Environmental Toxicology and Chemistry, Vol. 30, No. 8, pp. 1819–1826, 2011

Printed in the USA
DOI: 10.1002/etc.574

PCB-INDUCED CHANGES OF A BENTHIC COMMUNITY AND EXPECTED ECOSYSTEM RECOVERY FOLLOWING IN SITU SORBENT AMENDMENT

Elisabeth M.-L. Janssen,† Janet K. Thompson,‡ Samuel N. Luoma,‡ and Richard G. Luthy*†
†Department of Civil and Environmental Engineering, Stanford University, Stanford, California, USA
‡U.S. Geological Survey, Menlo Park, California

(*Submitted* 16 *December* 2010; *Returned for Revision* 31 *January* 2011; *Accepted* 5 *April* 2011)

Abstract—The benthic community was analyzed to evaluate pollution-induced changes for the polychlorinated biphenyl (PCB)-contaminated site at Hunters Point (HP) relative to 30 reference sites in San Francisco Bay, California, USA. An analysis based on functional traits of feeding, reproduction, and position in the sediment shows that HP is depauperate in deposit feeders, subsurface carnivores, and species with no protective barrier. Sediment chemistry analysis shows that PCBs are the major risk drivers at HP (1,570 ppb) and that the reference sites contain very low levels of PCB contamination (9 ppb). Different feeding traits support the existence of direct pathways of exposure, which can be mechanistically linked to PCB bioaccumulation by biodynamic modeling. The model shows that the deposit feeder *Neanthes arenaceodentata* accumulates approximately 20 times more PCBs in its lipids than the facultative deposit feeder *Macoma balthica* and up to 130 times more than the filter feeder *Mytilus edulis*. The comparison of different exposure scenarios suggests that PCB tissue concentrations at HP are two orders of magnitude higher than at the reference sites. At full scale, in situ sorbent amendment with activated carbon may reduce PCB bioaccumulation at HP by up to 85 to 90% under favorable field and treatment conditions. The modeling framework further demonstrates that such expected remedial success corresponds to exposure conditions suggested as the cleanup goal for HP. However, concentrations remain slightly higher than at the reference sites. The present study demonstrates how the remedial success of a sorbent amendment, which lowers the PCB availability, can be compared to reference conditions and traditional cleanup goals, which are commonly based on bulk sediment concentrations. Environ. Toxicol. Chem. 2011;30:1819–1826. © 2011 SETAC

Keywords—Functional ecology Bioavailability Sediment remediation Biodynamic modeling Benthic community

INTRODUCTION

Contaminated sediments that pose human health and ecological risks require cleanup actions that mitigate sediment toxicity and reduce exposure. The present study focuses on how sediment pollution by polychlorinated biphenyls (PCBs) may have changed the benthic community at Hunters Point (HP) in San Francisco Bay, California, USA, and to what degree sediment remediation might be necessary to allow for a recovery of the benthic community.

Contaminants in sediment that cause chronic toxicity can simplify community structures by reducing the abundance of sensitive species and increasing the abundance of tolerant species [1–3]. If changes in the benthic community can be attributed to major pollutants like PCBs, then sediment remediation strategies could more directly be evaluated based on their capability to mitigate exposure and risk and to allow for recovery. Even though general theories are being developed to use information about benthic community composition to evaluate ecosystem integrity and recovery, a mechanistic understanding of the physiological, ecological, and environmental characteristics involved remains to be developed [4–6].

Basing ecosystem recovery on principles of functional ecology offers a promising assessment method to identify pollution-induced changes in the benthic community structure [6,7]. Function-based analysis offers an enriched description of the benthic community by grouping species by their functional traits in addition to taxonomic analysis. This is also referred to as trait-based ecological risk assessment (TERA) [7]. While the response of one species in an ecosystem may not necessarily be representative of the response of the whole system, changes in the species groups performing a common ecosystem function can represent a serious alteration in ecological processes. Analysis by feeding traits is especially interesting because feeding groups reflect differences in dietary routes, and hence exposure to the contaminated environment [8–10]. In such circumstances, biodynamic modeling can estimate bioaccumulation and mechanistically reconcile dietary routes with contaminant exposure [11,12].

Sediment remediation should reduce risk by reducing exposure and then allow the benthic community to recover. An in situ sediment treatment with a strong sorbent like activated carbon is a novel and alternative remediation technique. The activated carbon repartitions hydrophobic organics like PCBs in the sediment. Consequently, total contaminant concentrations in the sediment are not altered but the presence of a strong sorbent like activated carbon lowers the availability of the hydrophobic contaminants from the sediment to water and biota [11,13,14]. Currently, bulk sediment concentrations are used to set sediment cleanup goals or risk thresholds like the sediment quality guidelines (SGQ [12,15,16]). To make the measure of bioavailability for in situ treatment more interpretable by sediment managers, a direct comparison between reduced bioavailability and reduced bulk sediment concentrations should be established. Furthermore, little attention has been devoted to the potential of sorbent amendments to result in recovery of the pollution-impacted benthic community. Small-scale, in situ studies show that sorbent amendments can reduce PCB bioaccumulation from sediment [13,17], but long-term monitoring

All Supplemental Data may be found in the online version of this article.
* To whom correspondence may be addressed (luthy@stanford.edu).
Published online 10 May 2011 in Wiley Online Library (wileyonlinelibrary.com).

data of ecosystem response after full-scale field amendments do not exist currently.

The present study suggests that sediment pollution changed the benthic community at HP. This hypothesis was tested by community analysis at HP and 30 reference sites based on functional ecology, which considers the organism's strategies for tolerating contaminant exposure. The present study further proposes that a modeling framework can link different feeding strategies to exposure. This second hypothesis is tested by employing a biodynamic model to estimate PCB availability and tissue concentrations for organisms with different feeding traits. Finally, the present study considers that the model can be employed to directly compare reduced PCB bioavailability with reduced PCB sediment concentrations. This third hypothesis was tested by modeling the reduction in PCB availability at HP that is required to achieve exposure conditions comparable to the reference sites and the cleanup goal where lower PCB sediment concentrations are present.

MATERIALS AND METHODS

Benthic community surveys

Benthic samples were collected in September 2008 at South Basin at HP, offshore at an intertidal mudflat. Additionally, benthic samples were collected at up to 30 shallow subtidal reference sites following the spring (April) and fall (August) recruitment periods in 2007 and at intertidal reference sites in August 2007 and 2008 (Fig. 1). The reference sites were sampled to determine the benthic community structure for the reference area and the benthic species pool that could be available for recruitment into HP. Therefore, the distribution of reference sites encompassed the spatial extent of sites where larvae could be delivered to HP within one tidal cycle (currents available at http://sfports.wr.usgs.gov/SFPORTS). The selection criteria for the reference sites were based on the physical habitat (water depth, temperature, tidal exposure, salinity, sediment grain size, and composition) to match the conditions at HP most closely but with minimal pollution of the sediment. The sediment at HP and the reference sites show similar total organic carbon (TOC) values with 1.4 ± 0.4 and $1.1 \pm 0.2\%$ and similar fraction of fines with 80 ± 20 and $77 \pm 11\%$, respectively. The reference sites were grouped into biotopes based on geographical proximity to evaluate spatial variability (Fig. 1, circled areas). More information about the geographical locations of the benthic sampling sites can be found in the Supplemental Data (Table S1). Four shallow intertidal reference sites that are most similar to water depth at HP were sampled in August 2007 and 2008. Few locations of this muddy high-intertidal habitat are present in Central San Francisco Bay and these sites represent a significant portion of this habitat.

Samples were collected with a $0.05\,m^2$ van Veen sampler, sieved through a 0.5 mm screen, preserved in 10% buffered formalin, and transferred to 70% ethyl alcohol after 1 to 2 weeks of preservation. Common organisms were identified to the species level. A few rare species were identified only to genus or family depending on the existing taxonomic knowledge. All sorting and taxonomic work was performed by Susan McCormick (Georgetown, California). Samples were double-sorted and voucher specimens maintained for future reference.

Benthic community analysis

Benthic communities were analyzed and compared on the basis of number of species, total abundance of organisms, the Shannon-Wiener Diversity index with standard error (Jackknife technique with $n = 10{,}000$, Pisces Conservation 2007 Software), and abundance of dominant species. Benthic communities were further compared using nonparametric multidimensional scaling with PRIMER 6 [18,19]. Abundance data for all species were square-root transformed and Bray-Curtis dissimilarities were computed between each pair of stations. The resulting matrix was ordinated by nonmetric multidimensional scaling to display the variation among the assemblages.

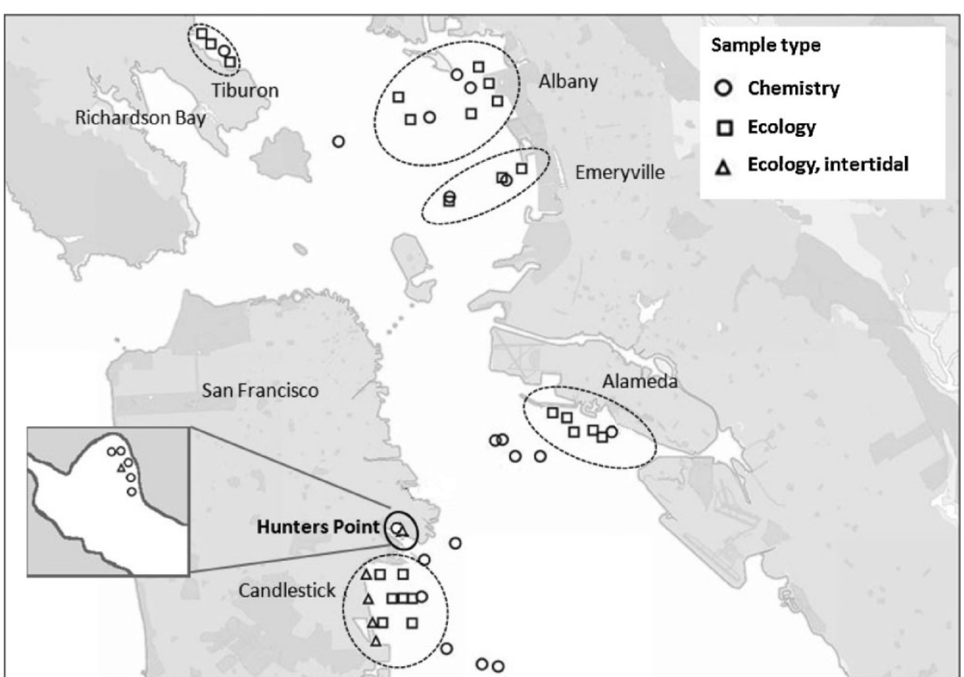

Fig. 1. Benthic survey sample sites (squares) including intertidal benthic sites (triangles) and sediment chemistry sites (circles) in the Central San Francisco Bay and at Hunters Point, CA, USA, regional biotopes are circled.

Functional ecology analysis

Benthic survey samples were further analyzed by means of functional ecology. Functional groups were defined for feeding mode, reproductive mode, and position in and protection from the sediment.

Feeding mode

Functional feeding groups refer to strategies of acquiring food and have important implications for the exposure to chemical pollution. Exposure can vary widely among filter feeders (also known as suspension feeders), deposit feeders, and carnivorous organisms based on their relative interaction and exposure to water as well as consumption of the sediment and prey that may contain varying concentrations of contaminants. Functional groups of feeding were defined with decreasing level of exposure as follows: subsurface carnivores (carnivores that feed on subsurface and some surface organisms); subsurface deposit feeders (species that feed by directly ingesting the sediment below the surface); surface deposit feeders (species that feed on organic matter that either grows on the surface such as benthic algae or is transported to the surface from the water column); surface carnivores (species that do not burrow below the surface of the sediment for their prey); filter and surface feeders combined (species that actively or passively remove particles from the water column and have the ability to harvest surface particles); and filter feeders only. The abundance and presence of species from subsurface and surface deposit feeders were combined because PCB concentrations in subsurface and surface sediments are similar, as previously observed at HP [13,20].

Reproductive mode

Reproductive strategy of a species determines how the offspring will be exposed to contaminants in their early developmental stage. Designation of reproductive strategy also considers the possibility that individuals have been transported as larvae from other, noncontaminated sites. Functional groups of reproduction were defined with expected decreasing level of exposure as follows: species that lay their eggs on the sediment surface with fully developed juveniles being released from the egg or species that produce clones by fission; species that brood their young away from the sediments and release fully functional juveniles; and species that broadcast their sperm and eggs into the water column and produce pelagic larvae or brood eggs and release pelagic larvae.

Position in and protection from sediment

A species' body position in the subsurface sediment, surface layer, free water column, or some combination of the above defines the organism's passive exposure to the contaminated environment. Functional positional groups were defined with decreasing level of exposure as follows: no barrier (species that expose their tissue directly to the environment and are free-living); tubed with tissue (bare animals with sediment tubes that usually have a mucous proteinaceous lining); chitin armor (species with similar tubes but with a chitinous body armor); chitin barrier and free-living (on the sediment surface); shell barrier (calcium carbonate); and cuticle covering (limited to nematodes). A detailed table of species identified in the benthic samples and their respective functional categories can be found in the Supplemental Data (Tables S2, S3).

Percentages of abundance were arcsin square-root transformed before an analysis of variance (ANOVA) test was performed to compare sites. Nematodes were analyzed separately because of their variety of life history patterns and the sampling method of the present study (Supplemental Data: Details on Nematodes).

Sediment characteristics

Information about contamination levels in the surface sediment (top 5 cm) were selected from the database of the San Francisco Estuarine Institute (SFEI, http://legacy.sfei.org) and the Validation Study for South Basin at HP [15] (Fig. 1). More information about the location and properties of the sediment sample sites can be found in the Supplemental Data (Tables S4–S7). Average values of chemical concentration were computed using all stations selected and compared to HP.

Concentrations of sediment contaminants other than total PCBs included heavy molecular weight polyaromatic hydrocarbons (HPAHs), light molecular weight PAHs (LPHAs), total dichloro-diphenyl-trichloroethanes (DDTs), dieldrin, Cu, Pb, As, Hg, and Ni. Sediment concentrations were subjected to a principal component analysis (Microsoft Excel XLSTAT, employing Pearson (n), significance level 0.05) to verify that the pollution levels at HP are distinctly different (higher) compared to the reference sites. Concentrations of all contaminants were further compared to effect-range low (ERL) and effect-range medium (ERM) values suggested as SQGs to identify possible pollutants of concern that distinguish HP from the reference sites. The contaminant-specific ERL and ERM values suggest threshold concentrations in sediment for the approximately 10th and 50th percentile concentrations, respectively, where adverse biological responses have been observed. Effect-range medium quotients (ERMq) were calculated by dividing the average sediment concentration of each contaminant by the respective ERM value to assess the number of pollutants that exceed the SQGs [21] (Fig. S1). The values of ERMq were then divided by the sum of all ERMq (ΣERMq) for each site to assess the relative contribution of each pollutant.

Ecosystem recovery scenarios

A biodynamic model was used to estimate total PCB tissue concentrations in three marine invertebrates. The model has been used previously for PCB bioaccumulation [11,12] and incorporates species-specific physiological parameters (e.g., feeding rates, growth, elimination rates, assimilation efficiency):

$$\frac{dC_{org}}{dt} = C_{sed} \cdot IR \cdot AE_{sed} + C_w \cdot k_w - C_{org,t} \cdot (k_e + k_g)$$

with C_{org} the PCB concentration in the organism (μg/g dry tissue); C_{sed} the PCB concentration in the sediment (μg/g dry weight); IR the ingestion rate (g particles / g dry weight · d); AE_{sed} the assimilation efficiency of PCBs from sediment [-]; k_w the aqueous uptake rate constant (L/g · d); C_w the aqueous PCB concentration (μg/L); k_e the rate constant of loss (1/d); and k_g the growth rate constant (1/d).

Feeding strategies

Polychlorinated biphenyl tissue concentrations were predicted for three invertebrates with different feeding strategies: the deposit feeding polychaete *Neanthes arenaceodentata*, the filter and surface-deposit feeding clam *Macoma balthica*, and the filter feeding mussel *Mytilus edulis*. The species- and PCB-specific physiological parameters used for the model predictions have been established earlier [11,12,17,22–26]. The exposure of filter feeding organisms through ingestion of particulate organic matter (POM) is variable, depending on origin, concentration, and composition of POM in water

Table 1. Polychlorinated biphenyl (PCB) sediment concentrations

Location	PCB concentration in sediment [ppb]
Hunters Point	1570 [15] [a]
Oakland Harbor (hot spot)	476 [28]
Clean-up goal	200 [27]
Effect-range medium	180 [16]
Effect-range low	23 [16]
Reference sites in Central Bay	9[b]

[a] Bracketed numbers are reference citations.
[b] San Francisco Estuary Institute.

[6]. The present study makes a conservative assumption that the PCB concentration of POM is similar to the PCB concentration of the surface sediment (Supplemental Data: Details on filter feeders).

Exposure conditions and expected remedial success

Various exposure conditions were tested to assess the response of PCB uptake relative to the dietary exposure pathway of the different model organisms. The exposure scenarios further allow comparing PCB tissue concentrations of organisms at HP to exposure conditions with different levels of PCB contamination. The contaminant tissue concentrations were calculated for the present condition at HP; the conditions at the selected reference sites; the ERL for PCBs; the ERM for PCBs; the PCB cleanup goal for HP [27]; and the present concentrations at a site in Oakland Harbor (sample station identification IT-6 and IM-1; [28]) where elevated PCB concentrations have been observed. Respective PCB sediment concentrations range over two orders of magnitude (Table 1). To estimate the PCB porewater concentration under different exposure conditions, general partitioning theory was applied. The average partitioning coefficient (K_d) was estimated as the ratio of total PCB sediment concentrations and porewater concentrations for HP as 1.01×10^5 L/kg (Supplemental Data, Table S8). The K_d values at HP range from 3.59×10^4 to 1.89×10^5 L/kg and we assume that the K_d values at the reference sites are within this range because the TOC content and fraction of fines is similar among all sites and similar equilibrium partitioning is inferred. The PCB concentration in the overlying water is approximately one order of magnitude lower than the porewater concentration [13] and was estimated accordingly. For the Oakland Harbor site, a comparable TOC content relative to HP has been reported but the fraction of fines is very low (23%, [28]). Nevertheless, the site was used as another exposure scenario in the modeling.

Biodynamic modeling (Eqn. 1) was further used to estimate the PCB tissue concentrations expected at HP after an activated carbon amendment. Information about the relative reduction of PCB availability after activated carbon addition was obtained for the three test species from the literature (Table 2). The expected remedial response was estimated for favorable field and treatment conditions; that is, the site should be depositional, should allow for mixing activated carbon into the upper sediment layer, and should be dominated by the fast desorbing fraction (nonblack carbon-like particles).

Finally, the model was used to approximate the reduction of PCB availability required to achieve tissue concentrations at HP that are comparable to tissue concentrations expected under other exposure scenarios (reference sites, SQG thresholds ERL and ERM, cleanup goal, Oakland Harbor, California, USA). The activated carbon amendment reduces the assimilation efficiency of PCB from sediment (AE_{sed}) as well as the aqueous PCB concentration (C_w). The required reductions in PCB availability is the fraction by which both parameters (AE_{sed} and C_w) have to be reduced, while keeping the total PCB sediment concentration at HP constant ($C_{sed} = 1570$ ppb).

RESULTS AND DISCUSSION

Benthic community surveys

The Shannon-Wiener species diversity index is one of the traditional ways of comparing community structures among different sites. The Shannon-Wiener index for the reference sites ranged from 2.89 ± 0.14 to 2.09 ± 0.35 in April and August 2007, respectively. The indices for the intertidal reference sites ranged from 1.84 ± 0.60 to 2.1 ± 0.24 in August 2007 and 2008, respectively. These reference site communities show similar dominant species. The most dominant species was the amphipod *Ampelisca abdita* (median abundance of subsamples >60% of the total abundance of subsamples), which was present at the majority of sites. Unlike the benthic community at HP, the second through tenth most abundant species at the reference sites have similarly high abundances (each >1% of the total abundance, Supplemental Data, Fig. S2).

The Shannon-Wiener species diversity index at HP (1.84 ± 0.05) was similar to the value at the intertidal reference sites. The benthic community at HP is dominated by four species of Corophidae amphipods (median abundance of samples >60% of the average total abundance of samples) and nematodes (median abundance of samples >30% of the average

Table 2. Values of physiological and model parameters for the benthic invertebrates

Parameter (unit)[a]	*Neanthes arenaceodentata*	*Macoma balthica*	*Mytilus edulis*
FR (L water/g dry wt/d)		2 [17] [b]	45 [24]
AE_{aq} (%)		50 [17]	20 [23]
k_w (L water/g dry wt/d, $= FR \times AE_{aq}$)	0.5 [12]	1	9
IR (g sediment/g dry wt/d)	3.5 [11]	0.25 [17]	0.02[c]
AE_s (%)	7 [11]	20 [17]	10 [22]
k_e (1/d)	0.04 [12]	0.05 [17]	0.144 [23]
k_g (1/d)	0 [11]		0.002 [25]
Expected time to attain steady state based on model predictions (d)	56	100 [17]	40
Lipid content of dry weight[d] (%)	5 [11,12]	18 [26]	10 [25]
Remedial success of AC (given as % reduction of bioaccumulation)	90 [11]	84 [17]	90 [25]

[a] FR = filtration rate; AE_{aq} = aqueous assimilation efficiency; k_w = aqueous uptake rate constant; IR = ingestion rate; AE_s = sediment assimilation efficiency; k_e = elimination rate constant; k_g = growth rate constant.
[b] Bracketed numbers are reference citations.
[c] Supplemental Data gives details on filter feeder.
[d] With the dry weight of the *N. arenaceodentata* being 10% of the wet weight.

total abundance of samples). Overall, the highest species abundances were seen at HP (average of >1,000 nematodes/ 0.05 m^2), assuming the nematodes were one species. An isopod (*Paranthura japonica*) and a polychaete (*Exogone lourei*) were the only other species with notable abundances (each >1% of the total abundance) and all remaining species were much less abundant at HP (Supplemental Data, Fig. S2).

Data in Fig. 2 show the number of species as a function of total abundance for all individual sites and medians for the biotopes. The benthic communities among the biotopes showed no significant difference for these measures. About 80% of all sites, including HP, had 20 to 30 species (reference sites April 2007: 26 ± 4.9 and August 2007: 21 ± 4.8; intertidal reference sites August 2007: 22 ± 4.8 and August 2008: 25 ± 5.2; HP August 2008: 28 ± 2).

The results of nonmetric multidimensional scaling shows that the reference sites and HP are not significantly different (stress >0.10) when analyzed for species and abundances (Supplemental Data Fig. S3). Although HP clusters separately from all reference sites and is most closely related to the intertidal reference sites, the distinctions are not sufficient to make the differences significant.

Details of species abundance at the sampling site are available at http://www.werc.usgs.gov/BenthicAtlas.

Functional ecology

Given the similar number of species and diversity indices between sites and the similarity between stations in the multidimensional scaling analysis, it might be concluded that the benthic communities did not differ greatly. However, the functional traits of the species at these sites reveal significant differences between the reference sites and HP. Data in Fig. 3 show the abundance of selected functional groups normalized by total community abundance for the reference communities and HP. Representative functional groups were selected that show the clearest pathway to contaminant exposure based on their direct interaction with the contaminated sediments, i.e., deposit feeders, subsurface carnivores, and species with no protective barrier.

The relatively low abundance (≤11%) of deposit feeders at HP relative to the reference sites (≤77%) supports the hypothesis that species that consume the sediment and live with some portion of their body within the sediment are less abundant and

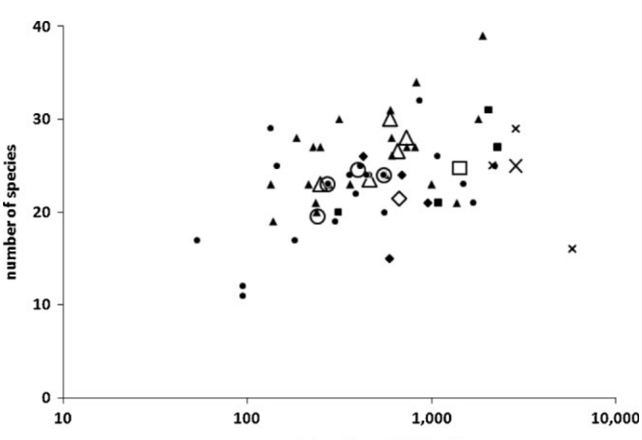

Fig. 2. Number of species relative to the total abundance for individual stations at reference sites April 2007 (triangles) and August 2007 (circles), intertidal reference sites August 2007 (diamonds) and August 2008 (squares), Hunters Point, CA, USA, August 2008 (×) and medians for each biotope with respective open symbols and Hunters Point (X).

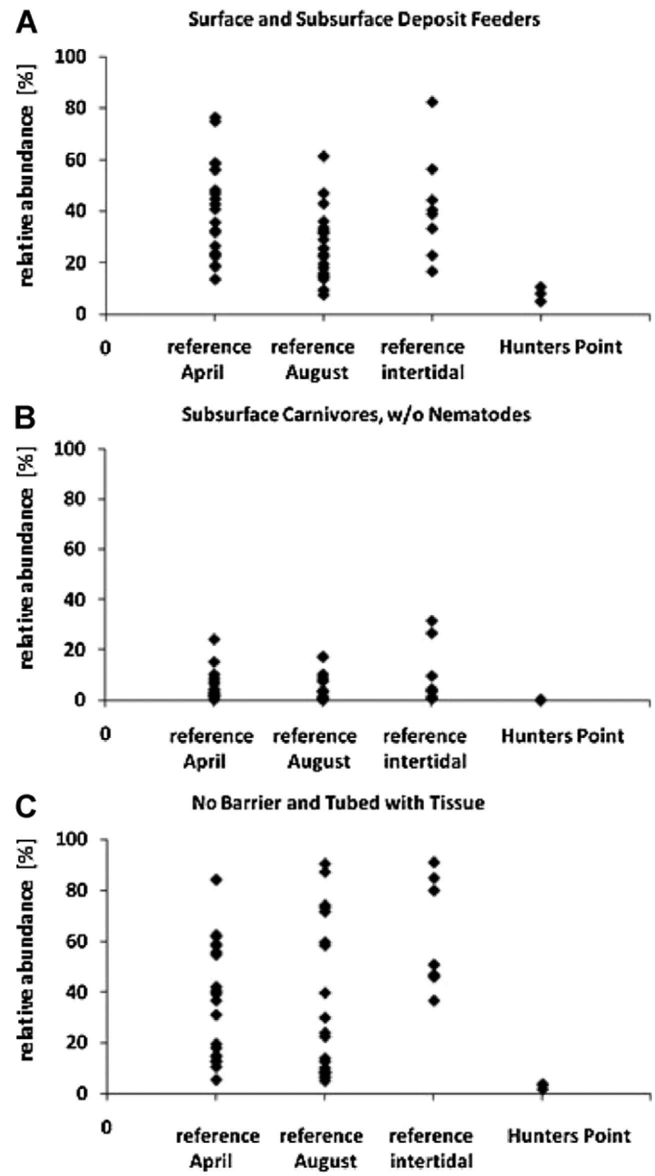

Fig. 3. Normalized abundance (%) of species and species groups as the ratio of individual abundance over total abundance for each sample station (n) at the reference sites in April ($n = 21$), August ($n = 20$), and at the intertidal sites ($n = 4$) compared to Hunters Point ($n = 3$).

differentially absent from HP (Fig. 3A). The percent abundance of deposit feeders at the three reference areas, seasons, and years was not significantly different from each other but different from the percent abundance at HP (ANOVA, $p < 0.05$, except comparison of HP and intertidal reference sites 2007: $p < 0.1$).

The relative abundance of carnivorous species was similar among reference sites but the reference sites were dissimilar from HP (ANOVA, $p < 0.05$, Fig. 3B). The overall abundance of subsurface carnivores was low, probably due to the trophic transfer efficiency typical of such communities (≈10%) [29]. The number of carnivore species (surface and subsurface) also shows fewer carnivores at HP (one species) than at the reference sites (three to six species at each site, ANOVA, $p < 0.01$). Subsurface carnivores may be less common at HP either because these species prey on organisms with high PCB exposure or due to lesser abundance of prey in the subsurface. Finally, data in Fig. 3C show that species without a protective barrier are significantly less common at HP (3%) than at the intertidal reference sites (August 2007: 66%; August 2008: 46%) and at the remaining reference areas (34%, ANOVA, $p < 0.05$).

The analysis shows that the benthic community at HP was entirely composed of surface feeders, whereas only 73% and 87% of the benthos at the intertidal reference sites were surface feeders in 2007 and 2008, respectively. The majority of the individuals at HP (97%) are ovoviparous or viviparous, with larvae being brooded and protected before being released as fully viable juveniles on the sediment surface. In contrast, this group makes up only 64% at the reference sites and 66% and 81% at the intertidal reference stations in 2007 and 2008, respectively.

Details for the total and relative abundances of the other functional groups can be found in the Supplemental Data (Figs. S4, S5).

Sediment characteristics

The contaminant concentrations are consistently lower at the reference sites compared to HP, except for PAHs (Fig. 4). Although the locations of the chemical and biological stations are not perfectly matched, the low concentrations and low variability of contaminant concentrations (reflected by error bars in Fig. 4) in the reference areas are consistent with the hypothesis that the benthic sampling sites represent a region of lower pollution than HP with similar sedimentary characteristics. Hunt et al. [30] also defined two stations in this region as appropriate reference sites for San Francisco Bay on the basis of low contaminant concentrations and low toxicity in traditional tests. Principal component analysis further supports that elevated pollution separates HP from the references area (Supplemental Data, Fig. S6, Table S9). Data in Fig. 4 show that the ERL is exceeded for all contaminants at the reference sites, except for total PCBs, LPAHs, and lead. Only Ni concentrations exceeded the ERM (51 ppm) by a factor of 1.5 at the reference sites. High Ni concentrations are expected throughout San Francisco Bay, including HP, most likely due to the dominance of ultramafic rocks (serpentine) in the local watershed [31].

Besides Ni, only PCBs and Hg concentrations at HP exceeded concentrations at the reference sites and the ERM of 180 ppb and 0.71 ppm, respectively. Concentrations at HP exceeded the ERM for PCBs by a factor 8.7 and the ERM for Hg by a factor of 1.5. The relative contribution of each contaminant to the sum of the ERMq (ΣERMq) is 57% for PCBs and 10% for Hg. At the reference sites PCBs only contribute 2% to the ΣERMq (Fig. S7). As shown by the data in Fig. 4, the total PCB concentrations reach $1,570 \pm 325$ ppb at HP and the average concentration at the reference sites is 8.8 ± 2 ppb; 0.6% of the concentration at HP.

Previous assessments of the response of macroinvertebrates to sediment contaminants in San Francisco Bay suggest some pollution-induced impacts in the bay as a whole [5,10,32]. However, a common challenge is that observed impacts covaried with multiple pollutants and other environmental factors, such as salinity and sediment texture [5,6,10], and could not be directly linked to specific contamination. Here, no correlation between texture and contamination was found at the study sites (Supplemental Data, Figs. S8, S9).

The ERMq's are not a direct proof of casualty, but SQGs are often employed to compare different levels of contamination and dominant pollutants where physical characteristics of the sediments are similar [16,21,30]. Here, ERMq's strongly support the concept that PCBs are probably the major risk drivers that differentiate the references sites from HP. Polychlorinated biphenyls are persistent, hydrophobic, and bioaccumulative contaminants, which are widely distributed and listed as endocrine disrupters and probable carcinogens to humans [33]. Even when aqueous PCB concentrations in overlying water are below the acutely toxic level ($\approx$7 ppm [1]), chronic effects have been observed on organism growth, reproduction, survival, and life span as well as teratogenic effects in wildlife [34–36].

Ecosystem recovery scenarios

The PCB tissue concentrations predicted with the biodynamic model are presented for different feeding strategies and exposure scenarios.

Feeding strategies. Bioaccumulation in the organisms' lipids increases linearly with PCB exposure (Fig. 5). Data in Fig. 5

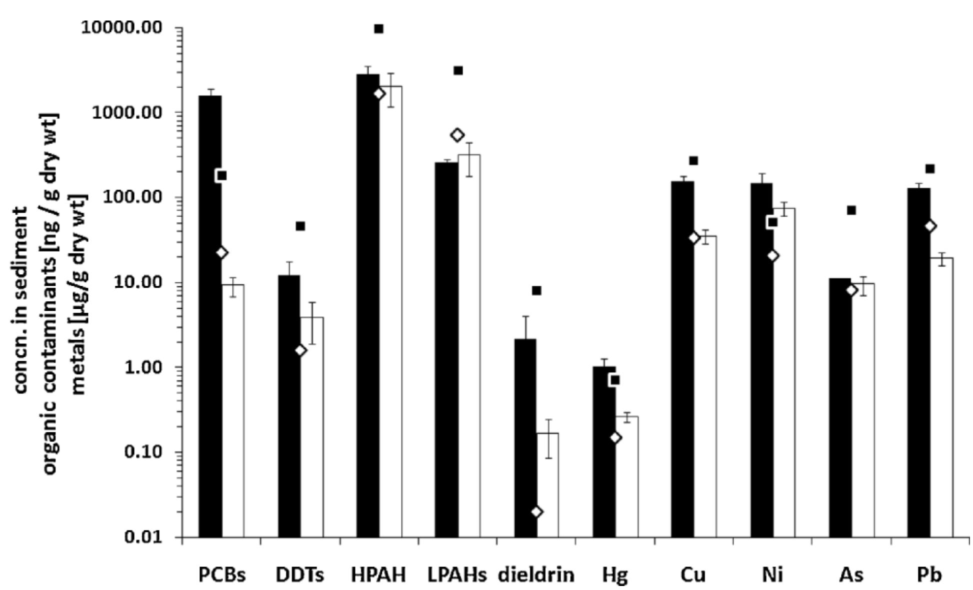

Fig. 4. Average sediment concentrations of predominant pollutants at Hunters Point (solid bar) and on average at the reference sites in the Central Bay (open bar), CA, USA. The sediment quality guidelines thresholds of contaminant-specific ERL (open diamond) and ERM (solid square) are shown for organic contaminants (ppb) and metals (ppm). Error bars represent one standard deviation. Note the logarithmic scale.

Environ. Toxicol. Chem. 30, 2011

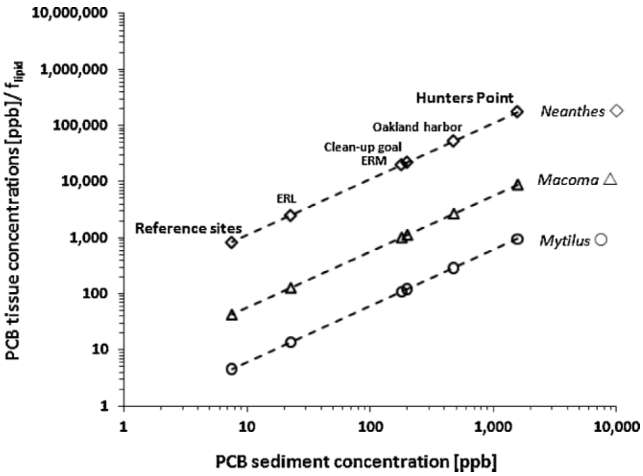

Fig. 5. Lipid-normalized PCB tissue concentrations estimated with the biodynamic model for *Neanthes arenaceodentata* (open diamond, deposit feeder), *Macoma balthica* (open triangle, surface deposit and filter feeder), *Mytilus edulis* (open circle, filter feeder) corresponding to the six different exposure scenarios and for Hunters Point, CA, USA. ERL = effect-range low; ERM = effect-range medium.

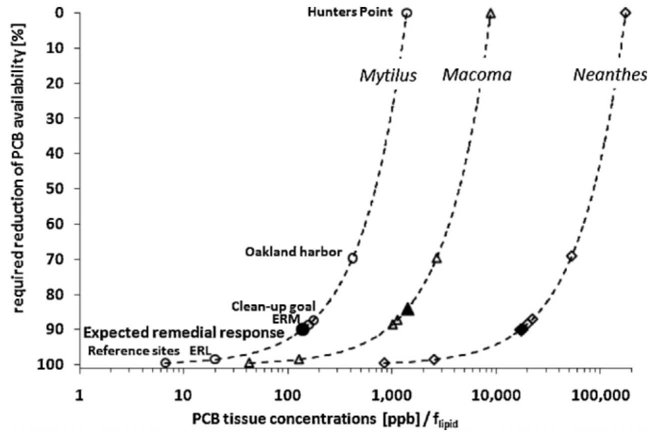

Fig. 6. Required reduction of PCB availability at Hunters Point, CA, USA, as estimated with the biodynamic model to achieve desired lipid-normalized PCB tissue concentrations for *Neanthes arenaceodentata* (open diamond, deposit feeder), *Macoma balthica* (open triangle, surface deposit and filter feeder), *Mytilus edulis* (open circle, filter feeder) corresponding to the six different exposure scenarios and the expected remedial response (full symbols) and for Hunters Point. ERL = effect-range low; ERM = effect-range medium.

also show the steady-state concentration in each of three species that might be expected at several sites with different PCB sediment concentrations in San Francisco Bay and at the SQG thresholds (ERL and ERM). The model predicts that PCB tissue concentrations are two orders of magnitude less at the reference sites compared to HP for all three feeding groups. The PCB tissue concentrations predicted for exposure conditions at the reference sites are slightly lower than for exposure to sediment with PCB concentrations at the ERL threshold. These predictions for the reference sites are only approximately 0.5% of the body burden expected at HP.

The polychaete *N. arenaceodentata* acquires food by bulk deposit feeding and accumulates the highest amount of PCBs. Polychlorinated biphenyl concentration in lipids are approximately 20 times higher in this deposit feeder compared to the facultative deposit feeding clam *M. balthica*. This clam shows a higher overall lipid content (18%) than the deposit feeder (5%) and interacts less with the contaminated sediment because the ingestion rate (IR) is approximately an order of magnitude lower. The filter-feeding mussel *M. edulis* accumulates 130 times less PCBs than the deposit feeder and approximately 6 times less than the facultative deposit feeding clam. Even though the mussel's filtration rate (FR) is approximately 20 times higher compared to the clam's, the mussel's PCB uptake by sediment ingestion is significantly lower (compare Table 2). *Mytilus edulis* is also most efficient in eliminating PCBs, i.e., highest elimination rate constant (k_e) of the three species.

The comparison of bioaccumulation among these species demonstrates how their different functional attributes lead to different levels of exposure and internal tissue concentrations.

Exposure scenarios and expected remedial success. The biodynamic model was further used to approximate the reduction of PCB availability required at HP to achieve tissue concentrations expected at the other sites and the SQG thresholds. Data in Fig. 6 demonstrate that the PCB availability has to be reduced by 60 to 75% to attain PCB exposure at HP that is similar to the conditions at the Oakland harbor site. A 90% reduction of PCB availability would achieve exposure conditions at HP that would comply with the SQGs (ERM) and the cleanup goal for HP. To lower PCB tissue concentrations to levels estimated for the reference sites, the PCB availability has to be reduced by 99.5%, which is probably not feasible with the tested activated carbon amendment in the field. Previous studies have shown that the expected remedial response upon activated carbon amendment at HP would reduce PCB availability by up to 85 to 90% [12,24,37] for the three model organisms under ideal field and treatment conditions. This remedial response essentially corresponds to lowering PCB sediment concentrations from the present 1,570 ppb to 150 ppb, which would comply with the cleanup goal for HP (200 ppb).

CONCLUSION

The comparison of benthic communities based on functional ecology proved to be a sensitive approach to identify pollution-induced changes at HP and to identify species groups that are affected. The community differences among the contaminated site and reference conditions were detectable only because of the careful choice of multiple reference sites with similar physical habitats representing background condition with relatively low pollution. Different feeding traits can be linked to different levels of PCB bioaccumulation by employing the biodynamic model, which reflects that the deposit feeder accumulates 20 times more PCBs that the facultative deposit feeder and 130 times more PCBs than the filter feeder. Because PCBs appear to be the major risk drivers that differentiate the reference sites from HP, sediment remediation to reduce PCB availability should benefit the benthic community. The expected remedial response of an area-wide, in situ sorbent amendment with activated carbon may reduce the PCB availability at HP by 85 to 90%. Polychlorinated biphenyl tissue concentrations would remain slightly higher than at the reference sites but would comply with the cleanup goal for HP and the SQGs (ERM). The predicted reduction of bioaccumulation is expected to be indicative of the community response. Remediating the PCB hotspot at HP could ultimately narrow the exposure to fish bay-wide, as recently demonstrated with a PCB food web model, which also would lower risk and benefit human health [38]. This type of comparative analysis of bioavailability and bulk sediment concentrations can assist decision-making for the cause of action for sediment remediation at cleanup sites in general and for sorbent amendments specifically, a promising yet novel remediation technique being evaluated at several pilot tests in the field [39].

SUPPLEMENTAL DATA

The Supplemental Data includes nine tables and nine figures. (1,622 KB).

Acknowledgement—Project funding was provided through the Strategic Environmental Research and Development Program (SERDP, ER-1552) and the U.S. Geological Survey Toxic Substances Hydrology Program.

REFERENCES

1. Fuchsman PC, Barber TR, Lawton JC, Leigh KB. 2006. An evaluation of cause-effect relationships between polychlorinated biphenyl concentrations and sediment toxicity to benthic invertebrates. *Environ Toxicol Chem* 25:2601–2612.
2. Pearson T, Gray J, Johannessen P. 1983. Objective selection of sensitive species indicative of pollution-induced changes in benthic communities. 2. Data analysis. *Mar Ecol-Prog Ser* 12:237–255.
3. Hyland JL, Balthis WL, Engle VD, Long ER, Paul JF, Summers JK, Van Dolah RF. 2003. Incidence of stress in benthic communities along the U.S. Atlantic and Gulf of Mexico coasts within different ranges of sediment contamination from chemical mixtures. *Environ Monit Assess* 81:149–161.
4. Pachepsky E, Crawford JW, Bown JL, Squire G. 2001. Towards a general theory of biodiversity. *Nature* 410:923–926.
5. Thompson B, Adelsbach T, Brown C, Hunt J, Kuwabara J, Neale J, Ohlendorf H, Schwarzbach S, Spies R, Taberski K. 2007. Biological effects of anthropogenic contaminants in the San Francisco Estuary. *Environ Res* 105:156–174.
6. Burton GA, Johnston EL. 2010. Assessing contaminated sediments in the context of multiple stressors. *Environ Toxicol Chem* 29:2625–2643.
7. Baird DJ, Rubach MN, Van den Brink PJ. 2008. Trait-based ecological risk assessment (TERA): The new frontier? *Integr Environ Assess Manag* 4:2–3.
8. Horne MT, Finley NJ, Sprenger MD. 1999. Polychlorinated biphenyl- and mercury-associated alterations on benthic invertebrate community structure in a contaminated salt marsh in southeast Georgia. *Arch Environ Contam Toxicol* 37:317–325.
9. Pearson TH, Rosenberg R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr Mar Biol Ann Rev* 16:229–311.
10. Thompson B, Lowe S. 2004. Assessment of macrobenthos response to sediment contamination in the San Francisco Estuary, California, USA. *Environ Toxicol Chem* 23:2178–2187.
11. McLeod PB, Luoma SN, Luthy RG. 2008. Biodynamic modeling of PCB uptake by *Macoma balthica* and *Corbicula fluminea* from sediment amended with activated carbon. *Environ Sci Technol* 42:484–490.
12. Janssen EML, Croteau MN, Luoma SN, Luthy RG. 2010. Measurement and modeling of polychlorinated biphenyl bioaccumulation from sediment for the marine polychaete *Neanthes arenaceodentata* and response to sorbent amendment. *Environ Sci Technol* 44:2857–2863.
13. Cho YM, Smithenry DW, Ghosh U, Kennedy AJ, Millward RN, Bridges TS, Luthy RG. 2007. Field methods for amending marine sediment with activated carbon and assessing treatment effectiveness. *Mar Environ Res* 64:541–555.
14. Millward RN, Bridges TS, Ghosh U, Zimmerman JR, Luthy RG. 2005. Addition of activated carbon to sediments to reduce PCB bioaccumulation by a polychaete (*Neanthes arenaceodentata*) and an amphipod (*Leptocheirus plumulosus*). *Environ Sci Technol* 39:2880–2887.
15. Battelle E, Neptune Co. 2004. Hunters Point Shipyard Parcel F Validation Study Draft Final Report. San Francisco Bay, CA, USA.
16. Long ER, MacDonald DD, Smith SL, Calder FD. 1995. Incidence of adverse biological effects within rages of chemical concentrations in marine and estuarine sediments. *Environ Manage* 19:81–97.
17. Janssen EML, Oen AMP, Luoma SN, Luthy RG. 2011. Assessment of field-related influences on polychlorinated biphenyl exposures and sorbent amendment using polychaete bioassays and passive sampler measurements. *Environ Toxicol Chem* 30:173–180.
18. Clark KR, Warwick R. 2001. Change in marine communities: An approach to statistical analysis and interpretation. Plymouth Marine Laboratory, Plymouth, UK.
19. Clarke K. 1993. Non-parametric multivariate analyses of changes in community structure. *Aust J Ecol* 18:117–143.
20. Cho YM, Ghosh U, Kennedy AJ, Grossman A, Ray G, Tomaszewski JE, Smithenry DW, Bridges TS, Luthy RG. 2009. Field application of activated carbon amendment for in-situ stabilization of polychlorinated biphenyls in marine sediment. *Environ Sci Technol* 43:3815–3823.
21. Long ER, Ingersoll CG, MacDonald DD. 2006. Calculation and uses of mean sediment quality guideline quotients: A critical review. *Environ Sci Technol* 40:1726–1736.
22. Bjørk M, Gilek M. 1999. Efficiency of polychlorinated biphenyl assimilation from water and algal good by the blue mussel (*Mytilus edulis*). *Environ Toxicol Chem* 18:765–771.
23. Bjørk M, Gilek M. 1997. Bioaccumulation kinetics of PCB 31, 49, and 153 in the blue mussel, *Mytilus edulis* L. as a function of algal food concentration. *Aquat Toxicol* 38:110–123.
24. Tomaszewski JE, McLeod PB, Luthy RG. 2008. Measuring and modeling reduction of DDT availability to the water column and mussels following activated carbon amendment of contaminated sediment. *Water Res* 42:4348–4356.
25. Gilek M, Bjørk M, Nåf C. 1996. Influence of body size on the uptake, depuration, and bioaccumulation of polychlorinated biphenyl congeners by Baltic Sea blue mussels, *Mytilus edulis*. *Mar Biol* 125:499–510.
26. Wenne R, Polak L. 1989. Lipid composition and storage in the tissues of the bivalve, *Macoma balthica*. *Biochem Syst Ecol* 17:583–587.
27. Brajas & Associates, Tetra Tech EM. 2008. Final Feasibility Study Report for Parcel F. Hunters Point Shipyard, San Francisco, California, San Francisco, CA, USA.
28. McFarland VA, Clarke JU, Lutz CH, Jarvis S, Mulheam B, Reilly FJ. 1994. Bioaccumulation potential of contaminants from bedded and suspended Oakland harbor deepening project sediments to San Francisco Bay flatfish and bivalve mollusks, San Francisco, CA, USA. Paper EL-94-7. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.
29. Lindeman RL. 1942. The trophic-dynamic aspect of ecology. *Ecology* 23:399–417.
30. Hunt JW, Anderson BS, Phillips BM, Newman J, Tjeerdema RS, Fairey R, Puckett HM, Stephenson M, Smith RW, Wilson CJ, Taberski KM. 2001. Evaluation and use of sediment toxicity reference sites for statistical comparisons in regional assessments. *Environ Toxicol Chem* 20:1266–1275.
31. Hornberger MI, Luoma SN, van Geen A, Fuller C, Anima R. 1999. Historical trends of metals in the sediments of San Francisco Bay, California. *Mar Chem* 64:39–55.
32. Davis JA, Hetzel F, Oram JJ, McKee LJ. 2007. Polychlorinated biphenyls (PCBs) in San Francisco Bay. *Environ Res* 105:67–86.
33. U.S Environmental Protections Agency. 1996. Integrated Risk Information System, PCBs. CASN 1336-36-3. Washington, DC.
34. Fisher SW, Chordas SW, Landrum PF. 1999. Lethal and sublethal body residues for PCB intoxication in the oligochaete, *Lumbriculus variegatus*. *Aquat Toxicol* 45:115–126.
35. Murdoch M, Chapman P, Johns D, Paine M. 1997. Chronic effects of organochlorine exposure in sediment to the marine polychaete Neanthes arenaceodentata. *Environ Sci Technol* 16:1494–1503.
36. Landrum PF, Faust WR, Eadie BJ. 1989. Bioavailability and toxicity of a mixture of sediment-assiciated chlorinated hydrocarbons to the amphipod *Pontoporeis hoyi*. In Cowgill UM, Williams LR, eds, *Aquatic Toxicology and Hazard Assessment*, Vol 12. American Society for Testing and Materials, Philadelphia, PA, pp 315–329.
37. McLeod PB, Van den Heuvel-Greve MJ, Luoma SN, Luthy RG. 2007. Biological uptake of polychlorinated biphenyls by *Macoma balthica* from sediment amended with activated carbon. *Environ Toxicol Chem* 26:980–987.
38. Gobas FAPC, Arnot JA. 2010. Food web bioaccumulation model for polychlorinated biphenyls in San Francisco Bay, California USA. *Environ Toxicol Chem* 29:1385–1395.
39. Ghosh U, Luthy RG, Cornelissen G, Werner D, Menzie CA. 2011. In-situ sorbent amendments: A new direction in contaminated sediment management. *Environ Sci Technol* 45:1163–1168.