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Influences of Dietary Uptake and Reactive Sulfides on Metal Bioavailability from Aquatic Sediments

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Understanding how animals are exposed to the large repository of metal pollutants in aquatic sediments is complicated and is important in regulatory decisions. Experiments with four types of invertebrates showed that feeding behavior and dietary uptake control bioaccumulation of cadmium, silver, nickel, and zinc. Metal concentrations in animal tissue correlated with metal concentrations extracted from sediments, but not with metal in porewater, across a range of reactive sulfide concentrations, from 0.5 to 30 micromoles per gram. These results contradict the notion that metal bioavailability in sediments is controlled by geochemical equilibration of metals between porewater and reactive sulfides, a proposed basis for regulatory criteria for metals.

A central problem in biogeochemistry and environmental management is understanding the uptake and transfer into food chains of aquatic contaminants, including heavy metals. Toxic metals introduced into aquatic environments by human activities typically accumulate in sediments. A common notion is that the association of metals with reactive sulfides controls toxicity in sediments by controlling porewater metal concentrations. An acid-extractable fraction of (iron) sulfides, called AVS (acid-volatile sulfide), can form thermodynamically stable metal sulfide precipitates in sediments, and thereby governs the behavior of divalent metals such as Cd, Cu, Ni, Pb, and Zn (*1*, *2*). Acute metal toxicity to benthic animals did not occur in experiments when there was sufficient AVS in the sediment to sequester all of the simultaneously extracted metal [termed SEM or EM (*3*)]; that is, when [EM − AVS] < 0 or EM/AVS < 1 (*2*, *4*). These experiments typically used conditions that increased the likelihood that porewater metal concentrations were high and that porewater controlled metal bioavailability. To evaluate bioavailability more realistically, we studied bioaccumulation of environmentally relevant concentrations of metals (Cd, Ni, and Zn), and we used a sediment column with vertical stratification of oxygen concentrations, which is typical of natural conditions. The ratios of porewater to sediment metal concentrations in our experiments were in the range of those commonly observed in nature, and experimental conditions were such that the animals were likely to feed on the sediments.

We examined Cd, Ni, and Zn accumulation in four types of benthic invertebrates: the filter-feeding clam *Potamocorbula amurensis,* the facultative deposit-feeding clam *Macoma balthica,* the surface deposit-feeding worm (polychaete) *Neanthes arenaceodentata,* and the head-down deep deposit–feeding polychaete *Heteromastus filiformis*. All are common in estuaries. Sulfide-rich anoxic sediment was obtained from a tidal mudflat in San Francisco Bay and was held under anoxic conditions (mixed with deaerated filtered seawater and kept under a N_2 atmosphere). In order to vary AVS, an aliquot of this sediment was mixed with 0.45-μm filtered seawater (salinity 25) and oxidized by bubbling continuously with air for 3 days. The AVS was readily oxidized and declined from 30 to 0.5 μmol g^{-1}. Oxidized and anoxic sediments were mixed at varying ratios to produce four levels of AVS (0.5, 7.5, 15, and 30 μmol g^{-1}). In one series of experiments, sediments containing a single nominal AVS concentration of 7.5 μmol g^{-1} were enriched with four levels of a Cd-Ni-Zn mixture (*5*). Unenriched sediment (AVS = 7.5 μmol g^{-1}) was used as a control. In a second series, the four nominal AVS treatments each received one concentration of the metal mixture (*5*). Before the introduction of test animals, a vertical redox gradient was established in each experimental microcosm by 1 week of equilibration with an oxidized water column (*6*). AVS, EM, and porewater metal concentrations were determined at the beginning and end of the 18-day bioaccumulation experiment (*7*). Animals were allowed to defecate their gut contents before tissue metal analysis.

Results for Cd were illustrative of those for all three metals (Fig. 1). Both clams accumulated significantly more Cd in all treatments than did controls, although the molar concentrations of extractable Cd in sediment (0.02 to 0.2 μmol g^{-1}) were substantially lower than the AVS concentrations (0.5 to 30 μmol g^{-1}). In all treatments, [EM − AVS] was less than zero, and concentrations of Cd in porewater were low (Fig. 1). Tissue Cd concentrations in *M. balthica* and *P. amurensis* increased linearly with increasing extractable Cd in sediment (Fig. 2; slope ± SE for *M. balthica* = 0.14 ± 0.02, $P < 0.001$). Tissue Cd followed Cd in porewater only when the metal concentration in sediment was varied, but not when only AVS was varied (Fig. 1); nor did Cd uptake increase when AVS decreased. No significant association ($P > 0.05$) was found between tissue Cd concentrations and Cd in porewater or [EM − AVS] (Fig. 2). Thus, extractable Cd in sediment, not AVS, controlled Cd bioaccumulation in our experiments. Similarly, bioaccumulation of Ni and Zn by the clams was also best related with extractable metal concentrations ($P < 0.001$). AVS and porewater metals had no apparent relation to the bioaccumulation of Ni and Zn when the confounding influence of sediment metal concentration was eliminated by varying only AVS.

Bioaccumulation from porewater governed Cd bioaccumulation in *N. arenaceodentata* (Fig. 1). This result was most obvious in the sediment containing the lowest AVS and highest Cd in porewater. Tissue Cd concentrations in *N. arenaceodentata* increased with porewater concentrations ($P < 0.001$) but not with extractable Cd ($P > 0.05$). However, Ni and Zn bioaccumulation by *N. arenaceodentata* did not follow porewater concentrations or the predicted influences of AVS. As with the clams, Ni and Zn uptake increased with concentrations of extractable metals (slope ± SE for Ni = 0.18 ± 0.02 and for Zn = 0.30 ± 0.06, $P < 0.001$). The uptake of Cd predominantly from porewater in *N. arenaceodentata* contradicted biokinetic model predictions that dietary uptake accounts for >98% of Cd bioaccumulation in the polychaete *Nereis succinea* (*8*).

No significant bioaccumulation of Cd was found for the head-down deposit-feeding worm *H. filiformis*, which was exposed only to variable AVS. Because *H. filiformis* inhabits deep sediments and does not actively aerate its burrow, Cd availability to this species may be strongly influenced by the high AVS in its microhabitat. *H. filiformis* did accumulate Ni (at a level ~11 times that of the

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control) and Zn (at a level of ~2 times that of the control) in proportion to metal concentration in sediments and not to EM/AVS.

A poor association between porewater metals and bioaccumulation of metals by animals was observed in 10 of the 12 treatments (four species times three metals). One explanation is that organisms usually accumulated metals mainly through direct ingestion of sediment, regardless of AVS content. A radiotracer experiment was conducted to directly test the bioavailability of dietary Cd and Ag sulfide. We also compared metal sulfide bioavailability to that of metals on oxidized particles. The latter may constitute an important particle type eaten by benthic animals. Radioactive ^{109}Cd and ^{110m}Ag were coprecipitated with FeS onto glass beads (10 μm in diameter) and cured for 18 hours according to an established method (*9*). An aliquot of the metal sulfide was reoxidized by bubbling with filtered air for 18 hours. The sulfide-coated and reoxidized particles were suspended separately in seawater (salinity 30) and fed to *M. balthica* and the mussel *Mytilus edulis* for 5 min. The assimilation efficiency of ingested ^{109}Cd and ^{110m}Ag was determined by a pulse-chase protocol (*10*).

Both bivalves assimilated 14 to 19% Cd from metal sulfide particles, which is comparable to assimilation from reoxidized particles (10 to 20%) (Table 1). Deposit-feeding *M. balthica* also assimilated Ag from metal sulfide particles with an efficiency of 15 to 28%; assimilation from reoxidized particles was 16 to 21%. The filter-feeding mussel *Mytilus edulis* assimilated Ag from both particle types with lower efficiency than did *M. balthica*, but metal availability from sulfide precipitates was not lower than from reoxidized particles in either bivalve. Ingested particulate metal sulfides could be solubilized if gut fluids were mildly acidic, by retention for more than a day in a mildly oxidized gut (*11*), or by association with the high concentrations of organic ligands that are typical of invertebrate gut fluids (*12*). Additionally, bivalves allocate ingested food to the digestive gland, where intensive intracellular digestion occurs, which could further enhance metal assimilation (*13*). Whatever the mechanism, it is apparent that the AVS-independent correlation of tissue and sediment metal concentrations could be caused by direct metal accumulation from diet, even if most metals in that diet were in sulfide form. Others (*14*) have also shown the importance of dietary metal uptake in benthic invertebrates.

If porewater metal concentrations were solely responsible for metal bioavailability, organisms living in sediments should be protected from metal effects when AVS is in excess of extractable metals. We applied this concept to Ag bioavailability as observed in ~200 different samplings between 1977 and 1998 from a South San Francisco Bay mudflat (*15*). The sediments from this mudflat contained molar concentrations of extractable Ag (extracted with 0.4 N HCl) of ≦0.02 μmol g^{-1}, and total Ag rarely exceeded 0.03 μmol g^{-1} (*15*, *16*).

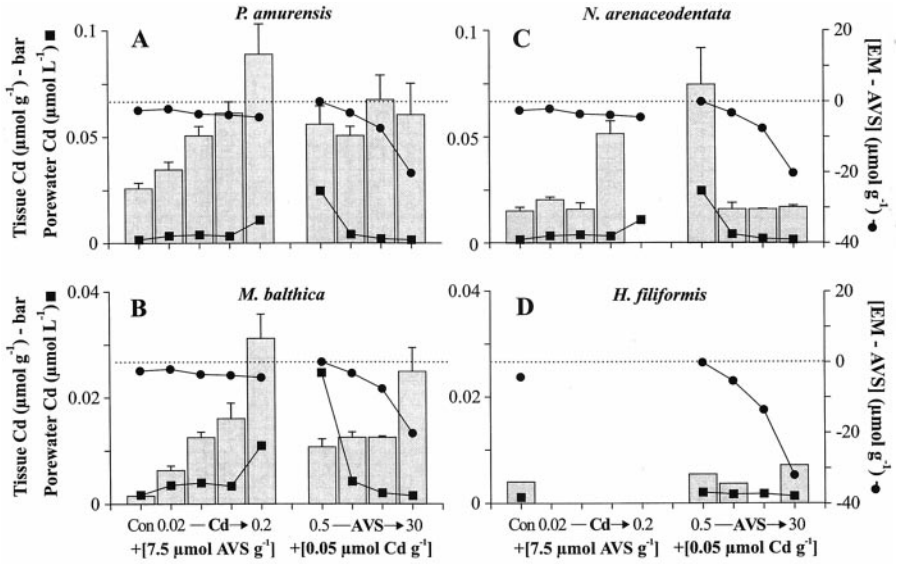

Fig. 1. Mean tissue Cd concentrations (bars) in the clams *P. amurensis* (**A**) and *M. balthica* (**B**) and the polychaetes *N. arenaceodentata* (**C**) and *H. filiformis* (**D**), related to extractable Cd, AVS [EM − AVS] (circles), and porewater Cd (squares) determined at the end of an 18-day bioassay. Error bars represent SDs around the mean (n = six replicates of three individuals for each clam; n = four replicates of one to six individuals for *N. arenaceodentata*; and n = 1 of 20 individuals for *H. filiformis*). No *N. arenaceodentata* data were available for the highest Cd treatment because of mortality. The dotted line represents [EM − AVS] = 0, below which animals should be protected from the effects of toxic metals, according to previous studies (*2*). The [EM − AVS] or porewater Cd values are from the depth where the animals feed most actively (*6*). Con, control treatment without metal enrichment (AVS = 7.5 μmol g^{-1}).

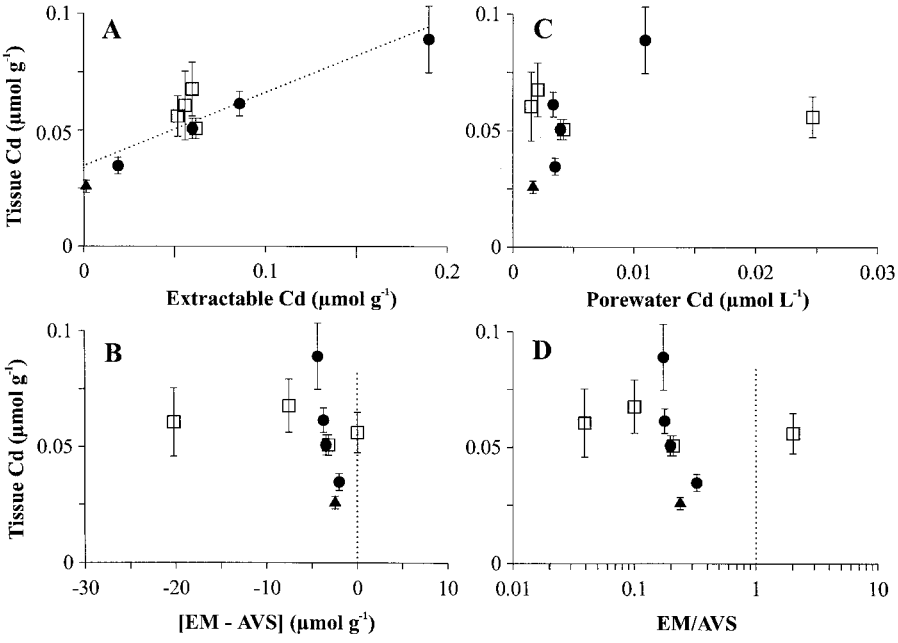

Fig. 2. Mean tissue Cd concentration (μmol g^{-1} dry weight) in *P. amurensis* when extractable Cd was varied and AVS was held constant (circles), or when AVS was varied and extractable Cd was held constant (squares). The control (triangles) was from the unenriched sediment with a nominal AVS of 7.5 μmol g^{-1}. The relationships of (**A**) tissue Cd to extractable Cd, (**B**) [EM − AVS], (**C**) porewater Cd, and (**D**) EM/AVS are shown. Error bars represent SDs around the mean (n = six replicates of three individuals). The vertical dotted line in (B) and (D) represents either [EM − AVS] = 0 or EM/AVS = 1. A significant relationship was found only between tissue Cd and extractable Cd (y = 0.32x + 0.03, P < 0.001).

Recently, AVS levels in this mudflat have been ~0.4 μmol g^{-1} at the surface (0 to 2 cm) and ~10.7 μmol g^{-1} in deep (2 to 6 cm) sediments. Sediment characteristics have not changed since 1977 in ways that would cause AVS to change (*15*). Given these conditions, the AVS-based criteria would predict that Ag should not be bioavailable and the sediments should be nontoxic (*17*). Yet, when extractable Ag concentrations were 0.01 to 0.02 μmol of Ag g^{-1} between 1977 and 1985, *M. balthica* accumulated 0.5 to 1.5 μmol of Ag g^{-1} (baseline is <0.01 μmol of Ag g^{-1} of tissue). These animals were also unable to produce gametes necessary for reproduction during the time of greatest contamination (*15*). Silver concentrations in sediment have declined since 1985 to 0.002 μmol of Ag g^{-1}, bioaccumulation has declined, and reproduction has recovered. Although we cannot prove that Ag alone caused the changes in reproduction at this field site, no environmental variables other than Ag and Cu contamination correlate with reproductive change. If *M. balthica* assimilates 15 to 28% of Ag from reduced and oxidized particles (Table 1), Ag bioaccumulation from ingestion of either surficial sediments or suspended materials could explain the departure from AVS-based predictions.

Our results are based on bioaccumulation as a measure of dose; we assume that dietary exposure as measured by bioaccumulation is at least generally related to chronic toxicity. The earlier experimental designs largely precluded exposure via diet, partly by excluding a vertical redox gradient typical of natural sediments (*6*, *18*). Extremely high quantities of metals were added in most such experiments [for example, up to 177 μmol of Cd g^{-1} (*4*), compared to <0.2 μmol of Cd g^{-1} used here], and equilibration times were short. These experimental features could result in acute toxicity from artificially high porewater metal concentrations before the implications of dietary exposure are clear. In addition, many of the earlier studies manipulated the metal-versus-AVS relationships by manipulating metal levels, not AVS levels, which confounded the controlling variable.

Table 1. Assimilation efficiency of Cd and Ag by *M. balthica* (small = 13 ± 1 mm, large = 25 ± 2 mm) and *Mytilus edulis* (35 ± 2 mm) fed Cd and Ag sulfides coated on glass beads (M-S^{2-}) and reoxidized particles (M-Ox). SD, standard deviations around the mean (n = five replicates of four individuals for *M. balthica* and n = five replicates of one individual for *Mytilus edulis*).

Organism	Particle type	Cd		Ag	
		Mean	SD	Mean	SD
*M. balthica**					
Small	M-S^{2-}	13.7	1.9	14.7	2.3
	M-Ox	18.2	6.5	15.5	5.7
Large	M-S^{2-}	15.7	2.6	27.6	4.5
	M-Ox	20.5	5.0	20.5	4.6
Mytilus edulis†					
	M-S^{2-}	19.1	2.6	2.6	0.4
	M-Ox	10.3	1.1	3.5	0.6

*No significant difference in Cd and Ag assimilation efficiency between the two particle types was found (P > 0.05). †The difference in assimilation efficiency between the two particle types was small but statistically significant (P = 0.048 for Ag and P < 0.001 for Cd).

Metal concentrations in porewater may be mostly controlled by equilibration with metal sulfides in sediments, but metal exposure is not necessarily controlled only by porewater. We contend that exposure of most organisms will occur principally through ingestion of particles. The benthic community does not behave as one entity; physiology, life habit, and feeding type have great influence on how organisms interact with the geochemical characteristics of sediments. The AVS-based approach may be appropriate for protecting some benthic organisms from acute toxicity associated with exposure to very high porewater metal concentrations in extremely contaminated sediments (*2*). However, the most common regulatory and scientific challenge is the need to determine the ecological implications of sediments contaminated at less than extreme levels. Important uncertainties remain in the application of the AVS-equilibrium concept as either a regulatory tool or a scientific generalization.

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3. AVS is defined as the molar concentration of sulfide liberated from wet sediments with cold weak acid treatment (typically 1 N HCl for 40 min). Extracted metal (EM) or simultaneously extracted metal (SEM) (*2*) is the concentration of metals simultaneously extracted with AVS. The affinity of metals to AVS follows the order Ag > Cu > Pb > Cd > Zn > Ni (*1*). In this paper, when an individual metal is compared to AVS, all metals with greater affinity for AVS than the metal of concern are defined as EM. For example, when [EM − AVS] is considered for Cd, the sum of extracted Ag, Cu, Pb, and Cd defines EM. This accounts for the consumption of AVS by Ag, Cu, and Pb before formation of CdS (*2*).
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5. The four levels of added metals in the first series of treatments (AVS = 7.5 μmol g^{-1}) were as follows: 0.02, 0.05, 0.1, or 0.2 μmol of Cd g^{-1}; 0.8, 2.4, 4.0, or 5.6 μmol of Ni g^{-1}; and 2, 7, 10, or 14 μmol of Zn g^{-1}. Baseline concentrations of these metals (the control treatment) were 0.002 μmol of Cd g^{-1}, 0.2 μmol of Ni g^{-1}, and 0.8 μmol of Zn g^{-1}. The enriched metal levels in the second series of treatments were 0.05 μmol of Cd g^{-1}, 2.4 μmol of Ni g^{-1}, and 7 μmol of Zn g^{-1}.
6. During 7-day sediment consolidation and 18-day incubation, AVS in the surface sediments (<0.5 cm) decreased by 65 to 95% because of oxidation. Initial AVS was maintained at depth (0.5 to 7 cm). No significant vertical or temporal changes in extractable metal were observed. Concentrations of Cd, Ni, and Zn in porewater showed vertical gradients as predicted by [EM − AVS]. Tissue metal concentration in each species of animal was compared to the geochemical parameters from the horizon in the sediment column where the animal feeds and/or obtains oxygen. The bioaccumulation in *P. amurensis*, *M. balthica*, and *N. arenaceodentata* was compared to metal concentrations in surface sediments (0 to 1.5 cm) in Figs. 1 and 2, and bioaccumulation in *H. filiformis* was compared to metal concentrations in deep sediments (3 to 7.5 cm).
7. Metal analyses were performed with ultra-clean techniques. Both AVS and porewater samples were processed under a N_2 atmosphere. AVS, EM, and porewater metals were analyzed at five depth intervals from 8-cm cores taken from each experimental container. Porewater samples were collected by filtration of supernatant though 0.45-μm syringe filters, after centrifugation of sediments at 3600*g* for 30 min. Tissue samples were digested with concentrated HNO_3 according to the method of Brown and Luoma [C. L. Brown and S. N. Luoma, *Mar. Ecol. Prog. Ser.* **124**, 129 (1995)]. Metal analyses were conducted by inductively coupled plasma (ICP)–atomic emission spectrometry, graphite furnace atomic absorption spectrometry, or ICP–mass spectrometry with standard addition methods.
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9. The particles were prepared according to the method of Schoonen and Barnes [M. A. A. Schoonen and H. L. Barnes, *Geochim. Cosmochim. Acta* **55**, 1505 (1991)]. Approximately 10 nM ^{110m}Ag and ^{109}Cd (10 to 20 μl) were coprecipitated during the formation of FeS onto glass beads (10 μm in diameter) by 1 ml of a 1.0 mM solution of $Fe(NH_4)_2(SO_4)_2$ and 4 ml of 0.25 mM Na_2S. Radioisotopes remained associated (>99%) with particles in the oxidized aliquot. Animals were fed radioactive particles for only 5 min to ensure that the precipitates retained their phase association. The oxidation of CdS is negligible in a 5-min exposure to oxidized water, and the solubility of Ag_2S is slower than that of CdS [D. M. DiToro, J. D. Mahony, D. J. Hansen, W. J. Berry, *Environ. Toxicol. Chem.* **15**, 2168 (1996)].
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