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TRACKING MULTIPLE PATHWAYS OF HUMAN EXPOSURE TO PERSISTENT MULTIMEDIA POLLUTANTS: Regional, Continental, and Global-Scale Models

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■ **Abstract** Widespread observations of organic pollutant compounds in vegetation, soil, animals, and human tissue have motivated research on more accurate characterizations of chemical transport over regional, continental, and global scales. Efforts to assess human and ecosystem exposure to contaminants from multiple environmental media have been evolving over the last several decades. In this review, we summarize the development and evolution of the multimedia mass-balance approach to pollutant fate and exposure evaluation and illustrate some of the calculations used in multimedia assessments. The concepts that form the foundation of Mackay-type mass-balance compartment models are described, and the ongoing efforts to use multimedia models to quantify human exposures are discussed. A series of case studies of varying complexity are used to illustrate capabilities and limitations of selected multimedia approaches. We look to the future and consider current challenges and opportunities in the field of multimedia contaminant fate and exposure modeling.

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INTRODUCTION

In the late-1950s, scientists began to recognize that certain chemical pollutants were capable of persisting in the environment for a long time, migrating between air, water, soils, and sediments, and accumulating to levels that could harm wildlife and humans. Carson brought this issue into the public eye in 1962 with her classic book *Silent Spring* (1), which described the potential environmental impacts of the insecticide DDT. Shortly thereafter, the discovery of polychlorinated biphenyls in fish throughout Sweden by Jensen (2) showed that industrial chemicals designed for use in closed systems were also entering the environment and accumulating to significant concentrations.

Prior to this time, the field of contaminant fate and exposure assessment was concentrated piecemeal on assessing chemical behavior in air, water, or soil, but this paradigm ran counter to the emerging realizations about the behavior of chemicals in the environment. A novel approach was required that described interactions between the seemingly distinct components of the ecosystem—the atmosphere, hydrosphere, lithosphere, and biosphere. In an effort to both articulate these issues and set new directions for environmental policy, the Council on Environmental Quality, which coordinates federal environmental initiatives in the United States, recommended in 1985 that long-term environmental research focus on 1. contaminant transfer rates among environmental media (i.e., soil, water, and air); 2. geohydrological processes at soil/water/air interfaces; 3. the role of biological, physical, and chemical processes in pollutant transport; and 4. the scientific basis for quantitative risk assessment (3). As illustrated in Figure 1, the multimedia approach requires comprehensive assessments that locate all points of chemical release to the environment, characterize mass-balance relationships, and track the contaminants through the entire environmental system to exposure of individuals or populations. A complete assessment of this type can be used to identify where in the chain of events control efforts would be most effective.

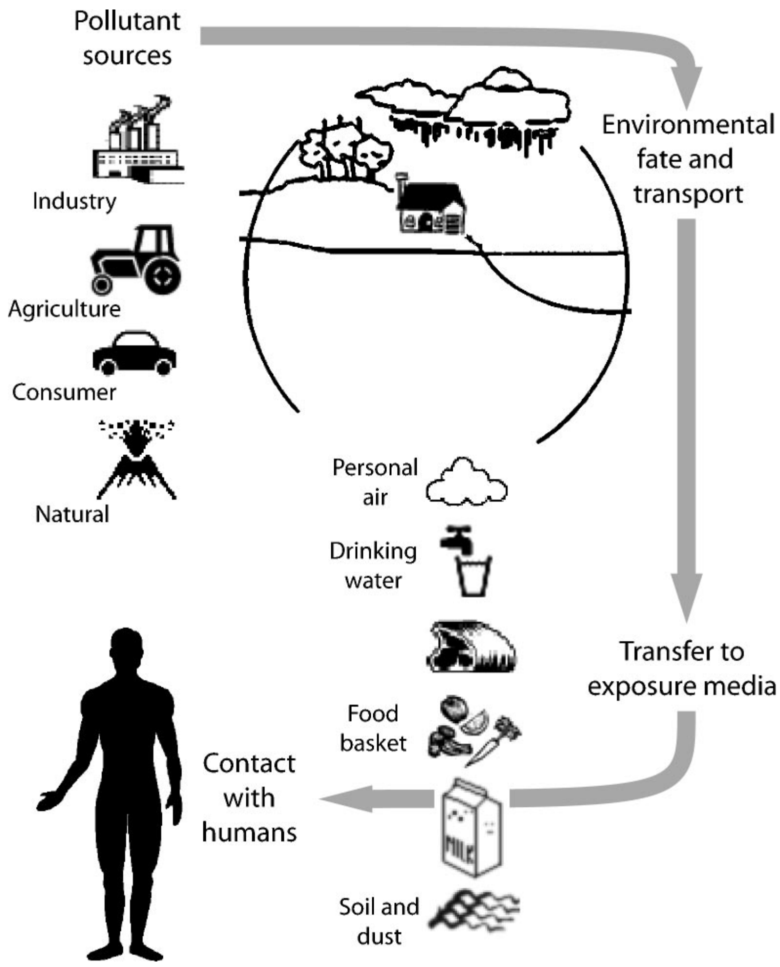

Figure 1 A conceptual illustration of the migration of pollutants from sources, through the multimedia environment and into exposure media, followed by contact with humans.

The emergence of the multimedia paradigm has focused attention on the long-term behavior and effects of chemicals released from modern industrial economies into the environment. Organic-chemical, inorganic-chemical, and radionuclide contamination of soils, the release of volatile and semivolatile organic compounds to air and to soil, and toxic-chemical runoff to surface water are all multimedia problems. Since 1985 an entire discipline of multimedia modeling of contaminants has evolved, and many useful techniques and modeling tools have been developed. Multimedia fate models are now widely applied for many types of environmental assessments. In this review, we document the development and application of multimedia models to questions about chemical transport and fate, cumulative exposure assessment, human and ecological health risk, and the behavior of persistent pollutants. We also explore questions about the models' reliability and how to address uncertainty in results and provide policy makers with useful guidance. These models have become widely accepted despite being impossible to validate in the conventional sense because of their inherent structure. We discuss reasons for this acceptance and suggest future research directions that will ensure the models remain useful conceptual and policy-making tools.

The History and Motivation for Multimedia Models

It is difficult to identify the true origin of multimedia models; however, it is clear that the need to assess human exposure to global fallout in the 1950s required considering transport through and among air, soil, surface water, vegetation, and food chains (4–9). Papers in the health physics and radioecology literature did not specifically refer to these models as multimedia, but they included many of the attributes that we now consider part of the multimedia-model genre. In the 1970s, growing concerns about the impacts of metal species such as lead, cadmium, mercury, and arsenic resulted in efforts to develop global and regional mass-balance models for these metals (10–12).

Regional/global mass-balance models for organic chemicals clearly emerged in the late 1970s with the publications of the first edition of *Chemodynamics* (13) and the seminal papers by Mackay describing the application of fugacity principles to environmental problems (14–16). In the early 1980s, Bennett (17–19) applied the source-to-dose methods of the health physics field to metals and organic chemicals with what he called the "exposure-commitment method," which was primarily an empirical model. By the late 1980s multimedia modeling had become more established and was the subject of several reviews, national workshops, and studies (20–24).

The most widely used multimedia models are the mass-conservative Mackay-type compartment models (14–16, 22, 24, 25). These models are most appropriate for treating transport and transformation of chemicals emanating from nonpoint sources over relatively long time and length scales at low concentrations.

An important extension of the multimedia paradigm was the introduction of comprehensive models that linked environmental contamination to multiple pathways for human or wildlife exposure. McKone & Layton (26) were among the first to link regional multimedia mass-balance models to deterministic multipathway exposure models to assess the health risks of the U.S. Army's program for managing the disposal of explosives and propellants. By the early 1990s, the paradigm was sufficiently well accepted by scientists and regulators that the California Environmental Protection Agency adopted a multimedia approach for setting clean-soil goals through its CalTOX program (27–31), and the European Union adopted a multimedia multipathway framework for chemical risk assessment (32). The CalTOX model also introduced formal consideration of uncertainty and variability in multimedia models.

A Brief Overview of Multimedia, Multipathway Modeling

Multimedia fate and exposure models synthesize information about partitioning, reaction, and intermedia-transport properties of a chemical in a representative or generic environment with information about exposed humans or wildlife to assess impacts, such as health risk. The environment is treated as a set of compartments that are homogeneous subsystems exchanging water, nutrients, and chemical contaminants with other adjacent compartments. There are two basic features that make compartment models suitable for an integrated model of transport and transformation in multimedia environments: (*a*) Each compartment forms a unit in which one can balance gains and losses attributable to sources, transfers to and from other compartments, and chemical transformations, and (*b*) each compartment forms a unit in which chemical partitioning can be evaluated against equilibrium criteria.

A cumulative multipathway exposure assessment for humans relates contaminant concentrations in multiple environmental media to concentrations in the media with which a human population has contact (for example, personal air, tap water, foods, household dusts, and soils). The potential for harm is assessed either as the average daily intake or uptake rate or as time-averaged contact concentration.

Multimedia contaminant fate and exposure models have been useful to decision makers because these models provide an appropriate quantitative framework to evaluate our understanding of the complex interactions between chemicals and the environment. The greatest challenge for multimedia models is to provide useful information without creating overwhelming demands for input data and producing outputs that cannot be evaluated. The multimedia modeler must struggle to avoid making a model that has more detail than can be accommodated by existing theory and data while also including sufficient fidelity to the real system to make reliable classifications about the source-to-dose relationships of environmental chemicals.

Overview of the Chapter

The first section of this review focuses on the fundamental assumptions of compartment-based mass-balance models and their mathematical formulation. Simple examples illustrate multimedia fate calculations using the fugacity concept. In the next section, we describe multipathway exposure models used to calculate rates of exposure by humans or wildlife from measured or modeled contaminant concentrations. Finally, we describe and review efforts to combine the contaminant fate and exposure models to conduct source-to-dose calculations for multimedia environmental contaminants and discuss some of the philosophical and technical challenges currently facing model developers.

MULTIMEDIA CONTAMINANT FATE: THEORY AND MODELS

We present here an overview of mass-balance contaminant fate and transport calculations using the fugacity approach. The text by Mackay (25) provides a comprehensive discussion of this topic and is highly recommended for the interested reader. Additional information on many of the concepts discussed below and alternative formulations of mass-balance equations using rate constants and concentrations rather than the fugacity concept are available in Schwarzenbach et al. (33) and Thibodeaux (13).

Phase Equilibrium and Chemical Partitioning in Environmental Systems

In this section, we explore the philosophical approach used in developing multimedia contaminant fate models. The complexity of multimedia environments and our lack of data and knowledge limit our ability to quantify multimedia processes. As a result, the approach to model development is to simplify the chemical-environment system as much as possible and systematically build up a more detailed understanding of the system. At their core, all multimedia models of contaminant fate in the environment are based on a set of three postulates. The first two are axiomic, and the third provides a convenient framework for avoiding being overwhelmed by the complexity and variability of the natural environment.

1. CONSERVATION OF MASS This axiom provides the basis for writing mass-balance equations, which are indispensable tools in a wide range of engineering applications. Chemicals put into the environment will accumulate, particularly if removal processes are slow. Adoption of this postulate represents recognition that, although vast, the assimilative capacity of the environment for chemicals is finite.

2. CHEMICAL EQUILIBRIUM According to the Second Law of Thermodynamics, chemicals are driven to disperse into the environment such that the chemical-environment system reaches thermodynamic equilibrium. In a closed system with multiple solvating phases at equilibrium a solute will be distributed in such a way that the system has reached a minimum of free energy. The change in free energy associated with movement of the solute from one region to another is directly proportional to the difference in chemical potential between the regions.

3. THE USE OF LINKED COMPARTMENTS The environment can be usefully described as a set of linked compartments or boxes. In contrast to the axiomic laws above, this is clearly a gross simplification of the real system under study. The underlying assumption is that the overall fate of chemicals of interest is more strongly controlled by partitioning between the various phases available to the chemical than by spatial differences in properties within individual compartments

of the system. This postulate is consistent with a philosophical approach to model development that views a useful model as one that captures the characteristics of a system that are assumed to be important and omits those that are assumed to be extraneous.

With these three postulates as a foundation, multimedia mass-balance models describing the partitioning and ultimate fate of chemicals in the environment can be assembled. Figure 2 provides a simple generic example of a regional environment consisting of five discrete compartments. In order to formulate the equations that quantitatively describe the system illustrated in Figure 2, and recognizing the second postulate above, we introduce the fugacity concept.

Fugacity and Fugacity Capacity

Fugacity is a metric for quantifying chemical activity at low concentrations. Fugacity, f, can be viewed as the "escaping tendency" of a chemical in a phase, has dimensions of pressure, and is related to concentration, C, by a proportionality constant, fugacity capacity, Z:

$$C = Zf. \qquad 1.$$

In the International System (SI) of units, Z has units of mol m^{-3} Pa^{-1}. From the ideal gas law, it can be shown that Z for the vapor phase is $1/RT$ where R is the gas constant and T is absolute temperature.

Fugacity is a criterion of equilibrium. When a chemical reaches equilibrium distribution between two available phases, the fugacities of the chemical in the phases are equal. Equilibrium partitioning between two phases can also be described by a dimensionless partition coefficient K_{12}, which can be measured under laboratory conditions as the ratio of concentrations C_1 and C_2. Applying the relationship between concentration and fugacity, and recognizing that $f_1 = f_2$ at equilibrium:

$$K_{12} = \frac{C_1}{C_2} = \frac{f Z_1}{f Z_2} = \frac{Z_1}{Z_2}. \qquad 2.$$

Z can thus be determined experimentally for many phases by measuring partition coefficients between the phase of interest and a phase with known Z.

The environment shown in Figure 2 includes air, water, and soil and sediment particles. As shown in Table 1, Z values for these four phases can be derived from the Z value for air and appropriate partition coefficients measured in the laboratory. Several assumptions are required to arrive at these expressions; these include (*a*) that chemicals in the vapor phase obey the ideal gas law, (*b*) that chemicals in the aqueous phase form ideal dilute solutions, and (*c*) that octanol can be used as a surrogate to describe chemical partitioning to lipids and the organic carbon component of soil and sediments, as described by Karickhoff (34) and reviewed by Seth et al. (35).

TABLE 1 Fugacity capacities (**Z** values) for environmentally relevant phases

Phase	**Definition of Z (mol/m^3 Pa)**	
Air	$Z_A = 1/(R \times T)$	$R = 8.314$ Pa × m^3/mol × K T = temperature (K) H = Henry's Law constant (Pa × m^3/mol)
Water	$Z_W = 1/H = C^S/P^S$	C^S = aqueous solubility (mol/m^3) P^S = vapor pressure (Pa) y_S = fraction of organic carbon in soil k = Karickhoff constant = 0.41 L/kg
Soil solids	$Z_S = Z_W \times y_S \times k \times K_{OW} \times \rho_S/1000$	K_{OW} = octanol-water partition coefficient ρ_S = density of soil (kg/L) 1000 converts L to m^3
Sediment solids (x)	$Z_X = Z_W \times y_X \times k \times K_{OW} \times \rho_X/1000$	y_X = fraction of organic carbon in sediment ρ_X = density of sediment (kg/L)

Partitioning in a Closed System—Level I Models

Treating the environment as an open or closed system and making different assumptions about compartment equilibria and temporal trends in concentration allow multimedia models of varying complexity to be assembled. Four levels of complexity suggested by Mackay (25) are summarized in Table 2. As one progresses from Level I to Level IV calculations, the fidelity of the calculation to the actual chemical-environment system increases, but it is at the cost of additional requirements for input data to describe both the environment and the chemical. At the lowest level of complexity, we model a closed system at equilibrium.

From the fugacity capacities in Table 1 and the law of conservation of mass, a simple fugacity calculation can be used to describe the equilibrium partitioning of a fixed number of moles (M) of chemical in a closed environmental system consisting of four compartments (which may be composed of several distinct phases) with defined volumes (V).

$$\begin{aligned} M &= V_1C_1 + V_2C_2 + V_3C_3 + V_4C_4 = V_1Z_1f + V_2Z_2f + V_3Z_3f + V_4Z_4f \\ &= f(V_1Z_1 + V_2Z_2 + V_3Z_3 + V_4Z_4). \end{aligned} \qquad 3.$$

Where C is the concentration of chemical in each compartment, the prevailing fugacity of the chemical in the system can be calculated from the general equation:

$$f = \frac{M}{\sum V_i Z_i}. \qquad 4.$$

TABLE 2 Summary of fugacity calculations of different levels of complexity used to describe multimedia contaminant fate

Type of fugacity calculation	Key assumptions	Information garnered
Level I	Equilibrium partitioning Steady state Closed system	General partitioning tendencies for persistent chemicals
Level II	Equilibrium partitioning Steady state Open system	Estimate of overall persistence Important compartments for removal processes Relative importance of advection and degradation as removal pathways
Level III	Nonequilibrium Steady state Open system	Influence of mode of emission on fate and transport Refined assessment of overall persistence and loss pathways
Level IV	Nonequilibrium Dynamic Open system	Influence of mode of emission on fate and transport Time course of response of contaminant inventory by compartment to any time-varying condition

This simple equilibrium partitioning calculation is termed a Level I fugacity calculation (25). The results of the Level I calculation provide insight into the influence of chemical properties on environmental partitioning and a rapid assessment of the environmental media into which the chemical is likely to partition.

Figure 3 provides sample results from a Level I calculation for the homologous series of chlorinated benzenes as adapted from MacLeod & Mackay (36). Equilibrium partitioning of members of the series at each of the seven possible chlorination levels are plotted as a percentage in the air, water, soil, and sediment of a generic environment with properties similar to a jurisdictional region such as the state of Ohio or the country Greece.

As indicated in Figure 3, the less chlorinated congeners of the series of chlorinated benzenes partition almost exclusively to air. With increasing chlorination level, there is an increased tendency for chlorobenzenes to partition out of the atmosphere and into the organic carbon component of soils. This simple assessment of the equilibrium partitioning tendencies of chlorobenzenes is consistent with the observed environmental fate of these compounds (36).

Equilibrium Partitioning in an Open System—Level II Models

As illustrated above, Level I fugacity calculations provide an indication of the likely long-term partitioning of persistent contaminants in the environment, and they can be carried out with only a basic knowledge of chemical partitioning characteristics. However, in most cases it is desirable to describe the real environment-chemical system with more fidelity. At Level II complexity, sources are balanced with removal by chemical transformation and advection at the system boundaries, but the assumption of equilibrium partitioning among compartments is retained. In fugacity models, the rate of chemical transport and/or degradation is generally described using D values such that the product (Df) equals the removal rate in mol/h. D values therefore have SI units of mol/(Pa h) and are analogous to first-order rate constants. Because they are compartment models, multimedia models can account for area and volume sources but not point sources of pollution. In Level II models, sources are introduced as continuous inputs to one or more compartments.

TRANSFORMATION PROCESSES—CHEMICAL REMOVAL Chemical transformations may occur as a result of biotic or abiotic processes that include biotransformation, photolytic decomposition, hydrolytic transformation, and oxidation/reduction. These processes change the chemical identity of a compound and its fundamental properties, and thus its partitioning characteristics and behavior in the environment. The rate of transformation of contaminants in the environment ultimately determines their potential for persistence. For organic chemicals in particular, the United Nations Environment Program uses half-lives for transformation as criteria for classifying chemicals as persistent in the environment (37). Experimental methods (38) and estimation methods (39, 40) are available for specifying these fate processes in a variety of media. However, because transformation rates are highly variable and difficult to measure, they are among the most uncertain parameters used in multimedia models.

PHYSICAL DISPLACEMENT OF POLLUTANTS—ADVECTION In addition to transformation, Level II models account for physical displacement that removes chemicals at the boundaries of the environment being modeled. The Level II assumption of equilibrium makes assessment of displacement at compartment boundaries within the environmental system unnecessary. Physical displacement occurs by advection and turbulent diffusion. Advective removal refers to chemical displaced by entrainment in a moving medium, such as in a moving air mass, a flowing river, or in suspended sediments settling out of the water column to the bottom of a lake. Unlike degradation, chemicals that advect or diffuse out of the environmental system under consideration may return and, therefore, may not be permanently removed. In fact, for regions that are adjacent to relatively highly contaminated areas, advection into the region may be the dominant emission source term.

LEVEL II FUGACITY CALCULATIONS Degradation D values (D_R) can be calculated from an estimated pseudo first-order rate constant (k) for chemical transformation in the compartment of interest (i).

$$D_{Ri} = k_i V_i Z_i, \tag{5.}$$

where (V) is the volume of the environmental compartment. The rate constant k must be selected to represent the overall rate of degradation of the chemical, which as noted above may be taking place by several competing mechanisms.

Advection D values (D_A) are calculated as the product of the flow rate of the medium in which the chemical is entrained (G, m^3/h) and its fugacity capacity.

$$D_{Ai} = G_i Z_i. \tag{6.}$$

The mass-balance condition at Level II is between sources of the chemical to the environment (E, mol/h) and removal processes. For a regional environment consisting of four compartments,

$$E = D_1 f + D_2 f + D_3 f + D_4 f, \tag{7.}$$

where D_i is the total D value for removal from a given compartment by chemical transformation and advection. The equilibrium assumption makes it possible to calculate a single fugacity for the system, which is the same in all compartments:

$$f = \frac{E}{\sum D_i} = \frac{E}{\sum (G_i + V_i k_i) Z_i}. \tag{8.}$$

Figure 4 shows Level II mass-balance diagrams for benzene and hexachlorobenzene in the generic evaluative environment used previously for the Level I calculations.

Hexachlorobenzene (HCB) is far less reactive in the environment than benzene, and it has a higher potential for persistence and transport out of the region of emission, as is evident in Figure 4. Whereas 80% of the removal of benzene from the environment is by transformation, specifically degradation in air, greater than 97% of removal of HCB is by advection in the atmosphere for the environmental system under consideration. The low rate of degradation and dominance of atmospheric transport as a removal process indicate that HCB is likely to travel long distances from areas of disposal or use and be present in remote environments, as is indeed observed (41).

Nonequilibrium Partitioning in Open Systems—Level III Models

A Level III fugacity model includes the rates of inter-media transport between environmental compartments. The mass-balance condition is applied to each environmental compartment. This requires quantification of diffusion and advection rates at the compartment boundaries. Although there is no requirement for equilibrium partitioning between adjacent compartments, it is still assumed that chemicals achieve equilibrium partitioning among the available phases within a compartment.

One of the major advantages of fugacity models is their ability to represent diffusive and advective transfer processes among environmental media of different composition. When the modeler relies on concentration-based algorithms, these mass-transfer calculations require the use of flux matching, careful unit conversions, and selection of appropriate partition factors at compartment boundaries. Because fugacity has the same units in all media, this difficulty is avoided in fugacity-based models.

DIFFUSIVE TRANSPORT AMONG COMPARTMENTS In a fugacity model, the net diffusive flux, in mol/m^2-h, across the interfacial area separating compartments is:

$$\mathit{flux} = Y_{12}(f_1 - f_2), \tag{9.}$$

where Y_{12} is the fugacity mass-transfer coefficient across the boundary between compartments 1 and 2 with units mol/(m^2-Pa-h) and f_1 and f_2 are the fugacities of compartments 1 and 2. Equation 9 is analogous to the flow of electrons in a circuit in which $(f_1 - f_2)$ plays the role of a voltage difference, Y_{12} is a conductance, and the mass flux is the equivalent of electrical current. The fugacity mass-transfer coefficient depends on the mass-transfer coefficient on either side of the interface and the fugacity capacities of the two media that form the interface.

$$Y_{12} = \left(\frac{1}{Z_1 U_1} + \frac{1}{Z_2 U_2}\right)^{-1}, \tag{10.}$$

where U_1 and U_2 are the mass-transfer coefficients (m/h) in the boundary layers in compartments 1 and 2 and Z_1 and Z_2 are the fugacity capacities of compartments 1 and 2.

ADVECTIVE TRANSPORT AMONG COMPARTMENTS The inter-compartment transfer of contaminants by advection is also modeled as a flux at the compartment boundary. To be consistent with the area normalized description of diffusion, this flux (mol/m^2-h) is modeled as the product of the velocity of the moving phase (m/h) and the contaminant concentration in that phase (mol/m^3).

$$\mathit{Advection\ flux} = \mathit{velocity} \times Z_{ik} f_i, \tag{11.}$$

where Z_{ik} and f_i represent the fugacity capacity of the moving phase and the fugacity of the chemical, respectively, in compartment i. For example, the flux of contaminant from air to surface soil through particle deposition is the product of the particle deposition velocity, v_d; the fugacity capacity of air particles, Z_{ap}; and the total fugacity of the bulk air compartment, f_a:

$$\mathit{Flux}\ \text{(air to ground-surface soil)} = v_d \times Z_{ap} \times f_a. \tag{12.}$$

Examples of inter-compartmental advection processes typically included in multimedia models are rainfall, deposition of atmospheric aerosol particles, resuspension of particles from soil, water-borne erosion of soil, runoff of precipitation, infiltration of water through soil, deposition of sediment particles in surface water, resuspension of sediment particles from the sediment layer, and surface water flows.

Once equations describing all inter-compartmental transfers of contaminants have been derived, mass-balance equations equating input and removal rates can be written for each compartment (i) of the environmental system.

$$E_{\mathrm{i}} + \sum flow_{j\to i} = \sum flow_{i\to j} + \sum flow_{i\to \mathrm{sink}}. \qquad 13.$$

On the left-hand side of Equation 13 are chemical inputs to compartment i by direct emission, E_i, and the total rate of inter-compartmental transfer to compartment i, $\Sigma flow_{j\to i}$. Removals from the compartment occur by inter-compartmental transfer ($\Sigma flow_{i\to j}$) and by advection out of the system or chemical transformation ($\Sigma flow_{i\to \mathrm{sink}}$). For an environment consisting of n compartments, one can write n equations of this type and solve them algebraically to obtain the unknown fugacity of chemical in each compartment.

Dynamic, Open, and Nonequilibrium Systems—Level IV Models

By removing the assumption of steady-state conditions, we attain the Level IV system. Removal rates and rates of inter-media transport between environmental compartments are used to define a time-dependent description of mass distribution. The rate of chemical input to each compartment can be continuous or time varying. This makes assessment of transient effects possible, such as seasonal variations in emissions and/or climate and soil conditions.

MULTIPATHWAY EXPOSURE MODELS

Multimedia fate models provide additional insight into human and environmental impacts when exposure pathways are explicitly included in the model framework. Exposure assessment is the process of measuring or modeling the magnitude, frequency, and duration of contact between a potentially harmful agent and a target population, including the size and characteristics of that population (27, 42–46). In this section we describe a standard framework for organizing and calculating multipathway exposure to multimedia pollutants. Figure 5 illustrates how links between ambient environmental media and exposure media are included in an exposure model. Human exposures to pollutants released to the ambient environment result from contacts with contaminated air, water, soils, and food. Exposures may be dominated by contacts with a single medium or may reflect concurrent contacts with multiple exposure media. Here, we focus on models that describe the exposure of human individuals or populations to chemical agents dispersed in a multimedia environment.

An exposure assessment begins with exposure concentrations in contact media, which include the envelope of air surrounding the exposed person, the water and food ingested, and the layer of soil, water, or other substances that contacts the skin surface. The magnitude and relative contribution of each exposure pathway must be considered in order to assess total human exposure to a harmful substance and determine the best approach for more refined characterization of the exposure. For example, consider exposure of an individual to a semivolatile hazardous air pollutant that is released into air. In the atmosphere, the chemical will partition between the vapor phase and the condensed phase (the surface of airborne particles). Both the vapors and the particles containing the associated pollutant can be transported to the indoor or outdoor air surrounding a person, who may inhale the pollutant. However, the gas-phase and particle-phase pollutant may be transported with different efficiencies. The same pollutant could be transferred by deposition and runoff to surface water that provides drinking water to a population of individuals or transferred by deposition to vegetation that feeds the population or to the agricultural animals that supply meat and milk. Each of these scenarios represents a pathway from the air emission to contact with a human. Each pathway has an associated exposure surface and subsequent route of intake or uptake. The true potential for exposure cannot be quantified unless the pathways that account for a substantial fraction of the routes of intake and uptake for the receptor population are identified.

Exposure Events

The nature and magnitude of exposures to environmental contaminants depend largely on two things: (*a*) human factors and (*b*) the concentrations of contaminants in the exposure media. Human factors include all behavioral, sociological, and physiological characteristics of an individual that determine their contact rates with food, air, water, and soils. For example, about 35 percent of the U.S. population eats homegrown vegetables, which must be reflected in the treatment of this exposure pathway. Activity patterns, which are defined by allocation of an individual's time at different activities and locations, are also significant because they directly affect the magnitude of exposure to substances present in different indoor and outdoor environments.

Multimedia models that include multipathway exposures (such as the CalTOX model) use exposure events to construct cumulative intake from human factors and exposure concentrations. An *exposure event* occurs when human activities bring them into contact with an exposure medium within a specified microenvironment. Important attributes that define an exposure event are pollutant concentration in environmental media (i.e., ambient air, surface water, and soil), transport pathways from environmental media to exposure media, the duration of and number of exposure media contacts, and the timescale of interest for health effects. The timescale of possible health effects, i.e., whether chronic or acute, and the possibility of spatial and temporal variations in pollutant concentration are important considerations in compiling an exposure assessment. For some pollutants, such as criteria air pollutants, which have acute health effects associated with short-term exposure, the assessment must describe the number and duration of peak concentration events. Exposure events may need to be aggregated over periods as short as one hour or less. In contrast, for pollutants with chronic health impacts, it is more important to characterize long-term cumulative exposures over several years or decades.

The timescale and spatial scale of gradients in pollutant concentrations also provide critical insight into the resolution appropriate for modeling an exposure event. If a pollutant shows little spatial variation in concentration over a large region, even if there is time variation in the concentration, there is little need for including a number of geographic regions in the assessment. Similarly, for a pollutant whose concentration does not vary significantly in time, even if it shows large spatial variation, it may be possible to use longer timescales in an exposure assessment. The duration of the exposure event and the duration of human activities are also important considerations in the structure of the exposure-event model.

Cumulative Exposure Models

Multipathway exposure assessments use the intake or potential dose model that has been adapted from a general Environmental Protection Agency model (47). For each combination of environmental media and contact media, a transfer factor is calculated that defines the quantitative relationship between contaminant concentrations in the two media. These transfer factors are classified in terms of the route of exposure (inhalation, ingestion, and dermal) and the exposure medium/environmental medium link, for example, soil to house dust. There is thus a defining equation for each combination of environmental medium, exposure medium, and exposure route. These relationships are expressed in the CalTOX multimedia exposure model as an average daily potential dose rate (ADDpot), in mg/kg-d relative to a specified environmental medium concentration:

$$ADD_{pot} = C_{\mathrm{Env}} \times \mathrm{ITF} \times \frac{\mathrm{CR}}{\mathrm{BW}} \times \frac{\mathrm{ED} \times \mathrm{EF}}{\mathrm{AT}}, \qquad 14.$$

where C_{Env} is the concentration in an environmental medium, mg/kg; ITF is the ratio of concentration in an exposure medium to concentration in environmental medium, unitless; CR is the contact rate with the exposure medium, kg/day; BW is the representative body weight, kg of the exposed individual or cohort; ED is the exposure duration, year; EF is exposure frequency, days/year; and AT is the averaging time for the exposure, days. AT is based on the timescale for health effects. For cancer as an endpoint, AT is lifetime; for other chronic agents it is less than lifetime, but often years.

APPLICATIONS OF MULTIMEDIA FATE AND EXPOSURE MODELS

Multimedia fate models are now widely used for screening-level chemical assessments, for setting goals for soil cleanup standards, for assessing the regional and global fate of persistent organic chemicals, and for life-cycle impact assessment.

In many of these applications the models are used for assessing environmental fate and potential human or ecological impacts. Emerging uses for these models include exposure tracking based on toxic release data and premanufacturing chemical classification. The U.S. Clear Air Act of 1990 and efforts in the European Union to conduct risk assessments for existing and new industrial chemicals have led to the development of specific application multimedia models. In this section, we provide an overview of some of the currently available multimedia fate and exposure models and examples of their applications.

Generic Screening or Evaluative Models

Many multimedia fate and exposure models are based on environmental parameters that are not representative of any specific geographical area. These generic models are used as a laboratory for evaluating the likely behavior of pollutants and how this relates to basic chemical properties. The focus is on comparative assessments of chemicals and interpreting how partitioning properties and degradability determine transport and fate processes. Early examples of evaluative regional fate models are the suite of fugacity models developed by Mackay in the early 1990s (22) and the Equilibrium Criterion or EQC model, which defined a standard four-compartment environment of air, water, soil, and sediment (48). Although these models were originally developed to track the fate of nonpolar, nondissociating organic compounds, their use has expanded to metals, inorganic species, surfactants, and polar organic chemicals.

Generic multipathway human-exposure models coupled with multimedia fate models have been developed in the United States and Europe. The California Environmental Protection Agency developed CalTOX for conducting generic assessments of chemical mobilization from hazardous waste sites and subsequent human exposure (28). In Europe, the Netherlands National Institute for Public Health and the Environment (RIVM) developed SimpleBox, a generic contaminant fate model based on concentrations and first-order rate constants but consistent with fugacity concepts (49, 50).

Recently, generic models of contaminant fate have been adapted to conduct rapid screening-level assessments of large numbers of chemicals for persistence (P) and potential for long-range transport (LRT). The P and LRT attributes have been identified as cause for global concern and have been used as a basis for international bans imposed on specific chemical compounds (51). Webster et al. (52) describe a generic Level III fugacity model for comparing chemicals in terms of persistence in the entire environment, rather than half-lives in individual media. Bennett et al. (53) and Pennington (54) proposed and applied similar models to calculate multimedia persistence. Generic models for assessing long-range transport potential have been developed by Bennett et al. (55), who introduced the concept of “characteristic travel distance” for multimedia chemicals, and by Beyer et al. (56). Faced with a proliferation of generic models for assessing P and LRT, Wania & Mackay (57) conducted a round-robin comparison of the models and found that, although the magnitude of P or LRT scores differed among the models, the relative ranking of a standard group of chemicals according to P or LRT was consistent.

Regional and Spatially Resolved Models

Multimedia models have evolved from evaluative models to regional mass-balance models, multi-region models, and global models. Regional multimedia mass balance models have the same framework as generic models but include geographical databases representing a specific political or ecological region. Multi-region models include more spatial resolution by linking several regional mass-balance models and have been applied on local, continental, and global scales.

REGIONAL-SCALE MASS-BALANCE MODELS An advantage of regional models over evaluative models is that results can be directly compared with reported concentrations of contaminants in a specific area. For example, ChemCAN is a generic Level III fugacity model with the addition of a database of environmental parameters for 24 ecological regions of Canada (58). ChemCAN has been evaluated for a limited number of contaminants in the southern Ontario, Canada region (36) and also applied in Japan (59). The current version 4.0 of CalTOX includes a geographical database for several specific regions of California, 10 ecoregions of the United States, and political regions representing 48 of its states. The SimpleBox model was adapted into the EUSES framework (see below) and became a regional fate model for the European states. More recently regional-scale models that provide more detailed treatment of densely populated urban environments, forest canopy compartments, and coastal waters have been developed (60, 61).

Decision makers and model developers convened an international workshop in 1994 to assess the strengths and weaknesses of the available regional multimedia models and make recommendations about future research in regional environmental contaminant fate modeling (24). The workshop participants used simple model-comparison exercises to demonstrate the fundamental similarities between ChemCAN, CalTOX, SimpleBox, and HAZCHEM, an adaptation of ChemCAN applied to the European Member states (62). Among the recommendations from the workshop were more evaluation studies comparing model results against reported data on environmental concentrations, improved methodologies for conducting sensitivity and uncertainty analysis, and new model inter-comparison exercises when the science advances and models are refined.

MULTI-REGION MASS-BALANCE MODELS The current set of regional multimedia fate models represent a region using a single set of environmental parameters representing the multiple environmental media. This limits their ability to track the movement of contaminants among different geographic regions. An emerging class of multimedia models includes spatial resolution by connecting a set of discrete regional fate models into a single larger model. The approach of linking different model units was first illustrated in contaminant fate models for river systems such as the Fraser River in British Columbia, Canada, (63) and the GREATER model for rivers in the European continent (64).

Regional models have also been linked to describe contaminant fate and transport on a continental-scale based on both air and water transport. The Berkeley-Trent North American contaminant fate model (BETR North America) includes 24 linked regions and an accompanying geographical information system (GIS) database of long-term flow patterns of air and water on the continental scale (65, 66). BETR North America has been applied to describe the entire use history and long-term fate of the persistent organochlorine pesticide toxaphene, including its atmospheric transport and deposition to the Laurentian Great Lakes (67). In another example of combining air- and water-shed transport, the generic CoZMo-POP framework has been adapted to describe contaminant fate in the Baltic Sea and its terrestrial drainage basin (68).

GLOBAL-SCALE MULTIMEDIA MODELS Wania & Mackay (69) introduced a multimedia global distribution model for persistent organic chemicals. They treated the global environment as nine connected climate zones and described contaminant fate in each zone with a fugacity mass-balance model. The model has been applied to describe the global fate and transport of the pesticide hexachlorocyclohexane (70) with a focus on transport and deposition to the Arctic.

Evaluative models of global contaminant fate have also been developed by Scheringer (71–74) to assess persistence and spatial range as endpoints in screening level assessments of environmental hazard for chemicals. These models are simpler than the Wania-Mackay global model and have proved useful in studies to determine how temperature gradients in the global environment affect the long-term fate of persistent contaminants.

Assessing Exposures to Humans and Ecosystems

In comparison to the proliferation of models of contaminant fate in the environment, there have been fewer efforts to develop source-to-dose relationships for humans using multimedia models coupled to multipathway exposure models. In the United States, CalTOX is an example of a model that evolved from an exposure model for hazardous sites into a comprehensive fate and human exposure model with integrated probabilistic assessment of uncertainty and variability (27, 75, 76). A predecessor to the CalTOX model called GEOTOX (26) provided an early example of linking regional multimedia mass-balance models to deterministic multipathway exposure models. In Europe, SimpleBox has been incorporated into a model framework for fate, exposure, and risk assessment of multiple chemicals. This system was first developed in the Netherlands as the Unified System for the Evaluation of Substances (USES) (77). The European community has adopted USES as a decision support framework and for conducting chemical risk assessments, the European Union System for the Evaluation of Substances (EUSES).

EXAMPLE OF COMPARATIVE HUMAN-EXPOSURE ASSESSMENT To illustrate the use of multimedia models for comparative exposure assessment, we apply the CalTOX model to two organic pollutants, hexachlorobenzene (HCB) and pentachlorophenol (PCP). CalTOX is used to estimate concentrations in the air (gas and particles), plants, soil (surface and vadose zones), and water compartments. Then the resulting potential multipathway human exposures are calculated using methods described above. This calculation is applied to a generic landscape of 1 million square kilometers, and the exposure calculations represent the general population, i.e., they do not account for workers who may be exposed to these chemicals occupationally. For each chemical, we model continuous releases for a period of 15 years. The illustrative multimedia releases used for both chemicals in these examples are 100,000 mol/d to air, 10,000 mol/d applied to the soil surface, and 1000 mol/d to surface water. The pollutants in this example are structurally similar but differ significantly in chemical properties and exposure pathways (Figure 6).

As demonstrated in earlier examples, HCB is a hydrophobic, persistent compound capable of long-range atmospheric transport on a global scale. HCB binds to soil and to suspended sediments in water and accumulates to some extent in the bottom sediments of lakes, rivers, and estuaries. It can bioaccumulate in fish, marine mammals, and birds, and it is also transferred into grasses, vegetables, and other plants. Like HCB, PCP is a manufactured chemical not found naturally in the environment. It is persistent in the environment and generally binds to soil particles, but its mobility in soils depends on the soil's acidity. PCP does not readily evaporate into the air and is not readily soluble in water. In soils and surface waters, microorganisms break down PCP into other compounds. PCP is subject to photolytic decomposition in surface water and air, but it still persists for hours or days in air, soils, and surface waters. It is present in fish, but tissue levels are usually low because PCP breaks down in the fish tissues.

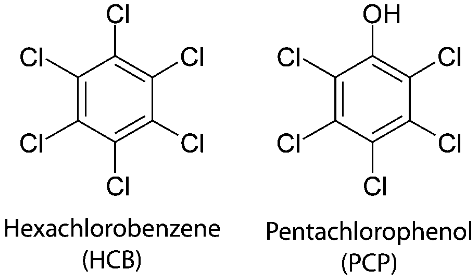

Figure 6 Chemical structures of HCB and PCP.

Figure 7 shows the relative contributions of different pathways to total HCB exposures calculated with CalTOX, compared to those for PCP. We see from this example that the human population is potentially exposed to HCB by a combination of ingestion and inhalation pathways with a minor contribution from dermal contact. Both pollutants can be transferred into foods from the atmosphere (by gaseous exchange and dry and wet deposition of particles), from soils (by uptake in the roots), and from surface water (by direct bioconcentration into aquatic organisms or by irrigation of crops). Detailed pathways analysis indicates exposure is due to eating low levels in contaminated terrestrial food products, eating contaminated fish, drinking milk or eating dairy products or meat from cattle grazing on contaminated pastures, drinking small amounts in contaminated water, breathing low levels in contaminated air, eating or touching contaminated soil, and absorbing small amounts from water while showering and bathing. Human populations are also exposed to PCP by eating contaminated food, such as fish and terrestrial food crops, and by dermal contact with soils and bathing water. PCP is relatively more efficiently transferred from soils into foods than HCB, as indicated by higher potential daily intake from foods grown on contaminated soils. Exposure due to inhalation of contaminated air is a significantly less important pathway than for HCB, as a result of the lower vapor pressure of PCP.

Sustainability and Life-Cycle Impact Assessment

The issues of long-term environmental sustainability, design for environment, and life-cycle impact assessment (LCIA) all require quantitative measures of hazard as weighting factors for pollutant releases. Because the scope of these issues does not allow for full-scale, site-specific risk assessments, analysts rely on generic and regional multimedia models as assessment tools. Generic versions of both CalTOX and USES have been used to conduct comparative assessments of fate and exposure in support of LCIA, sustainability, and comparative risk assessments (78, 79).

MODEL EVALUATION STRATEGIES

Multimedia fate and exposure models support decisions to tolerate, regulate, or monitor existing and new industrial and agricultural chemicals. In this role, fate/exposure models provide prospective analysis of future risk and retrospective analysis of the links between health outcomes and environmental releases. In using models to support regulation and monitoring policies, decision makers struggle with the questions of how likely they are to make unwarranted choices and what the associated health, economic, and political consequences of those choices are. To confront these questions, decision makers rely on modelers to quantify the reliability of their model predictions. Here we document current methods used to assess the performance of multimedia models. We describe the criteria for establishing model reliability and examine methods for model sensitivity and uncertainty analysis. Finally, a process for model evaluation suitable for multimedia fate models is suggested.

As is the case for all models, multimedia models have inherent capabilities and limitations. The limitations arise because models are simplifications of the real system that they describe, and all assessments using the models are based on imperfect knowledge of input parameters. Thus multimedia assessments have inherent uncertainty. This realization provides insight into how the models should be applied and helps decide whether and/or how to make the models more detailed. Confronting the uncertainties requires a model performance evaluation that estimates the degree of uncertainty in the assessment and illustrates the relative value of increasing model complexity by providing a more explicit representation of uncertainties or by assembling more data through field studies and experimental analysis.

Sensitivity and Uncertainty Analyses

Sensitivity and uncertainty analyses are powerful tools for assessing the performance and reliability of models. As applied to mathematical models, sensitivity analysis is quantification of changes in model results as a result of changes in individual model parameters. Uncertainty analysis is the determination of the variation or imprecision in the output function based on the collective variation of the model inputs. A full discussion of sensitivity and uncertainty analysis is provided in the texts by Morgan & Henrion (80) and the volume edited by Saltelli et al. (81). The goal of a sensitivity analysis is to rank input parameters, model algorithms, or model assumptions on the basis of their contribution to variance in the model output. Sensitivity analyses can be either local or global. A local sensitivity analysis is used to examine the effects of small changes in parameter values at some defined point in the range of outcome values. A global sensitivity analysis quantifies the effects of variation in parameters over their entire space of outcome values.

SENSITIVITY OF MULTIMEDIA MODELS Eisenberg et al. (82), Eisenberg & McKone (83), and Hertwich et al. (84) have studied parameter variability and sensitivity in multimedia exposure models. For cumulative exposure assessments based on multimedia models, these studies indicate that output variance arises primarily from chemical-specific input parameters and secondarily from human-exposure factors. Landscape characteristics, such as climate, hydrologic conditions, and soil properties, are generally of minor importance. Among chemical properties, environmental half-lives are the most sensitive properties. Among exposure factors, food intakes are the most sensitive parameters for hydrophobic compounds, water ingestion for water-soluble compounds, and inhalation rates for volatile compounds.

SOURCES OF UNCERTAINTY Uncertainty in model predictions arise from a number of sources: specification of the problem; formulation of the conceptual model; estimation of input values; and calculation, interpretation, and documentation of the results. Of these, only uncertainties due to estimation of input values (parameter uncertainty) can be quantified in a straightforward manner based on variance propagation techniques. Mis-specification of the problem and incorrect model formulation give rise to the wrong models. Having the wrong model results in errors that are potentially large, systematic, and often difficult to discover. As a result, the uncertainties resulting from these errors are potentially much larger and more difficult to characterize than parameter uncertainties. Efforts have been made to assess mis-specification and formulation errors using tools such as decision trees or based on elicitation of expert opinions [for example, the case study by Ragas et al. (85)].

UNCERTAINTY IMPORTANCE AND RANKING A framework for the analysis of uncertainty in environmental models was developed by Morgan & Henrion (80) and Finkel (86) and has been applied by Hertwich et al. (87) to multimedia exposure models. This framework distinguishes among parameter uncertainty, model uncertainty, decision rule uncertainty, and natural variability in any of the parameters and calls for a separate treatment of the different types of uncertainty. More recently, Huijbregts et al. (88) have looked at geographical scenario uncertainty in generic fate and exposure factors. In evaluating parameter uncertainty and variability, Hertwich et al. (87) considered both uncertainty in chemical-specific input parameters as well as the variability in exposure factors and landscape parameters. They determined how the uncertainty and variability of these parameters impact estimates of potential dose for 236 different chemicals. The chemicals were grouped into five dominant exposure medium/route combinations (inhalation of ambient air; ingestion of water, meat, vegetation, or fish; and dermal contact with water). A Monte Carlo analysis was conducted for one representative chemical in each group. From this process, it was determined variance in calculated dose for a specific chemical is typically one to two orders of magnitude. For comparison, the point estimates in the potential dose for the full 236 chemical set spans 10 orders of magnitude. This demonstrates that in spite of the large uncertainties, the potential dose calculations for these chemicals offer a significant information gain relative to a simpler exposure index or the use of toxicity data alone.

Model Evaluation and Confidence Building

A widely held opinion is that the only reliable models are ones that have been validated. However, there continues to be wide disagreement and confusion in the scientific and regulatory communities about what it means to validate a model and if true validation is even possible. Recent papers have made convincing arguments that comparison of model output to observations, or one's view of reality, is not a sufficient measure of acceptability on its own to validate a model (89–93).

Hodges & Dewar (92) have proposed a classification scheme for models that distinguishes between those that can be validated and those that cannot. Models describing systems in which inputs and outputs are all directly and readily measurable and exhibit constancy in structure over time are the only ones that can be validated. Conditions not specified in the model must be constant within the limits of applicability of the model. Validity accrues when predictions made by the model are found accurate for conditions not originally considered when the model was constructed. In contrast, models that describe systems in which the structure is not constant in time or conditions not specified in the model are not constant can never be truly validated. These models cannot be validated, but as Hodges & Dewar point out (92), they are not useless. Some applications for non-validated models include assisting in decision making by stimulating intuition, illustrating an idea, summarizing data, or providing an incentive for improving data quality, and formulating hypotheses for subsequent testing.

Viewed in these terms, it is clear that multimedia models belong to a class of models that cannot be truly validated because the environmental systems and human activities described by these models comprise a system with operative processes that cannot be fully described, that does not exhibit constancy in structure in time, and has features that are not constant within its range of applicability. It is thus impossible to conduct the controlled experiments needed for true validation of multimedia models. For example, a multimedia contaminant fate model might be applied to predict the atmospheric concentration of benzene attributable to a specific source (for example, a refinery). However, this is not a general outcome that can be validated. For a given refinery in a given region, the model can be evaluated by shutting down the refinery and observing the resulting effect on benzene concentrations. However, acceptable model performance in the evaluation exercise does not necessarily mean that the model is generally applicable to describe the contribution of benzene emissions to total atmospheric concentrations of benzene for all refineries in all regions.

As has been pointed out by Oreskes et al. (89), models of this type are common in earth sciences, economics, and engineering as well as in the policy arena, but they cannot be fully verified or validated because descriptions of the operative processes are always incomplete. This limitation does not mean that multimedia models should be exempt from performance evaluation. On the contrary, the fact that the models cannot be validated requires a more thoughtful and systematic process for building confidence among model users. It is possible to build confidence in these models through a series of evaluation exercises, and they can be used to put bounds on the likely range of outcomes. The greater the number and the diversity of confirming observations that can be made, the more probable it is that the conceptualization embodied in the model is not flawed. Confirming observations do not demonstrate the veracity of the model, but they do support the probability that the model is useful and the hypotheses that it represents are not false. Although validity may not accrue with these evaluation exercises, user confidence will increase.

Confidence is further enhanced if the user can easily inspect or verify the operation of the algorithms and data transformations and determine whether the model is internally consistent and contains no logical flaws or technical errors, such as incorrect code implementation. Easy access to the raw data used as inputs, transformed data and the steps of data transformations used in the calculation, and the computer coded algorithms underlying these data transformations will thus enhance user confidence in the model. The availability of clear documentation for model structure and the possibility of performing calibration against an external standard (test data sets) or an internal standard (parallel algorithms to perform the same calculation) all increase user confidence in a model.

For multimedia models in particular, credibility is further enhanced by clearly quantifying the effects of variability and uncertainty in input parameters on model predictions. Communicating the uncertainties associated with contaminant fate and exposure assessments enhances their credibility by highlighting model inputs that control the outcome of the assessment for individual chemicals. Estimates of the uncertainty associated with specific model outputs can be used to inform the decision-making process and direct future refinements of the model or experimental studies to add additional information to the assessment.

Discussion and Conclusions

There is increasing international concern about the presence and possible accumulation of environmentally persistent chemical, physical, and biological pollutants that may threaten the sustainability of ecosystems and the health of humans. These contaminants are often traceable to human activities that support our modern lifestyle such as combustion for energy generation, construction practices, industrial processes, and agriculture activities. Many of the most persistent agents are not contained in a single environmental system, but they are multimedia pollutants that migrate in and between the air, water, soil, sediments, and biota of the environment. Environmentally persistent multimedia agents remind us that all pollutants are, to some extent, multimedia pollutants. These contaminants can accumulate in environmental media that lead to human exposure, e.g., air, indoor dusts, soils, and foodstuffs.

In this review, we described multimedia contaminant fate models at a range of complexity levels and multipathway human exposure models that work in concert with the fate models or with field monitoring data. These models have been useful conceptual tools for understanding the fate and transport processes that determine the behavior of contaminants in the environment, and how they may ultimately contact humans. From a policy perspective, these models can be used to inform decisions aimed at avoiding unacceptable detrimental impacts from development on both human health and ecosystems.

Multimedia models are impossible to truly validate, but they have established a high level of credibility because they are based on sound thermodynamic principles and they have been evaluated in a series of case studies. Our review highlights that there are many limitations on our ability to develop complex and reliable models of contaminant fate and exposure. The environment is incredibly intricate and conditions vary widely both spatially and temporally. Many of the contaminants that are released to the environment are poorly characterized. Properties that control their environmental fate and partitioning range over several orders of magnitude, and in some cases lie near the boundaries of measurable limits, which means there is considerable uncertainty about the nature of these chemicals even under controlled laboratory conditions. Thus, the multimedia modeler must struggle to avoid making a model that has more detail than can be supported by existing theory and data while also including sufficient fidelity to the real system to make reliable classifications about the source-to-dose relationships of environmental chemicals.

New and existing multimedia exposure models are taking on an increasingly important role in the regulatory environment, and as such, we need to ensure their credibility and adequacy. There is a need for novel strategies and tools that take advantage of new biomarker and exposure data to evaluate the performance of source-to-dose models. There is also a need for methods to use existing models to identify the type and quantity of information that would contribute to the greatest reduction in uncertainty in model outcomes. For example, none of the exposure models currently available provide an integrated simulation of major transport processes and indoor/outdoor relationships for toxic substances in air, water, food, and soil.

In looking to the future use of these models, we must address the trade-offs among complexity, reliability, and confidence. As these models evolve and proliferate, we must continue to evaluate their capabilities and limitations. There is an ongoing need to establish and improve upon the confidence placed in these models by decision makers. There is also the opportunity to build more complex and spatially explicit multimedia models. Are these two trends compatible? The increasing capability of personal computers makes possible more complex models, and some equate complexity with credibility. But often the opposite is the case—complexity makes the models much more difficult to verify and evaluate and makes it particularly difficult to assess data limitations. Future multimedia fate and exposure models will have to find an acceptable balance that increases the model's fidelity to the real environmental system through added complexity but retain the reliability and user confidence that have been established by the current generation of models.

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LITERATURE CITED

1. Carson R. 1962. *Silent Spring*. Boston, MA: Houghton Mifflin
2. Jensen S. 1966. Report of a new chemical hazard. *New Sci.* 32:612
3. Counc. Environ. Qual. 1985. *Report on Long-Term Environmental Research and Development*, Exec. Off. Pres. US, CEQ, Washington, DC
4. Eisenbud M. 1963. *Environmental Radioactivity*. New York: McGraw-Hill
5. Eisenbud M. 1973. *Environmental Radioactivity*. New York: Academic. 2nd ed.
6. Eisenbud M. 1987. *Environmental Radioactivity from Natural, Industrial, and Military Sources*. New York: Academic. 3rd ed.
7. Ng YC. 1982. A review of transfer factors for assessing the dose from radionuclides in agricultural products. *Nuclear Saf.* 23:57–71
8. Whicker FW, Kirchner TB. 1987. Pathway: a dynamic food-chain model to predict radionuclide ingestion after fallout deposition. *Health Phys.* 52:717–37
9. UN Sci. Comm. Eff. At. Radiat. 1993. *Sources and effects of ionizing radiation: United Nations Scientific Committee on the effects of atomic radiation. UN UNSCEAR Rep Gen. Assem.*, New York
10. Garrels RM, Mackenzie FT, Hunt C. 1975. *Chemical Cycles and the Global Environment: Assessing Human Influences*. Los Altos, CA: Kaufmann
11. Nriagu JO. 1978. *The Biogeochemistry of Lead in the Environment*. New York: Elsevier/North-Holland Biomed.
12. Nriagu JO. 1979. *The Biogeochemistry of Mercury in the Environment*. New York: Elsevier/North-Holland Biomed.
13. Thibodeaux LJ. 1979. *Chemodynamics, Environmental Movement of Chemicals in Air, Water, and Soil*. New York: Wiley
14. Mackay D. 1979. Finding fugacity feasible. *Environ. Sci. Technol.* 13:1218–23
15. Mackay D, Paterson S. 1981. Calculating fugacity. *Environ. Sci. Technol.* 15:1006–14
16. Mackay D, Paterson S. 1982. Fugacity revisited—the fugacity approach to environmental transport. *Environ. Sci. Technol.* 16:A654–60
17. Bennett BG. 1981. The exposure commitment method in environmental pollutant assessment. *Environ. Monit. Assess.* 1:21–36
18. Bennett BG. 1982. Exposure of man to environmental nickel—an exposure commitment assessment. *Sci. Total Environ.* 22:203–12
19. Bennett BG. 1983. Exposure of man to environmental PCBs—an exposure commitment assessment. *Sci. Total Environ.* 29:101–11
20. Cohen Y. 1986. *Pollutants in a Multimedia Environment*. New York: Plenum
21. Allen DT, Cohen Y, Kaplan IR. 1989. *Intermedia Pollutant Transport: Modeling and Field Measurements*. New York: Plenum
22. Mackay D. 1991. *Multimedia Environmental Models: The Fugacity Approach*. Chelsea, MI: Lewis
23. Natl. Acad. Sci. 1994. *Science and Judgment in Risk Assessment*. Washington, DC: Natl. Res. Counc./Natl. Acad.
24. Cowan CE, Mackay D, Feijtel TCJ, Van De Meent D, Di Guardo A, et al. 1995. *The Multi-Media Fate Model: A Vital Tool for Predicting the Fate of Chemicals*. Pensacola, FL: Soc. Environ. Toxicol. Chem.
25. Mackay D. 2001. *Multimedia Environmental Models: The Fugacity Approach*. Chelsea, MI: Lewis. 2nd ed.

26. McKone TE, Layton DW. 1986. Screening the potential risk of toxic substances using a multimedia compartment model: estimation of human exposure. *Regul. Toxicol. Pharmacol.* 6:359–80
27. McKone TE, Daniels JI. 1991. Estimating human exposure through multiple pathways from air, water, and soil. *Regul. Toxicol. Pharmacol.* 13:36–61
28. McKone TE. 1993. *CalTOX, a multimedia total-exposure model for hazardous-wastes sites part I: executive summary. Rep. UCRL-CR-111456PtI*, Lawrence Livermore Natl. Lab., Livermore, CA
29. McKone TE. 1993. *CalTOX, a multimedia total-exposure model for hazardous-wastes sites part II: the dynamic multimedia transport and transformation model. Rep. UCRL-CR-111456PtII*, Lawrence Livermore Natl. Lab., Livermore, CA
30. McKone TE. 1993. *CalTOX, a multimedia total-exposure model for hazardous-wastes sites part III: the multiple-pathway exposure model. Rep. UCRL-CR-111456PtIII*, Lawrence Livermore Natl. Lab., Livermore, CA
31. Maddalena RL, McKone TE, Layton DW, Hsieh DPH. 1995. Comparison of multimedia transport and transformation models: regional fugacity model vs. CalTOX. *Chemosphere* 30:869–99
32. Natl. Inst. Public Health Environ. Prot. 1994. *Uniform system for the evaluation of substances (USES), version 1.0. Rep. VROM Distrib. 11144/150*, RIVM; Minist. Hous., Spat. Plan. Environ. (VROM); Minist. Welf., Health, Cult. Aff. (WVC), The Hague, Neth.
33. Schwarzenbach RP, Gschwend PM, Imboden DM. 1993. *Environmental Organic Chemistry*. New York: Wiley
34. Karickhoff SW. 1981. Semiempirical estimation of sorption of hydrophobic pollutants on natural sediments and soils. *Chemosphere* 10:833–46
35. Seth R, Mackay D, Muncke J. 1999. Estimating the organic carbon partition coefficient and its variability for hydrophobic chemicals. *Environ. Sci. Technol.* 33:2390–94
36. MacLeod M, Mackay D. 1999. An assessment of the environmental fate and exposure of benzene and the chlorobenzenes in Canada. *Chemosphere* 38:1777–96
37. UN Environ. Progamme. 2001. *Governments Give Green Light to Phase Out of World's Most Hazardous Chemicals. Stockholm Convention on Persistent Organic Pollutants* http://irptc.unep.ch/pops/POPs_Inc/press_releases/pressrel-01/pr5-01.htm
38. Howard PH. 1989. *Handbook of Environmental Fate and Exposure Data for Organic Chemicals*. Chelsea, MI: Lewis
39. Lyman WJ, Reehl WF, Rosenblatt DH. 1982. *Handbook of Chemical Property Estimation Methods*. New York, NY: McGraw-Hill
40. Mackay D, Boethling RS. 2000. *Handbook of Property Estimation Methods for Chemicals:Environmental and Health Sciences*. Boca Raton, FL: Lewis
41. Bailey RE. 2001. Global hexachlorobenzene emissions. *Chemosphere* 43:167–82
42. Natl. Acad. Sci. 1991a. *Frontiers in Assessing Human Exposure to Environmental Toxicants*. Washington, DC: Natl. Res. Counc./Natl. Acad.
43. Natl. Acad. Sci. 1991b. *Human Exposure Assessment for Airborne Pollutants: Advances and Opportunities*, Washington, DC: Natl. Res. Counc./Natl. Acad.
44. US Environ. Prot. Agency. 1992. Guidelines for exposure assessment: notice. *Fed. Regist.* 57:22888–938
45. Lioy P, Pellizzari E. 1996. Conceptual framework for designing a national survey of human exposure. *J. Expo. Anal. Environ. Epidemiol.* 5:425–44
46. Zartarian VG, Ott WR, Duan N. 1997. A quantitative definition of exposure and related concepts. *J. Expo. Anal. Environ. Epidemiol.* 7:411–37
47. US Environ. Prot. Agency. 1989. *Risk Assessment Guidance for Superfund Vol. I, Human Health Evaluation Manual (Part A). Rep. EPA/540/1-89/002*, EPA, Off.

Emerg. Remedial Response, Washington, DC

48. Mackay D, Di Guardo A, Paterson S, Cowan CE. 1996. Evaluating the environmental fate of a variety of types of chemicals using the EQC model. *Environ. Toxicol. Chem.* 15:1627–37
49. van de Meent D. 1996. *SimpleBox: a generic multimedia fate evaluation model. Rep. 672720 001,* Natl. Inst. Public Health Environ. Prot. (RIVM), Bilthoven, Neth.
50. Brandes LJ, Hollander HD, van de Meent D. 1996. *SimpleBox 2.0: a nested multimedia fate model for evaluating the environmental fate of chemicals. Rep. RIVM 719101029,* Natl. Inst. Public Health Environ. Prot. (RIVM), Bilthoven, Neth.
51. UN Environ. Programme. 1997. *Status Report on UNEP's and Other Related Activities on Persistent Organic Pollutants.* http://www.chem.unep.ch/pops/indxhtms/status.html
52. Webster E, Mackay D, Wania F. 1998. Evaluating environmental persistence. *Environ. Toxicol. Chem.* 17:2148–58
53. Bennett DH, Kastenberg WE, McKone TE. 1999. General formulation of characteristic time for persistent chemicals in a multimedia environment. *Environ. Sci. Technol.* 33:503–9
54. Pennington D. 2001. An evaluation of chemical persistence screening approaches. *Chemosphere* 44:1589–601
55. Bennett DH, McKone TE, Matthies M, Kastenberg WE. 1998. Evaluating the spatial range of persistent organic pollutants in a multi-media environment. *Environ. Sci. Technol.* 32:4023–30
56. Beyer A, Mackay D, Matthies M, Wania F, Webster E. 2000. Assessing long-range transport potential of persistent organic pollutants. *Environ. Sci. Technol.* 34:699–703
57. Wania F, Mackay D. 2000. *A comparison of overall persistence values and atmospheric travel distances calculated by various multimedia fate models. Rep. 2/2000,* Wania Environ. Chem. Corp., Toronto, ON. http://www.utsc.utoronto.ca/∼wania/
58. Mackay D, Di Guardo A, Paterson S, Kicsi G, Cowan CE, Kane DM. 1996. Assessment of chemical fate in the environment using evaluative, regional and local-scale models: illustrative application to chlorobenzene and linear alkylbenzene sulfonates. *Environ. Toxicol. Chem.* 15:1638–48
59. Kawamoto K, MacLeod M, Mackay D. 2001. Evaluation and comparison of multimedia mass-balance models of chemical fate: application of EUSES and ChemCAN to 68 chemicals in Japan. *Chemosphere* 44:599–612
60. Priemer DA, Diamond ML. 2002. Application of the multimedia urban model to compare the fate of SOCs in an urban and forested watershed. *Environ. Sci. Technol.* 36:1004–23
61. Wania F, Persson J, Di Guardo A, McLachlan MS. 2000. *Cozmo-Pop. A fugacity-based multi-compartmental mass-balance model of the fate of persistent organic pollutants in the coastal zone. Rep. 1/2000,* Wania Group., Univ. Toronto., Can., http://www.utsc.utoronto.ca/∼wania/
62. Eur. Cent. Ecotoxicol. Toxicol. Chem. 1994. *A Mathematical Model for Use in Risk Assessment of Substances. Rep. Special Rep. N. 8,* ECETOC, Brussels, Belg.
63. Gobas FAPC, Pasternak JP, Lien K, Duncan RK. 1998. Development and field validation of a multimedia exposure assessment model for waste load allocation in aquatic ecosystems: application to 2,3,7,8-tetrachlorodibenzo-P-dioxin and 2,3,7,8-tetrachlorodibenzofuran in the Frazer River watershed. *Environ. Sci. Technol.* 32:2442–49
64. Feijtel T, Boeije G, Matthies M, Young A, Morris G, et al. 1998. Development of a geography-referenced regional exposure assessment tool for European rivers—GREATER. *J. Hazard. Mater.* 61:359–65
65. MacLeod M, Woodfine DG, Mackay D, McKone T, Bennett D, Maddalena R.

2001. BETR North America: a regionally segmented multimedia contaminant fate model for North America. *Environ. Sci. Pollut. Res.* 8:156–63

66. Woodfine DG, MacLeod M, Mackay D, Brimacombe JR. 2001. Development of continental scale multimedia contaminant fate models: integrating GIS. *Environ. Sci. Pollut. Res.* 8:164–72
67. MacLeod M, Woodfine DG, Brimacombe J, Toose L, Mackay D. 2001. A dynamic mass budget for toxaphene in North America. *Environ. Toxicol. Chem.* 21:1628–37
68. Wania F, Persson J, Guardo AD, McLachlan M. 2000. *The POPcycling-Baltic model: a non-steady state multicompartment mass-balance model of the fate of persistent organic pollutants in the Baltic Sea environment. Rep. U-96069,* Nor. Inst. Air Res. (NILU), http://www.utsc.utoronto.ca/∼wania/
69. Wania F, Mackay. D. 1995. A global distribution model for persistent organic chemicals. *Sci. Total Environ.* 160/161:211–32
70. Wania F, Mackay D. 1999. Global chemical fate of alpha-hexachlorocyclohexane. 2. Use of a global distribution model for mass balancing, source apportionment, and trend prediction. *Environ. Toxicol. Chem.* 18:1400–7
71. Scheringer M. 1996. Persistence and spatial range as endpoints of an exposure-based assessment of organic chemicals. *Environ. Sci. Technol.* 30:1652–59
72. Scheringer M. 1997. Characterization of the environmental distribution behavior of organic chemicals by means of persistence and spatial range. *Environ. Sci. Technol.* 31:2891–97
73. Scheringer M, Wegmann F, Fenner K, Hungerbuhler K. 2000. Investigation of the cold condensation of persistent organic pollutants with a global multimedia fate model. *Environ. Sci. Technol.* 34:1842–50
74. Scheringer M, Hungerbuhler K, Matthies M. 2001. The spatial scale of organic chemicals in multimedia fate modeling—recent developments and significance for chemical assessment. *Environ. Sci. Pollut. Res.* 8:150–55
75. McKone TE, Bogen KT. 1991. Predicting the uncertainties in risk assessment: a California groundwater case study. *Environ. Sci. Technol.* 25:1674–81
76. McKone TE. 1993b. The precision of QSAR methods for estimating intermedia transfer factors in exposure assessments. *SAR QSAR Environ. Res.* 1:41–51
77. Natl. Inst. Public Health Environ. 1999. *Uniform system for the evaluation of substances 3.0 (USES 3.0).* RIVM, Minist. Hous., Spat. Plan. Environ. (VROM), Minist. Health, Welf. Sports (VWS), Bithoven, Neth.
78. Huijbregts MAJ, Thissen U, Guinee JB, Jager T, Kalf D, et al. 2000. Priority assessment of toxic substances in life cycle assessment. Part I: calculation of toxicity potentials for 181 substances with the nested multi-media fate, exposure and effects model USES-LCA. *Chemosphere* 41: 541–73
79. Hertwich EG, Mateles SF, Pease WS, McKone TE. 2001. Human toxicity potentials for life-cycle assessment and toxics release inventory risk screening. *Environ. Toxicol. Chem.* 20:928–39
80. Morgan GM, Henrion M. 1990. *Uncertainty: A Guide to Dealing with Uncertainty on Quantitative Risk and Policy Analysis*. Cambridge, UK: Cambridge Univ. Press
81. Saltelli A, Chan K, Scott EM, eds. 2000. *Sensitivity Analysis*. New York: Wiley
82. Eisenberg JNS, Bennett DH, McKone TE. 1998. Chemical dynamics of persistent organic pollutants: a sensitivity analysis relating soil concentration levels to atmospheric emissions. *Environ. Sci. Technol.* 32:115–23
83. Eisenberg JNS, McKone TE. 1998. Decision tree method for the classification of chemical pollutants: incorporation of across chemical variability and within chemical uncertainty. *Environ. Sci. Technol.* 32:3396–404

84. Hertwich EG, McKone TE, Pease WS. 1999. Parameter uncertainty and variability in evaluative fate and exposure models. *Risk Anal.* 19:1193–204
85. Ragas AMJ, Etienne RS, Willemsen FH, van de Meent D. 1999. Assessing model uncertainty for environmental decision making: a case study of the coherence of independently derived environmental quality objectives for air and water. *Environ. Toxicol. Chem.* 18:1856–67
86. Finkel AM. 1990. *Confronting Uncertainty in Risk Management—A Guide for Decision-Makers*. Washington, DC: Resourc. Future
87. Hertwich EG, McKone TE, Pease WS. 2000. A systematic uncertainty analysis of an evaluative fate and exposure model. *Risk Anal.* 20:439–54
88. Huijbregts MAJ, Guinee JB, Reijnders L. 2001. Priority assessment of toxic substances in life cycle assessment. III: Export of potential impact over time and space. *Chemosphere* 44:59–65
89. Oreskes N, Shraderfrechette K, Belitz K. 1994. Verification, validation, and confirmation of numerical-models in the earth-sciences. *Science* 263:641–46
90. Oreskes N, Belitz K, Shraderfrechette K. 1994. The meaning of models—response. *Science* 264:331
91. Oreskes N. 1998. Evaluation (not validation) of quantitative models. *Environ. Health Perspect.* 106:1453–60
92. Hodges JS, Dewar JA. 1992. *Is it you or your model talking: a framework for model validation. Rep. R-4114-AF/A/OSD*, Rand Publ., Santa Monica, CA
93. Beck MB, Ravetz JR, Mulkey LA, Barnwell TO. 1997. On the problem of model validation for predictive exposure assessment. *Stoch. Hydrol. Hydraul.* 11:229–54

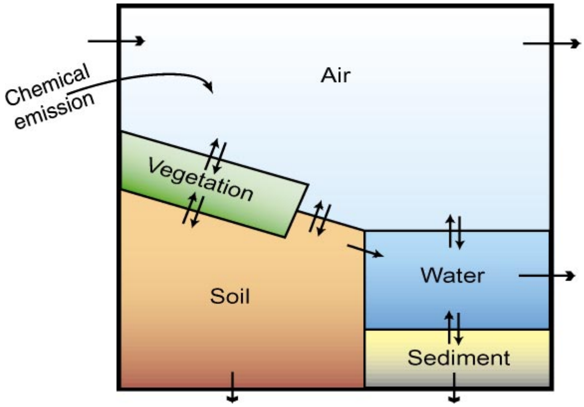

Figure 2 A conceptual multimedia environment. The arrows represent chemical transfer and transport pathways between the compartments.

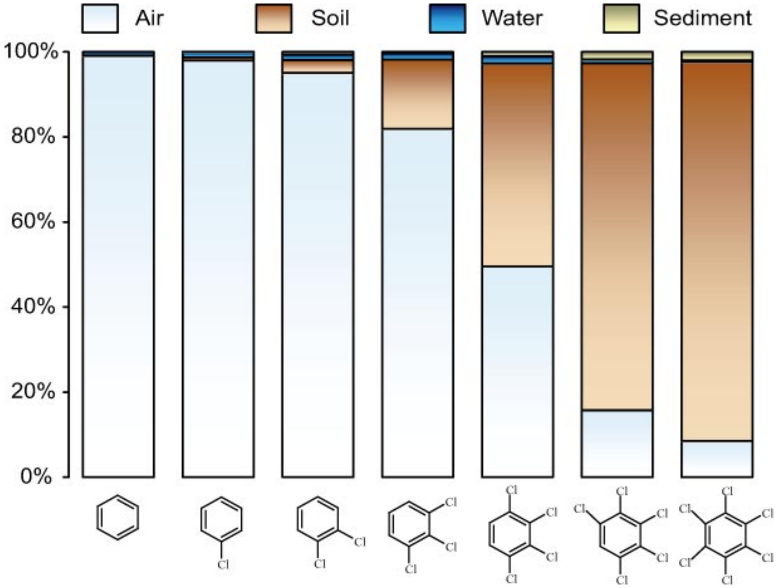

Figure 3 Example of chemical partitioning for the chlorinated benzene series.

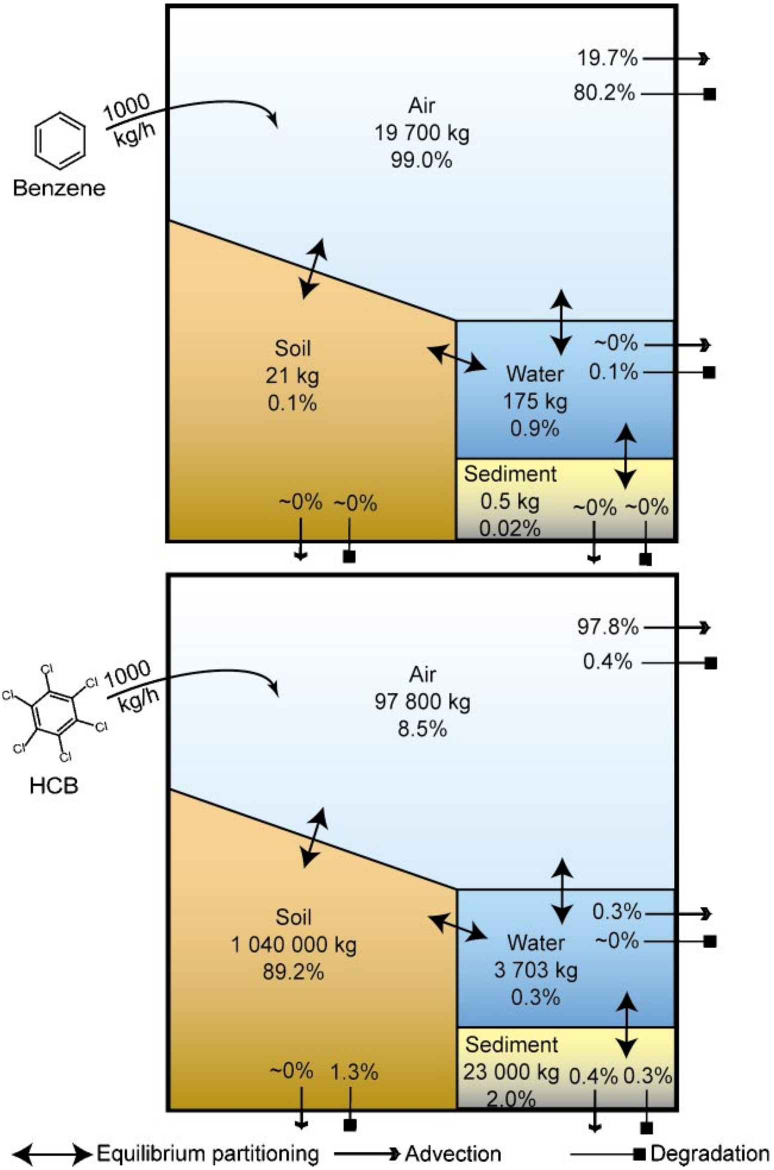

Figure 4 Level II fugacity calculations for benzene and hexachlorobenzene.

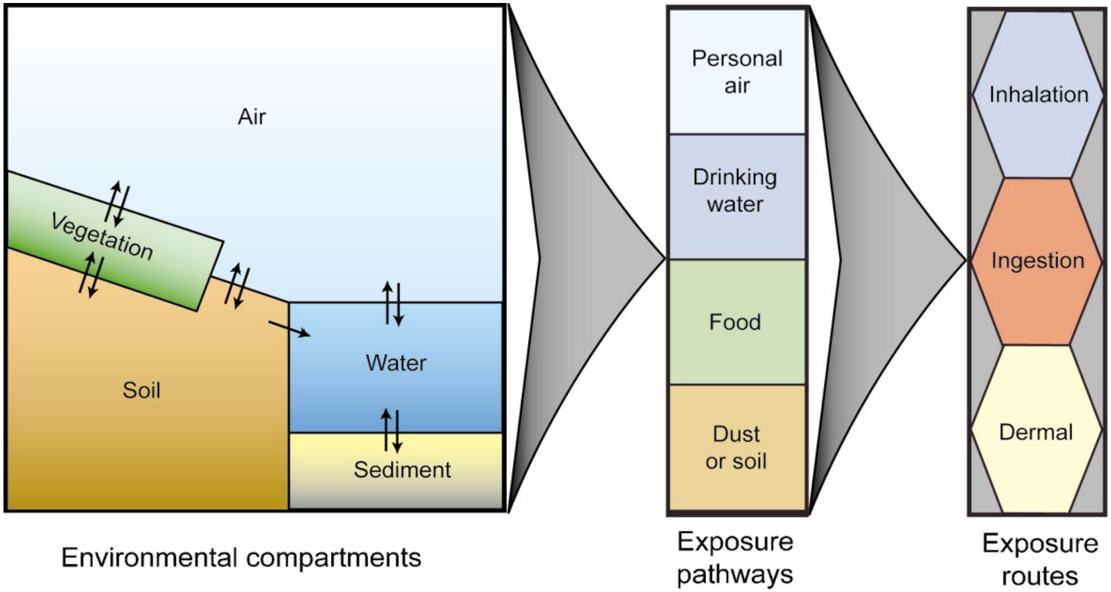

Figure 5 An illustration of the links between emissions, environmental media, and human contact.

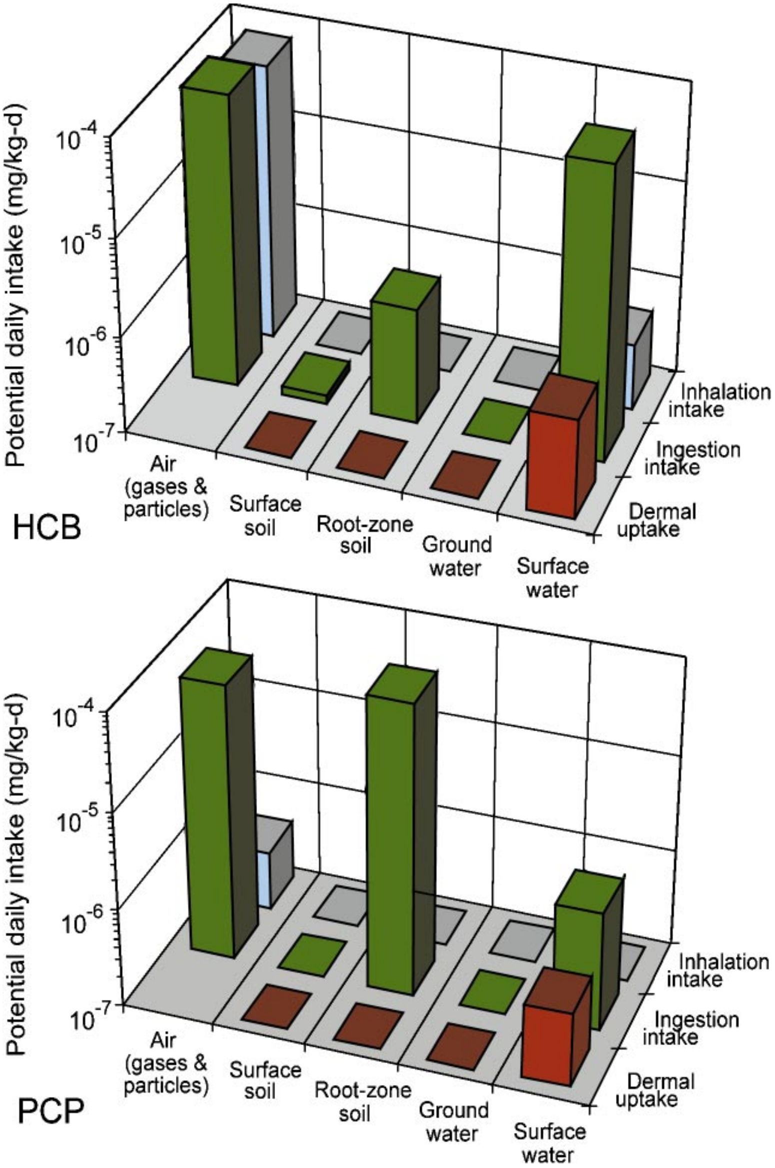

Figure 7 Classification of HCB and PCP intake by exposure route and environmental pathway in the example calculation.