved and maintained monthly by K. Ward, Golden Gate National Recreation Area, NPS, and corrected to NAVD 88 elevations]

[1] Crissy Field marsh expansion study, final report, 2004: San Francisco, California, Philip Williams and Associates, (PWA), Ref# 1623, 70 p.. Available at: *http://library.presidio.gov/archive/documents/CrissyField_Exp_Chpt_1%20to%203_accesible.pdf*

Surveys of vegetation at Crissy marsh coincident with the recent hydrologic record suggest that the closure events are responsible for vegetation dieback in habitats dominated by cordgrass (*Spartina foliosa*) (Ward and Ablog, unpub. data, 2006[2]). Loss of cordgrass populations was most pronounced in early summer 2005, after an extended closure period. Since then, cordgrass has not extensively recolonized this lower zone. Flooding damage was observed visually in summer 2005 in cordgrass cross sections, but it remained unclear whether this damage was a function of the flooding level, flooding duration, higher temperatures, or the related effect of sulfide toxicity (Crawford and Braendle, 1996). As a result of these multiple interacting stressors, the lack of cordgrass recovery may be due to a change in physical conditions (erosion; Callaway, unpub. data, 2008[3]), biogeochemical conditions, (sulfide toxicity and oxygen stress; Koch and others, 1990), or herbivory by coots (*Fulica americana*) or other waterfowl (Ward and Ablog, unpub. data, 2006).

Sulfur chemistry plays a dominant role in the energy and carbon flow of salt marshes, because microbial sulfate reduction is a dominant respiratory process in these systems (Howarth, 1984). The respiratory end product of microbial sulfate reduction is sulfide. At high concentrations, sulfide can be toxic to wetland plants (Koch and others, 1990). Thus, a primary focus of this study is the linkage between closure events and their effect (if any) on sulfur biogeochemistry and how this linkage may affect cordgrass recolonization in Crissy marsh.

Wetlands are also known to be important areas of toxic methylmercury (MeHg) production (Gilmour and others, 1998; Marvin-DiPasquale and Agee, 2003), a process facilitated by some sulfate-reducing bacteria (Compeau and Bartha, 1985), as well as some iron-reducing bacteria (Fleming and others, 2008; Kerin and others, 2008), in anoxic sediment. Due to mercury (Hg) contamination in SF Bay associated with historic mining and industrial activity, concerns have been raised regarding the potential for enhanced MeHg production and bioaccumulation associated with extensive wetland restoration efforts in the region (Davis and others, 2003; Beutel and Abu-Saba, unpub. data, 2004[4]). Subsequently, a second key focus of this study was examining the linkage between closure events at Crissy marsh and their effect on MeHg production and the biogeochemical cycling of Hg.

The Golden Gate National Recreation Area and its partner agencies (Presidio Trust, Golden Gate National Parks Conservancy) have been cooperating to assess the possibility of future marsh expansion; however, the area available for marsh expansion is still limited. Any plans for the expansion of Crissy marsh to a larger footprint require additional data on hydrologic and biogeochemical processes to optimize the marsh design for vegetated marsh habitat, while also minimizing MeHg production and export. We examined key biogeochemical processes of sulfur, carbon, and Hg occurring throughout subhabitats of Crissy marsh during and between closure events. This report details the methods used and the resulting data for sediment, water, and plant-sampling components of the study.

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[2] Crissy Field restoration project; final report, summary of monitoring data 2000–2004: National Park Service Golden Gate National Recreation Area, 103 p. Available on request from Kristen Ward, National Park Service, Fort Mason, Building 201, San Francisco, CA 94123

[3] Wetland sediment dynamics at Crissy Field marsh: Final annual report to the National Park Service, February 8, 2008, 19 p. Available on request from Kristen Ward, National Park Service, Fort Mason, Building 201, San Francisco, CA 94123

[4] Beutel, M., and Abu-Saba, K., 2004, South Bay Salt Ponds Restoration Project; mercury technical memorandum: Brown and Caldwell and Larry Walker and Associates report prepared for the South Bay Salt Ponds Restoration Project Management Team, 47 p. Available at: *http://www.southbayrestoration.org/pdf_files/Final BC Mercury Technical Memo Aug 4 2004.pdf*.

Site Description

Crissy marsh is located in Golden Gate National Recreation Area at the waterfront base of the historic Presidio of San Francisco, just southeast of Fort Point National Historic Site, at the base of the Golden Gate Bridge (fig. 2). Prior to its tidal restoration in 1999, the historic wetland had been reclaimed by diking and filling beginning in the late 1800s by the U.S. Army and culminated with filling of the remaining wetlands for use as the site of the 1915 Panama-Pacific International Exposition.

Crissy marsh comprises four major subhabitats within its 18-acre footprint—subtidal, low intertidal (cordgrass dominant), mid to high intertidal (pickleweed dominant), and upland transitional (coastal scrub). Approximately 80 percent of the acreage is at subtidal elevations below mean low water (MLW), and 15 percent of the acreage is outside of the tidal frame above mean higher high water (MHHW), leaving only ~5 percent of the acreage within the intertidal elevation zone (between MLW and MHHW). Data collection focused only on tidally influenced habitats along an elevation gradient—subtidal, low intertidal, and high intertidal—determined by vegetation classification.

Figure 2. California state map indicating the location of Crissy marsh (star) within San Francisco Bay, California.

To address substrate variability within the marsh, eight stations (A–H) were chosen to anchor transects across three general elevation categories—subtidal (<2 ft North American Vertical Datum of 1988 or NAVD 88), low intertidal (~2–4 ft NAVD 88), and high intertidal (~4–6 ft NAVD 88). These stations, shown in figure 3, were selected to represent a range of sediment textures and organic-matter concentrations and corresponded, where possible, with NPS stations for water quality analysis. Table 1 lists the U.S. Geological Survey (USGS) and NPS station codes, horizontal coordinates, and sediment sampling depth interval for each sampling location. Sediment and pore-water samples were collected at two depth intervals below the sediment surface, 0–2 cm (surface) and 8–10 cm (deep). In many locations throughout the marsh, a compacted sediment layer at depth (~20–40 cm) impeded sediment collection below.

Surface water chemistry was assessed temporally by monthly collections from February 2007 through October 2008 at a single central location (site WQ4) and spatially by a single synoptic sampling of all locations in March 2007. Table 2 lists the locations, dates, and tidal status of all sampling events, as well as the surface-water measurements made for each sampling event. To assess the speciation and exchange of aqueous Hg between Crissy marsh and San Francisco Bay, a diurnal (24-hour) hydrologic flux study was conducted, with sampling focused on the tidal inlet (site WQ9) and two end-member sources: an upland spring-fed creek, Thompson Reach of the

Tennessee Hollow Drainage (THD), and a well-mixed central location, site WQ4. Velocity was measured in all three locations with a Global Water velocity meter (sensitivity >0.5 cm/s), but velocities and fluxes were ultimately modeled and estimated with a USGS model of water flows based on current and historic field data (L. Erikson, unpub. data, 2008).

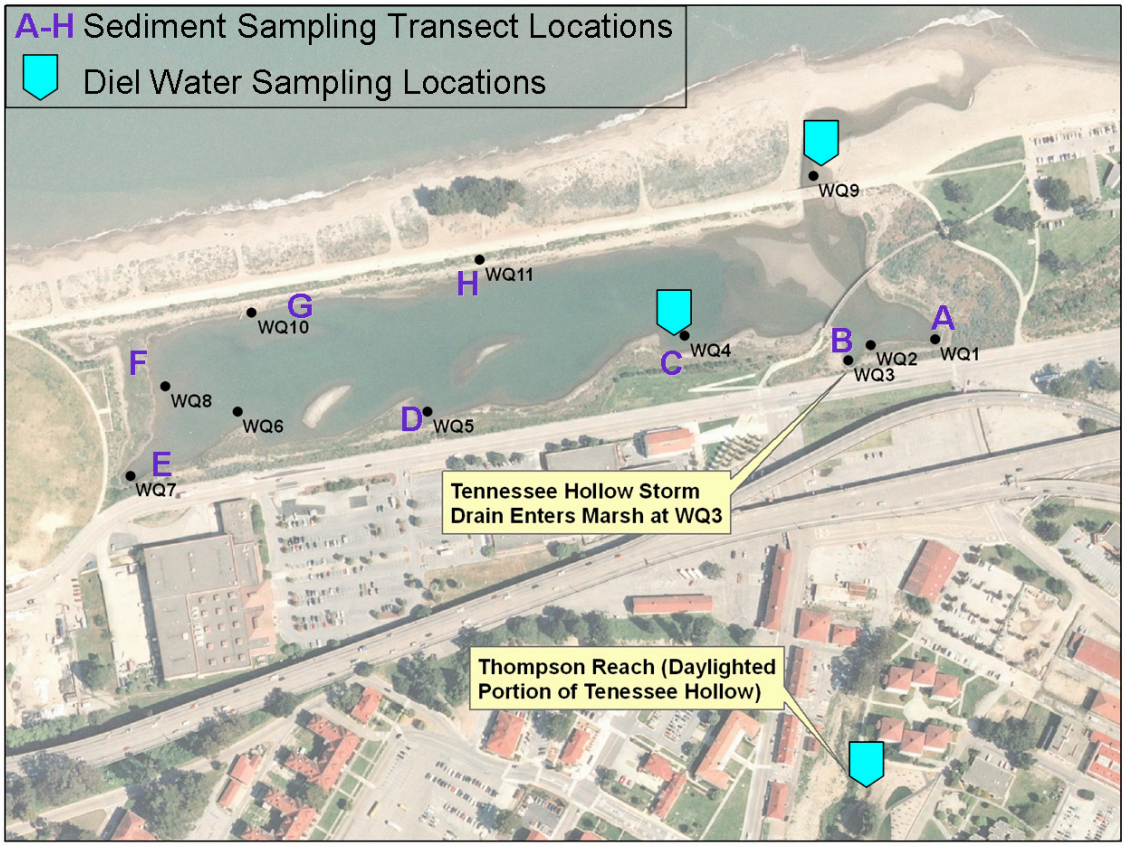

Figure 3. Map of study sampling locations and National Park Service water quality monitoring stations (WQ#), Crissy marsh, San Francisco Bay, California.

Table 1. Sediment sampling locations and codes, Crissy marsh, San Francisco Bay, California. Sampling dates from February 2007–October 2008.

[Latitude (northing) and longitude (westing) are given in degrees decimal minutes (DD MM.MMM) in North American Datum of 1983 (NAD 83); coordinates refer to midpoints on transect, thus, low-intertidal elevations]

USGS station	NPS water quality station	Subhabitat	Sediment depth (cm)	Sample code	Northing	Westing
A	WQ1	High Marsh	0-2	A-H-2	37 48.274	122 27.199
	WQ1	High Marsh	8-10	A-H-10		
	WQ1	Low Marsh	0-2	A-L-2		
	WQ1	Low Marsh	8-10	A-L-10		
	WQ1	Subtidal	0-2	A-S-2		
	WQ1	Subtidal	8-10	A-S-10		
B	WQ2	High Marsh	0-2	B-H-2	37 48.265	122 27.235
	WQ2	High Marsh	8-10	B-H-10		
	WQ2	Low Marsh	0-2	B-L-2		
	WQ2	Low Marsh	8-10	B-L-10		
	WQ2	Subtidal	0-2	B-S-2		
	WQ2	Subtidal	8-10	B-S-10		
C	WQ4	High Marsh	0-2	C-H-2	37 48.271	122 27.334
	WQ4	High Marsh	8-10	C-H-10		
	WQ4	Low Marsh	0-2	C-L-2		
	WQ4	Low Marsh	8-10	C-L-10		
	WQ4	Subtidal	0-2	C-S-2		
	WQ4	Subtidal	8-10	C-S-10		
D	WQ5	High Marsh	0-2	D-H-2	37 48.234	122 27.529
	WQ5	High Marsh	8-10	D-H-10		
	WQ5	Low Marsh	0-2	D-L-2		
	WQ5	Low Marsh	8-10	D-L-10		
	WQ5	Subtidal	0-2	D-S-2		
	WQ5	Subtidal	8-10	D-S-10		
E	WQ7	High Marsh	0-2	E-H-2	37 48.216	122 27.597
	WQ7	High Marsh	8-10	E-H-10		
	WQ7	Low Marsh	0-2	E-L-2		
	WQ7	Low Marsh	8-10	E-L-10		
	WQ7	Subtidal	0-2	E-S-2		
	WQ7	Subtidal	8-10	E-S-10		
F	WQ8	High Marsh	0-2	F-H-2	37 48.261	122 27.588
	WQ8	High Marsh	8-10	F-H-10		
	WQ8	Low Marsh	0-2	F-L-2		
	WQ8	Low Marsh	8-10	F-L-10		
	WQ8	Subtidal	0-2	F-S-2		
	WQ8	Subtidal	8-10	F-S-10		
G	WQ10	High Marsh	0-2	G-H-2	37 48.289	122 27.514
	WQ10	High Marsh	8-10	G-H-10		
	WQ10	Low Marsh	0-2	G-L-2		
	WQ10	Low Marsh	8-10	G-L-10		
	WQ10	Subtidal	0-2	G-S-2		
	WQ10	Subtidal	8-10	G-S-10		
H	WQ11	High Marsh	0-2	H-H-2	37 48.298	122 27.442
	WQ11	High Marsh	8-10	H-H-10		
	WQ11	Low Marsh	0-2	H-L-2		
	WQ11	Low Marsh	8-10	H-L-10		
	WQ11	Subtidal	0-2	H-S-2		
	WQ11	Subtidal	8-10	H-S-10		

Table 2. Surface water sampling locations, schedule, and data collected from Crissy marsh, San Francisco Bay, California.

[All surface-water samples collected at 10 cm depth. Chl-a, chlorophyll-a; F-MeHg, filtered methylmercury; F-THg, filtered total mercury; pH, acidity; Pha, phaeophytin; POC, particulate organic carbon; Sal, salinity; SC, specific conductivity; Temp, temperature; U-MeHg, unfiltered methylmercury; U-THg, unfiltered total mercury]

USGS station	Water quality station	Date	Status	pH	Temp	Sal	SC	TSS	Chl-a	Pha	POC	U-THg	U-MeHg	F-THg	F-MeHg
C	WQ4	1-Feb-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	28-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	5-Apr-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	9-May-07	Impounded	X	X	X	X	X	X	X	X	X	X		
C	WQ4	6-Jun-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	17-Jul-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	15-Aug-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	27-Sep-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	11-Oct-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	21-Nov-07	Impounded	X	X	X	X	X	X	X	X	X	X		
C	WQ4	17-Dec-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	13-Feb-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	26-Mar-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	4-Apr-08	Impounded	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	11-Apr-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	18-Apr-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	2-May-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	22-May-08	Impounded	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	30-May-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	27-Jun-08	Impounded	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	3-Jul-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	5-Aug-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
C	WQ4	10-Oct-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
A	WQ1	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
B	WQ2	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
B2	WQ3	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
C	WQ4	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
D	WQ5	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
E	WQ7	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
F	WQ8	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
Inlet	WQ9	8-Mar-07	Slack Ebb	X	X	X	X	X	X	X	X	X	X		
Inlet	WQ9	5-Aug-08	Slack Ebb	X	X	X	X	X	X	X	X	X	X	X	X
THD		5-Aug-08	Non-Tidal	X	X	X	X	X	X	X	X	X	X	X	X

Methods

Field Sampling

Surface Water

Surface water was sampled monthly beginning in February 2007 through October 2008. Samples were collected from the horizontally well mixed central station WQ4; this station has been found to have consistent tidal mixing among the 11 NPS water quality stations over a 12-hr tidal cycle as indicated by salinity data (Ward and Ablog, unpub., 2006). Field samples were collected within the subtidal zone where water depth ranged from 0.8 to 1 m. With careful attention to minimizing site disturbance, water samples were collected by pole and by technicians in waders, using the "dirty hands/clean hands" technique (Gill and Fitzgerald, 1985). Combusted 1-L amber bottles were submerged 10–15 cm below the water surface, rinsed twice with site water, and then filled and brought to shore for preservation and analysis. A YSI–85 datasonde and Orion 250A pH meter were employed at the time of sampling to measure real-time water temperature, salinity, conductivity, dissolved oxygen, and pH levels at the same depth as the water samples (10–15 cm below the water surface). Samples were returned to the laboratory in a cooler on wet ice. In addition to these monthly samples collected at WQ4, sampling of eight stations was performed during March 2007 in concert with NPS efforts to map water quality patterns within the marsh (NPS, unpub. data, 2007).

Sediment Pore Water

Sediment pore water was sampled with multiple approaches during February–May 2007 to assess the best method for this sandy, shallow sediment. Pore water was collected from both the surface (0–2 cm) and subsurface/root zone (8–10 cm). Samples collected side-by-side with different methods were compared statistically to assess the variability within each method and the differences between methods for sulfate, sulfide, and chloride (as a conservative tracer).

Pore water diffusion samplers (or "peepers", described in Windham-Myers, 2005) were deployed for 10–14 days to allow for complete equilibration as predicted by modeled diffusion rates based on sediment porosity, concentration gradient, and sample-well geometry. Upon retrieval, pore water was sampled immediately from the wells by syringe and preserved for sulfide with a sulfide-antioxidant-buffer (SAOB; EPA, 1996) and for sulfate and chloride by flash-freezing on dry ice. On the same day of retrieval, sediment cores were taken to ~15 cm depth and immediately sectioned into the corresponding sample depths (0–2 and 8–10 cm). These depth intervals were collected from multiple cores, transferred into a combusted glass mason jar until full, and returned to the laboratory on wet ice. Subsequently, the sediment was manually homogenized and sediment-pore water was collected via centrifugation under anoxic conditions and filtration through a 0.45 μM membrane filter in an anoxic glovebag, followed by preservation of pore-water sulfide, sulfate, and chloride (hereafter, PWSU, $PWSO_4$ and PWCl, respectively), as described above.

After two sampling events, it was apparent that the peeper approach was generating unreasonable data for PWSU, $PWSO_4$ and PWCl concentrations, the last of which should have largely tracked overlying water. Rather, peeper-PWSU was spurious between replicate samples and often an order of magnitude higher than centrifuged-PWSU, suggesting a membrane effect. Conversely, peeper-$PWSO_4$ and peeper-PWCl were frequently lower than expected. For example, peeper wells positioned above the sediment in overlying water often had PWCl well below surface water concentrations. The centrifuged-sediment approach is predisposed to sampling macropore environments (Harvey and others, 1995) and was more consistent between samples. Although the

replicate quality-assurance was improved, the whole-sediment centrifugation approach was time-consuming and involved post-collection sample processing as opposed to field collection and preservation. To reduce sampling time, a third approach to pore-water collection was tested against the centrifugation approach.

In May 2007, two types of drivepoint sampling approaches were tested. The first involved a battery-operated peristaltic pump (Geopump) with Masterflex tubing and fritted polycarbonate filter tip packed with combusted glass wool. The second involved a mini-drivepoint sampler with replicate stainless steel tubes, each having a three-way valve, and with samples being pulled by syringe. Both drive-point sampling devices are described in Duff and others (1998) and both were found to provide an acceptable level of reproducibility among replicate samples (field duplicates relative percent difference (RPD) <20 percent) and similar PWCl concentrations to overlying water and centrifuged pore water (RPD <40 percent). Based on failure of the peristaltic pump to draw adequate volumes of pore water in some of the fine-grained sediment, the manual, mini-drivepoint sampler was deployed for all eight transect stations for the remainder of 2007 and throughout 2008. Only these drivepoint sampled data are reported in the remainder of this report. All pore water analyte preservation methods were the same as described above.

Sediment

Sediment sampling began in February 2007, with an emphasis on variability across the elevation transect from high intertidal to low intertidal to near-shore subtidal. In 2007, sediment cores were collected by hand with a polycarbonate coring tube (10 cm diam). Sediment was extruded upward with a rubber plunger, and two discrete depth intervals were collected (0–2 and 8–10 cm) and transferred into combusted glass mason jars where the sediment was homogenized. Subsamples (n=3 for each sample date) were then collected in the field for organic content, grain size, bulk density, porosity, oxidation-reduction potential (ORP), pH, total reduced sulfur (TRS), THg, and MeHg. In 2008, a single surface sediment sample (0–2 cm) was collected with a 2-cm-tall core ring, following the same field-based protocol. Sediment and overlying water temperature was measured in the field in duplicate with calibrated digital thermometers.

Plant Structure and Physiology

Plant structure for cordgrass (*Spartina foliosa*) communities was sampled in low-intertidal locations of all eight transects in July 2007 and July 2008 and consisted of three metrics—plant stem density, maximum plant height, and live root density in the surface 0–10 cm sediment depth interval. All sites were dominated by cordgrass (>75 percent cover) in the intertidal elevations, except for Station F, which was dominated by pickleweed (*Sarcocornia pacifica*), and Station E, which was dominated by alkali bulrush (*Scirpus maritima*). Aboveground metrics were assessed within 1 m of the site marker. Root density was assessed at the m^2 scale based on the average of three below-ground cores (10 cm diam) to 10 cm depth. In March through July 2008, the effect of impoundment events on plant health was assessed with assays of root concentrations of ethanol and acetaldehyde—two sensitive and short-lived fermentative end-products that indicate lack of oxygen in the root zone and represent conditions of low-carbon-use efficiency (Crawford and Braendle, 1996). Root samples were rinsed quickly in the field and transferred to pre-weighed vials placed immediately on dry ice to preserve initial conditions per Li and others (2004).

Continuous Water Level and Water Quality Status

Since 2004, two instruments have been mounted in the thalweg of the drainage channel (at ~1 ft NAVD 88, just above the lowest expected water elevations in the marsh) under the Crissy marsh footbridge to provide a continuous record of environmental conditions at the site: a pressure

transducer to monitor waterlevels and a datasonde equipped with four water quality sensors for salinity (0–70 ppt), conductivity (0–100 mS/cm), dissolved oxygen (D.O.; 0–500 percent of saturation), and temperature (–5°C to 50°C). For continued maintenance since 2004, the pressure transducer (Druck model no. PS9800) and datalogger unit (Aquistar model no. DL-1) are visually inspected and an open-air calibration check is performed on a monthly basis. The datasondes used to record water quality have included a Hydrolab Datasonde (model series 4a) or Minisonde (May 2001–March 2001) and Eureka Manta and Manta2 (from March 2007). The datasonde was removed from the field, cleaned, and calibrated every 3–4 weeks, with D.O membranes (Hydrolab instruments) replaced each time to remove fouling effects. Stratification of channel flow appeared to be minimal, because the flow was commonly turbulent.

Hydrologic Flux

Model Grid, Bathymetry, and General Setup

Tidal flows through the marsh inlet were simulated by employing the FLOW module in the Delft3D numerical model (Roelvink and van Banning, 1994). Spatial extent of the curvilinear grid (198 x 111 grid cells) and bathymetry are shown in figure 4. Cell size ranges from 44.0 m in the offshore region to 2.5 m in the inlet region. The offshore open boundary is forced with spatially varying astronomic tide constituents (water-level amplitude and phase), which were obtained by nesting the domain in a larger calibrated model of the San Francisco Bay system, while the western and eastern open lateral sections are described by Neuman water-level-gradient boundaries (water-level gradients).

Bathymetry outside of the marsh was generated from measurements obtained by using a coastal profiling system and by walking the beach with a pole-mounted real-time kinetics geographical positioning system (RTK-GPS) in September 2008 (Li Erikson, unpub. data, 2008). Marsh and inlet bathymetry was estimated from earlier surveys conducted in 2004 and 2006 (PWA, 2006[5]; Towill, 2004[6]). The thalweg of the marsh inlet is dynamic and shifts from the nearly north-south-trending orientation after mechanical excavation to the east due to accumulation of sediments from longshore transport. The channel configuration employed in these model simulations is an estimate of the conditions present for the time of the simulations.

[5] Crissy Field/ East Beach physical monitoring update—fall 2006: San Francisco, California, Philip Williams and Associates (PWA), Ref. #1386.07, p. 41. Available on request through *http://www.pwa-ltd.com/resources/resource_ftp.html*.

[6] Surveyors report, 2004, Crissy Field benchmark elevation comparisons, available as Towill No. 10748, from Towill, Inc., 5099 Commercial Circle, Suite 100, Concord, CA.

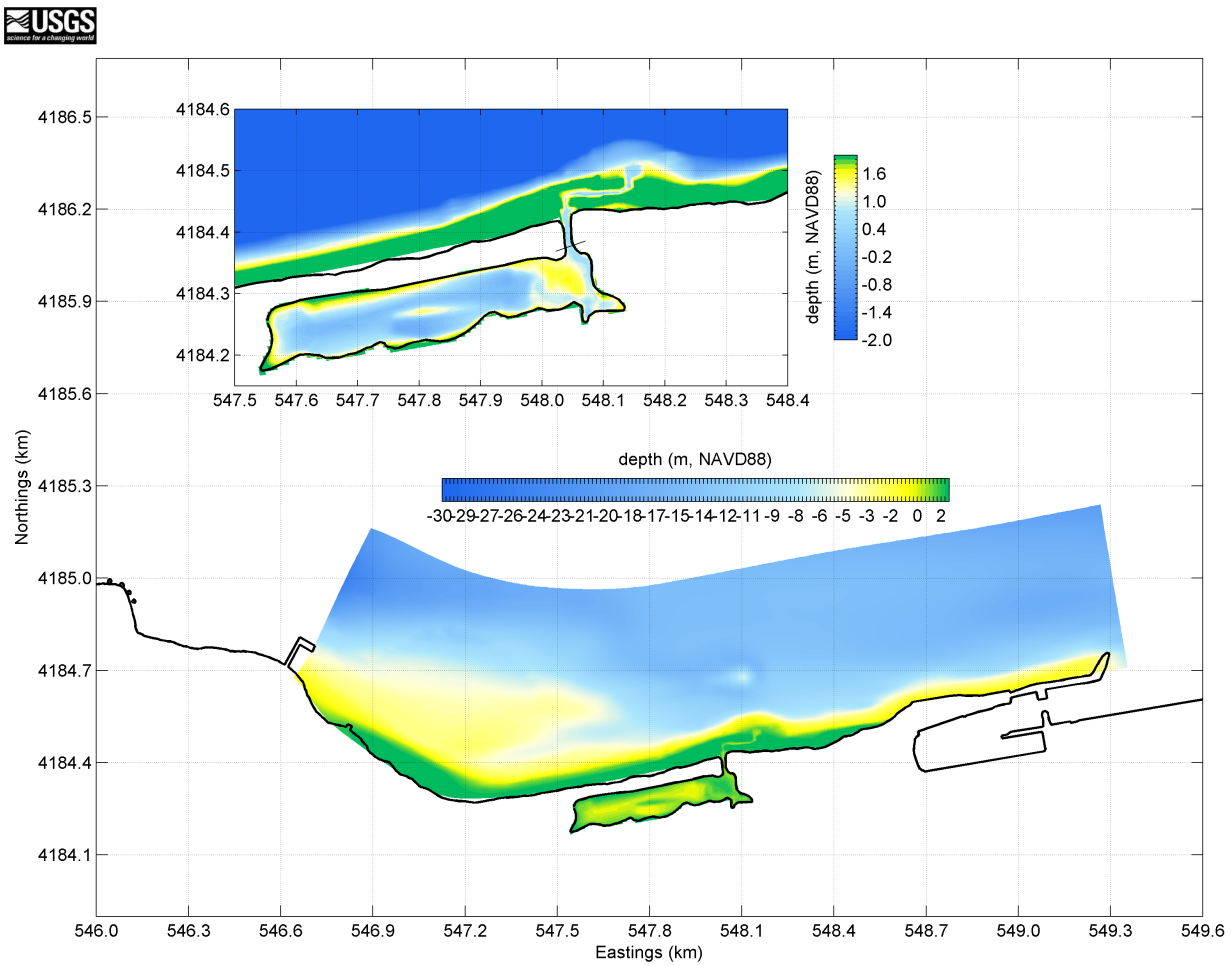

Figure 4. Grid and bathymetry used in model simulations of Crissy marsh, San Francsico Bay, California.

[Inset shows Crissy marsh and transect across which volumetric flow estimates and measurements are presented]

Comparison of Measured and Modeled Currents and Water Levels within the Inlet

A 2-MHz Nortek Aquadopp current profiler was deployed at the seabed underneath the pedestrian bridge on two separate occasions (September 8–11 and September 19–22, 2008) to measure water levels and currents in the inlet. The instrument, located at latitude 37°48.335' N., longitude 122°27.249' W., measured water depths with a pressure transducer and currents at 20 cm vertical bins by way of Doppler shift. There was little vertical structure in the observed flows and, hence, the model was run in a vertically averaged mode and compared to measured depth-averaged values.

Measured and model-predicted water levels and currents along the axis of the channel are compared in figure 5. Water levels were fairly well represented (root mean square error = 0.2 cm) during both time periods (CF5A = September 8–11, CF5B = September19–22). The magnitude and phase of the peak currents also compared well for both measurement periods, but the lower peak of the inward (negative) and outward (positive) directed currents were not represented in the model for the stronger spring tide during September 19–22, 2008. This may be due to an incorrect

representation of the shoal at the entrance of the inlet. Detailed topography and bathymetry was not available during the time period of these measurements.

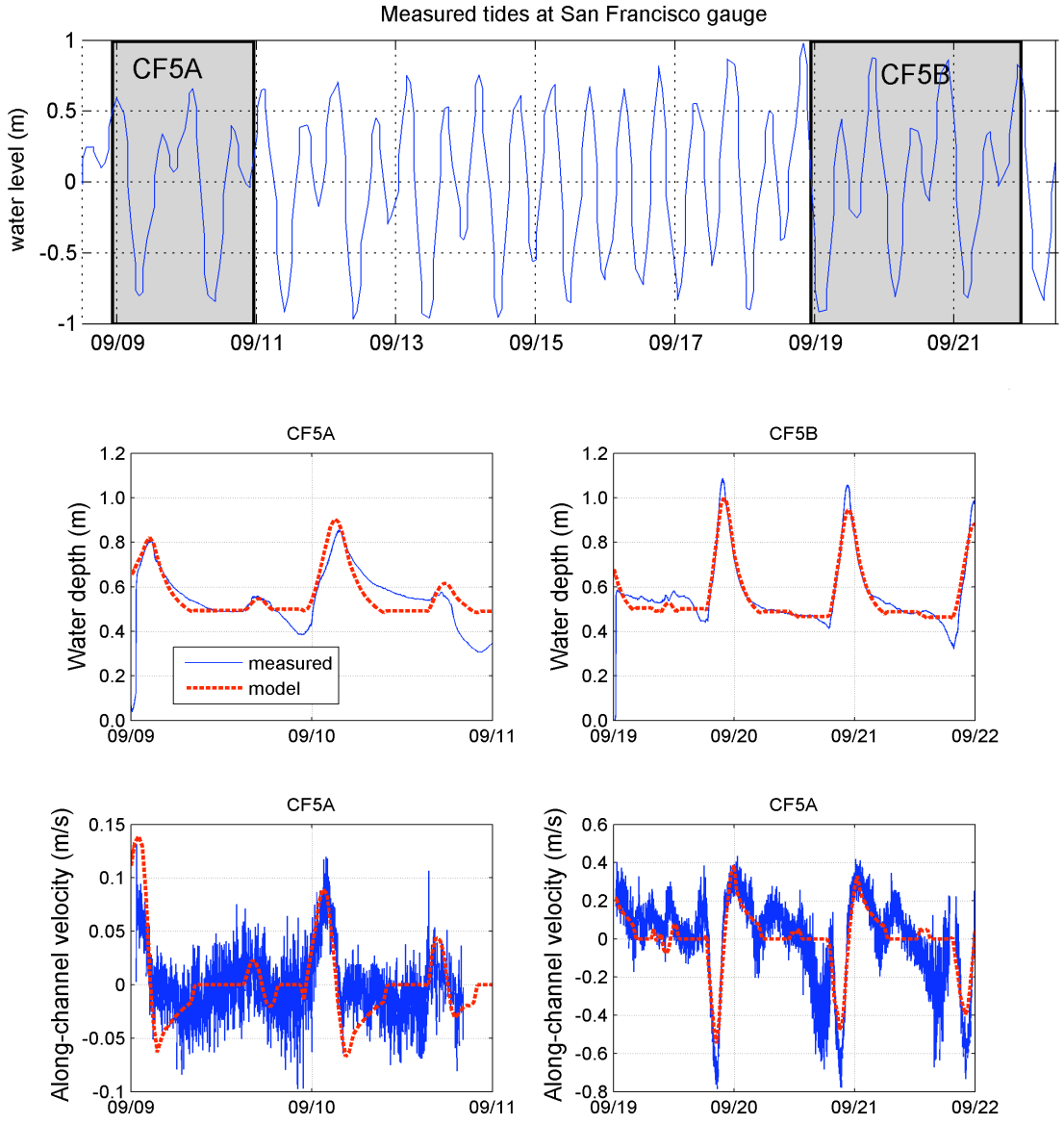

Figure 5. Comparison of measured and modeled water levels and currents within Crissy marsh inlet, San Francisco Bay, California.

[Upper plot highlights the time periods when measurements were obtained, as well as the stage of the tide measured at the nearby San Francisco gauge (NOAA/NOS station 9414290). Lower four panels compare measurements and model predictions. CF5A and CF5B refer to the sampling time periods of September 8–11 and September 19–22, 2008, respectively.]

Model-estimated Volumetric Water Flux during August 4–5, 2008

Figure 6 shows water depth and discharge as predicted by the model for August 4–5, 2008, the time period of the diel mercury sampling event. Flows into the marsh are depicted as negative discharge values, while flows out of the marsh are represented as positive discharge values (right-hand side of figure). Discharge (flows, m^3/s) are the total volumetric water flux across the channel at CF5 (fig. 4). The width of the channel at this location is approximately 12 m.

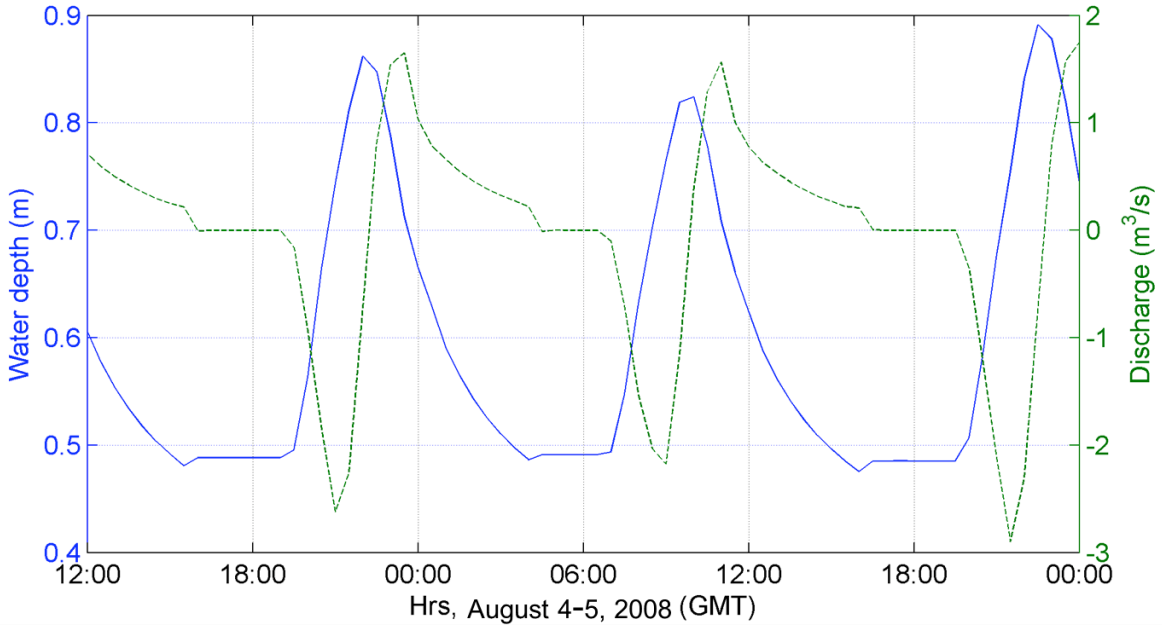

Figure 6. Water depth and instantaneous discharge through the inlet channel estimated by the numerical model, Crissy marsh, San Francisco Bay, California.

Sensitivity of Volumetric Water Flux/Tidal Flushing to Stage of Tide Cycle

Tidal exchange during the August 4–5, 2008 time period is compared to discharge at the same location for spring and neap tide cycles in figure 7. The model indicates that net discharge over a full tide cycle (24.84 hrs) during the August 4–5 sampling period was slightly outward (2,492 m^3), while the net flux during maximum spring and neap tides was inward.

Sensitivity of Volumetric Flux\Tidal Flushing to Channel Configuration

The sensitivity of volumetric flux to the channel configuration was tested by running the exact same simulation for the August 4–5 sampling period but with a channel configuration similar to that observed following mechanical excavation (channel oriented nearly directly north-south). As shown in figure 8, the net volumetric water flux (outward) increased by 113 percent (Q_{net} = 1,696 m^3) compared to the original estimated channel configuration, indicating that tidal flushing is sensitive to channel configuration and inlet elevation.

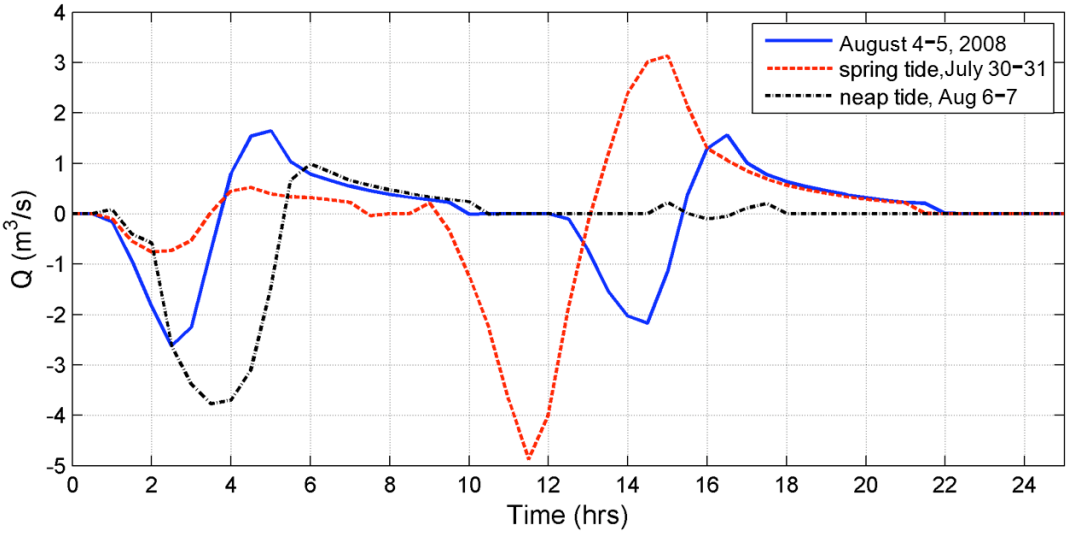

Figure 7. Volumetric water flows through Crissy marsh inlet, San Francisco Bay, California, for August 4–5, 2008, and corresponding spring and neap tides.

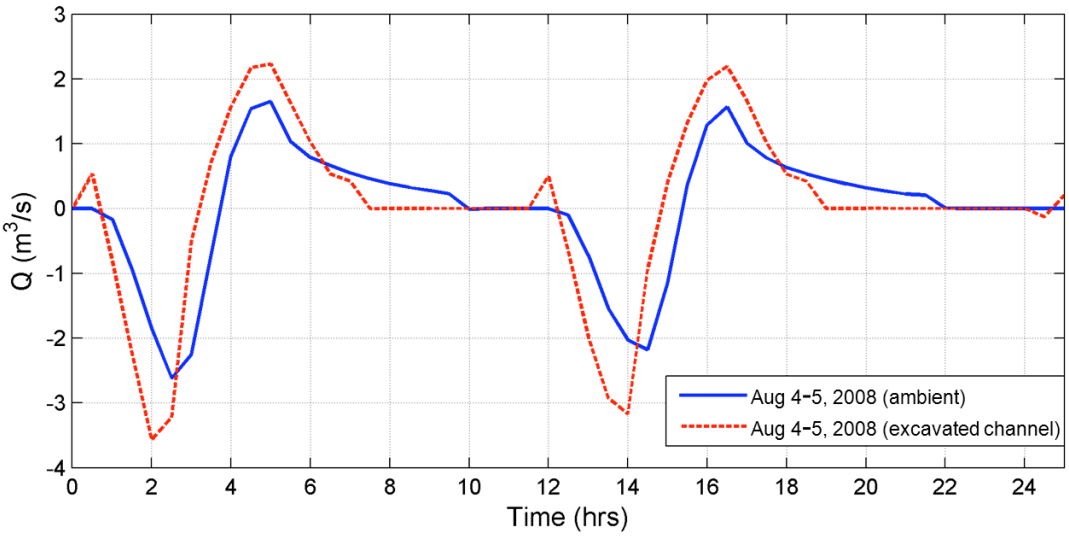

Figure 8. Comparison of volumetric water flux for different channel configurations, Crissy marsh, San Francisco Bay, California.

Laboratory Analyses—Surface Water

Point-based Water Quality Measurements

Field-based, site-specific, water-quality measurements were taken with all water samples collected by the USGS and the NPS. In addition to field measurements of salinity, conductivity, D.O., and temperature taken with a calibrated hand-held YSI-85, pH was measured with a field meter (Orion 250A) at the same time that water samples were collected for processing. In 2007, water samples collected by the USGS were analyzed in the laboratory as non-filtered water samples for THg and MeHg concentrations only. Samples simultaneously collected by NPS (NPS, unpub.

data, 2007–08) were sent to a commercial laboratory (Analytical Sciences, Petaluma, CA) for analysis of total suspended solids (TSS), chlorophyll-a, phaeophytin, sulfate, phosphate, nitrate, ammonium, and bacterial pathogens (total coliforms, *E. coli* and *Enterococcus*). In 2008, when the NPS water quality sampling and laboratory analysis had ceased, the USGS continued analyzing surface-water field conditions, using the same hand-held datasonde (YSI 85, calibrated and maintained by NPS), and began performing its own analysis of TSS, particulate organic matter (POM), and algal concentrations (via chlorophyll-a and phaeophytin pigments) on aliquots from the two 1-L bottles collected for Hg analyses.

In 2008, within 2 hrs of collection, after the return of the refrigerated samples to the laboratory, water samples were shaken to homogenization and then processed. Samples for chlorophyll-a and phaeophytin concentration were processed immediately under dark conditions. Aliquots of 100 ml were filtered under low vacuum through glass fiber filters (GF/F, 47mm), in duplicate. Filters were folded, wrapped in aluminum foil, and frozen at –80°C until acetone extraction and spectral analysis on a calibrated Shimadzu UV-VIS spectrophotometer within 60 days, per Mantoura and others (1997). Total suspended solids (TSS) concentration was assessed by filtration of a known volume of homogenized sample through pre-weighed GF/F filters (approx. 0.7 μm poresize). Samples were lyophilized (freeze-dried) and reweighed for dry weight of suspended particles. These samples were then combusted at 450°C for 5 hr, cooled in a dessicator, and reweighed to determine the POM concentration by difference (APHA, 1981b).

Aqueous Concentrations of Total Mercury and Methylmercury

In 2007, all water samples were preserved in the field (0.2 percent v/v sulfuric acid, final concentration), and analyses were run within 6 months of collection. In 2008, water samples were transported to the laboratory cold (on ice) and filtered within 2 hrs of collection for dissolved (<0.45 μm membrane filter-passing, hereafter "F-") and particulate (>0.7 μm GF/F filter, hereafter "P-") THg and MeHg concentrations, for a total of 4 fractions (F-THg, F-MeHg, P-THg, P-MeHg). Unfiltered (hereafter "U-") THg (U-THg) and MeHg (U-MeHg) surface water concentrations were calculated for all samples based on the sum of the filter-passing plus particulate fractions. A subset of samples was also assayed as non-filtered (whole-water) samples for THg and MeHg to compare methods. Aqueous samples were analyzed for F-THg and U-THg using Environmental Protection Agency (EPA) Method 1631 (EPA, 2002), which involves sample pre-oxidation, purge and trap, and cold vapor atomic fluorescence spectrometry (CVAFS), with quantification carried out on a Tekran 2600 Automated Mercury Analysis System (Tekran Instruments Corp., Seattle, WA). The same procedure was used to quantify P-THg concentrations using particulates collected on precombusted glass-fiber filters (Olund and others, 2004). Quality control included reagent blanks, field blanks, field duplicates, analytical duplicates, and matrix spikes. Because there are no certified reference waters for THg at concentrations relevant to environmental samples, calibration coefficients were verified against a certified sediment standard for THg (IAEA 405, described in sediment methods). At least 10 percent of all THg analyses were run in replicate and agreed within ±20 percent (acceptance criteria for the batches run). The method detection limit (as run) was 0.2 ng/L. The RPD for all analyical duplicates (F-THg, UF-THg, P-THg) was 5±6 percent (n=9). The average matrix spike recovery was 98±12 percent (n=5).

Aqueous samples were analyzed for F-MeHg and U-MeHg by distillation, aqueous ethylation, purge and trap, and CVAFS (DeWild and others, 2001), with quantification being conducted on a MERX automated methylmercury analysis system (Brooks Rand, Seattle, WA). P-MeHg was assayed by a methanol extraction (further described in sediment methods) of solids collected on precombusted glass fiber filters (~0.7 μm poresize), followed by centrifugation, buffering to the proper pH for ethylation, and continued measurement as described above (DeWild and others, 2004). Quality control included reagent blanks, field blanks, field duplicates, analytical

duplicates, and matrix spikes. At least 10 percent of all MeHg analyses were run in replicate and agreed within ±20 percent (acceptance criteria for the batches run). The method detection limit (as run) was 0.02 ng/L. The RPD for all analytical duplicates (F-MeHg, U-MeHg, P-MeHg) was 15±8 percent (n=9). The average matrix spike recovery was 104±5 percent (n=5).

Laboratory Analyses—Sediment

Mercury Speciation

Sediment mercury speciation assays included THg and MeHg. Samples collected for Hg speciation were frozen in the field immediately after collection and maintained frozen until analyzed. The percent of THg that was MeHg (percent MeHg) was used as an index of Hg(II)-methylation efficiency, per Gilmour and others (1998).

Total Mercury

Sediment THg was assayed according to Olund and others (2004). In brief, thawed, homogenized subsamples were digested in Teflon bombs with a 1:3 solution of concentrated nitric acid and hydrochloric acid (aqua regia), after which they were treated in sequence with bromine monochloride, hydroxylamine hydrochloride, and stannous chloride ($SnCl_2$) to convert all Hg species to gaseous elemental Hg^0. The gaseous Hg^0 was purged from aqueous solution, captured on a gold trap, thermally desorbed, and then quantified via CVAFS using a Tekran Model 2600 Automated Mercury Analysis System (Tekran Instruments Corp., Seattle, WA).

This method has an absolute detection limit of 0.3 ng. The standard reference material (SRM) routinely used as a quality assurance standard was IAEA-405 (estuarine sediment), with a certified value for THg of 810 ng/g dry wt. Average (± standard deviation) SRM recovery was 95±4 percent (n=4) for all batches of sediment samples assayed. Matrix spikes were conducted in duplicate (per batch) by adding a known amount of $HgCl_2$ solution to the sediment before digestion. Average matrix spike recoveries were 109±9 percent (n=4). While most samples were run only once, method analytical precision was tested on approximately 10 percent of all samples, by assaying in triplicate. The average RPD of these triplicate assays was 10±9 percent (n=5).

Methylmercury

Field frozen samples were thawed, homogenized, and weighed into polycarbonate centrifuge tubes. MeHg was extracted from sub-samples using 25 percent potassium hydroxide in methanol (CH_3OH), per Florida Department of Environmental Protection methods (Tate, 2009). The extractant was then pH-adjusted with acetate buffer and ethylated with sodium tetraethylborate. The ethylated MeHg was purged from aqueous solution, trapped, thermally desorbed, separated on a gas chromatographic column, reduced to elemental Hg^0 using a pyrolytic column, and detected using CVAFS, with quantification being conducted on a MERX automated methylmercury analysis system (Brooks Rand, Seattle, WA).

This method has an absolute detection limit of 0.08 ng/g of wet or dry sediment (as processed). The SRM routinely used as a quality assurance standard was IAEA-405 (estuarine sediment), with a certified value for MeHg of 5.4 (4.96–6.02) ng/g dry wt (as inorganic Hg). Average (± standard deviation) SRM recovery was 112±7 percent (n=5) for all batches of sediment samples assayed. Matrix spikes were conducted in duplicate (per batch) by adding a known amount of MeHgCl solution to the sediment prior to extraction. Average matrix spike recoveries were 82±12 percent (n=4). While most samples were run only once, method analytical precision was tested on selected samples by assaying select environmental samples in triplicate. The average RPD of these triplicate assays was 11±11 percent (n=6)

Additional Ancillary Sediment Geochemical Measures

Additional sediment samples were collected for geochemical characterization of conditions at the time of field collection.

Total Reduced Sulfur

Whole-sediment total-reduced sulfur (TRS) was assayed from zinc-acetate preserved and frozen samples within 6 months of collection. Samples were heated with potassium chromate, and released sulfide gases were trapped in a balch tube with zinc acetate then analyzed colorimetrically for concentration, per Cline (1969) and Fossing and Jørgensen (1989). Further details are given in Marvin-DiPasquale and others (2008). Method blanks were consistently below the lowest standard (100 nM), matrix spike recovery averaged 115±35 percent (n=3 sample pairs), and RPD between analytical duplicates was 11±9 percent (n=40 sample pairs).

Dry Weight, Porosity, Bulk Density, and Organic Content

The sediment parameters—bulk density, dry weight, porosity, and organic content—were from a single sediment sub-sample. Duplicate sub-samples were taken for these four parameters. A sub-sample of 3.0 cm^3 of homogenized wet sediment was obtained with a 3.0 cm^3 plastic syringe that had the needle end cut off of the syringe barrel. This sub-sample was transferred into a pre-weighed crucible and weighed. Sediment bulk density (g/cm^3) was then calculated as the weight:volume ratio.

Sediment dry weight and porosity were measured using standard drying techniques (APHA, 1981a). The crucible was then placed in an oven overnight at 105°C, placed in a dessicator to cool, then reweighed. The sediment percent dry weight was then calculated as [(dry sed weight)/(wet sed weight) x 100]. Sediment porosity (mL pore water per cm^3 of wet sediment) was calculated as the volume of water lost upon drying divided by the original sediment wet volume.

Organic content was calculated via the Loss on Ignition (LOI) assay (APHA, 1981b). The aforementioned dry sample in its crucible was placed in a combustion oven at 450°C for four hrs, burning off all organic constituents and leaving only mineral material. After cooling and reweighing, the weight loss was calculated and used to assess the total organic content of the sample.

Quality assurance was achieved by assaying all samples in duplicate. The relative deviation for duplicate analysis (average ± standard error) follows: (1) bulk density, 5±2 percent (n=7 sample pairs); (2) dry weight, 3±4 percent (n=7 sample pairs); (3) porosity, 5±2 percent (n=7 sample pairs); and (4) organic content, 14±9 percent (n=7 sample pairs).

Oxidation-reduction Potential

Sediment ORP was measured in the field immediately after homogenization and after sub-sampling approximately 10 cm^3 of sediment into a 20 mL screw-top serum vial. Measurements were made with a platinum band silver-silver chloride ORP electrode (model EW05990-55, Cole Parmer, Vernon Hills, IL; or Orion 9180BN, Thermo Scientific) used in conjunction with a hand-held pH/mV multi-meter (Model 59002-00, Cole Parmer, Vernon Hills, IL; or Orion 250A). Electrode accuracy was tested prior to use with freshly made buffer solutions (pH=7 and pH=4) saturated with quinhydrone, per the manufacturer's instructions (Cole-Parmer document #P1937). A cleaned probe was then fully inserted into the vial of homogenized sediment, then wrapped in parafilm to minimize any oxygen transfer while swirling the probe. Before recording the millivolt (mV) reading, equilibration was allowed within 10 minutes or until a stable reading was achieved. All measurements were corrected for temperature (19–22°C), and replicate readings (n=2) were collected immediately on the same vial after electrode removal and cleaning. The ORP meter

values were converted to E_h values (the potential, in millivolts, of the standard hydrogen electrode), using the following conversion provided by the ORP electrode manufacturer:

E_h = ORP (meter value) + E_R and E_R = (-0.718 x T) + 219.97 mV,

where E_R is the standard potential for a normal reference electrode and T is the temperature (°C).

pH

Sediment pH was measured immediately after homogenization and after sub-sampling approximately 10 cm^3 of sediment into a 20 mL screw-top serum vial. Measurements were made with a pH electrode used in conjunction with a hand-held pH/mV multi-meter (Model 59002-00, Cole Parmer, Vernon Hills, IL). The electrode was calibrated daily with fresh commercial pH=7 phosphate buffer and then rinsed clean with reagent water. The probe was then fully inserted into the homogenized sediment, gently swirled to ensure the exclusion of any air pockets for approximately 45–60 s, until the meter indicated a stable reading. Replicate readings (n=2) were collected immediately on the same vial after electrode removal and cleaning.

Pore Water Sulfide

As described earlier, pore water sulfide (PWSU) was preserved with a SAOB solution on duplicate samples in the field after collection with a mini-drivepoint sampler. These samples were kept on wet ice until returned to the laboratory. PWSU concentrations were measured with a sulfide-specific ion selective electrode (ISE) as modified from EPA Method 9215 (EPA, 1996). Calculations were made based on calibration standards prepared from sodium sulfide stock solutions, and quality assurance included laboratory reagent blanks, field blanks, matrix spikes, and analytical duplicates. Matrix spikes were within QA guidelines (18±14 percent), and the relative deviation for all samples assayed in duplicate was 13±5 percent (mean ± standard error; n=9 sample pairs).

Pore Water Sulfate and Chloride

Sulfate and chloride analyses were conducted on filtered samples of sediment pore water that were collected and stored under anaerobic conditions in the field and transferred to 13 cm^3 crimp-sealed serum vials and stored frozen (–80°C) until analysis. Sulfate and chloride were measured on an ion chromatograph (Dionex Model DX-300, Sunnyvale, CA) equipped with an auto-suppressor, an IonPac AG4A-SC guard column, AS4A-SC analytical column, and mobile phase consisting of 1.8 mM Na_2CO_3 and 1.7 mM $NaHCO_3$. Quality assurance included calibration standards prepared from crystalline sodium acetate, laboratory reagent blanks, field blanks, and analytical duplicates. The relative deviation for all samples that were assayed in duplicate and that were above the method detection limit was 6±8 percent (mean ± standard error; n=9 sample pairs) and 2±2 percent (n=9 sample pairs) for sulfate and chloride, respectively.

Laboratory Analyses—Plant

Plant Biomass Distribution and Tissue Quality

Plant structure of *Spartina foliosa* populations was assessed in July 2007 and 2008 at eight transect locations by measuring both maximal height and stem density within a 1-m radius of the plot marker. *S. foliosa* root density (g m^{-2}) at each transect location in the 0–10 cm surface sediment depth interval was assessed by separation and collection of root tissues within three replicate cores

(10 cm diam). Roots were rinsed, dried, and weighed in the laboratory as described by Windham-Myers and others (2009).

Root concentrations of ethanol and acetaldehyde were assessed on field-frozen root samples collected in triplicate for each treatment of site and date. Pre-weighed crimp-sealed glass serum bottles (interior volume = 9.15 ml) contained ~0.8–2.0 g of fresh root tissue. As described by Li and others (2004), sample bottles were autoclaved to force ethanol and acetaldehyde into gas phase and inhibit bacterial activity from generating additional fermentative products. Partition ratios were calculated empirically with known standards of ethanol and acetaldehyde, per Kimmerer and MacDonald (1987). A gas-phase sub-sample (100 µl) was drawn from the serum vial with an airtight syringe and injected into a Shimadzu 405 gas chromatograph equipped with a flame ionization detector (GC-FID). Gas concentrations were corrected by partitioning coefficients, sample volume, and sample moisture content. Even when normalized by root mass and moisture content, variability between triplicate samples ranged as high as 82 percent RPD for ethanol and 133 percent for acetaldehyde. Given this wide variation of sample response, an average of all three samples was used as the average estimate of ethanol and acetaldehyde concentration in plant cytoplasm. Matrix spike recoveries were difficult to evaluate with this internal sample variability, but by injecting previously sampled vials and re-autoclaving them, spike recoveries were found to be within the QA guidelines of <25 percent.

Data Reporting

All data are reported as tables in Appendix A. Data include means and propagated error terms for all datasets and are reported as standard deviation, unless otherwise noted.

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Appendix A

All project data are represented in the accompanying workbook files as spreadsheets *http://pubs.usgs.gov/of/2010/1299/of2010-1299_appendix_a/.*

Table A1. Sediment and plant characteristics, Crissy marsh, San Francisco Bay, California—2007.

Table A2. Sediment biogeochemistry, Crissy marsh, San Francisco Bay, California — 2007: temporal and spatial responses.

Table A3. Sediment biogeochemistry, Crissy marsh, San Francisco Bay, California — 2008: temporal and spatial responses.

Table A4. Plant fermentative respiration, Crissy marsh, San Francisco Bay, California — 2008: temporal and spatial responses.

Table A5. Surface water quality, Crissy marsh, San Francisco Bay, California — 2008: monthly time series.

Table A6. Surface water concentrations of Hg species, Crissy marsh, San Francisco Bay, California — 2008: monthly time series.

Table A7. Hourly surface water characteristics for August 4–5, 2008, Crissy marsh, San Francisco Bay, California: 24-hr diurnal study.

Table A8. Bi-hourly tidal channel fluxes of surface water for August 4–5, 2008, Crissy marsh, San Francisco Bay, California: 24-hr diurnal study.