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

**M. Marvin-DiPasquale¹, R. Stewart¹, N.S. Fisher², P. Pickhardt², R.P. Mason³, A. Heyes⁴,
L. Windham-Meyer⁵**

¹U.S. Geological Survey, Menlo Park, CA.

²Stony Brook Univ., Stony Brook NY

³Univ. of Connecticut, Groton, CT

⁴Univ. of Maryland., Chesapeake Biological Laboratory, Solomons, MD

⁵Lehigh Univ., Bethlehem, PA

November 7th, 2005

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

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

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 critical 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 has been undertaken, which focuses 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 region (low biota Hg) and the Cosumnes River (CR) tributary (high biota Hg) (Fig. 1). In addition to comparing these large regional areas, the current study is 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 central Delta FT region. The study is laid out in three phases over three years. Phases I and II both consist of field sampling components designed directly contrast the CR and FT regions, with phase I weighted more towards spatial comparisons and Phase II weighted more towards temporal (seasonal) comparisons. In Phase III attention is focused exclusively on the CR region and Hg-cycling dynamics associated with the floodplain during the period immediately before, during and after seasonal inundation.

Scientific goals and objectives of the project

The current proposal has three primary objectives: (1) to conduct field-based and controlled laboratory studies 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 FT), (2) to examine these processes at three key sub-habitats (EM, OW, SAV) within both regions and in the CR floodplain during the period associated with seasonal inundation, and (3) to design these studies so as 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 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 was added to the original list. The research elements of the project are conceived and designed such that all of the listed hypothesis 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. Described in terms of our study regions, the hypotheses are:

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

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

HYP III. FT 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.

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 FT.

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).

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.

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 research 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

While the official project start on the original contract is listed as July 1, 2003, final funding and signatures were not in place until late August, 2003, resulting in almost a two month delay in actual project initiation. The current official completion date is June 30, 2006.

Project status

Apart from the initiation delay noted above, the project is largely on schedule. However, the field sampling originally slated for late October 2005, was postponed. This was to be the last sampling event directly comparing Hg-cycling dynamics in both CR and FT. The last minute group decision to cancel this fieldtrip was based on multiple factors, largely stemming from the fact that Dr. Mason (a co-PI on the project) recently left the Univ. of MD and moved his laboratory to the Univ. of Connecticut. This move represented a major transition for Dr. Mason, which necessitated a prolonged period of 'down-time', from approximately August thru October 2005, as the MD lab was essentially shut down and pack up, and the UCONN lab was set-up. Since Dr. Mason's group is largely responsible for running the bulk of the environmental Hg samples collected as part of our SFB-Delta project, there has understandably developed somewhat of a sample analysis backlog, resulting from this major institutional transition, and the fact that new laboratory technicians needed to be hired and trained. Subsequently, the PI's on the SFB-Delta Hg project were faced with conducting our final CR/FT comparison, with a large portion of the previously collected Hg data yet to be analyzed. The PI's conferred and unanimously agreed that it was scientifically wiser to postpone this final regional comparison until more of the previously collected data is analyzed, least we conclude that we might approached the final field sampling differently in some way. Apart from making this group decision, we have also recently submitted an amendment proposal to CBDA, requesting (among other things) that a one year extension be granted for the project completion date, in light of the above situation. Further, technicians are now in place, and the Hg analytical capacity has just recently come back on-line, at Dr. Mason's new UCONN laboratory. So every effort is being made to work through the remaining samples that are awaiting analysis.

Project milestones achieved

To date we have: a) completed Phase I field sampling (Dec. '03 and June '04), which focused on examining the spatial variability of Hg dynamics at the regional and sub-habitat levels, b) conducted two of three planned sampling events for Phase II (March and Aug. '05), which focused on expanding our assessment of the seasonal variability associated with Hg processes in both regions and sub-habitats, c) held a three day meeting of all lead scientists in December '04 to assess data and project progress to date, d) submitted an amendment proposal to CBDA in late 2004 that added a wetland plant ecologist (L. Windham) to our research team (proposal funded), e) conducted additional sampling of OW, SAV and EM sub-habitats adjacent to the Dutch Slough Restoration Project site (ca. 8 km west of FT in the central Delta) during Aug. 05, as requested by CBDA, to provide direct process-level Hg cycling information for that project, and to determine to what extent our overall findings in the nearby FT sampling area can be extrapolated to the area surrounding the Dutch Slough restoration area.

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 given above (HYP I – VI), 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 FT.

The data collected to date supports this hypothesis, as a number of key benthic parameters are statistically greater in CR than in FT, including concentrations of a) sediment reactive inorganic Hg(II)¹, b) sediment MeHg, and c) pore water MeHg, as well as d) ²⁰³Hg(II) derived rates of microbial MeHg production (Table 1). This regional comparison includes all data associated with the first three field collections. Other relevant significant results include the fact that both the sediment organic content and pore water sulfate concentrations are higher in FT than in the CR region, which leads to generally higher microbial sulfate reduction rates (SRR) and higher concentrations of sedimentary total reduced sulfur (TRS) (Table 1). 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 the freshwater tributaries, as discussed below.

Apart from these overall regional differences, sediment MeHg concentrations were greatest at the EM and SAV sub-habitat levels, in the CR region (Fig. HI-1). The third CR sub-habitat, the OW zone, exhibited the lowest sediment/porewater partitioning coefficient (Fig. HI-1). This indicates that a larger proportion of MeHg is in the aqueous phase as opposed to attached to the solid phase, for the CR OW habitat, and as compared with all other sub-habitat types in either region. Both of these factors suggest that MeHg moves from the sediment to the water column more readily in the CR region, compared with FT.

Methylmercury Production and Degradation in Sediments

The production of MeHg is dependent on both the activity of the bacteria involved in the Hg(II)-methylation process (largely the sulfate reducers) and on the availability of ‘reactive’ inorganic Hg(II) for methylation. While the higher average SRR values in FT might lead one to predict higher rates of MeHg production in this region, compared to the CR, the negative influence of reduced-S concentrations on reactive Hg(II) pool size may actually mitigate this trend. This is most readily reflected in the current data set by the significant negative relationship between SRR and the %Hg(II)_R, both LOG transformed (Fig. HI-2), which implies as the rate of microbial sulfate reduction increases (and the concentration of reduced-S species increases), 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,

¹ ‘Reactive’ inorganic Hg(II) is defined here as the pool of Hg(II) in whole (non-digested) sediment that is readily reduced to gaseous elemental Hg⁰ by excess SnCl₂, under mildly acidic and anoxic (N₂ 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.

NAWQA stream/river sites, and the CBDA sponsored Petaluma River marsh project (Yee et al. 2005).

The balancing of microbial rates and Hg-substrate availability in mediating in situ rates of MeHg production and degradation for the three sub-habitat types examined, is illustrated in Fig. HI-3. The radiotracer derived rate constants k_{mp} and k_{md} (Fig. HI-3a, d), give a measure of the activity of the native microbial population involved in MeHg production or degradation, respectively, when supplied a readily available form of Hg(II) or MeHg. Microbial activity associated with MeHg production (k_{mp} values) was the highest in the organic rich SAV sediments, in both regions. In contrast, k_{md} values varied widely and showed no strong sub-habitat difference. The pool size of Hg(II)R, a surrogate measure of the pool of inorganic Hg(II) readily available for methylation, was highest in the EM sites from the CR region (Fig. HI-3b), which might reflect the influence of emergent marsh plants transferring oxygen into their root zone. The resulting calculated rate of MeHg production (MP), which accounts both for the measure of in situ microbial activity and the in situ pool size of Hg(II)R, suggests that the highest MP rates are associated with SAV habitats in both regions (Fig. HI-2c), and that MP rates are generally higher in CR, compared to FT. To calculate in situ rates of MeHg degradation (MD, Fig. HI-3f), the concentration of in situ porewater MeHg (Fig. HI-3e) was assumed to be the fraction of whole sediment MeHg that is most readily available for MD. The highest concentrations of porewater MeHg were associated with OW and SAV sub-habitats in the CR region (Fig. HI-3e). Used in conjunction with the radiotracer derived k_{md} values, calculated rates of MD was highest in the SAV habitat of the CR. (Fig. HI-3f).

Total Mercury and Methylmercury in the Water Column

The higher rates of MeHg production and MeHg concentrations in CR sediments appears to translate into higher average MeHg concentrations in unfiltered water column samples from the CR, compared to FT (Table 2). However, the fact that average THg in unfiltered water samples was also higher in CR, and that regional differences in unfiltered samples were not pronounced, indicates that regional differences in particle-associated Hg fractions reflect regional differences in suspended particulate matter concentrations. While the spatial and temporal variability in the water column Hg speciation data was quite high (not shown), over 80% of dissolved THg was < 2 ng/L, and over 87% of dissolved MeHg was < 0.2 ng/L for both regions during the first three sampling events. The % MeHg for filtered water was similar for both regions (averages of CR = 8.3%, FT = 7.6%), and greater than for the unfiltered water (CR = 4.8%, FT = 4.7%), which is expected given the lower average partitioning coefficient between particles and the aqueous phase for MeHg, compared to inorganic Hg(II).

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

The data collected to date is not sufficient to either support or refute this hypothesis. We conducted total-Hg, MeHg and DOC sediment/water flux experiments at both CR and FT during the March 2005 field sampling. Whole sediment cores ($n = 3$ per site) were collected from the two OW sites in each region. All Teflon core liners and parts were used, and incubations were initiated on the day of collection, with four time points being sub-sampled over the ensuing 24-hr period. The results were highly variable and non-linear in most cases (data not shown), making

the calculation of dependable flux rates difficult. Sediment/water Hg flux experiments conducted using the same whole core approach are typically quite challenging, labor intensive, and often exhibit highly variable spatial flux rates (Kuwabara et al. 2002, 2003; Topping et al. 2004). It is doubtful, at this point of the program, that we have enough in human and financial resources to carry out this measurement with enough replication, and at enough locations, to rigorously test the above hypothesis. However, earlier measures of mercury species flux, using diver-deployed benthic chambers, suggests that the CR region exhibits a higher areal MeHg flux than does FT (Gill et al. 2003).

HYP III. FT 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 to date is not sufficient to either support or refute this hypothesis. Mercury stable isotope ($^{201}\text{Hg(II)}$ and Me^{199}Hg) amendment experiments were conducted on-site in the open water regions of both CR and FT during the June '04 sampling trip, to assess rates of microbial and photo- MeHg degradation, Hg(II) -methylation, and dissolved gaseous Hg^0 production in the water column. These experiments were conducted in situ using Teflon bottles, with sub-samples being collected at multiple time points over a 24-hour incubation. However, a mass balance of the initial isotope concentrations added to each bottle was not achieved at the end of the experiment, suggesting significant loss of isotope either from the bottle or during sample processing. This makes analysis of the experimental data associated with these water column Hg-transformation measurements problematic. Dr. Mason's group is currently working on a new improved method for conducting these types of measurements, which uses collapsible Teflon bag, instead of bottles, for the incubation. These methods are being worked out in the context of Dr. Mason's other funded projects, but will likely be applied to this study during the final field trip that will compare the two study regions.

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 FT.

The data collected to date generally refutes this hypothesis, although, a significant portion of the biota samples collected from the field have yet to be fully analyzed, so this conclusion is tentative. Levels of MeHg in the CR food web are approximately 2X higher than those in FT. Enrichment of Hg with each trophic level was statistically similar between the two food webs, suggesting that Hg accumulation in these regions was relatively insensitive to differences in habitats, feeding behavior and species-specific accumulation (i.e. kinetics) at the food web base. This suggests that the difference in Hg in sport fish was driven by differences in either a) the quantity of MeHg at the food web base or b) by mechanisms controlling MeHg uptake into algae at the food web base. Controlled laboratory experiments of Hg(II) and MeHg uptake kinetics demonstrates that water chemistry alone may control the transfer of mercury species into the base of the food web in certain instances. Thus, based on these preliminary data we tentatively conclude that the quantity of MeHg entering the food web base is the primary factor leading to higher MeHg concentrations in biota for the CR region, and not regional differences in feeding behavior, species-specific Hg accumulation kinetics, or food web structure.

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 the CR and FT study sites. 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. HIV-1 & Table 3). However, 2/4 of species (diatom and cryptomonad) had significantly higher MeHg bioconcentration factors in FT water relative to CR water. The remaining 2 phytoplankton species (green alga and blue-green alga) had similar VCFs in both water types (Fig. HIV-2 & Table 3). While it is unclear as what accounted for the cases where differences were observed, FT water is both higher in ionic strength and in DOC. We are currently hypothesizing that the latter may have a significant effect on MeHg uptake kinetics in certain phytoplankton species, and we are planning to conduct some follow up experiments to test 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 in each water type (Table 4), although some amphipod experiments are still pending. Direct, aqueous accumulation of Hg(II) and MeHg by invertebrates is generally similar (data not shown).

Fish accumulation and retention of Hg(II) from dietary sources (Hg exposed phytoplankton fed to either *Daphnia pulex* or *Hyallorella sp.*, which were subsequently fed to fish) exhibited similar AE's for each water type and in both fish species tested (Fig. HIV-3A & 3B). The AE of MeHg from invertebrate diets by mosquitofish was greater in FT water (Figure HIV-3C), but the AE of MeHg by redear sunfish was greater in CR water (Fig. HIV-3D). 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 (Fig. HIV-4A & 4B). Despite the greater uptake rate constants in CR water, final fish burdens are overwhelmingly determined by high AE's of MeHg from invertebrate diets.

Field Based Studies of Trophic Mercury Enrichment

Food webs were characterized for both CR and FT in December 2003 and June 2004 (a complete dataset is currently available for December 2003). Similar species of invertebrates and fish were collected in both regions and feeding relationships were characterized using stable isotopes of carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$). Although similar species were found in both regions, $\delta^{13}\text{C}$ values suggested some differences in the connectivity among habitats. For example, the FT open water and SAV habitats were relatively disconnected, while the CR habitats were more connected (Fig. HIV-5). All of the fish species sampled in FT appeared to feed upon invertebrates living in the SAV (shared similar $\delta^{13}\text{C}$ values), which were isotopically distinct from the invertebrates feeding in the open water (zooplankton). Conversely for the CR, fish and their prey (e.g. threadfin shad and zooplankton) feeding in the open water were only slightly more depleted in $\delta^{13}\text{C}$ than the fish and their prey feeding in the SAV habitat. This suggests some degree of C exchange among the habitats, or that the habitats shared similar C sources. These results suggest that constituents such as stable isotopes and Hg that are transformed

through habitat specific biogeochemical cycles may be transported or shared among habitats in the CR and not in FT.

Concentrations of MeHg in both FT and CR food webs increased with trophic position, as measured by $\delta^{15}\text{N}$ values (Fig. HIV-6). Mercury levels were considerably more variable among individual fish in CR than in FT. This variability could not readily be explained by fish size, but might be explained by the observed heterogeneity in MeHg within CR habitat types.

In order to directly compare Hg exposure between regional food webs, MeHg enrichment as a function of trophic level (adjusted for the regional differences in food web base $\delta^{15}\text{N}$ signal) was plotted for SAV biota from both regions (Fig. HIV-7). Trophic enrichment of MeHg in SAV food webs was statistically similar among regions (i.e. regression slopes both ~ 0.2) and similar to food webs from other aquatic systems studied throughout the world. However, the absolute level of MeHg contamination for the CR SAV food web was approximately 2X that of the FT SAV food web. Because the relative accumulation of MeHg as a function of trophic position was the same in both regions, the higher MeHg concentrations in the CR food web could only be explained by regional differences in either a) the quantity of MeHg available for uptake at the food web base, or b) the mechanisms controlling accumulation into the food web base. Analysis of all water column data collected for the first three field trips does confirm that concentrations of filtered MeHg are significantly higher ($P = 0.035$) in the CR region, compared to the FT region, based on a 3-way ANOVA that included region, sampling date and sub-habitat type.

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 data necessary to evaluate this hypothesis will be collected during Phase III of the project. Sampling of the CR floodplain will be initiated sometime in the winter-spring period, once the initial substantial inundation of this area is imminent or is just beginning. Logistical planning for this sampling has already begun.

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 is can neither fully support nor refute the above hypothesis. Regional differences in plant community characteristics and interactions with Hg dynamics have been identified. Most notably, while plants from the CR appear to have higher Hg levels in their tissue (per unit weight), Hg concentrations are similar between regions on an area (m^2) basis. Further, total-Hg flux through the plant biomass pool appear faster in FT, compared to CR, due to faster plant turnover and decomposition rates and higher ionic strength water conditions. However, it is not readily apparent at this point if these differences in plant-Hg processes directly mediate regional differences in biota Hg concentrations.

SAV Biomass and Mercury Concentrations

The SAV communities from CR and FT differ in three distinct ways. First, FT is strongly 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). Second, where sampled, the standing stock of live biomass (g m^{-2}) is 2-3X greater in FT (March = 2.0 ± 0.4 , July = 3.6 ± 1.4) compared to CR (March = $1.1 \pm .3$, July = 1.7 ± 0.6). No significant seasonal differences in live biomass were observed, but the total amount of dead biomass (g m^{-2}) increased 2X in FT between March (10.8 ± 4.0) and July (19.2 ± 4.1), implying a complete turnover of vegetation and higher primary production rates, compared to CR. Third, live *Egeria* in FT has 2X the leaf density (40 ± 18 leaves/10cm stem) compared to CR *Egeria* (22 ± 13 leaves /10cm), which may allow for a greater amount of epiphytic algal growth, and higher rates of organic decomposition (as observed below) in the FT region.

Plant tissue Hg concentrations were 2-6X greater for the CR (Fig. HVI-1a), with the comparatively low FT *Egeria* Hg concentrations quite homogeneous throughout the waterbody ($n = 10$ sites, Relative Std. dev. = 36%). When normalized by areal biomass density, Hg pools in the SAV sites sampled were similar (~ 185 - 265 ng/m^2) between both sites and seasons (Fig. HVI-1b). However, these levels are roughly 20,000-30,000X lower than the average total-Hg pool calculated for the surface 0-1 cm depth interval of sediment in the CR ($2.0 \times 10^5 \text{ ng/m}^2$) and FT ($2.0 \times 10^5 \text{ ng/m}^2$) regions. Even though only a small percentage of these sediment THg pools were found to be 'reactive' inorganic Hg(II) (i.e. %Hg(II)_R [avg]; CR = 3.3%; FT = 1.7%), this still represents Hg(II)_R levels approximately 380-1000X higher than from plant biomass (assuming $\sim 200 \text{ ng Hg/m}^2$) at the SAV sites assessed, even if all of the plant-Hg was assumed to be Hg(II)_R.

SAV Decomposition – Hg release and mass loss

Three separate sets of laboratory controlled decomposition rate experiments (each 1-month in duration) were performed. In Experiment #1 (Fig. HVI-2a), the decomposition of cut-up *Egeria* segments in filtered ($<100\mu\text{m}$) anoxic site water was measured, using both plant material and water collected during March 2005 from both sites (i.e. 4 reciprocal plant/site-water treatments). In Experiment #2 (Fig. HVI-2b), *Egeria* decomposition in filtered site water was measured under both anoxic and oxic conditions (April 2005 plant and water sample collection). In Experiment #3 (Fig. HVI-2c), decomposition rate of both *Egeria* and Coontail were compared at three salinity levels (1, 10, and 20 ppt) in material collected from both sites during July 2005.

Initial decomposition rates (k_d) were consistently greater for FT *Egeria* ($k_d = 0.07 \text{ d}^{-1}$), compared to CR *Egeria* ($k_d = 0.05 \text{ d}^{-1}$), and 70-90% of *Egeria* biomass was decomposed after one month in all three experiments. In contrast, only $36 \pm 10\%$ of coontail biomass had decomposed after one month of incubation ($k_d < 0.04 \text{ d}^{-1}$) in Experiment #3.

Measured rates of Hg leaching during plant decomposition varied by experiment. In Experiment #1, *Egeria* incubated in filtered site water released $\sim 200 \text{ pg g}^{-1}$ dry weight, or 20-40% of its Hg burden over 28-days. Plant biomass decomposition and associated Hg leaching rates were stimulated under oxic (compared to anoxic) incubation conditions in Experiment #2 (Fig. HVI-2b) in both regions. In Experiment 3, salinity had a variable effect on both SAV decomposition and Hg release rates, but there was a striking difference between SAV species. Whereas *Egeria* decomposed quickly under all salinities, and released Hg into solution at rates of 20-100 pg Hg g^{-1} dry weight, the coontail decomposed very slowly and actually removed Hg from solution rather than released it (Fig. HVI-2c).

IV. Potential Management Implications of Findings to Date

As stated in the CALFED Mercury Strategy document (Weiner et. al 2003), a major challenge with respect to mercury, which confronts agencies and ecosystem managers involved with ecological restoration in the SFB, is “to avoid increasing – and to eventually decrease – biotic exposure to methylmercury.” By providing baseline process specific information regarding Hg cycling in two representative and contrasting regions, the current research programs aims to provide the kind of data that will inform 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 project (<http://www.dutchslough.org/homepage.html>) near our FT study area, b) the McCormick-Williamson Tract Floodplain Restoration Project, near our CR study area, which is part of DWR’s larger ‘North Delta Flood Control and Ecosystem Restoration Project’ (<http://baydeltaoffice.water.ca.gov/ndelta/northdelta/index.cfm>), and c) the DWR ‘Flooded Islands’ Project (<http://baydeltaoffice.water.ca.gov/ndelta/floodedislands/index.cfm>) which would significantly impact the hydrology and salinity conditions in the SFB-Delta in general, and FT in particular.

Of all the hypothesis originally posed, to address why biota Hg levels in SFB tributaries appear to be elevated relative to those in the Central Delta, the one that the current data seem most strongly to support is Hypothesis I. To restate it in more general terms: ***‘Benthic conditions are more conducive for net MeHg production in the tributaries (e.g. CR) than in the central delta (e.g. FT.)’***. This essentially implies that sediment and water geochemistry mediate the extent to which a given area is a ‘hot spot’ for MeHg production or not. This is in contrast to Hypothesis IV, which would have implicated regional differences in food web structure or function, and for which we also have a substantial amount of data collected to date. 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, then would management actions that had more of a significant impact on regional food web structure directly.

In terms of sub-habitat types, it appears that areas of high SAV density are also zones of enhances MeHg production. So management actions that would decrease SAV density, such as certain alternatives that are being considered in FT as part of the Flooded Islands project, would also decrease areas of MeHg production. 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, on how particular landscape manipulations might impact Hg cycling over the long term.

Despite the observed species differences in algal Hg accumulation rates in both regions, the observation that MeHg enrichment (as a function of trophic level) was similar for both regions suggests that minor differences in species-specific bioaccumulation or habitat structure do not significantly impact on how Hg propagates up the food web. Instead, either the absolute amount of Hg available for uptake at the food web base or the mechanisms controlling uptake into the food web base are more likely responsible for regional differences in Hg levels at the top of the food chain. Consequently, management actions that significantly reduce or increase the amount of aqueous MeHg available for uptake by photoplankton would likely result in a corresponding decrease or increase, respectively, in MeHg levels in sport fish for that region.

These results also do not imply that food web considerations are unimportant. Quite the contrary. The 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 HgT from the dissolved phase to biota occurs between water and primary producers, implicating phytoplankton as the primary entry point of Hg in both regions studied. 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 affect the trophic transfer of Hg into bulk phytoplankton (into the foodweb base), and subsequently into primary grazers (invertebrates), and ultimately into fish.

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. Again, a management action being considered for the Flooded Islands project would be toraising salinity (and/or turbidity) in FT, which would potentially decrease the density of *Egeria* in this area of the central Delta. Based on our plant decay experiments, this would results in less Hg cycling through *Egeria*, but release from any *Egeria* remaining may be increased due to the higher salinities. In contrast, management actions that would decrease FT salinity would result in slower Hg release from decaying *Egeria*, base on our salinity experimental results. In, contrast, lowering salinity in FT would presumably slow the cycling of Hg, from senescent plant material. However, it is currently unclear how these changes in plant-Hg dynamics would ultimately impact Hg accumulation in fish.

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, E. Warren, K. Sigler, M-N. Croteau, F. Parchaso, E. Moon, and B. Richards at USGS (Menlo Park, CA) for field and laboratory analytical support; M. Stepanova and E. Freimuth at Stony Brook Univ. (Stony Brook, NY) for laboratory analytical support; C. Miller, D. Heyes and E.H. Kim at the Univ. of MD (Solomons, MD) for field and laboratory analytical support; C. Jeffres, 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 ²⁰³Hg(II) for Hg-cycling studies, and R. Cooper and J. Marty at The Nature Conservancy (Galt, CA) for access to and logistical support at the Cosumnes River Preserve site.

VI. Literature Cited

- 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., 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>
- Gill, G., K.Y. Choe, R. Lehman, and S. Han., 2003. Sediment-Water Exchange and Estuarine Mixing Fluxes of Mercury and Monomethyl Mercury in the San Francisco Bay Estuary and Delta (Task 4B). Final Report Submitted to the Calfed Bay-Delta Program, Sacramento, CA. Published On-Line: <http://loer.tamug.tamu.edu/calfed>
- 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, Nevada, 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.
- 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>
- 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.

Table 1. Select sediment (*Sed*) and pore water (*Pw*) biogeochemical parameters in Frank's Tract (FT) and the Cosumnes River (CR) study regions. Values represent the mean of all data (regardless of habitat) collected for that parameter during the first three field collections (Dec. '03, June'04, and March '05). Mean standard errors 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 two-tailed t-test.

Parameter	Units	Matrix	CR	FT	df	P
Total mercury (HgT)	ng/g dry sed	<i>Sed</i>	192 (33)	137 (23)	42	n.s.
Total mercury (HgT)	pg/L	<i>Pw</i>	7516 (1322)	5065 (942)	40	n.s.
reactive inorganic Hg(II)	ng/g dry sed	<i>Sed</i>	2.49 (0.48)	1.05 (0.18)	42	< 0.01
reactive inorganic Hg(II)	Percent (%)	<i>Sed</i>	3.27 (1.74)	1.71 (0.59)	42	n.s.
Methylmercury (MeHg)	ng/g dry sed	<i>Sed</i>	1.17 (0.14)	0.45 (0.05)	39	< 0.0001
Methylmercury (MeHg)	percent (%)	<i>Sed</i>	0.016 (0.006)	0.007 (0.002)	39	n.s.
Methylmercury (MeHg)	pg/L	<i>Pw</i>	694 (104)	322 (61)	38	< 0.005
Methylmercury (MeHg)	percent (%)	<i>Pw</i>	13.0 (1.9)	12.3 (3.3)	36	n.s.
Hg(II)-methylation rate ^a	pg/g dry sed/d	<i>Sed</i>	2.96 (0.65)	1.25 (0.30)	42	< 0.05
MeHg degradation rate ^b	pg/g dry sed/d	<i>Sed</i>	0.38 (0.15)	0.08 (0.02)	38	n.s.
Chloride [Cl ⁻]	mmol/L	<i>Pw</i>	0.20 (0.04)	3.89 (0.77)	42	< 0.0001
Sulfate [SO ₄ ²⁻]	μmol/L	<i>Pw</i>	21 (5)	212 (33)	42	< 0.0001
microbial sulfate reduction	nmol/g dry sed/d	<i>Sed</i>	6.9 (1.8)	157.4 (78.8)	42	n.s.
total reduced sulfur (TRS)	μmol/g dry sed	<i>Sed</i>	7.2 (1.1)	102.0 (22.8)	42	< 0.0005
Sulfide (HS ⁻)	μmol/L	<i>Pw</i>	0.84 (0.21)	1.66 (0.39)	41	n.s.
Crystalline Fe(III)-oxides	mg/g dry sed	<i>Sed</i>	1.67 (0.39)	0.29 (0.14)	42	< 0.005
Organic content	% weight loss on ignition	<i>Sed</i>	8.3 (0.8)	17.4 (3.5)	42	< 0.05
Dissolved Organic Carbon ^c	mg/L	<i>Pw</i>	29.5 (4.2)	37.0 (4.9)	26	n.s.
Grain Size < 0.64 μm (silt)	percent (%)	<i>Sed</i>	47.0 (4.9)	23.4 (4.1)	42	< 0.001

^a Calculated from the ²⁰³Hg(II) radiotracer derived rate constant (k_{meth}) obtained from 4 hr amendment incubation experiments at in situ temperature, and the in situ concentration of reactive inorganic mercury (Hg(II)_{R}), such that $\text{MeHg production (MP)} = \text{Hg(II)}_{\text{R}} - \text{Hg(II)}_{\text{R}} * \text{EXP}(-k_{\text{meth}} * t)$, where $t = 1$ day.

^a Calculated from either the ¹⁴CH₃Hg⁺ or CH₃²⁰³Hg⁺ radiotracer derived rate constant (k_{deg}) obtained from 4 hr amendment incubation experiments at in situ temperature, and the in situ concentration of pore water MeHg, such that $\text{MeHg degradation (MD)} = \text{MeHg}_{\text{pw}} - \text{MeHg}_{\text{pw}} * \text{EXP}(-k_{\text{deg}} * t)$, where $t = 1$ day.

^c DOC means represents those for the first two field sampling trips only

Table 2. Water column Mercury fractions for the Cosumnes River (CR) and Frank's Tract (FT) study regions. Data represents the mean ± (std. error) of all data (including all sub-habitats) collected from the first three field trips: Dec. '03, June '04 and March '05.

Aqueous Hg Fraction	Units	CR			FT		
		Mean	± s.e.	N	Mean	± s.e.	N
Total Hg (unfiltered)	ng/L	7.3	(0.9)	20	3.0	(0.4)	26
Total Hg (filtered)	ng/L	1.5	(0.2)	20	1.2	(0.1)	25
MeHg (unfiltered)	ng/L	0.42	(0.25)	21	0.11	(0.02)	26
MeHg (filtered)	ng/L	0.13	(0.03)	21	0.08	(0.01)	25
MeHg (unfiltered)	(% of THg)	5.1	(2.2)	20	4.7	(1.2)	26
MeHg (filtered)	(% of THg)	8.3	(1.1)	20	7.6	(1.1)	24

Table 3. Calculated volume concentration factors (VCF) for three different phytoplankton species and one cyanobacterium (or blue-green algae) 24 h after initial cell exposures to aqueous inorganic Hg(II) and organic (CH₃Hg(II)). Values are means of 3 replicates \pm 1 SD.

Phytoplankton Species	VCFs \pm 1 SD			
	Inorganic Hg ²⁺		Organic MeHg	
	CR H ₂ O	FT H ₂ O	CR H ₂ O	FT H ₂ O
<i>Chlamydomonas reinhardtii</i> (green algae)	26,300 \pm 1,760	26,500 \pm 3,600	426,000 \pm 260,000	359,800 \pm 89,000
<i>Cryptomonas ozolini</i> (cryptomonad)	47,600 \pm 4,600	43,400 \pm 2,900	*140,100 \pm 32,800	*338,900 \pm 48,300
<i>Cyclotella meneghiniana</i> (benthic diatom)	*4,800 \pm 300	*2,800 \pm 100	*132,300 \pm 5,700	*406,800 \pm 128,700
<i>Synechocystis</i> sp. (cyanobacterium/blue-green)	18,700 \pm 7,000	22,400 \pm 10,000	1,204,000 \pm 442,000	1,744,000 \pm 850,000

*Significant differences in Hg accumulation between water types in single factor ANOVA at the $P \leq 0.05$ level.

Table 4. Calculated assimilation efficiencies (AE) for *Daphnia pulex* (a pelagic zooplankton species) and a *Hyallela* sp. (a macroinvertebrate typically associated with submerged aquatic vegetation) after consuming two species of phytoplankton. Both phytoplankton cells were exposed previously to either inorganic Hg(II) or organic CH₃Hg(II) and then resuspended in fresh water prior to feedings. Values are approximate means of 3 replicates.

Invertebrate species	Approximate Assimilation Efficiency (%)			
	Inorganic Hg ²⁺		Organic MeHg	
	CR H ₂ O	FT H ₂ O	CR H ₂ O	FT H ₂ O
<i>Daphnia pulex</i> fed	72.5%	70.8%	\approx 99.0%	\approx 99.0%
<i>C. meneghiniana</i> <i>Daphnia pulex</i> fed	75.5%	70.8%	\approx 90.1%	\approx 99.0%
<i>C. reinhardtii</i> <i>Hyallela</i> sp. fed	*	*	\approx 99.0%	86.3%
<i>C. meneghiniana</i> <i>Hyallela</i> sp. Fed	*	*	\approx 99.0%	79.2%
<i>C. reinhardtii</i>				

*experiments pending

Appendix I - Products to date

I. Publications

A. Manuscripts Submitted

Pickhardt, P. C., M. Stepanova, and N. S. Fisher. (*in review*). Contrasting uptake routes and tissue distributions of inorganic and methylmercury in mosquitofish, *Gambusia affinis*, and redear sunfish, *Lepomis microlophus*. Environmental Toxicology & Chemistry.

B. Manuscripts In Preparation

Pickhardt, P. C., and N. S. Fisher. (*in prep*). The accumulation and trophic transfer of mercury to *Daphnia pulex* and a *Hyallela* sp. in two natural waters. Submission goal: Environmental Science & Technology or similar.

Pickhardt, P. C., and N. S. Fisher. (*in prep*). Accumulation dynamics of inorganic Hg(II) and monomethylmercury (CH₃Hg(II)) in four species of freshwater phytoplankton in two natural waters. Submission goal: Environmental Science & Technology.

Pickhardt, P. C., E. Freimuth, and N. S. Fisher. (*in prep*). The effects of organic and inorganic mercury consumption on the population dynamics of *Daphnia pulex* and implications for trophic transfer. Submission goal: Bulletin of Environmental Contamination and Toxicology.

II. Public Presentations

A. Oral Presentations

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.

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.

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, Paul (SUNY Stony Brook). Accumulation of inorganic and organic mercury in phytoplankton and the subsequent trophic transfer to crustaceans. November, 15th, 2005. Scheduled to be presented at the upcoming SETAC annual meeting in Baltimore, MD, in a session titled 'Metals and Bioaccumulation'.

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.

B. Poster Presentations

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'.

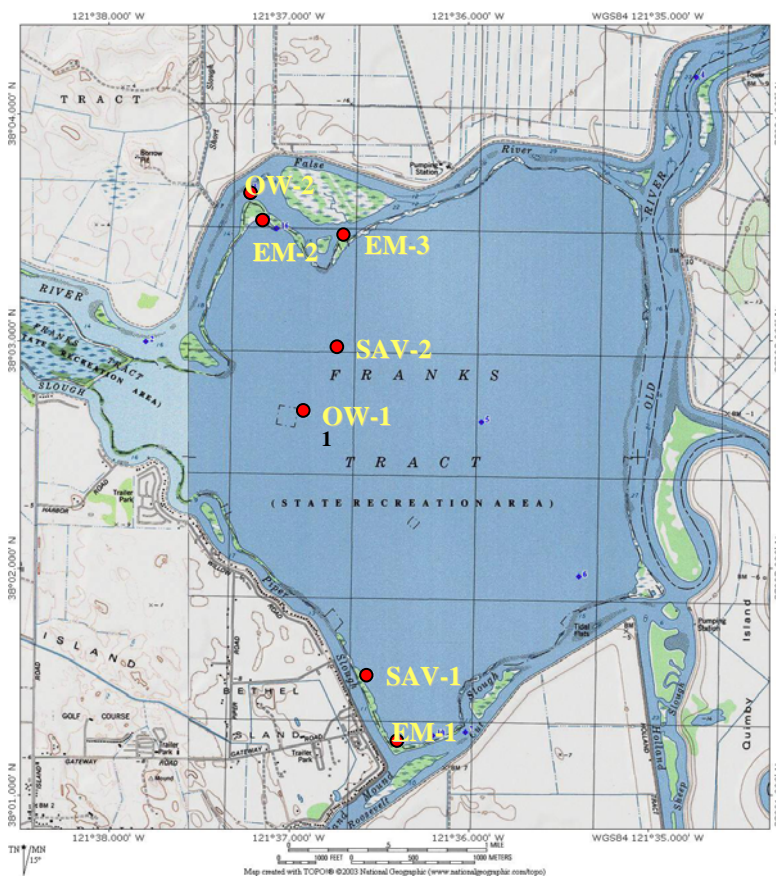
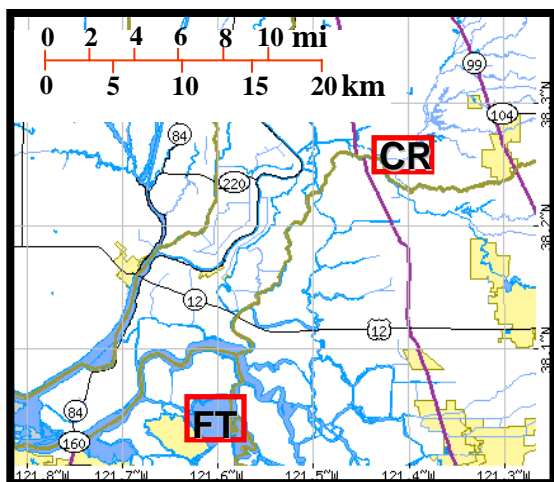
C. Other Presentations / Products

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.

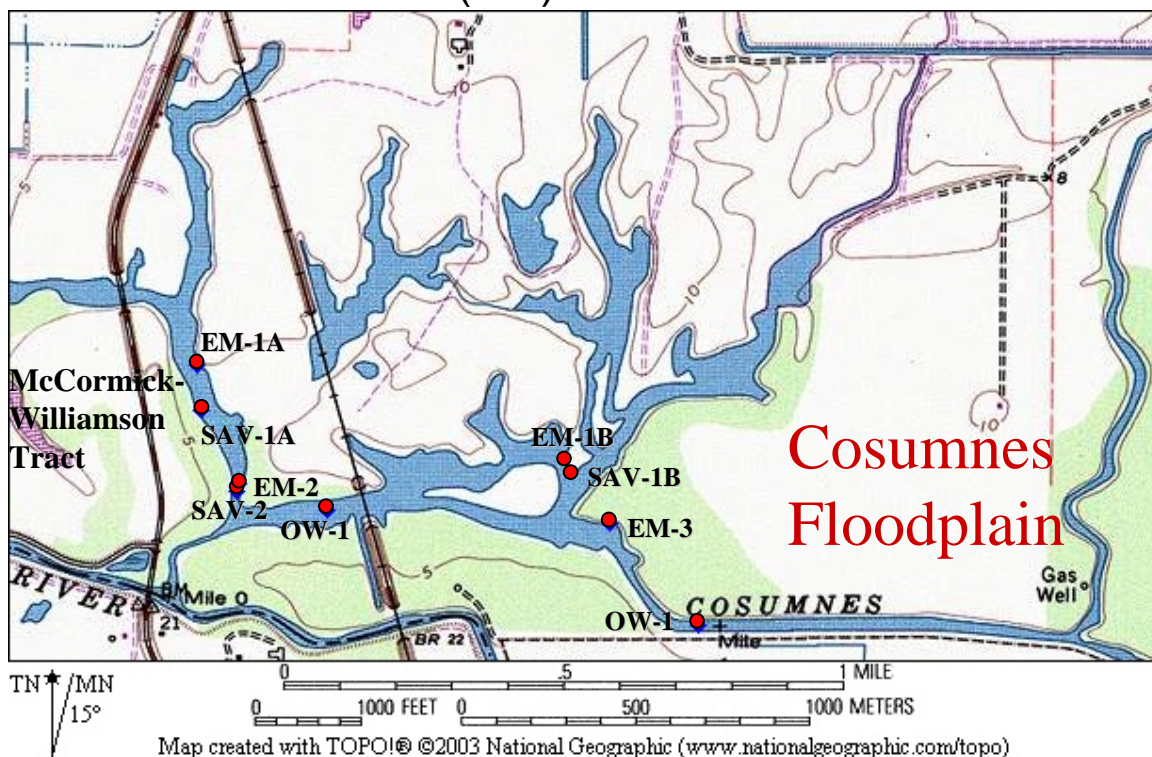
Figure 1.

Frank's Tract (FT)

The SFB-Delta



Cosumnes River (CR)



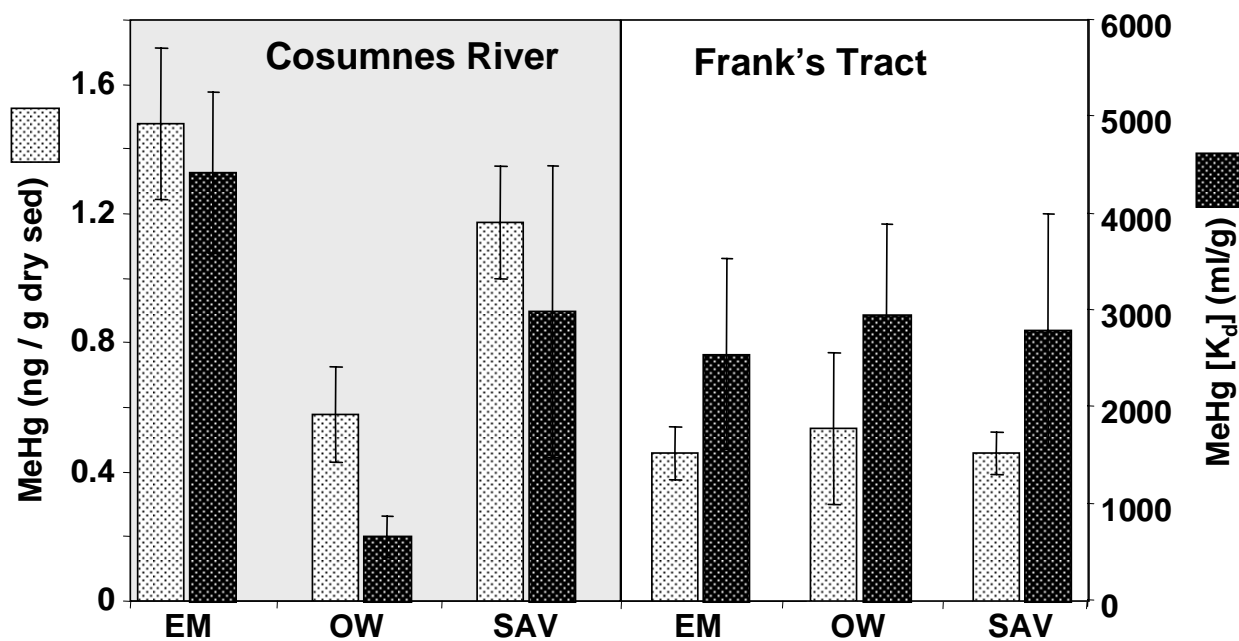


Figure HI-1. Average values for sediment MeHg concentration and sediment:porewater partitioning coefficient (k_d) in surface sediment samples collected from three sub-habitat types, Emergent Marsh (EM), Open Water (OW) and Submerged Aquatic Vegetation (SAV), in both study regions, during the first three field sampling events (Dec'03, Jun'04, and Mar'05).

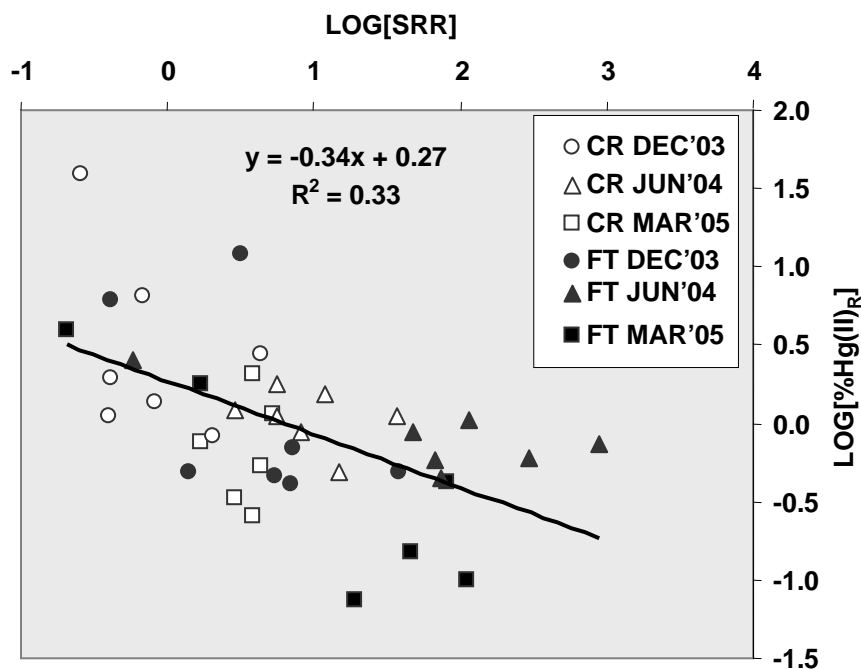


Figure HI-2. LOG[X]-LOG[Y] Linear regression plot of of microbial sulfate reduction (SRR) vs the percentage of reactive inorganic mercury (%Hg(II)_R) for all surface sediment samples collected during the first three field collections from both study regions (CR and FT). The resulting linear regression slope is statistically significant ($P < 0.05$).

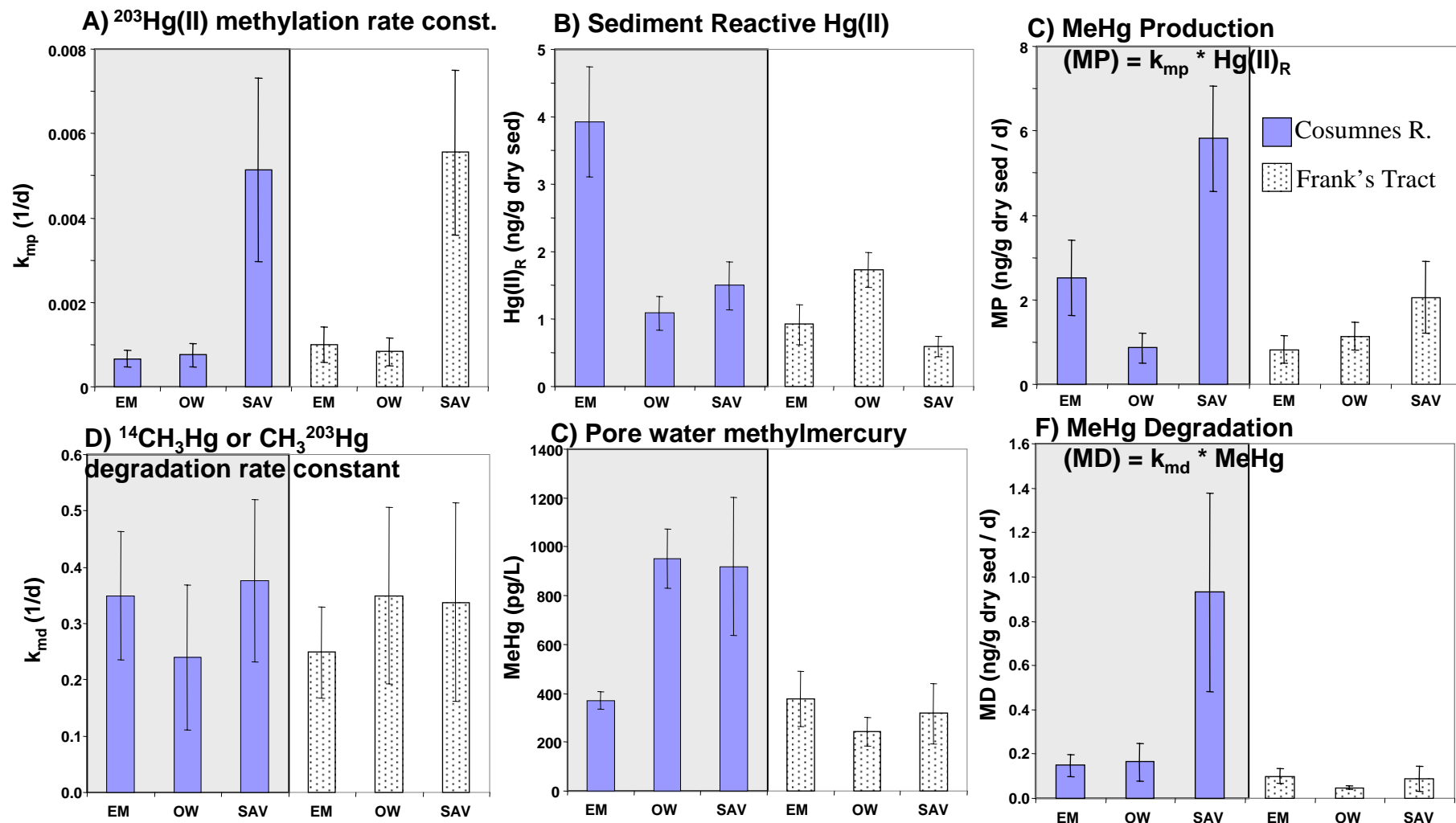


Figure HI-3. Average values for composite surface sediment samples collected from three sub-habitat types, Emergent Marsh (EM), Open Water (OW) and Submerged Aquatic Vegetation (SAV), in both study regions, during the first three field sampling events (Dec'03, Jun'04, and Mar'05). Parameters include A) $^{203}\text{Hg(II)}$ radiotracer derived methylmercury (MeHg) production rate constants, B) reactive inorganic mercury (Hg(II)_{R}), C) calculated rates of MeHg production, D) $^{14}\text{CH}_3\text{Hg}$ or $\text{CH}_3^{203}\text{Hg}$ radiotracer derived MeHg degradation rate constants, E) MeHg concentrations, and F) calculated potential rates of MeHg degradation.

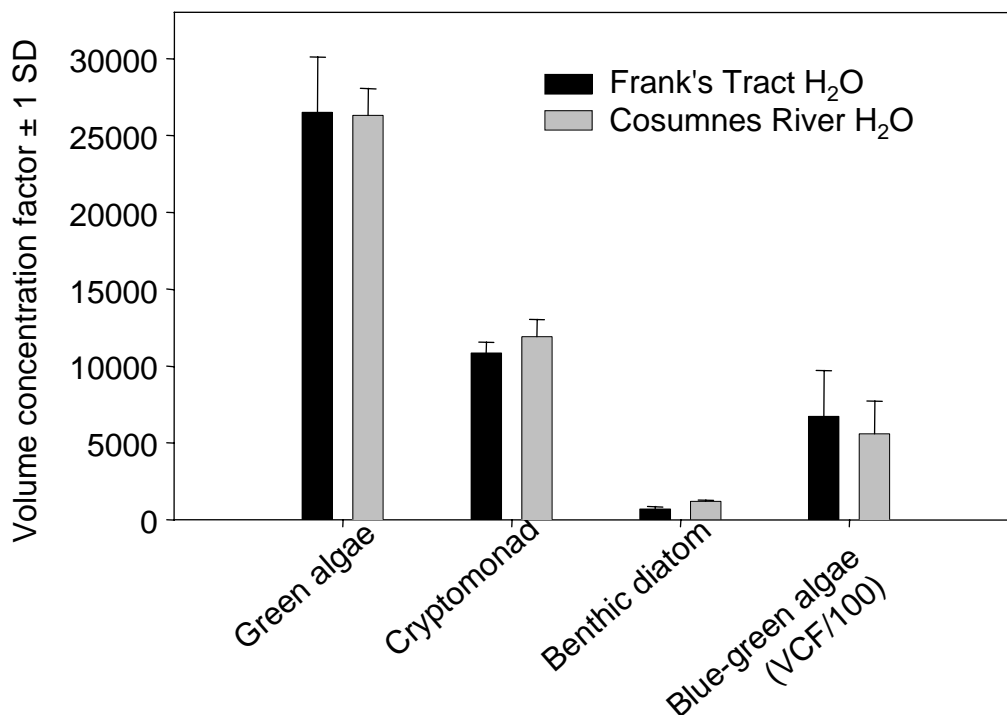


Figure HIV-1. Phytoplankton accumulation of inorganic Hg(II) using water collected from both study regions. No significant differences in bioconcentration between the two water types was observed.

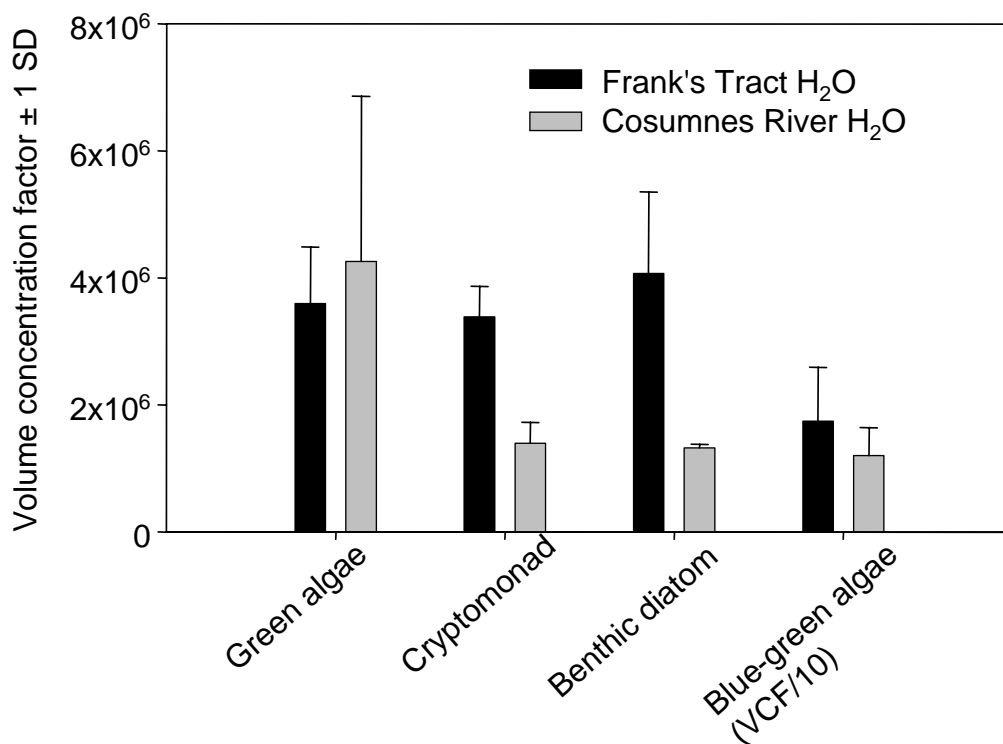


Figure HIV-2. Phytoplankton accumulation of MeHg using water collected from both study regions. A greater bioconcentration of MeHg was found in FT water for 2 out of 4 species.

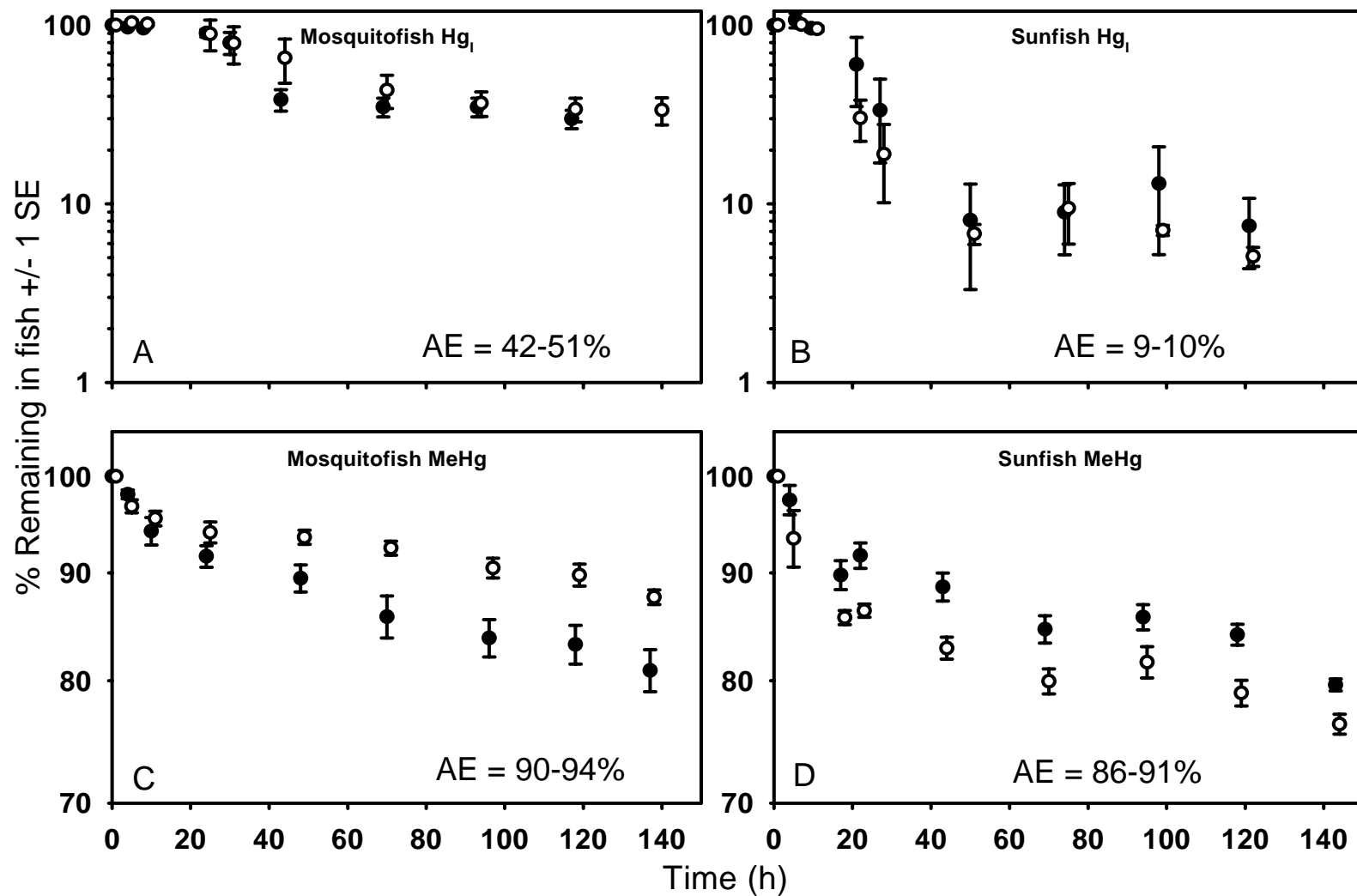


Figure HIV-3. Fish Depuration of $Hg(II)$ and MeHg From Dietary Exposures. No significant differences for $Hg(II)$, contrasting AE's for MeHg between H_2O types (Cosumnes River -closed circles, Frank's Tract - open circles).

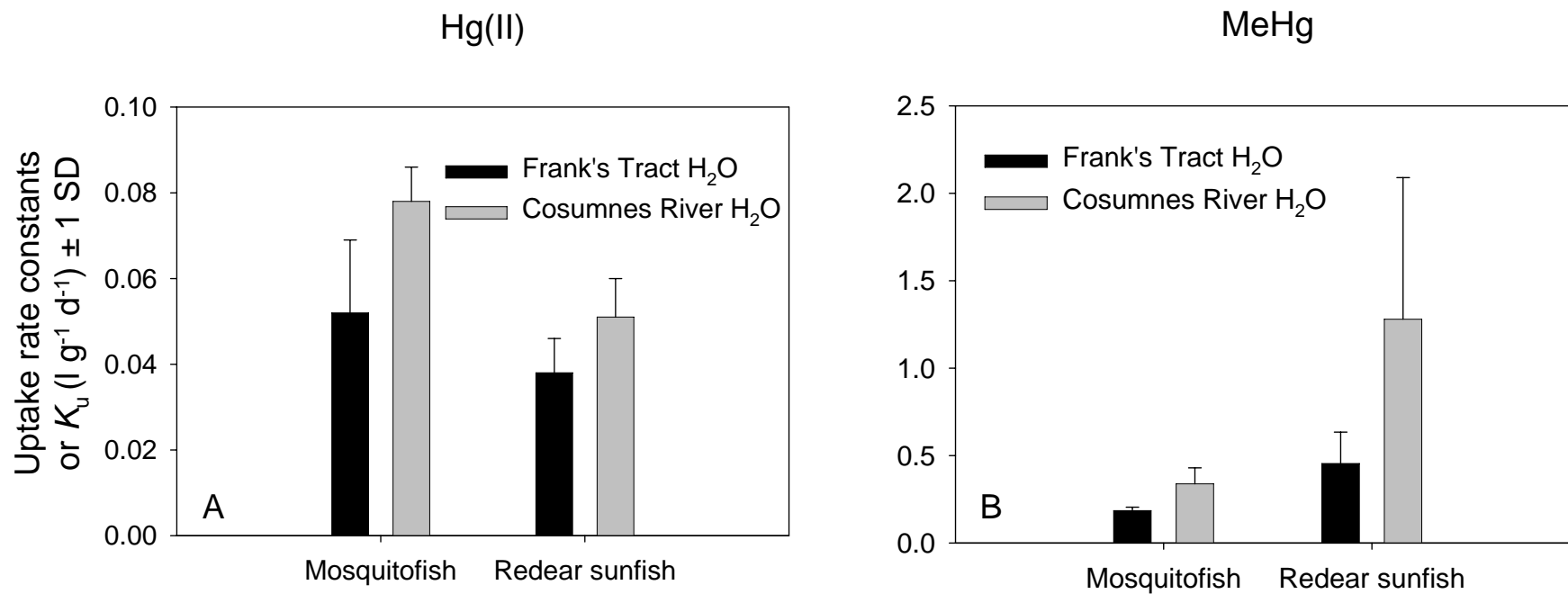


Figure HIV-4. Aqueous accumulation of Hg(II) and MeHg by 2 fish species in the 2 water types: Uptake rate constants were always greater in Cosumnes River water.

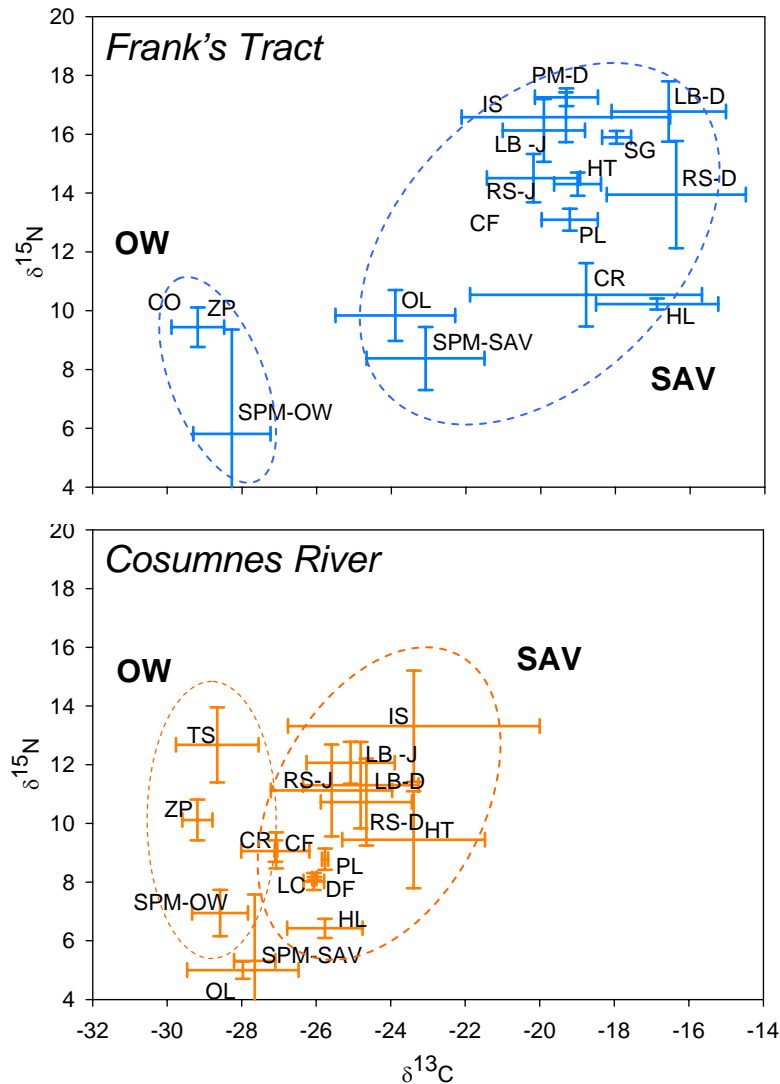


Figure HIV–5. Stable isotope plot showing feeding relationships among fish, invertebrates and suspended particulate matter (SPM) in Frank's Tract (blue) and Cosumnes River (orange) in December 2003. Values are means (\pm SD). Open water (OW) and submerged aquatic vegetation (SAV) food webs are operationally identified using stable isotopes. SPM-OW and SPM-SAV – open water and SAV SPM; OL – oligochaete *Sparganophilus eiseni*; HL – *Hyaella azteca*; CO – *Corophium* spp.; LC – leech, DF – Damsel fly, CR – Chironomidae midges; PL – flatworms; ZP – bulk zooplankton ($> 150 \mu\text{m}$); 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.

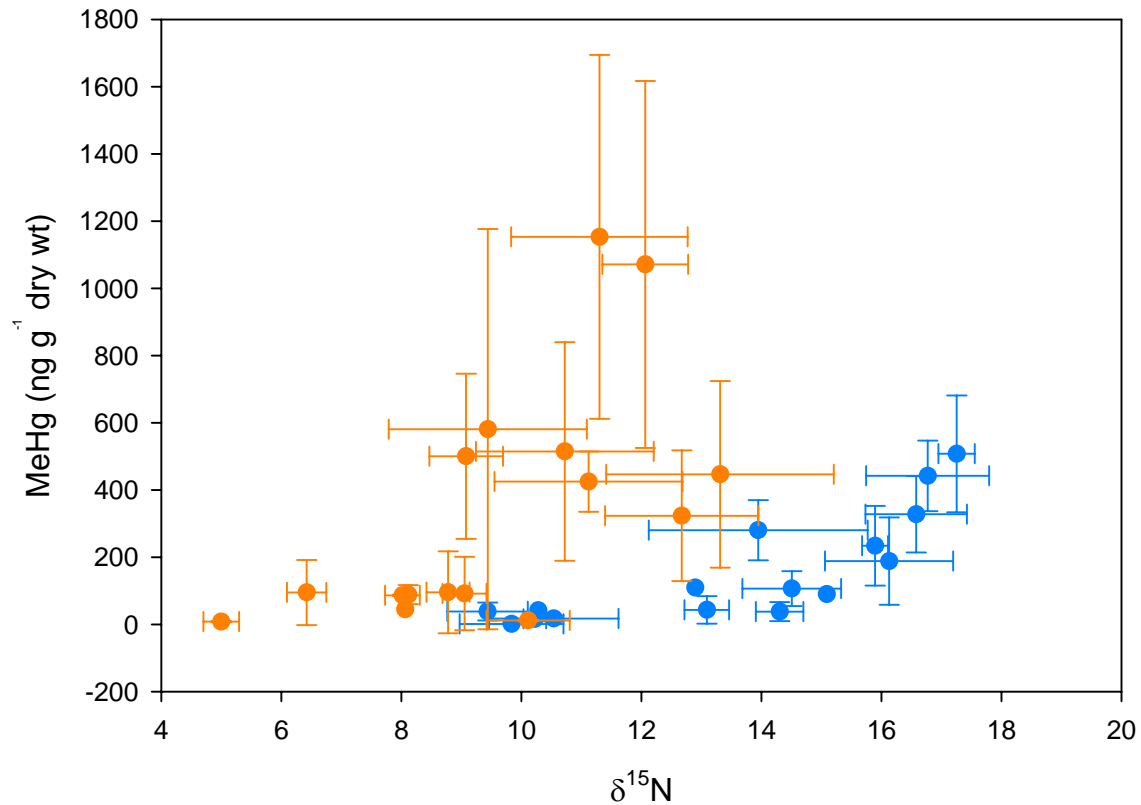


Figure HIV–6. Methyl mercury concentration as a function of trophic level (as measured by $\delta^{15}\text{N}$ stable isotope) in open water and SAV food webs (combined) from Frank's Tract (blue symbols) and Cosumnes River (orange symbols). Each point represents a mean (\pm SD) of fish species or composites of invertebrates within each region.

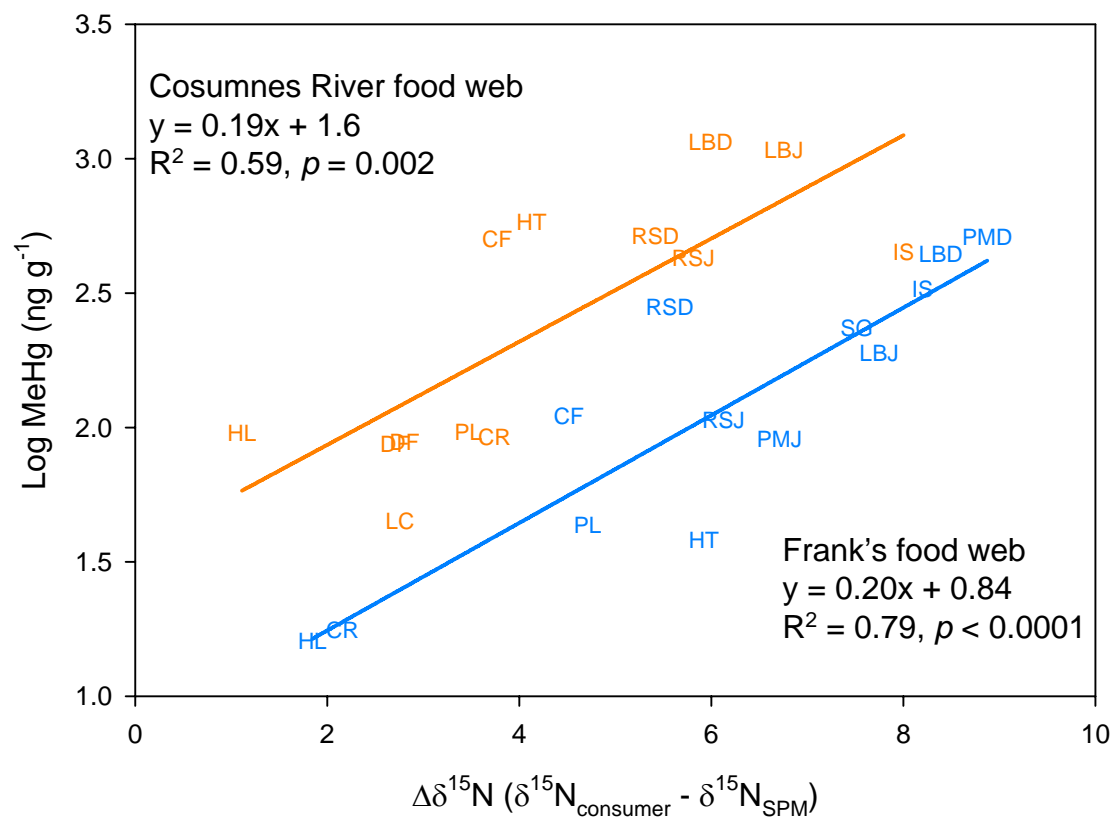


Figure HIV-7. Trophic enrichment of methyl mercury in SAV food webs in Frank's Tract (blue) and Cosumnes River (orange) in December 2003 and June 2004 (LB and RS only). Values are means for individual fish or invertebrate composites. HL – *Hyaella azteca*; LC – leech, DF – Damsel fly, CR – Chironomidae midges.; PL – flatworms; 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.

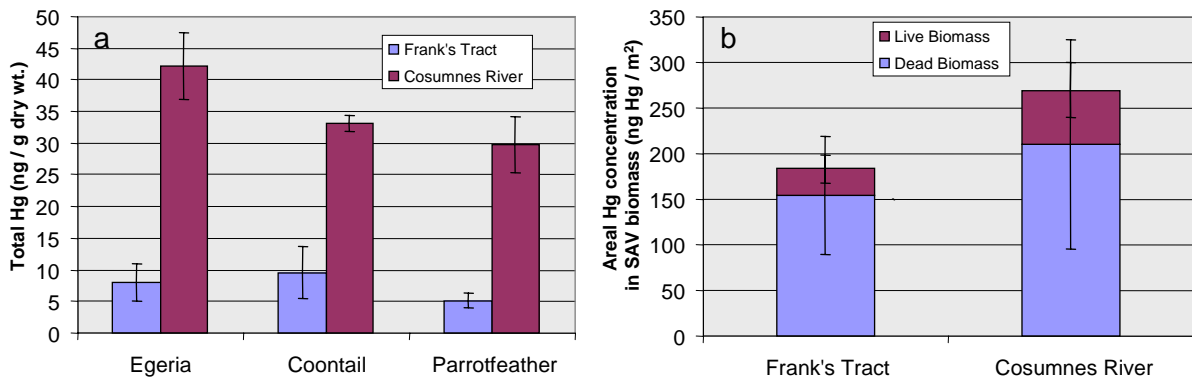


Figure HVI-1. a) Total Hg (T-Hg) concentrations in cleaned, EDTA-rinsed fresh *Egeria densa* from both study regions. b) areal pool size of THg in living and dead *Egeria*.

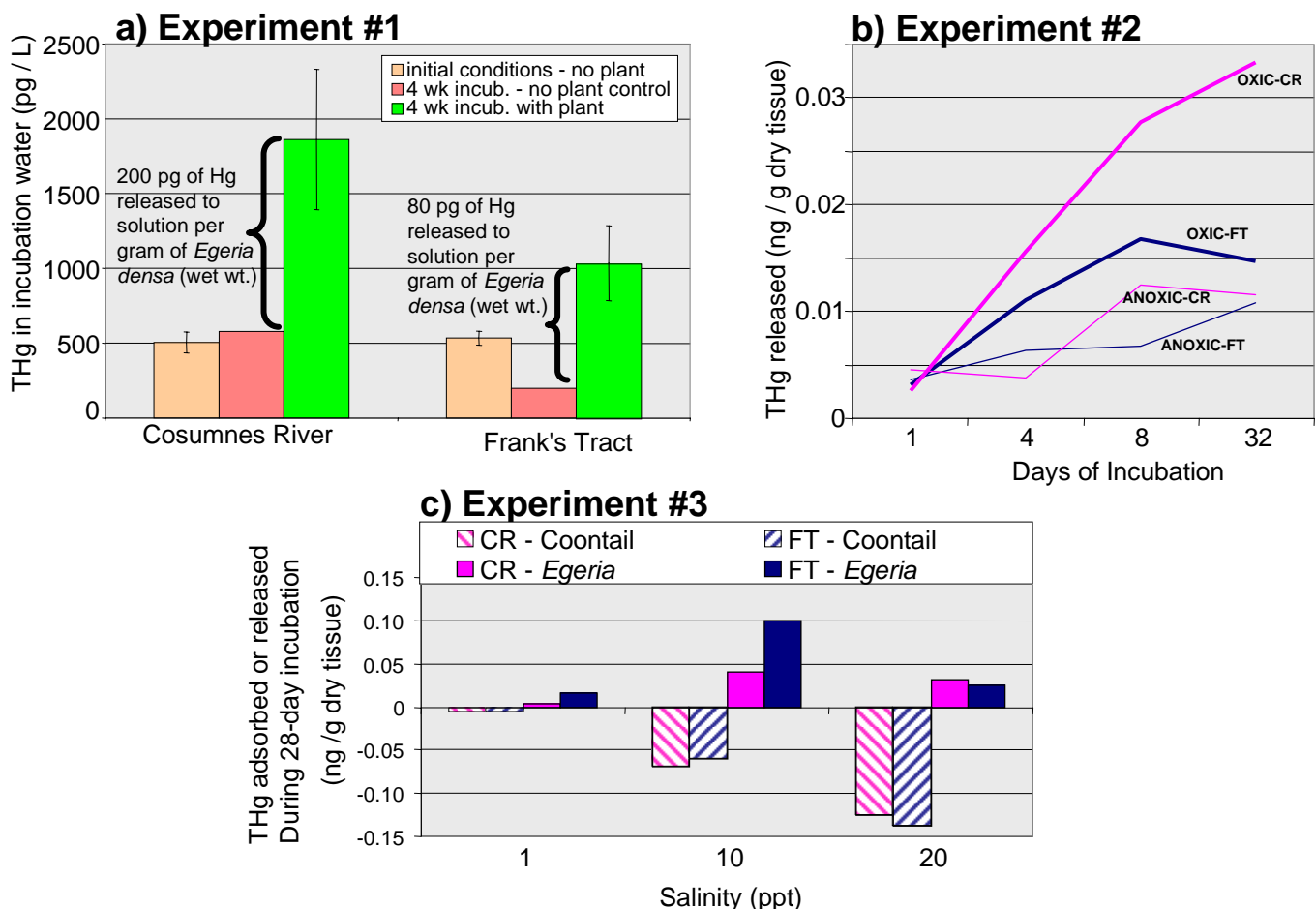


Figure HVI-2. Plant decomposition Experiments #1 thru #3 involving fresh, EDTA-rinsed segments of *Egeria* or Coontail incubated under various aqueous conditions at 30 °C with continuous shaking. a) Total Hg (THg) released from *Egeria* to the aqueous phase. Both incubation water and plant material was collected from the region indicated, in each case. b) Time course of THg released from *Egeria* under oxic and anoxic conditions using site specific water as the aqueous phase. c) the release or absorption of total-Hg from/to coontail and *Egeria* when incubated at different salinity levels in artificially prepared aqueous media.