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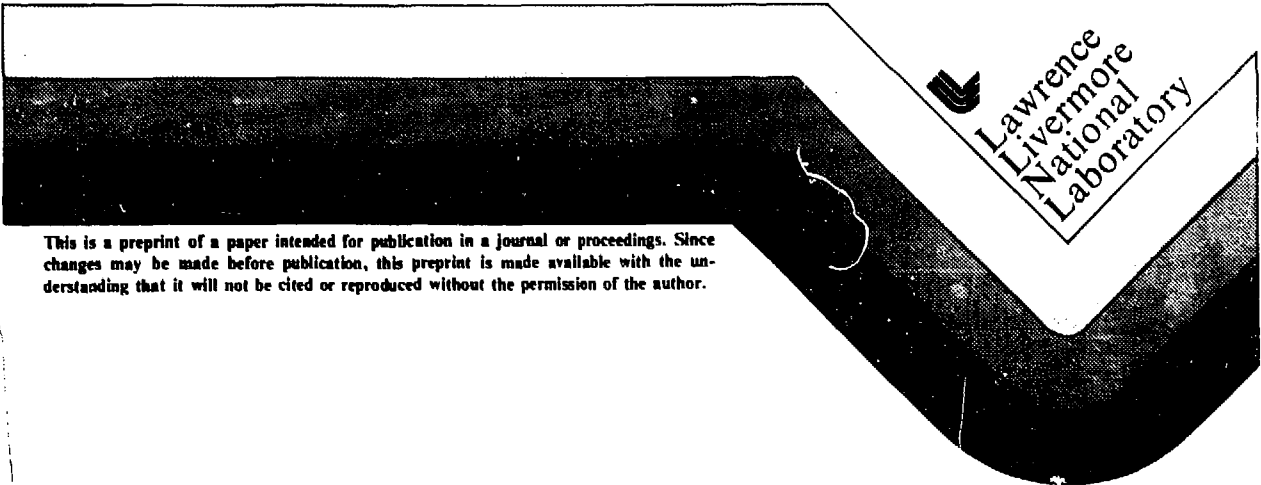
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The Use of Environmental Health-Risk
Analysis for Managing Toxic Substances

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Introduction

The impact of a toxic substance on public health depends upon its inherent toxicity, the size and location of the source term, its behavior once released to the environment and the number and type of exposure pathways. Often the screening of toxic substances is based on quantity and toxicity (for a review of such measures see Smith *et al.*¹). The role of the environment to enhance or mitigate exposures through transport and transformation is neglected. In addition, efforts to model environmental transport often focus on only one or two environmental media. Multiple or competing pathways are not considered. This paper illustrates the use of a generic multi-media model for estimating potential health risks. This approach uses the partitioning, reaction and inter-phase transport characteristics of a chemical in a representative or a generic environment as a basis for health-risk estimates. Such an approach can aid the management of toxic substances in several ways. It is a tool for screening, ranking and preliminary standards setting. It can be used to compare the public health risks of several compounds, to compare the impact of source location (i.e. air, water, soil) for the same compound or interpret the importance of uncertain environmental transport properties. In this paper, I use the approach to compare the magnitude and types of exposures that result from additions to soil of tritium, arsenic and TCDD (2,3,7,8-tetrachlorodibenzo-p-dioxin).

In the sections that follow, the process of environmental health-risk analysis using a multi-media model is developed in three phases. In the first phase, the concept of using compartment models for modelling chemical fate is described. This concept, originally suggested by Baughman and Lassiter,² has been used by others, notably MacKay,³ MacKay and Paterson,^{4,5} and Klopffer *et al.*^{6,7} The cross-media model I use, referred to as GEOTOX, was adapted from a model developed by McKone *et al.*⁸ for ranking the health-risk of heavy metals and radionuclides in a generic global landscape. The second phase involves the development of decay constants and inter-media transfer coefficients for use in the GEOTOX model. These factors are related to the chemical and physical properties of the compound or element being assessed. The work of MacKay and Paterson,^{4,5} Jury *et al.*⁹ and Lyman *et al.*¹⁰ provided the background for this process. In the third phase, environmental concentrations are linked to human exposures and cancer risk using an exposure model that accounts for intake via seven pathways: inhalation; ingestion of drinking water; fruits and vegetables; milk, meat and dairy products; fish; and soil; and dermal absorption. Following the development of the models, there is an applications section in which the models are applied to tritium, arsenic and TCDD. This exercise quantifies the relative health risk associated with steady state concentrations of a radionuclide, a trace element and an organic compound. Environmental data for this exercise come from information compiled for the northeastern and central U.S. by Layton *et al.*¹¹

Development of a Multi-Media Compartment Model

Compartmental systems consist of a finite number of homogenous, lumped

subsystems that exchange a physical quantity (thermal energy, a chemical contaminant, a nutrient, etc.) with each other or with the external environment in a way that the quantities or concentrations of material within the compartments may be described by a set of first-order differential equations. Compartmental systems may be used to model either the kinetics of one substance, in which case the compartments occupy different spaces and the intercompartment transfers represent flows of material from one location to another; or the kinetics of two or more substances (such as a radionuclide and its decay products), in which case different compartments may occupy the same space and some of the intercompartment transfers represent transformation from one substance to another.

The general form of the compartment equations used in the current assessment is:

$$\frac{dN_i^n}{dt} = -\lambda_i^n N_i^n - \sum_{\substack{j=1 \\ j \neq i}}^m T_{ij}^n N_i^n - T_{io}^n N_i^n + \sum_{\substack{j=1 \\ j \neq i}}^m T_{ji}^n N_j^n + S_i^n \quad (1)$$

where

- N_i^n = time varying inventory of species n in compartment i, moles;
- λ_i^n = decay constant for species n in compartment i accounting for radioactive decay, chemical decomposition, etc., yr⁻¹;
- T_{ij}^n = transfer rate of species n from compartment i to compartment j, yr⁻¹;
- T_{io}^n = transfer rate of species n from compartment i to some external system, yr⁻¹;
- S_i^n = source term for the introduction of species n to compartment i, mole/yr; and
- m = number of compartments.

When Eq. 1 is written m times for m compartments, the following matrix equation is obtained

$$\underline{N}(t) = \underline{T} \underline{N}(t) + \underline{S}(t) \quad (2)$$

in which \underline{N} and \underline{S} are vectors of length m and \underline{T} is a matrix of size m by m. A steady-state solution for this system of equations can be obtained by defining a constant source \underline{S} . A time-varying solution to this system is set up by allowing the source vector $\underline{S}(t)$ to be a function of time. A transient solution for the vector $\underline{N}(t)$ is obtained using numerical integration. The chemical fate analysis provides estimates for the transfer coefficients T_{ij}^n and decay constants.

The choice of compartment structure requires a balance between two competing concepts, simplicity of the model and the value of the information provided. Because the model is required to provide a framework for assessing the environmental fate of chemical species, it should, even in its most simple form, include at least three compartments: air, water, and soil. However, because much of the exchange of chemicals occurs at interfaces such as soil/water, soil/atmosphere and water/sediment and because of the role of the surface biota as a possible reservoir for

chemicals, I choose to use a minimum of seven compartments. These are atmosphere, land biota, upper soil layer, lower soil layer, ground water zone, surface water, and surface-water bottom sediments.

Each compartment is made up of at least two and as many as three phases: solid, liquid, and gas (where applicable). A compartment is described by its total mass, total volume, solid mass, liquid mass and gas mass (where applicable). Mass flows among the compartments consist of solid flows and liquid phase flows. Figure 1 illustrates the conceptual layout of the environment as it is viewed by the seven-compartment GEOTOX model. In the paragraphs that follow, I describe how the mass, volume, liquid and solid exchanges among the compartments are obtained.

Atmosphere. This compartment is used to predict the movement and concentration of substances in the lower atmosphere. The GEOTOX model treats chemicals in soil as moving to the lower atmosphere by vapor diffusion or as sorbed to soil particles suspended as windblown dust. The height of the lower atmosphere is 1000 m. The load of dust particles in this region is assumed to be 100 mg/m³ with a residence time of three days.

Land Biota. The GEOTOX model treats chemical turnover in the biosphere in terms of the dry mass production per unit area. It is assumed that the dry mass portion of vegetation (excluding carbon) is derived from soil matter and returns to soil matter as the plant decays.

Upper and Lower Soil Layers. The soil layers are characterized by five parameters: thickness, bulk density, porosity, water content and fraction organic carbon. Typically, the upper soil layer has a lower bulk density, higher porosity, and higher organic carbon content than the lower soil layer. The lower soil layer is the vadose zone between the upper soil layer and the water table. It can be viewed as a filter layer for precipitation passing through the upper soil layer on its way to the groundwater compartment.

Groundwater. This compartment consists of the saturated material of the lowest soil horizon and saturated bedrock. All pore spaces are occupied by water. Water and chemical migration in this zone are typically in a horizontal direction. Groundwater moves in response to gravity from areas of infiltration (recharge) to points of discharge where the water table intersects the land surface. However, transport of substances dissolved in subsurface waters is not at the same rate as groundwater motion. Several processes retard the migration of chemicals in groundwater. Among these processes are absorption, ion exchange, precipitation, colloidal filtration, and irreversible mineralization. Taken together these processes are referred to as "sorption" and accounted for by a first order constant, which expresses the ratio between solid and liquid phase concentrations.

Surface Water and Sediments. These compartments are used to represent the fate of chemicals in lakes, reservoirs, ponds, sloughs, streams and estuaries. The surface water compartment is composed of water, biota, and suspended sediments in chemical equilibrium. Water enters this compartment from run-off, precipitation, and ground water discharge and leaves by

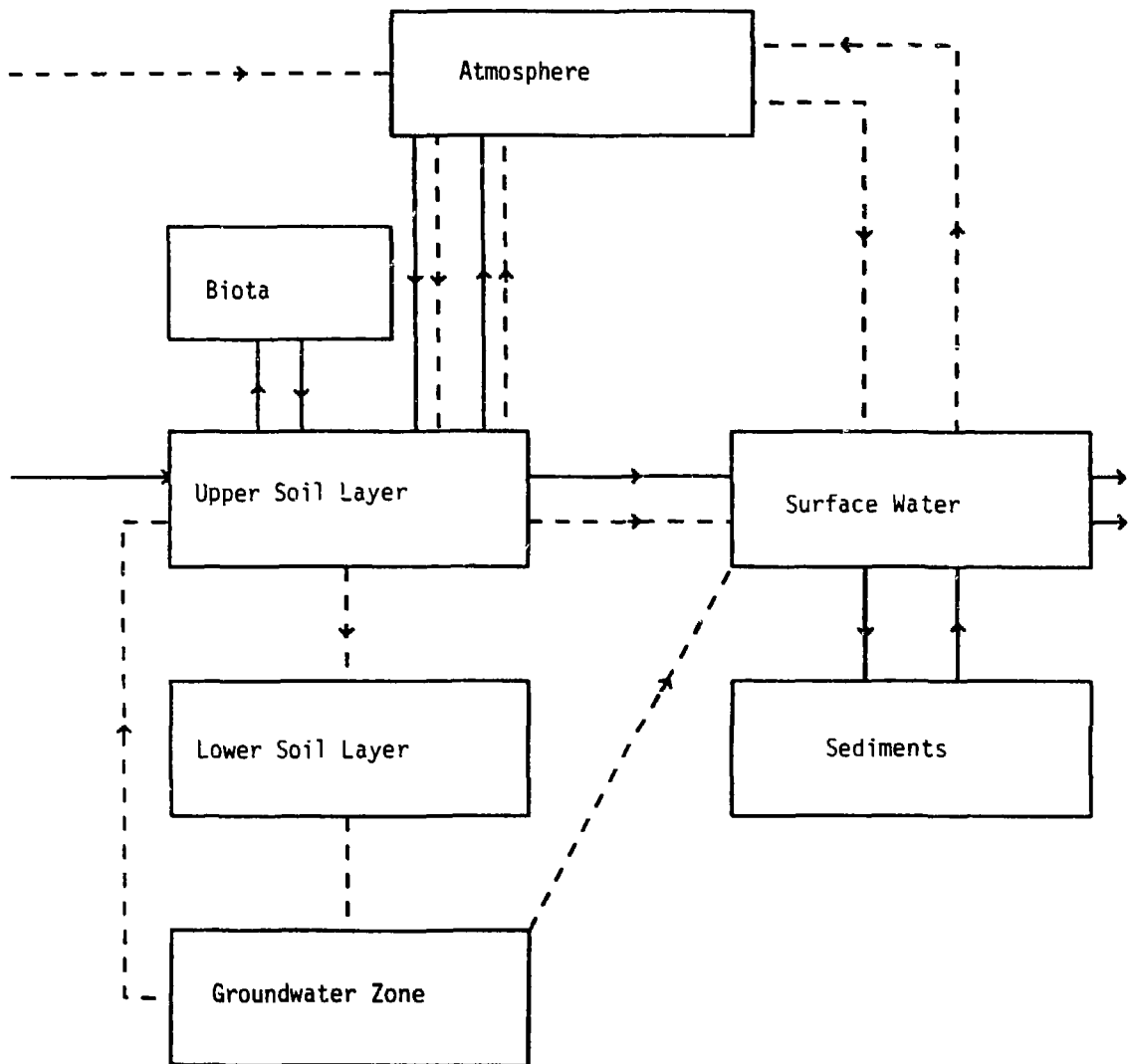


Figure 1. Interactions among the seven compartments in the GEOTOX model. Solid lines refer to solid phase flows, dashed lines to liquid phase flows.

evaporation and downstream flow. Solids enter the compartment from atmospheric deposition and surface erosion and leave by deposition to sediments and downstream advection. The sediment compartment exists at the lower boundary of the surface water compartment. This compartment is composed of bottom sediments and water in a ratio that is determined by the porosity of the sediments. Chemical species move into and out of this compartment by deposition from and resuspension to the overlying surface water and by mass diffusion.

Comparison of exposure potential among a number of chemicals requires the definition of a hypothetical environment or "unit world".³ The masses, volumes and intercompartmental transfers of liquids and solids in the GEOTOX model depend upon a number of landscape parameters. These parameters and the values used for the comparative analysis in this paper are listed in Table I. These properties are taken from a review of landscape characteristics in three regions of the U.S. by Layton *et al.*¹¹ They reflect landscape properties that are characteristic of the central and northeastern United States.

Calculation of Transfer Rates and Decay Constants

Each environmental compartment is made up of at least two and as many as three phases: solid, liquid and/or gas. Chemical exchange among compartments occurs by advection in the solid phase, advection in the liquid phase and/or diffusion. A given chemical species is assumed to be in chemical equilibrium among the three phases of a single compartment. However, there is no requirement for equilibrium between adjacent compartments. As an example of this process consider the upper soil layer. It is composed of three phases: soil water, soil matter, and soil gas. An organic chemical added to the soil distributes itself among these three phases in such a way that chemical and physical equilibrium are achieved. Among the potential chemical pathways from the upper soil compartment are liquid advection as a dissolved component of soil water runoff, solid advection in dust suspensions, and diffusion from soil gas to the atmosphere.

Mathematically, the rate of movement of a chemical from compartment i to compartment j can be represented by the expression:

$$T_{ij} = \alpha_{ij} N_i + \beta_i F_{ij} N_i + \beta_i F_{ij} N_i \quad (3)$$

where

- α_{ij}^n = rate constant that relates the molar inventory of chemical n in compartment i to the rate of mass diffusion from compartment i to compartment j , yr^{-1} ;
- β_i^{nl} = partition coefficient that expresses the fraction of molar inventory of species n in compartment i that is in the liquid phase, kg^{-1} ;
- F_{ij}^l = rate of mass flow of liquid from compartment i to compartment j , kg/yr ;

Table I. Landscape parameters used to determine mass, volume and liquid and solid transfers of the compartments.

Compartment	Parameter	Value
Air	Height	1000 m
	Absolute humidity	6.6 g/m ³
	Precipitation ^a	100 cm/yr
	Residence time of dust particles	3 d
	Ambient temperature	285 K
Biota	Dry mass inventory	3 x 10 ⁷ kg/km ²
	Dry mass production	2 x 10 ⁶ kg/km ² -yr
Upper soil	Thickness	0.25 m
	Bulk density	1.5 kg/L
	Water content	0.45 kg/L
	Air content	0.03 L/L
	Evapotranspiration	58 cm/yr
	Surface run-off	35 cm/yr
	Mechanical erosion	3.0 x 10 ⁵ kg/km ² -yr
Organic carbon fraction	0.02	
Lower soil	Thickness	2.4 m
	Bulk density	2.0 kg/L
	Water content	0.3 kg/L
	Air content	0.02 L/L
	Organic carbon fraction	1 x 10 ⁻³
Groundwater	Water inventory	2 x 10 ⁹ L/km ²
	Rock porosity	0.1
	Rock density	2.3 kg/L
	Irrigation withdrawal	2 cm/yr
	Recharge	5 cm/yr
	Organic carbon fraction	3 x 10 ⁻⁴
Surface water	Fraction of the land surface	0.02
	Average depth	6 m
	Suspended sediment load	340 mg/L
	Sediment-deposition rate	150 kg/m ² -yr
Sediments	Thickness	0.05 m
	Bulk density	1.5 kg/L
	Porosity	0.2
	Sediment resuspension rate	150 kg/m ² -yr
	Organic carbon fraction	0.02

^a 98% of the total precipitation falls on land and 2% falls on surface waters.

β_i^{ns} = partition coefficient that expresses the fraction of species n in compartment i that is in the solid phase, kg^{-1} ; and
 F_{ij} = rate of solid flow from compartment i to j, kg/yr .

The terms F_{ij}^s and F_{ij}^l in Eq. 3 are dependent on landscape parameters, whereas, the α and β terms are dependent on environmental and chemical properties. The assumption of chemical equilibrium among the three phases of a compartment requires that the total molar inventory of a compartment satisfy the following expressions:

$$N_i^n = M_i^s C_i^{ns} + M_i^l C_i^{nl} + V_i^g C_i^{ng} \quad (4)$$

$$N_i^n = \gamma_i^{ns} C_i^{ns} = \gamma_i^{nl} C_i^{nl} = \gamma_i^{ng} C_i^{ng} \quad (5)$$

where

M_i^s, M_i^l = mass of the solid and liquid phases in compartment i, kg;
 C_i^{ns}, C_i^{nl} = concentration of species n in the solid and liquid phases of compartment i, mole/kg;
 V_i^g = volume of the gas phase in compartment i, m^3 ;
 C_i^{ng} = concentration of species n in the gas phase of compartment i, mole/ m^3 ;
 $\gamma_i^{ns}, \gamma_i^{nl}$ = constants relating the concentration of species n in the solid and liquid phase of compartment i to the molar inventory, kg; and
 γ_i^{ng} = a constant relating the concentration of species n in the gas phase of compartment i to the molar inventory, m^3 .

The γ constants are derived from chemical partition factors. They can be interpreted as the effective mass or volume capacity of each phase for containing a chemical distributed in equilibrium among all phases. The distribution of a chemical between solid and liquid can be described with a sorption constant, K_d^n , that relates the amount of chemical sorbed to soil, rock, or sediment to the amount in the water:

$$K_d^n = \frac{C_i^{ns}}{C_i^{nl}} = \frac{\text{mole/kg (solid)}}{\text{mole/kg (liquid)}} \quad (6)$$

For organic compounds the sorption coefficient is the product of the organic carbon fraction of the soil f_{oc} and the organic carbon portion factor K_{oc}^{12} .

The distribution of a chemical between water and air is expressed using the Henry's law constant K_h^n :

$$K_h^n = \frac{C_i^{ng}}{C_i^{nl}} = \frac{P}{RTS} = \frac{\text{mole}/\text{m}^3 \text{ (gas)}}{\text{mole}/\text{kg} \text{ (liquid)}} \quad (7)$$

where

P = vapor pressure of the pure chemical, torr;
R = gas constant, $\text{torr}\cdot\text{m}^3/\text{mole}\cdot\text{K}$;

T = temperature, Kelvins; and
 S = water solubility of the chemical, mole/kg(water).

Combining Eq. 4 through 7 provides a set of expressions for the Y factors in terms of known parameters:

$$\gamma_i^{ns} = M_i^s + M_i^{\ell}/K_d^n + V_i^g K_h^n/K_d^n \quad (8)$$

$$\gamma_i^{nl} = M_i^s K_d^n + M_i^{\ell} + V_i^g K_h^n \quad (9)$$

$$\gamma_i^{ng} = M_i^s K_d^n/K_h^n + M_i^{\ell}/K_h^n + V_i^g \quad (10)$$

By inspection it can be determined that the β factors are simply the inverse of the corresponding Y factors. However, relating the α factor to γ_j^{ng} requires more effort.

The net molar flow (in mole/yr) of a species from the gas phase of one compartment to that of another is represented by the expression

$$\text{Molar Flow} = \frac{AD_G^n}{d} (C_i^{ng} + C_j^{ng}) = \alpha_{ij}^n N_i^n - \alpha_{ji}^n N_j^n \quad (11)$$

where

A = landscape area, m^2 ;
 D_G^n = diffusion coefficient for species n in air, m^2/yr ; and
 d^G = boundary layer thickness, m.

In general, the α terms can be related to A, D_G , d and γ_i^{ng} using Eq. 11 and 5 so that

$$\alpha_{ij}^n = \frac{AD_G^n}{d\gamma_j^{ng}} \quad (12)$$

A similar approach is used for liquid diffusion from bottom sediments to surface water. The boundary layer thickness, d, was assigned a value of 1 cm for atmospheric diffusion and 2 cm for diffusion from bottom sediments (see Layton *et al.*)¹¹.

Decay and transformation processes in the GEOTOX code are modelled as a first-order removal processes. The decay constant λ_i^n in Eq. 1 is used to account for radioactive decay, hydrolysis, photolysis, oxidation and biodegradation.

Human Exposure and Cancer Risk Model

The final step in the screening analysis is to link environmental concentrations to potential human exposure and health risk. Exposure is expressed as daily intake per unit body weight averaged over an individual's lifetime. Two age groups, adult and child, are assumed in the analysis. The general model used to calculate exposure has the form:

$$E = \frac{1}{70 \text{ yr}} \left[\frac{10 \text{ yr}}{17 \text{ kg}} \sum_{i=1}^P I_i^c + \frac{60 \text{ yr}}{70 \text{ kg}} \sum_{i=1}^P I_i^a \right] \quad (13)$$

where

E = lifetime average exposure, for p pathways, ng/kg-d; and
 I_i^C, I_i^A = daily intake by pathway i for a child and adult, ng/d.

This formula reflects the use of steady-state environmental concentrations as a basis for exposure. The lifetime average exposure is not intended to represent the exposure to any specific individual, but to an individual who reflects the physiology, life-style and food consumption of the population for screening purposes.

Seven potential pathways are used in the analysis. These are, 1) inhalation; 2) drinking water; 3) ingestion of fruits, vegetables and grain; 4) ingestion of meat and dairy products; 5) ingestion of fish; 6) ingestion of soil and 7) dermal absorption. Parameters used to describe the reference adult and child for these exposure pathways are given in Table II. Intake of a chemical by inhalation is the product of atmospheric concentration and breathing rate. Intake of chemical through drinking water is the product of water concentration and daily water intake. I assume the drinking water supplies are split equally between surface and ground water. The intake of a chemical species through fruits, vegetables and grains is the product of biota dry mass ingestion, the chemical concentration in soil and the plant/soil partition coefficient k_{ps} . The remaining pathways involve more detailed assumptions that are dealt with in the following paragraphs.

The ingestion of meat and dairy products and of fish present the opportunity for bioaccumulation through the food chain. Meat and dairy produce exposures are based on the intake of chemicals by cattle. For an organic compound such as TCDD, the human intake is the product of daily human intake of animal fat, the rate of intake of the chemical by cattle and the fat/diet partition factor K_{fd} . For other compounds human intake is the product of human ingestion of animal mass, the rate of intake of the chemical by cattle and a meat or milk/diet partition factor K_{md} . It is assumed that cattle ingest 16 kg/d of vegetation, 0.72 kg/d soil and 27 L each of surface and ground water.¹¹ The intake of chemicals through fish is based on the assumption that they reach chemical equilibrium with their surroundings - either surface waters or bottom sediments. The concentration in fish tissue is the product of surface or bottom water concentrations and a bioaccumulation factor, BCF. The daily human intake of fish is assumed to be divided equally between bottom feeders and surface feeders.

Soil ingestion and dermal absorption are direct pathways from soil to humans that are important in the absence of other soil-mediated pathways. Ingestion of soil occurs throughout one's life, but is most significant during childhood. Dermal absorption results from the accumulation of contaminated soil on skin. Dermal absorption of tritiated water from the atmosphere is set equal to the intake by inhalation.¹³ The direct uptake of other compounds from the atmosphere through skin is neglected. Dermal absorption from contaminated soil on the skin is based on an estimate that both a child and adult will daily absorb the contaminant content of roughly 40 mg of soil.¹¹

Table II Parameters used in the exposure model^a.

Parameter	Child	Adult
Age range (yr)	0 - 10	10 - 70
Average weight (kg)	17	70
Inhalation rate (m ³ /d)	10	22
Drinking water intake (L/d)	0.7	1
Ingestion of fruits, vegetables and grains (kg/d) ^b	0.15	0.13
Ingestion of milk and dairy products (kg/d)	0.5	0.3
Ingestion of meat (kg/d)	0.1	0.3
Ingestion of animal fat (kg/d)	0.06	0.17
Ingestion of fish (kg/d)	2.0×10^{-3}	6.5×10^{-3}
Ingestion of soil (kg/d) ^a	1.0×10^{-6}	6.0×10^{-7}

^a Values compiled from ICRP¹⁴ and USNRC¹⁵ except for soil ingestion, which comes from Layton et al.¹¹

^b Dry mass ingestion.

The general procedure for calculating cancer risk from non-radioactive exposure is as follows:

$$\text{Lifetime cancer risk} = 1 - \exp(-\text{potency factor} \times E) \quad (14)$$

The cancer potency factor for TCDD is 0.16 per ng/kg-d¹⁶ and that for arsenic is 1.4×10^{-5} per ng/kg-d.¹⁷ The risk factor for tritium is based on the intake (in Ci) of tritiated water. The NCRP¹⁸ reports the half-time of tritium in humans as approximately 10 days. This results in an equilibrium body burden of 14 μ Ci for each μ Ci/d taken in or a whole-body dose rate of 21 mrad/yr per μ Ci/d uptake. Using the ICRP¹⁹ risk factor of 1.65×10^{-4} per rem for a 70 yr exposure period and a quality factor of unity for the radiation exposure gives a lifetime risk of 2.4×10^{-4} per μ Ci/d uptake.

Applications

The transport and exposure models discussed above are applied here to three species - arsenic, tritium and TCDD - in order to compare their relative environmental health risk. No effort is made to simulate a real environment or exposures to any specific population group. The aim is, rather, to provide information on the characteristic behavior of each substance and the resulting exposure to an individual living their full life and receiving all their air, water and food from the contaminated landscape. The physical chemical properties used for this exercise are listed in Table III.

Table IV shows the results of an analysis in which the environmental risks of the three candidate materials were evaluated in a landscape with properties given in Table I. This exercise considers exposures that result

Table III. Physical chemical properties of Arsenic, TCDD and Tritium.

Properties	Arsenic	TCDD	Tritium
Molecular weight (gm/mole)	74.9	322	20.0 ^a
Vapor pressure (torr)	-- ^b	10 ⁻⁶ c	17.5 ^d
Solubility in water (mole/liter)	--	6.2 x 10 ⁻¹⁰ c	--
Sorption Coefficient, K _d	1300 ^e	--	--
Log K _{ow}	--	6.84 ^c	--
K _{oc}	--	4.3 x 10 ⁶ f	--
K _{ps} plant(DM)/soil	0.1 ^e	3.0 x 10 ⁻⁵ g	1 ^h
K _{fd} fat/diet (cattle)	--	0.92 ⁱ	--
K _{md} meat and milk/diet (cattle)	0.04 ^e	--	1
BCF in fish	75 ^e	10 ^c	1
Diffusion constant in water (m ² /yr)	--	-- ^j	7.7 x 10 ⁻²
Diffusion constant in air (m ² /yr)	--	-- ^j	7.4 x 10 ²
Radioactive half-life (yr)	--	--	12.26
Chemical half-life (yr)	--	12 yr ^k	--

^a This is the molecular weight of tritiated water, THO.

^b The double dash indicates information that is not required for the specific chemical species.

^c From Schaum¹⁶

^d The vapor pressure of water at 20°C is used.

^e Based on data compiled in the Handbook of Geochemistry.²⁰

^f Estimated from the correlation $\log K_{oc} = \log K_{ow} - 0.21$ in Karickhoff et al.²¹

^g Assumes the water content of the plant has the same concentration as soil water, that the plant dry mass contains the same chemical inventory as the fresh mass and that the dry mass is 1/4 the fresh mass.

^h Tritium partitioning is based on fresh mass.

ⁱ $\log K_{fd} = -3.457 + 0.500 (\log K_{ow})$, Kenaga.²²

^j Diffusion of TCDD is not treated because of its low volatility and solubility.

^k This value is from Schaum.¹⁶ Degradation of TCDD is assumed to occur only in soil.

Table IV. Results of the environmental health risk screening analysis.

Arsenic	Exposures in ng/kg-d per mole/km ² introduced to soil			Cancer potency ^a (ng/kg-d) ⁻¹	Environmental health risk ^b (gm/km ² -yr) ⁻¹
	Pathway	Adult	Child		
Inhalation	0.20	0.37	0.22		
Drinking water	0.68	2.0	0.86		
Vegetables, fruit grain	18.0	84.0	27.0		
Meat and dairy	5.3	22.0	7.6		
Fish	1.3	1.6	1.3		
Soil ingestion	0.10	0.73	0.19		
Total ^c	26.	110.	37.	1.4 x 10 ⁻⁵	7.0 x 10 ⁻⁶

Tritium	Exposures in Ci/d per mole/km ² introduced to soil			Cancer potency ^a (ng/kg-d) ⁻¹	Environmental health risk ^b (gm/km ² -yr) ⁻¹
	Pathway	Adult	Child		
Inhalation	9.7	4.4	8.9		
Drinking water	42	29	40		
Vegetables, fruit grain	31	37	32		
Meat and dairy	32	32	32		
Fish	0.38	0.12	0.34		
Dermal absorption	9.7	4.4	8.9		
Total ^d	120	110	120 (1.7) ^e	2.4 x 10 ⁻²	2.0 x 10 ⁻³

TCDD	Exposures in ng/kg-d per mole/km ² introduced to soil			Cancer potency ^a (ng/kg-d) ⁻¹	Environmental health risk ^b (gm/km ² -yr) ⁻¹
	Pathway	Adult	Child		
Inhalation	3.9	7.4	4.4		
Drinking water	0.001	0.003	0.001		
Vegetables, fruits grains	0.001	0.005	0.001		
Meat and dairy	0.66	2.2	0.87		
Fish	2.7	3.4	2.8		
Soil ingestion	0.01	0.08	0.02		
Dermal absorption	0.009	0.03	0.01		
Total	7.3	13.0	8.1	1.56 x 10 ⁻¹	3.9 x 10 ⁻³

^a For tritium the cancer potency is converted to (ng/kg-d)⁻¹ equivalent.

^b By definition, the risk to a hypothetical individual from all pathways corresponding to a source of 1 g/km²-yr uniformly added to the upper soil compartment.

^c Dermal absorption is not considered a pathway for arsenic.

^d Soil ingestion is not considered for tritium.

^e Value in parentheses is the corresponding exposure in ng/kg-d.

from a continuous addition of 1 mole/km² of each species to the soil compartment. Table IV lists adult, child and lifetime average exposures for each pathway assuming that all air, food and water are provided by the contaminated landscape. The last two columns in Table IV provide two screening measures. The first, the cancer potency, is the lifetime risk of cancer, expressed in (ng/kg-d)⁻¹. This is the health risk given that absorption in the human system has occurred. Using this measure it can be concluded that tritiated water is approximately 2000 times more potent than arsenic and that TCDD is roughly 7 times more potent than tritiated water. The second measure, referred to as the "environmental health risk," is the risk per unit flux (g/km²-yr) of the toxin to the generic landscape. This measure combines the health risk with information on exposure pathways, environmental partitioning and persistence. This measure indicates that the environmental risk of tritiated water is 300 times that of arsenic and the environmental risk of TCDD is about 2 times that of tritiated water.

Application of the screening model to three chemically different carcinogens reveals that the environmental health risk does not scale with the cancer potency. The environmental health risk of TCDD relative to tritiated water and arsenic is roughly an order of magnitude less than its cancer potency relative to these compounds. The difference is attributable in large part to the immobility of TCDD relative to tritium and the lower persistence of TCDD compared to arsenic. Because it is a stable element, arsenic continuously added to the landscape can only be removed as a suspended or dissolved sediment in surface water. Also, the environmental risk of tritiated water relative to TCDD is higher than the corresponding cancer potency. This can not be attributable to differences in environmental persistence. They have comparable half-lives. However, TCDD is strongly hydrophobic and partitions within the environment so as to be relatively unavailable. Tritium, on the other hand, behaves like water and becomes available through all exposure pathways as is revealed in Table IV. Thus, the environment is more efficient in providing exposures per unit quantity of tritiated water than it is for TCDD.

Conclusions

This paper presents a set of simple models used to assess health risks based on toxicity, environmental mobility and persistence. These models use a representative landscape in order to describe the steady-state distribution of arsenic, tritiated water, and TCDD as a result of continuous additions to soil. This information is used to assess potential exposures. In a risk-management strategy, this type of information can be used to identify high risk toxins, to perform sensitivity studies or to quantify the value of reducing uncertainties.

Application of the screening model to three chemically different carcinogens reveals that the environmental health risk does not scale with direct measures of toxicity. This is expected. As estimated here, the environmental health risk of TCDD relative to tritiated water and arsenic is roughly an order of magnitude less than its cancer potency relative to these compounds. The difference is attributable in large part to the immobility of TCDD relative to tritium and the lower persistence of TCDD compared to

arsenic.

It should be emphasized that the models presented here are intended as risk screening tools, not simulation models. The purpose of this paper is to present a simple procedure for using the relative behavior of toxic species under prototype conditions as a basis for risk management. The goal is to provide information that is both easy to access and useful for appropriating research and regulatory resources. However, before such procedures can be applied confidently to a broad spectrum of toxic materials, more experiments are needed to verify general model assumptions. If the models become validated in this way, they can function as regionally calibrated tools for assessing the likely magnitude and variability of environmental health risks.

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