

ISSUE PAPER ON METAL EXPOSURE ASSESSMENT

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NOTICE

This paper has been developed in support of an ongoing effort within the U.S. Environmental Protection Agency (EPA) to develop an integrated framework for metals risk assessment. In September 2002, the cross-Agency technical panel, organized under the auspices of the Agency's Science Policy Council, discussed plans for the development of the framework and associated guidance with the Agency's Science Advisory Board (SAB). During the advisory, the SAB affirmed the importance of incorporating external input into the Agency's effort. *As part of the effort to engage stakeholders and the scientific community and to build on existing experience*, the Agency commissioned external experts to lead the development of papers on issues and state-of-the-art approaches in metals risk assessment for several key topics. Topics identified include: environmental chemistry; exposure; ecological effects; human health effects; and bioavailability and bioaccumulation. (Some individual EPA experts contributed specific discussions on topic(s) for which he or she has either specific expertise or knowledge of current Agency practice). Although Agency technical staff, as well as representatives from other Federal agencies, reviewed and commented on previous drafts, the comments were addressed at the discretion of each respective author or group of authors. Therefore, the views expressed are those of the authors and should not be construed as implying EPA consent or endorsement.

This draft paper is being made available for public comment consistent with EPA's commitment to provide opportunities for external input. Science-based comments received on this paper will be made available to authors for final disposition. The material contained in this paper may be used in total, or in part, as source material for the Agency's framework for metals risk assessment and EPA's evaluation of this material will therefore include consideration of the Assessment Factors recently published by EPA for use in evaluating the quality of scientific and technical information. The draft framework, as an Agency document, will undergo scientific peer review by the SAB.

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TABLE OF CONTENTS

1. OVERVIEW	1
1.1 Problem Statement	1
1.2 Metal Qualities, Quantities, and Sources	1
1.3 Conclusion	3
2. ISSUE TOPICS	3
2.1 Aquatic Receptors	3
2.1.1 Sources, Phases, and Exposure Pathways	3
2.1.2 Dealing With Differences Between Laboratory Test Exposures and Field Exposures	4
2.1.3 Mixtures	5
2.1.4 Exposure Estimation Tools	7
2.1.5 Dynamic vs. Steady State Assumptions	10
2.1.6 What To Do When Information Is Limited	11
2.2 Terrestrial Receptors	13
2.2.1 Sources, Phases, and Exposure Pathways	13
2.2.2 Differences Between Laboratory Test and Field Exposures	16
2.2.3 Mixtures	17
2.2.4 Considering Background	17
2.2.5 Estimating Exposure	17
2.2.6 What To Do When Information Is Limited	21
2.3 Human Health Receptors	21
2.3.1 General Considerations	21
2.3.2 Air Pathway	23
2.3.3 Surface Dust and Soil Pathway	24
2.3.4 Food Pathway	26
2.3.5 Water Pathway	27
2.3.6 Transplacental Pathway	28
2.3.7 Exposure Estimation Tools	28
2.3.8 Mixtures	29
2.3.9 What To Do When Information Is Limited	29
3. APPLICATION TO A REGULATORY CONTEXT	29
3.1 Aquatic Receptors	29
3.2 Terrestrial Receptors	30
3.3 Human Health Receptors	30
4. NATURAL AND BACKGROUND METAL CONCENTRATION	30
4.1 Introduction	30
4.2 Site-Specific Assessment of Baseline Exposures	32

5. RESEARCH NEEDS	33
5.1 Aquatic Receptors	33
5.2 Terrestrial Receptors	33
5.3 Human Health Receptors	35
6. CONCLUSIONS	35
7. LITERATURE CITED	37

LIST OF TABLES

Table 1.	Source Categories of Metals for Six Sentinel Species (Herbivores, Carnivores, Vermivores).....	15
Table 2.	Percent Contribution of Incidental Soil Ingestion to Oral Dose for Wildlife at Different Soil Ingestion Rates and Bioaccumulation Factors and a Bioavailability of 100%.....	20
Table 3.	Estimated Global Averages of Natural Levels and Typical Present-Day Levels of Lead in the Environment.....	31
Table 4.	Mean Baseline Metal Concentrations in Soils ($\mu\text{g/g}$) by State.....	34

LIST OF FIGURES

Figure 1.	Potential pathways for metal exposure in terrestrial ecosystems.....	12
Figure 2.	Conceptual model for direct and indirect exposure of ecological receptors to metals in soil zones.....	12
Figure 3.	Distinct digestive systems of mammalian carnivore and fermenter.....	20
Figure 4.	Diagram of the hand dust exposure pathway for exposures of children to lead.....	22

1. OVERVIEW

1.1 Problem Statement

The overarching goal of this working group is to develop a consistent means of assessing hazards and risks of metals by applying available data within a consistent framework. The topic of this specific paper is metal exposure assessment.

In risk assessment, a potentially harmful metal is identified as a hazard and the most relevant concentration-response relationship is established. Also defined are the details of the exposure, because exposures vary in their effectiveness in delivering a dose across a biological boundary (Ott, 1985; Ryan, 1998). No risk exists unless an effective exposure to a receptor occurs.

Generally, metal exposure assessment involves considering exposure route and pathways, considering phase associations and chemical forms, and then expressing exposure concentration in a way that best reflects the amount of bioavailable metal. Routes include inhalation, ingestion, and absorption. Pathways describe the specifics of any exposure, e.g., inhalation of lead in dust by children playing in a park near a busy highway. Phase association and chemical speciation influence metal movement through pathways and metal's capacity to deliver a dose. Consequently, exposure routes and pathways, appropriate expressions of metal concentrations, phase association, metal speciation, and bioavailability will be discussed in this paper.

1.2 Metal Qualities, Quantities, and Sources

Assuming that a plausible exposure pathway exists, the dose realized by a receptor will be partially determined by speciation, phase association, and bioavailability of a particular metal or metal compound. Basic metal qualities determine these characteristics (Jones and Vaughn, 1978; Nieboer and Richardson, 1980; Brezonick et al., 1991) and relevant trends are explicable quantitatively with Hard and Soft Acid and Base (HSAB) Theory-based metrics, or more qualitatively by classifying metals as Class B, intermediate, or Class A metals. As described earlier in the chemistry paper, Class B metals (e.g., the extreme Class B metal ions, Hg^{2+} and Ag^+) tend to form strong covalent bonds with ligands. They contrast with Class A metals (e.g., the extreme Class A metal ions, Mg^{2+} and Ca^{2+}), which form weak, ionic complexes with ligands. The ligands can be dissolved inorganic ligands that influence speciation or with ligand groups on biomolecules of receptors that influence biological effect. Such metal classification also provides useful prediction of solid phase associations; for example, Class B metals are most prone to have their bioavailability influenced by sediment sulfides because they form extremely insoluble sulfides. Consequently, HSAB Theory provides a foundation for predicting trends in key aspects of metal exposure.

The source of a metal also influences the realized dose by determining the amount, exposure route, species, phases, and ultimately bioavailability of the metal. Lead halides, once emitted at high rates as fine aerosols in automobile exhaust, gain deeper access in the lungs and

are more soluble than lead from roadside soil that, with time, becomes associated with particulate sulfates (Cordasco et al., 1995). Lead bioavailability in ingested soils is also strongly influenced by solid phase chemistry (Casteel et al., 1997; Ruby et al., 1992, 1999). Some sources generate metal mixtures, and exposure assessment must enable estimation of potential risk associated with metal mixtures.

Major metal sources include diverse manufacturing, mining, combustion, and pesticide activities. Major atmospheric sources are oil and coal combustion, mining and smelting, steel and iron manufacturing, waste incineration, phosphate fertilizers, cement production, and wood combustion (Haygarth and Jones, 1992). Metals from these atmospheric sources can find their way into soils, sediments, and water. Other major sources to aquatic and terrestrial systems include chlor-alkali, acid, pigment, electronics, and copper sulfate production.

Some exposure assessments do not involve anthropogenic releases of metals to the environment *per se*. They focus on changes in exposure to ambient metals that result from other aspects of human activities. Acidification of freshwater changes aluminum phase association and speciation, resulting in an increased dose to aquatic biota (e.g., Campbell et al., 1992). Intensive irrigation and consequent evaporative concentration of naturally occurring selenium in arid regions of the U.S. can lead to toxic exposures (e.g., Wu et al., 1995).

Exposure assessment can be difficult if the dose received by the receptor is influenced by ambient in addition to anthropogenic metal concentrations. Considerable effort has been expended trying to distinguish between natural or background metal exposure and those concentrations resulting from the source being assessed (see Section 4). Differences in stable isotopes composition between natural and anthropogenic metal can generate insight about the relative contribution of natural and introduced metal to exposure. For example, the relative amounts of lead accumulated by a terrestrial invertebrate from mining or automobile sources were quantified with isotopic ratios (Newman et al., 1994).

Exposure assessment can be difficult if the major exposure pathway involves trophic exchange within a food web. But, here again, stable isotope techniques can contribute if knowledge of the trophic status of organisms is important to define within an exposure pathway. The increase in concentration of ^{15}N relative to ^{14}N with each trophic transfer allows one to estimate a species' trophic status by simply measuring the ^{14}N and ^{15}N in its tissues. As an important example, the concentration of mercury within a trophic web was predicted based on ratios of nitrogen isotopes in species' tissues (Cabana et al., 1994). Nitrogen isotope techniques can also define composition of the human diet, e.g., the relative amount of sustenance from marine fish versus terrestrial sources (Scheninger et al., 1983) and, in so doing, help define exposure to metals such as mercury.

1.3 Conclusion

This paper explores the best approaches for characterizing exposure pathways and routes, estimating the most relevant exposure concentrations, linking exposure to dose, and coping with natural or background concentrations. Relevant issues for human, terrestrial, and aquatic receptors will be described separately because the relative importance of exposure routes and pathways differs among these receptors. Regardless of the receptor being discussed, an important goal herein is to identify and address issues unique or especially relevant to metals and metalloids. Approaches will be highlighted that are immediately useful or potentially useful in the near future. Shortcomings or inaccuracies of current approaches will be identified.

2. ISSUE TOPICS

2.1 Aquatic Receptors

2.1.1 Sources, Phases, and Exposure Pathways

The extent to which a metal is taken up by an organism via respiration/inhalation, absorption, and ingestion exposure routes is difficult to define for all relevant aquatic species. Inhalation/respiration, which for aquatic species means general exchange across respiratory surfaces, can involve diverse gill and lung types. Absorption includes movement across the skin; the walls of diverse structures such as spongocoels, the cniderian gastrovascular cavity, and the echinoderm water-vascular system; the filtration-based feeding structures of many crustacea, insects, and polychaetes; diverse phytoplankton cell membranes; and plant roots and leaves. Ingestion pathways can involve direct consumption or consumption after exchange through a trophic web. The diversity of potential receptors makes definition of exposure pathways more difficult for aquatic receptors than for humans.

The respiration/inhalation route is a challenge to define in exposure assessments involving aquatic receptors because of the different types of respiratory organs, the dynamic nature of the respiratory process in water, and the intimate contact between a receptor and metals dissolved in waters. Metals can pass across respiratory surfaces from overlying or interstitial waters. Respiratory surfaces include fish gills, various molluscan and arthropod gill types, the pseudo-lungs of pulmonate gastropods, cells surrounding the sponge spongocoel, mammalian and avian lungs, and plant leaves. Further complicating the issue, some respiratory organs can also be involved in locomotion, excretion, ion regulation, and food capture, sorting and ingestion.

The adsorption route can involve uptake across a phytoplankton cell membrane, amphibian skin, arthropod exoskeleton, the general integument of an infaunal clam or annelid, or the egg membrane of a bird or amphibian. Absorption can occur from overlying or interstitial waters: these sources have very distinct chemistries that influence metal bioavailability. General tools for estimating exposure concentrations for each of these situations are discussed in Section 2.1.4.

Similarly, the ingestion route is difficult to define for aquatic receptors because of the diversity of feeding modes and food sources such as sediments, suspended solids, microflora, animal tissues, and plant tissues. Combining of respiratory, locomotive, or feeding structures complicates description of the ingestion pathway for some species. The presence of life stages that feed differently also confounds exposure assessment.

2.1.2 Dealing With Differences Between Laboratory Test Exposures and Field Exposures

Temporal aspects of exposure should be considered in assessments of any toxicant; however, rapid speciation and phase changes associated with changes in pH/eH make this issue particularly germane to metals. Fluctuating or pulsed exposures occur in situations such as snow melts, acidic precipitation events, rapid changes in eH/pH associated with photosynthesis and respiration, hypolimnetic discharge from stratified reservoirs, biocide (e.g., copper sulfate) spraying, and ingestion of prey items with seasonally high metal concentrations.

Most laboratory-derived effects data are associated with exposures of a standard duration, and very gross extrapolation is usually done to accommodate other durations. The most common and crudest extrapolation involves the assumption that, within the range of actual exposure durations, an estimate for one exposure duration suffices. The assumption may be made that the 96-hour exposure duration for an acute LC₅₀ (lethal concentration, 50 percent) test grossly reflects the consequences of a 45- or 70-hour field exposure. Solomon et al. (1996) approximated the effects of pulsed atrazine exposure in the field with laboratory 48- or 96-hour LC₅₀ data regardless of the actual duration of the short term exposure pulses noted in the field. In more relevant examples, this approach was taken for extrapolating from laboratory exposure data to field exposures for copper and cadmium (Hall et al., 1998) and tributyltin (Hall et al., 2000). The same expedient approximation might be done for chronic exposures. An effect metric produced from a chronic test featuring 28 days of laboratory exposure may be used to represent all chronic field exposures.

More accurate extrapolation methods can be applied to conventional laboratory data if the duration of a field exposure was different from that used in the laboratory. Inverse (van den Heuvel et al., 1991; Mayer et al., 2002), exponential (Gaddum, 1953), and power (Sprague, 1970) relationships have been applied to adjusting data for laboratory test durations to other field exposure durations. All assume a particular model and none accommodate latent mortality, i.e., post-exposure mortality that is due to damage during exposure.

When it is necessary to reduce uncertainty even further, a rich array of survival modeling techniques is available (e.g., Miller, 1981) that can easily include metal duration and exposure concentration in predictions (e.g., Newman and Dixon, 1996; Crane et al., 2002). Unlike conventional methods that note proportion dying at the end of an exposure duration, these methods note the times to death for each exposed individual. Survival time models fit to laboratory test data can be applied to different field exposure durations and, with straightforward adjustments, can also accommodate latent mortality. Application of survival time methods is

becoming increasingly common in ecotoxicology, especially at higher tiers of risk assessments, and carries the promise of more effective prediction for exposures of different durations.

Water quality differences in laboratory test and field situations produce differences in metal speciation and consequent bioactivity. Provided that test media characteristics are measured and reported, equilibrium models such as Minteq (Brown and Allison, 1987) can estimate the free ion concentration or activity of the test and field waters. The Free Ion Activity Model (FIAM) discussed in Section 2.1.4 can be used to identify the most relevant exposure concentration in the field, with the most relevant species in most instances being the free ion. Differences in metal solid phase association in laboratory test and field sediments present a more complex task during exposure assessment. Tools described in Section 2.1.4 provide potential ways of extrapolating from laboratory sediment tests to field exposures.

Because it has been recognized that receptors often are exposed to a distribution of concentrations, not a single concentration, a stochastic approach is emerging in risk assessments for aquatic systems (e.g., Aquatic Dialog Group, 1994). Most stochastic exposure assessments use Monte Carlo methods with known concentration distributions or bootstrap methods if knowledge of underlying distributions is absent. Consistent application of such a stochastic approach to metal exposure assessment will significantly improve risk characterizations.

2.1.3 Mixtures

In risk assessments, metal mixture exposure is often addressed within the hazard index calculation. As relevant examples, the EPA Superfund (U.S. EPA, 1997a) and specific mixture guidance (U.S. EPA, 2000) documents describe this approach for mixtures of similarly acting chemicals. The unfortunate consequence relative to this paper is that exposure and effect concentrations cannot be split neatly apart for separate discussion.

The common hazard index approach to estimating total metal exposure sums hazard quotients for each metal of concern and produces a hazard index: $\text{Hazard Index} = \sum_{i=1}^x \frac{E_i}{\text{RfD}_i}$, where E_i = exposure concentration (or intake) for the i^{th} metal and RfD_i = some effect reference concentration (or dose) for the i^{th} metal. The underlying assumption of a (pseudo)linear relation between exposure concentration and effect is used to justify this summing. However, summing concentrations becomes problematic with more than a few metals because the hazard index increases as more and more metals are included. Because most exposure concentration–effect models are sigmoidal, the assumption of pseudo-linearity also produces an upwardly biased hazard index in many cases. Many metals that at low concentrations would have a negligible joint effect according to a sigmoidal model, in combination, will produce a large hazard index according to a pseudo-linear approximation of the exposure concentration–effect models. This artifact is particularly troublesome for metals because many have background concentrations which are included in these summations. Furthermore, some metals are essential elements and the assumption of a monotonic, psuedo-linear relationship is especially inappropriate. Indeed,

using the hazard index approach, an essential metal present at such low concentrations as to produce a deficiency would be handled in calculations as if it were having a toxic effect!

Concentration summation might seem plausible in some cases if the RfD values reflected true effect thresholds, no dose/concentration–effect models were available, and the metals of interest caused the same effect by a common mechanism. In taking this approach, it would be important to know whether the metals were acting independently or similarly. Similarly acting metals could be summed, but the justification for summing metals with independent action is unclear. In the case of independent joint action for metals that are present at concentrations below their thresholds, it is difficult to understand why they should not simply be omitted from the hazard index calculation instead.

Deciding if metals should be summed in the hazard index requires some means of defining the nature of the metals' joint action. Such a decision could be aided by knowledge of the relevant metals' modes of action, essentialities, and capacities to act as analogs for other metals. Quantitative prediction of the degree of similar joint action of metals might also be approximated using similarities for ligand binding tendencies of the metals in mixture (Newman and McCloskey, 1996; Ownby and Newman, 2003).¹

Bias emerging from summing concentrations, instead of effects, can be avoided if the complete concentration-effect models are used to produce RfD estimates (e.g., conventional concentration-effect model metrics for acute effects such as the LC₅₀ or benchmark doses [U.S. EPA, 1995c]). A simple approach exists if one knows whether the metals in mixture have independent or similar joint action (Finney, 1947).

If the metals act independently, their combined effect (expressed here as a proportion affected or responding, P_{A+B}) can be estimated from the expected effects (P_A and P_B) calculated for each metal alone (A and B) based on their respective probit models, $\text{Probit}(P_A) = \text{Intercept}_A + \text{Slope}_A(\log \text{Concentration}_A)$ and $\text{Probit}(P_B) = \text{Intercept}_B + \text{Slope}_B(\log \text{Concentration}_B)$. The predicted joint effect would be the following:

$$\text{Predicted } P_{A+B} = P_A + P_B(1 - P_A) = P_A + P_B - P_A P_B$$

This model can be expanded if more than two metals are present in the mixture:

$$P_{A+B+C\dots} = 1 - (1 - P_A)(1 - P_B)(1 - P_C)\dots$$

¹Such a model predicting metal or metal mixture bioactivity based on metal ion characteristics is a Quantitative Ion Character-Activity Relationship, or QICAR (Newman et al., 1998). It is analogous to Quantitative Structure-Activity Relationships (QSARs) developed for organic compounds.

If the metals act similarly in the mixture, their slopes will tend to be parallel ($\text{Slope}_A = \text{Slope}_B = \text{Slope}$) and a simple estimate of relative potency (ρ_B) can be derived (Finney, 1947):

$$\log \rho_B = \frac{(\text{Intercept}_B - \text{Intercept}_A)}{\text{Slope}}$$

The combined effect of A and B is predicted with the following equation:

$$\text{Probit}(P_A + P_B) = \text{Intercept}_A + \text{Slope}(\log[A] + \rho(\log[B]))$$

The difficulty with using these approaches to avoid biases in the hazard index is identifying whether the metals in mixture are acting similarly or independently. Such a determination could be made with a literature review, consideration of published modes and sites of action for the metals, or prediction of potential interactions using HSAB-based Quantitative Ion Character-Activity Relationship (QICAR) models (e.g., Ownby and Newman, 2003). Literature on essential metals and their analogs might also provide insight. For the reasons just provided, one might consider grouping similarly acting metals together in exposure assessment of metal mixtures.

2.1.4 Exposure Estimation Tools

Descriptions of exposure pathways involving respiratory surfaces should include chemical speciation, chemical kinetics, and binding with biological ligands. The FIAM states that, in general, metal ion availability and effect are correlated with free ion concentration or activity (Campbell 1995, Brown and Markich 2000). Consequently, knowledge of free ion concentration or activity is crucial to fully describing exposure. The biological ligand model suggests that the bioactivity of a metal is a result of its interaction with biological ligands. It follows that a dissolved metal ion's bioactivity is a function of its complexation with dissolved ligands (which determines the free ion concentration or activity), and the affinity and stability of its complexes with biological ligands (which determines the realized dose or effect). Any expression of exposure concentration should consider these issues.

Equilibrium speciation can be used to estimate the free ion concentration or activity. In some cases, the free ion can be measured directly during an exposure assessment. Combining insights from the FIAM approach with those from Biotic Ligand Model (BLM) and HSAB theories allows general prediction of metal activity on biological surfaces as different as fish gills (Pagenkopf, 1983; Reid and McDonald, 1991; Janes and Playle, 1995), green algae (Crist et al., 1988; Parent and Campbell, 1994), and bacteria (Azenha et al., 1995). Entry across the integument could also be addressed with conceptual tools of the FIAM and BLM. As an example, surface adsorption was described in this manner for the chironomid exoskeleton (Krantzberg and Stokes, 1988).

The FIAM model is a powerful tool as long as it is applied with enough understanding to anticipate or recognize exceptions. As examples of important exceptions, charged uranium complexes are toxic in addition to the free ion (Markich et al., 2000) and the neutral mercury complex, HgCl_2^0 , is bioavailable due to its high lipophilicity (Simkiss, 1996).

In the absence of sufficient information about speciation, exposure concentrations for dissolved metals such as beryllium, cadmium, copper, lead, and zinc might be normalized based on water hardness. Most practitioners fit linear models ($\log \text{ of Toxicity Endpoint} = \log a + b [\log \text{ of Hardness}]$) and backtransform them to produce a normalizing function, $\text{Toxic Endpoint} = 10^a (\text{Hardness})^b$. This power function contains an easily corrected backtransformation bias that should be removed from predicted hardness-adjusted exposure concentrations (Newman, 1991).

Defining the particulars of metal exposure by ingestion is complicated by the diversity of the feeding modes and digestive systems of candidate receptors, as well as the possibility of exposure after passage through a trophic web. For the simple transfer of metals from algae to zooplankton, Reinfelder and Fisher (1991) noted high assimilation efficiencies for Class B and intermediate metals which have relative affinities for ligand donor atoms of $S > N > O$ and consequent strong binding to soluble, cytoplasmic algal components. Class A metals with binding affinities of $O > N > S$ showed contrastingly low assimilation efficiencies. Such a general trend could be used in predicting potential exposure for one metal based on information about other metals, or the potential for exposure to different metals as a result of trophic transfer.

Ingestion exposure assessments have higher levels of uncertainty for more complicated situations, and it is a challenge to accurately determine the extent to which metals associated with different solid phases are taken up by detritivores or suspension feeders. Choosing the best expression of exposure concentration for a metal becomes crucial to reducing uncertainty.

Several approaches have been taken for sediment-associated metals. One approach normalizes or compares the amount of “acid-volatile sulfide” (AVS; 1 N HCl extractant) in sediment to the amount of simultaneously extracted metal (SEM); when the molar concentration of AVS exceeds that of metals ($\text{SEM-AVS} < 0$), then the metals are expected to associate with the solid phase rather than interstitial water. It is further assumed that this low activity of metals in interstitial water is indicative of low toxicity to benthic organisms (Di Toro et al., 1990). The ability of this SEM-AVS approach to predict toxicity of metals in sediment has been explored closely for Class B or borderline Class B metals (Di Toro et al., 1990; Ankley et al., 1991; Carlson et al., 1991; Berry et al., 1996; Hansen et al., 1996; Ankley et al., 1996) which form extremely insoluble sulfides. While the correspondence of SEM-AVS to toxicity was found to be strong in these studies, some have questioned the adequacy of the approach because of its reliance on the chemistry of bulk anoxic sediment to characterize toxicity to organisms that often live in oxygenated burrows and/or ingest and reprocess sediment. For example, Lee et al. (2000) exposed benthic organisms to a range of sediments with varying concentrations of AVS and SEM. In these studies, the researchers found that whole body burdens of metals correlated better with total metal concentration in sediment than with metal concentration in interstitial water for some benthic organisms, and that exposure-dependent uptake of metals occurred even when

sulfide was in excess of metal and metal concentrations in interstitial water were low. These findings call into question the assertion that metals in sediments with excess sulfide are in fact biologically unavailable. At the same time, the only toxic responses observed by Lee et al. occurred in sediments where SEM was greater than AVS (as would be predicted by the SEM-AVS approach), and these responses were not correlated with metal accumulation. The lack of correspondence between whole body burden of metal and toxic response in these and other studies makes clear an important difficulty in expressing metal bioavailability and in evaluating metal exposure in sediment. Better understanding of the mechanisms of metal accumulation from sediment and their relationship to toxic effects is badly needed.

Other tools exist for determining the exposure concentration of sediment-bound metals. Metal concentrations in chemical (Tessier et al., 1984; Babukutty and Chacko, 1995; Fan and Wang, 2001), acid (Luoma and Bryan, 1978; Langston, 1980), or biomimetic (Chen and Mayer, 1998; Mayer et al., 2001; Weston and Maruya, 2002) extracts provide acceptable estimates in some instances, but no consensus exists yet about their best use for different types of metals or metalloids. Based on the premise that iron oxides in oxic sediments lower metal bioavailability, iron in a 1 N HCl sediment extract has been used to normalize metal exposure concentrations (Luoma and Bryan, 1978). Increasing concentrations of organic carbon can decrease metal bioavailability (Creclous et al., 1982), so normalization of sediment metal concentrations to organic carbon content has been useful in other cases. The more readily extracted metals from sequential chemical extraction schemes tend to be the most bioavailable (Tessier et al., 1984; Young and Harvey, 1991) and can be used as exposure metal concentrations.

Exposure assessment for benthic receptors could be enhanced if a clearer consensus were reached about the utility of each of these methods for different classes of metals, biological species, and sediment types, e.g., oxic versus anoxic or organic rich versus low organic sediments. Explicit recommendations about the best metal assessment scenarios to apply in different situations would benefit those trying to accurately quantify the most relevant exposure concentrations. Yet the needs highlighted by Luoma (1989) more than a decade ago remain unanswered for assessing exposure to sediment-bound metals: “1) improved computational or analytical methods for analyzing distribution of metal among components of the sediments; 2) improved computational methods for assessing the influences of metal form in sediments on sediment-water metal exchange; and 3) a better understanding of the processes controlling bioaccumulation of metals from solution and food by metazoan species directly exposed to the sediments.” Although this lack of fundamental knowledge precludes definitive recommendations, the National Academy of Science (2002) has tabulated the qualities of several of the options just described, producing the following general conclusions (Table 4-2 and associated text):

- No approach is acceptable as a universal method “that meets all needs for characterizing the complex processes that determine bioavailability.”

- Normalization to organic carbon content, sediment grain size, or cation exchange capacity of sediments does not generally lead to understanding bioavailability but is sometimes used for regulatory purposes.
- Extractions, AVS, or sequential extractions are better accepted for soils than sediments. They can establish useful correlations but not mechanistic insight about bioavailability. Fractions should be considered operationally defined and not strict measures of phase association.
- Biomimetic methods are simple but generalizations about results are difficult to make. Consequently, results from such studies tend to have site-specific utility.
- AVS/SEM or SEM-AVS methods can be helpful for inferring biological uptake by correlation. The approach was proposed for regulatory purposes due to its simplicity but controversy exists about generalizations from the approach.

As Section 1.3 explains, exposure assessment can be complicated if exposure assessment requires estimation of trophic exchange within a complex food web. Fortunately, well-established N isotope techniques can aid in defining the trophic status of a species. The ^{15}N concentration increases relative to ^{14}N concentration with each trophic exchange, and statistical models can link trophic status to metal concentration (e.g., Cabana et al., 1994). This approach can and should be applied more widely in exposure assessments involving trophic transfer of metals in aquatic systems.

2.1.5 Dynamic vs. Steady State Assumptions

Linkages of exposure concentration to realized dose often carry the assumption of a steady state concentration within the receptor and media. Steady state concentrations for some metals can be reached within the receptor's life span, but this is not always the case. Concentrations can vary with time and, in such instances, steady state concentrations are irrelevant. Instead, a range of exposure concentrations might be expected and a stochastic approach might be more appropriate than a simple deterministic model.

In some situations, equilibrium-based speciation models can provide inadequate description and chemical kinetics need to be considered. Strong gradients in pH, eH, and ionic composition of waters exist and can result in speciation gradients. Contaminated groundwater can seep into a flowing stream with consequent shifts in associated metal species and phase associations. Changes in photosynthesis and respiration produce diurnal and annual shifts in the eH/pH of water and surficial sediment. Bioturbation and infaunal activities produce dynamic microhabitats with strong chemical gradients in sediments and waters at the sediment-water interface. As one example, burrows of the polychaete *Arenicola* sp. create oxygenated microenvironments in sediments. Related shifts in equilibrium concentrations of metal species are not always rapid enough to be handled adequately with equilibrium-based computations. As a general illustration of this point, adding calcium to an equilibrated solution of copper and an

organic ligand does not immediately result in a new steady state. The process can be slow enough to make kinetics relevant (Hering and Morel, 1988). Regardless of these important exceptions, most situations involving dissolved metals can be addressed adequately by assuming near-equilibrium conditions (Hoffmann, 1981), as long as one is sufficiently aware that some conditions can lead to important non-equilibrium dynamics. The equilibrium approach should be abandoned when kinetics seems important.

Another context in which kinetics becomes an important issue is exposure at the gill surface microlayer, where gradients in pH and ionic composition exist due to respiration, excretion, and ion regulation. Dissolved aluminum toxicity to freshwater fish is a good illustration of this point (Playle and Wood, 1990). The deposition of aluminum on gills is determined largely by the rapid shift from dissolved, ionic aluminum to an $\text{Al}(\text{OH})_3$ precipitate on the gill surface due to the more alkaline state of gill microlayer water. How much aluminum speciation and phase association changes is a function of the initial pH of the bulk water. Exposure assessment should discuss metal speciation kinetics in such situations or inaccuracies will emerge in subsequent effects assessment.

2.1.6 What To Do When Information Is Limited

Many uncertainties emerge during attempts to define metal exposure of aquatic receptors. Two overarching recommendations are warranted by this high uncertainty. First, more widespread use of stochastic expressions for exposure is recommended. Second, all else being equal, one should favor receptors with the clearest, and therefore most tractable, exposure pathways.

Specific recommendations emerge from the above discussions of existing tools for defining exposure:

- Uncertainty in extrapolation from laboratory exposures to field situations can be reduced using the methods mentioned in Section 2.1.2 to adjust for differences in exposure durations.
- Latent mortality can be important for metals that cause general cell damage (e.g., gill lesions) but is not currently included in estimates of exposure consequences. Presently, all effects are restricted to the exposure period and ignore the post-exposure period. Effects after exposure ends could be included with minor changes to toxicity testing procedures, i.e., noting effects for a period beyond the duration of exposure.
- Expression of dissolved metal exposure can be enhanced by applying chemical speciation calculations and the FIAM concept; however, exposure assessors should be open to addressing chemical kinetics when warranted.
- Linking HSAB, FIAM, and BLM theory, the QICAR approach (Newman et al., 1998) has promise for extrapolation among metals and for predicting metal interactions. However,

more research is needed for this approach before general application is recommended.

- Exposure characterization for metal mixtures could be improved with the methods described in Section 2.1.3. The general application of hazard index methods which assume concentration additivity is not recommended, for reasons discussed in Section 2.1.3.
- Exposure assessment for benthic receptors could be enhanced if a clearer consensus were reached about the relative utility of each of the methods described in Section 2.1.4. At present, the best consensus statement seems to be that of the National Academy of Science (NAS, 2002).
- Uncertainty associated with exposure via food web transfer can be reduced using nitrogen isotopes to quantify trophic status of pertinent members of the trophic web.

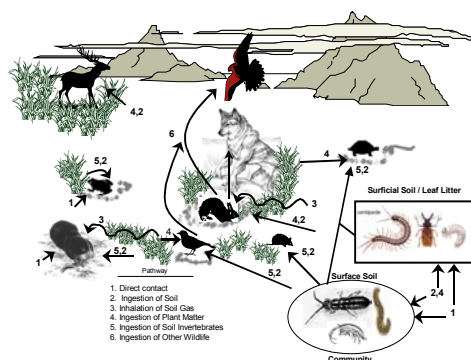


Figure 1. Potential pathways for metal exposure in terrestrial ecosystems.

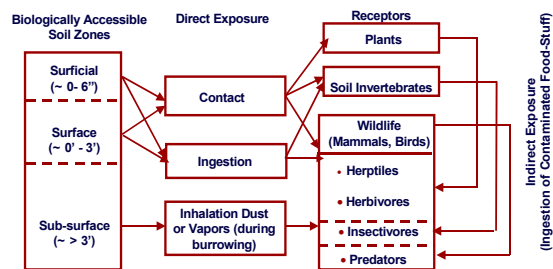


Figure 2. Conceptual model for direct and indirect exposure of ecological receptors to metals in soil zones.

2.2 Terrestrial Receptors

2.2.1 Sources, Phases, and Exposure Pathways

Conceptual exposure models for metal in soils are given in Figures 1 and 2 (see previous page). While these show a number of discrete pathways, they are oversimplifications. Yet they do indicate that metal exposure in terrestrial systems requires a consideration of where metal-contaminated soil is located, as well as how biota may come into contact with the metals present in the soils. As the conceptual models illustrate, exposure can be direct (e.g., as is the case for soil invertebrates) or indirect (as is the case for animals that feed higher in the food web). The importance of the pathways shown in the two figures depends on the metal as well as the form of that metal. While these simple conceptual models illustrate relationships within the terrestrial system, there can be connections between the terrestrial, wetland, and aquatic environments.

Terrestrial communities that can be exposed to metals include microbial communities, plants, invertebrates, and terrestrial wildlife (including terrestrial life stages of some amphibians, reptiles, birds, and mammals). Microbial communities can also be exposed to metals in soil, although in general, microbial communities are not evaluated in ecological risk assessment. Microbial communities can affect the form or speciation of metals, and play an important role in cycling nutrients in soil. Plants' leaf areas can be exposed via aerial deposition of metals, and a wide variety of plant species have been shown to take up metals from soil. Cataldo and Wildung (1978) and Raskin et al. (1994) summarized the mechanisms by which plants can take up metals. Greszta (1982) demonstrated uptake of copper, zinc, lead, and cadmium into a variety of tree species. The highest accumulation of metals was detected in the roots, although other parts of the plant also accumulated metals to varying degrees. Mitrofanov (1993) demonstrated metal uptake by a variety of non-wooded forested plants. With the exception of a few hyperaccumulators, most plant species do not bioconcentrate metals to a great degree. Hyperaccumulators have been used in phytoremediation programs to address metal-contaminated soils.

Wildlife and invertebrates exposed to metals in soil have been shown to accumulate metals from soil and food ingestion. Talmage and Walton (1990) used small mammals as indicators of metals in the environment. Pascoe et al. (1994) also measured metal concentrations in small mammals exposed to mining waste. These types of field studies will likely be important components of determining actual exposures to wildlife species. They can be used to determine the degree to which specific metals are being accumulated in smaller mammals and can also be used to support food-web exposure models to other wildlife species. Empirical studies such as that of Pascoe et al. (1994) can be used to establish relationships between metals in soils and in small mammals.

Factors or mechanisms that control food chain exposures to metals are bioavailability, bioaccessibility, and bioaccumulation. The availability of metals in soils depends on whether exposure occurs via pore water or other pathways external to the organisms. Bioaccessibility of metals to animals and plants that live on or in the soils can be influenced by a number of soil parameters such as pH, cation exchange capacity, and organic carbon. These soil factors tend to be less important for soils that are incidentally ingested into wildlife species.

Many assessments make the assumption that metals are 100% bioavailable or bioaccessible from soil. Many studies have indicated that following the introduction of metals to soils, the relative bioavailability of those metals to invertebrates, plants, and wildlife tends to decrease with time. This reflects a variety of geochemical and physical processes. Metals can be loosely bound to the surface of the soil particle or can be an intrinsic part of the soil particle, not available for uptake. Pascoe et al. (1994, 1996) measured metals concentration in small mammals exposed to mining waste contamination in a wetland and used this information to evaluate bioavailability of metals in soil. They measured and modeled metals concentrations in small herbivorous mammals given concentrations in soil, food, and water, and estimated bioavailability to these organisms. The bioavailable fraction of metals was only a small fraction of the total metal concentration in soil. Research on bioavailability and absorption is ongoing for application in human health risk assessments. Some of this research may be applicable to wildlife (e.g., for small mammals), but in some cases, the soil matrix and digestive systems of wildlife may be different enough that the information is inapplicable.

Most metals can be taken up by terrestrial receptors but usually to concentrations lower than that in soil. However, some organisms are hyperaccumulators of specific metals. These can be plants (Raskin et al., 1994); however, some invertebrates such as isopods and pulmonate mollusks also accumulate copper to very high concentrations (Hughes, 1981). The form of the metal in the environment, or its speciation, also influences its uptake.

Insight into the relative importance of pathways for metals can help focus data gathering or modeling efforts. In addition, an examination of the pathways most critical for exposure helps identify key data gaps in information or methods that can be addressed through focused research programs. Typically, default assumptions are used during screening-level assessments. An example of where this has been done for metals is the ECO SSL document (U.S. EPA, 2003). That methodology is not repeated here but it is the most current summary of “default values.” As site-specific information is acquired and applied, the relative importance of a pathway can change and may need to be reevaluated before gathering additional data.

The bioaccumulation of metals into plants and in soil invertebrates is discussed elsewhere in this document. These are starting places for evaluating exposure to wildlife species. Food and the incidental ingestion of soil are the two most important pathways for most wildlife species and their relative importance varies by species of animal as well as by metal (U.S. EPA, 2003). This has already been evaluated elsewhere (Menzie-Cura and TN&A, 2000) (Table 1). The U.S. EPA’s (2003) ECO SSL workgroup has developed a white paper that addresses inhalation and dermal contact exposures for wildlife. Its conclusion was that there are certain chemicals and exposure situations for which these pathways are important. However, under most situations these pathways are less important sources of exposure for metals than are food pathways or incidental ingestion of soil.

Table 1. Source Categories of Metals for Six Sentinel Species (Herbivores, Carnivores, Vermivores). Based on Application of Food-chain Models with Default Exposure Assumptions

Species	Contaminants Taken Up Predominantly From Soil (> 75%)	Contaminants Taken Up From Both Soil and Food	Contaminants Taken Up Predominantly From Food (> 75%)
Meadow vole (herbivore)	Al, Co, Fe, Vn	Sb, Cr, Ag	As, Ba, Cd, Cu, Pb, Mn, Ni, Se, Zn
Mourning dove (herbivore)	Al, Sb, Co, Fe, Ag, Vn	As, Ba, Cr, Pb, Mn, Ni	Cd, Cu, Se, Zn
Long-tailed weasel (carnivore)	As, Ag	Al, Ba, Co	Cd, Cr, Cu, Fe, Pb, Ni, Se, Zn
Red-tailed hawk (carnivore)	Ag	Al, As, Ba, Co, Mn, Vn	Cd, Cr, Cu, Fe, Pb, Ni, Se, Zn
Short-tailed shrew (vermivore)	Al, Fe, Vn	As, Ba, Co	Cd, Cu, Pb, Mn, Se, Ag, Zn
Woodcock (vermivore)		Al, As, Ba, Co, Fe, Vn	Cd, Cu, Pb, Mn, Se, Ag, Zn

TN&A and Menzie-Cura reached the following conclusions regarding exposure pathways evaluated using standard models and default assumptions:

- Chemicals for which uptake into food (either plants or soil invertebrates) tends to be the most important route of exposure include cadmium, copper, lead, nickel, selenium, and zinc.
- Chemicals for which incidental soil ingestion (and some food uptake) tends to be the most important pathway include aluminum, iron, and vanadium.
- Incidental soil ingestion is a proportionally more important pathway for herbivores.
- Uptake into soil invertebrates (e.g., earthworms) is a proportionally more important pathway for animals that feed on these organisms. (Note: This assessment reflects work done with earthworms and may not apply to other hard-bodied soil invertebrates.)

Experience at metals-contaminated sites indicates that the above generalizations must be viewed with caution. As site-specific information is acquired, the relative importance of pathways may change. For example, site-specific data may show that the accumulation of a chemical into plants or soil invertebrates is much lower than indicated by the default assumptions. In such cases, the incidental ingestion of soil would become proportionally more important. The bioavailability of metals in incidentally ingested soil is also variable, as discussed

later. Therefore, when the exposure is being driven by incidental ingestion, refinements of exposure estimates can benefit from a better understanding of bioavailability.

While most of the pathways begin with soil contact, other pathways (that may begin with the soil) can be important. For example, metals can reach terrestrial receptors through the air. Aerial deposition of particulate matter containing metals from incineration or power generation sources can reach soil and vegetation, resulting in exposure of terrestrial receptors. Terrestrial wildlife can then contact or ingest the metals on vegetation or soil. Bache et al. (1991) addressed metals in grasses at varying downwind distances from a municipal refuse incinerator. Exposure can result from metals on the surface of the vegetation or taken up into the vegetation. Terrestrial receptors can also inhale particles. This exposure, although possible, is usually not addressed in ecological risk assessment because it is assumed to be insignificant in comparison to soil ingestion exposures.

Research on exposure of soil invertebrates and plants to metals in soils has typically involved relatively few species, and this influences our understanding of exposure as reflected in the conceptual models. It is helpful to consider the anatomy, physiology, and ecology of the various animals and plants where information is limited on exposure studies. For example, exposure of soil invertebrates likely depends strongly on such characteristics. Earthworms and other soft-bodied invertebrates may come into steady state with the soil porewater concentrations. However, hardbodied bugs (e.g., springtails) do not; rather, they are exposed through their diet. Better understanding of the soil invertebrate food will enable better (i.e., more realistic) estimations of exposure. While there has been an increased emphasis on evaluating exposure to arthropods, most exposure work has been performed using relatively few earthworm species.

2.2.2 Differences Between Laboratory Test and Field Exposures

The importance of matrix effects is well recognized and generally dealt with under the topic of bioavailability, which is covered in another paper. However, the most obvious issue with respect to exposure assessment for essentially all of the terrestrial receptors is the fact that most of the available toxicity data have been generated for the readily bioavailable forms of the metals. These data are useful for evaluating the relative toxicity potential of metals but do not reflect the exposure that occurs in soil systems. In the bioavailability paper, there is a detailed discussion of the geochemical factors that influence the environmental fate and exposure of metals in soils. As is clear from that paper, there is considerable uncertainty when laboratory data on metals toxicity is extrapolated to soil exposures. To address this extrapolation problem, recent evaluations that focus on soil exposures are incorporating a processing step wherein soils amended with specific metals are allowed to age or weather. Various procedures for doing this have been developed (McLaughlin et al., 2002; Ronald T. Checkai, U.S. Army CBDCOM/ERDEC, personal communication, 2003). An alternative approach is to work with field soils where actual contamination is known to exist. This latter approach is especially useful for site-specific assessments but often lacks information on the mechanistic basis for binding of the metals to the soils.

Other aspects of exposure must also be considered. Some metals are essential for the normal growth, development, and reproduction of organisms. These include essential macronutrients (calcium, magnesium, potassium, and sodium) and essential micronutrients (chromium, cobalt copper, iron manganese, molybdenum, nickel, selenium, vanadium, and zinc) (Chapman and Wang, 2000). Therefore, when evaluating exposures that are potentially harmful, one should make a distinction between the amounts that are required, total exposures, and incremental increases above the amounts needed by the organism, i.e. the exposure zone within which harmful effects may be manifested.

Because of spatial variations in natural background, as discussed later, biological communities that reside in different regions may be adapted to the levels of metals present in the soils of those regions. Laboratory organisms may be more or less sensitive than native organisms adapted to a particular soil (Janssen et al., 2000). If background levels of metals in soil are naturally high, and laboratory-cultured organisms have had a lower level of metals exposure, use of laboratory organisms could result in effects that would not be evidenced in the environment. On the other hand, organisms may become acclimated to particular higher levels of metals during culturing (Janssen et al., 2000).

2.2.3 Mixtures

While there is limited information available about exposure of terrestrial receptors to mixtures of metals, this issue has been receiving increased attention (Raskin et al., 1994; Kabata-Pendias and Pendias, 2000). For mammals, some insights can be gained from the work performed on domestic animals (NRC, 1980).

2.2.4 Considering Background

Metals are naturally occurring in the environment, and there are areas where elevated concentrations of particular metals exist naturally in soil. Background concentrations in soils can vary by as much as five orders of magnitude depending on soil type, geography, and other factors (Chapman and Wang, 2000). Biological communities can adapt to the presence of metals (Janssen et al., 2000). A dose that is toxic in one natural setting may be tolerated in another. Therefore, it is important to consider whether exposure to background concentrations is important. If the local community is adapted to background, even if it is an elevated background, an exposure would have to be outside of that site-specific background to be of concern.

2.2.5 Estimating Exposure

Microbial Communities. Exposures of soil microbial communities to metals is most commonly evaluated with respect to soil pore water. The underlying assumption is that the metals must be available to the pore water before being available to the organisms. This is obviously an oversimplification but may suffice as a rough approximation. Therefore, measures of metals in soil pore water or in selected aqueous extracts of soils may be useful as measures of exposure of the soil microbial community. The bioremediation literature is a useful source of information for evaluating exposure of microbial communities in soil systems. Exposures of the

microbial communities are influenced by soil characteristics. Most investigations measure soil parameters (e.g., organic matter and moisture content) that can be used to help judge exposure of microbial communities (Hull et al., 1999).

Plants. Some aspects of plants exposure to metals can be measured quite easily, such as bulk metals concentrations, soil moisture content and soil pH. Uptake through leaf surfaces depends on whether deposited particles are dissolved or washed off, and hence depends on properties of the particle (Hughes, 1981). Factors affecting uptake into roots are soil pH, the presence of other ionic species (e.g., chloride, hydroxide), organic matter, clay content, and cation exchange capacity (Martin and Coughtrey, 1981). Various authors cited in Martin and Coughtrey (1981) have demonstrated that, even if the source of metals is aerial deposition, the soil is the sink for these contaminants and exposure is predominantly through the soil. A great deal of research with regard to plant exposure and uptake of metals has been conducted for crop species.

Terrestrial vascular plants have long been used to evaluate exposure to metals. Plant species have been used to identify elevated metal exposures in solid “terrestrial” matrices such as contaminated soils, sludges, and other solid industrial wastes, as well as their leachates or eluates. There has been less laboratory investigation of the bioaccumulation of chemicals from soils into plants for hazardous waste site application. In some cases, bioaccumulation is examined in plant growth tests. At the end of such a test, the plants are harvested and the tissues analyzed. Most of the standard laboratory tests focus on species of some commercial interest, including lettuce (*Lactuca sativa*), millet (*Panicum miliaceum*), oat (*Avena sativa*), rye grass (*Secale cereale*), soybean (*Glycine max*), radish (*Raphanus sativus*), sorghum (*Sorghum vulgare*), bush bean (*Phaseolus vulgaris*), tall fescue (*Festuca arundinacea*), cucumber (*Cucumis sativa*), rice (*Oryza sativa*), and wheat (*Triticum aestivum*). The sites at which terrestrial plant tests have been applied include industrial facilities such as chemical, pesticide, and ordnance manufacturing sites; Superfund sites; hazardous waste sites; spill sites; mining facilities; and military facilities. The most common matrix tested with plants is soil. However, the literature also includes references on sludges, ashes, mine tailings, spoil sites, and asphalt disposal pits. Some work has been carried out to evaluate exposure to natural (rather than crop) plant species (Larry Kapustka, ecological planning and toxicology, inc., personal communication, 10/2002). This is an important area of future investigation. It would be useful to determine the extent to which standard plant test species capture the range of exposures at issue for plants in terrestrial systems.

Davies (1992) has developed models to predict heavy metal (cadmium, copper, lead, and zinc) accumulation in the radish (*Raphanus sativus* L.) based on heavy metal soil concentrations and analyte-specific extraction methods. With the exception of zinc, soil pH did not significantly improve the predictive power of the models. An accurate model for copper could not be produced.

Invertebrates. Exposure of invertebrates to metals occurs in soil litter and soil. Significant research is available on the uptake of metals by invertebrates, in particular earthworms (Martin and Coughtrey, 1981). Sample et al. (1998a) developed uptake models to

predict concentrations in earthworms from soil concentrations. These can be used to estimate the exposure of vermivorous wildlife (e.g., song birds, voles, and shrews).

For selected metals (arsenic, cadmium, copper, mercury, manganese, lead, and zinc), the best estimate of tissue concentration in earthworms was given by a simple ln-ln regression analysis (Sample et al., 1998a). The addition of soil pH data to the regression model did not markedly improve fit. If soil calcium results were incorporated into the regression model, a better fit was obtained for cadmium and lead but not for other metals. Tissue concentrations were inaccurately estimated for the transition metals nickel and chromium, by both simple and multiple regression models. However, for most analytes, the UFs reported by Sample et al. (1998a) were generally in agreement with the results of previously published studies. For general estimates, Sample et al. (1998a) concluded that log-linear regression models may be used as bioaccumulation models for arsenic, cadmium, copper, mercury, manganese, lead, and zinc in earthworms, but recommend the use of the upper 95 percent prediction limit on the simple regression for the conservative estimates that are required for screening ecological risk assessment.

Wildlife. Food chain modeling can be used to estimate the exposure of wildlife to metals based on ingestion of soil, food, and water. The U.S. DOE (1994), among others, present the basic format of the model. Measured concentrations of metals in soil, surface water, and food items can be used in the model. Concentrations in food can also be estimated or modeled—e.g., the earthworm models developed by Sample et al. (1998a) or the small mammal bioaccumulation models developed by Sample et al. (1998b). These models can be modified to account for bioavailability of metals from each exposure medium, if information is available.

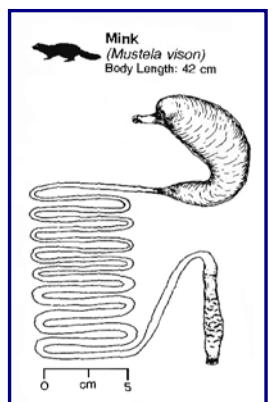
In general, the two most important pathways for wildlife are food and incidental ingestion of soil (U.S. EPA, 2003). As noted earlier, these vary by species and by metal. The relative importance of these pathways is dictated by the fraction of metal-contaminated soil in the diet and the amount of accumulation of metal into food items. The bioavailability of the metals in the ingested soil is an important consideration for situations in which soil is a potentially significant contributor to the exposure. Table 2 provides a simple scheme for judging the relative contribution of food and soil before accounting for bioavailability. The values in the table are the percent contribution of exposure associated with the incidental soil ingestion pathway. The highest values are in the lower left-hand side of the table. Incidental ingestion of soil becomes proportionally more important for exposure to wildlife when a) the bioaccumulation factor from soil to food (e.g., to plants or soil invertebrates) is less than “1” and b) the fraction of soil in the diet is greater than 1%. With respect to refinements of exposure estimates, an understanding of the bioavailability of metals in incidentally ingested soils would be most useful for the exposure combinations in the lower left-hand region of the table and would not be particularly useful for combinations in the upper right-hand region.

Table 2. Percent Contribution of Incidental Soil Ingestion to Oral Dose for Wildlife at Different Soil Ingestion Rates and Bioaccumulation Factors and a Bioavailability of 100%

	Bioaccumulation Factor From Soil to Food Organism			
% Soil in Diet	0.01	0.01	1.0	10
0.01	0.99%	0.099%	0.01%	0.001%
0.1	9.1%	0.99%	0.1%	0.01%
1.0	50%	5%	1%	0.1%
10	92%	52.6%	10%	1%

Attention is paid to the bioavailability of metals for which incidental soil ingestion is the predominant pathway and where ecological risk is indicated, attention is given to the bioavailability of these metals. Very little information is available on this for most wildlife species. Menzie-Cura and TN&A (2000) have reviewed the issues related to the development of such information. One of the most significant challenges is that the bioavailability of metals may be influenced by differences in digestive physiology and anatomy across the broad and diverse range of mammalian and avian species. For example, metals present in soils may be more or less bioavailable within the gut of an herbivore that relies on fermentation as compared to the comparatively simpler gut of a carnivore that is designed to break down proteins. These gut systems differ in chemistry (including pH) and residence time. Illustrations of anatomies are shown in Figure 3 to illustrate this point for just two of the many variations in the anatomy of wildlife digestive systems. These potential differences among species must be considered when developing data on the bioavailability of metals in soils incidentally ingested by wildlife. This is yet to be explored with any rigor.

Mammal Carnivore Digestive System



Hindgut (Caecum) Fermenter

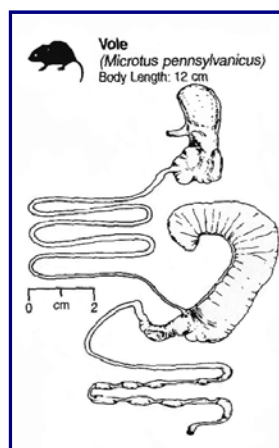


Figure 3. Distinct digestive systems of mammalian carnivore and fermenter.

2.2.6 What To Do When Information Is Limited

As noted above, the ECO SSLs and the methodologies developed to derive them provide an appropriate set of defaults for judging exposure when other information is missing (U.S. EPA, 2003). However, these approaches have not been developed for all ecological receptors.

2.3 Human Health Receptors

2.3.1 General Considerations

Assessment of human exposures to metals includes 1) determining routes of exposure; 2) determining the concentrations of the metals in the primary media of human contact; 3) estimating the intakes of the metal in those media through all relevant physiological routes of entry to the body; 4) describing and, where possible, quantifying relationships between exposure concentrations and intakes, as well as the sources of variability that affect these relationships; and 5) identifying sources of uncertainty and, where possible, quantifying uncertainty in estimates of exposure.

While the above elements of exposure assessment apply equally well to organic contaminants, typical exposures to metals offer unique challenges to human exposure assessments that are intended to serve risk assessment. Because metals are natural constituents of the earth's crust, human exposures to metal contaminants emanating from specific anthropogenic sources nearly always occur in context with exposure to local natural background levels. An added complication in assessing exposures to metals is that human exposures to any given metal typically are usually to a mixture of various chemical species of the metal (e.g., oxidation states or metal compounds), often in various physical or chemical associations (i.e., complexes) with organic or inorganic components of exposure medium (e.g., sediment, soil, water, or air). Furthermore, the types and amounts of chemical species and complexes can vary from one exposure medium to another. This complexity must be considered in risk estimates that attempt to associate environmental exposures to experimentally derived, dose-response relationships (e.g., reference dose, reference concentration, or cancer slope factor), because these are often based on experimental studies in which the exposure context may have been much different and is usually less complex.

The topics of physical and chemical speciation of metals, and their fate and transport in the environment, are discussed elsewhere in this report, and issues therein are not specific to human exposures. The possible exception to this is speciation of metals in treated drinking water, which is an exposure medium less relevant to ecological receptors. Therefore, this discussion focuses on the major challenges to assessing intakes of metals at the major routes of entry into the human body. These challenges, by and large, relate to difficulties and uncertainties in associating concentrations of metals, as typically measured in the field, with the behavioral (e.g., hand-to-mouth activity) and physiological (e.g., respiration) processes that result in intake.

To illustrate the complexity of human exposure pathways, Figure 4 shows an example of a prominent pathway of exposure of children to lead, the dust ingestion pathway (U.S. EPA, 1986). The primary medium of contact is surface hand dust, which can be ingested as a result of hand-to-mouth activity. However, lead transfers from antecedent media (e.g., exterior and interior dust, soil, atmospheric deposition, paint) contribute to the levels of lead in the primary medium. Ideally, assessment of exposures through this pathway would be made by measuring concentrations and intakes of hand dust, the primary exposure medium; however, in practice, measurements of lead in antecedent media (e.g., indoor dust and soil) often must serve as surrogates for the primary medium. Models must then be used to predict intakes of lead from the primary medium from measurements made in antecedent media.

The routes of entry of metals into humans that are most frequently encountered in environmental assessments include ingestion, inhalation, dermal and transplacental. Other routes can be important in specific circumstances, for example, explosions of metal-bearing materials can result in intra- or trans-dermal exposures (Robinson et al., 1983). Primary environmental media of contact include air, surface dust, food, water, and sediment. Other media that may sometimes be relevant to (or complicate) environmental risk assessment include medicines, cosmetics, and dietary supplements; paints, pigments, and other surface coatings; and tobacco.

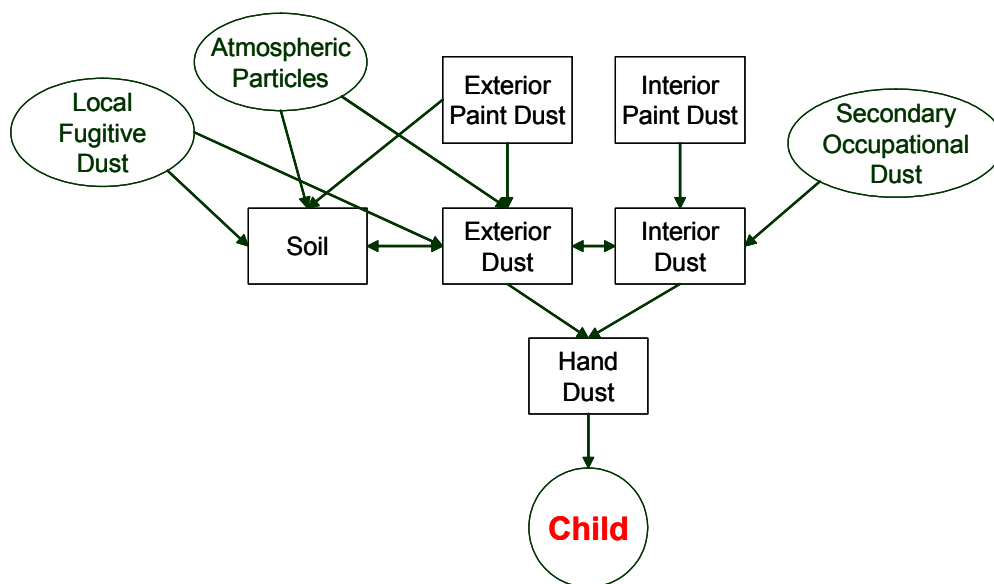


Figure 4. Diagram of the hand dust exposure pathway for exposures of children to lead (from U.S. EPA, 1986).

2.3.2 Air Pathway

The major pathways for human intakes of metals in which air serves as the primary medium of contact are inhalation and dermal. Other indirect pathways in which air serves as an antecedent medium include 1) deposition of metals to surface dusts and intake from ingestion, inhalation, or dermal contact; 2) deposition to surface water and sediment and intake from ingestion and dermal contact; and 3) uptake of deposited metals into aquatic and/or terrestrial biota, entrance into the human food chain, and intake from ingestion. Issues related to the indirect pathways are discussed elsewhere in this report where surface dust, water, and diet are discussed as primary media of contact.

Although it is often assumed that the dermal pathway makes only a minor contribution to internal doses from exposure to air-borne inorganic forms of metals, this assumption rarely has an empirical basis for specific metals. Few studies have attempted to quantify the extent or kinetics of dermal penetration of metals deposited on the skin (Hursh et al., 1989; Ilyin et al., 1975; Stauber et al., 1994; Wester et al., 1992; Hostynek et al., 1993); thus, quantifying the dermal contribution in risk estimates is usually largely guesswork. Dermal penetration may be a highly significant pathway for air immersion exposures to lipid-soluble inorganic forms of metals, such as mercury vapor (Hursh et al., 1989).

Although, in most instances in which air-borne metals have resulted in environmental contamination, ingestion of surface dust tends to be the dominant contributor to human health risk, this may not always be the case. Bioavailability of inhaled metals can be much higher than for other routes of intake. This can result in relatively high internal doses from inhalation even when inhalation intakes are similar to intakes from other routes. An example of this is the large contribution made by cigarette smoking to the body burden of cadmium (e.g., Ellis et al., 1979; Friis et al., 1998). Infants and children can be particularly vulnerable to air-borne metal particulates because differences in airway geometry and airstream velocities tend to result in higher deposition fractions of inhaled particulates in infants and children than in adults at similar exposure levels (James et al., 1994; Phalen et al., 1985; Xu and Yu, 1986).

Human exposures to air-borne metals are usually to metal-bearing particulates, which necessitates measurements of particle sizes in the breathing zone of receptors of concern to achieve accurate estimates of deposition rates in the respiratory tract. Exceptions to this are exposures to mercury vapor or other gaseous forms of metals, such as arsine gas. In most applications, monitoring of metals for use in environmental risk assessment consists of measuring total metal (i.e., unspciated) captured in the PM₁₀ fraction (the fraction of particles that are less than or equal to 10 microns in diameter) of samples collected in stationary samplers. This approach introduces three uncertainties into risk estimates that are based on these assumptions. Air concentrations measured at stationary samplers may not accurately reflect air exposures to receptors of concern (Azar et al., 1975). Consider, for example, a scenario at a lead mining or smelter site in which human receptors drive off-road vehicles (e.g., dirt bikes) over dust-laden waste piles. Stationary monitors almost certainly do not accurately capture levels of exposure to air-borne metal-bearing particulates from such activities. While these exposures may

be intermittent, the magnitude of the exposures could make an appreciable contribution to metal body burden and related health risk.

An additional consideration is that, although a PM₁₀ sample captures the bulk of the respirable particulates (i.e., those of such size that they would not be removed from the air stream in the nasopharyngeal region of the respiratory tract), deposition patterns—and thus the fate of inhaled particulates deposited in the respiratory tract—are substantially affected by particle size, below a diameter of 10 microns (James et al., 1994). For example, a substantial fraction of the inhaled particles larger than 1 micron can be expected to be deposited in the upper respiratory tract and subsequently transferred by mucociliary transport to the gastrointestinal tract, where fractional absorption may be very much different from that of particles absorbed from the respiratory tract. Measurement of unspiciated metals introduces additional uncertainties into risk estimates for inhaled metals because the physiological solubility of a metal affects the mechanisms, rate, and extent of absorption of metals from the respiratory tract (Oberdoerster, 1992; Bailey and Roy, 1994). Lack of information about the particle sizes, chemical form, and solubility of the air-borne metals in the breathing zone of receptors of concern can have important implications for accurate modeling of rates of absorption, internal dose of inhaled metal particulates, and risk (Khoury and Diamond, 2003).

2.3.3 Surface Dust and Soil Pathway

Surface dusts and soil are particularly important media of human contact with metals because both can serve as repositories for air-borne metal particulates. As a result, exposures through the surface dust or soil pathways can occur long after cessation of emissions of metals to air. Humans can be exposed to metals in surface dust and soil as a result of ingestion or dermal contact with soil and dust, and by inhalation of suspended dust particles. Infants and children are particularly vulnerable to exposures to metals through the surface dust pathway because of their crawling and play activity in close proximity to surface dust, and because they often mouth their hands (e.g., finger sucking) and objects in their environment; this results in higher intakes of surface dust than in adults (e.g., Barnes, 1990). Children also absorb a larger fraction of the ingested dose of some metals (e.g., lead) than do adults (Mushak, 1991). This conclusion is consistent with evidence for a relatively strong dependence of blood lead concentrations in children on interior dust lead and/or hand wipe lead levels (Lanphear et al., 1998; Succop et al., 1998). The importance of the dust ingestion pathway is not limited to metals; however, because this pathway is often a major contributor to risk at metal-contaminated sites, uncertainties in estimates of surface dust ingestion can make a major contribution to uncertainties in exposure and risk estimates (Griffin et al., 1999). Several studies have attempted to estimate the amount of soil ingested by children and adults (e.g., Binder et al., 1986; Calabrese et al., 1989, 1997; Clausen et al., 1987; Davis et al., 1990; see U.S. EPA, 2002a, for review). The amount of soil ingested by children can be expected to vary with numerous factors, including age, activity patterns, and the accessibility to soil and dust. Estimates for children 2 to 6 years of age range from 39 milligrams per day to 271 milligrams per day; however, considerable uncertainty is associated with estimates derived from each study. Current estimates have been based on short-term measurements (e.g., days) which cannot be expected to capture intra-individual variability

and may not reflect long-term trends. Furthermore, these estimates do not represent pica behavior, in which much higher amounts of soil may be ingested (Barltrop, 1966; Calabrese and Stanek, 1993). Studies conducted thus far do not distinguish between the quantity of dust ingested and the quantity of soil ingested. This parameter is important in connecting measured soil metal concentrations with surface dust ingestion that occurs in the indoor and outdoor environment (U.S. EPA, 1994). Few studies of soil ingestion in adults have been conducted; however, the estimates support the general assumption that average daily soil ingestion rates of adults who do not participate in activities in which intensive exposure to surface dust and soil occur (e.g., occupational gardening, construction work) are lower than for children (Calabrese et al., 1990; Hawley, 1985).

Although dermal contact with metals in soil represents a potential route of exposure, the relatively low lipid solubility of most metals is generally thought to limit absorption through the skin (Hostynek et al., 1998; Paustenbach, 2000) far more than for more lipid-soluble organic chemicals (an exception being Hg^0). However, as previously noted, few studies have actually attempted to quantify the extent or kinetics of dermal penetration of metals deposited on the skin, and the applicability to these studies to metal species and complexes that occur in surface dust or soil is highly uncertain.

Metals in the soil surface do not tend to migrate into subsurface soils unless the metal retention capacity of the soil is overloaded or metal interaction with other contaminants enhances mobility (U.S. EPA, 1992). The migration of metals through the soil is affected by the chemical form of the metals, which affects solubility. Metal migration is highly dependent on soil properties. Thus, the concentrations of the metal contaminants in soil can be expected to vary with depth, as can the size of metal-bearing particles. For example, higher concentrations of lead and smaller particles are found near the soil surface (Duggan and Inskip, 1985; Duggan et al., 1985; Fergusson and Ryan, 1984; see Chaney et al., 1988, for review). Bioavailability of lead, as well as other chemicals, increases with decreasing particle size (Barltrop and Meek, 1979). One must consider these two variables, concentration and bioavailability, in interpreting measurements of metal concentrations obtained from soil samples at different depths. The depth of soil that an individual may be exposed to will depend on the individual's activities. In the case of inadvertent ingestion of surface dust, it is reasonable to assume that children may be exposed to the surface soils and the primary contact medium of concern is hand dust. In this case, core samples of the top 6 inches of soil may substantially underestimate the actual concentration in the primary medium of contact. This may not be the case for a home gardener who may be exposed to subsurface soil. The effects of particle size on metal concentrations and bioavailability also has implications for sample preparation, which often includes sieving of the soil sample to remove larger particles.

2.3.4 Food Pathway

Food can be a major contributor to human metal exposures and associated health risks. Failure to accurately account for the dietary contribution can result in significant errors in exposure and risk estimates for metals for which diet makes a major contribution to exposure (Choudhury et al., 2001). Human dietary exposures to metals may occur from various processes. Deposition of metals from ambient air may result in contamination of food crops. Animals grazing in areas where metal contamination exists can also be affected by the uptake of metals from soils or surface water. Metals can also migrate into surface water and sediments, and be taken up by aquatic species that may be consumed by humans. Human food crops can also take up metals from soils and surface water. Crops can also be contaminated with metals through direct deposition of airborne particles. Exposure from this pathway is affected by food handling practices, such as washing of fruits and vegetables; these practices can be expected to have a substantial local variability, and data in support of quantitative estimates are limited. Metals can also enter food during harvesting or processing of produce and livestock, during food storage from metals in food containers, and during preparation of foods for meals.

Estimation of intakes of metals in food requires information on the levels of the metal in food and the amount of food consumed. Although large-scale surveys of metals contents of foods and food consumption patterns have been conducted (e.g., U.S. DHHS, 1996; Egan et al., 2002; O'Rourke et al., 1999; Ryan et al., 2001; Thomas et al., 1999; U.S. FDA, 2000), these surveys have severe limitations, noted below, for applications to risk assessment. With few exceptions, such applications have not been empirically evaluated against biomarkers of exposure (Clayton et al., 1999, 2002; Choudhury et al., 2001). In general, the food consumption surveys estimate short-term consumption (e.g., 1 to 3 days) and do not capture intra-individual variability that would affect long-term averages. Furthermore, food consumption patterns can be expected to change over time; thus, patterns discerned at any given time may not accurately represent historical exposures. This becomes important when one is reconstructing doses and risks associated with exposures to metals that have a long residence time in the body (e.g., lead and cadmium). An additional challenge is the integration of data from separate metal residue surveys and food consumption surveys (e.g., Tomerlin et al., 1997). Estimates of dietary intakes of metals in food based on national or regional data cannot be expected to accurately reflect intakes of metals in locally harvested foods, including produce, fish and game. This can be a particularly important limitation when the receptors of concern are subsistence fishermen or hunters. These surveys may also not reflect metals that enter the food pathway during local food preparation or storage (e.g., in the home). Such local contributions of metals to food can be assessed with local food consumption/residue surveys, for example measurements of metals in foods as prepared in the home (e.g., duplicate diet studies).

Theoretically, risk from the dietary pathway could be estimated based on total exposures or based on the increment in exposure above a background level that would be expected in the absence of the exposure source of concern. Where it is possible to accurately determine a background exposure level, metal intakes (i.e., dose) can be apportioned into background and other source categories. However, this will be useful for similarly apportioning risk only if a

dose-response relationship for the health endpoint of concern has been derived that encompasses the dose expected from background exposures. In most cases, the EPA expresses dose-response relationships for non-cancer endpoints as an uncertainty bound on a No Observed Adverse Effect Level (NOAEL), which by itself provides no information about the slope of the dose-response relationship or, therefore, the incremental change in risk expected for an incremental increase in exposure above the background level. Thus, for most metals, risk-based decision making can be more reliably based on estimates of total, rather than incremental, exposures.

2.3.5 Water Pathway

The major pathways for human intakes of metals in which water serves as the primary medium of contact are ingestion, dermal, and inhalation. As previously noted, information on the extent or kinetics of dermal penetration of metals deposited on the skin is, in general, inadequate to support risk assessments of this pathway. Inhalation of inorganic metal contaminants in water can result from aerosolization, or in special cases from volatilization (e.g., Hg⁰). The aerosolization pathway can be a major source of intake of inorganics under certain conditions. For example, sea spray can be a significant contributor to iodide intakes in populations that live near the seashore (Whitehead, 1984). A pathway more typical of human exposure to contaminated water is showering, in which aerosolization can occur at the water tap. Although models have been developed to predict human inhalation exposures to volatile organics from showering (e.g., McKone, 1987; Moya et al., 1999; see Guo, 2002, for review), comparable models do not exist for aerosolized metals (Wilkes, 1998).

The ingestion pathway is generally considered to be the dominant contributor to humans' intakes of inorganic metal species in water. Unique to humans, among to ecological receptors, is exposure to and consumption of water specifically treated for human consumption (i.e., drinking water). Treatment of ambient water for human consumption can change the speciation of some metals, e.g., chromium (III) to chromium (VI). It also removes dissolved organic carbon and suspended organic sediments that can form complexes with metals; thus, the exposure context of metals in human drinking water is very different from that of ambient water. Of greatest importance is that inorganic forms of metals in drinking water consist of the more bioavailable, water-soluble species. Treatment also removes bacteria that can participate in organification reactions of toxicologic significance to humans (e.g., methylation of inorganic mercuric mercury). Metal contaminants can enter drinking water at various stages in the treatment and delivery of the water to human receptors, including from distribution systems within homes and, in the case of lead, from glassware and water storage containers (Graziano et al., 1996). Thus, exposure estimates based on measurements in municipal water supplies may not accurately represent exposures at the location where the drinking water is consumed.

Humans can be exposed to metals dissolved in ambient surface water or in association with suspended sediments; the latter can serve as a long-term repository for water-borne metal particulates. Such exposures can occur during swimming or other recreational activities, or during occupational activities in which the sediments are disturbed or resuspended in the water column. Children can be particularly vulnerable to exposure to sediment-borne metals as a result

of swimming or play activities at or near the shoreline. The chemical species in ambient surface water can be expected to be much more diverse than in treated drinking water or in ground water, because of the presence of organic carbon, inorganics, and suspended organic material that can serve as ligands or reactants for metals. Speciation will also vary with pH of the surface water.

2.3.6 Transplacental Pathway

Maternal exposure can be important for establishing the body burden of certain metals prior to birth, as a result of transplacental transfer, or during infancy, through breast milk. For metals that have long retention times in tissues, the maternal tissues can serve as a reservoir for exposures during fetal development or infancy, long after cessation of maternal exposure. This can be a particularly important exposure pathway for metals that accumulate in the inorganic matrix of bone (e.g., lead, strontium, uranium), as mobilization of bone minerals to develop the fetal skeleton can result in a transfer of maternal bone stores of metals to the fetus (e.g., Gulson et al., 1999a, 1999b; Tolstykh et al., 1998). Transplacental exposures cannot be directly estimated from environmental measurements, and such assessments require the use of pharmacokinetics models. A few models of transplacental transfer of lead in humans have been developed; models for other metals are not available for use in risk assessment. The lead models reported to date rely on assumptions of a steady state between maternal and fetal blood lead concentrations (U.S. EPA, 1994, 1996; Leggett, 1993; O'Flaherty, 1993).

2.3.7 Exposure Estimation Tools

Several models for simulating exposures to specific metals are available for use in risk assessment. The U.S. EPA Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK model; see U.S. EPA, 1994; White et al., 1998) was specifically developed for translating exposure measurements into risk estimates at sites contaminated with lead. Several studies offer comparisons between model predictions and observed relationships between blood lead concentrations and exposure levels (Hogan et al., 1998; Bowers and Mattuck, 2001). A fully stochastic version of the exposure model in the IEUBK model that implements one- and two-dimensional Monte Carlo simulations of lead exposure is also available, and has had limited application in uncertainty analysis at lead-contaminated sites (Goodrum et al., 1996; Griffin et al., 1999). A stochastic human exposure model for lead, linked to a lead pharmacokinetics model (O'Flaherty, 1993), has also been reported (Beck et al., 2001). Less complex models linking adult exposures to lead and blood lead concentrations are also available (Bowers et al., 1994; Carlisle and Wade, 1992; Carlisle et al., 2000; Stern, 1996, 1997; U.S. EPA, 1996); the U.S. EPA approach (1996) has had extensive application at lead-contaminated sites. An exposure model for arsenic has also been reported (Cohen et al., 1998). A more generic exposure model, RESRAD, was developed by the U.S. DOE for risk assessment of radionuclides (U.S. DOE, 2001; LePoire et al., 2000). Although this model was developed for deriving surface radionuclide soil action levels, the extensive human exposure module is applicable to other metal contaminants. The model also includes functionality for Monte Carlo simulation. Several efforts are underway to develop stochastic human exposure models that, while not specifically intended

for assessing exposures to metals, would be applicable to such endeavors; these include the U.S. EPA Stochastic Human Exposure and Dose Simulation (SHEDS) model and Lifeline™. The U.S. EPA has developed a Dietary Exposure Potential Model (DEPM) that links national food consumption and chemical residue data to allow estimates of average dietary intakes of metals and other food contaminants (Tomerlin et al., 1997).

2.3.8 Mixtures

Human exposures to environments contaminated with metals often involve exposures to mixtures of metals. Interactions between metals can alter the absorption, toxicokinetics, and toxicodynamics of metals (ATSDR 2001a, 2001b; Diamond et al., 1998; Mushak, 1991; Reeves and Chaney, 2002; Stillman and Presta, 2000). The potential for such interactions should be considered in risk estimation. To do so requires that exposures to metals having a high potential for interactions be estimated. A methodology is also needed for gathering, evaluating, and integrating information on interaction potential into risk estimates (ATSDR 2001a, 2001b; Durkin et al., 1995; Mumtaz and Durkin, 1992; Mumtaz et al., 1994). Toxicokinetic and toxicodynamic aspects of metal interactions, as they relate to human health effects, are discussed further in the Issue Paper devoted to human health effects.

2.3.9 What To Do When Information Is Limited

The absence of information needed to fully characterize human exposures to metals introduces uncertainty into exposure and risk estimates. To the extent possible, sources of uncertainty need to be identified and described; when possible, their impacts on exposure and risk estimates must be quantified (U.S. EPA, 2001c). The latter requires mathematical models that can reliably simulate the important parameters that affect human exposures. Monte Carlo simulation and other techniques can then be applied to explore the quantitative impacts of variability and uncertainty in exposure and risk estimates (e.g., Beck et al., 2001; Clewell et al., 1999; Griffin et al., 1999; Stern, 1997).

3. APPLICATION TO A REGULATORY CONTEXT

3.1 Aquatic Receptors

Exposure assessments can take advantage of the tools described in Section 2.1, including the different expressions of exposure concentrations, the methods for adjusting for differences in laboratory test and field exposure durations, N isotope methods to define trophic structure, prediction and metal grouping based on HSAB theory, and alternate methods of handling exposures to metal mixtures. Equilibrium-based methods are generally applicable, but kinetics should be considered, as described above.

3.2 Terrestrial Receptors

Exposure-related information can be developed at various levels of sophistication and detail. Most risk-based approaches begin with some form of screening-level assessment. This might be used for different purposes in different programs but the approach is generally the same. For metals in terrestrial systems, the methods applied in the development of the ECO SSLs are probably the best and most well-thought-out that are available for invertebrates, plants, and wildlife. (Note that the ECO SSLs effort did not establish methods for microbes, amphibians, or reptiles.) The scientific consensus is that there is sufficient uncertainty about the nature of exposure that, while the ECO SSLs are intended to be useful as screening measures, they should not be considered as accurate indicators of actual exposure in field conditions. There are simply too many factors that can influence exposure. As a result, screening levels should be used as a starting place or as a convenient guide for site management when other factors are driving decisions. However, if the decisions are strongly dependent on the actual risk associated with metals in soils, case-specific risk assessments should be undertaken.

3.3 Human Health Receptors

Exposure assessments provide the crucial linkage between dose-response analysis for metals and risk estimates. Exposure assessment tools are used in the regulatory context to support environmental decision-making at the national and local (e.g., site-specific) levels. For example, exposure assessments are used to set cleanup levels at sites where soil contamination exists. They are often used to support regulations regarding point emission sources, safety of consumer products, or use of a pesticide. Exposure assessments usually vary in scope and level of detail, to match the decision-making demands. The issues raised in Section 2.3 pertain to uncertainties associated with both national-level and site-specific exposure assessments.

4. NATURAL AND BACKGROUND METAL CONCENTRATION

4.1 Introduction

Metals are naturally occurring elements in the earth's crust. Over the geological time scale, biogeochemical processes have resulted in a steady-state cycling of metals within the biosphere in which the speciation, distribution, and effects of each metal are fairly well regulated. Recent technological developments have resulted in the budgets of natural cycles of most elements now being dominated by industrial inputs to many ecosystems. Environmental metal pollution is a global phenomenon that has significantly changed the natural concentrations of many trace metals in various environmental media, with long-range transport processes now delivering industry-derived metals to the most remote parts of the world. For the atmosphere, water, and to a limited extent vegetation and soils, current baseline levels may be far removed from the true natural background concentrations, as illustrated by the data for lead (Table 3). During the early 1970s, for instance, industrial sources accounted for over 90% of the airborne lead, the deposition of which supplied a large fraction of the lead economy of many ecosystems (Nriagu and Pacyna, 1988). The notion that current baseline levels are the same as the naturally

occurring concentrations permeates both the scientific literature and current policy debate on framework for metal assessment.

Table 3. Estimated Global Averages of Natural Levels and Typical Present-Day Levels of Lead in the Environment (modified from NAS, 1980)

Medium	Estimated Natural Level	Typical Levels in Present-Day Environment
Air		
Rural/remote	0.01–0.1 ng/m ³	0.1–100 ng/m ³
Inhabited		10–500 ng/m ³
Soil		
Rural/remote	5–25 µg/g	5–50 µg/g
Inhabited		10–1,000 µg/g
Water		
Fresh	0.5–10 ng/L	2–50 ng/L
Ocean	0.05–2 ng/L	1–15 ng/L
Foods		
	5–50 ng/kg	10–200 ng/kg

Everybody in the United States is exposed to a wide variety of metals in air, food, drinking water, and soils derived from both natural and anthropogenic sources. The distinction between natural and anthropogenic sources can sometimes be further blurred when a previously deposited metal is recycled by natural processes, thereby becoming an important source in some ecosystems. It has been reported, for instance, that the recycling phenomenon accounts for about a third of the annual flux of mercury in the global ecosystems (U.S. EPA, 1997b). The general magnitude of exposures from natural versus industrial sources for individuals is uncertain because the levels of metals in environmental media from either source are highly variable and not well documented. The data for lead in Table 3 are order-of-magnitude approximations to be revised as our knowledge improves. However, the data suggest that current relative ratios of anthropogenic to natural concentrations for lead are in this order: air > crops > water > soil. The greatest increase in concentrations above natural levels has occurred in the two environmental media that contribute most to our current lead exposures. Levels of lead in soils are expected to have increased slightly, if at all, but this pool contributes much less to direct human exposure in typical adults. Recent studies suggest that human-induced changes in relative concentrations of mercury, cadmium, and to a lesser extent other toxic metals in the environmental media parallel those for lead. For many metals, present-day (baseline) exposures are expected to be above the natural background doses and the biological implications of the historical increase in levels of human exposure are unknown. Speculations that sensitive species in some ecosystems may be impaired in undetected ways by present-day baseline levels of toxic metals (Patterson, 1980)

cannot be dismissed without further research to determine the true natural background metal concentrations in the country's ecosystems. A likely collateral effect of background contribution to human metal intake would be to push the total exposure over the toxicity threshold.

Equating current baseline concentrations with natural background levels of toxic metals can result in underestimation of risk in some locations. In terms of policy relevance, this potential error needs to be addressed in national regulatory assessments; in site-specific assessment of badly contaminated area, the effect of historical changes in baseline levels should be of less concern. A metal emitted from industrial sources may have distinctive chemical features (such as redox state) or isotopic composition which can be used to ascertain the relative contribution of such sources to a particular exposure pathway. Most often, there are no distinguishing features between metals from natural and industrial sources and indirect methods have to be used to estimate the relative risks of exposures to the naturally and anthropogenically derived metals. These "indirect methods" tend to be ad hoc with limited scientific support.

Oxidative weathering of minerals in rock formations can lead to exaggerated concentrations of elements that readily form oxy-anions (especially selenium, arsenic, molybdenum, antimony, vanadium, and chromium) in natural waters. For instance, natural sources have resulted in arsenic and selenium concentrations in groundwater and surface water in some parts the country that exceed established toxicity thresholds. The unique tendency of the oxy-anions of many elements derived from natural sources to accumulate in water supplies should be recognized in any regulatory framework on background metal exposures.

Every environmental action begins with the simple question: Is the site contaminated? For metals that occur naturally in the earth's crust, this question cannot be answered without adequate accounting for the natural occurrence and background exposures to the metals. Treatment of background exposures to metals in current frameworks for national level assessments is neither systematic nor scientifically rigorous. All too often, spot samples are said to reflect pollution in particular ecosystems even though the natural levels have not been considered. On the other hand, little attention is paid to the fact that some metals (such as mercury) are disproportionately deposited in some parts of the country (enough to create area-wide risks, for mercury as an example), and limited attempt is made to differentiate between the effects of natural versus perturbed (in terms of metal inputs) ecological cycles on metal exposures. In view of the importance in long-term policy choices, there is increasing need to establish the concentration ranges of specific metals to be regarded as natural in particular environmental media in various parts of the country, i.e., the "background values" for the metals.

4.2 Site-Specific Assessment of Baseline Exposures

Risk assessment and risk management under the enabling Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) have used various strategies to estimate the baseline metal levels at the Superfund sites (U.S. EPA, 1989, 1991, 1995a, 1995b). A number of recent documents dealing with background issues at CERCLA sites have been published by the U.S. EPA (2001a, 2001b, 2002b, 2002c, 2002d). The focus on metals in these documents is the characterization of baseline levels in soils, and guidance manuals have been

prepared on sampling and statistical analysis of baseline levels of metals in soils at CERCLA sites (U.S. EPA, 2001b, 2002d). Data on typical baseline levels of several metals in every state have recently been compiled (Table 4); they can be used in screening level analysis, for example, at CERCLA sites. Soils, however, are not generally a primary exposure route for metals, especially for human adults. The guidance for site-specific evaluation of baseline levels of metals in soils is inapplicable to non-soil media, which tend to be more dynamic and are more likely to be influenced by upstream and distal sources. Assessment of exposures from non-soil pathways requires more complex spatial and temporal sampling strategies, analysis of releases and transport, and different ways of combining and analyzing data (U.S. EPA, 2002d). The use of baseline reference locations, a critical element in soil assessment, may also be inappropriate. A systematic strategy for evaluating site-specific baseline exposure from non-soil sources should go a long way towards improving the risk estimate for metals.

5. RESEARCH NEEDS

5.1 Aquatic Receptors

The most pressing research needs involve defining the best ways of expressing exposure concentration, accommodating differences in exposure durations, and combining exposure concentrations when exposure involves metal mixtures. An aggressive and focused effort is needed to assess the relative merits of different methods used to express exposure concentrations in sediments and suspended solids. When unacceptable uncertainty associated with using the durations of standard laboratory tests warrants it, survival time methods or other extrapolation methods can and should be applied. More careful comparison of the simple extrapolation methods described herein to richer survival or time-to-event models is essential. Without such comparisons, an unacceptable level of uncertainty may persist in assessments. Finally, the application of the hazard index approach to metal mixtures should be carefully scrutinized and alternate methods explored more frequently.

5.2 Terrestrial Receptors

Several generalizations are currently applied to evaluating wildlife exposure (bulleted in Section 2.2.1) that introduce error if applied to specific scenarios. Further work is needed to define or reduce the associated uncertainty of using such generalizations. Further reductions in uncertainty could be obtained if better means of producing laboratory test data for soil systems that better reflected the actual forms of metals in field soils. Finally, there is a paucity of information for terrestrial receptors with which to assess the joint effect of metals in mixtures.

Table 4. Mean Baseline Metal Concentrations in Soils (µg/g) by State (from U.S. EPA, 2002d)

State	Al	As	Ba	Be	Cd	Co	Cr	Cu	Mn	Ni	Pb	Se	V	Zn
Alabama	23100.0	4.7	200.0	0.6		4.4	30.6	9.6	420.0	11.0	9.3	0.3	38.0	26.0
Arkansas	33429.0	9.7	336.0	0.9		12.0	53.1	17.0	731.0	18.0	21.0	0.7	52.0	39.0
Arizona	32933.0	9.6	364.0	1.0	0.4	9.9	37.3	23.0	447.0	23.0	16.0	0.4	42.0	51.0
California	75633.0	5.1	598.0	1.1	0.4	14.0	119.9	39.0	640.0	48.0	26.0	0.2	118.0	113.0
Colorado	61557.0	6.7	662.0	1.4		6.8	41.7	21.0	343.0	13.0	31.0	0.4	74.0	87.0
Connecticut	85000.0	4.1	400.0	0.5		7.5	40.0	15.0	450.0	13.0	5.0	0.8	60.0	40.0
Deleware	22500.0	1.4	400.0	0.5		3.3	30.0	5.0	85.0	6.0	15.0	0.3	20.0	23.0
Florida	9944.0	3.0	48.0	0.6	0.1	1.6	15.4	5.6	86.0	8.5	12.0	0.3	11.0	12.0
Georgia	38250.0	5.0	232.0	0.6		6.9	32.4	21.0	252.0	17.0	19.0	0.4	43.0	47.0
Iowa	64667.0	7.3	617.0	1.3		11.0	64.7	31.0	603.0	26.0	19.0	0.4	97.0	57.0
Idaho	58500.0	6.4	757.0	1.1		12.0	52.1	28.0	580.0	22.0	22.0	0.3	90.0	83.0
Illinois	48714.0	7.1	551.0	0.7		9.8	48.4	24.0	646.0	19.0	39.0	0.5	62.0	67.0
Indiana	50000.0	7.5	500.0	0.7		10.0	46.8	27.0	518.0	18.0	18.0	0.4	74.0	56.0
Kansas	61818.0	6.8	694.0	1.0		8.9	49.0	25.0	452.0	17.0	32.0	0.4	77.0	67.0
Kentucky	54123.0	7.8	349.0	1.1		11.0	79.8	17.0	483.0	23.0	16.0	0.5	66.0	35.0
Louisiana	42188.0	7.6	441.0	0.6		8.6	60.8	33.0	470.0	33.0	16.0	0.7	76.0	55.0
Massachusetts	34083.0	8.6	203.0	1.3	0.2	7.8	39.5	16.0	439.0	13.0	13.0	1.9	87.0	54.0
Maryland	39167.0	3.8	393.0	1.3		7.5	47.9	20.0	291.0	13.0	22.0	0.2	63.0	39.0
Maine	65385.0	9.4	319.0	1.6		10.0	71.2	28.0	581.0	30.0	19.0	0.7	98.0	80.0
Michigan	10964.0	4.2	127.0	0.7	0.9	4.6	13.8	12.0	230.0	12.0	9.2	0.3	44.0	33.0
Minnesota	49457.0	5.5	571.0	0.7	0.3	7.2	25.4	20.0	583.0	14.0	9.9	0.3	72.0	38.0
Missouri	42094.0	10.0	499.0	1.0		12.0	50.0	19.0	940.0	20.0	23.0	0.5	72.0	53.0
Mississippi	45368.0	8.8	390.0	0.9		12.0	53.2	20.0	471.0	21.0	18.0	0.5	68.0	45.0
Montana	70938.0	8.8	739.0	1.1		7.5	63.3	29.0	366.0	20.0	14.0	0.4	101.0	69.0
Nebraska	59474.0	5.5	711.0	1.1		5.9	32.5	15.0	306.0	15.0	16.0	0.4	62.0	54.0
North Carolina	60105.0	4.8	356.0	0.6		15.0	64.8	34.0	563.0	24.0	17.0	0.4	107.0	56.0
North Dakota	62857.0	7.0	682.0	0.9		6.9	53.2	23.0	530.0	20.0	13.0	0.4	83.0	64.0
New Hampshire	66667.0	4.4	500.0	2.3	0.6	5.3	18.4	12.0	633.0	10.0	28.0	0.3	57.0	23.0
New Jersey	10075.0	7.0	54.0	0.3	0.3	1.7	13.9	14.0	221.0	3.8	35.0	0.9	30.0	22.0
New Mexico	54423.0	5.9	727.0	1.0		8.8	55.5	21.0	367.0	28.0	18.0	0.3	72.0	44.0
Nevada	66078.0	9.0	822.0	1.3		8.4	36.8	25.0	481.0	15.0	25.0	0.3	78.0	69.0
New York	58800.0	6.4	666.0	1.4	0.2	9.1	66.9	36.0	418.0	21.0	20.0	0.3	132.0	82.0
Ohio	54615.0	12.0	469.0	1.0		13.0	55.0	28.0	550.0	25.0	23.0	0.6	88.0	69.0
Oklahoma	39200.0	7.0	430.0	1.1		7.1	46.0	16.0	465.0	15.0	18.0	0.3	50.0	50.0
Oregon	94412.0	5.1	682.0	0.9		16.0	121.6	53.0	725.0	23.0	15.0	0.3	168.0	70.0
Pennyslvania	63438.0	13.0	366.0	1.4		15.0	52.8	37.0	609.0	24.0	23.0	0.5	80.0	81.0
Rhode Island	100000	3.5	500.0	0.5		10.0	50.0	15.0	500.0	15.0	15.0	0.9	70.0	30.0
South Carolina	39143.0	3.9	151.0	1.4		3.5	21.4	16.0	87.1	7.8	5.0	0.3	45.0	25.0
South Dakota	74333.0	8.5	1043.0	1.4		7.7	58.7	29.0	1013.0	28.0	16.0	0.5	108.0	75.0
Tennessee	31894.0	16.0	193.0	0.8	0.2	14.0	40.3	17.0	1112.0	18.0	23.0	0.6	49.0	57.0
Texas	41958.0	6.4	404.0	0.9		5.3	39.6	15.0	303.0	12.0	14.0	0.3	52.0	39.0
Utah	45638.0	8.0	493.0	0.9		6.6	45.6	26.0	371.0	13.0	35.0	0.3	70.0	96.0
Virginia	60438.0	5.1	436.0	0.9		9.7	54.3	33.0	441.0	17.0	36.0	0.4	77.0	233.0
Vermont	56667.0	3.6	333.0	1.7		12.0	66.7	18.0	800.0	25.0	20.0	0.4	70.0	43.0
Washington	66834.0	4.5	606.0	0.9	0.8	18.0	49.9	31.0	760.0	23.0	14.0	0.3	160.0	78.0
Wisconsin	48000.0	4.4	543.0	2.0		7.7	40.3	12.0	365.0	14.0	12.0	0.3	48.0	44.0
West Virginia	67000.0	8.6	360.0	1.0		14.0	46.0	22.0	770.0	23.0	17.0	0.5	65.0	60.0
Wyoming	56125.0	6.5	756.0	0.7		8.3	47.9	21.0	416.0	16.0	17.0	0.5	84.0	57.0

5.3 Human Health Receptors

Categories of uncertainty that affect exposure assessments include lack of or insufficient data, measurement error, and uncertainties in models used to integrate data into estimates of metal intakes. While uncertainties related to lack of data can sometimes be rectified by collection of additional data, in general, the latter two categories of uncertainty, measurement error and model uncertainty, can be reduced only with research to improve sampling and measurement approaches and models.

Major sources of uncertainty encountered in risk estimates of metals that could be affected by additional research to improve measurement capabilities or exposure models include 1) uncertainty regarding the chemical and physical forms of metals in the primary media of exposure; 2) relatively large uncertainties about rates of soil and surface dust ingestion, including estimates of central tendencies, both short-term and long-term, inter- and intra-individual variability (e.g., within age- and across ages), and relative contributions of surface dust and soil; 3) insufficient information about the types and frequencies of activities that place children in contact with contaminated soils, dusts, or surfaces (e.g., hand-to-mouth behavior, rates of contact with surfaces); 4) uncertainties in models used to predict concentrations in surface dust, a primary medium of contact, from measurements made in surface soil samples, surface dust wipe samples, and surface dust vacuum samples; 5) uncertainty in estimates of dietary intakes of metals; and 6) uncertainties in models used to predict the contribution of locally harvested foods to metal intakes (e.g., uptake of metals from soil, intakes of home-grown or home-harvested foods).

6. CONCLUSIONS

Specific conclusions can be found throughout this paper, but several general conclusions can be highlighted here. The source influences the realized dose by determining the amount of metal, exposure route, species and phase association, and, ultimately, the metal bioavailability. Metal speciation, phase association, and bioavailability can be explored for diverse metals using HSAB theory.

Aquatic receptors can be exposed via inhalation/respiration (i.e., exchange across respiratory surfaces), absorption, or ingestion. Exposure assessment is difficult and prone to high uncertainty because of the diversity of relevant aquatic receptors and their divergent respiratory and feeding modes. Most laboratory-derived exposures involve a set duration and speciation/phase-association different from field exposures. Most of our current data and methods handle metal mixtures poorly. Methods are described in Section 2.1 for reducing the associated uncertainty. For dissolved metals, these involve equilibrium speciation calculation of dissolved species and application of the FIAM, BLM, and QICAR models. For metals in solid phases, a variety of extraction methods are available but their relative utilities remain poorly defined.

Terrestrial receptors can be exposed by these same routes. Some generalizations can be made with caution. They include the following: 1) chemicals for which uptake into food (either

plants or soil invertebrates) tends to be the most important route of exposure include cadmium, copper, lead, nickel, selenium, and zinc; 2) chemicals for which incidental soil ingestion (and some food uptake) tends to be the most important pathway include aluminum, iron, and vanadium; 3) incidental soil ingestion is a proportionally more important pathway for herbivores; and 4) uptake into soil invertebrates (e.g., earthworms) is a proportionally more important pathway for animals that feed on these organisms.

Human receptors can be exposed by ingestion, absorption, inhalation and transplacental transport. The primary media of contact are inhalation and dermal when exposure involves metal for air. Surface dusts and soil are particularly important media of contact because they are the media eventually receiving air-borne metals. The soil/dust ingestion pathway is particularly important in exposures to children. Food can also be a major contributor to human exposure: the water pathway can be important because humans consume water that is conditioned (i.e., by removal of chelating organic compounds and modifying quality) prior to consumption. Maternal transfer of metals via the placenta or milk can increase fetal or infant exposure, particularly for metals incorporated into bone, i.e., lead, strontium, and uranium.

Realistically, the predominant metal exposure route of ecological risk concern is ingestion for higher-order terrestrial receptors, and in many cases for aquatic receptors. The respiratory/inhalation route is often more important for aquatic receptors and less important for terrestrial receptors. The absorption route is often minimal due to the often effective barrier that most external epithelia provide to organisms. A potential exception might be that of plants whose external root tips and associated root micro-environments resemble the intestinal microvilli. The soil matrix often exposes these systems to metal solutions.

Metal exposure assessments are complicated by natural and background metal concentrations. The false notion that current baseline levels are the same as the naturally occurring concentrations befuddles the assessment literature and current policy.

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