Evaluation of Mercury Transformations and Trophic Transfer in the San Francisco Bay/Delta: Identifying Critical Processes for the Ecosystem Restoration Program

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April 2nd, 2007

Annual Report of Progress for Project #ERP-02-P40
To
The California Bay-Delta Authority (CBDA)
Sacramento, CA

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I. Introduction to the Project

Background

Due to extensive mercury contamination of the San Francisco Bay (SFB) watershed, resulting largely from historic mining operations, there has been a substantial recent effort to assess the extent of, and controls on, mercury (Hg) contamination in biota, water and sediment of this ecosystem. An important recent finding has been that certain indicator species (e.g. inland silverside, small mouth bass) have substantially lower mercury concentrations in the central Delta region than in most (if not all) tributaries (Davis et al. 2003, 2004; Slotton et al. 2000, 2002), even though benthic methylmercury (MeHg) production appears quite active in central Delta sediments (Gill et al. 2002, Marvin-DiPasquale and Agee 2003). It is unclear why this apparent paradox exists. To find out what factors underlie this striking regional trend, an intensive hypothesis driven process-level investigation was undertaken, which focused efforts on two representative regions of the SFB-Delta that contrast with respect to previously observed trends in biota Hg levels, namely, Frank's Tract (FT) in the Central Delta (CD) region (low biota Hg) and the Cosumnes River (CR) tributary (high biota Hg) **(Fig. 1)**. In addition to comparing these large regional areas, the study was designed to evaluate Hg dynamics and controls at the sub-habitat spatial scale. The three habitat types included in this regional comparison are a) emergent marsh [EM] dominated by tule (*Scirpus sp.*), b) zones of dense submerged aquatic vegetation [SAV], typically in the form of the invasive species *Egeria densa,* and c) non-vegetated open water [OW] areas. Seasonally inundated floodplains, represents a fourth habitat type investigated, which is an important component of the CR region, but largely absent in the CD region. The study was laid out in three phases over four years. Phases I and II both consisted of field sampling components designed to directly contrast the CR and CD regions, with Phase I weighted more towards spatial comparisons and Phase II weighted more towards temporal (seasonal) comparisons. In Phase III, research was focused exclusively on the CR region and Hg-cycling dynamics associated with the Cosumnes R. floodplain (CRF) during the period immediately before, during and after seasonal inundation.

Scientific goals and objectives of the project

The study has three primary objectives: (1) to conduct field-based and controlled laboratory research designed to examine the relative importance of specific ecosystem processes that control MeHg production, transport, and bioaccumulation in two representative, yet contrasting, SFB regions (CR and CD), (2) to examine these processes at three key sub-habitats (EM, OW, SAV) within both regions and in the CRF during the period associated with seasonal inundation, and (3) to address a suite of five explicit testable hypothesis that could explain the observed regional trend in biological mercury contamination. The primary goals of the project are to a) fill critical data gaps with respect to a process based understanding of Hg cycling and bioaccumulation in the SFB watershed, and b) compliment ongoing research efforts of other CALFED sponsored mercury investigation teams, who are using more concentration-oriented approaches over larger spatial scales.

Current working hypotheses

During the initial project conceptualization phase, five hypotheses (**Table 1**) were put forth that could potentially explain the previously reported regional trend in biological mercury contamination (i.e.

low in the central Delta, high in the tributaries). A project amendment proposal was subsequently funded in mid-2005, which added the expertise of Dr. Lisamarie Windham (a wetlands ecologist), and a sixth hypothesis (**Table 1**) was added. The research elements of the project were conceived and designed such that all of the listed hypotheses can be tested over the course of the study. These hypotheses are not mutually exclusive, and may act in an additive capacity resulting in the observed regional Hg bioaccumulation trend. Our original hypotheses were cast in terms of comparing the CR and FT sampling areas. However, at the request of CBDA we included a single sampling (July 2005) of the Dutch Slough (DS) area which is located in the Central Delta (CD) near our FT sampling location (**Fig. 1**). Thus, for the purposes of this report, we have recast the hypothesis as a comparison of the CR region with the more generally termed CD region (FT plus DS). We have likewise recast a majority of the graphics, data tables and statistical analyses in these terms (except where explicitly noted otherwise).

Management goals and objectives addressed by the project

By filling critical gaps in our process-level understanding of how mercury cycles and bioaccumulates in the SFB-Delta, this project supports CBDA's Ecosystem Restoration Program goal #6, "*to improve or maintain water and sediment quality*" (CALFED Bay-Delta Program 2000). Moreover, many of the Core Components (CC) identified in the CBDA sponsored 'Mercury Strategy for the Bay-Delta Ecosystem' document (Weiner et al. 2003) are addressed in the context of this project. These include: three aspects of CC #1 *[Quantification and evaluation of mercury and methylmercury sources]* - a) quantification of Hg pools in the Bay and Delta, b) identification of current key sources and sinks of MeHg in the Bay-Delta, and c) evaluation of the reactivity and bioavailability (for methylation) of Hg from different sources; two aspects of CC # 3 *[Quantification of effects of ecosystem restoration on methylmercury exposure]*, a) characterization of the biogeochemical cycling of Hg in wetlands, with emphasis on understanding processes and factors controlling MeHg abundance, and b) to determine if the net production of MeHg and biological exposure to MeHg vary among existing types of wetlands; one aspect of CC #5 *[Assessment of ecological risk]* - the identification of habitats, areas, and trophic pathways associated with elevated MeHg exposure; and two aspects of CC #6 *[Identification and Testing of Potential Management Approaches for Reducing Methylmercury Contamination]* – a) development of an empirical understanding of processes and habitat factors affecting MeHg production and exposure, and b) to determine which of the factors controlling MeHg production and exposure can be managed in the Bay-Delta ecosystem.

II. Project Timetable and Progress

Starting and target completion dates

The original project start and completion dates were July 1, 2003 - June 30, 2006. As part of Amendment #3, we received a 1 year extension to assay additional samples (described below). The current project completion date is June 30, 2007.

Project status

All field sampling and approximately 98% of all sample analysis has been completed, and we are actively working on the final data synthesis and the final report, which is due at the end of June 2007. We are also in the process of putting together a USGS Fact Sheet regarding this project's findings and the associated management implications.

Project milestones achieved

To date we have: a) completed Phase I & II field sampling (Dec. '03 - July '05) , which focused on the spatial (Phase I) and temporal (Phase II) variability associated with Hg dynamics in the three primary sub-habitats (listed above) within both FT and the CR main-stem, b) conducted an additional field sampling of these same sub-habitat types in Dutch Slough (July 2005), at the request of CBDA, to assess if Hg dynamics in the zone surrounding the pending Dutch Slough wetland restoration site are similar to those found in the nearby FT study area, c) completed Phase III sampling (Nov. '05 – July '06) focused on the CRF (included experiments on Hg-uptake in caged fish on and off of the floodplain), d) have done extensive work on the concentrations of and controls on the 'reactive' inorganic Hg(II) pool in both sediments and overlying water, as part of Amendment #3, e) expanded our investigation of Hg-bioaccumulation into larval fish in the CRF, as part of Amendment #3, by obtaining and assaying preserved samples from two previous years of caged larval fish studies conducted by UC Davis researchers, f) completed an extended suite of laboratory studies assessing the biokinetics of mercury trophic transfer (from water → phytoplankton → zooplankton → prey fish), g) published two papers associated with these laboratory trophic transfer studies (Pickhardt and others, 2006; Pickhardt and Fisher, 2007), and h) presented data associated with this project at a wide range of public forums over the last 4 years (see **Appendix I**).

III. Project Highlights and Results

This project has resulted in a plethora of process-level information regarding how Hg is cycled in different regions, in key sub-habitats, and through food webs of the SFB-Delta ecosystem. In order to best highlight our results to date, we consider the original hypotheses (**Table 1**), and relate whether the information collected to date appears to support, refute, or is insufficient to draw a conclusion, with respect to each hypothesis. We then briefly describe the data that leads us to that conclusion, and present any other relevant data associated with that hypothesis topic.

HYP I. Benthic conditions are more conducive for net MeHg production in CR than in the CD.

The data supports this hypothesis, as a number of key benthic parameters collected during Phase I and II are statistically greater in CR than in the CD, including a) sediment microbial Hg(II)-methylation rates, b) whole sediment and pore water MeHg concentrations, c) sediment reactive inorganic mercury ($Hg(II)_R$)[1], and d) pore water THg (**Table 2**). Other significant results include the fact that both the sediment organic content and pore water sulfate concentrations are higher in the CD than in the CR region, which leads to generally higher microbial sulfate reduction rates (SRR) and higher concentrations of sedimentary acid volatile sulfur (AVS) and total reduced sulfur (TRS) (**Table 2**). This regional trend in SRR and TRS may in part underlie the reason the central delta tends to have generally lower rates of MeHg production than do the freshwater tributaries, as discussed below. Apart from these overall regional differences, sediment MeHg concentrations and production rates were generally highest in the vegetated EM and SAV sub-habitats, compared to the non-vegetated OW zone, particularly in the CR (**Fig. 2E and 2F**).

[1] 'Reactive' inorganic mercury' ($Hg(II)_R$) is defined here as the pool of Hg(II) in whole (non-digested) sediment that is readily reduced to gaseous elemental Hg^0 by excess $SnCl_2$, under mildly acidic and anoxic (N_2 purged) conditions, during a 15 minute 'purge & trap' period. It is used in the calculation of MeHg production (potential) rates, as the surrogate measure of the Hg(II) fraction that is most readily available to sediment bacteria for Hg(II)-methylation.

Temporal and Spatial Methylmercury Production and Degradation in Sediments

The production of MeHg is a function of both the activity of the bacteria involved in the Hg(II)-methylation process and on the availability of $Hg(II)_R$ for methylation. The $^{203}Hg(II)$-methylation rate constant (k_{meth}), derived from our standard $^{203}Hg(II)$ amendments to sediment samples, provides a measure of the activity of Hg(II)-methylating bacteria at a given site and/or time. Values of k_{meth} were higher in CR than in the CD, with differences among sub-habitats following the general pattern of SAV > EM > OW, particularly during the July period when peak rates were observed in both regions (**Fig. 2A, 2B**). In addition, the pool size of $Hg(II)_R$ was higher in CR than in the CD for the vegetated sub-habitats (EM and SAV), but generally not in the OW zone (**Fig. 2C, 2D**). The combined effect of both factors led to higher calculated rates of MeHg production in the CR region for all sub-habitats and times, with particularly high rates in the EM and SAV sites during July (**Fig. 2E, 2F**). The actual sediment MeHg concentrations generally followed these same spatial trends (**Fig. 2G, 2H**), although the temporal changes in their magnitude were muted relative to the relative temporal changes measured for MeHg production rates.

A primary focus of this research was to decipher which environmental factors exerted the strongest control on both the activity of the Hg(II)-methylating bacteria and on the $Hg(II)_R$ pool size. Of everything examined, field/incubation temperature explained the most variability (22%) in the LOG transformed k_{meth} values (**Fig. 3**). There was no consistent relationship with k_{meth} and microbial sulfate reduction rate or with sediment organic content, across the complete data set. However, within a given region or for a given sampling period, such positive correlations were not uncommon. Since it is well established that both of these factors are important controls on microbial Hg(II)-methylation, these findings suggest that for large temporal/spatial ecosystem studies such as this one, simple relationships are often obscured due to the multiple controls on the microbial methylation process.

Sediment grain size has frequently been reported as an important control on THg concentration throughout the SFB (Conaway and others 2003, Alpers and others 2006) and elsewhere, with higher THg concentrations associated with smaller particles. Such a relationship was also found to exert a significant control on $Hg(II)_R$ concentration (LOG transformed) (**Fig. 4**). However, sediment geochemical conditions also appear to mediate the $Hg(II)_R$ pool. This is most readily reflected by the significant negative relationship between the percentage of THg that is $Hg(II)_R$ ($\%Hg(II)_R$) and the pool size of total reduced sulfur (**Fig. 5A**), and the positive relationship between $\%Hg(II)_R$ and sediment redox (**Fig. 5B**). These trends imply that as the rate of microbial sulfate reduction increases, and the concentration of reduced-S species increases (e.g. **Table 2**), the fraction of inorganic Hg(II) that is readily available for methylation decreases. A very similar trend exists in a number of our other ecosystem projects, including those in Louisiana wetlands, NAWQA stream/river sites, and the CBDA sponsored Petaluma River marsh project (Yee et al. 2005, 2007).

MeHg degradation was also investigated during Phase I and II, using $Me^{203}Hg$ radiotracer incubations to calculate the site specific rate constants (k_{deg}) for this process. Porewater MeHg concentrations were used in conjunction with k_{deg} to calculate MeHg degradation rates (excluding July 2005, when no pore water MeHg was measured). No regional differences were detected for either values of k_{deg} or MeHg degradation (**Table 2**). Temporally, values of k_{deg} peaked during June in both regions and in all sub-habitats (not shown), prior to the peak of k_{meth} in July (**Fig. 2A, 2B**). When calculate as described above, MeHg degradation rates ranged from 1% to 133% of MeHg production rates, with a median value of 10%, and an average (± std error) of 21 ± 4% (n =40). When whole sediment MeHg concentrations were used in conjunction with k_{deg} values, calculated site-specific MeHg degradation rates typically far exceeded calculated rates of MeHg production, with MeHg degradation/production

ratios ranging from 1-1314, with a median value of 46, and an average (± std. err.) of 158 ± 38 (n =57). These calculations suggest that, like inorganic Hg(II), much of the whole sediment MeHg pool may not be readily available to short term microbial degradation, and may be complexed with refractory dissolved organic matter. Thus, the use of the dissolved MeHg pool in such calculations may be a more appropriate approach until a better surrogate measure of the microbially available MeHg pool is developed (akin to $Hg(II)_R$).

Total Mercury and Methylmercury in the Water Column

The water column MeHg data also supports HYP I. The higher rates of MeHg production and MeHg concentrations in CR sediments do appear to manifest themselves in 50% higher average filtered and unfiltered MeHg concentrations in the water column of the CR region compared to the CD (**Table 3**), although the spatial and temporal variability these parameters was quite high (not shown). Over 80% of dissolved THg concentrations were < 2 ng/L, and over 87% of dissolved MeHg was < 0.2 ng/L for both regions. The %MeHg in filtered water was similar for both regions, and was greater than for the % MeHg in unfiltered water, which is expected given the lower average partitioning coefficient between particles and the aqueous phase for MeHg, compared to THg (**Table 3**). The average THg in unfiltered and filtered water samples was also higher in the CR, although the distribution coefficients (K_d values) were not. The higher average total suspended sediment load in the CR (**Table 3**) may be driving these observed regional differences in water column THg.

HYP II. Physical and/or geochemical conditions mediating MeHg benthic flux to the overlying water column are more favorable in the CR than in the CD.

Calculations of MeHg diffusive flux from the sediment to the overlying water column support this hypothesis, as regional average MeHg flux was 2x higher in CR than in FT (**Fig. 6A**). When both region and habitat were considered the OW and SAV sites within the CR were appreciably elevated compared to all other region/habitat combinations (**Fig. 6C**). In addition to these diffusive flux estimates, the direct measurement of THg, MeHg and DOC benthic flux was attempted during the March '04 field trip, using the whole core flux method (Kuwabara et al. 2002, 2003; Topping et al. 2004). Multiple cores (n=3) were taken from one OW and one SAV site in each region (FT and CR). The results were equivocal, the variability among cores within a site was very high for both THg and MeHg, and the data (not shown) was insufficient to either support or refute HYP II..

HYP III. The CD has a higher net loss of MeHg from the water column, due to either microbial and/or photo-degradation, resulting in a lower net transfer of MeHg into the base of the food web in this region, compared to the CR.

The data collected is not sufficient to either support or refute this hypothesis, although measured MeHg degradation rates in the water column appear low. Two separate water column experiments were conducted using Hg stable isotope amendments ($^{201}Hg(II)$ and $Me^{199}Hg$) to assess water column rates of microbial and photo- MeHg degradation, Hg(II)-methylation, and dissolved gaseous Hg^0 production. In the first case (June '04 sampling trip), amendment experiments (24-hr incubations with multiple time point sub-sampling and light/dark treatments) were conducted on-site, in Teflon bottles, in the open water regions of both CR and FT. In the second case (March 2005), similar experiments were conducted using Teflon bags (Whalin and Mason, 2006) at the Univ. of MD, using site water from FT and CR. In both sets of experiments and for both sites, observed rates of MeHg loss were either low (0.8-4% d^{-1}) or not detectable. These results may in part reflect the high load of TSS in both regions (**Table 3**). In a majority of the cases the changes in Hg species with time was non-linear

and resulted in non-significant slopes (data not shown).

HYP IV. Regional and/or sub-habitat differences in food web dynamics, such as habitat utilization, feeding behavior, food chain length and composition, and/or species-specific Hg bioaccumulation rates at the food web base, account for the higher MeHg concentrations in CR biota compared to the CD.

The data collected is sufficient to reject this hypothesis under current operating conditions in the Delta. However, laboratory experiments and field studies identify important processes that could potentially modulate uptake of mercury into the base of the food web given different external inputs of MeHg in the region.

Laboratory Studies of Mercury Biodynamics in Food Webs

Species-specific kinetics of Hg accumulation in 4 phytoplankton species, 1 pelagic zooplankter, 1 macroinvertebrate associated with SAV, and 2 fish species were investigated in controlled laboratory studies, using filtered natural waters from CR and FT. Both dietary and direct aqueous accumulation of Hg from the two water types was determined for invertebrates and fish.

In all 4 phytoplankton species, there were no significant differences in inorganic Hg(II) accumulation by cells, measured by volume concentration factors (VCFs), in the two water types after 24 hours (**Fig. 7** or Table 1 within Pickhardt & Fisher 2007). However, 2/4 of species (diatom and cryptomonad) had significantly higher MeHg bioconcentration factors in FT water relative to CR water and all species showed 1-2 orders of magnitude greater VCFs for MeHg than for Hg(II) (**Fig. 8**). The green alga (*Chlamydomonas reinhardtii*) accumulated significantly higher concentrations of MeHg in FT water relative to CR water, but high variance in the VCFs for this species did not produce significant differences in VCFs (Pickhardt & Fisher 2007). The cyanobacterium that was used in our experiments did not produce significantly different VCFs in either water type for Hg(II) or MeHg (Pickhardt & Fisher 2007). While it is unclear as to what accounted for the cases where differences were observed, FT water is both higher in ionic strength and in DOC (**Table 3**). We are currently hypothesizing that the latter may have a significant effect on MeHg uptake kinetics in certain phytoplankton species, and future studies should prioritize a rigorous test of this hypothesis.

Invertebrate accumulation of inorganic Hg(II) and MeHg from dietary sources (Hg exposed phytoplankton) and assessed as assimilation efficiency (AE) of ingested food are generally similar between FT and CR waters (**Table 4**). Direct, aqueous accumulation of Hg(II) and MeHg (presented as uptake coefficients or k_u) by invertebrates is generally similar (**Table 4**). Importantly, loss rates (or k_e) of Hg(II) were much higher than for MeHg, regardless of the source (aqueous or dietary) (**Table 4**). Experiments to test for the sub-lethal and reproductive toxicities to *Daphnia pulex* of Hg(II) and MeHg in FT and CR waters suggested greater negative impacts on reproductive parameters in FT water (**Table 5**, from Pickhardt & Fisher in preparation).

Fish accumulation and retention of Hg(II) from dietary sources (Hg exposed phytoplankton fed to either *Daphnia pulex* or *Hyallela sp*., which were subsequently fed to fish) exhibited similar AE's for each water type and in both fish species tested (**Table 6**). The AE of MeHg from invertebrate diets by mosquitofish was greater in FT water, but the AE of MeHg by redear sunfish was greater in CR water (**Table 6**). Overall, AEs for ingested MeHg in fish were greater than AEs for ingested Hg(II), with differences ranging from a factor of about 2 for mosquitofish to a factor of about 10 for redear sunfish. Calculated uptake constants measuring direct, aqueous accumulation of Hg(II) and MeHg by the two fish species, from the two water types, was greater in CR under all treatment combinations (**Table 6**).

Despite the greater uptake rate constants in CR water, final fish burdens are overwhelmingly determined by high AE's of MeHg from invertebrate diets (Pickhardt et al., 2006). Efflux rates for both forms of mercury, regardless of the source, were generally in the 1-2% d^{-1} range for both fish species. Regarding tissue distribution within the fish, the greatest pool for MeHg was in the muscle and skeletal tissue, whereas inorganic Hg from diet was predominantly associated with the intestines following dietary uptake; Hg(II) obtained from the aqueous phase was generally evenly distributed among the tissues of the fish (Table 3 in Pickhardt et al. 2006).

Field Based Studies of Trophic Mercury Enrichment

The patterns of trophic enrichment through habitat specific food webs were statistically similar among CR and CD regions, but concentrations of MeHg in the CR region SAV food web were approximately 2.6 times higher than those in the FT SAV food web (**Fig. 9**). These differences in concentrations were consistent among food webs and specific species compared between regions (i.e. primary producers, primary consumers, fish). Factor differences in MeHg concentrations between paired species in each region (through space and time) ranged from 2.5 in epiphytic algae to 2.9 in redear sunfish and largemouth bass (**Table 7**). These factor differences were similar to those previously reported for 350 mm largemouth bass (Jay Davis, SFEI). The consistency of trophic enrichment among food webs and among different species pairs strongly suggests limited species-specific or habitat-specific controls on mercury bioaccumulation among the different regions. Differences in mercury bioaccumulation among regions appear to be set at the base of the food webs. Indeed, surface water dissolved MeHg and sediment MeHg concentrations also different by a factor of 2-3. The enhanced uptake of MeHg into phytoplankton in FT (see laboratory studies) appears to be insufficient to counter the influence of higher aqueous MeHg concentrations in the CR region. This consistency of bioaccumulation among species within food webs that is apparently linked to aqueous MeHg is consistent with other recent studies nationally and internationally.

Food webs in both regions were highly dependent on SAV habitat and epiphytic algae at their base (**Fig. 10**). Direct comparison between primary consumers in open-water (bulk zooplankton) and in SAV (amphipods) during March and July 2005 showed slightly higher MeHg concentrations in bulk zooplankton from the open-water habitat. However, these differences were only marginally statistically higher in CR and not significantly different in the CD. Franks Tract had significantly higher biomass of epiphytes (chlorophyll_a normalized to mass or surface area) and significantly less MeHg per mass than CR (see A. Lorenzi poster). The role of biomass dilution from primary producers as a mechanism driving differences in mercury bioaccumulation between the different regions was evaluated in March and July 2005. Densities of *Egeria densa* and associated live biomass and leaf surface area, epiphytic algae biomass and mercury concentrations, and aqueous mercury concentrations, were all used to calculate the partitioning of mercury among different phases (dissolved MeHg, suspended particulate MeHg (phytoplankton) and epiphytic MeHg (attached phytoplankton). This provided some insight into the potential role of biomass dilution in modifying SAV bioaccumulation in the CD region. Results shown in **Table 8** suggest that biomass dilution, predominantly from the growth of epiphytic algae, could deplete aqueous MeHg in the SAV resulting in overall lower MeHg concentrations in the base of the SAV food web relative to habitats where aqueous MeHg was not limited. It is unlikely that biomass dilution alone was the primary factor controlling regional differences in mercury bioaccumulation since open-water biota in the CD, not affected by biomass dilution, was also low compared to the CR.

HYP V. Larval fish reared on the CR floodplain have higher Hg levels than those reared in the

Cosumnes R. proper, due to increased MeHg production and Hg exposure during seasonal inundation of the floodplain (i.e. the reservoir effect).

The sediment data supports HYP-V, as average sediment MeHg production rates and concentrations associated with the floodplain are significantly higher than those measured in the CR main channel (all habitat types included) (**Table 9**). **Figure 11** illustrates the Cosumnes R. hydrograph, sediment and water sampling dates, and the period during which the floodplain is hydologically connected to the Cosumnes R. The temporal and spatial trends for sediment MeHg production rates and concentrations, during Phase III sampling, are given in **Fig 12**.

In contrast to the sediment data, the water column MeHg concentrations did not appear to be particularly elevated, compared to the CR main stem during Phase III sampling (**Fig. 13**), and thus in itself water column MeHg concentration data does not support HYP-V. Indeed, some of the highest filtered and unfiltered MeHg concentrations were measured below the floodplain in the CR main channel towards the end of the sampling period.

Another aspect of mercury chemistry examined during Phase III was the interaction between $Hg(II)_R$ and the quality/quantity of dissolved organic carbon (DOC) in overlying water. Soon after initial floodplain inundation (January) the quantity of DOC was similar among sites within the floodplain and in the CR main channel. However by June, after the floodplain became hydrologically disconnected from the river (late May, **Fig. 11**), DOC concentrations rose considerably at sites within the floodplain (**Fig. 14A**). The quality of DOC also changed, as the percent of the DOC pool that was associated with the hydrophobic organic acid (HPOA) fraction began to decrease (**Fig. 14B**). Since the HPOA fraction is generally associated with aromatic-C, largely derived from lignin, this suggests an increased input of DOC from phytoplankton blooms in the summer, compared to the winter/spring. The observed negative relationship between %HPOA and $Hg(II)_R$ LOG[Kd] (**Fig. 14C**) suggests that the changing temporal quality of the DOC pool at least partially controlled the distribution of $Hg(II)_R$ between suspended particulate and the dissolved phase, and resulted in more $Hg(II)_R$ adsorbed to particles in the summer, compared to the winter and spring. These findings have implications for the MeHg production process on the floodplain and elsewhere, where the dissolved $Hg(II)_R$ pool in overlying water is in contact with active zones of MeHg producing bacteria in surface sediments.

The caged fish experiment and associated biological data are insufficient to support or refute HYP-V. Juvenile salmon caged on the CRF for 9 weeks grew in length (**Fig. 15A**) and at some sites in muscle tissue mass (**Fig. 15B**). Mercury concentrations in muscle tissue increased significantly relative to initial concentrations and were similar to other resident adult fish in the region or the greater San Francisco Bay with concentrations reaching over 2 µg/g (dry wt.) at the 'below FP' site (**Fig. 16**). Because of major differences in growth among the sites (e.g. growth (d^{-1}) after 52 days of exposure: upper FP = 0.019, lower FP = 0.006, River above FP = 0.018, River below FP = -0.007) it is difficult to determine if the differences in THg concentrations in the salmon were due to growth or exposure (concentrations of mercury in food or ingestion rates) at the different sites. Zooplankton, common prey of the juvenile salmon, collected during floodplain inundation had THg concentrations that did not vary statistically among caging sites and were similar to samples collected in the CR at other times during the study, except for the 'below FP' site where concentrations reached over 1.5 µg/g (dry wt.) at the end of the 9 week exposure period (**Fig. 17**). Indeed the elevated zooplankton mercury levels at the 'below FP' site corresponded to the higher water column MeHg concentrations found at that site, suggesting higher exposures for biota below the floodplain. Further, accumulation patterns in zooplankton over time at the 'below FP' site shared a striking similarity to caged fish at that site (**Figs. 16 and 17**). Given the study was not designed to account for fluxes on and off the floodplain it is not

possible to determine if the elevated mercury levels recorded at the 'below FP' site were coming from the floodplain itself or from upstream within the CR mainstem.

HYP VI. Regional differences in plant-Hg interactions, such as Hg uptake and leaching rates by various plant species, gaseous elemental Hg efflux by plants, and/or plant community composition and density, leads to regional differences in Hg cycling pathways, MeHg production, and ultimately to differences in Hg levels in biota.

The data collected to date can neither fully support nor refute HYP VI, although significant regional differences in plant community characteristics and plant-Hg interactions have been identified.

SAV Biomass and Mercury Concentrations

The regional SAV communities differ in five distinct ways. (1) FT is dominated by the exotic waterweed *Egeria densa*, whereas CR is a mixture of three species: *E. densa* (hereafter, *Egeria*), *Ceratophyllum demersum* (Coontail), and *Myriophyllum aquaticum* (Parrotfeather). (2) Where sampled, the standing stock of live biomass (g m^{-2}) is 2-3X greater in FT (**Fig. 18**). While no significant seasonal differences in live biomass was observed, dead biomass (g m^{-2}) increased 2X in FT between March and July, implying a complete turnover of vegetation and higher primary production rates in FT, compared to CR (**Fig. 18**). (3) Live *Egeria* in FT has 2X the leaf density per stem compared to CR *Egeria*, which may allow for a greater amount of epiphytic algal growth. (4) Plants from the CR have higher Hg levels in their tissue (**Fig. 19**). (5) On an area (m^2) basis, plant-Hg concentrations are similar between regions due to greater plant densities in FT (**Fig. 20**).These SAV THg levels are 200-300X lower than the THg pool in surface sediment 0-1 cm depth interval of sediment in the both regions. Even though only a small percentage the sediment THg pools was 'reactive' inorganic Hg(II) (i.e. $\%Hg(II)_R$ [avg]; CR = 3.3%; FT = 1.7%), this still represents sediment $Hg(II)_R$ levels 4-10X higher than from SAV biomass, even if all of the plant-Hg was assumed to be $Hg(II)_R$ upon plant decomposition.

Cleaned (ephiphyte-free) new-growth *Egeria* samples (collected July '05) were assayed for MeHg content, which was higher in the CR (15.0 ± 8.5 ng/g d.w.) compared to FT (4.8 ± 1.8 ng/g d.w.) (n=4 from each region). These MeHg levels represented a high proportion of the THg pool (31±18%, n =8) in these plant tissues. These regional differences in the SAV are similar to those observed for the epiphyte and other biological MeHg data (**Table 7**). Higher biomass density and higher leaf area associated with *Egeria* patches in FT, appear to support a greater epiphyte densities (per m^2) in FT compared to CR. This suggests the possibility of regional differences in the extent of MeHg biodilution in the epiphytic foodweb base (i.e. higher biodilution in FT), as discussed above.

SAV Decomposition and Hg release

A series of SAV decomposition experiments were performed in the laboratory under controlled conditions, to examine biomass loss, DOC release and Hg release rates from senescent SAV. Each experiment was 1-month in duration and differed slightly in experimental conditions and goals. All used DI-rinsed new-growth plant material. Key findings include: a) SAV decomposition rates (biomass loss) and THg release rates were highest under oxic water conditions, b) plant species collected from FT (incubated in FT water) degraded faster than the same species collected from CR (incubated in CR water), c) plant species specific degradation rates follow the general trend [*Egeria* > coontail > *ludwigia* > tule], and decrease with increasing plant biomass C/N ratio (**Fig. 21**), e) the $\%Hg(II)_R$ in the dissolved phase decreased with time and was likely complexed by the increasing

concentrations of (presumably high molecular weight) DOC, e) salinity alone had little effect on both SAV degradation and HgT release rates. The C:N ratios of *Egeria* (SAV) and Tule (an emergent marsh macrophyte) are significantly lower in FT than in the CR (**Fig. 21**), suggesting a greater supply of available nitrogen (N) in FT for plant uptake. This may be a primary factor responsible for the faster growth and decomposition rates of *Egeria* and Tule in FT, compared to the CR. Overall, we conclude that the THg pool associated with plant biomass turns over faster in FT, due to higher growth and decomposition rates.

Gaseous Mercury Flux from Emergent Marsh Plants and Water

A limited number of gaseous mercury (Hg^0) flux rates were measured from leaf surfaces of emergent marsh plants and from open water surfaces (**Fig. 22**). Overall, maximum Hg^0 flux rates observed from *Scirpus acutus* leaves (March and July '05) were approximately 2x higher than maximum rates measured from water surfaces (October '06). Further, Hg^0 flux from plant surfaces are even greater in terms of landscape surface area, as the surface area of plant leaves are 2-4 fold greater than their associated 2-dimensional planar footprint. The flux of Hg^0 associated with the water surfaces was entirely dependent on incident UV radiation, whereas Hg^0 fluxes from vegetation was strongly correlated with transpiration rates (and CO_2 uptake). We conclude that large zones of emergent vegetation could represent major areas of Hg^0 flux, compared to the flux associated with nearby water/air interfaces.

IV. Potential Management Implications of Findings to Date

As stated in the CALFED Mercury Strategy (Weiner et. al 2003), a major challenge that confronts agencies and ecosystem managers involved with SFB ecological restoration is "to avoid increasing – and to eventually decrease – biotic exposure to MeHg." By providing baseline process specific information regarding Hg cycling in two representative and contrasting regions, the current research program provides data that directly informs ongoing and planned restoration and management actions in both the central Delta and in the tributaries, such as a) the Dutch Slough tidal marsh restoration and the Sherman Island Demonstration Project both focused on central delta wetland restoration and sequestering carbon in accreting peat, b) the McCormick-Williamson Tract Floodplain Restoration, near the CR study area, as part of DWR's 'North Delta Flood Control and Ecosystem Restoration Project', c) the 'Flooded Islands' Project which would significantly impact the hydrology and salinity conditions in the SFB-Delta in general, and FT in particular.

The current study shows clearly that the regional differences observed in fish Hg concentrations are linked to the amount of MeHg that enters the base of the food web. This in turn appears to be linked to MeHg concentrations in water, which in turn is linked to the both the rates of net MeHg production in the sediment and the transfer rate of MeHg from the sediment to the overlying water. As such, management actions that either increase the net rate of benthic MeHg production or the flux rate of MeHg from sediment to water, will likely result in higher water column MeHg concentrations and higher initial concentrations of MeHg transferred to the base of the foodweb, and ultimately in higher MeHg concentrations into top level trophic species of concern. Conversely, management actions that would decrease the net rate of benthic MeHg production or the flux rate of MeHg from sediment to water would likely result in lower levels of Hg contamination in top level trophic species. There is clearly a balance between microbial activity and the availability in Hg(II) for methylation, that needs to be recognized when considering the impact of a particular management action on net MeHg production. However, we are only now starting to truly appreciate this balance, and how particular

landscape manipulations might impact Hg cycling over the long term. The biggest implications of these findings is that management actions that significantly impact sediment and/or water geochemistry, will likely have a more direct effect on increasing or decreasing MeHg production and uptake into the food web, than would management actions that had more of a significant impact on regional food web structure directly.

These results also do not imply that food web considerations are unimportant. Food web Hg-biodynamics laboratory studies indicated that the majority of MeHg accumulated in invertebrates and fish originates from consumed diet items. Additionally, the largest bioconcentration of MeHg from the dissolved phase to biota occurs between water and primary producers (particularly epiphytic algae), implicating suspended and attached forms of phytoplankton as the primary entry points of Hg in both regions studied.

There are three general areas of ecosystem management that the findings of this study may have direct relevance to: A) increased tidal wetlands in the central delta region, B) changing hydrology and salinity in the central delta that would decrease invasive SAV (i.e. *Egeria*) and C) increased flood plain habitat in the tributaries. While it is often challenging to extrapolate the findings of one study to hypothetical changes, we attempt here to link the findings of our current study to these three ecosystem management activities, and to highlight what we do and do not know.

A) Increased tidal wetlands in the central delta region

Our findings generally support the widely held contention that vegetated sites are more active zones of MeHg production than open water non-vegetated areas. Thus, an increase in tidal wetlands may well lead to increased MeHg production in those locations. However, an important process that was not explicitly studied as part of the current research is the net flux (mass transfer) of THg/MeHg from tidal wetlands or the floodplain environment to the larger adjacent water body. This can be an important term controlling the ultimate concentration of water column MeHg. Thus, it is unclear to what extent an increase in MeHg production in newly created wetland areas would affect the overall balance of water column MeHg concentrations in the larger central delta region. Wetlands are also known to be important zones of net particle deposition. Thus, one possibility is that the creation of extensive new wetland habitat effectively traps much of the THg associated with particles originating from upstream. Further, our plant Hg^0 flux measurements suggest that wetland plants can effectively release gaseous Hg^0 to a significant extent. Increased tidal wetlands in the central delta should increase this process; however it is yet unclear how this increased Hg^0 flux associated with wetland plants will affect the net Hg cycle in this region.

B) Changing hydrology and salinity in the central delta that would decrease invasive SAV

Our results indicate that sediments associated with dense SAV are comparatively active areas for MeHg production. Thus, steps taken that would decrease the density of Egeria would likely decrease the extent of MeHg production in the sediments associated with these areas.

Our studies of plant-Hg interactions suggest that submerged aquatic vegetation (SAV), such as *Egeria*, serves as a quantitatively small reservoir of total-Hg, compared to the sediments. However, where plant biomass is dense, this may still represent an important component of the local Hg cycle, as plant material represents a very labile and temporally dynamic substrate (compared to sediment), which is intimately a part of the existing epiphytic based food web. As a result, a high percentage of reactive Hg(II) may be readily taken up and/or released to the surrounding water as part of the seasonal growth and senescent cycles associated with aquatic plants. While the extent to which this

occurs and is not fully understood, management actions that would significantly increase or decrease SAV biomass density, would presumably also increase or decrease the importance of this Hg reservoir with respect to regional/local Hg cycling.

Changes in SAV densities will alter food web structure in the central delta, but based on our data it may have limited impacts on regional differences in bioaccumulation of mercury in the open water food web, assuming aqueous MeHg concentrations remain the same or decline. Our results suggest that biodilution by epiphytic material in SAV could mitigate mercury bioaccumulation on a local level. Thus, a decrease in SAV density could potentially remove any local biodilution effect.

A management action being considered for the Flooded Islands project would be to raising salinity (and/or turbity) in FT. Based on our plant decay experiments salinity alone appeared to have little impact on both SAV decay or THg release; however, Hg cycling through *Egeria* could be indirectly impacted if nitrogen pools were altered, as plant C/N ratios appear to be more directly linked to differences in plant decay rates among regions. However, it is currently unclear how these changes in plant-Hg dynamics would ultimately impact Hg accumulation in fish.

We also demonstrated appreciable differences between phytoplankton species Hg uptake kinetics. So management decisions that lead to appreciable shifts in phytoplankton species composition, such as those that might be expected by manipulations of salinity regimes, could potentially affect the trophic transfer of Hg into bulk phytoplankton (into the food web base).

C) Increased floodplain habitat in the tributaries

The creation of new floodplain habitat will likely promote MeHg production on those lands during periods of inundation, as suggested by the sediment MeHg production and concentration data. However, it is unclear how effectively this pool of sediment MeHg is able to be transferred to the overlying water, as there was very little difference observed in water column MeHg concentrations above, within or downstream of the floodplain, until the floodplain became hydrologically disconnected from the river. We did find elevated aqueous MeHg and MeHg in pelagic biota downstream of flooded habitats that increased with duration of flooding. However, further bioenergetic modeling is required to determine if elevated fish tissue concentrations pose any risk to wildlife that utilize the floodplain or consume fish coming off the floodplain. Dr. Stewart will include in the Final Report an assessment of Hg-biodynamics in striped bass consuming floodplain reared salmon to better assess potential downstream impacts. Further, we currently have no estimates of a) benthic MeHg flux associated with floodplain sediments, b) water column MeHg photodegradation rates (these should be high in shallow water floodplain type environments), c) feeding rates for floodplain biota, nor c) mass flux from the floodplain to the down-stream environment. So the net impact of increased floodplain habitat on the larger system remains unknown, but should be an area of active future research.

V. Acknowledgements

The authors would like to thank the following group of people who's professionalism and energy significantly contributed to this scientific program: J.L. Agee, L.H. Kieu, J.R. Flanders, H.A. Harms, S. Becking, N. Ladizinski, B. Topping, A. Jew, S. Wren, E. Warren, A. Lorenzi, K. Sigler, M-N. Croteau, F. Parchaso, E. Moon, and B. Richards at USGS (Menlo Park, CA) for field and laboratory analytical support; S. PalmaM. Stepanova and E. Freimuth at Stony Brook Univ. (Stony Brook, NY) for laboratory analytical support; C. Miller, A. Heyes and E.H. Kim at the Univ. of MD (Solomons, MD) for field and laboratory analytical support; G. Bernier at the Univ. of CT for analytical support; S. Peters and J. Wollenberg (Lehigh Univ., PA) for providing Phase III Hg^0 flux measurements; C. Jeffres, E. Buckland, P. Crain and J. Mount at UC Davis (Davis, CA) for field and logistical support; D. Barfuss at GA State Univ. (Atlanta, GA) and NIEHS (Project # RO-1 ES05980 granted to D. Barfuss) for supplying radiotracer $^{203}Hg(II)$ for Hg-cycling studies; and R. Cooper, J. Buck and J. Marty at The Nature Conservancy (Galt, CA) for access to and logistical support at the Cosumnes River Preserve site.

VI. Literature Cited

Alpers, C.N., Hunerlach, M.P., Marvin-DiPasquale, M.C., Antweiler, R.C., Lasorsa, B.K., De Wild, J.F., and Synder, N.P., 2006, Geochemical Data for Mercury, Methylmercury, and Other Constituents in Sediments from Englebright Lake, California, 2002: U.S. Geological Survey, Data Series Report 151, 107 p.

CALFED Bay-Delta Program. 2000. Ecosystem Restoration Program Plan: Strategic Plan for Ecosystem Restoration. Sacramento, California.

Conaway, C. H., S. S., R. P. Mason, and A. R. Flegal, 2003, Mercury speciation in the San Francisco Bay Delta: Marine Chemistry, v. 80, p. 199-225.

Davis, J.A., D. Yee, J.N. Collins, S.E. Schwarzbach, and S.N. Luoma. 2003. Potential for Increased Mercury Accumulation in the Estuary Food Web, In: Larry R. Brown, editor. Issues in San Francisco Estuary Tidal Wetlands Restoration. San Francisco Estuary and Watershed Science. Vol. 1, Issue 1, Article 4.. On-line at: http://repositories.cdlib.org/jmie/sfews/vol1/iss1/art4

Davis, J.A., B.K. Greenfield, G. Ichikawa, and M. Stephehson. 2004. Mercury in Sport Fish From the Delta Region (TASK 2A). Final Report Submitted to the CALFED Bay-Delta Program for the Project: An Assessment of the Ecological and Human Health Impacts of Mercury in the Bay-Delta Watershed. On-line at: http://loer.tamug.tamu.edu/calfed/FinalReports.htm

Gill, G.A., Bloom, N.S., Capellino, S., Driscoll, C.T., Dobbs, C., McShae, L., Mason, R., and Rudd, J.W.M., 1999, Sediment-Water fluxes of mercury in Lavaca Bay, Texas: Environ. Sci. Technol., v. 33, p. 663-669.

Gill, G., M. Stephenson, K. Coale, C. Foe, and M. Marvin-DiPasquale. 2002. Conceptual Model and Working Hypotheses of Mercury Cycling and Transport in the Bay-Delta Ecosystem and its Tributaries. Report Submitted to the CALFED Bay-Delta Program for the Project: An Assessment of the Ecological and Human Health Impacts of Mercury in the Bay-Delta Watershed. On-line at: http://loer.tamug.tamu.edu/calfed/DraftReports.htm

Kuwabara, J. S., M. Marvin-DiPasquale, W. Praskins, E. Byron, B. R. Topping, J. L. Carter, S. V. Fend, F. Parchaso, D. P. Krabbenhoft, and M. S. Gustin, 2002, Flux of Dissolved Mercury Across the Sediment-water Interface in Lahontan Reservoir, Navada, U.S. Geological Survey, Water Resources Investigations Report 02-4138.

Kuwabara, J. S., C. N. Alpers, M. Marvin-DiPasquale, B. R. Topping, J. L. Carter, A. R. Stewart, S. V. Fend, F. Parchaso, G. E. Moon, and D. E. Krabbenhoft, 2003, Sediment-Water Interactions Affecting Dissolved-mercury Distributions in Camp Far West Reservoir, California, U.S. Geological Survey. Water Resources Investigations Report 03-4140.

Marvin-DiPasquale, M. and J.L. Agee. 2003. Microbial mercury cycling in sediments of the San Francisco Bay-Delta. Estuaries 26(6): 1517-1528.

Pickhardt, P.C., Stepanova, M., and Fisher, N.S., 2006, Contrasting uptake routes and tissue distributions of inorganic and methylmercury in mosquitofish (*Gambusia affinis*) and redear sunfish (*Lepomis microlophus*): Environmental Toxicology and Chemistry, v. 25, no. 8, p. 2132–2142.

Pickhardt, P.C., and Fisher, N.S., 2007, Accumulation of inorganic and monomethylmercury by freshwater phytoplankton in two contrasting water bodies: Environmental Science and Technology, v. 41, p. 125-141.

Slotton D.G., T.H. Suchanek, and S.M. Ayer. 2000. Delta wetlands restoration and the mercury question: Year 2 findings of the CALFED UC Davis delta mercury study. IEP Newsletter 13: 34-44.

Slotton, D.G., S.M. Ayers, T.H. Suchanek, R.D. Weyand, A.M. Liston, C. Asher, D.C. Nelson, and B. Johnson. 2002. Effects of wetland restoration on the production and bioaccumulation of methylmercury in the Sacramento-San Joaquin Delta, California. Draft Final Report to the California Bay-Delta Authority. 49 pp.. On-line at: http://loer.tamug.tamu.edu/calfed/DraftReports.htm

Topping, B. R., J. S. Kuwabara, M. C. Marvin-Dipasquale, J. L. Agee, L. H. Kieu, F. Parchaso, S. W. Hager, C. B. Lopez, and D. P. Krabbenhoft, 2004, Sediment Remobilization of Mercury in South San Francisco Bay, California. U.S. Geological Survey, Scientific Investigations Report 2004-5196.

Weiner, J.G., C.C. Gilmour, and D.P. Krabbenhoft. 2003. Mercury Strategy for the Bay-Delta Ecosystem: A Unifying Framework for Science, Adaptive Management, and Ecological Restoration. Final Report to the California Bay Delta Authority. On-line at: http://science.calwater.ca.gov/library.shtml

Whalin, L.M., and Mason, R.P., 2006, A new method for the investigation of mercury redox chemistry in natural waters utilizing deflatable Teflon(R) bags and additions of isotopically labeled mercury: Analytica Chimica Acta, v. 558, no. 1-2, p. 211-221.

Yee, D., J. Collins, L. Grenier, J. Takekawa, S. Schwarzbach, M. Marvin-DiPasquale, D. Krabbenhoft, and J. Evens. 2005. Mercury and Methylmercury Processes in North San Francisco Bay Tidal Wetland Ecosystems. Annual Project Report to California Bay-Delta Authority. Sacramento. CA. Nov. 7th, 2005.

Yee, D., J. Collins, L. Grenier, J. Takekawa, S. Schwarzbach, M. Marvin-DiPasquale, D. Krabbenhoft, and J. Evens. 2007. Mercury and Methylmercury Processes in North San Francisco Bay Tidal Wetland Ecosystems. Annual Project Report to California Bay-Delta Authority. Sacramento. CA. April 4th, 2007.

Table 1. List of hypotheses.

I. Benthic conditions are more conducive for net MeHg production in CR than in the CD.

II. Physical and/or geochemical conditions mediating MeHg benthic flux to the overlying water column are more favorable in the CR than in the CD.

III. The CD has a higher net loss of MeHg from the water column, due to either microbial and/or photo-degradation, resulting in a lower net transfer of MeHg into the base of the food web in this region, compared to the CR.

IV. Regional and/or sub-habitat differences in food web dynamics, such as habitat utilization, feeding behavior, food chain length and composition, and/or species-specific Hg bioaccumulation rates at the food web base, account for the higher MeHg concentrations in CR biota compared to the CD.

V. Larval fish reared on the CRF have higher Hg levels than those reared in the Cosumnes R. proper, due to increased MeHg production and Hg exposure during seasonal inundation of the floodplain (i.e. the reservoir effect).

VI. Regional differences in plant-Hg interactions, such as Hg uptake and leaching rates by various plant species, gaseous elemental Hg efflux by plants, and/or plant community composition and density, leads to regional differences in Hg cycling pathways, MeHg production, and ultimately to differences in Hg levels in biota.

Table 2. Select sediment (*Sed*) and pore water (*Pw*) biogeochemical parameters in the Cosumnes River (CR) and Central Delta (CD) study regions. Values represent the mean of all data (regardless of habitat) collected for that parameter during the four Phase I and Phase II field collections (Dec. '03, June'04, March '05, and July '05). The standard errors of the mean are given in ( ). The statistical degrees of freedom (df) and the probability (P) that the two means are not statistically different (null hypothesis accepted) are given. Values of $P > 0.05$ are listed as non-significant (n.s.). Means comparison conducted as a Welch Modified Two-Sample t-Test.

Parameter	Units	Matrix	CR	CD	df	P
Total mercury (THg)	ng/g dry sed	*Sed*	202 (37)	130 (18)	41	*n.s.*
Total mercury (THg)	ng/L	*Pw*	7.7 (1.4)	4.2 (0.8)	30	< 0.05
Reactive inorganic Hg(II)	ng/g dry sed	*Sed*	1.70 (0.34)	0.90 (0.15)	41	< 0.05
Reactive inorganic Hg(II)	Percent (%)	*Sed*	2.44 (1.29)	1.50 (0.45)	36	*n.s.*
Methylmercury (MeHg)	ng/g dry sed	*Sed*	1.28 (0.15)	0.43 (0.05)	33	< 0.0001
Methylmercury (MeHg)	percent (%)	*Sed*	1.41 (0.43)	0.72 (0.21)	41	*n.s.*
Methylmercury (MeHg)	ng/L	*Pw*	0.73 (0.11)	0.33 (0.07)	29	< 0.005
Methylmercury (MeHg)	percent (%)	*Pw*	13.6 (2.1)	13.6 (3.6)	23	*n.s.*
^{203}Hg(II)-methylation rate constant (k_{meth}) [a]	1/day	*Sed*	8.7E-3 (3.8E-3)	5.5E-3 (2.0E-3)	45	*n.s.*
Hg(II)-methylation rate [b]	Pg/g wet sed/d	*Sed*	2.61 (0.84)	0.56 (0.11)	31	< 0.05
Me^{203}Hg-degradation rate constant (k_{deg}) [c]	1/day	*Sed*	0.31 (0.06)	0.27 (0.06)		*n.s.*
MeHg degradation rate [d]	Pg/g DRY sed/d	*Sed*	0.40 (0.16)	0.09 (0.02)	17	*n.s.*
Total mercury LOG k_d [e]	L/kg	*Sed / Pw*	4.31 (0.09)	4.57 (0.13)	30	*n.s.*
Methylmercury LOG k_d [f]	L/kg	*Sed / Pw*	3.18 (0.12)	3.27 (0.12)	33	*n.s.*
Chloride [Cl^-]	µmol/L	*Pw*	183 (32)	3274 (626)	28	< 0.0001
Sulfate [SO_4^{2-}]	µmol/L	*Pw*	22 (5)	161 (29)	30	< 0.0001
Microbial sulfate reduction	nmol/g dry sed/d	*Sed*	6.3 (1.5)	125.2 (60.5)	28	< 0.05
Acid volatile sulfur (AVS)	µmol/g dry sed	*Sed*	3.7 (0.9)	20.5 (5.7)	29	< 0.01
total reduced sulfur (TRS)	µmol/g dry sed	*Sed*	6.0 (0.9)	92.6 (18.6)	28	< 0.0001
Sulfide (HS-)	µmol/L	*Pw*	0.76 (0.15)	1.76 (0.34)	38	< 0.05
Ferrous Iron (Fe(II))	mg/L	*Pw*	6.28 (1.57)	1.82 (0.46)	35	< 0.01
Ferrous Iron (Fe(II))	mg/g dry sed	*Sed*	5.94 (0.61)	5.29 (0.66)	57	*n.s.*
Amorphous Fe(III)-oxides	mg/g dry sed	*Sed*	1.84 (0.28)	1.19 (0.30)	57	*n.s.*
Crystalline Fe(III)-oxides	mg/g dry sed	*Sed*	1.39 (0.39)	0.32 (0.11)	35	< 0.05
Organic content	% loss on ignition	*Sed*	7.3 (0.7)	14.7 (2.8)	31	< 0.05
Dissolved Organic Carbon	mg/L	*Pw*	31.6 (3.1)	46.7 (3.7)	55	< 0.005
Grain Size < 0.64 µm (silt)	percent (%)	*Sed*	42.5 (4.4)	23.1 (3.3)	55	< 0.001

[a] The ^{203}Hg(II) radiotracer derived microbial Hg(II)-methylation rate constant (k_{meth}) obtained from 4 hr amendment incubation experiments (kill corrected), using 3 g of homogenized sediment, conducted at in situ temperature.

[b] Calculated from k_{meth} and the in situ concentration of reactive inorganic mercury ($Hg(II)_R$), such that MeHg production (MP) = $Hg(II)_R - Hg(II)_R * EXP(-k_{meth} * t)$, where t = 1 day.

[c] The Me^{203}Hg radiotracer derived microbial MeHg-degradation rate constant (k_{deg}) obtained from 4 hr amendment incubation experiments (kill corrected), using 3 g of homogenized sediment, conducted at in situ temperature.

[d] Calculated from k_{deg} and the in situ concentration of pore water MeHg, such that MeHg degradation (MD) = $MeHg_{pw} - MeHg_{pw} * EXP(-k_{deg} * t)$, where t = 1 day.

[e] The THg partitioning coefficient between solid phase sediment and porewater, and is calculated as: $LOG(THg_{dw}/THg_{pw})$; where THg_{dw} = sediment THg (ng/kg) dry wt., and THg_{pw} = pore water THg (ng/L).

[f] The MeHg partitioning coefficient between solid phase sediment and porewater, and is calculated as: $LOG(MeHg_{dw}/MeHg_{pw})$; where $MeHg_{dw}$ = sediment MeHg (ng/kg) dry wt., and $MeHg_{pw}$ = pore water MeHg (ng/L).

Table 3. Water column Mercury fractions and ancillary parameters for the Cosumnes River (CR) and the Central Delta (CD = FT + DS) study regions. Data represents the mean ± (std. error) of all data (including all sub-habitats) collected in Dec. '03, June '04, March '05 and July '05. The statistical degrees of freedom (df) and the probability (P) that the two means are not statistically different (null hypothesis accepted) are given. Means comparison conducted as a Welch Modified Two-Sample t-Test.

Parameter	Units	CR	CD	df	P
Total mercury (unfiltered)	ng/L	9.7 (1.4)	3.3 (0.3)	27	< 0.0001
Total mercury (filtered)	ng/L	1.6 (0.2)	1.1 (0.1)	43	< 0.05
LOG kd for Total mercury [a]	L/kg	5.5 (0.1)	5.3 (0.1)	54	*n.s.*
Methylmercury (unfiltered) [c]	ng/L	0.15 (0.02)	0.10 (0.01)	42	*n.s., (<0.06)*
Methylmercury (filtered)	ng/L	0.16 (0.03)	0.08 (0.01)	30	< 0.05
LOG kd for methylmercury [b]	L/kg	4.8 (0.2)	5.2 (0.2)	32	*n.s.*
Methylmercury (unfiltered) [c]	% of THg	2.5 (0.5)	4.6 (1.0)	50	*n.s.*
Methylmercury (filtered)	% of THg	8.8 (1.3)	8.4 (1.1)	55	*n.s.*
Total Suspended Sediment	mg/L	29.2 (7.0)	11.5 (2.8)	34	< 0.05
Dissolved Organic Carbon	mg/L	3.0 (0.3)	4.5 (0.3)	43	< 0.005

a The Total-mercury (THg) portioning coefficient between water column suspended particles and the dissolved phase. Calculated from: LOG[(UFTHg-FTHg)/(FTHg*TSS)*10^6], Where: UFTHg = unfiltered THg (ng/L); FTHg = filtered THg (ng/L); and TSS = total suspended sediment (mg/L).

b The Methylmercury (MeHg) portioning coefficient between water column suspended particles and the dissolved phase. Calculated from: LOG[(UFMeHg-FMeHg)/(FMeHg*TSS)*10^6] Where: UFMeHg = unfiltered MeHg (ng/L); FTHg = filtered MeHg (ng/L); and TSS = total suspended sediment (mg/L).

c Analysis of this parameter excludes a single outlier data point (March 2005, CR region, site SAV-1), with a UFMeHg value of 5.21 ng/L, which equaled 45.3 %MeHg (as a % of THg) for that sample.

Table 4. Assimilation efficiencies, uptake (k_u), and efflux constants (k_e) for *Daphnia pulex* consuming *Cyclotella meneghiniana* or *Chlamydomonas reinhardtii* exposed to either Hg(II) or MeHg in two natural surface waters (Data from Pickhardt & Fisher, in preparation).

Hg species	Exposure route	Water type	AE (%)	Kinetic parameters	
				k_u (l g^{-1} h^{-1})	k_e (d^{-1})
Hg(II)	Aqueous	CR	N/A	0.79 ± 0.20	0.402 ± 0.027
	Diatom diet	CR	72.6 ± 9.4	N/A	0.195 ± 0.071
	Green diet	CR	75.5 ± 3.2	N/A	0.173 ± 0.013
	Aqueous	FT	N/A	1.00 ± 0.28	0.305 ± 0.023
	Diatom diet	FT	70.9 ± 8.1	N/A	0.168 ± 0.006
	Green diet	FT	71.9 ± 1.0	N/A	0.349 ± 0.345
MeHg	Aqueous	CR	N/A	0.92 ± 0.16	0.092 ± 0.058
	Diatom diet	CR	95.4 ± 10.8	N/A	0.026 ± 0.006
	Green diet	CR	91.9 ± 5.1	N/A	0.010 ± 0.004
	Aqueous	FT	N/A	2.10 ± 0.72	-0.046 ± 0.058
	Diatom diet	FT	116.3 ± 6.6	N/A	0.009 ± 0.015
	Green diet	FT	99.8 ± 18.0	N/A	0.025 ± 0.033

Values are means ± 1 SE.

Table 5. Life history/reproductive results from life table experiments with *Daphnia pulex* (Table & data from Pickhardt & Fisher in preparation).

Food level (mg/L)	Treatment or parameter	Length (mm)		*R_o (ind.)		Total clutches released		Average clutch size (#)		Age at 1st clutch (d)	
		CR	FT	CR	FT	CR	FT	CR	FT	CR	FT
0.1	Control (no Hg(II))	1.84 ± 0.04	1.84 ± 0.06	25.8 ± 2.3	21.5 ± 9.2	6.2 ± 0.5	5.5 ± 0.7	4.2 ± 0.3	3.8 ± 1.2	9.8 ± 1.1	10.0 ± 0.0
0.1	Hg(II)	1.83 ± 0.03	1.79 ± 0.01	26.0 ± 2.3	12.2 ± 4.3	6.3 ± 0.5	3.0 ± 1.4	4.2 ± 0.6	4.5 ± 1.3	9.0 ± 0.0	9.0 ± 0.0
0.1	Hg/Control (%)	99.3	97.3	100.8	56.8	101.6	54.5	100	118.4	91.8	90.0
0.1	Control (no MeHg)	1.87 ± 0.05	1.87 ± 0.08	15.2 ± 5.8	17.2 ± 4.5	4.0 ± 1.2	4.4 ± 0.9	3.7 ± 0.8	3.9 ± 0.7	14.4 ± 2.9	13.0 ± 3.9
0.1	MeHg	1.84 ± 0.05	1.82 ± 0.02	14.2 ± 4.05	9.8 ± 5.3	5.2 ± 1.1	4.5 ± 1.3	2.7 ± 0.4	2.0 ± 0.8	12.0 ± 0.7	12.3 ± 1.9
0.1	MeHg/Control (%)	98.7	97.5	93.4	56.7	130.0	102.3	73.0	51.3	83.3	94.2
0.6	Control (no Hg(II))	2.50 ± 0.05	2.44 ± 0.11	114 ± 36	59.8 ± 10.7	6.8 ± 1.1	5.6 ± 0.6	16.5 ± 4.0	10.7 ± 1.8	9.0 ± 0.0	10.0 ± 0.7
0.6	Hg(II)	2.21 ± 0.08	Dead	84.3 ± 15.9	18.6 ± 16.3	7.5 ± 0.6	2.2 ± 1.6	11.2 ± 1.3	8.0 ± 1.9	9.0 ± 0.0	10.2 ± 1.6
0.6	Hg/Control (%)	88.6	0	73.8	31.1	110.3	39.3	67.9	74.8	100	102.0
0.6	Control (no MeHg)	2.48 ± 0.06	2.47 ± 0.12	62.8 ± 29.6	85.4 ± 7.3	5.2 ± 2.1	6.4 ± 0.6	11.5 ± 2.3	13.4 ± 1.1	10.8 ± 0.5	11.0 ± 1.7
0.6	MeHg	2.52 ± 0.03	2.54 ± 0.07	74.2 ± 18.9	78.8 ± 27.2	6.2 ± 0.5	6.8 ± 1.6	12.0 ± 3.0	11.3 ± 2.4	10.0 ± 0.0	10.0 ± 0.0
0.6	MeHg/Control (%)	101.4	102.8	118.1	92.3	119.2	106.3	104.3	84.3	92.6	90.9

Values are means ± 1 SD. * Cumulative reproduction across all clutches per *Daphnia pulex* is represented by R_o.

Table 6. Assimilation efficiencies (AE) and uptake (k_u) and efflux (k_e) rate constants of inorganic (HgI) and methyl (MeHg) mercury from dietary and aqueous exposures in mosquitofish and redear sunfish in Cosumnes River water and Frank's Tract water (from Pickhardt et al. Environ. Toxicol. Chem., 2006).

Fish species	Hg species	Exposure route	Water type	AE (%)	% of Hg burden attributable to exposure route	Kinetic parameters	
						k_u (l g^{-1} d^{-1})	k_e (d^{-1})
Mosquitofish	Hg_I	Aqueous	CR		27.2	0.078 (0.008)	0.021 (0.021)
		Diet	CR	41.7 (15.3)	72.8		0.025 (0.023)
		Aqueous	FT		12.2	0.052 (0.017)	0.042 (0.025)
		Diet	FT	51.3 (26.6)	87.8		0.033 (0.032)
	MeHg	Aqueous	CR		0.8	0.338 (0.092)*	0.018 (0.006)
		Diet	CR	89.6 (8.2)	99.2		0.016 (0.003)
		Aqueous	FT		0.7	0.185 (0.020)*	0.019 (0.006)
		Diet	FT	94.1 (3.0)	99.3		0.016 (0.002)
Redear sunfish	Hg_I	Aqueous	CR		59.6	0.051 (0.009)*	0.030 (0.006)
		Diet	CR	8.5 (9.0)	40.4		0.003 (0.003)
		Aqueous	FT		45.5	0.038 (0.008)*	0.035 (0.01)
		Diet	FT	9.8 (3.8)	54.5		0.007 (0.007)
	MeHg	Aqueous	CR		1.6	1.28 (0.81)*	0.021 (0.005)
		Diet	CR	91.4 (2.2)*	98.4		0.018 (0.007)
		Aqueous	FT		0.6	0.454 (0.180)*	0.021 (0.006)
		Diet	FT	85.8 (2.2)*	99.4		0.015 (0.002)

[a] Mean values are presented for each parameter, followed by standard deviations in parentheses (except for % of Hg burden). Significant differences between Cosumnes River (CR) and Frank's Tract (FT) waters (Sacramento and Contra Costa counties, respectively, CA, USA) for a given Hg species and exposure route from one-way analysis of variance with $p \leq 0.05$ are designated with an asterisk. Hg_I = inorganic Hg; MeHg = methylmercury.

Table 7. Factor differences in mercury concentrations for species pairs and food webs from the CR and CD region.

	Epiphyte[1] MeHg ng/g dry wt.	Amphipods[2] MeHg ng/g dry wt.	SPM[1] MeHg ng/g wet wt.	Bulk[2] Zooplankton MeHg ng/g dry wt.	Fish[3] Muscle MeHg ng/g dry wt.	Food web[4] MeHg ng/g dry wt.	Predicted[5] predator fish MeHg ng/g dry wt.	350mm[6] Largemouth bass Hg µg/g wet wt.
CR	39	77	0.13	131	684	518	813	0.9
FT	16	41	0.12	61	237	200	274	0.3
Factor difference	**2.5**	**1.86**	**1.08**	**2.15**	**2.89**	**2.58**	**2.97**	**3.00**
	2-way ANOVA (region and sample date as factors) F=12.5 *p=0.024*	2-way ANOVA (region and sample date as factors) F=9.9 *p=0.004*	2-way ANOVA (region and sample date as factors) F=4.3 *p=0.52*	2-way ANOVA (region and sample date as factors) F=7.0 *p=0.013*	ANCOVA F=119 *p<0.0001*	ANCOVA F=84 *p<0.0001*		

[1] Least square means for epiphyte and suspended particulate matter (SPM) were based on samples collected in March and July 2005. Epiphytes were collected from Teflon sheets deployed for one month in the field.
[2] Least square means for bulk zooplankton and amphipod samples collected in December 2003, June 2004, March and July 2005.
[3] Adjusted ($\delta^{15}N$) least square means for both Redear sunfish and largemouth bass
[4] Adjusted ($\delta^{15}N$) least square means for selected species food web (see **Fig. 9**)
[5] Mercury concentrations for a predator fish (i.e. $\delta^{15}N = 12$, roughly equivalent to a 350 mm largemouth bass) predicted using linear relationships for regional food webs in **Fig. 9**.
[6] Source Jay Davis, SFEI.

Table 8. Estimates of partitioning of mercury among dissolved and epiphytic phases in Cosumnes River (CR) and Central Delta (CD) SAV habitat in March and July 2005. Values are means ± total standard deviation (propagated through all calculations).

Region	Date	Epiphyte MeHg ng/L	Suspended Particulate MeHg ng/L	Dissolved MeHg ng/L	Epiphyte MeHg %	Dissolved MeHg %
CR	March '05	0.26 ± 0.21	0.32 ± 0.33	0.25 ± 0.14	32 ± 27	30 ± 20
CD	March '05	0.77 ± 0.79	0.16 ± 0.13	0.085 ± 0.007	76 ± 91	8 ± 5.4
CR	July '05	1.0 ± 0.87	0.12 ± 0.05	0.21 ± 0.11	76 ± 80	15 ± 13
CD	July '05	1.3 ± 1.5	0.24 ± 0.15	0.16 ± 0.057	77 ± 137	9 ± 13

Table 9. Summary statistics for sediment mercury parameters in the Cosumnes River (CR) main stem (including all data collected during Phase I, II, and III; all sub-habitats; n = 33) and the Cosumnes River Flood Plain (CRF; Phase III sampling; n = 11). CRF data includes the Upper and Lower Floodplain (UF and LF) sampling sites shown on Fig. 1E. Values represent the mean ± (std. error). The statistical degrees of freedom (df) and the probability (P) that the two means are not statistically different (null hypothesis accepted) are given. Means comparison conducted as a Welch Modified Two-Sample t-Test.

Parameter	Units	CR	CRF	df	P
^{203}Hg(II)-methylation rate constant (k_{meth}) [a]	1/day	0.8E-2 (0.4E-2)	2.0E-2 (0.6E-2)	18	*n.s.*
MeHg Product. Rate	pg/g wet/d	2.6 (0.8)	9.4 (2.6)	12	< 0.05
Methylmercury	ng/g (dry)	1.3 (0.2)	8.5 (1.8)	10	< 0.005
Methylmercury	% of THg	1.4 (0.4)	6.0 (1.2)	12	< 0.005
Reactive Hg(II)	ng/g (dry)	1.9 (0.4)	1.8 (0.8)	15	*n.s.*
Reactive Hg(II)	% of THg	2.4 (1.1)	1.1 (0.4)	36	*n.s.*
Total Mercury (THg)	ng/g (dry)	205 (36)	147 (10)	33	*n.s.*

[a] As per Table 1.

Appendix I - Products to date

I. Publications

A. Papers Published

Pickhardt, P.C., and Fisher, N.S., 2007, Accumulation of inorganic and monomethylmercury by freshwater phytoplankton in two contrasting water bodies: Environmental Science and Technology, v. 41, p. 125-141.

Pickhardt, P.C., Stepanova, M., and Fisher, N.S., 2006, Contrasting uptake routes and tissue distributions of inorganic and methylmercury in mosquitofish (Gambusia affinis) and redear sunfish (Lepomis microlophus): Environmental Toxicology and Chemistry, v. 25, no. 8, p. 2132–2142.

B. Online Reports

Marvin-DiPasquale, M., Stewart, A.R., Fisher, N.S., Pickhardt, P., Mason, R.P., Heyes, A. and L. Windham-Meyer. 2005a. Evaluation Of Mercury Transformations and Trophic Transfer in the San Francisco Bay/Delta: Identifying Critical Processes for the Ecosystem Restoration Program: Annual Report of Progress for Project # ERP-02-P40. Submitted to the California Bay Delta Authority (CBDA). Sacramento, CA. November 7th, 2005. Available online at: *http://calwater.ca.gov/Programs/EcosystemRestoration/Ecosystem_MercuryAnnualReport2005.asp*

C. Manuscripts Submitted

D. Manuscripts In Preparation

Pickhardt, P. C., E. Freimuth, and N. S. Fisher. (*in prep*). The accumulation and sub-lethal effects of organic and inorganic mercury on *Daphnia pulex* in two natural surface waters.. Submission goal: Environmental Toxicology & Chemistry.

II. Public Presentations

A. Oral Presentations

Croteau, M.-N., A.R. Stewart, and S.N. Luoma. 2005. *Trophic enrichment of trace elements in aquatic food webs: A paradigm shift from the organics world*. Oral Presentation given at the 26th Annual Meeting of the Society of Environmental Toxicology and Chemistry (SETAC), Baltimore MD, November 13-17, 2005.

Fisher, Nick (SUNY Stony Brook). Radiotracer studies of mercury and methylmercury bioaccumulation in aquatic food chains in the California Bay-Delta system. May 24th, 2005. Presented in Monaco as part of the IAEA Coordinated Research Program on Nuclear Applications to Determine Bioaccumulation Parameters and Processes used for Establishing Coastal Zone Monitoring and Management Criteria.

Lorenzi, A.H. and A.R. Stewart, A.R. 2006. Variations in Methylmercury Accumulation by Epiphytes Associated with Submerged Aquatic Vegetation (SAV) in the San Francisco Bay Delta Region. Abstract submitted for the SETAC North America 27th Annual Meeting November 5-9, 2006; Montréal, Québec, Canada

Mark Marvin-DiPasquale, M. (USGS, Menlo Park, CA). Mercury Cycling Concepts Important in Adaptive Management of Wetland Restoration. Presented at the 3rd Annual CALFED Bay-Delta Science Conference, Oct. 2004.

Mark Marvin-DiPasquale, M. (USGS, Menlo Park, CA). Wetland Restoration and the Potential for Enhanced Mercury Methylation. Invited Presentation at UC Berkely. 29 October 2004.

Mark Marvin-DiPasquale, M. (USGS, Menlo Park, CA). Wetland Restoration and the Potential for Enhanced Mercury Methylation", Invited Presentation at Univ. of MD, Chesapeake Biological Lab, December 2, 2004.

Mark Marvin-DiPasquale, M. (USGS, Menlo Park, CA). Toxic Mercury in Aquatic Ecosystems: Why Quality Trumps Quantity. Presented at USGS Menlo Park (CA) as part of the USGS Western Region Public Lecture Series. September 29th, 2005.

Marvin DiPasquale, Mark and Robin Stewart (USGS, Menlo Park, CA). Evaluation of Mercury Transformations and Trophic Transfer in the San Francisco Bay/Delta. October 12th, 2005. Joint presentation to staff and invited guests of The Nature Conservancy, providing an overview of the CALFED SBF-Delta Hg project, as part of our agreement with that group, which allows us access to the Cosumnes Nature Preserve property.

Marvin-DiPasquale, M., R. Stewart, R. Mason, N. Fisher, P. Pickhardt, L. Windham. 2005c. Evaluation of Mercury Transformations and Trophic Transfer in the San Francisco Bay/Delta. Oral Presentation given at the California Bay-Delta Authority sponsored Mercury Project Review, Sacramento CA. Nov. 29 – Dec. 1, 2005.

Marvin-DiPasquale, M., B.D. Hall, J.R. Flanders, N. Ladizinski1, J.L. Agee, L.H. Kieu, L. Windham. 2006a. Ecosystem Investigations of Benthic Methylmercury Production: A Tin-Reduction Approach for Assessing the Inorganic Mercury Pool Available for Methylation. Oral presentation abstract for the *8th International Conference on Mercury as a Global Pollutant*. August 6-11th, 2006. Madison WI.

Marvin-DiPasquale, M., J.L. Agee, N. Ladizinsky, L. Windham-Myer, S. Wren, D. Yee, J. Collins, S. Olund, D. Krabbenhoft, R. Mason, A. Heyes. 2006b. Controls on Mercury-Methylation in Sediments From Freshwater, Delta, and Salt-Marsh Regions of the San Francisco Bay Watershed. Abstract for CBDA Science Conference, October 23-26, 2006; Sacramento, CA.

Mason, RP, Laurier FJ and Whalin, LM. The oxidation and reduction of mercury in surface waters: Studies using stable isotopes. Eos Trans. AGU, 87(36), Ocean Sci. Meet. Hawaii, February 2006, Suppl., Abstract OS16M-25

Pickhardt, Paul (SUNY Stony Brook). Accumulation dynamics of mercury in freshwater plankton. February, 24th, 2004. Presented as part of Stony Brook University's Center for Environmental Molecular Sciences seminar series.

Pickhardt, Paul (SUNY Stony Brook). Mercury cycling in aquatic ecosystems: An important link to human exposures. July 13th, 2004. Presented as part of Stony Brook University's Center for Environmental Molecular Sciences summer lecture series for its "REU" students.

Pickhardt, P. and N. Fisher. 2005. *Accumulation of inorganic and organic mercury in phytoplankton and the subsequent trophic transfer to crustaceans.* Oral Presentation given at the 26th Annual Meeting of the Society of Environmental Toxicology and Chemistry (SETAC), Baltimore MD, November 13-17, 2005. Available online at: http://abstracts.co.allenpress.com/pweb/setac2005/document/?ID=56234

Pickhardt, P (SUNY Stony Brook). Mercury Accumulation Dynamics in Biota from the San Francisco Bay Delta System. February, 17th, 2006. Marine Sciences Research Center Friday Seminar Series, Stony Brook University.

Pickhardt, P.C. and N.S. Fisher. 2006. Uptake of mercury by freshwater phytoplankton and trophic transfer to crustacean grazers. Oral presentation abstract for the *8th International Conference on Mercury as a Global Pollutant*. August 6-11th, 2006. Madison WI.

Stepanova, Maria (SUNY Stony Brook). Uptake and Retention of Dietary and Aqueous Methyl and Inorganic Mercury by Two Fish Species. October 3rd, 2005. Presented at Stony Brook University as part of the requirements for completion of her M.S. degree, associated with this research.

Stewart, A.R., M.-N. Croteau, S. Luoma. 2005b. *Trophic enrichment of trace elements in aquatic food webs: A paradigm shift from the organics world.* Oral Presentation given at the 26th Annual Meeting of the Society of Environmental Toxicology and Chemistry (SETAC), Baltimore MD, November 13-17, 2005. Available online at: http://abstracts.co.allenpress.com/pweb/setac2005/document/?ID=56969

Stewart, A.R., M. Marvin-DiPasquale, A.H. Lorenzi, C.A. Jeffres, and E.M. Buckland. 2006b. Mercury accumulation in juvenile Chinook salmon caged on the Cosumnes floodplain and Cosumnes River: Reconciling mercury uptake and growth dilution. Abstract for CBDA Science Conference, October 23-26, 2006; Sacramento, CA

Stewart, A.R., K. Higgins, M. Marvin-DiPasquale, PC Pickhardt. 2006c. Processes driving methylmercury (MeHg) accumulation at the base of aquatic food webs from two regions of the San Francisco Bay Delta. Abstract submitted for the SETAC North America 27th Annual Meeting November 5-9, 2006; Montréal, Québec, Canada

Windham, L., A. Jew and M. Marvin-Dipasquale. 2006a. The uptake, release and remobilization of mercury by wetland plants – implications for the "reactive" pool of mercury available for methylation. Poster presentation abstract for the *8th International Conference on Mercury as a Global Pollutant*. August 6-11th, 2006. Madison WI.

Windham-Myers, L., A. Jew, S.L. Wren, and M. Marvin-DiPasquale. 2006b. Plant-mercury interactions: The role of submerged and emergent macrophytes in mercury cycling of San Francisco Bay and Delta wetlands. Abstract for CBDA Science Conference, October 23-26, 2006; Sacramento, CA

B. Poster Presentations

Bernier, G., E.-H. Kim, A. Heyes, R.P. Mason, C.L. Miller, and M. Marvin-DiPasquale. 2006. The Biogeochemical Cycling and Fate of Mercury and Methylmercury in the San Francisco Delta Region . Poster presentation abstract for the *8th International Conference on Mercury as a Global Pollutant*. August 6-11th, 2006. Madison WI.

Marvin-DiPasquale, M, J.L. Agee, L.H. Kieu, N. Ladizinski, L. Windham, D. Yee, J. Collins, S. Olund, D. Krabbenhoft, R. Mason, A. Heyes, C. Miller. 2005b. Mercury-Methylation Dynamics In Sediments From Freshwater, Delta and Saltmarsh Regions of the San Francisco Bay Watershed. Poster Presentation at the California Bay-Delta Authority sponsored Mercury Project Review, Sacramento CA. Nov. 29 – Dec. 1, 2005.

Mason, R.P., A. Heyes, C.L. Miller and M. Marvin-DiPasquale. 2005. A Comparison of the Biogeochemical Cycling of Mercury and Methylmercury within Frank's Tract and the Consumes River. Poster Presentation at the California Bay-Delta Authority sponsored Mercury Project Review, Sacramento CA. Nov. 29 – Dec. 1, 2005.

Pickhardt, P., M. Stepanova, and N. Fisher. 2005. *Accumulation Dynamics of Mercury in Phytoplankton and the Subsequent Transfer to Crustaceans and Fish in Cosumnes River and Frank's Tract Water.* Poster Presentation at the California Bay-Delta Authority sponsored Mercury Project Review, Sacramento CA. Nov. 29 – Dec. 1, 2005.

Stepanova, Maria (SUNY Stony Brook). Uptake and retention of aqueous and dietary inorganic and methylmercury in two fish species. November 14-18, 2004. Presented at the SETAC World Congress conference in Portland, OR. in the special session entitled 'Metals and Bioaccumulation'.

Stewart, A.R., K. Higgins, P.C. Pickhardt, M-N. Croteau, and A. Heyes. 2006a. Trophic enrichment of methylmercury in aquatic food webs of the San Francisco Bay Delta: How important is biology and food web structure? Poster presentation abstract for the *8th International Conference on Mercury as a Global Pollutant*. August 6-11th, 2006. Madison WI.

Stewart, R., K. Sigler, P. Pickhardt, M.-N. Croteau, and A. Heyes. 2005a. *Trophic Enrichment of Methylmercury (MeHg) in Food Webs of the Tributaries and Central Delta of the San Francisco Bay Watershed*. Poster Presentation at the California Bay-Delta Authority sponsored Mercury Project Review, Sacramento CA. Nov. 29 – Dec. 1, 2005.

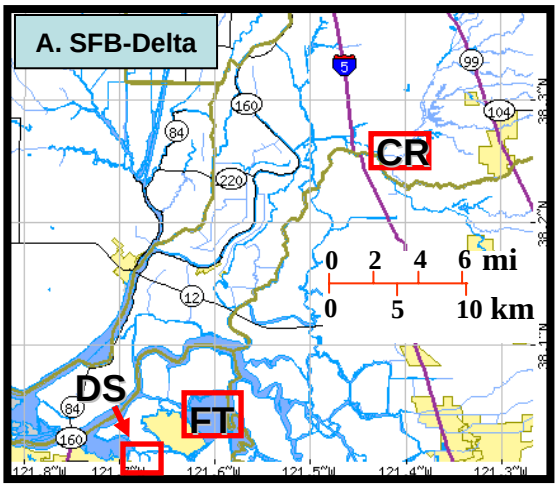
Windham, L. and M. Marvin-DiPasquale. 2005. *The Role of Submerged and Emergent Macrophytes in the Mercury Cycle of San Francisco Bay Wetlands.* Poster Presentation at the California Bay-Delta Authority sponsored Mercury Project Review, Sacramento CA. Nov. 29 – Dec. 1, 2006.

Wren, S.L., M. Marvin-DiPasquale, L. Windham-Myers, G. Aiken, N. Ladizinsky, and M. Cox. 2006. Reactive mercury studies associated with the Cosumnes River floodplain. Abstract for CBDA Science Conference, October 23-26, 2006; Sacramento, CA

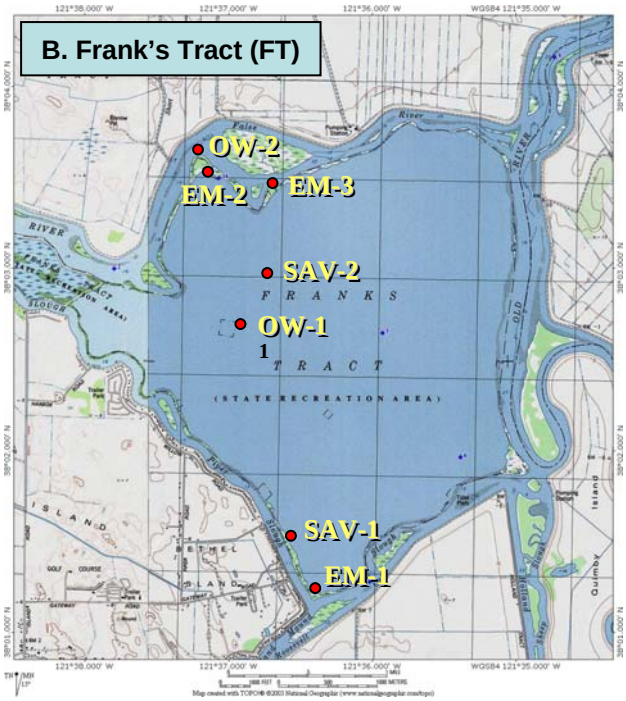
C. Other Presentations / Products

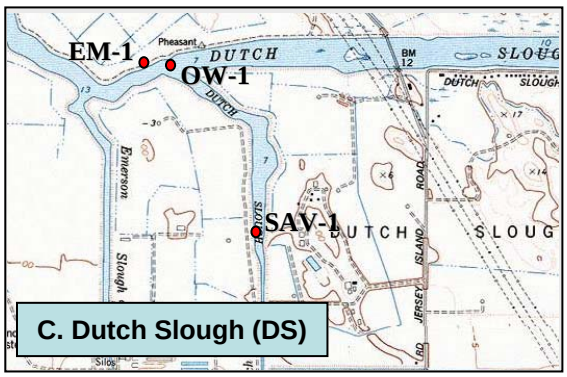
Freimuth, Erika (high school intern at SUNY Stony Brook). The Effects of Organic and Inorganic Mercury Consumption on the Population Dynamics of *Daphnia pulex* and its Implications for the Trophic Transfer of Mercury. Paper submitted November 2004 to the national InSTAR science competition, based results from her research project in Dr. Fisher's laboratory regarding the sub-lethal effects of Hg_I and MeHg on *Daphnia pulex* in Cosumnes River and Frank's Tract waters.

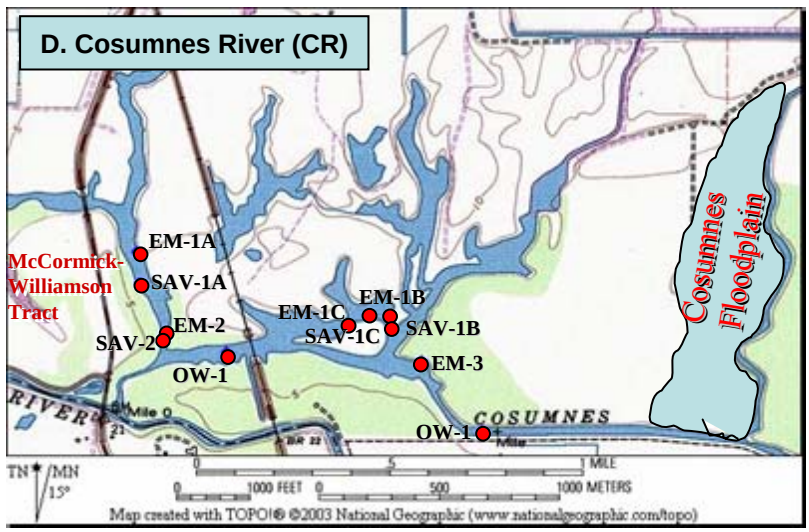
Figure 1.



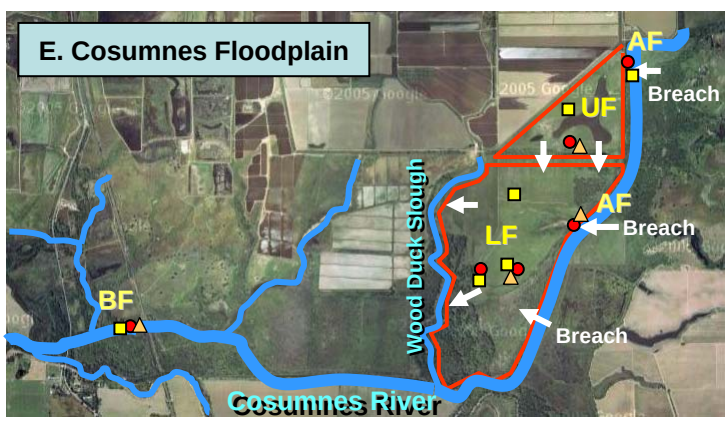
A. SFB-Delta: Map showing the three sampling regions: Frank's Tract (FT), Dutch Slough (DS) and Cosumnes River (CR).







B. Thru D. Specific sites sampled during Phase I and II, coded by habitat type: emergent marsh (EM), open water (OW) [non-vegetated], and *Egeria* dominated submerged aquatic vegetation (SAV). The DS region was only sampled during July 2005, and is included with FT data in graphical and statistical assessments of the 'Central Delta' (CD) region compared to the CR region, unless specifically indicated otherwise.



E. Cosumnes Floodplain: The map identifies the location of sediment (●) and water (□) sampling sites and caged fish experiments (△), during Phase III. Also indicated are the three sampling habitats: Above floodplain (AF), where Cosumnes River water enters the floodplain through levee breaches; Upper floodplain (UF), outlined by the large red triangle; lower floodplain (LF) outlined by the red irregular polygon; and below floodplain (BF) in the Cosumnes R. main channel. The with white arrows indicate water flow onto the floodplain through levee breaches, from UF to LF through levee breaches, and into Wood Duck Slough which reconnects back to the Cosumnes R. mainstem.

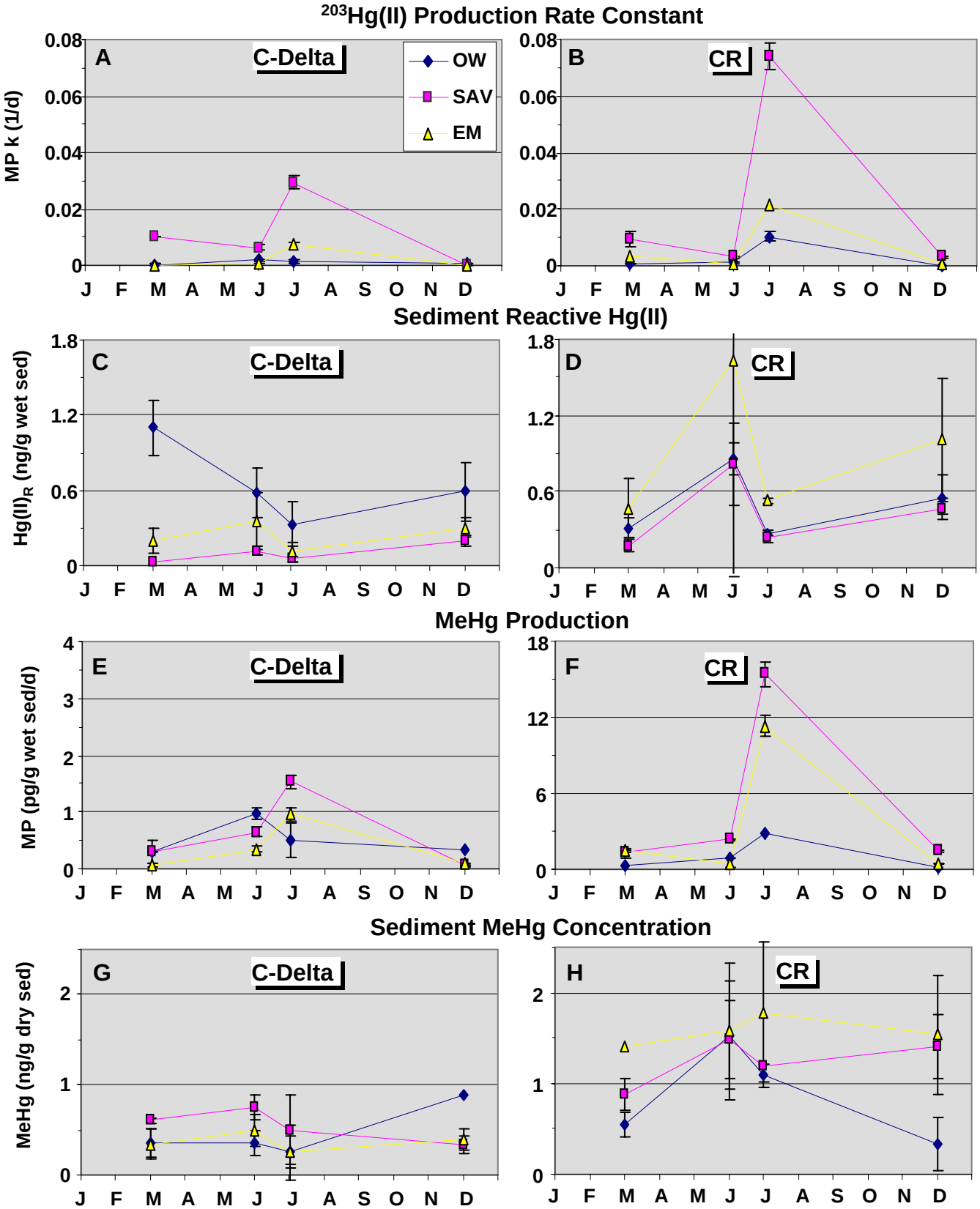


Figure 2. Composite temporal plots for the Central Delta (C-Delta) and Cosumnes R. (CR) regions by primary sub-habitats for the parameters: ^{203}Hg(II)-Methylation rate constant (A & B), Sediment reactive Hg(II) (C & D), calculated MeHg production rates (E & F), and sediment MeHg concentration (G & H). Note the difference in scales for E & F.

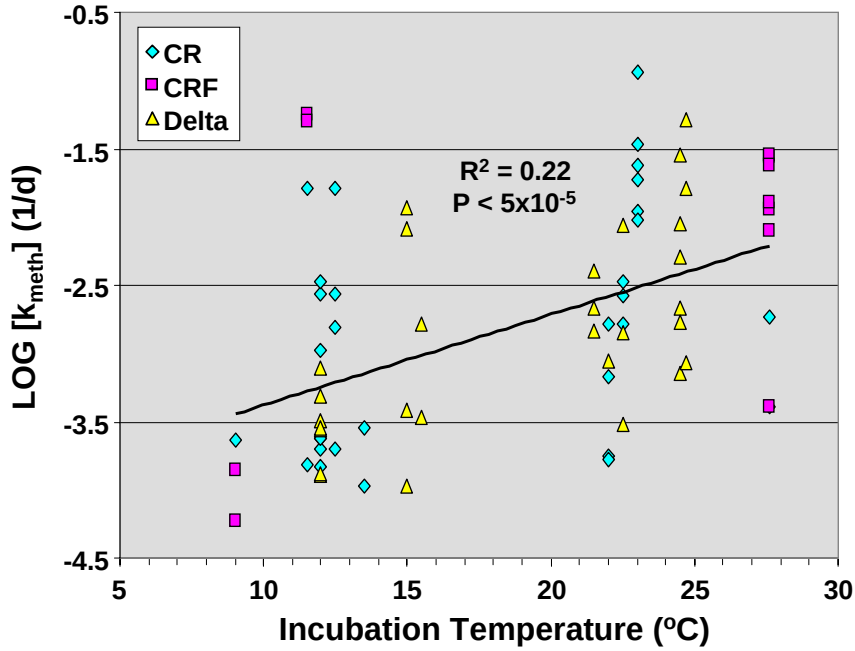


Figure 3. TheLOG transformed ^{203}Hg(II)-methylation rate constant (k_{meth}) vs incubation temperature used in radiotracer experiments. The incubation temperature was closely matched to the average in-situ field temperature (±1 ºC) measured at the time of sediment collection. The regression R^2 and probability (P) of a non-significant slope is indicated.

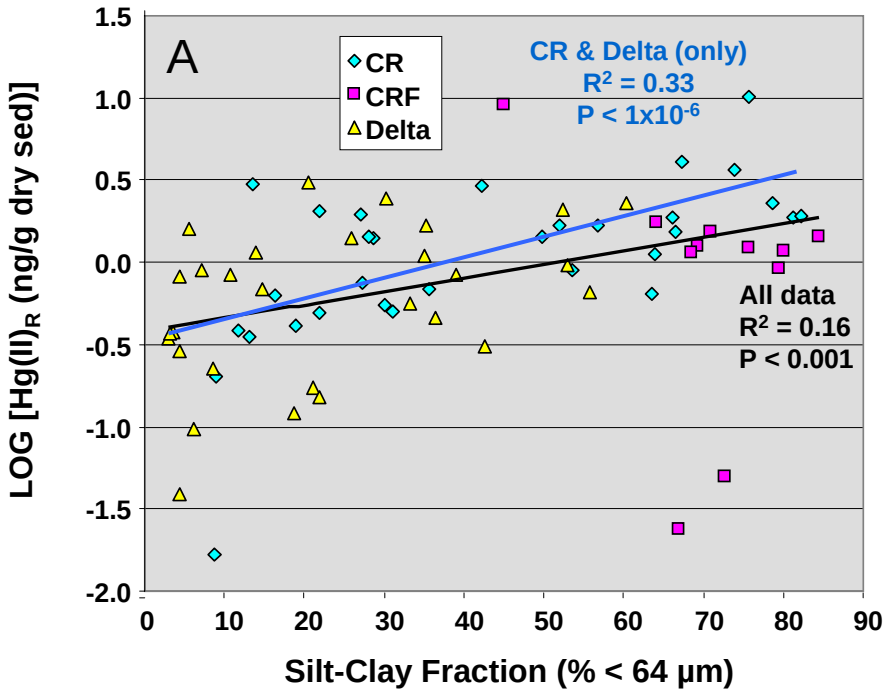


Figure 4. The reactive-mercury ($Hg(II)_R$) concentration (LOG scale) vs sediment silt-clay fraction, plotted by region (A). The two regression lines reflect the use of all data (black line) and the omission of the CR-Floodplain data (blue line). The regression R^2 and probability (P) of a non-significant slope is indicated.

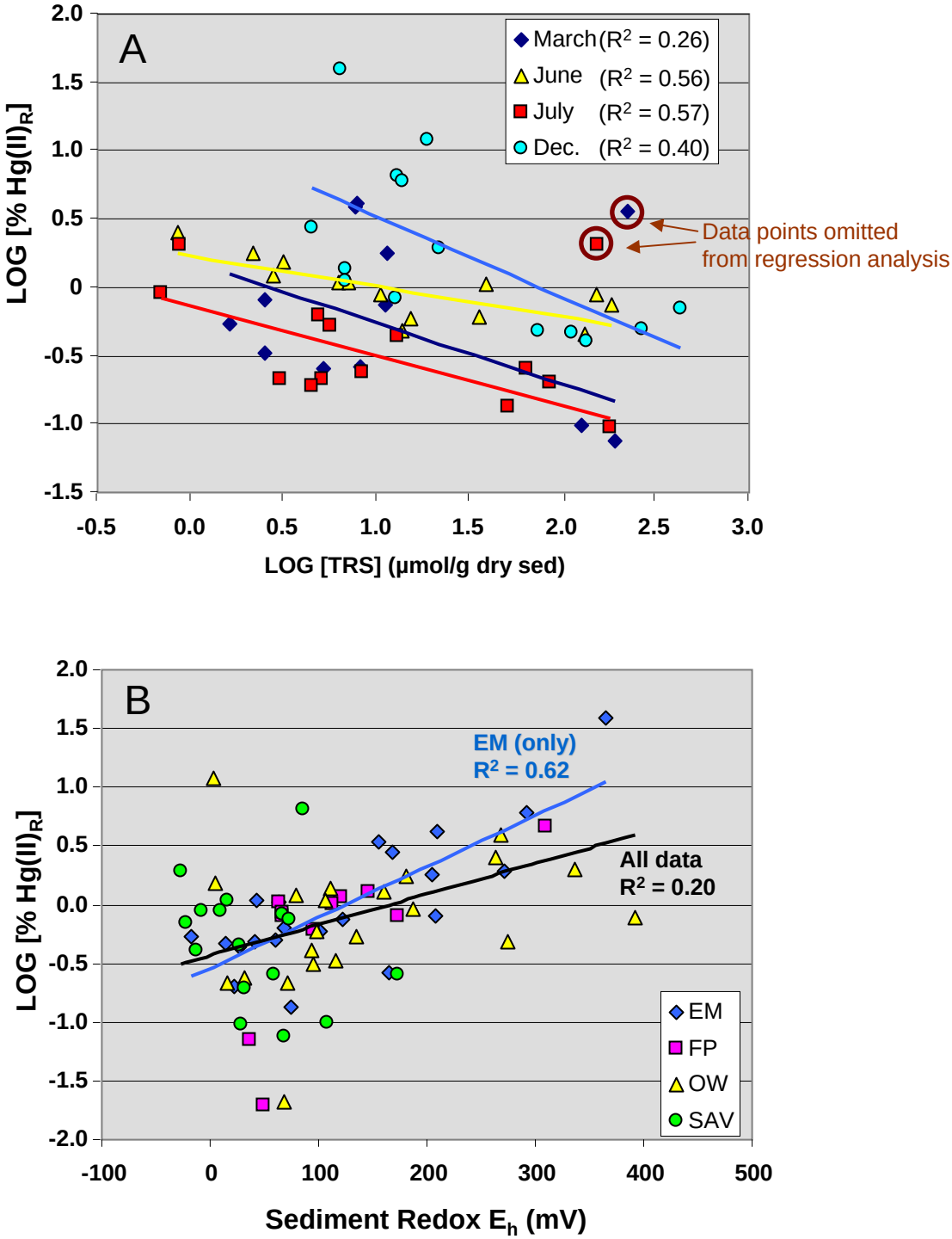


Figure 5. The percentage (%) of the total-Hg that is $Hg(II)_R$ (LOG scale) as a function of total reduced sulfur (TRS), plotted by month (CD and CR data only; no CRF data) (A). The bottom figure (B) depicts % $Hg(II)_R$ (LOG scale) as a function of sediment redox potential, plotted by habitat type. The two regression lines reflect the use of all data (black line) and the emergent marsh (EM) data only (blue line). All regression lines shown have a significant slope at the $P < 0.05$ level or lower.

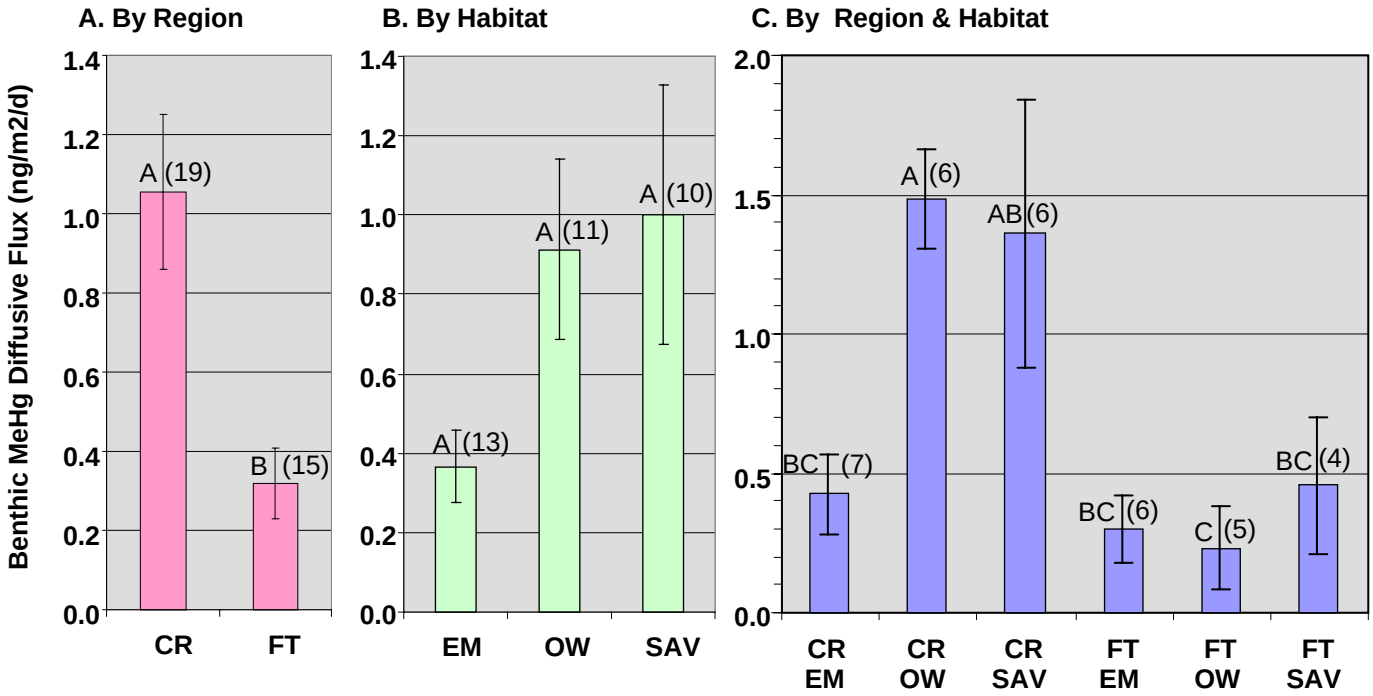


Figure 6. Calculated average methylmercury (MeHg) diffusive flux rates across the sediment/water interface by A) Region, B) Habitat type, and C) both Region & Habitat, based on data collected as part of the Dec. 2003, June 2004 and March 2005 field trips. Flux (F) for each site is calculated as: $F = D^* (\delta C/\delta x)$, where $(\delta C/\delta x)$ is the MeHg concentration gradient, calculated as the difference between the pore water MeHg concentration ($MeHg_{pw}$) and the overlying water dissolved (filtered) MeHg concentration (FMeHg), and x is assumed to be 1 cm; D = the molecular diffusion coefficient ($2x10^{-6}$ cm^2*s^{-1}) for dissolved organic matter (DOM) based on the assumption of a MeHg-DOM complex (Gill et al. 1999). As such, the magnitude of these flux calculations should be considered minimum estimates. If modeled as the neutral species MeHgCl, each average estimate would be 6.5X higher. The value (#) associated with each bar represents the number of samples, and the error bars reflect standard errors of the mean. Within a given graph, bars that share any common letter are NOT significantly different ($P < 0.05$) by the Tukey multiple means ranking method.

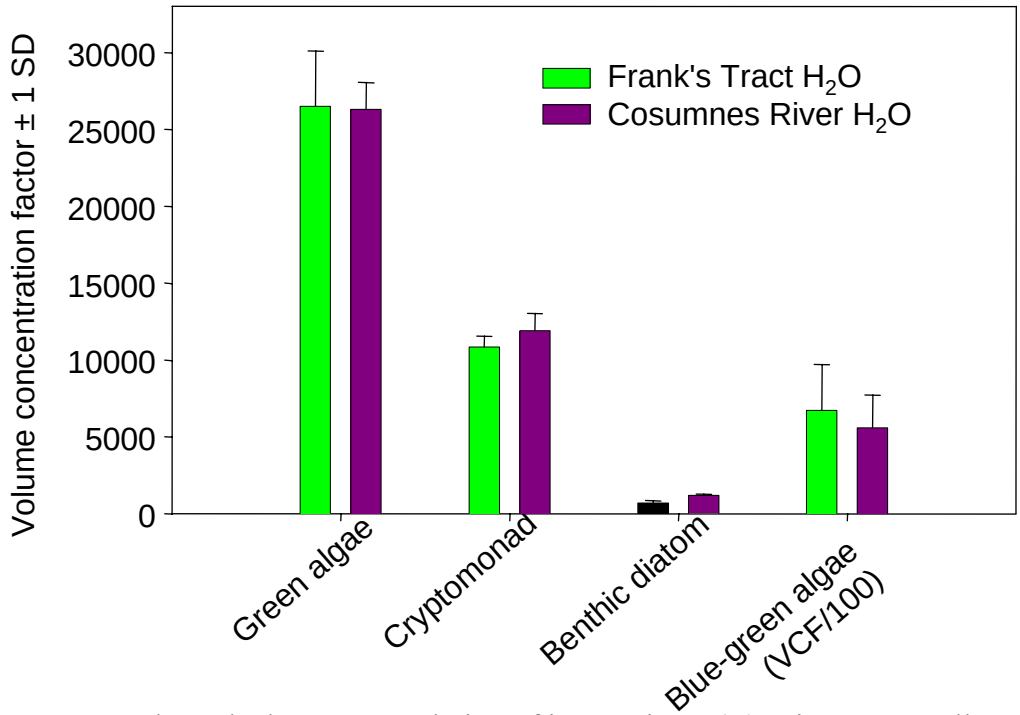


Figure 7. Phytoplankton accumulation of inorganic Hg(II) using water collected from both study regions. No significant differences in bioconcentration between the two water types was observed.

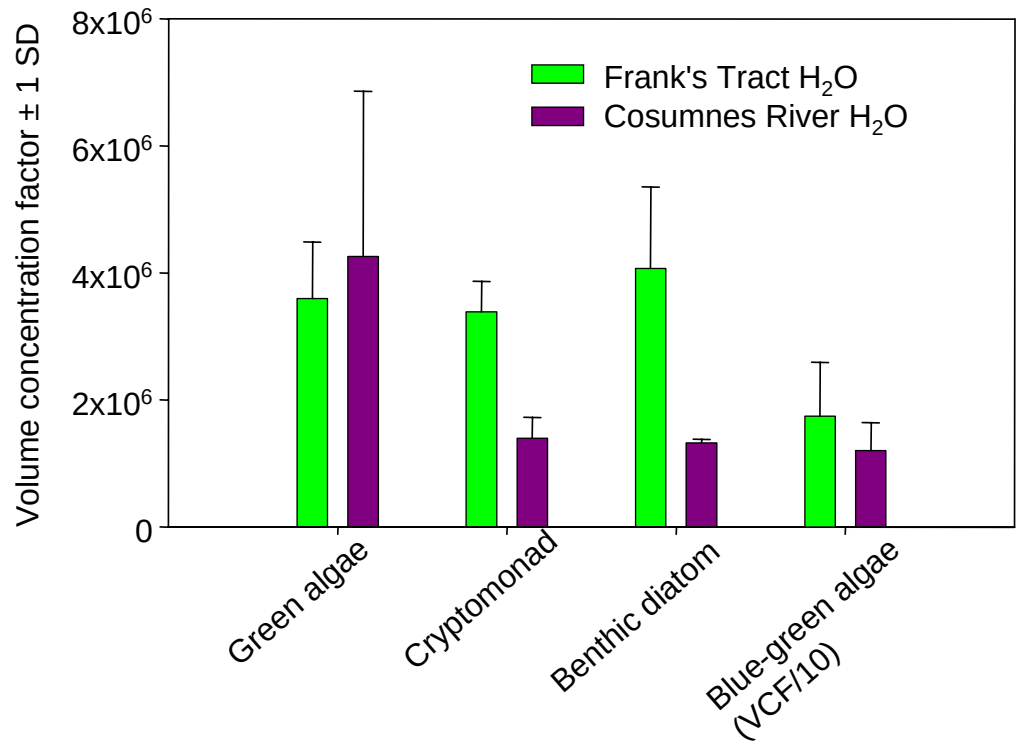


Figure 8. Phytoplankton accumulation of MeHg using water collected from both study regions. A greater bioconcentration of MeHg was found in FT water for 2 out of 4 species.

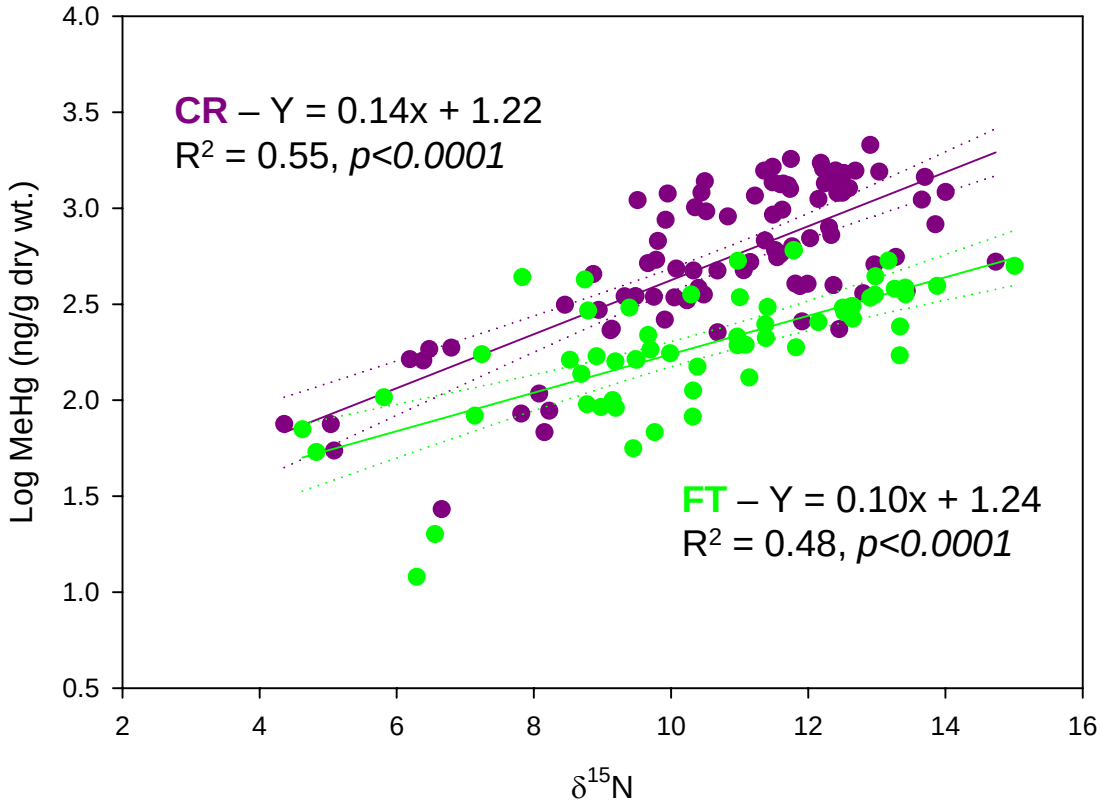


Figure 9. Trophic enrichment of MeHg in individuals of selected species from submerged aquatic vegetation (SAV) food webs from CR and FT sampled in December 2003 and June 2004. Species sampled included amphipods (*Hyalella* and *gammarid* sp.), damsel flies, redear sunfish and large mouth bass. Slopes were marginally not statistically different (FT slope was lower than CR), but intercept for CR food web was significantly higher (ANCOVA, F=83.7, *p<0.0001*)

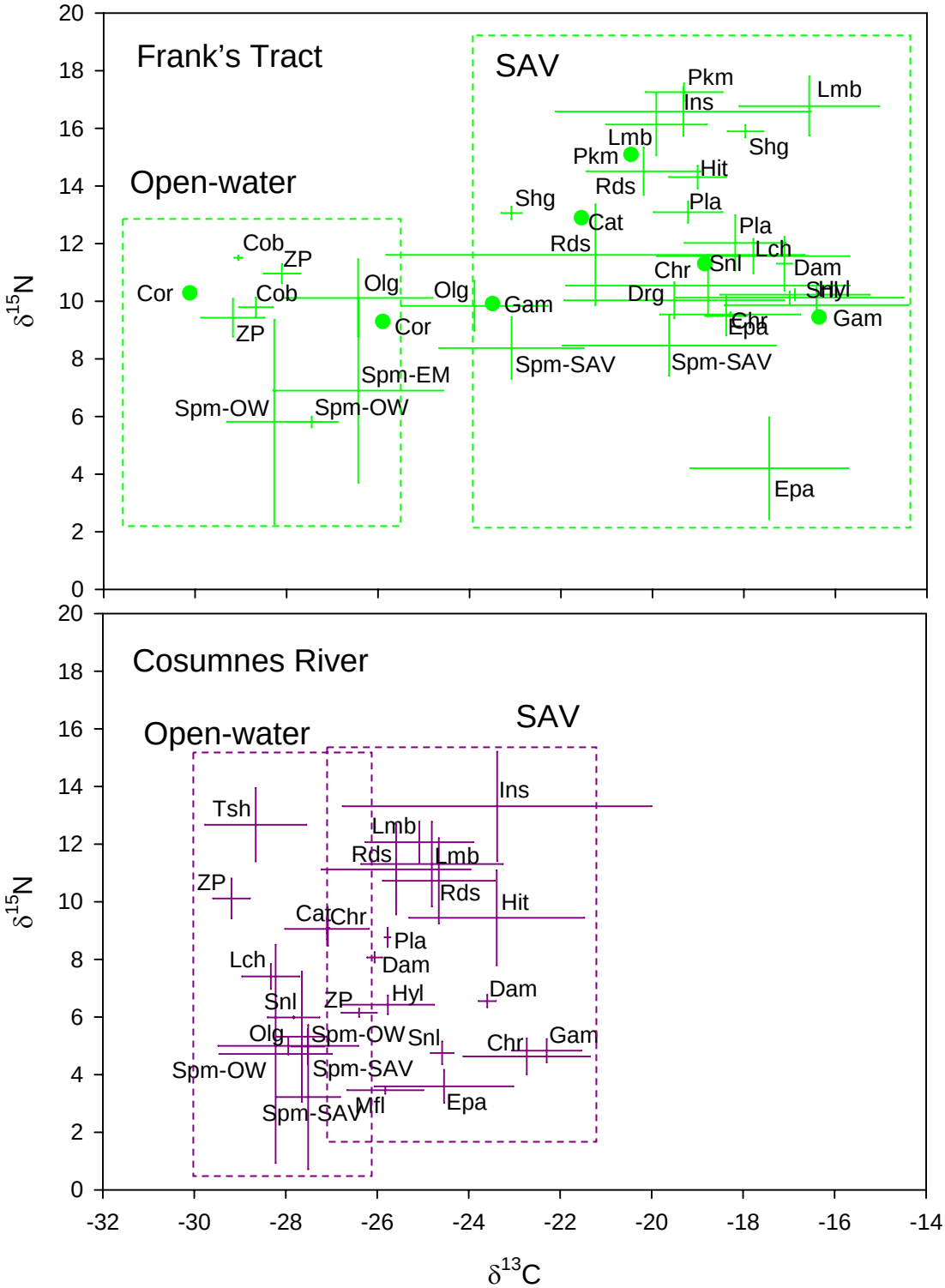


Figure 10. Stable isotope plot showing feeding relationships among fish, invertebrates and suspended particulate matter (SPM) in Frank's Tract (green) and Cosumnes River (purple) in December 2003 and June 2004 (except for epiphytic algae, which was collect on Teflon sheets in March and July 2005). Values are means (±SD). Open water (OW) and submerged aquatic vegetation (SAV) food webs are operationally identified using stable isotopes. SPM-OW and SPM-SAV – open water and SAV SPM; Epa – epiphytic algae; OL – oligochaete *Sparganophilus eiseni*; HL – *Hyalella azteca*.; CO – *Corophium* spp.; LC – leech, DF – Damsel fly, CR – Chironomidae midges.; PL – flatworms; ZP – bulk zooplankton (> 150 µm); TS – threadfin shad; IS – inland silverside; HT – hitch; RS-D and RS-J - redear sunfish (Dec '03 and Jun '04); LB-D and LB-J – largemouth bass (Dec '03 and Jun '04), PM – pike minnow; SG – shimofori goby.

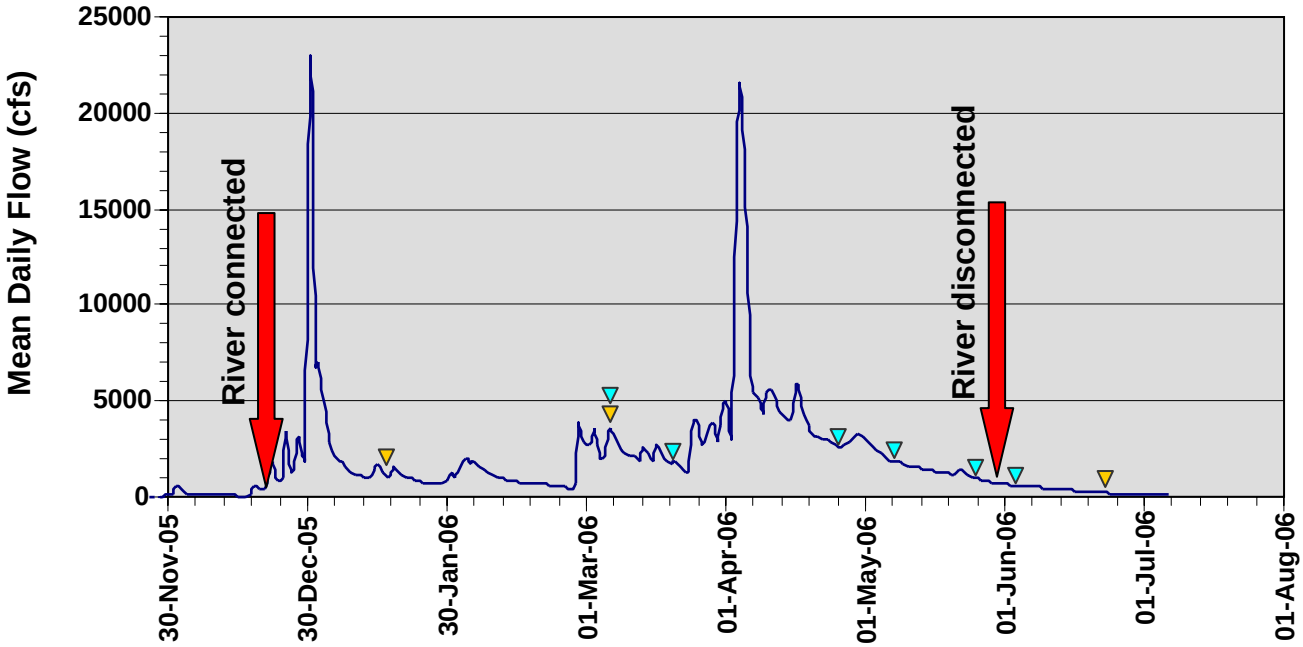


Fig. 11. Cosumnes R. Mean Daily Flow at Michigan Bar (MHB) gauging station (blue line). The red arrows indicate the dates when the floodplain became connected to, and disconnected from the Cosumnes R mainstem. Sediment was collected on three dates (gold triangles), along with water samples associated with the $Hg(II)_R$-DOC study. Additional water samples, associated with the caged larval fish studies were collected separately (blue triangles).

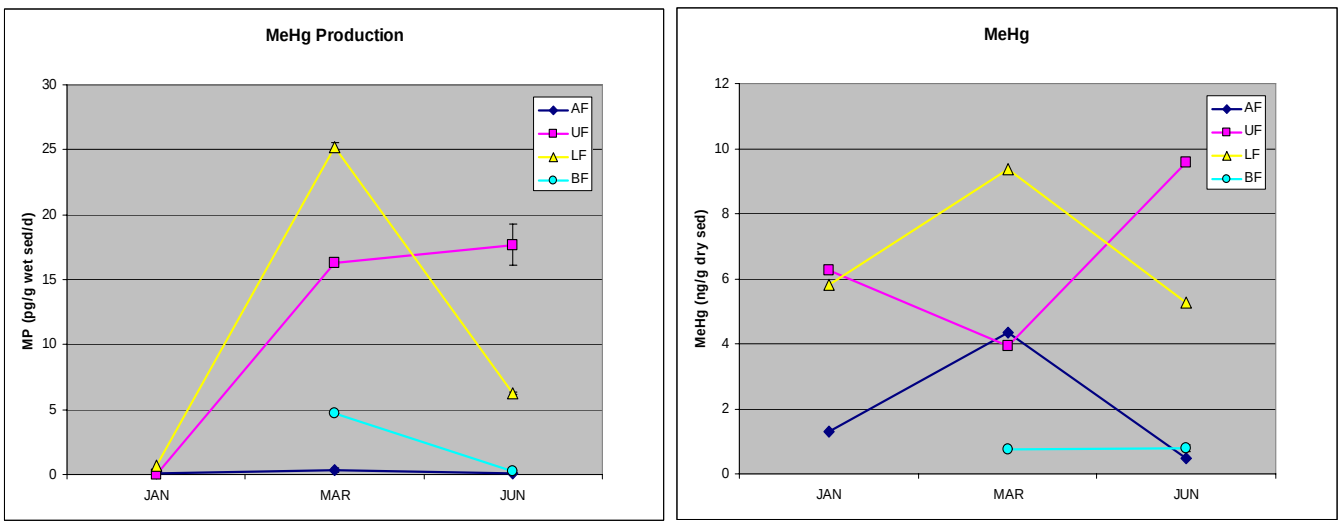


Fig. 12. Sediment methylmercury production (MP) rates and concentrations in the Cosumnes R. and the floodplain sampling during Phase III (January, March and June 2006). Cosumnes R. sites include those above and below the floodplain (AF and BF, respectively). Sites within the floodplain include those in the upper and lower floodplain (UF and LF, respectively), as per Fig. 1E. No BF sample was collected during January.

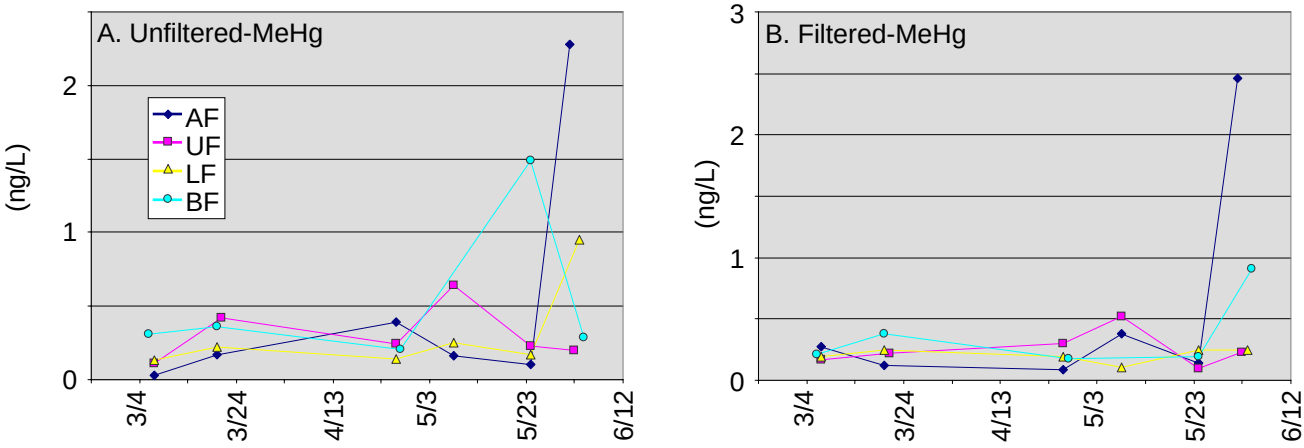


Fig. 13. Water column unfiltered and filtered methylmercury (MeHg) concentrations in the Cosumnes R. and Floodplain sampling during Phase III as part of the caged fish portion of the study. The Cosumnes R. sites include those above and below the floodplain (AF and BF, respectively). Sites within the floodplain include those in the upper and lower floodplain (UF and LF, respectively), as per Fig. 1E.

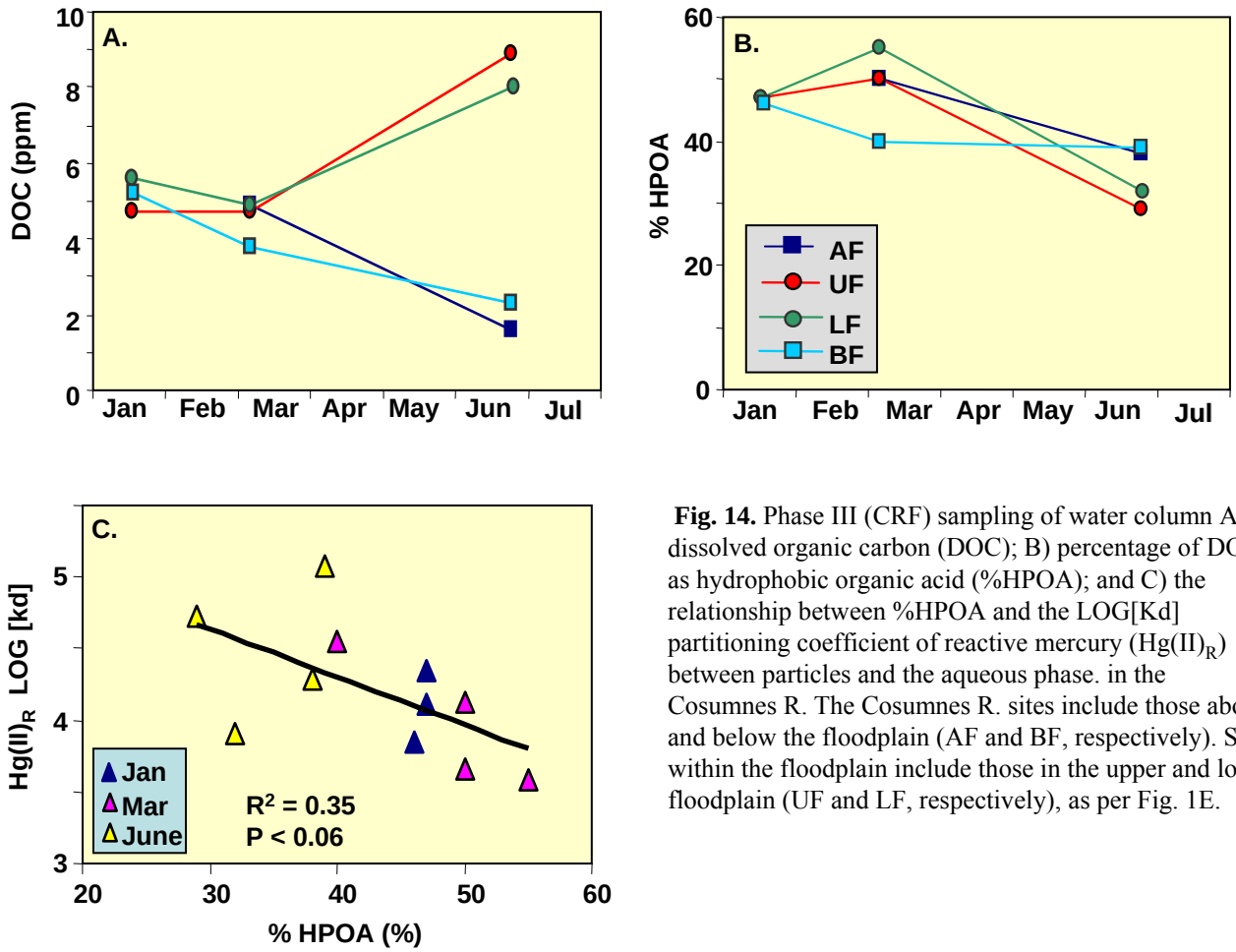


Fig. 14. Phase III (CRF) sampling of water column A) dissolved organic carbon (DOC); B) percentage of DOC as hydrophobic organic acid (%HPOA); and C) the relationship between %HPOA and the LOG[Kd] partitioning coefficient of reactive mercury ($Hg(II)_R$) between particles and the aqueous phase. in the Cosumnes R. The Cosumnes R. sites include those above and below the floodplain (AF and BF, respectively). Sites within the floodplain include those in the upper and lower floodplain (UF and LF, respectively), as per Fig. 1E.

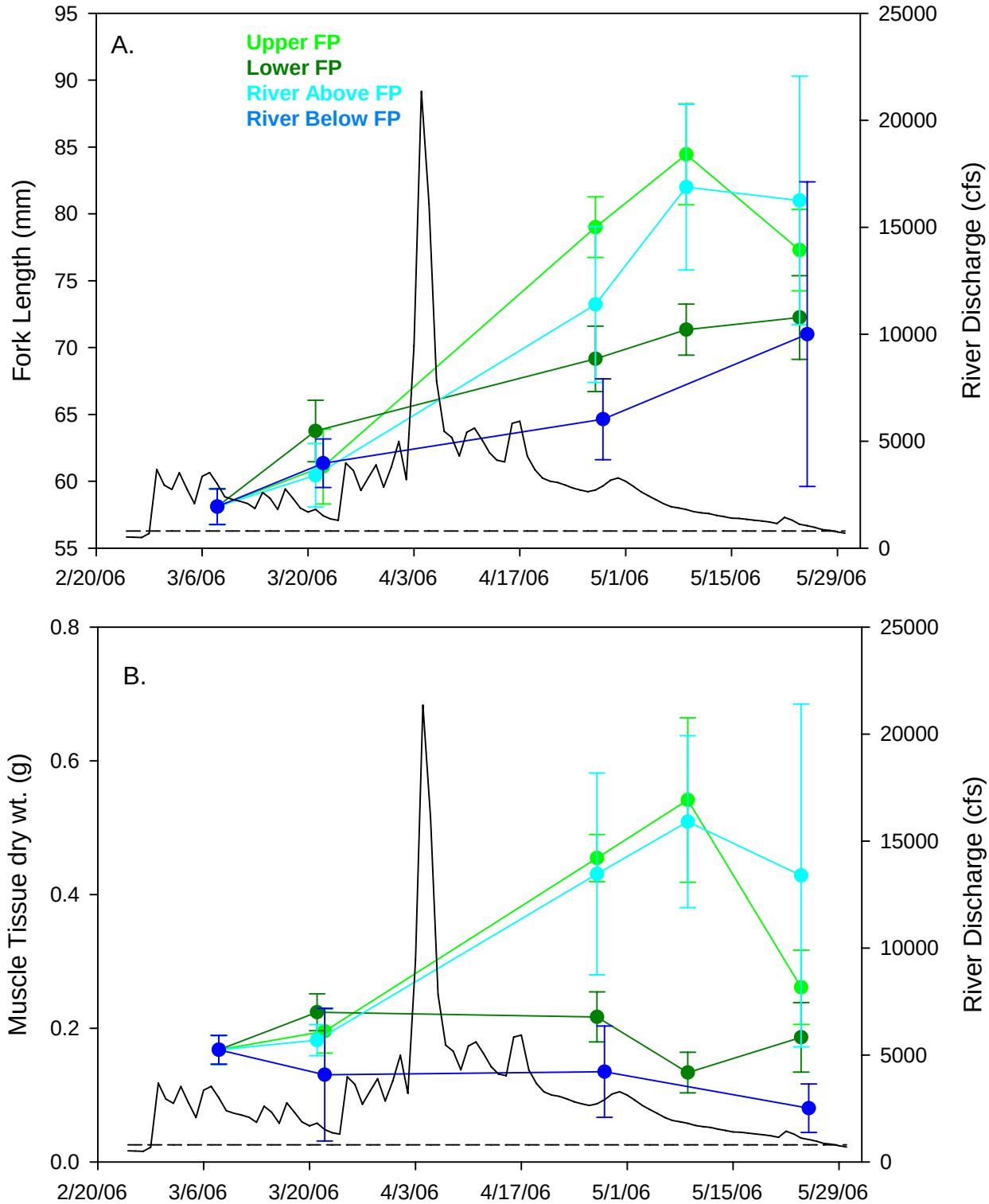


Figure 15. Growth of juvenile hatchery salmon caged at 2 sites on the Cosumnes River floodplain and two sites in the Cosumnes River mainstem and Cosumnes River discharge (cfs). Values are means ± 95% conf. intervals. A. Fork length (mm). B. Muscle tissue dry wt. (g).

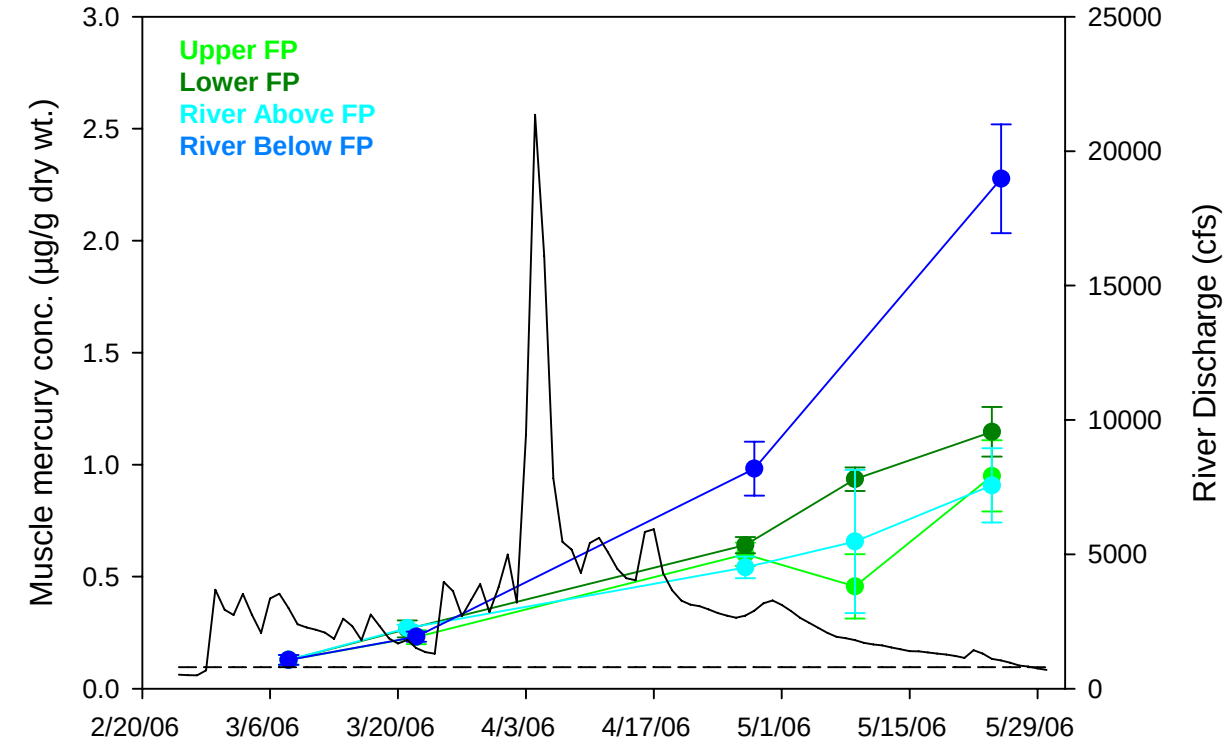


Figure 16. Mercury accumulation in muscle tissue over time in caged juvenile Chinook salmon on and off the Cosumnes River floodplain and Cosumnes River discharge (cfs). Values are means ± 95% conf. intervals.

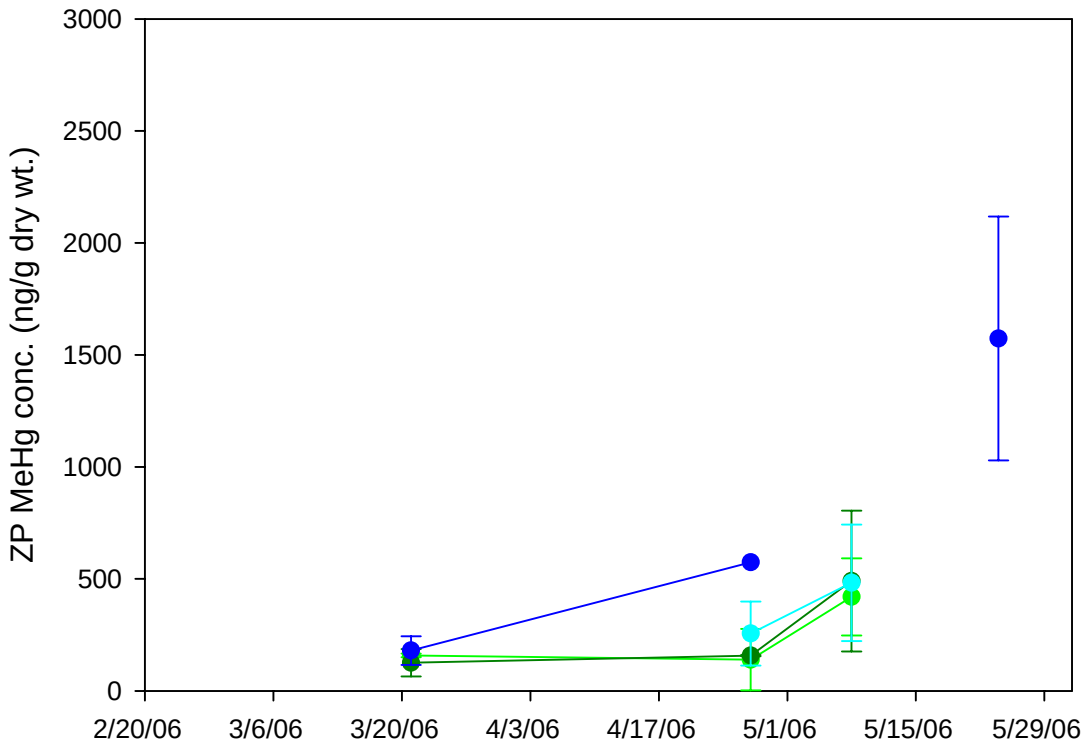


Figure 17. Mercury accumulation in bulk zooplankton over time at juvenile Chinook salmon caging sites on and off the Cosumnes River floodplain. Values are means ± 95% conf. intervals.

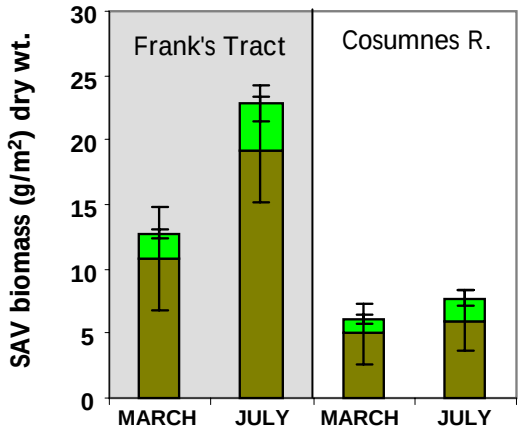


Figure 18. Total submerged aquatic vegetation (SAV) biomass (live and dead) per m^2 in March and July 2005 in Frank's Tract and the Cosumnes R.. Data represent 2 replicates from 2 locations in each region. Live biomass was separated from dead biomass based on color, epiphyte colonization and tissue turgidity. Error bars reflect the 95% CI.

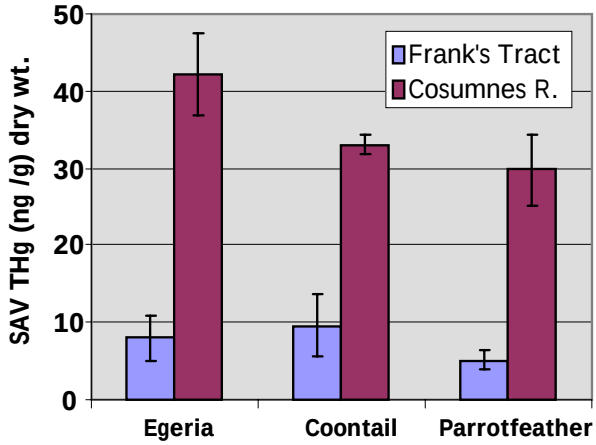


Figure 19. Total mercury concentrations in the dominant SAV species of the Cosumnes R. and Franks Tract (July 2005). Data represent 2 samples from each of 2 sampling locations (n=4) at each site except for *Egeria*, which represents 5 samples from each of 2 sampling locations at each site. 95% CI

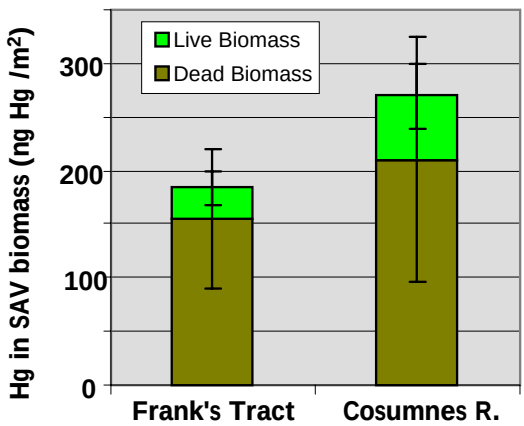


Figure 20. Total Hg pool in SAV biomass (live and dead) per m^2 in July 2005 in Franks Tract and the Cosumnes R. Error bars represent the 95% CI.

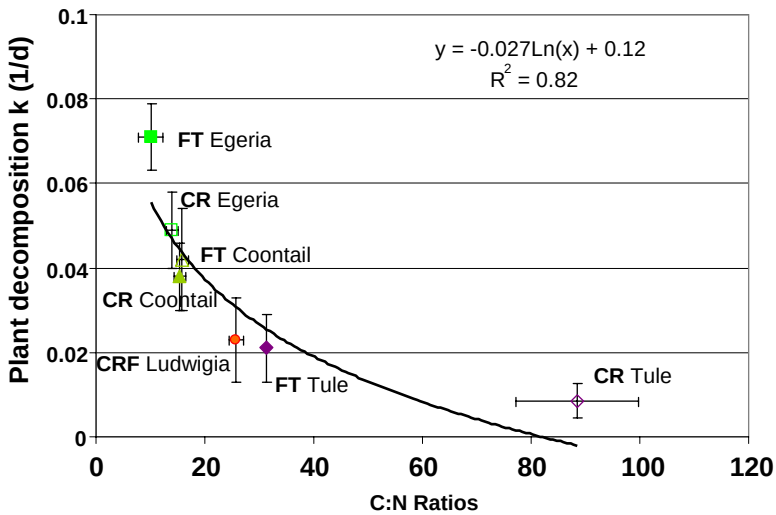


Figure 21. Plant specific decomposition (biomass loss) rate constants (k) as a function of plant tissue C:N Ratio. Each data point represent 3-5 samples for each C:N ratio, and 4 samples at each of 4 time points to calculate each k value. Samples listed with each value were collected from either Frank's Tract (FT), the Cosumnes R. main channel (CR), or the Cosumnes R. floodplain (CRF). Error bars represent the standard deviation of each measure.

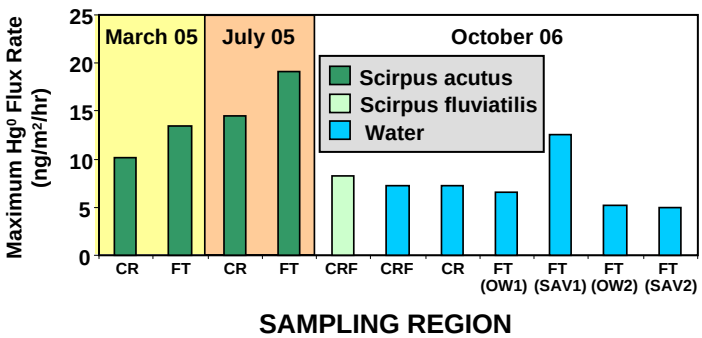


Figure 22. Maximum gaseous mercury (Hg^0) flux from vegetation and water surfaces. Fluxes from plant surfaces were measured using two different techniques (a LiCor1600 porometer with gold trap in March and July 2005, and a continuous Hg flux analyzer with polycarbonate chamber in October 2006).