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The Basic Science of Poisons

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soil and water through fallout during smelting. Natural soil concentrations of cadmium are less than 1 ppm and average about 0.4 ppm. Sewage sludge is often contaminated with cadmium, which then concentrates in plants grown on contaminated soils. The problem of heavy-metal contamination, especially cadmium, has been one of the most serious concerns impeding the use and disposal of domestic sewage sludge on agricultural lands. Cadmium also enters agricultural soils as a contaminant of phosphate fertilizers. There is some evidence for the leaching of cadmium in soils.

Among the metals, cadmium is one of the most readily absorbed and accumulated in plants grown on contaminated soil. The significance of this phenomenon is readily apparent in the relationship of cadmium concentrations in rice with the incidence of itai-itai disease. Although there is some question about the specific etiology of this disease, there is ample evidence from many studies of a positive correlation between rice cadmium content and the incidence of the disease in Japan (see, for example, Nogawa *et al.*, 1982, 1983).

Cadmium concentrations in fresh waters are usually less than 1 ppb, while concentrations in seawater range from 0.05 to 0.2 ppb and average about 0.15 ppb (Fleischer *et al.*, 1974). Higher concentrations of cadmium in surface water are usually due to metallurgical plants, plating operations, cadmium pigments, batteries, plastics manufacture, or from sewage effluent. Mine drainage and mineralized areas also contribute significantly to cadmium fluxes in the Mississippi River in the Missouri-Tennessee-Kentucky area.

Drinking water in soft-water areas can serve as a source of cadmium through corrosion of plumbing. However, this source is estimated to be small in relation to food intake. As in the association of selenium and mercury, there appears to be a protective effect with zinc and calcium against cadmium toxicity.

Lead. The use of lead, its mining, and its processing date back several centuries. Changing usage patterns rather than increased consumption determine present environmental inputs from man's use of lead. Batteries, gasoline additives, and paint pigments are major uses, but combustion of gasoline additives is the major source of environmental pollution by lead. Thus, lead is primarily an atmospheric pollutant that enters soil and water as fallout, a process determined by physical form and particle size. The net result is a buildup of lead near heavily traveled roads. In a study in Massachusetts, there was a close concordance between the use of leaded gasoline and umbilical cord blood lead levels. This suggests that further reductions in leaded gasoline usage would result in lower human exposure levels (Rabinowitz and Needleham, 1983).

Lead enters aquatic systems from runoff or as fallout of insoluble precipitates and is found in sediments. Typical fresh-water concentrations lie between 1 and 10 $\mu g/l$ while natural lead concentrations in soil range from 2 to 200 ppm and average 10 to 15 ppm. Deep ocean waters, below 1000 m, contain lead at 0.02 to 0.04 $\mu g/kg$ concentrations, but surface waters of the Mediterranean Sea and Pacific Ocean contain 0.20 and 0.35 $\mu g/kg$ levels (National Academy of Sciences, 1972). Drinking water concentrations of lead may be greatly increased in soft-water areas through corrosion of lead-lined piping and connections. However, average drinking-water intake is considerably less than food sources.

The biologic methylation of inorganic lead to tetramethyl lead by lake sediment microorganisms has been demonstrated (Wong *et al.*, 1975), but the significance of this observation remains unknown. It has not been possible to detect tetramethyl lead, trimethyl lead, or dimethyl lead in sediments or water that have high lead levels.

Arsenic. Arsenic is widely distributed in the environment. Man's input of arsenic into the global cycle occurs through smelting, coal burning, and the use of arsenical pesticides. Speciation of arsenic is an important consideration in the fate, movement, and action of this element. The chemical and biochemical transformations of arsenic include oxidation, reduction, and methylation, which affect the volatilization, adsorption, dissolution, and biologic disposition of the arsenic species involved.

Arsenic contamination of soils from point sources such as copper smelters or coal-burning power plants is easier to control than the dispersive use of arsenical pesticides, resulting in nonpoint-source pollution. Various forms of arsenic are used as pesticides. Chromated copper arsenate continues to grow in usage as a wood preservative (Fitzgerald, 1983). Arsenic acid (H_3AsO_4) is a leaf desiccant used in cotton production, lead and calcium arsenates were used as insecticides, and organic arsenicals, which include methanearsonic acid and its sodium salts as well as dimethyl arsenic acid (cacodylic acid), are used as postemergence herbicides. The transport of arsenic in the environment is largely controlled by adsorption/desorption processes in soil and sediments. Therefore, sediment movement is responsible for transfer of arsenic soil residues to their ultimate sinks in deep

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reproduction was decreased by approximately 80 percent by continuous exposure of 30 μg/l DEHP for up to 21 days. Reproduction in zebra fish and guppies was also decreased by low concentrations of DEHP.

Although the concentrations of phthalate esters in soil and water are quite low, the recent report of carcinogenicity and the continuing reports of the ubiquitous presence of residues in the environment led to concern about the potential for human health effects. Because phthalates have been found to occur in drinking water in the United States (Brass *et al.*, 1977) and are present in food, that concern is reinforced.

METALS

The toxicology of metals, including their use, occurrence, and effects, has already been presented in Chapter 19. In this section we will deal only with some aspects of the natural cycles of elements and conditions that alter the process involved. It is necessary to limit this treatment to the best-studied examples of environmentally important elements: mercury, cadmium, lead, arsenic, and selenium. (Note: arsenic and selenium are not metals, but the term will be used to include these metalloids.)

Mercury. Methyl mercury pollution of Minimata Bay and the subsequent human poisoning from consumption of contaminated seafood has stimulated much research on the origin and fate of methyl mercury. An important theme throughout the discussion of metals is the question of chemical species. Mercury, for example, exists in the inorganic form as free mercury, Hg^0, mercury ion in salts and complexes, Hg^{2+}; or as organic mercury compounds, such as phenylmercuric salts, which have been used as fungicides and herbicides, and the alkyl-mercury compounds including methyl mercury. In natural systems, a dynamic equilibrium that is determined by the physiochemical and biologic conditions of the soil-water system exists between the various chemical species. Some organic forms of mercury are man-made, such as the phenylmercuric compounds, while methyl mercury is produced by humans, sediment microorganisms, nonbiologically in sediments, and possibly by some fish. As expected, each species of mercury has its own set of physical, chemical, and toxicologic properties.

Mercury is transported to aquatic ecosystems via surface runoff and through the atmosphere. It is complexed or tightly bound to both organic and inorganic particles. Sediments with high sulfur content will strongly bind mercury. Organic acids such as fulvic and humic acid are usually associated with the mercury that is not bound to particles.

Methylation of mercury by microorganisms is a detoxication response that allows the organism to dispose of heavy metal ions as small organometallic complexes. Conditions for methylation by sediment microorganisms are strict and occur only within a narrow pH range. The rate of synthesis of methyl mercury also depends on the redox potential, composition of the microbial population, availability of Hg^{2+}, and temperature. Vitamin B_{12} derivatives are believed to be the methylating agents, since mechanistically they are the only methyl carbanion and methyl radical donating coenzymes known (Ridley *et al.*, 1977). An understanding of the biomethylation reaction mechanisms together with oxidation-reduction chemistry of elements allows predictions of the environmental conditions necessary for the biomethylation of mercury and several other metals. However, the best conversion rate for inorganic mercury to methyl mercury under ideal conditions is less than 1.5 percent per month (Jensen and Jernelov, 1969).

Little or no methyl mercury is found in sediments. Conversion of inorganic mercury to methyl mercury results in its desorption from sediment particles at a relatively fast rate. Demethylation by sediment microorganisms also occurs at a rapid rate when compared to methylation. Methyl mercury released in surface waters can undergo photodecomposition to inorganic mercury. However, methyl mercury can also be bioaccumulated by plankton algae and fish. In fish, the rate of absorption of methyl mercury is faster than that for inorganic mercury, and the clearance rate is slower with a net result of high methyl mercury concentrations in the muscle tissue. Selenium, which is present in seawater and seafood, readily complexes with methyl mercury and is believed to have an important protective action against the toxic effects of methyl mercury. In summary, the danger of methyl mercury poisoning, as occurred in Minimata, arises from direct methyl mercury contamination rather than methylation of environmental sources of inorganic mercury.

Cadmium. Cadmium has long been recognized as a toxic element. Its importance as an environmental contaminant was demonstrated in the outbreak of itai-itai disease caused by smelter wastes that contaminated rice paddies in Japan (see Chapter 19). Cadmium deposits are found as sulfides with zinc, copper, and lead deposits, and cadmium is recovered as a byproduct of smelting processes for those metals. A major environmental source of cadmium is vapor emissions that contaminate surrounding

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vaporization rate since air movement continuously replaces and mixes air around the evaporating surface. Many chemicals evaporate simultaneously with water, which leads some researchers to believe that chemicals such as DDT "codistill" with water. This phenomenon can be demonstrated in laboratory distillations at 100°C, but does not occur at normal environmental temperatures. Instead, water evaporation and DDT volatilization occur independently. Higher vapor loss from most soil surfaces correlates with pesticide volatilization, but is due to desorption of chemicals from soil adsorption sites by water molecules and the mass flow of chemical to the soil surface by the "wick effect." This phenomenon has been noted with chemicals such as 2,4-D esters, thiolcarbamates, triazines, organochlorine insecticides, and *N*-methyl-carbamates. Volatility of organic chemicals from water increases with decreasing water solubility. As a result, a chemical with both low vapor pressure in the solid phase and very low water solubility would be much more volatile from aqueous solution than might be expected. DDT is again the example.

Partitioning. The distribution relationships for a chemical between the environmental compartments of air, soil, water, and biota can be expressed by a series of partition coefficients (McCall *et al.*, 1983). The soil sorption constant (K_d) relates the amount of chemical sorbed to soil to the concentration in water (Table 26–2). Since organic matter is a key to the sorption process in many soils, as discussed above, the sorption characteristics of a chemical can be normalized by use of K_{oc}, which relates sorption properties to soil organic matter. Air-water distribution, expressed as K_w from Henry's law, is valid only for dilute solutions, which is the case for most environmental chemicals. The bioconcentration factor (BCF) is actually a measure of the partitioning of a chemical between water and fish but is a suitable indicator for biota in general. This information provides relative rankings of chemicals, although variations will occur among different organisms with respect to bioconcentration. A laboratory measure of the partitioning of a chemical between *n*-octanol and water provides K_{ow}, which is related to water solubility. K_{ow} values have been correlated with bioconcentration factors and soil sorption constants for a number of organic chemicals. Recently, retention times for chemicals on reversed-phase high-performance liquid chromatography (RP-HPLC) have been used to predict partitioning between the organic and aqueous phases of the environment (Swann *et al.*, 1983). Thus, with just a few measurements of the properties of a chemical (water solubility, vapor pressure, melting point, partitioning parameters, or RP-HPLC retention times) it is possible to guess a chemical's expected environmental distribution.

Bioaccumulation. Bioaccumulation is different from other environmental processes because it concentrates rather than diffuses the chemical in question. This concentration effect is expressed as the ratio of the concentration of a chemical in the organism to that in the medium (usually water). Bioaccumulation refers to both uptake of dissolved chemicals from water (bioconcentration) and uptake from ingested food and sediment residues. The two properties of a chemical that are responsible for high bioaccumulation ratio values are: (1) high partition coefficient, i.e., lipophilic, and (2) recalcitrance

Table 26–2. KEY PHYSICAL AND ENVIRONMENTAL PROPERTIES IN FATE ASSESSMENT*

PROPERTY	DEFINITION
Soil sorption coefficient (K_d)	$\frac{\mu g \text{ Chemical soil/g of soil}}{\mu g \text{ Chemical water/g of water}}$
Soil sorption constant (K_{oc})	$\frac{K_d}{\% \text{ Organic carbon}} \times 100$
Water-air ratio (K_w)	$\frac{\mu g \text{ Chemical/cm}^3 \text{ of water}}{\mu g \text{ Chemical/cm}^3 \text{ of air}}$
n-Octanol-water coefficient (K_{ow})	$\frac{\mu g \text{ Chemical/ml } n\text{-octanol}}{\mu g \text{ Chemical/ml of water}}$
Bioconcentration factor (BCF)	$\frac{\mu g \text{ of Chemical/g of fish}}{\mu g \text{ of Chemical/g of water}}$

* From Swann, R. L., *et al.*: A rapid method for the estimation of the environmental parameters octanol/water partition coefficient, soil sorption constant, water to air ratio, and water solubility. *Residue Rev.*, 85:17–28, 1983.

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classes of chemicals has been summarized by Hamaker and Thompson (1972) (Table 26–1).

Adsorption data for chemicals in soils is usually expressed by the Freundlich isotherm, $x/m = KC^n$: x/m is the amount of chemical sorbed per weight of the adsorbent, C is the equilibrium concentration of the chemical, and K and n are constants. The constant K represents the extent of adsorption while the value n sheds light on the nature of the adsorption mechanism and the role of the solvent, water.

Soil organic matter usually ranges from 0.1 to 7.0 percent and serves as the most important sorptive surface for nonionic chemicals. Above a few percent organic matter, all the soil mineral surfaces are effectively blocked and thus no longer function as adsorbents. Soil organic matter can be divided into two main groups: (1) nonhumic substances, which are fresh or incompletely decomposed plant and animal material, and (2) humic substances, which are more or less completely altered or resynthesized materials. The former serve as a source for the latter. Nonhumic materials include well-known organic chemical groups with definite characteristics: proteins, carbohydrates, organic acids, sugars, fats, waxes, resins, lignins, pigments, and low-molecular-weight compounds. These materials comprise 10 to 15 percent of the soil organic matter. Their composition and residence times are quite variable. Humic substances account for 85 to 90 percent of soil organic matter and their nature is not well understood. Humic substances are fractionated to give fulvic acid, which is soluble in both alkali and acid; humic acid, which is soluble in alkali but not in acid; and the humin fraction, which cannot be readily extracted with cold alkali. Humic acid and fulvic acid are aromatic polymers with molecular weights that range from 5000 to 100,000 and from 2000 to 9000, respectively. Functional groups that have been identified on humic substances are carboxyl, phenolic hydroxyl, alcoholic hydroxyl, carbonyl, and methoxy. Heterocyclic rings with oxygen and nitrogen atoms are also present. A hypothetic structure for humic acid has been proposed by Kononova (1966) as shown below.

Vaporization. Vaporization from soil, water, or plant surfaces is a major transport process for many chemicals. The volatility of a chemical is a function of its vapor pressure, but the rate of vaporization also depends on environmental conditions such as temperature, degree of adsorption, soil properties, and soil water content. Airflow over the evaporating surface affects

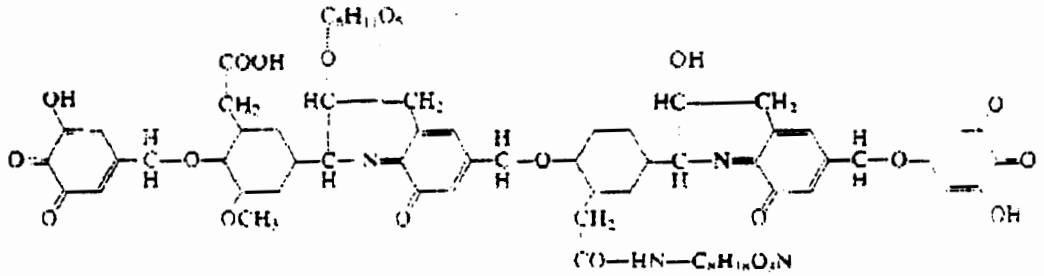

Table 26–1. CLASSES OF MATERIALS RELATED TO THE EFFECT OF pH ON ADSORPTION*

CLASS	EXAMPLE	pK_a	MOLECULAR FORM *Low* pH	MOLECULAR FORM *High* pH	pH *Effect*
Strong acid	Linear alkylsulfonates		Anion	Anion	Small
Weak acid	Picloram	3.7	Free acid	Anion	Large adsorption: pH approx. pK_a
Strong base	Diquat		Cation	Cation	Decrease at very low pH (18 N H_2SO_4)
Weak base	Ametryne		Cation	Free base	Increasing adsorption to pH approx. pK_a and then decrease
Polar molecules	Diuron		Nonionized	Un-ionized	Small
Neutral molecules	DDT	Nil	Nonionized	Un-ionized	Probably none

* From Hamaker, J. W., and Thompson, J. M.: Adsorption. In Goring, C. A. I., and Hamaker, J. W. (eds.): *Organic Chemicals in the Soil Environment*. Vol. 1. Marcel Dekker, Inc., New York, 1972. Reprinted by courtesy of Marcel Dekker, Inc.

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The domestic and urban use and disposal of chemicals also result in the contamination of soil and water. Domestic wastes are concentrated in sewage systems and landfill operations. Frequently, large buildups of heavy metals occur as a result. Pesticides and fertilizers used in suburban and some urban situations for lawn and home garden purposes or pest control in other situations also are serious problems when improperly used. Detergents may also cause difficulties as referred to above. The discovery that the process of purifying water can result in the chlorination of certain organic chemicals to produce chlorinated hydrocarbons that are potential carcinogens has generated concern. In some parts of the country concentrations of such chemicals above levels considered to be safe have been found in drinking-water systems.

Finally, metals, minerals, and plant or animal toxins are found in the environment as natural components of water and soil systems. Although they have always been there and always will be, human activities frequently result in excessive production or movement of such chemicals found naturally occurring in the environment and can result in concentrations detrimental to human or other organisms. Furthermore, the possibility of interaction of synthetic chemicals and pollutants with naturally occurring metals, minerals, and toxins must be considered.

Transport, Mobility, and Disposition

The fate and distribution of chemicals in the environment are determined by several variables that can interact in numerous ways. An analogy between pharmacodynamics and chemodynamics can be drawn to illustrate some basic similarities in each approach. First, one must appreciate the physicochemical properties of a chemical, such as water solubility, lipid solubility, partitioning behavior, vapor pressure, pKa for ionic species, chemical stability, etc., if one is to predict the behavior of a chemical in a system—be it human or an ecosystem. Second, the processes that act within the system must be considered. Transport, via serum proteins versus suspended sediments; circulation, via the circulatory system versus the hydrologic cycle; degradation, in liver versus soil microorganisms; and excretion, via urine, feces, and expired air versus dilution in water and air to nondetectable levels or deposition in ultimate sinks such as deep ocean sediments, are all processes that act on chemicals to determine the mobility and final disposition of a chemical in a system. The analogy can be carried one step further to include target organs or tissues affected by the chemical in comparison with the susceptible species of an ecosystem. The fundamental difference in this consideration is one of scale, in both time and dimension, which then requires models of varying scale. Several mathematical models have become available in recent years to predict the transport and fate of organic chemicals and metals in both aquatic and soil systems (Dickson *et al.*, 1982). The soil models are divided into unsaturated and saturated systems while other models deal with runoff and the transport of toxicants in surface waters. The value of such models is in understanding the properties that are most important in determining the environmental fate of a chemical under given conditions. A second objective is to predict exposure of various organisms through time to various toxicants and thus aid regulatory agencies in decision making.

Water Solubility. The water solubility and latent heat of solution are critical properties of a chemical that affect its environmental fate. Many environmental toxicants are hydrophobic, having solubilities in the parts-per-million (ppm, mg/l) to parts-per-billion (ppb, μg/l) range. Reported solubility values vary with the method used for determination (Gunther *et al.*, 1968). Water solubilities are affected by pH (for ionizable chemicals), presence of dissolved salts and organics, and temperature.

Soil Adsorption. Adsorption to particulate matter is a major mechanism by which chemicals are removed from solution. Adsorbent materials in soils and sediments can be divided into clay minerals and soil organic matter. Clay minerals include various hydrous silicates, oxides, and layer silicates. The clay minerals have been extensively studied and are characterized by physical structure or layering type, either 1:1 or 2:1, swelling ability, cation exchange capacity, and specific surface (m^2/g) (Weber, 1972). These parameters are important considerations in the behavior of organic cations, polar organic molecules, and metal ions in soils. High specific surface is associated with small particle size; therefore, the colloidal fraction of the soil is a dominant factor in chemical-soil interactions. Cation exchange capacity of the inorganic fraction is a function of the magnitude and distribution of the structural charge. Exchangeability is dependent on the adsorbed cations, usually sodium, potassium, or calcium, and the nature of the replacing cations.

The water associated with clay plays an important role in defining its characteristics. Adsorbed water on clay surfaces is more ordered than free water. Water on the clay surface may also be more ionized than otherwise. Thus, the hydrogen ion concentration of the clay surface is high. The effect of pH on the adsorption of

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