Critical Review

Why Is Metal Bioaccumulation So Variable? Biodynamics as a Unifying Concept

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Ecological risks from metal contaminants are difficult to document because responses differ among species, threats differ among metals, and environmental influences are complex. Unifying concepts are needed to better tie together such complexities. Here we suggest that a biologically based conceptualization, the biodynamic model, provides the necessary unification for a key aspect in risk: metal bioaccumulation (internal exposure). The model is mechanistically based, but empirically considers geochemical influences, biological differences, and differences among metals. Forecasts from the model agree closely with observations from nature, validating its basic assumptions. The biodynamic metal bioaccumulation model combines targeted, high-quality geochemical analyses from a site of interest with parametrization of key physiological constants for a species from that site. The physiological parameters include metal influx rates from water, influx rates from food, rate constants of loss, and growth rates (when high). We compiled results from 15 publications that forecast species-specific bioaccumulation, and compare the forecasts to bioaccumulation data from the field. These data consider concentrations that cover 7 orders of magnitude. They include 7 metals and 14 species of animals from 3 phyla and 11 marine, estuarine, and freshwater environments. The coefficient of determination (R^2) between forecasts and independently observed bioaccumulation from the field was 0.98. Most forecasts agreed with observations within 2-fold. The agreement suggests that the basic assumptions of the biodynamic model are tenable. A unified explanation of metal bioaccumulation sets the stage for a realistic understanding of toxicity and ecological effects of metals in nature.

Introduction

Bioaccumulation is often a good integrative indicator of the chemical exposures of organisms in polluted ecosystems (1). Bioaccumulation of metals and metalloids is of particular value as an exposure indicator because metals are not metabolized. But metal bioaccumulation can be complex. It is influenced by multiple routes of exposure (diet and solution) and geochemical effects on bioavailability. Variable patterns of accumulation occur among species. These include regulation of body concentrations of some metals by some species (2, 3), and vastly different concentrations among species and environments (4, 5). The links between bioaccumulation and toxicity are also complex (6, 7). Toxicity is determined by the uptake of metal internally and the species-specific partitioning of accumulated metal between metabolically active and detoxified forms (3, 8, 9). If a unifying mechanistic concept can capture the different geochemical and biological influences on bioaccumulation; then toxicity of the different metals and metalloids might be characterized by how different species sequester the bioaccumulated internal concentration (6).

Unifying concepts allow a field like environmental toxicology to advance by tying together diverse phenomena (10, 11). For example, fugacity is widely accepted as a unifying concept for the differences in bioaccumulation of organic contaminants (12). Fugacity does not fully explain, for every organic chemical, biological processes such as trophic transfer, food web influences, and ecological influences. Nevertheless, it must be considered first when trying to understand the differences in organic chemical concentrations among organisms, chemicals, and environments. The understanding of environmental toxicology of metals has lagged behind that of other chemicals because no simple generalization can explain the interacting influences of four factors on bioaccumulation: metal specificity, environmental influences, exposure route, and species-specific characteristics.

Here we propose that a biologically based conceptual model best captures the basic principles that drive metal and metalloid bioaccumulation. A biodynamic view of metal bioaccumulation processes unifies explanations of how and why trace element bioaccumulation differs among metals, species, and environments. Biodynamics are quantified by the dynamic multi-pathway bioaccumulation model (DYM-BAM) (13), also known as biokinetic or bioenergetic-based kinetic bioaccumulation models (14−17). The model assumes that net bioaccumulation is the result of a balance among three mechanisms: uptake rate from diet, uptake rate of dissolved forms, and loss rates. It further assumes that these rates can be realistically established in relatively simple controlled experiments for a range of concentrations and conditions. Site-specific concentrations and conditions can then be inserted into the model to calculate bioaccumulation. In other words, the model deconstructs bioaccumulation and quantifies its mechanistic components, then reconstructs a forecasted, site-specific outcome. The validity of the assumptions and the model depend on how well bioaccumulation forecasts match the bioaccumulation observed in the environment of interest.

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Here we test the comparability of predicted and observed bioaccumulation across a body of literature that considers different species, metals, and environments. We show that the predictions consistently agree well with independent field observations. Sensitivity analyses show that species, metal, and environment contribute similarly to the variability in bioaccumulation, and that dietary uptake is a key component for reasonable forecasts. These results contrast to the poor correlation often observed between environmental concentrations of metals and concentrations in biomonitor species (e.g., ref 18). As part of the validation exercise we discuss the history of this concept, why its power has only recently emerged, and its potential linkage to evaluating metal toxicity.

Methods

Biodynamic Models. Transport physiologists originated the concept that accumulation of chemical constituents (elements, amino acids, etc.) occurs as a balance of fluxes (biodynamics) (19). Radioecologists first applied these principles to contaminants, quantifying the bioaccumulation of radionuclides using simple exponential equations (20). Pentreath (21), for example, noted that ^{65}Zn bioaccumulation by mussels (*Mytilus edulis*) could be expressed using a linear differential equation with constant coefficients. If the activity in water was maintained constant, accumulation of a dissolved radioelement could be expressed as a balance between uptake rate and loss rate. If

$$C_t = C_{ss}(1 - e^{-Kt}) \qquad (1)$$

where C_t = concentration of nuclide at time, t, C_{ss} = the steady-state concentration in the organism, and K = the rate constant of loss (he called it excretion), then

$$dC_t/d_t = (k_u C_w) - (k_e C_t) \qquad (2)$$

which states that the change in concentration in an organism (C_t) over time (t) is a function of uptake minus loss. The uptake rate (from solution only) is defined by an uptake constant (k_u) in units of μg g^{-1}$_{tissue}$ d^{-1} per μg L$_{water}$$^{-1}$ (or L g^{-1} d^{-1}) multiplied by the concentration in water (C_w). The loss rate is defined by a proportional rate constant of loss (k_e in d^{-1}) multiplied by the concentration in the organism. Landrum et al. (22) noted the importance of using the proper units in defining k_u (g^{-1}$_{tissue}$ and L$_{water}$$^{-1}$ do not cancel although both are measures of mass). The same principles apply to determination of k_e. When defined on the basis of *proportional loss* (e.g., % loss per day, rather than concentration lost per day) the rate constant of loss is resolved in units that can be broadly extrapolated (d^{-1}).

It is now well accepted that organisms can accumulate metals from both water and food (15, 16, 23); and modern biodynamic models account for both pathways of accumulation. In a first-order rate coefficient model, uptake from the two sources is determined by summing unidirectional influx rates from water (I_w) plus unidirectional influx rates from food (I_F). In the DYMBAM model (13)

$$dC_t/dt = (I_w + I_F) - (k_e + g)(C_t) \qquad (3)$$

where

$$I_F = AE \times IR \times C_F \qquad (4)$$

and AE = assimilation efficiency (%), IR = ingestion rate (g g^{-1} d^{-1}), and C_F = metal concentration in food (e.g., phytoplankton, suspended particulate matter, sediment; μg g^{-1}).

The differential equations describing these processes can be solved to determine metal concentrations at steady state (C_{ss})

$$C_{ss} = [(k_u \times C_W) + (AE \times IR \times C_F)]/(k_e + g) \qquad (5)$$

where g is growth rate in d^{-1}.

As implemented, DYMBAM uses a bioenergetic-based model (22) or physiological model (24) for dietary uptake. Assimilation efficiency and ingestion rate are bioenergetic terms. A compartment model is used to describe uptake from solution. Most workers determine unidirectional influx of dissolved metal (or metalloid) empirically. The flux of water across the gills (the energetic term) may not fully explain differences in metal influx rates among species (25, 17). Uptake efficiency at the gill is also poorly known for most species. The combination of bioenergetic and compartment components suggests that "dynamic" or "biodynamic" are probably the most accurate designations for these models.

The major advantage of eq 5 compared to alternatives is its simplicity and the reliability of an independent, empirical determination of the key coefficients. Multi-compartment pharmacokinetic models, for example Redeker et al. (26), obtain coefficients by estimating an optimal fit to an uptake/loss curve from a combination of coefficients.

Validation: Constraining the Choices of Literature. Hundreds of publications on metal bioaccumulation exist. Most of these do not allow robust comparisons between the laboratory and the field. Some are controlled studies without complementary field data (or vice versa). Many are uptake studies that do not quantify dietary exposure or rate constants of loss. Most consider net outcomes (uptake over many days) that result from the combined influence of more than one process (e.g., uptake and loss). For the present compilation (Table 1) we sought papers that independently determined uptake and loss rates suitable for model coefficients, using laboratory protocols designed for that purpose (16, 27, 28). The only papers selected were those that determined unidirectional influx rates and unidirectional rate constants of loss. Papers were not selected that did not quantify assimilation efficiency from food.

We used only papers wherein the same species used in the physiology studies was collected from a resident population in nature. Transplant experiments and laboratory experiments were excluded from comparison with the forecasts. This avoided complexities caused by manipulation and time constraints. It was necessary that tissues were analyzed for metals using quality assured analytical techniques. The field collections in all papers followed established biomonitoring protocols (1, 29).

We also only selected papers that included metal concentrations determined from solution, particulate material, and/or other food sources from the same system in which the resident organisms were collected. Dissolved metal data were used only if they were quality assured. Ultra-clean sample collection and analytical techniques were essential. All environmental concentrations were within the bounds of concentrations found using modern, rigorous protocols in similar systems (30). In some cases the biomonitors were not sampled at the exact same time/place as their environment. For example, Roditi et al. (31) used in their forecasts ultraclean determinations of metal concentrations in the water column of the Hudson and Niagara Rivers and Lakes Erie and Ontario. The physiological constants were determined for zebra mussels. Those predictions were compared to concentrations in zebra mussels (*Dreissena polymorpha*) resident in each river or lake. But the *D. polymorpha* were collected in a separate program—NOAA Mussel Watch.

Some studies forecasted bioaccumulation from one set of parameters for a single species, then applied that to several

TABLE 1. All Data, Including Flux Rates and Environmental Concentrations, Compiled for Use in Figure 1[a]

species (ref)	ecosystem	metal	Dietary Source AE %	IR gg⁻¹d⁻¹	C(fd) (μg g⁻¹	C(fd) Kd	Dissolved Source k_u Lg⁻¹d⁻¹	C(water) μg L⁻¹	Loss k_e d⁻¹	Predicted min μg g⁻¹	max μg g⁻¹	median μg g⁻¹	Observed min μg g⁻¹	max μg g⁻¹	median μg g⁻¹
						Marine Mussels									
Mytilus edulis(49)	San Francisco Bay	Cr	1	0.27	18		0.1	0.11	0.012	2.6	7.5	5.05			4.05
(15)	San Francisco Bay	Ag	4–12	0.27		1.5 × 10⁵	1.6–2.0	0.003–0.010	0.034	0.3	2.1	1.2	0.35	0.77	0.56
(15)		Cd	10–30	0.27		5 × 10³	0.35–0.38	0.07–0.20	0.014	2.7	10.1	6.4	4.4	9.4	6.9
(15)		Se	30–70	0.27		1 × 10⁴	0.032–0.039	0.03–0.07	0.022	1	5.6	3.3	2.5	6.7	4.6
(15)		Zn	15–30	0.27		2 × 10⁴	0.096–1.31	0.5–1.7	0.015	54	265	159.5	54	130	92
(15)	Long Island Sound	Ag	4–12	0.27		1.5 × 10⁵	1.6–2.0	0.004	0.034	0.43	0.8	0.615	0.04	0.44	0.24
(15)		Cd	10–30	0.27		5 × 10³	0.35–0.38	0.07–0.12	0.014	2.9	7	4.95	1.5	6.2	3.85
(15)		Zn	15–30	0.27		2 × 10⁴	0.096–1.31	0.32–1.00	0.015	34	157	95.5	52	142	97
						Freshwater Bivalves									
Corbicula fluminea (88)	Freshwater Delta of San Francisco Bay	Cu	16	0.028	35		0.224		0.004			88	45	155	100
Dreissena polymorpha (31)	Hudson River	Ag	4.5	0.35	0.45		3.67	0.0009	0.08			0.09			0.08
(31)		Cd	22.7	0.35	2.2		1.98	0.0187	0.01			21.7			18.7
(31)		Cr	1.5	0.35	42		0.95	0.158	0.02			12.6			16.8
(31)	Niagra River	Ag	4.5	0.35	0.43		3.67	0.0004	0.08			0.15			0.76
(31)		Cd	22.7	0.35	1.81		1.98	0.0058	0.01			15.7			4.9
(31)		Cr	1.5	0.35	4.2		0.95	0.091	0.02			5.6			8
(31)	Lake Erie	Ag	4.5	0.35	0.06		3.67	0.0001	0.08			0.04			0.04
(31)		Cd	22.7	0.35	0.76		1.98	0.0076	0.01			7.7			5.94
(31)		Cr	1.5	0.35	4.2		0.95	0.042	0.02			2.1			9.94
(31)	Lake Ontario	Ag	4.5	0.35	0.21		3.67	0.0005	0.08			0.08			0.08
(31)		Cd	22.7	0.35	0.29		1.98	0.0043	0.01			3.3			4.3
(31)		Cr	1.5	0.35			0.95	0.199	0.02			9.1			7.8
						Marine Clams									
Macoma balthica (89)	San Francisco Bay	Ag	12–22	50	0.4–0.7		0.3–0.4	.016–0.120	0.01	1.3	21	11.2			8
(89)		Cd	6–13	50	0.04–0.7		0.03–0.04	0.006–0.22	0.025	0.02	0.9	0.46			0.33
(89)		Co	8–20	50			0.03–0.04	0.02–3.5	0.026	.03	2.7	1.5			2.4
(14)		Se	86	25	0.1		na	na	0.02	2.2	4.3	3.25			3
(14)		Se	86	25	0.2		na	na	0.02			5			4.3
(14)		Se	86	25	0.4		na	na	0.02			6.7			8.6
						Marine Zooplankton									
Mixed copepods (32)	Mediter. Sea	Ag	10	0.33	0.02–0.15		10.4	0.0008–0.0018	0.16	0.07	0.2	0.13			0.1
(32)		Cd	40	0.33	0.03–0.11		0.67	0.006–0.010	0.16	1.61	3.86	2.2			0.9
(32)		Co	15	0.33	0.88–3.94		0.57	0.012–0.019	0.30	0.4	0.86	0.57			0.5
(32)		Se	55	0.33	0.66–4.04				0.19	0.7	1.76	1.34			3.3
(32)		Zn	60	0.33	1.1–9.0		2.68	0.20–0.56	0.08	86	262	167			325
mixed copepods (63)	San Francisco Bay	Se	51	0.42	1		0.024	0.24	0.16			1.1			3.2
Mysid (*Neomysis mercedis*) (63)	San Francisco Bay	Se	61	0.44	1		0.027	0.24	0.23			1.3			1.4

TABLE 1 (Continued)

species (ref)	ecosystem	metal	Dietary Source				Dissolved Source		Loss	Predicted			Observed		
			AE %	IR gg⁻¹ d⁻¹	C(fd) (μg g⁻¹)	C(fd) Kd	k_u Lg⁻¹ d⁻¹	C(water) μg L⁻¹	k_e d⁻¹	min μg g⁻¹	max μg g⁻¹	medium μg g⁻¹	min μg g⁻¹	max μg g⁻¹	median μg g⁻¹
						Freshwater Insects									
*Chaoborus americanus** (33)	Quebec Lakes Turcotte	Cd	40–58	0.046–0.17	7.4		nd	nd	0.037–0.013	1.54		1.54	6.34 ± 0.77	6.34	
Chaoborus punctipenis (90)	Hélène	Cd	6–40	0.05–0.22	0.99–1.1				0.003–0.08	0.19	0.79	0.5	0.13 ± 0.02	0.13	
(90)	Flavrian	Cd	6–40	0.05–0.22	2–5.5				0.003–0.08	0.71	2.95	1.83	1.30 ± 0.10	1.3	
(90)	Duprat	Cd	6–40	0.05–0.22	1.3–1.6				0.003–0.08	0.28	1.18	0.73	1.40 ± 0.10	1.4	
(90)	La Bruère	Cd	6–40	0.05–0.22	1.6–2.9				0.003–0.08	0.52	2.18	1.3	1.70 ± 0.10	1.7	
(90)	d'Alembert	Cd	6–40	0.05–0.22	3.1–4.2				0.003–0.08	0.61	2.52	1.55	2.30	2.3	
(90)	Vaudray	Cd	6–40	0.05–0.22	3.2–5.8				0.003–0.08	1.05	4.36	2.7	2.90 ± 0.10	2.9	
(90)	Marlon	Cd	6–40	0.05–0.22	20–23				0.003–0.08	3.92	16.34	10.13	5.60 ± 0.60	5.6	
						Gammarid Amphipods (Marine)									
Mixed (91)	Weser Estuary, Germany	Cd	na	na	na		26.9	0.0005	0.053	0.25	0.5	0.375	0.33	0.44	0.34
						Cirripedia – Barnacles									
Balanus amphitrite (92)	Hong Kong coastal waters	Cd	35–85	0.4	0.2–0.7	na	na	0.007	0.02	21	12	2	31	17	
(92)		Zn	70	0.4	140		na	na	0.003	2,610	11,560	7,080	3,100	11,000	6,550
Elminius modestus (50)	English Channel	Cd	20–50	0.44	1.5		0.11	0.028	0.02	7.6	10.6	9.1	15	27	21
(50)		Zn	40–90	0.44	70		0.31	0.65	0.002	1,500	4,400	2950	2,470	4,730	3600
						Marine Snow Crab									
Chionoectes opilio liver (39)	St Lawrence Estuary	Ag	80–100	0.003	3.3		na	na	0.00047	5.5	27.5	11	0.44	6.6	2.2
						Marine Teleost (plaice)									
Hippoglossoides platessoides liver (39)	St Lawrence Estuary	Ag	4–16	0.01	3.3		na	na	0.023			0.22			0.44
Hippoglossoides platessoides muscle		Ag	4–16	0.01	3.3		na	na	0.023			0.004			0.004

[a] Bioaccumulation was calculated from the physiological coefficients and the environmental concentrations (predicted), and then compared to the independent determinations of bioaccumulated metal or metalloid in the tissues of the resident animals from the system of interest (observed). Where forecasts covered a range of the conditions the median of the range of coefficients and the predicted range of bioaccumulation outcomes were used for the comparisons in Figure 1. Where studies forecast a range of bioaccumulated concentrations, the full range is shown.
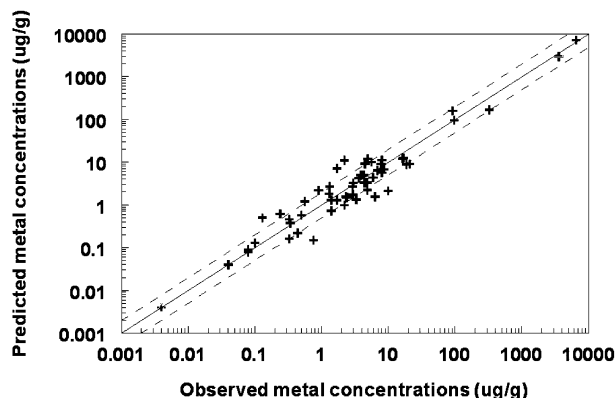
FIGURE 1. Forecasts from a biodynamic model compared to metal bioaccumulated by the same species in eleven different ecosystems. Dashed lines represent a deviation of 2× above or below 1:1 relationship. Most studies forecast a range of bioaccumulated concentrations that might be expected for each metal; incorporating expected differences in environmental conditions or behavior of the organism (e.g., choice of food) to capture the variability of natural systems. We used the median of the predicted range of bioaccumulation outcomes for the comparisons.

environments for which chemistry and biomonitor data were available (31−33). We used all forecasts where site-specific geochemistry was available, and the median among all where a full set of information was not published.

Results

Variability in Metal Bioaccumulation. Variability in trace metal bioaccumulation is widely known (34). Concentrations in the tissues of animals considered here varied by 7 orders of magnitude (Table 1). The highest concentrations were for Zn in the bodies of the barnacles *Semibalanus balanoides* and *Balanus amphitrite* (20 000−50 000 μg/g dry wt.) (35−37). The lowest were cadmium concentrations of 1.0 μg/g or less in bivalves from San Francisco Bay, copepods from the Mediterranean, or amphipods from the U.K. or the Weser Estuary, Germany (38; Table 1). Silver concentrations of less than 0.1 μg/g dry wt. were found in the muscle of fish and the tissue of bivalves from the Great Lakes (39).

The regression of DYMBAM-predicted vs observed concentrations across the full set of data (Figure 1) followed a 1:1 relationship and was highly significant ($R^2 = 0.98$; $p <$ 0.001). Eighty-eight percent of the data fell in the three decades between 0.1 and 100 μg/g. For that subset, R^2 was 0.94 ($p < 0.001$). So the strength of the fit between forecasts and observation was consistently strong over the entire range. What was most important was that only a small number of points deviated from the 2-fold variation ascribed by Landrum et al. (22) as acceptable for a useful relationship.

Forecasts and observations agreed over a suite of metals (Ag, Cd, Co, Cu, Cr, and Zn) and the metalloid, Se. The range of species included several bivalves: filter-feeding marine mussels (*Mytilus edulis* and *Perna viridis*) and freshwater clams and mussels (*Corbicula fluminea* and *Dreissena polymorpha*), as well as the deposit-feeding clam (*Macoma balthica*). It included marine crustaceans ranging from copepods to barnacles (*Elminius modestus* and *Balanus amphitrite*), to snow crabs (*Chionocetes opilio*); and a marine teleost fish, the American plaice *Hippoglossoides platessoides*. Environments ranged from Hong Kong coastal waters, San Francisco Bay, Long Island Sound in New York, Atlantic coastal waters off Quebec, the waters of the English Channel and the Mediterranean, as well as lakes from Quebec and rivers from northeastern North America. The range of environmental concentrations was from contaminated to relatively pristine, although no data from extremely con-
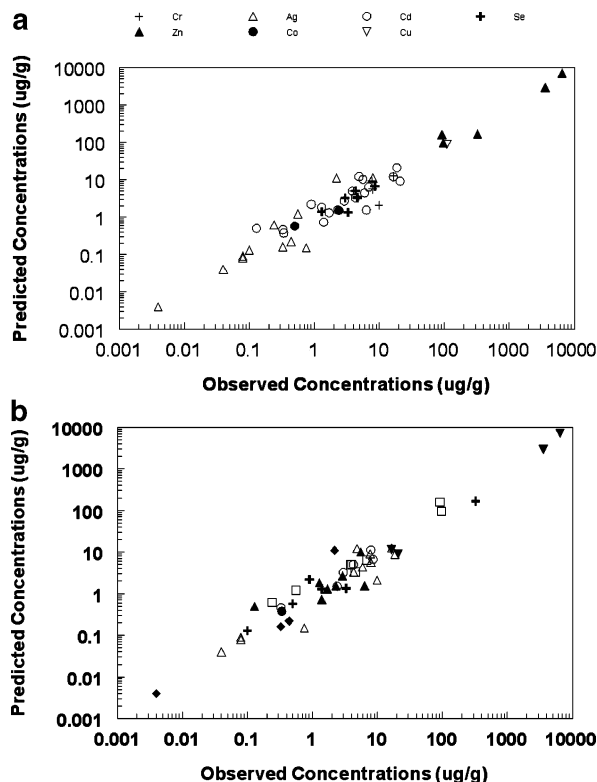

FIGURE 2. (a) Forecasts from a biodynamic model compared to metal bioaccumulated by the same species in eleven different ecosystems; identified by metal. (b) Forecasts from a biodynamic model compared to metal bioaccumulated by the same species in eleven different ecosystems; identified by taxon. Complete data set, genus/species names, and references are in Table 1. (□) Marine mussels; (○) marine clams; (▼) barnacles; (+) zooplankton; (△) freshwater mussels; (▲) freshwater, predaceous insects (*Chaoborus*); (◎) amphipods; (◆) crab and fish, muscle and viscera.

taminated environments were found. But all environmental concentrations are from nature, so they are lower than many of the concentrations used in short-term toxicity tests. Overall, the variability in the regression was equal to or smaller than that seen in relationships describing the conceptual basis of fugacity (40).

The different factors appeared to contribute similarly to variability in bioaccumulation. Comparing all data, different metals comprised the lowest values (Ag) and the highest values (Zn; Figure 2a). The range was 7 orders of magnitude. Bioaccumulation differed among metals within a single species (*Mytilus edulis*) from a single system (San Francisco Bay) by ∼400× (Figure 3).

Comparing all data, different organisms also comprised the lowest values (fish muscle) and the highest values (barnacles; Figure 2b), over the 7 orders of magnitude. When the bioaccumulation of a single metal (e.g., Zn, Figure 3) was compared across all species in which it was studied, the variability was ∼100×. For most metals, variability among species was of this magnitude. For the metalloid, Se, variability was less.

Variability within a single environment (all metals and all species studied in San Francisco Bay; Figure 3c) was also ∼400×. Values observed from nature and those predicted from the biodynamic models matched well in all these figures. This suggests that the basic causes of differences in bioaccumulation among metals, species, and environmental conditions are captured in the model, and can be empirically quantified using the protocols and measurements used to derive model forecasts.

Metals in *Mytilus edulis*
Cr, Ag, Cd, Se, Zn

Wang et al, 1996

Zinc across species

Diet + water

San Francisco Bay
Mussels, clams, mysids, mixed zooplankton
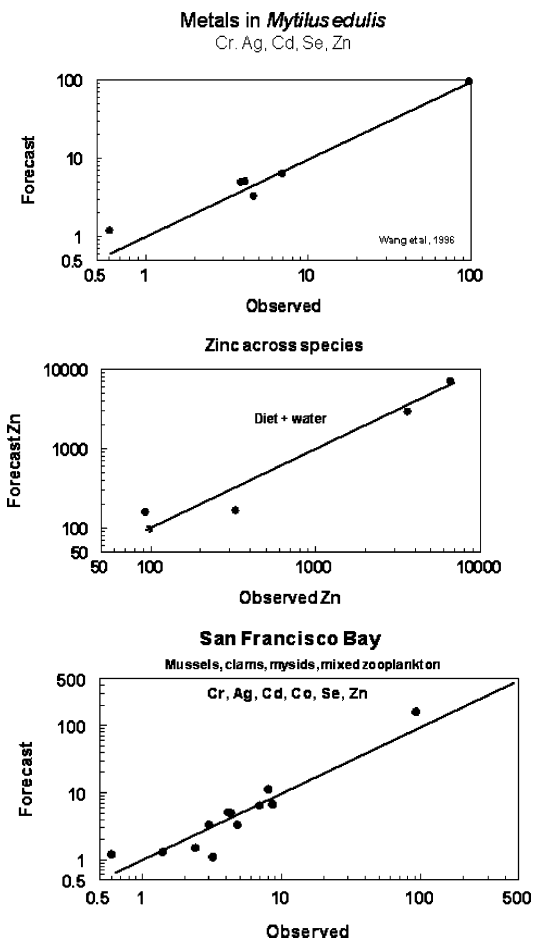Cr, Ag, Cd, Co, Se, Zn

FIGURE 3. Sensitivity of the variability in bioaccumulation to metal in one species, *Mytilus edulis*; to species for one metal (Zn); and in one environment (San Francisco Bay), for all metals and species.

We also tested the importance of the model component most often absent from bioaccumulation studies: dietary uptake. Figure 4a and b show model forecasts using both water and dietary routes of uptake, and forecasts for only dissolved uptake from the same species (where data were available). It is well-known that selenium bioaccumulation is primarily from food. So it is not surprising that forecasts from dissolved Se uptake alone were 50× lower than observations from nature (Figure 4a). Cadmium provides results more typical for most metals (Figure 4b). When both routes of uptake were considered, bioaccumulation of cadmium in nature agreed well with the forecasts. Cadmium bioaccumulation from dissolved sources was within 50% of total Cd bioaccumulation in the bivalves *M. edulis* and *M. balthica* from San Francisco Bay, *M. edulis* from Long Island Sound (*15*), and *D. polymorpha* from Lake Ontario (*31*) (4 of 9 cases, Table 1). In these circumstances, Cd concentrations on particulate material were lower, relative to concentrations in water, than in contaminated environments. In the Cd-contaminated Hudson River, by contrast, the model forecast 4 μg/g dw bioaccumulation from water and 21 μg/g dw overall in *D. polymorpha*. High particulate Cd appeared to explain the inability of dissolved uptake to explain total exposure. Four of nine cases had similar disparities. Biology appeared to explain the large disparity between dissolved uptake and total bioaccumulation in the barnacle *E. modestus* from the English Channel (Table 1; *50*). High assimilation efficiencies and high ingestion rates led to a forecast total bioaccumulation of 12 μg/g dw. Bioaccumulation from water alone was forecast to be 0.2 μg/g dw, because of the low k_u. Dietary uptake is thus often very important, although situations exist
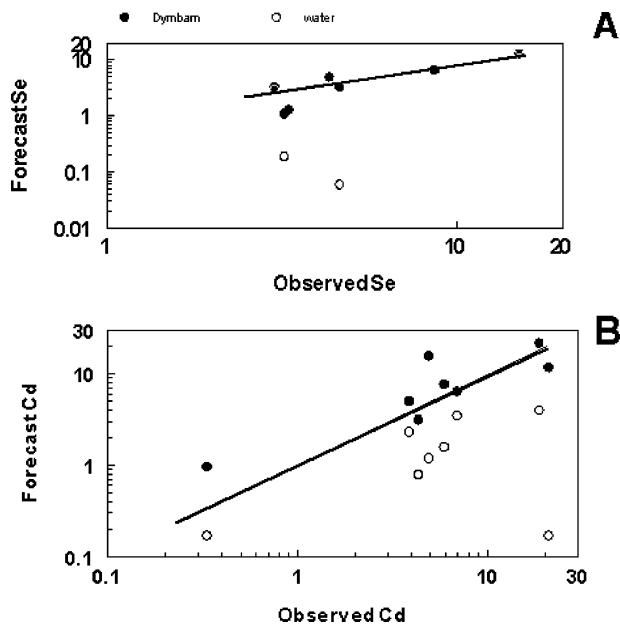

FIGURE 4. Observed vs forecast selenium (A) and cadmium (B) bioaccumulation (closed circles) from water and diet, contrasted to accumulation forecast from water alone, for those papers from which data were available (*15, 31, 32, 50, 88*).

where dissolved uptake alone describes most of bioaccumulation. The correlation between model forecasts and observations, overall, would be weak if only dissolved uptake was considered.

Discussion

History of Biodynamic Metal Bioaccumulation Studies. Biodynamic modeling has a long history, but only partial validation of the models was possible until recently. Cutshall (*20*), using data from Seymour (*41*), first tested the adequacy of a single-compartment, first-order exponential model against field data (^{65}Zn bioaccumulation in transplanted oysters near a nuclear power station). He concluded "single compartment exponential equations quite adequately fit the (dissolved) uptake data, the loss data, and (can be used to derive) steady-state concentrations". Later studies forecast concentrations of mercury (^{203}Hg) expected in a polychaete worm (*Nereis succinea*) and a decapod crustacean (*Palaemon debilis*) from a brackish water Hawaiian system (*42*). Mercury entered this estuary in pulses with runoff, then was flushed out. Rate constants of loss for inorganic mercury, from laboratory studies, seemed to fit loss of mercury from worm and shrimp tissues, after a pulse disappeared. Several other studies developed biodynamically predicted concentration factors from dissolved metal (radionuclide) uptake rates and loss rates (*21, 43−45*). Nearly all came to the conclusion that dietary uptake was necessary to explain the degree of bioaccumulation of Zn, Cd, Mn, and Hg observed in animals such as euphausiid crustaceans, mussels (*Mytilus edulis*), and fish (plaice *Pleuronectes platessa*) in nature. In general, however, opportunities for validation studies were rare (*20*). This was partly because reliable geochemical and biomonitor data were rare, and partly because there was no practical method to quantify dietary uptake.

During the 1990s high-quality data emerged (using clean techniques) for metal concentrations in water, particulate material, and food organisms of many sizes. Some systems still lack data (*30*) but they are increasingly available for major systems (*46, 47*). In 1991, a simple, repeatable protocol was published for quantifying absorption efficiency (aka, assimilation efficiency) from food by aquatic animals (*48*). Then the assimilation efficiency approach was used to differentiate

bioaccumulation of selenium from diet and water in bivalves from San Francisco Bay (*14*) and forecast overall bioaccumulation. The bioaccumulation forecasts were within a factor of 2 of selenium concentrations in bivalves from the Bay.

Wang et al. (*15*) published the full set of protocols for determining biodynamic coefficients in 1996. They also found favorable comparisons between forecast bioaccumulation and that observed under typical conditions in San Francisco Bay and Long Island Sound. Thomann et al. (*23*) used the parameters and data developed by Wang et al. (*15*) to expand his earlier modeling work with metals and organics. Thomann's conceptually similar, albeit more complex, model reiterated that dietary transfer had to be considered to accurately forecast bioaccumulation of most metals and metalloids.

How Biodynamics Explains Bioaccumulation. A rapidly growing body of work explores the explanatory power of metal biodynamics, whether employed quantitatively (*16*–*18*) or conceptually (*3*). Influential factors and processes include metal-specific chemistry, concentration, geochemical influences on bioavailability, exposure route, and species-specific physiological attributes.

Metal-Specific Chemistry and Abundance (Concentrations). Generally, bioaccumulated concentrations of Cd and Ag are nearly always lower than Zn concentrations. In part this is because Zn is more abundant in the environment than Cd and Ag. The response of bioaccumulation to varying environmental concentrations, in fact, is the basis of proposing that many animals are excellent exposure biomonitors (*1*). However, abundance in the environment is not the only important metal-specific aspect of bioaccumulation. Cr is as abundant as Zn in nature, judging from concentrations in many sediments. But Cr bioaccumulation is always substantially less than Zn bioaccumulation. This is because uptake rates of Cr from all dissolved forms (even Cr VI) are very slow compared to Zn uptake rates, and bioavailability from diet is relatively low as well (*49, 50*). It is possible that the slow reaction rate of Cr with organic ligands (*51*) is a bottleneck to Cr uptake. Whatever the explanation, biodynamic uptake rates empirically account for the mechanics of Cr and Zn uptake that explain a consistent difference seen in field accumulated concentrations.

Silver has an unusually wide range of bioaccumulation. Silver concentrations as low as 0.004 μg/g are observed in the tissues of some organisms (Table 1). But bioaccumulation of more than 100 μg/g dw has also been observed in invertebrates in polluted conditions (*52*). Strong association with sulfides or binding to sulfur ligands in solution can reduce Ag bioavailability in some circumstances. But when Ag is accessible to the organism, its uptake rates (e.g., k_us) are among the most rapid for any metal (*17*). So variability in Ag bioaccumulation is a reflection of metal-specific biology (fast uptake rates) and metal-specific geochemistry (as it influences availability to the organism) that is reflected in both nature and model forecasts.

Geochemical Influences. Speciation (*53*), particulate metal form (*51, 54*), and metal form in the tissues of prey (*48, 55*) all affect bioaccumulation. A major advantage of the biodynamic approach is that it is feasible to design simple experiments to quantify the influences of environmental factors. For example, the free ion activity model (FIAM; *56*), is generally useful for forecasting bioavailability of metals from solution, for those metals to which it applies. Lee et al. (*57*) showed that uptake rates of cadmium and zinc from solution in two bivalves were five times higher at a salinity of 5 than at a salinity of 30. They tied the differences to influences of salinity on metal speciation. Influx rates of zinc were greater than cadmium, on the basis of total metal concentrations. But cadmium rates were faster than zinc rates on the basis of free metal ion activities. Salinity also most strongly affects cadmium speciation. So salinity change affected cadmium uptake rates more than it did zinc rates. The data in Table 1 show that k_u values for Cd in the freshwater mussel *Dreissena polymorpha* are about 5 times greater than those in the marine mussel *Mytilus edulis*. The coefficients were determined in freshwater for *D. polymorpha* and seawater for *M. edulis* (these are the habitats of the species). So the experiments empirically captured the combination of biology and speciation that generally characterize uptake rates in each habitat, thus allowing reasonable forecasts for both species. Characterization of k_us on the basis of free ion concentrations (*58*) might ultimately be useful, especially if data for FIAM determinations from nature become more available.

FIAM and similar geochemical principles (e.g., SEM-AVS; *54*) alone do not address critical processes such as dietary bioaccumulation, however. Studies of biodynamics were the first to clearly quantify water and dietary bioaccumulation routes (*15, 28*). Later studies showed the quantitative importance of different factors influencing dietary bioaccumulation. Food choice can have large effects on assimilation efficiency, whereas the effects of food quantity are relatively small (*59*). Trophic transfer to predators is substantial for some elements (*60*–*63*). Biomagnification of Se, Hg, and perhaps Cd can occur (*64*–*66*). Effects of environmental circumstances such as pre-exposure to metals are complex (*67*–*69*), but can be quantified in different circumstances. Assimilation of Cd, Zn, and Cr increases as living materials comprise an increasing fraction of the suspended material ingested by clams (*Macoma balthica, Potamocorbula amurensis; 70*). Even strongly bound metals such as silver are assimilated from particulate material by mussels (*Mytilus edulis*) and clams (*Macoma balthica*), whether in sulfide or oxidized form (*71*).

Species-Specific Effects. Biodynamics also captures the biologically driven patterns that differentiate bioaccumulation among species (*3, 4*). Some animals are characterized by a rapid rate of metal excretion (high rate constant of loss) for some metals. Bioaccumulation in most such species changes little with environmental exposure. These organisms are described as regulators of metals. This is the case for zinc in the decapod crustacean *Palaemon elegans* (*3, 4*). Little change occurs in net Zn accumulation in *P. elegans* as environmental concentrations increase, because the animals lose Zn as fast as they take it up. This may have its roots in the inability of crustaceans to re-absorb amino acids, along with associated metal (*72*); and/or in the evolution of their metal detoxification strategy. If "regulation" is viewed as the net outcome of uptake and excretion, a rational basis is established for understanding when and where regulation might occur.

Organisms described as bioaccumulators, on the other hand, have low rate constants of loss. High concentrations are accumulated in tissues before the rate of excretion matches that of uptake. A remarkable example of a bioaccumulator is a barnacle for Zn. A high rate of ingestion by barnacles together with high Zn assimilation from food, is not matched by the extremely slow rate of zinc excretion until very high concentrations are attained in tissues (*73, 74, 50*). Mechanistically, the body zinc concentration escalates rapidly because zinc is stored, as it is taken up, in a very slowly exchanging form in detoxified pyrophosphate granules in the tissues around the midgut (*75*).

Mussels provide an interesting contrast with barnacles. Phillips and Rainbow (*76*) showed the range of Zn concentrations in the barnacle *Balanus amphitrite* and the mussel *Perna viridis* collected simultaneously from the same sites in Hong Kong waters. Rainbow et al. (*69*) have a similar set of data for *Balanus improvisus* and *Mytilus trossulus* where they co-occur in the Gulf of Gdansk (Table 2). In both cases

TABLE 2. Comparisons of Accumulated Zn Concentrations (μg g^{-1}) in Barnacles and Mussels Collected Simultaneously from the Same Location[a]

Hong Kong Site	Hang Hau	Chai Wan Kok	Kwun Tong	Tai Po Kau	Lai Chi Chong
Balanus amphitrite (barnacle)	11,990	9,353	7,276	4,381	2,726
	10,220−14,070	7,411−11,800	5,269−10,050	4,195−5,201	967−7,688
Perna viridis (mussel)	111	153	115	61	53
	75−147	59−247	79−151	42−80	39−67
Gulf of Gdansk Site	**Puck**	**Mechelinki**	**Gdynia**	**GN Buoy**	**Vistula Plume**
Balanus improvisus (barnacle)	3,293−14,106	4,466−14,386	6,088−10,048	4,197−7,448	5,610−12,217
Mytilus trossulus (mussel)	83.8−130	103−192	98.1−153	61.1−136	96.1−187

[a] Data (with 95% confidence limits) for *Balanus amphitrite* (concentration in barnacle body of 4 mg dry wt) and *Perna viridis* (mean soft tissue concentration) in Hong Kong waters are from ref *76*. Data (ranges of weight-adjusted mean concentrations) for *Balanus improvisus* and *Mytilus trossulus* in the Gulf of Gdansk, Baltic are from ref *69*.
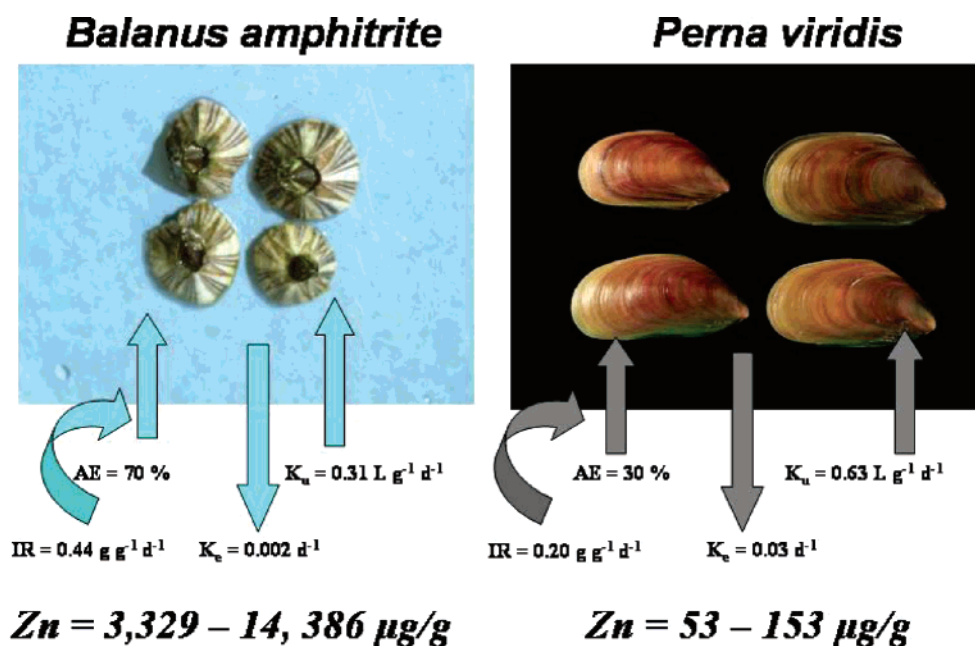

Balanus amphitrite

AE = 70 % K_u = 0.31 L g^{-1} d^{-1}

IR = 0.44 g g^{-1} d^{-1} K_e = 0.002 d^{-1}

Zn = 3,329 − 14, 386 μg/g

Perna viridis

AE = 30 % K_u = 0.63 L g^{-1} d^{-1}

IR = 0.20 g g^{-1} d^{-1} K_e = 0.03 d^{-1}

Zn = 53 − 153 μg/g

FIGURE 5. Zinc concentrations from nature, and physiological coefficients derived from laboratory experiments (*92, 50*), for a barnacle (*Balanus amphitrite*) and a mussel (*Perna viridis*) collected simultaneously from the same locations in Hong Kong coastal waters. AE is assimilation efficiency. IR is ingestion rate. k_u is dissolved influx rate constant. k_e is rate constant of loss. Higher AE, higher IR, and much slower k_e explain why higher Zn concentrations are always found in barnacles than in mussels when they co-occur.

barnacles bioaccumulated 40−100× more Zn than mussels under the same environmental conditions. Faster feeding rates, higher assimilation efficiencies, and slower rate constants of loss for Zn are typical of *Balanus spp.* compared to *M. edulis* and *P. viridis* (Figure 5). Wherever barnacles and mussels co-occur their differences in Zn bioaccumulation will be as conservative as their biological differences.

Differences in Se bioaccumulation between bivalves and pelagic crustaceans provide another example. Influx kinetics of Se are not typically much different among species. But the rate constant of loss is an order of magnitude slower in bivalves than in crustaceans (compare k_e for copepods and mysids from San Francisco Bay and the Mediterranean, to k_e for the bivalves *Mytilus edulis, Macoma balthica*, and *Dreissena polymorpha*; Table 1). Copepods bioaccumulate approximately 5-fold less Se in San Francisco Bay than do bivalves; consistent with the biodynamic differences. These differences are also propagated up the food chain and result in substantial differences in Se bioaccumulation among predators of the different species (*77*).

In summary, bioaccumulation varies widely among taxa, often reflecting basic differences in biology. Neither geochem-

ical model (FIAM) nor metal-specific ligand associations (*78*) capture these biological differences. But they are reflected by variations in the combination of a few empirically determined physiological rates that are included in biodynamic models.

Assumptions. Models offer the greatest promise for quantifying the interactions of complex factors. Models are most useful if they are simple yet robust. Protocols must exist to empirically determine coefficients. It is ideal if model forecasts can be directly compared with field observations. Biodynamic models fit these criteria. Nevertheless, all models have simplifying assumptions. These define the limits of the model and help define how to expand those limits. The agreement between biodynamic forecasts and independent field observations suggests that the basic assumptions of the biodynamic model are reasonable. Nevertheless, each should be considered carefully as experiments are designed.

Uptake Rate, k_u. To quantify uptake from the dissolved phase, it is assumed that k_u can be determined using short-term exposures to estimate unidirectional metal influx rates (*17*). Longer-term exposures underestimate k_u because they measure net accumulation, the balance between uptake and

efflux. It is also assumed that the influx rate determined in a short measurement is constant over time. The agreement between model forecasts and bioaccumulation in nature suggests the assumption of a constant uptake rate may best reflect the natural condition. Contradictory laboratory results could be a function of experimental conditions (e.g., extreme concentrations or behavioral or chemical effects of holding animals for long periods in small volumes of water; *79*).

The model also assumes that k_u describes a linear increase in uptake with concentration. Metal uptake rates in laboratory experiments linearly increase with concentration, up to dissolved concentrations that are at least an order of magnitude higher than those seen in nature. But it must be recognized that most metals traverse biological membranes via carriers or channels and a facilitated diffusion process (as compared to active transport; *8, 3, 24*). This means that influx rates could be saturable at very high concentrations. From the agreement of model forecasts with observations, it seems very unlikely that uptake rates in the field situations considered here were saturated. However, particularly in the case of laboratory experiments, it is important to better understand concentrations at which uptake rates might saturate for different metals, metalloids, and species. We might expect a convergence of the biodynamic view of bioaccumulation with that of the equilibrium-based models (*80*) as the concentration issue (nature vs laboratory) and the Michaelis–Menten characteristics of metal transport in different organisms are better understood.

Rate Constant of Loss, k_e. The proportional rate constant of loss, k_e, along with C_t, determines the efflux rate, termed the depuration rate by Landrum et al. (*22*). There is a myriad of ways that metals can associate with ligands within cells and within tissues or organs. Unidirectional whole body efflux seems to integrate many of these complexities so that physiological loss usually appears as from one or two compartments (*8*). If exposure time is months or more (typical of nature), it is assumed (and can be calculated; *15, 20, 81*) that the slow compartment of loss increasingly dominates the proportional distribution of the metal. The k_e describing the slow component of loss was used in all the models reported here. These simplifications seem to capture the major processes controlling steady state bioaccumulation (usually whole body) in the animals from nature in our literature compilation. Detailed physiological information (e.g., flux rates from specific binding sites or organs; carrier-specific dynamics) probably cannot be developed from such simple experiments, however. Some authors include separate rate constants of loss for metal accumulated from water and food (*82*). Such differences are often small in comparison to other factors (*83*), although there might be some exceptions.

Growth. Growth was not included in most model forecasts reported here, and did not seem necessary. That does not mean that growth considerations are never important. In most animals, growth rates are only occasionally high enough to affect the denominator in eq 5. A rapid addition of tissue mass is most common in young animals and usually only during certain seasons. Many biomonitor sampling regimes avoid periods of rapid growth, or biomonitors that grow rapidly (*29*). If growth is rapid, it is an essential consideration in the model (*50*).

The model also assumes that assimilation efficiency can be accurately determined from the proportion of label remaining after an ingested bolus of labeled food is defecated. The agreement between forecasts and field results appear to validate this assumption as well.

Links to Toxicity. Bioaccumulated metal is not necessarily toxic (*9, 3*). Sublethal toxic effects (reproductive impairment) were observed in bivalve populations from San Francisco Bay, coincident with an elevation of bioaccumulated metal

(*52, 84*). But relationships between bioaccumulation and adverse effects are complex when different species are compared, for different metals and even among environments. Understanding toxicity requires consideration of more than just total metal accumulated in tissues. A central theory is needed that links bioaccumulation (internal metal exposure), internal metal reactions, and the hierarchy of biological responses that define adverse ecological effects (*85*). One approach is the biotic ligand model (BLM) (*80*). It links metal speciation in solution with the amount of metal bound at the gill surface of fish. The proportion of binding sites on the gill that are occupied by metal determines the degree of disturbance to ion regulation. Two aspects of the BLM theory are fundamental to a workable central theory. Adverse effects are determined by (a) the amount of metal that is bioaccumulated, (b) at the site(s) of toxic "action". Total bioaccumulation is less important than accumulation of bioactive metal (*55*). From this spring several important questions. One is the question of a dynamic vs an equilibrium-based model. Is the toxic response determined simply by the proportion of active sites occupied by the toxicant (*80*)? How critical is the rate of delivery to the site? That is, are adverse effects only observed when the rate of uptake into the body exceeds the combined rates of detoxification and excretion or loss (*3, 7*)? How do different species balance key processes such as internal accumulation, loss rates, and detoxification? How important are mechanisms of toxicity other than disturbance of ionic regulation; especially when diet is a significant route of exposure (*86*)? To address any of these questions, bioaccumulation processes must be understood and quantified. Biodynamics provide a means to determine rates of delivery from both diet and water.

Metals probably manifest their adverse effects in nature by eliminating some species and not affecting others (*5, 87*). It seems important to determine which species are most likely to be eliminated. Those species are the useful indicators of metal effects (*5*) and the drivers of how ecosystems will change in response to metal contamination. Yet, we have vastly under-studied differences among taxa in bioaccumulation, detoxification, and the resulting responses to metals (*6*). Environmental toxicology might benefit greatly from adding an appreciation of the biological diversity of metal responses to the traditional emphasis on basic toxicology and geochemistry. Metal, taxon, environment, and exposure route, as captured in biodynamic models, provide a means to explain the variability of bioaccumulation in nature. This is central to understanding the implications of metal contamination in ecological terms.

Acknowledgments

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