Hazardous Waste Risk Assessment

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CHAPTER 5

Risk Assessment Techniques and Methods of Approach

Risk assessment is a process that seeks to estimate the likelihood of occurrence of adverse effects as a result of exposures to chemical, physical, and/or biological agents in humans and ecological receptors within an ecosystem. A number of techniques are available for performing risk assessments; most of the techniques are structured around decision analysis procedures to facilitate comprehensible solutions for even complicated problems.

Risk assessment can be used both to provide a baseline estimate of existing risks attributable to an agent or hazard and to determine the potential reduction in exposure and risk given various corrective actions. Potential risks are estimated by dtermining the probability or likelihood of occurrence of harm, the intrinsic harmful features or properties of specified hazards, the population at risk (PAR), the exposure scenarios, and the extent of expected harm and potential effects. A number of the risk assessment approaches commonly encountered in the literature of hazardous waste management are elaborated further in this chapter.

5.1 THE HEALTH RISK ASSESSMENT METHODOLOGY

Human health risk is the likelihood or probability that a given chemical exposure or series of exposures will damage the health of exposed individuals. Health risk assessment is defined as the characterization of the potential adverse health effects of human exposures to environmental hazards (NRC, 1983). In this process, the extent to which potential receptors have been or could be exposed to selected chemical(s) is determined; the extent of exposure is then considered in relation to the type and degree of hazard posed by the chemical(s), thereby permitting an estimate to be made of the present or potential health risk to the PAR. For hazardous

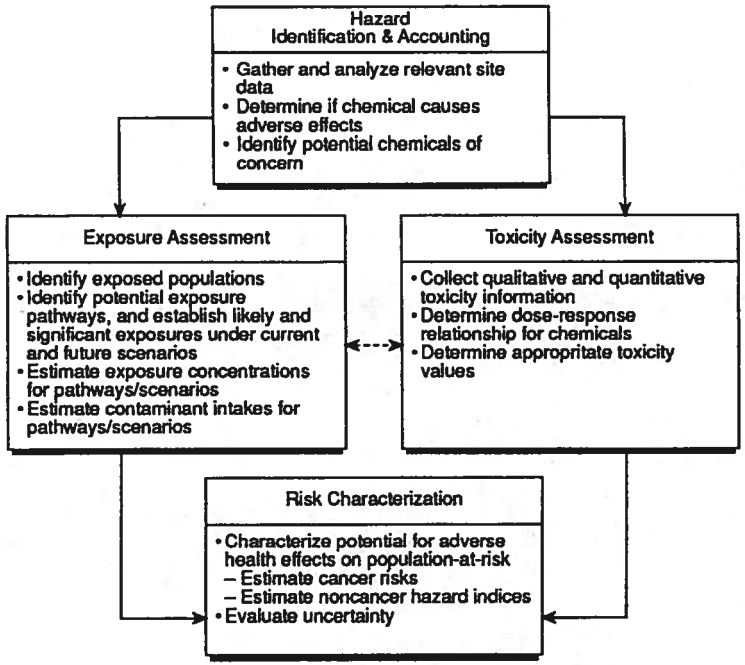

Figure 5.1 Components of the human health and chemical risk assessment process. (Adapted from U.S. EPA, 1989a.)

waste sites, a procedure generally followed for conducting a risk assessment consists of the following elements:

- Definition of the sources of contaminants
- Definition of the contaminant exposure pathways
- Identification of populations potentially at risk from contaminants
- Identification of acceptable exposure levels of contaminants
- Determination of frequency of exposure to potential receptors
- Determination of impacts or damage due to presence and exposures to chemicals

Figure 5.1 shows the basic components and steps used in carrying out a comprehensive health risk assessment. Risk assessment may be performed in response to either short-term (acute) exposures to toxic chemicals, long-term (chronic) exposures, or to combinations of these. It generally requires some level of effort in mathematical modeling, especially with respect to exposure point concentration estimation. Many sources of uncertainty surround the risk assessment process, especially because of possible incomplete exposure assessments, limited and questionable monitoring information, limitations on dose-response assessments, and/or the absence of complete toxicological profiles on some chemicals involved in the assessment.

5.1.1 Hazard Identification and Accounting

Hazard identification is the qualitative evaluation of the potential adverse health impacts of a chemical on potential receptors. The process involves identifying the chemicals of potential concern as well as the specific hazardous properties (such as persistence, bioaccumulative potentials, toxicity, and general fate and transport properties) exhibited. This primary stage of the risk assessment includes a compilation of the lists of all contaminants present at the site, the identification and selection of the chemicals of potential concern for the site, and the compilation of summary statistics for the key constituents selected for further investigation and evaluation.

Data Collection

Contaminants released to the environment are controlled by a complex set of processes including various forms of transport (e.g., intermedia transfers), transformation (e.g., biodegradation), and biological uptake (e.g., bioaccumulation or bioconcentration). Potential primary sources of contaminant release to the various environmental media include the following:

- Atmospheric contamination that may be the results of emissions of contaminated fugitive dusts and volatilization of chemicals
- Surface water contamination, resulting from contaminated runoff and overland flow of chemicals (from leaks, spills, etc.) and chemicals adsorbed to mobile sediments
- Groundwater contamination, as a result of the leaching of toxic chemicals from contaminated soils or vertical migration of chemicals from lagoons and ponds
- Soil contamination, whereby sources of surface soil contamination include intentional placement of wastes on or in the ground, or as a result of spills, lagoon failure, or contaminated runoff; chemicals can also be leached from surface soils to subsurface layers

Secondary sources will include those present in aquatic and terrestrial organisms and biota, due to earlier uptake from the primary sources. Table 5.1 summarizes the important source and target or receiving media associated with typical hazardous waste problems. All the source and impacted media should be thoroughly investigated. In fact, an adequate characterization of the site through implementation of a substantive data collection program is vital for an effective risk assessment; the types of site data and information required include (U.S. EPA, 1989i):

- Contaminant identities
- Contaminant concentrations in the key sources and media of interest
- Characteristics of sources, especially information related to release potential
- Characteristics of the physical and environmental setting that can affect the fate, transport, and persistence of the contaminants

In a typical scenario in which there is a release, contaminants may be transported via one or more media (including air, soils/sediments, surface water, and ground-

usually are all that there is from previous site investigations; these tend to be highly variable, and arithmetic or geometric averaging would not produce representative concentration estimates. Geostatistical techniques that account for spatial variations in concentrations may be employed for estimating the average concentrations at a site required for the long-term exposure assessment (e.g., Zirschy and Harris, 1986; U.S. EPA, 1988). A technique called block kriging is frequently used to estimate soil chemical concentrations in sections of a hazardous waste site in which sparse sampling data do exist. The site is divided into blocks (or grids), and concentrations are determined within blocks by using interpolation procedures that incorporate sampling data in the vicinity of the block. The sampling data are weighted in proportion to the distance of the sampling location from the block. Also, weighted moving-average estimation techniques based on geostatistics are applicable for estimating mean contamination present at a site. Because of the uncertainty associated with any estimate of exposure concentration, the upper confidence limit (i.e., the 95% upper confidence limit) on the average is frequently used in evaluations.

Treatment of Sample Non-Detects

All laboratory analytical techniques have detection limits (DLs) below which only "less than" values may be reported; the reporting of such values provides a degree of quantification. This is important because even at or near their detection limits, the concentration levels of some particular contaminants may be of considerable importance in a risk assessment. However, uncertainty about the actual values below the DL can bias or preclude subsequent statistical analyses. One approach in the calculation of applicable statistical values in a data evaluation involves the use of a value of one half of the sample quantitation limit (SQL) (or simply called the detection limit, DL). Half the DL is usually assumed (as a proxy or estimated concentration) for nondetectable (ND) levels (instead of assuming a value of zero or neglecting such values), provided there was at least one detected value from the analytical results (and/or if there is reason to believe that the chemical is possibly present in the sample at a concentration below the SQL). This method conservatively assumes that some level of the chemical is present and arbitrarily sets that level at $^1/_2$(SQL) (or $^1/_2$[DL]) when it is an ND value. In fact, the U.S. EPA (U.S. EPA, 1989i) suggests the use of the SQL value itself if there is reason to believe that the chemical concentration is closer to this value than to one half the SQL. It should be noted that although these assignments provide a degree of quantification, they may on the other hand considerably affect subsequent analyses and evaluations. Where it is apparent that serious biases may result, more spohisticated analytical and evaluation methods may be warranted.

5.1.2 Exposure Assessment

An exposure assessment is conducted to estimate the magnitude of actual and/or potential human exposures to chemical constituents, the frequency and duration of these exposures, and the pathways by which humans are potentially exposed to

chemicals from a hazardous waste site. The exposure estimates are used to assess whether any threats exist based on existing exposure conditions at or near a potentially contaminated site. Exposure assessment involves describing the nature and size of the population exposed to a substance (i.e., the risk group, which refers to the actual or hypothetical exposed population) and the magnitude and duration of their exposure. Several characteristics of the chemicals of concern will provide an indication of the critical features of exposure, as well as information necessary to determine the distribution, uptake, residence time, magnification, and breakdown of a chemical to new chemical compounds (Hallenbeck and Cunningham, 1988). Indeed, the physical and chemical characteristics of the chemicals can also affect the intake, distribution, half-life, metabolism, and excretion of such chemicals by potential receptors. The evaluation could concern past or current exposures or exposures anticipated in the future. Several techniques may be used for the exposure assessment, including

- Modeling of anticipated future exposures
- Environmental monitoring of current exposures
- Biological monitoring to determine past exposures

The exposure assessment phase of the health risk assessment involves the characterization of the physical and exposure setting, including contaminant distributions leading from sources on the site to the points of exposure; the identification of significant migration and exposure pathways; the identification of potential receptors, or the PAR; the development of site conceptual model(s) and exposure scenarios (including the determination of current and future land uses, and the analysis of environmental fate and persistence); the estimation/modeling of exposure point concentrations for the critical pathways and environmental media; and the estimation of chemical intakes for all potential receptors and significant pathways of concern. The process is used to estimate the rates at which chemicals are absorbed by organisms through all mechanisms including ingestion, inhalation, and dermal absorption. Populations potentially at risk are defined, and concentrations of the chemicals of concern are determined in each medium to which potential receptors may be exposed. Finally, using the appropriate site-specific exposure parameter values, the intakes of the chemicals of concern are estimated.

Exposure Pathways

An exposure pathway is the potential route that contaminants take to reach potential receptors. The route and duration of exposure greatly influences the impact on the receptor. Exposure duration may be short- (acute) or long term (chronic). Exposure pathways are determined by integrating information from the initial site characterization with knowledge about potentially exposed populations and their likely behavior. Table 5.3 indicates examples of the multiple pathways typically observed from hazardous waste sites. The significance of the migration pathway in a particular application is evaluated on the basis of whether the contaminant migration could cause significant adverse human exposures and impacts.

Table 5.3 Potential Multipathway Exposure Routes for Chemical Releases from Hazardous Waste Sites

Transport/Exposure (Contaminated) Medium of Concern	Primary (Direct) Exposure Routes	Examples of Secondary (Indirect) Pathways[a]
Air	Inhalation	Mother's milk
	Dermal absorption	Poultry, meat, and eggs diet
	Crop ingestion (from direct deposition)	Dairy products
Subsurface gas	Inhalation	Mother's milk
		Poultry, meat, and eggs diet
		Dairy products
Soil	Soil ingestion	Crop ingestion (from plant uptake)
	Dermal contact	Poultry, meat, and eggs diet
	Inhalation of particulates	Dairy products
Groundwater	Inhalation of volatiles	Crop ingestion (from plant uptake @ irrigation water)
	Water ingestion	Poultry, meat, and eggs diet (from use of water in feed)
	Dermal absorption	Dairy products (from use of water in feed)
Surface water	Inhalation of volatiles	Fish ingestion
	Ingestion of water	Crop ingestion (from plant uptake @ irrigation water)
	Ingestion of contaminated biota	Poultry, meat, and eggs diet (from use of water in feed)
Aquatic and terrestrial biota	Ingestion @ food chain	Fish ingestion

[a] Secondary pathways of exposure are those which result from assimilation of the chemical into a food source or that reaches the ultimate potential receptor via an intermediary.

An exposure pathway is considered complete only if all of the following elements are present:

- A source of contaminant
- A mode of transport (i.e., a mechanism of chemical release to the environment)
- A contaminant release pathway (including transport media) and exposure route
- Receptor contact at potential exposure points in affected media

For exposure to occur, a complete pathway is necessary, and a risk assessment must address all exposure pathways. The accuracy with which exposure is characterized

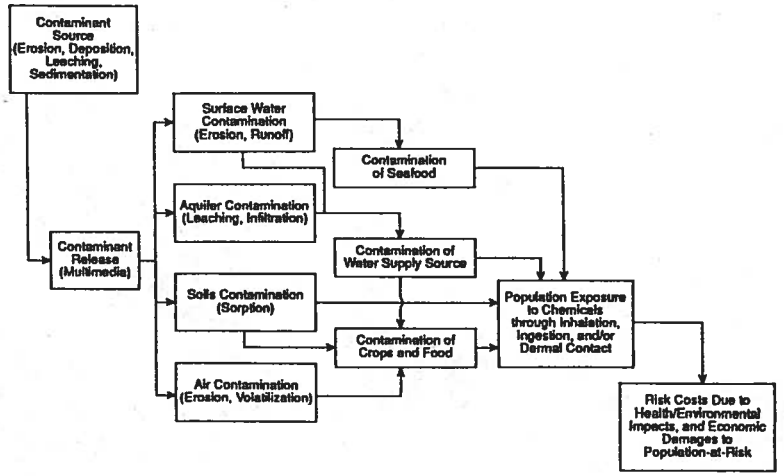

Figure 5.2 A simplified conceptual model for a typical exposure scenario. (Adapted from Asante-Duah, 1990.)

could be a major determinant of the ultimate validity of a risk assessment. Failure to identify an important pathway may seriously detract from the validity of any risk assessment.

Developing Exposure Scenarios

An exposure scenario is the qualitative connection between a contaminant source through one or more environmental media to some receptor population(s). The route of exposure to the population, such as inhalation, ingestion, or dermal contact, is identified as part of the exposure scenario. A schematic representation of the characterization process for developing an exposure scenario is shown, generically, in Figure 5.2. If numerous potential exposure scenarios exist, or if a complex exposure scenario has to be evaluated, it becomes helpful to use the event tree structure to cover potential outcomes and/or consequences; the event tree concept offers an efficient way to develop exposure scenarios. An example of an event tree structure is depicted in Figure 5.3. By using such an approach, the various exposure contingencies can be identified and organized in a systematic manner. Once developed, priorities can be established for focusing the available effort in the aspects of greatest concern. Table 5.4 illustrates the equivalent analytical protocol

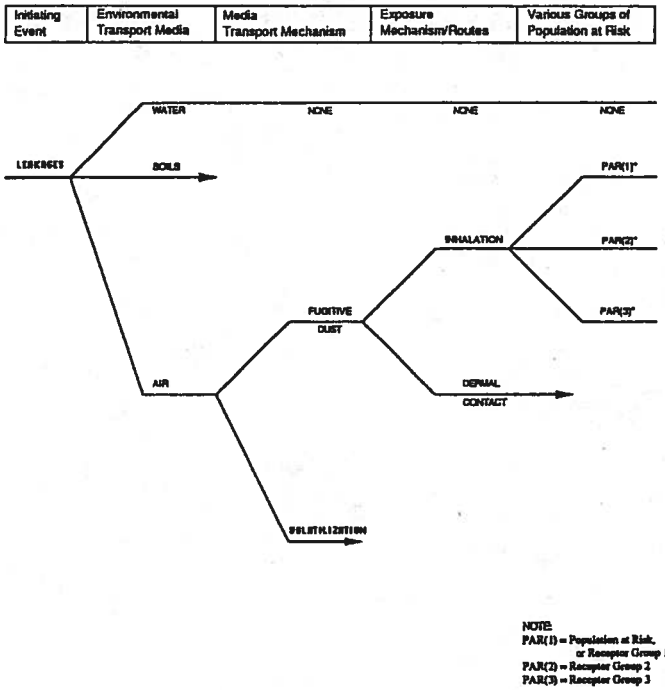

Figure 5.3 Development of exposure scenarios by use of the event tree concept.

for developing the set of exposure scenarios; it is noteworthy that this representation is analogous to the event tree structure.

The exposure scenario associated with a given hazardous situation may be reasonably well defined if the exposure is known to have already occurred; in most cases, however, the risk assessment is being undertaken to evaluate the potential risks due to exposures that may not yet have occurred, in which case hypothetical exposure scenarios are developed for this purpose.

Spectrum of Exposure Scenarios. Exposure scenarios are derived and modeled based on the movement of chemicals in the various environmental compartmental media. A schematic overview of selected processes that includes a conceptualization of typical mechanisms that may affect contaminant migration at a waste site is shown in Figure 5.4. For instance, precipitation may infiltrate the soil onsite and leach contaminants from the wastes and soil as it migrates through the material and the unsaturated soil zone. Infiltrating water may continue its vertical migration and encounter the water table at the top of the saturated zone; the mobilized contaminants may be diluted by the available groundwater flow. Once a contaminant enters the groundwater system, it is possible for it to be transported by groundwater to a discharge point. There also is the possibility of continued vertical migration of contaminants into the bedrock aquifer system. Contaminants may also be carried by surface runoff into surface water bodies. Air releases present additional release pathways. The following *potential* exposure scenarios may be considered representative of the exposure pattern anticipated from a contaminated site:

- Direct human exposure onsite via ingestion of dirt (including pica), inhalation of airborne contaminants, and/or absorption through the skin after dermal contact with contaminated soil.
- Direct human exposure offsite via inhalation of fugitive dust, ingestion of settled dust, and/or dermal contact with chemicals adsorbed onto soil particles.
- Direct human exposure resulting from onsite use of groundwater; exposure may be via ingestion of groundwater used for municipal or local water supplies, inhalation (e.g., during showering activities), and/or dermal contact (from use of the ground water for washing and showering).
- Direct human exposure resulting from offsite use of groundwater; exposure may be via ingestion of groundwater used for domestic water supplies, inhalation (e.g., during showering activities), and/or dermal contact from use of the groundwater for washing and showering.
- Direct human exposure resulting from offsite use of surface water (that has been contaminated from surface runoff and/or ground water discharge); exposure may be via ingestion of surface water, inhalation, and/or dermal contact (from use of the surface water for washing).
- Direct human exposure resulting from off-site recreational use of surface water (that has been contaminated from surface runoff and/or groundwater discharges); exposure may be via ingestion of surface water, inhalation, and/or dermal contact.
- Indirect human exposure resulting from bioaccumulation in *river* fish that is consumed by humans; aquatic life may be exposed to contaminants as a result of runoff and/or groundwater discharges into river(s).
- Indirect human exposure resulting from ingestion of game or livestock (as a result of bioaccumulation through the food chain).
- Indirect human exposure resulting from ingestion of dairy products from cattle that consumed feed and water containing surface residues of chemicals.
- Indirect human exposure resulting from ingestion of crops with bioaccumulated chemicals deposited onto soil, directly onto edible portions of plants, or accumulated through root uptake.
- Inter-human transfers, such as ingestion of human breast milk containing chemicals absorbed by the feeding mother.

atmospheric deposition through the air medium and also through releases of subsurface gas.

As pollutants are released into various environmental media, several factors contribute to their migration and transport. For instance, in the groundwater system, the solutes in the porous media will move with the mean velocity of the solvent by advective mechanism; in addition, other mechanisms governing the spread of contaminants include hydraulic dispersion and molecular diffusion (which is caused by the random Brownian motion of molecules in solution that occurs whether the solution in the porous media is stationary or has an average motion). Furthermore, the transport and concentration of the solute(s) are affected by reversible ion exchange with soil grains; the chemical degeneration with other constituents; fluid compression and expansion; and in the case of radioactive wastes, by the radioactive decay.

Estimation of Potential Receptor Exposures

Using applicable exposure models based on the physicochemical properties of the contaminants of potential concern, and conservative but realistic assumptions regarding contaminant migration and equilibrium partitioning, exposure to potential receptors may conservatively be estimated according to the following generic relationship:

$$EXP = IF \times C_m \times EDF \times CF$$

where

EXP = receptor dose or exposure (mg/kg/day)
IF = intake factor(s) (e.g., inhalation rate—m^3/day; ingestion rate—mg/day, L/day)
C_m = concentration in media (e.g., $\mu g/m^3$ or mg/L or mg/kg)
EDF = exposure duration factor(s)
CF = conversion factor

Values for IF and EDF may be obtained from standard reference manuals and documents (e.g., U.S. EPA, 1989b). These values are derived from information relating to the MEL. The MEL is used to calculate levels of a compound in an environmental media (i.e., soil, water, or air) such that its value is not likely to be exceeded during the course of specified categories of human activity and/or exposures. MEL values may be drawn from several sources, such as the guidelines for ADIs; virtually safe doses (VSDs); MCLs for drinking water; MCLGs for drinking water; threshold limit values (TLVs) for occupational exposures; the Food and Drug Administration (FDA) guidelines for concentrations in foods; no observed adverse effect levels (NOAELs); and acute toxicity values (e.g., LC_{50}, LD_{50}). The concentration in the various media, C_m, may be obtained from field measurements, estimated by simple mass balance analyses or other appropriate

contaminant transport models, or may be determined from equilibrium and partitioning relations. For instance, considering a situation in which groundwater is feeding into a surface water body, the concentration of a chemical in the surface water may be related to its concentration in groundwater by the following mass balance relationship:

$$C_{sw} = C_{gw}\left\{\frac{Q_{gw}}{\left(Q_{gw} + Q_{sw}\right)}\right\}$$

where

C_{sw} = concentration in surface water (mg/L)
C_{gw} = concentration in groundwater (mg/L)
Q_{gw} = flow rate of groundwater (cfs)
Q_{sw} = flow rate of surface water (cfs)

The site-related exposure point concentrations are determined once the exposure scenarios and potentially affected populations are identified. If the transport of compounds associated with the site is under steady-state conditions, monitoring data are generally adequate to determine potential exposure concentrations. If there are no data available, or if conditions are transient (such as pertains to a migrating plume in groundwater), models are better used to predict concentrations. Many factors — including the fate and transport processes affecting the chemicals of concern — must be considered when selecting the most appropriate model. In any case, in lieu of an established trend in historical data indicating the contrary, the site is considered to be in steady state with its surroundings.

Since exposure could be occurring over long time periods (say, up to a lifetime of about 70 years), it is important in a detailed analysis to consider whether degradation or other transformation of the chemical at the source would occur. In such cases, the chemical and its degradation properties should be reviewed. If significant degradation is likely to occur, exposure calculations become much more complicated. In that case, source contaminant levels must be calculated at frequent intervals and summed over the exposure period. Assuming first-order kinetics for instance, an approximation of the degradation effects can be obtained by multiplying the concentration by a degradation factor, DGF, defined by

$$DGF = \frac{\left(1 - e^{-kt}\right)}{kt}$$

where

k = chemical-specific degradation rate constant (d^{-1})
t = time period over which exposure occurs (d)

For a first-order decaying substance, k is estimated from the following relationship:

$$t_{1/2}[days] = \frac{0.693}{k} \text{ or } k[days^{-1}] = \frac{0.693}{t_{1/2}}$$

where $t_{1/2}$ is the half-life, which is the time after which the mass of a given substance will be one half its initial value. It should be recognized in carrying out all these manipulations, however, that in many cases when a substance undergoes degradation, it produces an end product that could be of potentially equal or greater concern (such as is the case when trichloroethylene biodegrades to vinyl chloride). Consequently, for simplicity, the decay factor will normally be ignored, except in situations where the end product is known to present no potential hazards to potential receptors.

Intake and Dose Calculation. Once exposure point concentrations in all media of concern have been estimated, the intakes and/or doses to potentially exposed populations need to be determined. Intake is defined as the amount of chemical coming into contact with the receptor's body or exchange boundaries (such as the skin, lungs, or gastrointestinal tract), and dose is the amount of chemical absorbed by the body into the bloodstream.

The absorbed dose differs significantly from the externally applied dose (called exposure or intake). Intakes and doses are normally calculated in the same step of the exposure assessment, where the former multiplied by an absorption factor yields the latter value. The methods by which each type of exposure is estimated are well documented in the literature of exposure assessment (e.g., DOE, 1987; U.S. EPA, 1988h; 1989b; 1989i; CAPCOA, 1990). The general equation for calculating chemical intakes by the PAR is expressed by the following relationship:

$$I = C \times CR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

I = intake, adjusted for absorption (mg/kg/day)
C = chemical concentration in media of concern (e.g., mg/kg; mg/L)
CR = contact rate (e.g., mg soil/day; liters water/day)
CF = conversion factor
FI = fraction of intake from contaminated source (unitless)
ABS_s = bioavailability/absorption factor (%)
EF = exposure frequency (d/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged — days)

Appendix B1 of this book contains a more detailed presentation of the exposure estimation equations. For a comprehensive evaluation, standard parameters suggested by the U.S. EPA are used, with appropriate and justifiable adjustments being made for problem-specific cases.

For each exposure pathway under consideration, an intake per event is developed. This value quantifies the amount of a chemical contacted during each exposure event, where "event" may have different meanings depending on the nature of the scenario under consideration (e.g., each d of inhalation of the chemical in air constitutes one inhalation exposure event). The quantity of a chemical absorbed into the bloodstream per event, represented by the dose, is calculated by additionally considering any pertinent physiological parameters (such as gastrointestinal absorption rates). When the level of dose (systemic absorption) from an intake is unknown, or cannot be estimated by a defensible argument, intake and dose are considered to be the same (i.e., 100% absorption into the bloodstream from contact is assumed). This approach provides a conservative estimate of the actual exposures. It assumes that the potential receptor is always in the same location, exposed to the same ambient concentration, and that there is 100% absorption on exposure. This would hardly ever represent any real-life situation, and lower exposures will be expected due to the fact that potential receptors will generally be exposed to lower or even near-zero levels for the times spent outside the "zones of influence."

Event-based intake values are converted to final intake values by multiplying the intake per event by the frequency of exposure events over the time frame being considered. Long-term (chronic) exposures are based on the number of events that are assumed to occur within an assumed 70-year lifetime (U.S. EPA, 1989i). Chronic daily intake (CDI) is the projected human intake over the long-term period and is calculated by multiplying the average, or the reasonably maximum exposure (RME) media concentrations, by the human intake and body weight factors. Subchronic daily intake (SDI), on the other hand, is the projected human intake over a short-term period such as only a portion of a lifetime (U.S. EPA, 1989i) and is also calculated by multiplying the RME (for the maximum or worst-case scenario) media concentrations by the human intake and body weight factors.

SDIs are used to evaluate subchronic noncarcinogenic effects, and CDIs are used to evaluate both carcinogenic risks and chronic noncarcinogenic effects. In general, the carcinogenic effects (and sometimes the chronic noncarcinogenic effects) from a contaminated site involve estimating the lifetime average daily dose (LADD). For noncarcinogenic effects, the average daily dose (ADD) is generally used. The ADD differs from the LADD in that the former is not averaged over a lifetime; rather, it is the average daily dose pertaining to the d of exposure. The maximum daily dose (MDD) will typically be used in estimating acute or subchronic exposures.

5.1.3 Toxicity Assessment

A toxicity assessment is conducted as part of a health risk assessment to qualitatively and quantitatively assess the potential for adverse human health effects from exposure to the chemicals of potential concern. The quantitative portion of the toxicity assessment entails identifying the relevant toxicity indices against which exposure point intakes and doses can be compared during the risk

characterization stage of the overall assessment. Such assessment may include a consideration of experimental studies that uses animal data for extrapolation to humans, as well as epidemiological studies. The qualitative aspect of the assessment includes summaries of the adverse human health effects, typical environmental levels or background concentrations, toxicokinetics, toxicodynamics, and ecotoxicology associated with each chemical of potential concern.

The toxicity assessment component of the risk assessment considers the types of adverse health effects associated with chemical exposures, the relationship between magnitude of exposure and adverse effects, and related uncertainties such as the weight of evidence of the carcinogenicity of a particular chemical in humans (Appendix C). A detailed toxicity assessment for chemicals found at contaminated sites is generally accomplished in two steps (1) hazard assessment; and (2) dose-response assessment. These steps are briefly discussed below. Appendix D in this book describes selected information sources for toxicity parameters.

Typically, risk assessments rely heavily on existing toxicity information developed for specific chemicals. Where toxicity information does not exist, decisions can be made to exclude the chemical from the evaluation or to estimate toxicological data from that of similar compounds (with respect to molecular weight and structure-activity). Structure-activity analysis is a technique which can be applied to derive an estimate for the toxicity of a chemical when direct experimental or observational data are lacking.

Hazard Assessment

Hazard assessment is the process of determining whether exposure to an agent can cause an increase in the incidence of an adverse health effect (e.g., cancer, birth defects, etc.), involving characterization of the nature and strength of the evidence of causation. It involves gathering and evaluating data on the types of health injury or disease that may be produced by a chemical and on the conditions of exposure under which injury or disease is produced. Hazard assessment may also involve characterizing the behavior of a chemical within the body and the interactions it undergoes with organs, cells, or even parts of cells. Data of the latter types may be of value in answering the ultimate question of whether the forms of toxicity known to be produced by a substance in one population group or in experimental settings, are also likely to be produced in all humans.

Methods commonly used for assessing the hazardous nature of substances include (Lave, 1982)

- Case clusters
- Structural toxicology
- Laboratory study of simple test systems
- Long-term animal bioassays
- Human (epidemiological) studies

Case clusters are based on the identification of an abnormal pattern of disease. This procedure tends to be more powerful in identifying hazards, especially when the

resulting condition is extremely rare; when the health condition is more common in the general population, the method is not very powerful. Since the PAR is essentially never known in detail, the case cluster method necessarily yields no conclusive evidence, only rather vague suspicions. Structural toxicology involves searching for similarities in chemical structure that might identify carcinogens. The close association between mutagens and carcinogens lead to a general presumption that mutagenic substances are also carcinogenic. Animal bioassays are laboratory experimentations, generally with rodents; statistical models are used to extrapolate from animal bioassays to humans. Epidemiology is a more scientific, systematic form of case cluster analysis with an attempt to control for confounding factors in the experimental design or statistical analysis.

Dose-Response Assessment

Dose-response assessment is the process of quantitatively evaluating the toxicity information and characterizing the relationship between the dose of the contaminant administered or received (i.e., exposure to an agent) and the incidence of adverse health effects in the exposed populations. It is the process by which the potency of the compounds is estimated by use of dose-response relationships. For carcinogens, this involves estimating the probability that an individual exposed to a given amount of chemical will contract cancer due to that exposure. Potency estimates may be given as "unit risk factor" (in $\mu g/m^3$ or ppm) or as "potency slopes" (in units of $[mg/kg/day]^{-1}$). Data are derived from animal studies or, less frequently, from studies in exposed human populations.

The risks of a substance cannot be ascertained with any degree of confidence unless dose-response relations are quantified, even if the substance is known to be toxic. There may be many different dose-response relationships for a substance if it produces different toxic effects under different conditions of exposure. Dose-response curves are functional relationships between the amounts of chemical substance and its morbidity/lethality. The response of a toxicant depends on the mechanism of its action; in the simplest case, the response, R, is directly proportional to its concentration, [C], so that

$$R = k[C]$$

where k is a rate constant. This would be the case for a pollutant that metabolizes rapidly, but even so, the response and the value of the rate constant would tend to differ for different risk groups of individuals and for unique exposures. If the toxicant accumulates in the body, the response is defined as

$$R = k[C]t^n$$

where t is the time and n is a constant. For cumulative exposures, the response would generally increase with time. Thus, the cumulative effect may show as linear until a threshold is reached, after which secondary effects begin to affect and enhance the

responses. The cumulative effect may be related to what is referred to as the "body burden" (BB). The body burden is determined by the relative rates of absorption (ABS), storage (ST), elimination (ELM), and biotransformation, (BT) (Meyer, 1983):

$$BB = ABS + ST - ELM - BT$$

Each of the factors involved in the quantification of the body burden is dependent on a number of biological and physiochemical factors. In fact, the response of an individual to a given dose cannot be truly quantitatively predicted, since it depends on many extraneous factors such as general health and diet of individual receptors or the PAR.

Three major classes of mathematical extrapolation models are often used for relating dose and response in the subexperimental dose range:

1. Tolerance distribution models, including Probit, Logit, and Weibull
2. Mechanistic models, including One-hit, Multi-hit, and Multi-stage
3. Time-to-occurrence models, including Log-normal and Weibull

Indeed, other independent models — such as linear, quadratic, and linear-cum-quadratic — may also be employed for this purpose. The details of these are beyond the scope of this book and are discussed elsewhere (e.g., CDHS, 1986). From the quantitative dose-response relationship, toxicity values are derived that can be used to estimate the incidence of adverse effects occurring in humans at different exposure levels.

Quantifying Toxicological Effects: Toxicity Parameters

Chemicals that give rise to toxic endpoints other than cancer and gene mutations are often referred to as "systemic toxicants" because of their effects on the function of various organ systems; the toxic endpoints are referred to as "noncancer or systemic toxicity." Most chemicals that produce noncancer toxicity do not cause a similar degree of toxicity in all organs, but usually demonstrate major toxicity to one or two organs. These are referred to as the target organs of toxicity for the chemicals (Klaassen et al., 1986; U.S. EPA, 1989d). In addition, chemicals that cause cancer and gene mutations also commonly evoke other toxic effects (i.e., systemic toxicity).

For the purpose of health risk assessment, chemicals are usually categorized into carcinogenic and noncarcinogenic groups. Noncarcinogens operate by threshold mechanisms, i.e., manifestation of systemic effects requires a threshold level of exposure or dose to be exceeded. Systemic toxicity is generally treated as if there is an identifiable exposure threshold below which there are no observable adverse effects. Chronic noncarcinogenic health effects are assumed to exhibit a threshold level — i.e., continuous exposure to levels below the threshold produce no adverse or noticeable health effects. This characteristic distinguishes systemic endpoints

from carcinogenic and mutagenic endpoints, which are often treated as nonthreshold processes. The threshold principle is not applicable for carcinogens, since no thresholds exist for this group. It is noteworthy, however, that among some professional groups, there is the belief that certain carcinogens require a threshold exposure level to be exceeded to provoke carcingenic effects.

Often, it becomes necessary to compare receptor intakes of chemicals with doses shown to cause adverse effects in humans or experimental animals. This can be achieved by estimating the MDD and LADD resulting from environmental exposures (expressed in mg/kg/day). The dose at which no effects are observed in human populations or experimental animals is referred to as the "no observed effect level" (NOEL). Where data identifying a NOEL are lacking, a "lowest observed effect level" (LOEL) may be used as the basis for determining safe threshold doses. For acute effects, short-term exposures/doses shown to produce no adverse effects are involved; this is called the "no observed adverse effect level" (NOAEL).

Traditionally, risk decisions on systemic toxicity have been made using the concept of "acceptable daily intake" (ADI) derived from an experimentally determined NOAEL. The ADI is the amount of chemical (in mg/kg body weight/day) to which a receptor can be exposed to on a daily basis over an extended period of time — usually a lifetime — without suffering a deleterious effect. A NOAEL is an experimentally determined dose at which there has been no statistically or biologically significant indication of the toxic effect of concern. In cases where a NOAEL has not been demonstrated experimentally, the term "lowest observed adverse effect level" (LOAEL) is used. For chemicals possessing carcinogenic potentials, the LADD is compared with the NOEL identified in the long-term bioassay experimental tests; for chemicals with acute effects, the MDD is compared with the NOEL observed in short-term animal studies. In assessing the chronic and subchronic effects of noncarcinogens and also noncarcinogenic effects associated with carcinogens, the experimental dose value (e.g., NOEL values) is typically divided by a safety (or uncertainty) factor to yield an RfD (or ADI).

For exposure of humans to noncarcinogenic effects of chemicals, the ADI is used as a measure of exposure considered to be without adverse effects. For carcinogenic effects, and assuming a no-threshold situation, an estimate of the excess cancer per unit dose, called the unit cancer risk (UCR), or the cancer slope factor (SF) is used. Overall, the quantitative evaluation of toxicological effects consists of the following specific steps:

- Compilation of toxicological profiles (including the intrinsic toxicological properties of the chemicals of concern, which may include their acute, subchronic, chronic, carcinogenic, and/or reproductive effects)
- Determination of appropriate toxicity indices (e.g., ADIs, RfDs, and SFs [or cancer potency factors, CPFs])

The toxicity parameters are dependent on the route of exposure; however, oral RfDs and SFs will normally be used for both ingestion and dermal exposures.

Derivation of Reference Doses (RfDs) or Acceptable Daily Intakes (ADIs)

RfD (or the ADI) is defined as the maximum amount of a chemical that the human body can absorb without experiencing chronic health effects; it is generally expressed in units of milligrams per kilogram of bodyweight per d (mg/kg/day). Although often used interchangeably with the ADI, the RfDs are based on a more rigorously defined methodology. It is an estimate of continuous daily exposure of a noncarcinogenic substance for the general human population (including sensitive subgroups) which appears to be without an appreciable risk of deleterious effects. Subchronic RfD is used to refer to cases involving only a portion of the lifetime, whereas chronic RfD refers to lifetime exposures. The RfD is a "benchmark" dose operationally derived from the NOAEL by consistent application of general "order-of-magnitude" uncertainty factors (UFs) (also called "safety factors") that reflect various types of data sets used to estimate RfDs. In addition, a modifying factor (MF) is sometimes used which is based on professional judgment of the entire data base of the chemical. More generally stated, RfDs (and ADIs) are calculated by dividing a NOEL (i.e., the highest level at which a chemical causes no observable changes in the species under investigation), a NOAEL (i.e., the highest level at which a chemical causes no observable adverse effect in the species being tested), or a LOAEL (i.e., that dose rate of chemical at which there are statistically or biologically significant increases in frequency or severity of adverse effects between the exposed and appropriate control groups), which are derived from human or animal toxicity studies by one or more UFs and MFs.

To estimate the risk of acute exposures, levels of acceptable short-term exposure may be developed, representing, for instance, the maximum 1-d exposure levels that are anticipated not to result in adverse effects in most individuals. Where available, acute ADIs can be based on EPA's 1-d drinking water health advisories. Where worker exposures are involved, OSHA's permissible exposure limits (PELs), or TLVs where no PEL has been established, serve as ARARs for acute exposures. Also, in rather rare cases where only TLV data may be all that is available, acceptable intake levels may be established/derived by correcting for continuous exposure and further dividing by a safety factor (of 100) to account for highly sensitive segments of impacted populations.

When no toxicological information exists for a chemical, concepts of a structure-activity relationship may have to be employed to derive acceptable intake levels by influence and analogy to closely related or similar compounds. In such cases, some reasonable degree of conservatism is suggested in any judgement call to be made.

Approach for Estimating RfDs (or ADIs). RfDs are typically calculated using a single exposure level and UFs that account for specific deficiencies in the toxicological data base. Both the exposure level and UFs are selected and evaluated in the context of the available chemical-specific literature. After all toxicological, epidemiologic, and supporting data have been reviewed and evaluated, a key study is selected that reflects best availabe data on the critical effect. Dose-response data points for all reported effects are examined as a

component of this review. U.S. EPA (1989d) discusses specific issues of particular significance in this endeavor — including the types of response levels (ranked in order of increasing severity of toxic effects as NOEL, NOAEL, LOAEL, and frank effect level [FEL], defined as overt or gross adverse effects) considered in deriving RfDs for systemic toxicants.

The RfD (or ADI) is determined by use of the following equation:

$$Human\ dose\,(e.g., ADI\ or\ RfD) = \frac{Experimental\ dose\,(e.g., NOAEL)}{(UF \times MF)}$$

or, specifically

$$RfD = \frac{NOAEL}{(UF \times MF)}$$

The UF (also safety factor) used in calculating the RfD reflects scientific judgment regarding the various types of data used to estimate RfD values. It is used to offset the uncertainties associated with extrapolation of data, etc. Generally, the UF consists of multiples of 10, each factor representing a specific area of uncertainty inherent in the available data. For example, a factor of 10 may be introduced to account for the possible differences in responsiveness between humans and animals in prolonged exposure studies. A second factor of 10 may be used to account for variation in susceptibility among individuals in the human population. The resultant UF of 100 has been judged to be appropriate for many chemicals. For other chemicals, with data bases that are less complete (for example, those for which only the results of subchronic studies are available), an additional factor of 10 (leading to a UF of 1000) might be judged to be more appropriate. For certain other chemicals, based on well-characterized responses in sensitive humans (as in the effect of fluoride on human teeth), a UF as small as 1 might be selected (Dourson and Stara, 1983).

In general, the following guidelines are useful in selecting uncertainty and modifying factors for the derivation of RfDs (Dourson and Stara, 1983; U.S. EPA, 1986b; U.S. EPA, 1989b; U.S. EPA, 1989d):

Standard UFs —

- Use a tenfold factor when extrapolating from valid experimental results in studies using prolonged exposure to average healthy humans. This factor is intended to account for the variation in sensitivity among the members of the human population, due to heterogeneity in human populations, and is referenced as "10H." Thus, if NOAEL is based on human data, a safety factor of 10 is usually applied to the NOAEL dose to account for variations in sensitivities between individual humans.
- Use an additional tenfold factor when extrapolating from valid results of long-term studies on experimental animals when results of studies of human exposure are not available or are inadequate. This factor is intended to account for the uncertainty

involved in extrapolating from animal data to humans and is referenced as "10A." Thus, if NOAEL is based on animal data, the NOAEL dose is divided by an additional safety factor of 10, to account for differences between animals and humans.

- Use an additional tenfold factor when extrapolating from less than chronic results on experimental animals when there are no useful long-term human data. This factor is intended to account for the uncertainty involved in extrapolating from less than chronic (i.e., subchronic or acute) NOAELs to chronic NOAELs and is referenced as "10S."
- Use an additional tenfold factor when deriving an RfD from a LOAEL, instead of a NOAEL. This factor is intended to account for the uncertainty involved in extrapolating from LOAELs to NOAELs and is referenced as "10L."
- Use an additional up to tenfold factor when extrapolating from valid results in experimental animals when the data are "incomplete." This factor is intended to account for the inability of any single animal study to adequately address all possible adverse outcomes in humans and is referenced as "10D."

MF —

- Use professional judgment to determine the MF, which is an additional UF that is greater than zero and less than or equal to ten. The magnitude of the MF depends upon the professional assessment of scientific uncertainties of the study and data base not explicitly treated above; e.g., the completeness of the overall data base and the number of species tested. The default value for the MF is 1.

In general, the choice of the UF and MF values reflect the uncertainty associated with the estimation of an RfD from different human or animal toxicity data bases. For instance, if sufficient data from chronic duration exposure studies are available on the threshold region of the critical toxic effect of a chemical in a known sensitive human population, then the UF used to estimate the RfD may be set at unity (1). That is, these data are judged to be sufficiently predictive of a population subthreshold dose, so that additional UFs are not needed (U.S. EPA, 1989d).

Determination of the RfD for a Hypothetical Example

Using the NOAEL — Consider the case of a study made on 250 animals (e.g., rats) that is of subchronic duration, yielding a NOAEL dosage of 5 mg/kg/day. Then,

$$UF = 10H \times 10A \times 10S = 1000$$

In addition, there is a subjective adjustment (MF) based on the high number of animals (250) per dose group:

$$MF = 0.75$$

These factors then give UF × MF = 750, so that

$$RfD = \frac{NOAEL}{(UF \times MF)} = \frac{5}{750} = 0.007\,(mg/kg/day)$$

Using the LOAEL — If the NOAEL is not available, and if 25 mg/kg/day had been the lowest dose tested that showed adverse effects,

$$UF = 10H \times 10A \times 10S \times 10L = 10{,}000$$

Using again the subjective adjustment of MF = 0.75, one obtains

$$RfD = \frac{LOAEL}{(UF \times MF)} = \frac{25}{7500} = 0.003\,(mg/kg/day)$$

Quantification of Toxicological Effects of Lead as an Example. Because of the importance of chemicals such as lead in a risk assessment, and since no RfDs are available through the U.S. EPA data base systems, an approach based on the use of ADIs may be utilized to estimate the acceptable chronic intakes (AICs) for the different potential receptor groups identified for a subject site.

Surrogate measures for oral RfDs — Marcus (1986) has calculated the ADIs for infants and children to be 19 μg/day and for adults to be 48 μg/day. Standard body weights of 16, 29, and 70 kg are used for children under 6 years, children between 6 to 12 years, and adults, respectively. Based on this information, the following AICs are calculated and used as substitute for oral RfDs for lead from the contaminated site problem:

AIC for children aged < 6 years

$$= \frac{(19 \times 10^{-3})\,mg/day}{16\,kg} = 1.19 \times 10^{-3}\ mg/kg/day$$

AIC for children aged 6 – 12 years

$$= \frac{(19 \times 10^{-3})\,mg/day}{29\,kg} = 6.55 \times 10^{-4}\ mg/kg/day$$

AIC for adults

$$= \frac{(48 \times 10^{-3})\,mg/day}{70\,kg} = 6.86 \times 10^{-4}\ mg/kg/day$$

Surrogate measures for inhalation RfDs — The Health Effects Assessment (HEA) (from the Environmental Criteria and Assessment Office of the U.S. EPA) bases its inhalation AIC for lead on the air standard of 1.5 μg/m^3. Inhalation rates of 0.25, 0.46, and 0.83 m^3/h are used for children under 6 years (16 kg), children between 6 to 12 years (29 kg), and adults (70 kg), respectively. Based on this

information, the following inhalation AICs are calculated and used as substitute for inhalation RfDs for lead from the contaminated site problem:

AIC for children aged < 6 years

$$= \frac{\left[\left(1.5\mu g / m^3\right) \times \left(0.25 m^3 / h\right) \times \left(24 h / day\right) \times \left(10^{-3} mg / \mu g\right)\right]}{16\, kg}$$

$$= 5.63 \times 10^{-4}\, mg / kg / day$$

AIC for children aged $6 - 12$ years

$$= \frac{\left[\left(1.5\mu g / m^3\right) \times \left(0.46 m^3 / h\right) \times \left(24 h / day\right) \times \left(10^{-3} mg / \mu g\right)\right]}{29\, kg}$$

$$= 5.71 \times 10^{-4}\, mg / kg / day$$

AIC for adults

$$= \frac{\left[\left(1.5\mu g / m^3\right) \times \left(0.83 m^3 / h\right) \times \left(24 h / day\right) \times 10^{-3} mg / \mu g\right]}{70\, kg}$$

$$= 4.29 \times 10^{-4}\, mg / kg / day$$

Similar procedures may be used for estimating applicable toxicity parameters for noncarcinogenic effects of other chemicals of concern.

Inter-Conversions of RfD Values. RfD values for inhalation exposure are usually reported both as a concentration in air (mg/m^3) and as a corresponding inhaled dose (in mg/kg/day). RfD values for oral exposures are reported in mg/kg/day; an oral RfD value can be converted to a corresponding concentration in drinking water as follows:

$$mg / L \text{ in water} = \frac{oral\ RfD\left(mg / kg / day\right) \times body\ weight\ (kg)}{ingestion\ rate\ (L / day)}$$

Risk Characterization Considerations. In a risk characterization process, comparison is made between the RfD and the estimated exposure dose (EED). The EED should include all sources and routes of exposure involved. If the EED is less than the RfD, the need for regulatory concern may be small. An alternative measure also considered useful to risk managers is the "margin of exposure" (MOE), which is the magnitude by which the NOAEL of the critical toxic effect exceeds the EED, where both are expressed in the same units. Suppose the EED for humans exposed to a chemical substance (with a RfD of 0.005 mg/kg/day) under a proposed use pattern is 0.02 mg/kg/day (i.e., the EED is greater than the RfD), then:

$$NOAEL = RfD \times (UF \times MF) = 0.005 \times 1000 = 5 mg / kg\text{-}day$$

and

$$MOE = \frac{NOAEL}{EED} = \frac{5\,(mg / kg / day)}{0.02\,(mg / kg / day)} = 250$$

Because the EED exceeds the RfD (and the MOE is less than the UF $\times$ MF of 1000), the risk manager will need to look carefully at the data set, the assumptions for both the RfD and the exposure estimates, and the comments of the risk assessors. In addition, the risk manager will need to weigh the benefits associated with the case and other non-risk factors in reaching a decision on the regulatory dose (RgD), defined by

$$RgD = \frac{NOAEL}{MOE}$$

The MOE may be used as a surrogate for risk; as the MOE becomes larger, the risk becomes smaller.

Determination of SFs and UCRs

The cancer SF (also cancer potency factor or potency slope) is a measure of the carcinogenic toxicity of a chemical generally required for completing a health risk assessment. Exposure to any level of a carcinogen is considered to have a finite risk of inducing cancer associated with it, i.e., carcinogenic exposure is generally not considered to have a no-effect threshold. The SF is the cancer risk (proportion affected) per unit of dose and is usually expressed in milligrams of substance per kilogram body weight per d (mg/kg/day). For instance, to estimate risks from exposures in food, one multiplies the SF (risk per mg/kg/day), the concentration of the chemical in the food (ppm), and the daily intake (mg) of that food together; the total dietary risk is found by summing risks across all foods.

For evaluating risks from chemicals found in certain other environmental sources, dose-response measures are expressed as risk per concentration unit. These measures are called the unit risk for air (inhalation) and the unit risk for drinking water (oral). The continuous lifetime exposure concentration units for air and drinking water are usually expressed in micrograms per cubic meter ($\mu g/m^3$) and micrograms per liter ($\mu g/L$), respectively. If the fraction of the agent that is absorbed from the diet for humans and animals differs, a correction factor is applied when extrapolating the animal-derived value to humans.

Scientific investigators have developed numerous models to extrapolate and estimate low-dose carcinogenic risks to humans from high-dose carcinogenic effects usually observed in experimental animal studies. Such models yield an estimate of the upper limit in lifetime risk per unit of dose (or the UCR, or unit risk, UR). The U.S. EPA generally uses the linearized multistage model to generate UCRs. This model, known to make several conservative assumptions, results in highly conservative risk estimates, yielding overestimates of actual UCR for carcinogens; in fact, the actual risks may be substantially lower than that predicted by the upper bounds of this model (Paustenbach, 1986).

Structural similarity factors, etc. can be used to estimate cancer potency units for chemicals not having one, but that are suspected to be carcinogenic. This is achieved, for instance, by estimating the geometric mean of a number of similar compounds whose UCRs are known and using this as a surrogate value for the chemical with unknown UCR.

Derivations and Conversion of Cancer Potency Slope to Unit Risk Values. The unit risk estimates the upper-bound probability of a "typical" or "average" person contracting cancer when continuously exposed to 1 μg/m^3) of the chemical over an average 70-year lifetime. Potency estimates are also given in terms of "potency slopes"; a potency slope is the probability of contracting cancer due to exposure to a given lifetime dose in units of mg/kg/day. The potency, SF, can be converted to UCR (also UR, or unit risk factor, URF) by adopting several assumptions. The most critical factor is that the endpoint of concern must be a systematic tumor, so that potential target organs experience the same blood concentration of the active carcinogen regardless of the method of administration. This implies an assumption of equivalent absorption by the various routes of administration. The basis for these conversions is the assumption that at low doses, the dose-response curve is linear, so that

$$P(d) = SF \times \{dose\}$$

where

P(d) = response (probability) as a function of dose
SF = potency slope factor (mg/kg/day)$^{-1}$
{dose} = amount of chemical intake (mg/kg/day)

Inhalation potency factor — Risks associated with unit chemical concentration in air is estimated as follows:

$$\begin{aligned} &risk\ per\ \mu g / m^3 (air) \\ &= slope\ factor\left(risk\ per\ mg / kg / day\right) \times \frac{1}{body\ weight(kg)} \\ &\times inhalation\ rate\left(m^3 / day\right) \times 10^{-3} (mg / \mu g) \end{aligned}$$

Thus, the inhalation potency can be converted to a UCR value by applying the following conversion factor:

$$\{(kg - day) / mg\} \times \{1 / 70\,kg\} \times \{20\,m^3 / day\} \times \{1\,mg / 1000\,\mu g\} = 2.86 \times 10^{-4}$$

Thus, the lifetime excess cancer risk from inhaling 1 μg/m^3 concentration for a full lifetime is

$$UCR\left(\mu g / m^3\right)^{-1} = \left(2.86 \times 10^{-4}\right) \times SF$$

Alternatively, the potency, SF, can be derived from the unit risk as follows:

$$SF = \left(3.5 \times 10^3\right) \times UCR$$

The assumptions used involve a 70-kg body weight and an average inhalation rate of 20 m^3/day.

Risk-specific concentrations in air — Risk-specific concentrations of chemicals in air is estimated from the unit risk in air as follows:

$$\begin{aligned} Air\ concentration, \mu g / m^3 &= \frac{specified\ risk\ level\,(R) \times body\ weight\,(BW)}{SF \times inhalation\ rate \times 10^{-3}} \\ &= \frac{specified\ risk\ level\,(R)}{UCR} = \frac{1 \times 10^{-6}}{UCR\left(\mu g / m^3\right)^{-1}} \end{aligned}$$

The assumptions used involves a specified risk level of 10^{-6}, a 70-kg body weight, and an average inhalation rate of 20 m^3/day.

Oral potency factor — Risks associated with unit chemical concentration in water is estimated as follows:

$$\begin{aligned} &risk\ per\ \mu g / L (water) \\ &= slope\ factor\left(risk\ per\ mg / kg / day\right) \times \frac{1}{body\ weight\left(kg\right)} \times ingestion\ rate \\ &\left(L / day\right) \times 10^{-3} \left(mg / \mu g\right) \end{aligned}$$

Thus, the ingestion potency can be converted to a UCR value by applying the following conversion factor:

$$\{(kg - day) / mg\} \times \{1 / 70\,kg\} \times \{2\,L / day\} \times \{1\,mg / 1000\,\mu g\} = 2.86 \times 10^{-5}$$

Thus, the lifetime excess cancer risk from ingesting 1 μg/L concentration for a full lifetime is

$$UCR(\mu g / L)^{-1} = \left(2.86 \times 10^{-5}\right) \times SF$$

Alternatively, the potency, SF, can be derived from the unit risk as follows:

$$SF = \left(3.5 \times 10^4\right) \times UCR$$

The assumptions used involve a 70-kg body weight and an average water ingestion rate of 2 L/day.

Risk-specific concentrations in water — Risk-specific concentrations of chemicals in drinking water can be estimated from the oral slope factor; the water concentration corrected for an upper-bound increased lifetime risk of R is given by

$$mg/L\,in\,water = \frac{(specified\,risk\,level, R \times body\,weight, BW)}{(slope\,factor, SF \times ingestion\,rate)}$$

$$= \frac{specified\,risk\,level\,(R)}{UCR(oral)}$$

or

$$= \frac{1\times10^{-6}\times70\,kg}{slope\,factor(mg/kg/day)^{-1}\times2\,L/day} = \frac{3.5\times10^{-5}}{SF}$$

The assumptions used involve a specified risk level of 10^{-6}, a 70-kg body weight, and an average water ingestion rate of 2 L/day.

5.1.4 Risk Characterization

Risk characterization is the process of estimating the probable incidence of adverse impacts to potential receptors under various exposure conditions, including an elaboration of uncertainties associated with such estimates. It is the final step in the risk assessment process and the first input to the risk management process. Its purpose is to present the risk manager with a synopsis and synthesis of all the data that should contribute to a conclusion with regards to the nature and extent of the risk. The risk characterization involves the integration of the exposure and toxicity assessments to arrive at an estimate of risk to the exposed population, both qualitatively and quantitatively. The exposure estimates and toxicity values used in the risk characterization should either both be expressed as absorbed doses or both expressed as administered doses (or intakes).

Risk characterization involves the quantitative estimation of the actual and potential risks and/or hazards due to exposure to each key chemical constituent, and also the possible additive effects of exposure to mixtures of the chemicals of concern. During risk characterization, chemical-specific toxicity information is compared against both measured contaminant exposure levels and, in some cases, those levels predicted through fate and transport modeling to determine whether current or future levels at or near a site under investigation are of potential concern. The risks to potentially exposed populations from exposure and subsequent intake of the chemicals of potential concern are characterized by the calculation of noncarcinogenic hazard quotients and indices and/or carcinogenic risks. These parameters are then compared with applicable standards for risk decisions in hazardous waste management.

An adequate characterization of risks and hazards at a potentially contaminated site allows a site remediation process to be better focused. Cleanup criteria can be developed based on the "acceptable" level of risks to potential receptors. Exposures resulting in the greatest risk can be identified and site mitigation measures selected to address these issues. In this sense, the risk assessment process integrates the information obtained during a remedial investigation (RI) into a coherent set of goals for the feasibility study (FS) phase of the site investigation for a potentially contaminated site.

Aggregate Effects of Chemical Mixtures

There are numerous complexities and inherent uncertainties involved in the analysis of contaminated site problems. The wastes found at contaminated sites tend to be heterogeneous and variable mixtures that may contain several distinct compounds, distributed over wide spatial regions and several compartmental media. The toxicology of complex mixtures is not well understood, further complicating the problem. Large uncertainties exist regarding the potential for these compounds to cause various health and environmental effects. Nonetheless, there is the need to assess the cumulative health risks for several chemicals measured or predicted in any environmental medium (U.S. EPA, 1986b). The method of approach assumes additivity of effects for carcinogens when evaluating chemical mixtures or multiple carcinogens. Any carcinogens which are not included in the quantitative analysis due to lack of potency values should be identified and discussed qualitatively.

For multiple pollutant exposures to noncarcinogens and noncarcinogenic effects of carcinogens, constituents should be grouped by the same mode of toxicological action (i.e., those which induce the same toxicological endpoint, such as liver toxicity). Toxicological endpoints that will normally be considered in a hazard index with respect to chronic toxicity include cardiovascular systems (CVS); central nervous system (CNS); immune system; reproductive system (including teratogenic and developmental effects); kidney; liver; and respiratory system. Cumulative risk is evaluated through the use of a hazard index that is generated for each health "endpoint." Chemicals with the same endpoint should be included in a hazard index calculation. Strictly speaking, constituents should not be grouped together unless the toxicological endpoint is known to be the same. If any calculated hazard index exceeds unity, then the health-based criterion for the chemical mixture has been exceeded and the need for interim corrective measures must be addressed. The risk assessment process must address the multiple endpoints or effects and also the uncertainties in the dose-response functions for each effect. Generally, the risk assessment is facility specific and the calculated risks should be combined for pollutants originating from a given facility or group in the case-study affecting same receptor groups.

Adjustments for Absorption Efficiency

Absorption adjustments may be necessary in the risk characterization stage to ensure that the site exposure estimate and the toxicity value for comparison are both expressed as absorbed doses or both expressed as intakes. Adjustments may be necessary to match the exposure estimate with the toxicity value if one is based on an absorbed dose and the other is based on an intake (i.e., administered dose).

Adjustments may also be necessary for different vehicles of exposure (e.g., water, food, or soil). Furthermore, adjustments may be necessary for different absorption efficiencies, depending on the medium of exposure. In the absence of reliable information, 100% absorption is normally used for most chemicals; for metals, approximately 10% absorption may be considered as a reasonable upper bound for other than the inhalation exposure route. Adjustment procedures are discussed in the literature (e.g., U.S. EPA, 1989i).

Absorption factors should not be used to modify exposure estimates in those cases where absorption is inherently factored into the toxicity/risk parameters used for the risk characterization. Thus, "correction" for fractional absorption is appropriate only for those values derived from experiments/studies based on absorbed dose. Consequently, no "correction" due to incomplete absorption is appropriate when these standards are used. Correction for fractional absorption is appropriate in two cases in particular:

- Interaction with environmental media or other contaminants may alter absorption from that expected for the pure compound.
- Assessment of exposure via a different route of contact from what was utilized in the experimental studies establishing the SFs and RfDs.

Absorbed dose should be used in risk characterization only if the applicable toxicity parameter (e.g., SF or RfD) has been adjusted for absorption; otherwise, simply use intake (undjusted for absorption) for the calculation of risk levels.

Estimation of Carcinogenic Risks

The risk of contracting cancer can be estimated by combining information about the carcinogenic potency of a chemical and exposure to the substance. For potential carcinogens, risks are estimated as the incremental probability of an individual contracting cancer over a lifetime as a result of exposure to the potential carcinogen (i.e., the excess or incremental individual lifetime cancer risk). The carcinogenic risks are estimated by multiplying the cancer SF, which is the upper 95% confidence limit of the probability of a carcinogenic response per unit intake over a lifetime of exposure, by the estimated intakes — yielding incremental risk values. The carcinogenic effects of the chemicals of concern are calculated according to the following relationship (U.S. EPA, 1989i):

$$Risk, CR = CDI \times SF$$

where

CR = probability of an individual developing cancer (dimensionless)
CDI = chronic daily intake for long-term exposure (i.e., averaged over 70 year lifetime) (mg/kg/day)
SF = slope factor (1/[mg/kg/day])

This represents the linear low-dose cancer risk model and is valid only at low risk

levels (i.e., below estimated risks of 0.01). For sites where chemical intakes are high (i.e., potential risks above 0.01), the one-hit model is used; the one-hit equation for high carcinogenic risk levels is given by the following relationship (U.S. EPA, 1989i):

$$Risk, CR = 1 - \exp(-CDI \times SF)$$

where the terms are same as defined above. On the other hand, the acceptable incremental cancer risk for a chemical is estimated by the following relationship:

$$Acceptable\ incremental\ cancer\ risk = virtually\ safe\ dose(VSD) \times slope\ factor(SF)$$

where VSD represents an acceptable chemical dose or intake (in mg/kg/day).

Aggregate Effects of Multiple Carcinogenic Chemicals. The aggregate cancer risk equation for multiple chemicals is obtained by summing the risks calculated for the individual chemicals. Thus, for multiple compounds,

$$Total\ risk = \sum_{i=1}^{n} \left(CDI_i \times SF_i \right)$$

for the linear low-dose model for low risk levels, or

$$Total\ risk = \sum_{i=1}^{n} \left(1 - \exp\left(-CDI_i \times SF_i \right) \right)$$

for the one-hit model used at high carcinogenic risk levels,

where

CDI_i = chronic daily intake for the i^{th} contaminant
SF_i = slope factor for the i^{th} contaminant
n = total number of carcinogens

Aggregate Effects of Multiple Carcinogenic Chemicals and Multiple Exposure Routes. For multiple compounds and multiple pathways, the overall total cancer risk for all exposure pathways and all contaminants considered in the risk evaluation will be

$$Overall\ total\ risk = \sum_{j=1}^{p} \sum_{i=1}^{n} \left(CDI_{ij} \times SF_{ij} \right)$$

for the linear low-dose model for low risk levels, or

$$Overall\ total\ risk = \sum_{j=1}^{p} \sum_{i=1}^{n} \left(1 - \exp\left(-CDI_{ij} \times SF_{ij} \right) \right)$$

for the one-hit model used at high carcinogenic risk levels,

where

CDI_{ij} = chronic daily intake for the i^{th} contaminant and j^{th} pathway
SF_{ij} = slope factor for the i^{th} contaminant and j^{th} pathway
n = total number of carcinogens
p = total number of pathways or exposure routes

The CDIs are estimated from the equations given previously for chemical intakes, whereas the SF values are obtained from various sources or databases, including the Integrated Risk Information System (IRIS) and the Health Effects Assessment Summary Tables (HEAST), available through the U.S. EPA, or are derived from fundamental toxicological data.

As a rule of thumb, incremental risks of between 10^{-4} and 10^{-7} are generally perceived as acceptable levels for the protection of human health and the environment, with 10^{-6} used as point of departure. Due to the realization that people may be exposed to the same constituents from sources unrelated to a specific site, it is preferred that the estimated carcinogenic risk $<< 10^{-6}$.

Population Excess Cancer Burden. The two important parameters or measures for describing carcinogenic effects are the individual cancer risk and the estimated number of cancer cases — the cancer burden. The unit risk factor multiplied by the environmental concentration, or the potency slope multiplied by the CDI as discussed above, gives the estimated individual cancer risk (i.e., the added lifetime probability that an exposed individual would contract cancer due to the source in question). The individual cancer risk from simultaneous exposure to several carcinogens is assumed to be the sum of the individual cancer risks from each individual chemical. The risk experienced by the individual receiving the greatest exposure is referred to as the "maximum individual risk." The number of cancer cases due to a specific source of emission can be estimated by multiplying the individual risk experienced by a group of people by the number of people in that group. Thus, if 10 million people experience an estimated cancer risk of 10^{-6} over their lifetimes, it would be estimated that 10 (i.e., 10 million $\times$ 10^{-6}) additional cancer cases could occur. The number of cancer incidents in each receptor area can be added to estimate the number of cancer incidents over an entire region. Thus, the excess cancer burden, B_{gi}, is given by

$$B_{gi} = R_{gi} \times P_g$$

where

B_{gi} = population excess cancer burden for i^{th} chemical for exposed group, G
P_g = exposed population group (i.e., the number of persons)
R_{gi} = excess lifetime cancer risk for i^{th} chemical for the exposed population group, G

Assuming cancer burden from each carcinogen is additive, then the total population group excess cancer burden is

$$B_g = \sum_{i=1}^{N} B_{gi} = \sum_{i=1}^{N} R_{gi} \times P_g$$

and

$$\textit{Total population burden}, B = \sum_{g=1}^{G} B_g = \sum_{g=1}^{G} \left\{ \sum_{i=1}^{N} B_{gi} \right\} = \sum_{g=1}^{G} \left\{ \sum_{i=1}^{N} R_{gi} \times P_g \right\}$$

Where possible, cancer risk estimates should be expressed in terms of both individual and population risk. For the population risk, the individual upper-bound estimate of excess lifetime cancer risk for an average exposure scenario is multiplied by the size of the potentially exposed population.

Calculation of Noncarcinogenic Hazards

The overall potential noncarcinogenic effects posed by the chemicals of concern is usually expressed by the hazard index (HI). The noncarcinogenic effects of the chemicals of concern are calculated according to the following relationship (U.S. EPA, 1989i):

$$\textit{Hazard quotient}, HQ = \frac{E}{RfD}$$

where

E = chemical exposure level or intake (mg/kg/day)
RfD = reference dose (mg/kg/day)

Aggregate of Multiple Noncarcinogenic Effects of all Chemicals. The sum total of the hazard quotients for all the chemicals of concern (affecting the same organ) gives the hazard index for a given exposure pathway. The applicable relationship is

$$\textit{Total hazard index}, HI = \sum_{i=1}^{n} \frac{E_i}{RfD_i}$$

where

E_i = exposure level (or intake) for the i^{th} contaminant
RfD_i = acceptable intake level (or reference dose) for i^{th} contaminant
n = total number of noncarcinogens

Aggregate of Multiple Noncarcinogenic Effects of all Chemicals and Multiple Exposure Routes. For multiple compounds and multiple pathways, the overall total noncancer risk for all exposure pathways and all contaminants considered in the risk evaluation will be

$$\textit{Overall total hazard index} = \sum_{j=1}^{p}\sum_{i=1}^{n}\frac{E_{ij}}{RfD_{ij}}$$

where

E_{ij} = exposure level (or intake) for the i^{th} contaminant and j^{th} pathway
RfD_{ij} = acceptable intake level (or reference dose) for i^{th} contaminant and j^{th} pathway

The E values are estimated from the equations given previously for chemical intakes, whereas the RfD values are obtained from databases such as IRIS and HEAST, available through the U.S. EPA, or are derived from fundamental toxicological information. RfDs have been established by the U.S. EPA as thresholds of exposure to toxic substances below which there should be no adverse health impact. These thresholds have been established on a substance-specific basis for oral and inhalation exposures, taking into account evidence from both human epidemiologic and laboratory toxicologic studies.

In accordance with the U.S. EPA guidelines on the interpretation of hazard indices, for any given chemical there may be potential for adverse health effects if the hazard index exceeds unity (1). The "acceptable level" itself (i.e., the RfD) incorporates a large margin of safety, so that it is possible that no toxic effects may occur even if the "acceptable level" is exceeded. However, in interpreting the results, a reference value of HI less than or equal to 1 should be taken as the acceptable reference or standard. For HI values greater than unity (i.e., HI > 1), the higher the value, the greater is the likelihood of adverse noncarcinogenic health impacts. Indeed, if HI > 1.0, it may be necessary to segregate chemicals by organ-specific toxicity and recalculate the values, since strict additivity without consideration for target organ toxicities could overestimate potential hazards (U.S. EPA, 1989i); consequently, the HI is calculated after putting chemicals into groups with same physiologic endpoints. On the other hand, due to the realization that people may be exposed to the same constituents from sources unrelated to a specific site, it is preferred that the estimated noncarcinogenic hazard index be << 1.

Distinction Between Chronic and Subchronic Noncarcinogenic Effects. The chronic noncancer HI is represented by the following modification to the general equation presented above:

$$\textit{Overall chronic hazard index} = \sum_{j=1}^{p}\sum_{i=1}^{n}\frac{CDI_{ij}}{RfD_{ij}}$$

where

CDI_{ij} = chronic daily intake for the i^{th} contaminant and j^{th} pathway
RfD_{ij} = chronic reference dose for i^{th} contaminant and j^{th} pathway

The subchronic noncancer hazard index is represented by the following modification to the general equation presented above:

$$\textit{Overall subchronic hazard index} = \sum_{j=1}^{p}\sum_{i=1}^{n}\frac{SDI_{ij}}{RfD_{sij}}$$

where

SDI_{ij} = subchronic daily intake for the i^{th} contaminant and j^{th} pathway
RfD_{sij} = subchronic reference dose for i^{th} contaminant and j^{th} pathway

Appropriate chronic and subchronic toxicity parameters and intakes are used for completion of such estimates.

Uncertainties Recognition

Considerable uncertainty is inherent in the overall risk assessment process. Uncertainties arise due to the use of several assumptions and inferences to complete a risk assessment. For instance, health risk assessment involves extrapolations and inferences to predict the occurrence of adverse health effects under certain conditions of exposure to chemicals in the environment, based on knowledge of the adverse effects that occur under a different set of exposure conditions (e.g., different dose levels and species). Because of these types of extrapolation and projections, there is uncertainty in the conclusions that are arrived at, due in part to the several assumptions that are part of this process. The following pertinent limitations and uncertainties relate to several components of the health risk assessment process:

- Uncertainties in extrapolations relevant to toxicity information (including inherent limitations in toxicity data — arising for several reasons such as differences in the general knowledge of the toxic effects of different chemicals and uncertainties in interspecies and intraspecies extrapolation) and differences of exposure conditions (with respect actual scenarios). Some chemicals have been extensively studied under a variety of exposure conditions in several species, including humans; others may have limited investigations done on them. Because data that specifically identify the hazards to humans associated with exposure to the various chemicals of concern under the conditions of likely human exposure do not exist, it is necessary to infer those hazards by extrapolating from data obtained under other conditions of exposure, generally in experimental animals. This introduces three types of uncertainties: that related to extrapolating from one species to another (i.e., uncertainties in interspecies extrapolation), those relating to extrapolation from a high-dose region curve to a low-dose region (i.e., uncertainties in intraspecies extrapolation), and those related to extrapolating from one set of exposure conditions to another (i.e., uncertainties due to differences in exposure conditions).
- Representativeness of sampling data (including limitations in determining exposure concentrations and modeling) to data of the actual population being sampled. Uncertainties arise from random and systematic errors in the type of measurement and sampling techniques used. For instance, it is critical that sample detection limits are

lower than both the applicable standards or criteria and the concentration which may present health risks; however, this often becomes a source of uncertainty in sample analysis. Professional judgment is also frequently used to fill data gaps, based on engineering and scientific assumptions, which also has some inherent uncertainties associated with it.

- Limitations in model form, including how close to reality the model function and output are, together with model imperfections. Exposure scenarios and constituent transport models contribute uncertainty to risk assessments; transport models typically oversimplify reality, contributing to uncertainty. The natural variability in environmental and exposure-related parameters causes variability in exposure factors and, therefore, in exposure estimates developed on this basis.
- Considerable uncertainty associated with the toxicity of chemical mixtures. That is, the effects of combining two chemicals may be synergistic (effect when outcome of combining two chemicals is greater than the sum of the inputs), antagonistic (effect when the outcome is less than the sum of the two inputs), or under potentiation (i.e., when one chemical has no toxic effect, but when combined with another chemical that is toxic produces a much more toxic effect). Indeed, chemicals present in a mixture can interact to yield a new chemical or one can interfere with the absorption, distribution, metabolism, or excretion of another. Notwithstanding all these, in general, risk assessments assume toxicity to be additive.

In general, uncertainty is difficult to quantify, or at best, the quantification of uncertainty is itself uncertain. Thus, the risk levels generated in a risk assessment are useful only as a yardstick and decision-making tool for prioritization of problems, rather than being construed as actual expected rates of disease, or adversarial impacts in exposed populations. It is used only as an estimate of risks, based on current level of knowledge coupled with several assumptions. Quantitative descriptions of uncertainty, which could take into account random and systematic sources of uncertainty in potency, exposure, intakes, etc., would help present the spectrum of possible true values of risk estimates, together with the probability (or likelihood) associated with each point in the spectrum.

Model Uncertainties. Because of the various limitations and uncertainties, the results of a risk assessment cannot be considered an absolutely accurate determination of risks. Most of the techniques used for compensating for the uncertainties (such as the use of large safety factors, conservative assumptions, and extrapolation models) are designed to err on the side of safety. For these reasons, many regulatory agencies tend to use the so-called linearized multistage model for conservatism. In fact, several models have been proposed for the quantitative extrapolations of carcinogenic effects to low dose levels. However, among these models, the U.S. EPA recommends a linearized multistage model (U.S. EPA, 1986a). The linearized multistage model conservatively assumes linearity at low doses. Alternative models that are generally less conservative do exist which do not assume a linear relationship. There is often no sound basis, in a biological sense, for choosing one model over another. When applied to the same data, the various models can produce a wide range of risk estimates. The model recommended by the U.S. EPA produces among the highest estimates of risk and thus provides a greater margin of protection for

human health. Moreover, this model does not provide a "best estimate" or point estimate of risk, but rather an upper-bound probability that the actual risk will be less than the predicted risk 95% of the time. However, given that no single model will apply for all chemicals, it is important to identify risk models on a case-by-case basis. In fact, Huckle (1991) suggests a presentation of the best estimate of risk (or range, with an added margin of safety) from two or three appropriate models, or a single value based on "weight-of-evidence," rather than using simply the linearized multistage model. Exceptions may occur, however, for cases of poorly studied chemicals.

Uncertainties in Uncertainty Adjustments. Experimental studies to determine the carcinogenic effects due to low exposure levels usually encountered in the environment generally are not feasible. This is because such effects are not readily apparent in the relatively short time frame over which it is generally possible to conduct such a study. Consequently, various mathematical models are used to extrapolate from the high doses used in animal studies to the doses encountered in exposure to ambient environmental concentrations. Extrapolating from a high dose (of animal studies) to a low dose (for human effects) introduces a level of uncertainty which could be significantly large. For instance, NOAELs and SFs from animal studies are usually divided by a factor of 10 to account for extrapolation from animals to humans and by an additional factor of 10 to account for variability in human responses. Given the recognized differences among species in responses to toxic insult, and between strains of the same species, it is apparent that additional uncertainties will be introduced when quantitative extrapolations and adjustments are made in the dose-response evaluation.

Potential for Risk Underestimation. It is always possible that a chemical whose toxic properties have not been thoroughly tested may be more toxic than originally believed or anticipated. For instance, a chemical not tested for carcinogenicity or teratogenicity may in fact display those effects. Furthermore, a limitation of analysis for selected "indicator chemicals" may have some limiting (even if insignificant) effects. The following factors, among others, can typically underestimate health impacts associated with chemicals evaluated in a risk assessment:

- Lack of potency data for some carcinogenic chemicals
- Risks due to compounds formed in environmental media (such as transformation products) that are not quantified
- All risks are assumed to be additive, although certain combinations of exposure may have synergistic (greater than additive) effects

Potentials for Overestimating Risks. A number of factors may cause an analysis to overestimate risks, including

- Many unit risk and potency factors are often considered plausible upper-bound estimates of carcinogenic potency, whereas the true potency of the chemical could be considerably lower.

- Exposure estimates are often very conservative.
- Possible antagonistic effects, for chemicals in which the combined presence reduces toxic impacts, are not accounted for.

5.1.5 Potential Applications

In the course of typical investigations of potentially contaminated sites, efforts are made to adequately characterize the site so that appropriate corrective actions can be implemented. Generally, risk assessment techniques can be employed to better develop the site characterization, site assessment, and corrective action plans. The scope of applications for the health risk assessment methodology discussed may vary greatly; the following specific applications are identified as part of the more common uses:

- Preliminary screening for potential problems (incorporating an analysis of baseline risks, and a consistent process to document potential public health and environmental threats from potentially contaminated sites)
- Evaluation and ranking of potential liabilities from hazardous waste facilities and properties
- Corrective measures evaluation and selection of remedial alternatives (i.e., risks posed by alternative remedial actions can be assessed before implementation)
- Prioritization of hazardous waste sites for remedial action (i.e., this helps to prioritize cleanup actions by providing consistent data for the rank-ordering of potentially contaminated sites)
- Development of target cleanup criteria for potentially contaminated sites (i.e., this provides the basis to determine levels of chemicals that can remain at a site or in environmental media without impacting public health and the environment)
- Site selection in hazardous waste management for siting of hazardous waste management facilities, including disposal sites
- Field sampling design and identification of data needs and/or data gaps

Risk assessment provides a logical, rational, and methodologically consistent approach to making cost-effective decisions. It is therefore almost imperative to make risk assessment an integral part of all investigations for potentially contaminated sites and environmental media, except that the level of detail will be case specific, ranging from qualitative through semiquantitative to detailed quantitative analyses.

5.2 METHODS OF AIR IMPACTS ASSESSMENT

CERCLA (1980) and SARA (1986) mandate the characterization of all contaminant migration pathways from hazardous wastes into the environment and an evaluation of the resulting health and environmental impacts. Furthermore, there is increasing concern that air emissions from hazardous waste sites may present a significant source of human exposure to toxic or hazardous substances. In fact, significantly low-level air emissions could pose significant threats if toxic or

carcinogenic compounds are present at potentially contaminated sites, even under baseline or undisturbed conditions. Furthermore, emissions during remedial actions — especially ones involving excavation — may be much higher than baseline conditions. Emissions from RCRA and similar facilities may also pose significant threats to an impact zone. The emissions of critical concern relate to volatile organic chemicals (VOCs), semi-VOCs, particulate matter, and other chemicals associated with wind-borne particulates such as metals, PCBs, dioxins, etc. Volatile chemicals may be released into the gaseous phase from such sources as landfills, surface impoundments, contaminated surface waters, open/ruptured tanks or containers, etc. Also, there is the potential for subsurface gas movements into underground structures such as pipes and basements and eventually into indoor air. Additionally, toxic chemicals adsorbed to soils may be transported to the ambient air as particulate matter or fugitive dust.

Once released to the ambient air, a contaminant is subject to simultaneous transport and diffusion processes in the atmosphere; these conditions are significantly affected by meteorological, topographical, and source factors. Additional fundamental atmospheric processes (other than atmospheric transport and diffusion) that affect airborne contaminants include transformation, deposition, and depletion. The extent to which all these atmospheric processes act on the contaminant of concern determines the magnitude, composition, and duration of the release; the route of human exposure; and the impact of the release on the environment. Several methods exist for estimating air emissions (CAPCOA, 1990; CDHS, 1986; U.S. EPA, 1990b), including

- Direct emissions measurement
- Indirect emissions measurement
- Air monitoring and/or modeling
- Emissions (predictive) modeling

In all cases, site-specific data should be used whenever possible to increase the accuracy of the emission rate estimates. In fact, the combined approach of environmental fate analysis and field monitoring should provide an efficient and cost-effective strategy for investigating the air pathways impacts on potential receptors under varying meterological conditions.

Air Emissions Classification

Hazardous waste site air emissions may be classified as either point or area sources. Point sources include vents (e.g., landfill gas vents) and stacks (e.g., incinerator and air stripper releases); area sources are generally associated with fugitive emissions (e.g., from landfills, lagoons, and contaminated surface areas). Fugitives (associated with area sources) are released at ground level and disperse there, with less influence of winds and turbulence; point sources, generally, come from a stack and are emitted with an upward velocity, often at a height significantly above ground level. Thus, point sources are more readily diluted by mixing and diffusion, further to being at greater heights, so that ground-level concentrations are

APPENDIX B

Relevant Equations Commonly Utilized in Human Health Risk Assessments

B.1 ESTIMATION OF RECEPTOR EXPOSURES TO CHEMICALS: EQUATIONS FOR CALCULATING CHEMICAL INTAKES AND DOSES

Introduction

An analysis of the potential exposures associated with potentially contaminated site problems generally involve a complexity of integrated evaluations and issues to be addressed (Figure B.1). The primary pathways of general concern include inhalation exposures, dermal exposures, soil ingestion, water ingestion, and crops ingestion (for crops contaminated from direct deposition of contaminants); secondary pathways of interest will generally comprise of ingestion of mother's milk, fish ingestion, poultry and eggs ingestion, meat and dairy products ingestion, and crops ingestion (for crops contaminated from root uptake of chemicals). Consumption of locally produced and homegrown food sources (i.e., animals and crops) should be determined and fully incorporated in all multipathway risk assessment. The methods by which each type of exposure is estimated are well documented in materials prepared under the auspices of various regulatory agencies (e.g., CDHS, 1986; U.S. EPA, 1989a,b; CAPCOA, 1990). Receptor exposures for the different primary routes of contact are presented.

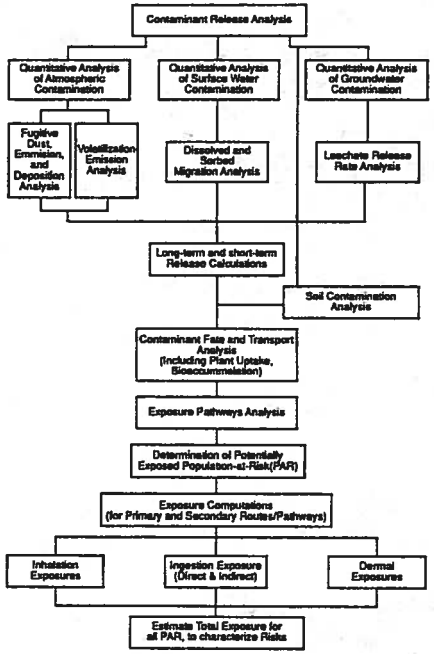

Figure B.1 Simplified schematic for the analysis of exposures associated with potentially contaminated site problem.

Inhalation

Chemical intake via the inhalation exposure pathway is conservatively estimated as follows:

$$\textit{Inhalation exposure}\,(mg/kg\text{-}day) = \frac{\{GLC \times RR \times CF\}}{BW}$$

where

GLC = ground-level concentration (mg/m^3)
RR = respiration rate (m^3/day)
CF = conversion factor (= 1 mg/1000 mg = 1.0E–03 mg/μg)
BW = body weight (kg)

Ingestion

Chemical intake through water, soils, crops, and dairy/beef ingestion exposure pathways are conservatively estimated as follows:

$$\textit{Water ingestion exposure}\,(mg/kg\text{-}day) = \frac{\{CW \times WIR \times GI\}}{BW}$$

$$\textit{Soil ingestion exposure}\,(mg/kg\text{-}day) = \frac{\{CS \times SIR \times GI\}}{BW}$$

$$\textit{Crop ingestion exposure}\,(mg/kg\text{-}day) = \frac{\{CS \times RUF \times CIR \times GI\}}{BW}$$

$$\textit{Dairy and beef products ingestion exposure}\,(mg/kg\text{-}day) = \frac{\{CD \times FIR \times GI\}}{BW}$$

where

CW = chemical concentration in water (mg/L)
WIR = water consumption rate (L/day)
CS = chemical concentration in soil (mg/kg)
SIR = soil consumption rate (kg/day)
RUF = root uptake factor
CIR = crop consumption rate (kg/day)
CD = concentration of chemical in diet (mg/kg); for grazing animals, the concentration of chemicals in tissue, CT, is CT = BCF × F × CD, where BCF is the bioconcentration factor (fat basis) for the organism, expressed as {mg/kg fat}/{mg/kg of diet}, and F is the fat content of tissues (in kg fat/kg tissue)
FIR = food (meat and dairy) consumption (kg/day)
GI = gastrointestinal absorption factor
BW = body weight (kg)

The total dose received by the potential receptors from chemical ingestions will in general be dependent on the absorption of the chemical across the gastrointestinal (GI) lining. The scientific literature provides some estimates of such absorption factors for various chemical substances. It is worthwhile to note, though, that for some chemicals for which a carcinogenic SF is available, GI absorption are already implicitly accounted for in some cases. For chemicals without published absorption values and for which absorption factors are not implicitly accounted for in toxicological parameters, absorption may conservatively be assumed to be 100%.

Dermal

For dermal exposure, the calculation of chemical intakes are carried out as follows:

$$\textit{Dermal exposure to soil}\,(mg/kg\text{-}day) = \frac{\{SS \times SA \times CS \times UF \times CF\}}{BW}$$

$$Dermal\ exposure\ to\ water\ (mg/kg\text{-}day) = \frac{\{WS \times SA \times CW \times UF\}}{BW}$$

where

SS = surface dust on skin (mg/cm^2/day)
CS = chemical concentration in soil (mg/kg)
CF = conversion factor (= 1.00E–06 kg/mg)
WS = water contacting skin (L/cm^2/day)
CW = chemical concentration in water (mg/L)
SA = exposed skin surface (cm^2)
UF = uptake factor
BW = body weight (kg)

Degradation Factor

Since exposure could be occurring over long time periods (up to an estimated human lifetime of 70 years or more), it is important in a detailed analysis to consider whether degradation or other transformation of the chemical at the source could occur. In such cases, the chemical and biological degradation properties of the contaminant should be reviewed. If significant degradation is likely to occur, exposure calculations become much more complicated. In that case, source contaminant levels must be calculated at frequent intervals and summed over the exposure period. For instance, assuming first-order kinetics, an approximation of the degradation effects can be obtained by multiplying the initial media concentration estimate by a degradation factor, DGF, defined by

$$DGF = \frac{\left(1 - e^{-kt}\right)}{kt}$$

where

k = chemical-specific degradation rate constant (days^{-1})
t = time period over which exposure occurs (days)

For a first-order decaying substance, k is estimated from the following relationship:

$$t_{1/2}[days] = \frac{0.693}{k}$$

where $t_{1/2}$ is the half-life, which is the time after which the mass of a given substance will be one half its initial value. It should be recognized in carring out all these manipulations, however, that in many cases when a substance is degraded, it produces an end product that could be of potentially equal or greater concern. Consequently, for simplicity, the decay factor will normally be ignored, except in situations where the end product is known to present no potential hazards to potential receptors.

Potentially Exposed Populations

An important step in the quantitative determination of the potential exposures involves the identification of the populations which may be potentially exposed to chemicals originating from a contaminated site. For instance, the difference in sensitivities between adults and children demands that they be treated separately in evaluating their exposure intakes and doses of chemicals/contaminants. Also, due to the variance in activity and behavior of children at different ages, child exposure of soils is usually broken down into two (or more) categories for such an evaluation, for example:

- Children aged up to 6 years (to include infants and preschool children)
- Children aged between 6 and 12 years (to include young children of school-going age)

For the purpose of a risk assessment and consistent with EPA guidance (U.S. EPA, 1989b), all population groups aged more than 12 years are normally included in the adult category.

Inhalation Exposures

Potential inhalation intakes are estimated based on the length of exposure, the inhalation rate of the exposed individual during the event, the concentration of contaminant in the air respired, and the amount retained in the lungs. Two major types of inhalation exposure pathways are generally considered (Figure B.2). The primary pathway is inhalation of airborne contaminants, in which all individuals within approximately 80 km (50 mi) radius of the site are potentially impacted. A secondary exposure pathway is inhalation of VOCs (i.e., airborne, vapor-phase chemicals) during domestic water use for showering. In fact, inhalation of VOCs may be considered for the groundwater sources only, since VOCs are not expected to remain in surface waters for the times required to reach service points of municipal/domestic water supply.

Inhalation of Volatile Compounds

Showering generally represents a system that promotes release of VOCs from water due to high turbulence, high surface area, and small droplets of water involved. Thus, the concentration of the contaminants in the shower air is assumed to be in equilibrium with the concentration in the water (DOE, 1987). In the case of volatile compounds released while bathing, the exposure relationship is defined by (U.S. EPA, 1988; 1989a,b)

$$INH = CW \times \left\{\left[\frac{ET_1}{(VS \times 2)}\right] + \frac{ET_2}{VB}\right\} \times IR \times RR \times VW \times ABS_s \times EF \times ED \times \frac{1}{BW} \times \frac{1}{AT}$$

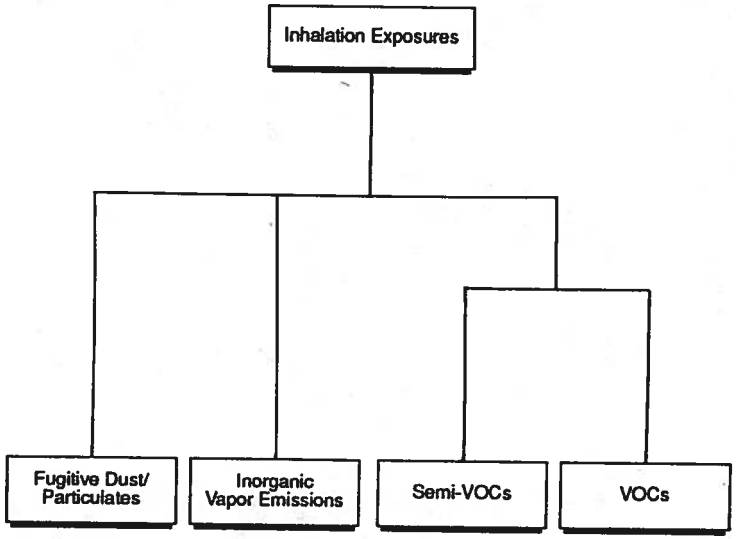

Figure B.2 Major inhalation exposure types.

where

INH = inhalation intake whiles showering (mg/kg/day)
CW = concentration of contaminant in water — adjusted for water treatment purification factor, T_f, which is the fraction remaining after treatment (i.e., $CW = CW_{source} \times T_f$) (mg/L)
ET_1 = length of exposure in shower (h/day)
ET_2 = length of additional exposure in enclosed bathroom (h/day)
VS = volume of shower stall (m^3)
VB = volume of bathroom (m^3)
IR = breathing/inhalation rate (m^3/h)
RR = retention rate of inhaled air (%)
VW = volume of water used in shower (L)
ABS_s = percent of chemical absorbed into the bloodstream (%)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

The concentration of contaminants in water may further be adjusted for environmental degradation by multiplying by a factor of e^{-kt}, where k (in $days^{-1}$) is the environmental degradation constant of the chemical, and t (in days) is the average time of transit through the water distribution system; this yields a new CW value equal to $(CW)(e^{-kt})$ to be used for the intakes computation.

Particulate Inhalation Exposure — Fugitive Dust

The following relationship is used to calculate intakes as a result of the inhalation of wind-borne fugitive dust by potential receptors (CAPCOA, 1990; U.S. EPA, 1988, 1989a, 1989b):

$$INH = CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT$$

where

INH = inhalation intake (mg/kg/day)
CA = chemical concentration in air (mg/m^3)
IR = inhalation rate (m^3/h)
RR = retention rate of inhaled air (%)
ABS_s = percent of chemical absorbed into the Bloodstream (%)
ET = exposure time (h/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

The contaminant concentration in air, CA, is defined by the ground-level concentration (GLC), represented by the respirable (PM-10) particles, expressed in $\mu g/m^3$.

Ingestion Exposures

The major types of ingestion exposure pathways are shown in Figure B.3. Exposure through ingestion is a function of the concentration of the pollutant in the substance or material ingested (soil, water, or food), the GI absorption of the pollutant in solid or fluid matrix, and the amount ingested. The potential intake due to the ingestion of contaminants present in materials ingested (such as contaminated water or soils or sediments) is determined by multiplying the concentration of the chemical in the medium of concern by the amount of fluid or solids ingested per day and the degree of absorption. In general, exposure to contaminants via the ingestion of contaminated fluids or solids may be estimated according to the following relationship (U.S. EPA, 1988; 1989a,b):

$$ING = CONC \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

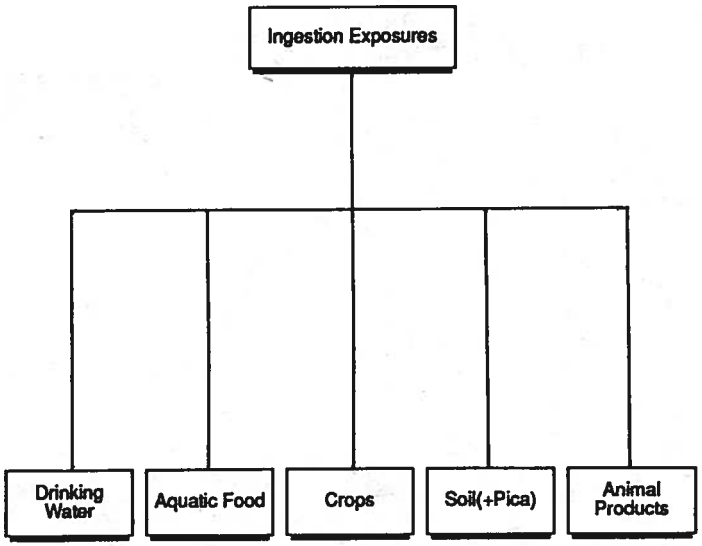

Figure B.3 Major ingestion exposure types.

where

ING = ingestion intake, adjusted for absorption (mg/kg/day)
CONC = chemical concentration in media of concern (mg/kg or mg/L)
IR = ingestion rate (mg or L media material/day)
CF = conversion factor (1.00E–06 kg/mg for solid media, or 1.00 for fluid media)
FI = fraction ingested from contaminated source (unitless)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Exposure Through Ingestion Of Drinking Water

The applicable relationship for the exposure intake through the ingestion of water is as follows:

$$ING_{dw} = CW \times IR \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

ING_{dw} = ingestion intake, adjusted for absorption (mg/kg/day)
CW = chemical concentration in water (mg/L)
IR = average water ingestion rate (L/day)
FI = fraction ingested from contaminated source (unitless)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Exposure Through Ingestion of Chemicals During Swimming Activities

The applicable relationship for the exposure intake through the ingestion of chemicals in surface water during recreational activities is as follows:

$$ING_r = CW \times CR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT$$

where

ING_r = ingestion intake, adjusted for absorption (mg/kg/day)
CW = chemical concentration in water (mg/L)
CR = contact rate (L/h)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
ET = exposure time (h/event)
EF = exposure frequency (events/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Exposure Through Ingestion Of Food

Exposure from the ingestion of food can be via the ingestion of plant products, fish, animal products, and mother's milk. The applicable relationship for the exposure intake through the ingestion of foods is as follows:

$$ING_f = CF \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

ING_f = ingestion intake, adjusted for absorption (mg/kg/day)
CF = chemical concentration in food (mg/kg or mg/l)
IR = average food ingestion rate (mg or L/meal)
CF = conversion factor (1.00E–06 kg/mg for solids and 1.00 for fluids)

FI = fraction ingested from contaminated source (unitless)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
EF = exposure frequency (meals/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Ingestion of Plant Products. Exposure through ingesting plants, ING_p, is a function of the type of plant, GI absorption factor, and the fraction of plants ingested that are affected by pollutants. The calculation is done for each plant type according to the following relationship (CAPCOA, 1990):

$$ING_p = CP_z \times PIR_z \times FI_z \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

ING_p = exposure intake from ingestion of plant products, adjusted for absorption (mg/kg/day)
CP_z = chemical concentration in plant type Z (mg/kg)
PIR_z = average consumption rate for plant type Z (kg/day)
FI_z = fraction of plant type Z ingested from contaminated source (unitless)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Bioaccumulation and Ingestion of Seafood. Exposure from the ingestion of fish from contaminated surface water bodies may be estimated by the following relation (U.S. EPA, 1987; 1988):

$$ING_{sf} = CW \times FIR \times CF \times BCF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

ING_{sf} = total exposure, adjusted for absorption (mg/kg/day)
CW = chemical concentration in surface water (mg/L)
FIR = average fish ingestion rate (g/day)
CF = conversion factor (= 1.00E–03 kg/g)
BCF = chemical-specific bioconcentration Factor (L/kg)
FI = fraction ingested from contaminated source (unitless)
ABS_s = bioavailability/gastrointestinal (gi) absorption factor (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)

BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Ingestion of Animal Products. Exposure through ingestion of animal products, ING_a, is a function of the type of meat ingested (including animal milk products and eggs), GI absorption factor, and the fraction of animal products ingested that are affected by pollutants. The calculation is done for each animal product type according to the following relationship (CAPCOA, 1990):

$$ING_a = CAP_z \times APIR_z \times FI_z \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

ING_a = exposure intake through ingestion of plant products, adjusted for absorption (mg/kg/day)
CAP_z = chemical concentration in food type Z (mg/kg)
$APIR_z$ = average consumption rate for food type Z (kg/day)
FI_z = fraction of product type Z ingested from contaminated source (unitless)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days).

Ingestion of Mother's Milk. Exposure through ingestion of mother's milk, ING_m, is a function of the average chemical concentration in mother's milk, the amount of mother's milk ingested, and GI absorption factor. This is estimated according to the following relationship (CAPCOA, 1990):

$$ING_m = CMM \times IBM \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

ING_m = exposure intake through ingestion of mother's milk, adjusted for absorption (mg/kg/day)
CMM = chemical concentration in mother's milk — which is a function of mother's exposure through all routes and the contaminant body half-life (mg/kg)
IBM = daily average ingestion rate for breast milk (kg/day)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Pica and Incidental Soil/Sediment Ingestion

Absorbed dose due to the incidental ingestion of contaminants sorbed on soils is determined by multiplying the concentration of the contaminant in the medium of concern by the amount of soil ingested per day and the degree of absorption, according to the following relationship (U.S. EPA 1988; 1989a,b; CAPCOA, 1990):

$$ING = CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT$$

where

ING = ingestion intake, adjusted for absorption (mg/kg/day)
CS = chemical concentration in soil (mg/kg)
IR = average ingestion rate (mg soil/day)
CF = conversion factor (1.00E–06 kg/mg)
FI = fraction ingested from contaminated source (unitless)
ABS_s = bioavailability/gastrointestinal (GI) absorption factor (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

In general, it normally is assumed that all ingested soil during receptor exposures come from a contaminated source, so that FI becomes unity.

Dermal Exposures

The major types of dermal exposure pathways are shown in Figure B.4. Dermal intake is determined by the chemical concentration in the medium of concern, the body surface area in contact with the medium, the duration of the contact, the flux of the medium across the skin surface, and the absorbed fraction.

Dermal Exposure — Absorption/Soils Contact

The dermal exposures to chemicals in soils and sediments from a site may be estimated by the following relationship (U.S. EPA 1989a,b; 1988; CAPCOA, 1990):

$$DEX = CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT$$

where

DEX = absorbed dose (mg/kg/day)
CS = chemical concentration in soil (mg/kg)
CF = conversion factor (1.00E–06 kg/mg)
SA = skin surface area available for contact, i.e., surface area of exposed skin (cm^2/event)

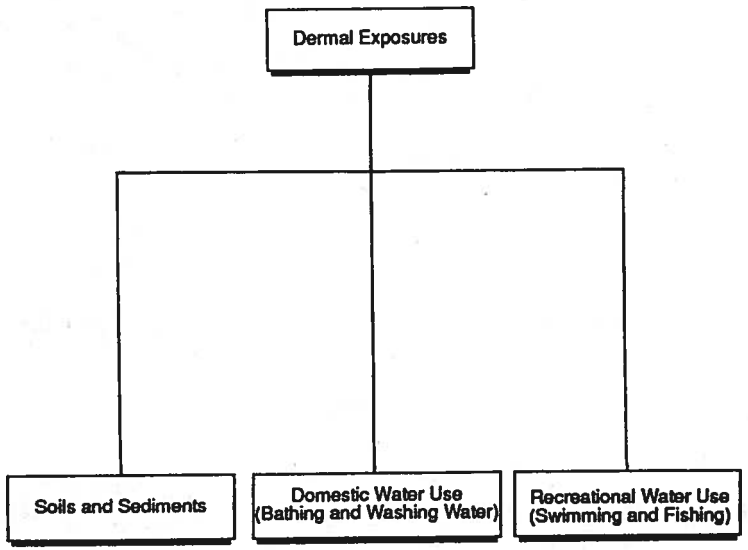

Figure B.4 Major dermal contact exposure types.

AF = soil to skin adherence factor, i.e., soil loading on skin (mg/cm^2)
ABS_s = skin absorption factor for chemicals in soil (%)
SM = factor for soil matrix effects (%)
EF = exposure frequency (events/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Dermal Exposure to Waters and Seeps

Dermal exposures to chemicals may occur during domestic use (such as bathing and washing) or through recreational activities (such as swimming or fishing). The dermal intakes of chemicals in ground- or surface water and/or from seeps from a site may be estimated by the following relationship (U.S. EPA, 1988; 1989a,b):

$$DEX = CW \times SA \times PC \times ABS_s \times CF \times ET \times EF \times ED \times 1/BW \times 1/AT$$

where

DEX = total exposure (mg/kg/day)
CW = chemical concentration in water (mg/L)

SA = skin surface area available for contact, i.e., surface area of exposed skin (cm^2)
PC = chemical-specific dermal permeability constant (cm/h)
ABS_s = skin absorption factor for chemicals in water (%)
CF = volumetric conversion factor for water (1 L/1000 cm3)
ET = exposure time (h/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Typical Computations

In general, default values may be obtained from literature for some of the parameters used in the estimation of intakes and doses. Table B.1 gives typical parameters commonly used; this is by no means complete. More detailed information can be obtained from several sources (e.g., U.S. EPA, 1987, 1988, 1989a; CAPCOA, 1990). A spreadsheet for automatically calculating intake factors for site-specific problems may be designed as shown in Table B.2. Numerical examples for potential receptor groups expected to be exposed through inhalation, soil ingestion (i.e., incidental or pica behavior), and dermal contact are discussed below. The same set of units are maintained throughout as given above.

Inhalation Exposures

The daily inhalation intake of fugitive dust for various population groups are calculated for both carcinogenic and noncarcinogenic effects. The assumed variables used in the numerical demonstration are given in Table B.1.

Carcinogenic Effects from a Contaminated Site— Estimation of LADD. For the fugitive dust inhalation pathway, the carcinogenic CDI (also the LADD) is estimated for the different population groups identified to represent the critical receptors.

The carcinogenic CDI for children aged up to 6 years is calculated to be

$$\begin{aligned}&CInh_{(1\text{-}6)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.25 \times 1 \times ABS_s \times 12 \times 365 \times 5 \times 1/16 \times 1/(70 \times 365)\\&= 1.34E\text{-}02 \times ABS_s \times [CA]\end{aligned}$$

The carcinogenic CDI for children aged 6 to 12 years is calculated to be

$$\begin{aligned}&CInh_{(6\text{-}12)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.46 \times 1 \times ABS_s \times 12 \times 365 \times 6 \times 1/29 \times 1/(70 \times 365)\\&= 1.63E\text{-}02 \times ABS_s \times [CA]\end{aligned}$$

The carcinogenic CDI for adult residents is calculated to be

$$\begin{aligned}&CInh_{(adultR)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.83 \times 1 \times ABS_s \times 12 \times 365 \times 58 \times 1/70 \times 1/(70 \times 365)\\&= 1.18E\text{-}01 \times ABS_s \times [CA]\end{aligned}$$

The carcinogenic CDI for adult workers is calculated to be

$$\begin{aligned}&CInh_{(adultW)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.83 \times 1 \times ABS_s \times 8 \times 260 \times 58 \times 1/70 \times 1/(70 \times 365)\\&= 5.60E\text{-}02 \times ABS_s \times [CA]\end{aligned}$$

Noncarcinogenic Effects from a Contaminated Site— Estimation of ADD. For the fugitive dust inhalation pathway, the noncarcinogenic CDI (also, the ADD) is estimated for the different population groups identified to represent the critical receptors.

The noncarcinogenic CDI for children aged up to 6 years is calculated to be

$$\begin{aligned}&NCInh_{(1\text{-}6)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.25 \times 1 \times ABS_s \times 12 \times 365 \times 5 \times 1/16 \times 1/(5 \times 365)\\&= 1.88E\text{-}01 \times ABS_s \times [CA]\end{aligned}$$

The noncarcinogenic CDI for children aged 6 to 12 years is calculated to be

$$\begin{aligned}&NCInh_{(6\text{-}12)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.46 \times 1 \times ABS_s \times 12 \times 365 \times 6 \times 1/29 \times 1/(6 \times 365)\\&= 1.90E\text{-}01 \times ABS_s \times [CA]\end{aligned}$$

The noncarcinogenic CDI for adult residents is calculated to be

$$\begin{aligned}&NCInh_{(adultR)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.83 \times 1 \times ABS_s \times 12 \times 365 \times 58 \times 1/70 \times 1/(58 \times 365)\\&= 1.42E\text{-}01 \times ABS_s \times [CA]\end{aligned}$$

The noncarcinogenic CDI for adult workers is calculated to be

$$\begin{aligned}&NCInh_{(adultW)}\\&= CA \times IR \times RR \times ABS_s \times ET \times EF \times ED \times 1/BW \times 1/AT\\&= [CA] \times 0.83 \times 1 \times ABS_s \times 8 \times 260 \times 58 \times 1/70 \times 1/(58 \times 365)\\&= 6.76E\text{-}02 \times ABS_s \times [CA]\end{aligned}$$

Table B.1 Example Case-Specific Parameters for Exposure Assessment

Parameter	Children Aged Up to 6	Children Aged 6–12	Adult	Reference Sources
Physical characteristics				
Average body weight	16 kg	29 kg	70 kg	a,b,c
Average total skin surface area	6980 cm^2	10,470 cm^2	18,150 cm^2	a,b,e,h
Average lifetime			70 years	a,b,c,e
Average lifetime exposure period	5 years	6 years	58 years	b,e
Activity characteristics				
Inhalation rate	0.25 m^3/h	0.46 m^3/h	0.83 m^3/h	b,e
Retention rate of inhaled air	100%	100%	100%	e
Frequency of fugitive dust inhalation				
Offsite residents, schools, and passers-by	365 days/year	365 days/year	365 days/year	b,e
Offsite workers	—	—	260 days/year	b,e
Duration of fugitive dust inhalation (outside)				
Offsite residents, schools, and passers-by	12 h/day	12 h/day	12 h/day	b,e
Offsite workers	—	—	8 h/day	b,e
Amount of soil ingested incidentally	200 mg/day	100 mg/day	50 mg/day	a,b,c,e,h,i
Frequency of soil contact				
Offsite residents, schools, and passers-by	330 days/year	330 days/year	330 days/year	b,e
Offsite workers	—	—	260 days/year	b,e
Activity characteristics				
Duration of soil contact				
Offsite residents, schools, and passers-by	12 h/day	8 h/day	8 h/day	b,e
Offsite workers	—	—	8 h/day	b,e
Percentage of skin area contacted by soil	20%	20%	10%	b,e,h
Material characteristics				
Soil to skin adherence factor	0.75 mg/cm^2	0.75 mg/cm^2	0.75 mg/cm^2	a,b,e,f,g
Soil to matrix attenuation factor	15%	15%	15%	d

Note: The exposure factors represented here are for potential maximum exposures (for conservative estimates) and could be modified as appropriate to reflect the most reasonable exposure patterns anticipated. For instance, soil exposure will be reduced by snow cover and rainy days, thus reducing potential exposures for children playing in contaminated areas.

[a] U.S. EPA (1989).
[b] U.S. EPA (1989).
[c] U.S. EPA (1988).
[d] Hawley (1985).
[e] Estimate based on site-specific conditions.
[f] Lepow et al. (1975).
[g] Lepow et al. (1974).
[h] Sedman (1989).
[i] Calabrese et al. (1989).

Table B.2 Example Spreadsheet for Calculating Case-Specific Intake Factors in an Exposure Assessment[a]

	Fugitive Dust Inhalation Pathway								Soil Ingestion Pathway							
Group	IR	RR	ET	EF	ED	BW	AT	INH Factor	IR	CF	FI	EF	ED	BW	AT	ING Factor
C(1-6)@NCarc	0.25	1	12	365	5	16	1825	1.88E–01	200	1.00E–06	1	330	5	16	1825	1.13E–05
C(1-6)@Carc	0.25	1	12	365	5	16	25550	1.34E–02	200	1.00E–06	1	330	5	16	25550	8.07E–07
C(6-12)@NCarc	0.46	1	12	365	6	29	2190	1.90E–01	100	1.00E–06	1	330	6	29	2190	3.12E–06
C(6-12)@Carc	0.46	1	12	365	6	29	25550	1.63E–02	100	1.00E–06	1	330	6	29	25550	2.67E–07
ResAdult@NCarc	0.83	1	12	365	58	70	21170	1.42E–01	50	1.00E–06	1	330	58	70	21170	6.46E–07
ResAdult@Carc	0.83	1	12	365	58	70	25550	1.18E–01	50	1.00E–06	1	330	58	70	25550	5.35E–07
JobAdult@NCarc	0.83	1	8	260	58	70	21170	6.76E–02	50	1.00E–06	1	260	58	70	21170	5.09E–07
JobAdult@Carc	0.83	1	8	260	58	70	25550	5.60E–02	50	1.00E–06	1	260	58	70	25550	4.22E–07

	Soil Dermal Contact Pathway							
SA	CF	AF	SM	EF	ED	BW	AT	DEX Factor
1396	1.00E–06	0.75	0.15	330	5	16	1825	8.87E–06
1396	1.00E–06	0.75	0.15	330	5	16	25550	6.34E–07
2094	1.00E–06	0.75	0.15	330	6	29	2190	7.34E–06
2094	1.00E–06	0.75	0.15	330	6	29	25550	6.30E–07
1815	1.00E–06	0.75	0.15	330	58	70	21170	2.64E–06
1815	1.00E–06	0.75	0.15	330	58	70	25550	2.19E–06
1815	1.00E–06	0.75	0.15	330	58	70	21170	2.08E–06

[a] Notations and units are as defined in the text.
INH Factor = inhalation factor for calculation of doses and intakes.
ING Factor = soil ingestion factor for calculation of doses and intakes.
DEX Factor = dermal exposure/skin adsorption factor for calculation of doses and intakes.
C(1-6)NCarc = noncarcinogenic effects for children aged 1 to 6 years.
C(1-6)@Carc = carcinogenic effects for children ages 1 to 6 years.
C(6-12)@NCarc = noncarcinogenic effects for children aged 6 to 12 years.
C(6-12)@Carc = carcinogenic effects for children aged 6 to 12 years.
ResAdult@NCarc = noncarcinogenic effects for resident adults.
ResAdult@Carc = carcinogenic effects for resident adults.
JobAdult@NCarc = noncarcinogenic effects for adult workers.
JobAdult@Carc = carcinogenic effects for adult workers.

Ingestion Exposures

The daily ingestion intake of soils for various population groups are calculated for both carcinogenic and noncarcinogenic effects. The assumed variables used in the numerical demonstration are given in Table B.1.

Carcinogenic Effects from a Contaminated Site — Estimation of LADD. For the soil ingestion pathway, the carcinogenic CDI (also the LADD) is estimated for the different population groups identified to represent the critical receptors.

The carcinogenic CDI for children aged up to 6 years is calculated to be

$$\begin{aligned}&CIng_{(1\text{-}6)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 200 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 330 \times 5 \times 1/16 \times 1/(70 \times 365)\\&= 8.07E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

The carcinogenic CDI for children aged 6 to 12 years is calculated to be

$$\begin{aligned}&CIng_{(6\text{-}12)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 100 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 330 \times 6 \times 1/29 \times 1/(70 \times 365)\\&= 2.67E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

The carcinogenic CDI for adult residents is calculated to be

$$\begin{aligned}&CIng_{(adultR)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 50 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 330 \times 58 \times 1/70 \times 1/(70 \times 365)\\&= 5.35E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

The carcinogenic CDI for adult workers is calculated to be

$$\begin{aligned}&CIng_{(adultW)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 50 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 260 \times 58 \times 1/70 \times 1/(70 \times 365)\\&= 4.22E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

Noncarcinogenic Effects from a Contaminated Site — Estimation of ADD. For the soil ingestion pathway, the noncarcinogenic CDI (also ADD) is estimated for the different population groups identified to represent the critical receptors.

The noncarcinogenic CDI for children aged up to 6 years is calculated to be

$$\begin{aligned}&NCIng_{(1\text{-}6)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 200 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 330 \times 5 \times 1/16 \times 1/(5 \times 365)\\&= 1.13E\text{-}05 \times ABS_s \times [CS]\end{aligned}$$

The noncarcinogenic CDI for children aged 6 to 12 years is calculated to be

$$\begin{aligned}&NCIng_{(6\text{-}12)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 100 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 330 \times 6 \times 1/29 \times 1/(6 \times 365)\\&= 3.12E\text{-}06 \times ABS_s \times [CS]\end{aligned}$$

The noncarcinogenic CDI for adult residents is calculated to be

$$\begin{aligned}&NCIng_{(adultR)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 50 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 330 \times 58 \times 1/70 \times 1/(58 \times 365)\\&= 6.46E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

The noncarcinogenic CDI for adult workers is calculated to be

$$\begin{aligned}&NCIng_{(adultW)}\\&= CS \times IR \times CF \times FI \times ABS_s \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 50 \times 1.00E\text{-}06 \times 1 \times ABS_s \times 260 \times 58 \times 1/70 \times 1/(58 \times 365)\\&= 5.09E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

Dermal Exposures

The daily dermal intake of soils for various population groups are calculated for both carcinogenic and noncarcinogenic effects. The assumed variables used in the numerical demonstration are given in Table B.1.

Carcinogenic Effects from a Contaminated Site — Estimation of LADD. For the soil dermal contact pathway, the carcinogenic CDI (also LADD) is estimated for the different population groups identified to represent the critical receptors.

The carcinogenic CDI for children aged up to 6 years is calculated to be

$$\begin{aligned}&CDEX_{(1\text{-}6)}\\&= CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 1.00E\text{-}06 \times 1396 \times 0.75 \times ABS_s \times 0.15 \times 330 \times 5 \times 1/16 \times\\&\quad 1/(70 \times 365)\\&= 6.34E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

The carcinogenic CDI for children aged 6 to 12 years is calculated to be

$$\begin{aligned}&CDEX_{(6\text{-}12)}\\&= CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT\\&= [CS] \times 1.00E\text{-}06 \times 2094 \times 0.75 \times ABS_s \times 0.15 \times 330 \times 6 \times 1/29 \times\\&\quad 1/(70 \times 365)\\&= 6.30E\text{-}07 \times ABS_s \times [CS]\end{aligned}$$

The carcinogenic CDI for adult residents is calculated to be

$$CDEX_{(adultR)} = CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT$$
$$= [CS] \times 1.00E\text{-}06 \times 1815 \times 0.75 \times ABS_s \times 0.15 \times 330 \times 58 \times 1/70 \times 1/(70 \times 365)$$
$$= 2.19E\text{-}06 \times ABS_s \times [CS]$$

The carcinogenic CDI for adult workers is calculated to be

$$CDEX_{(adultW)} = CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT$$
$$= [CS] \times 1.00E\text{-}06 \times 1815 \times 0.75 \times ABS_s \times 0.15 \times 260 \times 58 \times 1/70 \times 1/(70 \times 365)$$
$$= 1.72E\text{-}06 \times ABS_s \times [CS]$$

Noncarcinogenic Effects from a Contaminated Site — Estimation of ADD. For the soil dermal contact pathway, the noncarcinogenic CDI (also ADD) is estimated for the different population groups identified to represent the critical receptors.

The noncarcinogenic CDI for children aged up to 6 years is calculated as follows

$$NCDEX_{(1\text{-}6)} = CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT$$
$$= [CS] \times 1.00E\text{-}06 \times 1396 \times 0.75 \times ABS_s \times 0.15 \times 330 \times 5 \times 1/16 \times 1/(5 \times 365)$$
$$= 8.87E\text{-}06 \times ABS_s \times [CS]$$

The noncarcinogenic CDI for children aged 6 to 12 years is calculated to be

$$NCDEX_{(6\text{-}12)} = CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT$$
$$= [CS] \times 1.00E\text{-}06 \times 2094 \times 0.75 \times ABS_s \times 0.15 \times 330 \times 6 \times 1/29 \times 1/(6 \times 365)$$
$$= 7.34E\text{-}06 \times ABS_s \times [CS]$$

The noncarcinogenic CDI for adult residents is calculated to be

$$NCDEX_{(adultR)} = CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT$$
$$= [CS] \times 1.00E\text{-}06 \times 1815 \times 0.75 \times ABS_s \times 0.15 \times 330 \times 58 \times 1/70 \times 1/(58 \times 365)$$
$$= 2.64E\text{-}06 \times ABS_s \times [CS]$$

The noncarcinogenic CDI for adult workers is calculated to be

$$NCDEX_{(adultW)} = CS \times CF \times SA \times AF \times ABS_s \times SM \times EF \times ED \times 1/BW \times 1/AT$$
$$= [CS] \times 1.00E\text{-}06 \times 1815 \times 0.75 \times ABS_s \times 0.15 \times 260 \times 58 \times 1/70 \times 1/(58 \times 365)$$
$$= 2.08E\text{-}06 \times ABS_s \times [CS]$$

B.2 RISK CHARACTERIZATION: EQUATIONS FOR CALCULATING CARCINOGENIC RISKS AND NONCARCINOGENIC HAZARD INDICES

Estimation of Carcinogenic Risks

The methodology for calculating carcinogenic risks of chemicals in the environment is well documented in materials prepared under the auspices of various regulatory agencies (e.g., CDHS, 1986; U.S. EPA, 1989b; CAPCOA, 1990). Cancer risk is a function of the LADD and the chemical-specific potency slope. For inhalation, cancer risk can be calculated using unit risk factors (URFs) or unit risk (UR) values and ground-level concentrations (GLCs). Thus,

$$\text{Risk (for noninhalation pathways)} = \text{dose} \times \text{potency slope}$$

and

$$\text{Risk (for inhalation pathway)} = GLC \times UR$$

where dose (or sum of doses from all routes of exposure) is expressed in mg/kg/day; the chemical-specific potency slope is given in units of $(mg/kg/day)^{-1}$; GLC is expressed in $\mu g/m^3$; and the chemical-specific UR is in $(\mu g/m^3)^{-1}$.

More generally, the carcinogenic effects of the chemicals of concern are calculated according to the following relationship (CDHS, 1986; U.S. EPA, 1989b):

$$CR = CDI \times SF$$

where

CR = probability of an individual developing cancer (unitless)
CDI = chronic daily intake averaged over a lifetime, say 70 years (mg/kg/day)
SF = slope factor (1/[mg/kg/day])

This represents the linear low-dose cancer risk model and is valid only at low risk levels (i.e., below estimated risks of 0.01). For sites where chemical intakes are high (i.e., potential risks above 0.01), the one-hit model is used. The one-hit equation for high carcinogenic risk levels is given by the following relationship:

$$CR = \left[1 - \exp(-CDI \times SF)\right]$$

where the terms are the same as defined above for the low-dose model.

The aggregate cancer risk equation for multiple substances is subsequently obtained by summing the risks calculated for the individual chemicals using the above relationship(s). Thus, for multiple compounds,

$$Total\ risk = \sum_{i=1}^{n}\left(CDI_i \times SF_i\right)$$

for the linear low-dose model for low risk levels, or

$$Total\ risk = \sum_{i=1}^{n}\left(1 - \exp\left(-CDI_i \times SF_i\right)\right)$$

for the one-hit model used at high carcinogenic risk levels,

where
CDI_i = chronic daily intake for the i^{th} contaminant
SF_i = slope factor for the i^{th} contaminant
n = total number of carcinogens

For multiple compounds and multiple pathways, the overall total cancer risk for all exposure pathways and all contaminants considered in the risk evaluation will be

$$Overall\ total\ risk = \sum_{j=1}^{p}\sum_{i=1}^{n}\left(CDI_{ij} \times SF_{ij}\right)$$

for the linear low-dose model for low risk levels, or

$$Overall\ total\ risk = \sum_{j=1}^{p}\sum_{i=1}^{n}\left\{1 - \exp\left(-CDI_{ij} \times SF_{ij}\right)\right\}$$

for the one-hit model used at high carcinogenic risk levels,

where
CDI_{ij} = chronic daily intake for the i^{th} contaminant and j^{th} pathway
SF_{ij} = slope factor for the i^{th} contaminant and j^{th} pathway
n = total number of carcinogens
p = total number of pathways or exposure routes

The CDIs are estimated from the equations previously discussed in Appendix B.1 for calculating chemical intakes. The SF values are obtained from various sources or databases, including the IRIS and the HEAST, maintained by the U.S. EPA and other regulatory agencies. As a rule of thumb, incremental risks of between 10^{-7} and 10^{-4} are generally perceived as acceptable levels for the protection of human health.

Estimation of Noncarcinogenic Hazards

The methodology for calculating noncarcinogenic hazards of chemicals in the environment is well documented in materials prepared under the auspices of various regulatory agencies (e.g., CDHS, 1986; U.S. EPA, 1989b; CAPCOA, 1990). The noncarcinogenic effects of the chemicals of concern are calculated according to the following relationship (CDHS, 1986; U.S. EPA, 1989b):

$$Hazard\ quotient,\ HQ = \frac{E}{RfD}$$

where

E = chemical exposure level or intake (mg/kg/day)
RfD = reference dose (mg/kg/day)

The sum total of the hazard quotients for all chemicals of concern gives the HI for a given exposure pathway. The applicable relationship is

$$Total\ hazard\ index,\ HI = \sum_{i=1}^{n}\frac{E_i}{RfD_i}$$

where

E_i = exposure level (or intake) for the i^{th} contaminant
RfD_i = acceptable intake level (or reference dose) for i^{th} contaminant
n = total number of chemicals presenting noncarcinogenic effects

For multiple compounds and multiple pathways, the overall total noncancer risk for all exposure pathways and all contaminants considered in the risk evaluation will be

$$Overall\ total\ hazard\ index = \sum_{j=1}^{p}\sum_{i=1}^{n}\frac{E_{ij}}{RfD_{ij}}$$

where

E_{ij} = exposure level (or intake) for the i^{th} contaminant and j^{th} pathway
RfD_{ij} = acceptable intake level (or reference dose) for i^{th} contaminant and j^{th} pathway

The E values are estimated from the equations previously discussed in Appendix

B.1 for calculating chemical intakes. The RfD values are obtained from databases such as IRIS and HEAST maintained by the U.S. EPA and other regulatory agencies. In accordance with the U.S. EPA guidelines on the interpretation of HIs, for any given chemical, there may be potential for adverse health effects if the HI exceeds unity. For HI values greater than unity, the higher the value, the greater is the likelihood of adverse noncarcinogenic health impacts. In a comprehensive evaluation, it becomes necessary to introduce the idea of physiologic endpoints in the calculation process, in which case chemicals affecting the same target organs (i.e., chemicals determined to have the same physiologic endpoint) are grouped together in the calculation of total HI.

B.3 DEVELOPMENT OF HEALTH-BASED SITE CLEANUP CRITERIA: EQUATIONS FOR CALCULATING SOIL CLEANUP LEVELS FOR REMEDIAL ACTION PLANS

Introduction

The site cleanup level is a site-specific criterion that a remedial action would have to satisfy in order to keep exposures of potential receptors to levels at or below an AL. The ALs tend to drive the cleanup process for a contaminated site. The ALs are calculated for both the systemic toxicants and for the carcinogens; the more stringent of the two, where both exist, is selected as the site cleanup limit. This would represent the maximum acceptable contaminant level for site cleanup.

ALs for Systemic Toxicants

The governing equation for calculating action levels for noncarcinogens and noncarcinogenic effects of carcinogens present at a contaminated site is given by (U.S. EPA, 1987)

$$C_m = \frac{(RfD \times BW \times CF)}{(I \times A)}$$

where

C_m = AL in medium of concern (e.g., soil @ mg/kg)
RfD = reference dose (mg/kg/day)
BW = body weight (kg)
CF = conversion factor (e.g., 10^6 mg/kg for soil ingestion exposures)
I = intake assumption (e.g., soil ingestion rate @ mg/day)
A = absorption factor (dimensionless)

ALs for Carcinogenic Constituents

The governing equation for calculating ALs for carcinogenic constituents present at a contaminated site is given by (U.S. EPA, 1987):

$$C_m = \frac{(R \times BW \times LT \times CF)}{(SF \times I \times A \times ED)}$$

where

C_m = AL (equal to the RSD or VSD) in medium of concern (e.g., soil @ mg/kg)
R = specified (acceptable) risk level (dimensionless)
BW = body weight (kg)
LT = assumed lifetime (years)
CF = conversion factor (e.g., 10^6 mg/kg for soil ingestion exposures)
SF = cancer slope factor (1/[mg/kg/day])
I = intake assumption (e.g., soil ingestion rate @ mg/day)
A = absorption factor (dimensionless)
ED = exposure duration (years)

Allowable Soil Concentrations (ASCs)

To determine ASCs, the following relationships are used based on an algebraic manipulations of the HI or carcinogenic risk equations and the exposure estimation equations.

Noncarcinogenic Effects

The HI is given by

$$HI = \sum \left\{ \sum_{i=1}^{p} \frac{CDI}{RfD_p} \right\} = \frac{CDI_{inh}}{RfD_{inh}} + \frac{CDI_{ing}}{RfD_{ing}} + \frac{CDI_{der}}{RfD_{der}}$$

Assuming there is only one toxic constitiuent present in soils and that exposures via the dermal contact and ingestion routes only contribute to the total HI of 1 (a conservative assumption), then:

$$\sum CDI = RfD$$

or

B.1 for calculating chemical intakes. The RfD values are obtained from databases such as IRIS and HEAST maintained by the U.S. EPA and other regulatory agencies. In accordance with the U.S. EPA guidelines on the interpretation of HIs, for any given chemical, there may be potential for adverse health effects if the HI exceeds unity. For HI values greater than unity, the higher the value, the greater is the likelihood of adverse noncarcinogenic health impacts. In a comprehensive evaluation, it becomes necessary to introduce the idea of physiologic endpoints in the calculation process, in which case chemicals affecting the same target organs (i.e., chemicals determined to have the same physiologic endpoint) are grouped together in the calculation of total HI.

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where

C_m = AL in medium of concern (e.g., soil @ mg/kg)
RfD = reference dose (mg/kg/day)
BW = body weight (kg)
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$$C_m = \frac{(R \times BW \times LT \times CF)}{(SF \times I \times A \times ED)}$$

where

C_m = AL (equal to the RSD or VSD) in medium of concern (e.g., soil @ mg/kg)
R = specified (acceptable) risk level (dimensionless)
BW = body weight (kg)
LT = assumed lifetime (years)
CF = conversion factor (e.g., 10^6 mg/kg for soil ingestion exposures)
SF = cancer slope factor (1/[mg/kg/day])
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To determine ASCs, the following relationships are used based on an algebraic manipulations of the HI or carcinogenic risk equations and the exposure estimation equations.

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The HI is given by

$$HI = \Sigma\left\{\sum_{i=1}^{p}\frac{CDI}{RfD_p}\right\} = \frac{CDI_{inh}}{RfD_{inh}} + \frac{CDI_{ing}}{RfD_{ing}} + \frac{CDI_{der}}{RfD_{der}}$$

Assuming there is only one toxic constitiuent present in soils and that exposures via the dermal contact and ingestion routes only contribute to the total HI of 1 (a conservative assumption), then:

$$\Sigma CDI = RfD$$

or

$$\left(CDI_{ing} + CDI_{der}\right) = RfD_{oral}$$

$$\frac{\left(ASC \times IR \times CF \times FI \times ABS_{si} \times EF \times ED\right)}{\left(BW \times AT\right)}$$

$$+\frac{\left(ASC \times CF \times SA \times AF \times ABS_{sd} \times SM \times EF \times ED\right)}{\left(BW \times AT\right)} = RfD_{oral}$$

or

$$ASC = \frac{\left(BW \times AT\right) \times \left(RfD_{oral}\right)}{\left(CF \times EF \times ED\right)\left\{\left(IR \times FI \times ABS_{si}\right) + \left(SA \times AF \times ABS_{sd} \times SM\right)\right\}}$$

where

CDI = chronic daily intake, adjusted for absorption (mg/kg/day)
ASC = allowable chemical concentration in soil (mg/kg)
RfD_{oral} = oral reference dose (mg/kg/day)
IR = ingestion rate (mg/day)
CF = conversion factor (1.00E–06 kg/mg)
FI = fraction ingested from contaminated source (unitless)
ABS_{si} = bioavailability absorption factor for ingestion exposures (%)
ABS_{sd} = bioavailability absorption factor for dermal exposures (%)
SA = skin surface area available for contact, i.e., surface area of exposed skin (cm^2/event)
AF = soil to skin adherence factor, i.e., soil loading on skin (mg/cm^2)
SM = factor for soil matrix effects (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

Carcinogenic Effects

The cancer risk is given by

$$CR = \sum\left\{\sum_{i=1}^{p} CDI \times SF_p\right\} = \left(CDI_{inh} \times SF_{inh}\right) + \left(CDI_{ing} \times SF_{ing}\right) + \left(CDI_{der} \times SF_{der}\right)$$

Assuming there is only one toxic constituent present in soils and that exposures via the dermal and ingestion routes only contribute to the total CR of R (=10^{-6}, for example) (a conservative assumption), then

$$\sum CDI = \frac{R}{SF_{oral}} = RSD$$

or

$$\left(CDI_{ing} + CDI_{der}\right) = \frac{R}{SF_{oral}}$$

$$\frac{\left(ASC \times IR \times CF \times FI \times ABS_{si} \times EF \times ED\right)}{\left(BW \times AT\right)}$$

$$+\frac{\left(ASC \times CF \times SA \times AF \times ABS_{sd} \times SM \times EF \times ED\right)}{\left(BW \times AT\right)} = \frac{R}{SF_{oral}}$$

or

$$ASC = \frac{\left(BW \times AT\right) \times \left(RSD\right)}{\left(CF \times EF \times ED\right)\left\{\left(IR \times FI \times ABS_{si}\right) + \left(SA \times AF \times ABS_{sd} \times SM\right)\right\}}$$

where

CDI = chronic daily intake, adjusted for absorption (mg/kg/day)
RSD = risk-specific dose (mg/kg/day)
ASC = allowable chemical concentration in soil (mg/kg)
SF_{oral} = oral slope factor (1/mg/kg/day)
IR = ingestion rate (mg/day)
CF = conversion factor (1.00E–06 kg/mg)
FI = fraction ingested from contaminated source (unitless)
ABS_{si} = bioavailability absorption factor for ingestion exposures (%)
ABS_{sd} = bioavailability absorption factor for dermal exposures (%)
SA = skin surface area available for contact, i.e., surface area of exposed skin (cm^2/event)
AF = soil to skin adherence factor, i.e., soil loading on skin (mg/cm^2)
SM = factor for soil matrix effects (%)
EF = exposure frequency (days/years)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (period over which exposure is averaged, days)

RSCLs

The RSCL is estimated in the same way as the soil AL or other equivalent methods, but with a consideration for the aggregation of the individual chemicals

present at the case site. Then, assuming each compound contributes proportionately to the total carcinogenic risk and/or HI, the RSCL is estimated according to the following simplistic relationship:

$$RSCL = \frac{C_m}{N}$$

where

C_m = AL in medium of concern (e.g., soil @ mg/kg)
N = number of chemical contributors to overall HI or cancer risk, as appropriate

Also, the RSCL may alternatively be estimated by proportionately aggregating — or rather disaggregating — the target cancer risk (for carcinogens) or noncancer HI (for noncarcinogenic effects) between the chemicals of potential concern. This is carried out according to the following approximate relationships:

$$RSCL = \frac{(\% \times R \times BW \times LT \times CF)}{(SF \times I \times A \times ED)}$$

for carcinogens, and

$$RSCL = \frac{(\% \times RfD \times BW \times CF)}{(I \times A)}$$

for noncarcinogens. All the terms are the same as defined above, and % represents the proportionate contribution from a specific constituent to the target/acceptable risk level (for carcinogens) or HI (for systemic toxicants).

The assumption used here for allocating estimated excess carcinogenic risk is that all carcinogens have the same mode of biological actions and target organs; otherwise, excess carcinogenic risk is not allocated among carcinogens, but rather each assumes the same value in the computational efforts. Similarly, for the noncarcinogenic effects, the total HI is apportioned only between chemicals with the same toxicological endpoint.

B.4 REFERENCES

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Environmental Engineering Reference Manual

for the PE Exam

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can be measured is known as the *no observed adverse effect level* (NOAEL). However, at this level, the effects observed at the higher doses are not observed.

- *Lowest Observed Adverse Effect*: The dose where effects related to the response being measured first can be measured, and are the same effects as the effects observed at the higher doses, is known as the *lowest observed adverse effect level* (LOAEL).
- *Frank Effect*: The *frank effect level* (FEL) dose marks the point where maximum effects are observed with little increase in effect for increasing dose.

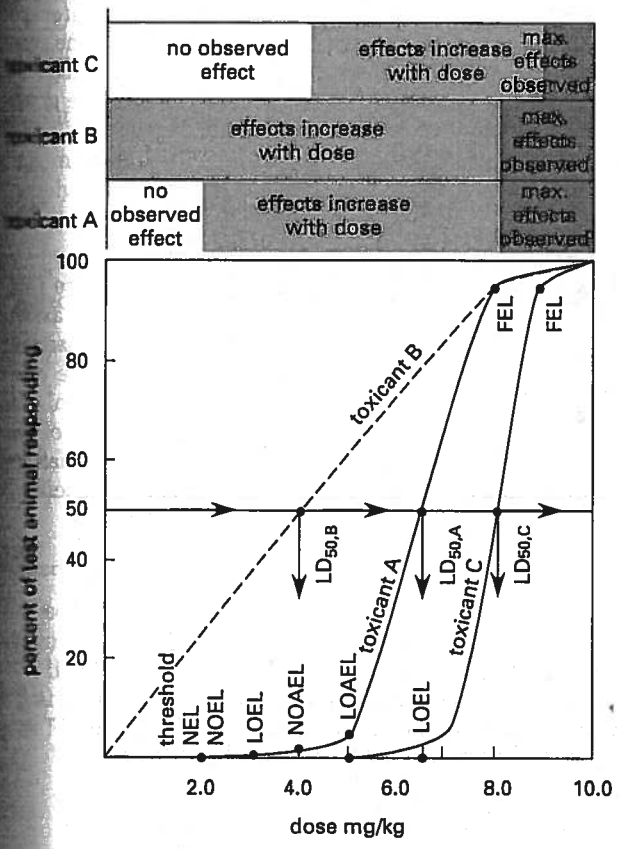

Figure 45.2 Dose-Response Relationships

For some toxicants, primarily those believed to be carcinogens, there is no apparent threshold; any dose is considered to have an effect even though such effect may be unmeasurable at low doses. Such toxicants have no safe exposure level. This is illustrated as toxicant B in Fig. 45.2. Lead is an example of a toxicant with no threshold dose.

Toxicants A and C in Fig. 45.2 have similarly shaped dose-response curves, which means that they show the same relative effects at both high and low doses (i.e., toxicant A is more toxic than toxicant C at both high and low doses). If the dose-response curves for toxicants A and C were to cross, then one toxicant would be more toxic at low doses and the other would be more toxic at high doses. Toxicant A is more toxic than toxicant C because a lower dose of toxicant A results in a pronounced response.

Acute and Chronic Toxicity Tests

To establish a dose-response relationship, toxicity tests are necessary. Both acute and chronic toxicity tests can be used for this purpose.

Acute toxicity tests are conducted for a relatively short duration, typically 24 hours to 14 days, with death of the test animal typically being the "response." Test animals typically include rats, mice, rabbits, and guinea pigs. The acute test must include a *control group* that is subjected to identical environmental conditions as the test animals except that it is not exposed to the toxicant. The response of the control group enables the responses at low toxicant doses to be differentiated from environmental or other factors that may affect the test animals.

The results of acute toxicity tests are typically expressed as the *lethal dose* or *lethal concentration*—the concentration of toxicant at which a specified percentage of the test animals died. The lethal dose is expressed as the mass of toxicant per unit mass of test animal. Thus, LD_{50} means the dose in milligrams of toxicant per kilogram of body mass at which 50% of the test animals died.

For acute tests involving inhalation as the exposure pathway, the concentration (in parts per million) of the toxicant in air is used. If the toxicant is in particulate form, the concentration in milligrams of toxic particles per cubic meter of air is used. Thus, LC_{50} means the concentration of the toxicant in air at which 50% of the test animals died.

The lethal concentration is associated with the inhalation pathway, while the lethal dose may be associated with either the ingestion or skin absorption pathway. However, this convention is not always followed, and the LD_{50} sometimes refers to the inhalation pathway. (Usually the pathway is noted with the lethal dose.)

Chronic toxicity tests can also be performed to determine the long-term dose-response relationship of a toxicant. Chronic tests may be conducted for 30 days or more, for years, or for the life of the test animal. Although death is often the end point in acute toxicity tests, in *chronic toxicity tests* the administered doses are selected such that most of the animals survive for the duration of the study. Chronic tests monitor diet consumption, perform urinalysis, observe changes in blood composition and chemistry, and perform gross and microscopic examination of major tissues and organs.

5. SAFE HUMAN DOSE

EPA Methods

Several approaches exist for selecting a safe human dose from the data obtained from toxicological and epidemiological studies. The approach most likely to be

encountered by the environmental engineer is the one recommended by the U.S. Environmental Protection Agency (EPA).

First, the differences between carcinogens and noncarcinogens (referred to by the EPA as *systemic toxicants*) are compared. Chemicals are classified as carcinogens if they can, or are believed to, produce tumors after exposure. *Carcinogens* generally do not exhibit thresholds of response at low doses of exposure, and any exposure is assumed to have an associated risk. The EPA model used to evaluate carcinogenic risk assumes no threshold and a linear response to any amount of exposure. Noncarcinogens (i.e., systemic toxicants) are chemicals that do not produce tumors (or gene mutations) but instead adversely interfere with the functions of enzymes in the body, thereby causing abnormal metabolic responses. *Noncarcinogens* have a dose threshold below which no adverse health response can be measured. Some substances can produce both carcinogenic and noncarcinogenic responses.

Noncarcinogens

The threshold below which adverse health effects in humans are not measurable or observable is defined by the EPA as the *reference dose* (RfD). The reference dose is used by the EPA and often results in more restrictive intake values than the values published by other organizations. The reference dose is the safe daily intake that is believed not to cause adverse health effects. The reference dose relates to the ingestion and dermal contact pathways and is route specific. For gases and vapors (exposure by the inhalation pathway), the threshold may be defined as the *reference concentration* (RfC). Sometimes the term "reference dose" is also used for exposure by the inhalation pathway. Note also that prior to 1993 the reference dose was referred to as the *acceptable daily intake* (ADI). The terms mean the same thing, but to distinguish between various aspects of risk assessment and management, the EPA adopted less value-laden terminology in 1993—hence the current use of RfD.

The reference dose is set for the most sensitive response when more than one response, or endpoint, is measured. For example, a chemical may affect the central nervous system, the cardiovascular system, or other particular organs. The most sensitive of these responses would be the basis for selecting the reference dose.

The reference dose (or reference concentration) is determined by dividing the NOAEL by the *uncertainty and modifying factors* that represent uncertainty of the procedure (UF and MF, respectively). The factors used by the EPA and the World Health Organization are given in Table 45.1.

$$\text{RfD} = \frac{\text{NOAEL}}{(\text{UF})(\text{MF})} \qquad 45.1$$

After an RfD has been identified for one or more toxicants, the *hazard ratio* can be determined to assess whether exposures indicate an unacceptable hazard. The hazard ratio (HR) is the *estimated exposure dose* (EED) divided by the reference dose for each of the toxicants from all routes of exposure. If the sum of the ratios exceeds 1.0, the risk is unacceptable. The hazard ratio should be considered a preliminary assessment.

$$\text{HR} = \frac{\text{EED}}{\text{RfD}} \qquad 45.2$$

Table 45.1 *Typical EPA Factors Used to Calculate Reference Dose*

designation	description	uncertainty factor (UF)
10A, interspecies variability	used when extrapolating from results of long-term studies on experimental animals when results from human exposure studies are inadequate or unavailable	10
10H, intraspecies variability	used when extrapolating from valid experimental results in studies using prolonged exposure to average healthy humans	10
10L	used when deriving a reference dose (RfD) from LOAEL values rather than NOAEL values	10
10S	used when extrapolating from subchronic results on experimental animals when long-term human results are inadequate or absent	10

Note: A professional judgment modifying factor (MF)—greater than zero and less than or equal to ten—is used to reflect the scientific uncertainties associated with the study not mentioned explicitly above.

Example 45.1

A chronic oral toxicity study was conducted to determine the effects of copper cyanide (CuCN) on rats. The quality of the data is judged by the toxicologist to be "average." The following results were obtained from the study.

NOEL	adverse effect observed at all doses
LOEL	adverse effect observed at all doses
NOAEL	adverse effect observed at all doses
LOAEL	25 mg/kg·day
FEL	100 mg/kg·day

In the absence of other, more reliable published information, what should be used as the reference dose (RfD) for human exposure from drinking water containing copper cyanide (CuCN)?

Solution

From Table 45.1, uncertainty factors 10A, 10H, and 10L apply. Therefore,

$$\text{UF} = (10)(10)(10) = 1000$$

Select an average modifying factor of MF = 5.

From Eq. 45.1,

$$\text{RfD} = \frac{\text{NOAEL}}{(\text{UF})(\text{MF})} = \frac{25\ \frac{\text{mg}}{\text{kg·day}}}{(1000)(5)}$$
$$= 5 \times 10^{-3}\ \text{mg/kg·day} \quad (5\ \mu\text{g/kg·day})$$

Example 45.2

A community water supply well was found to be contaminated with copper cyanide (CuCN), methanol (CH_3OH), and potassium cyanide (KCN). The concentrations in the water and the EPA reference doses are as follows.

toxicant	exposure (μg/L)	RfD (oral) (μg/kg·day)
CuCN	40	5
CH_3OH	1000	500
KCN	600	50

Is a 70 kg person who drinks 2 L of water daily exceeding the safe human dose for these noncarcinogens?

Solution

Determine the estimated exposure dose for the toxicants.

$$\text{EED}_{\text{CuCN}} = \frac{(E_{\text{CuCN}})(\text{CR})}{m} = \frac{\left(40\ \frac{\mu\text{g}}{\text{L}}\right)\left(2\ \frac{\text{L}}{\text{day}}\right)}{70\ \text{kg}}$$
$$= 1.14\ \mu\text{g/kg·day}$$

From Eq. 45.2,

$$\text{HR}_{\text{CuCN}} = \frac{\text{EED}_{\text{CuCN}}}{\text{RfD}_{\text{CuCN}}}$$
$$= \frac{1.14\ \frac{\mu\text{g}}{\text{kg·day}}}{5\ \frac{\mu\text{g}}{\text{kg·day}}}$$
$$= 0.228$$

Similarly, for the other toxicants,

toxicant	exposure, E (μg/L)	estimated exposure dose, EED (μg/kg·day)	RfD (μg/kg·day)	hazard ratio, HR
CuCN	40	1.14	5	0.228
CH_3OH	1000	28.57	500	0.057
KCN	600	17.14	50	0.343
sum				0.628

Because the sum of the hazard ratios is less than 1.0, the safe human dose is not exceeded.

Carcinogens

General

The distinguishing feature of cancer is the uncontrolled growth of cells into masses of tissue called *tumors*. Tumors may be *benign*, in which the mass of cells remains localized, or *malignant*, in which the tumors spread through the bloodstream to other sites within the body. This latter process is known as *metastasis* and determines whether the disease is characterized as cancer. The term *neoplasm* (new and abnormal tissue) is also used to describe tumors.

Cancer occurs in three stages: initiation, promotion, and progression. During the *initiation* stage, a cell mutates and the DNA is not repaired by the body's normal DNA repair mechanisms. During the *promotion* stage, the mutated cells increase in number and undergo differentiation to create new genes. During *progression*, the cancer cells invade adjacent tissue and move through the bloodstream to other sites in the body. It is believed that continued exposure to the agent that initiated genetic mutation is necessary for progression to continue. Many mutations are believed to be required for the progression of cancer cells to occur at remote sites in the body.

Direct Human Exposure

The EPA's classification system for carcinogenicity is based on a consensus of expert opinion called "weight of evidence."

The EPA maintains a database of toxicological information known as the Integrated Risk Information System

(IRIS). The IRIS data include chemical names, chemical abstract service registry numbers (CASRN), reference doses for systemic toxicants, carcinogen potency factors (CPF) for carcinogens, and the carcinogenicity group classification.

Table 45.2 *EPA Carcinogenicity Classification System*

group	description
A	human carcinogen
B1 or B2	probable human carcinogen B1 indicates that human data are available. B2 indicates sufficient evidence in animals and inadequate or no evidence in humans.
C	possible human carcinogen
D	not classifiable as to human carcinogenicity
E	evidence of noncarcinogenicity for humans

The dose-response for carcinogens differs substantially from that for noncarcinogens. For carcinogens it is believed that any dose can cause a response (mutation of DNA). Since there are no levels (i.e., no thresholds) of carcinogens that could be considered safe for continued human exposure, a judgment must be made as to the acceptable level of exposure, which is typically chosen to be an excess lifetime cancer risk of 1×10^{-6} (0.0001%). *Excess lifetime cancer risk* refers to the incidence of cancers developed in the exposed animals minus the incidence in the unexposed control animals. For whole populations exposed to carcinogens, the number of total excess cancers, EC, is the product of the probability of excess cancer, R, and the total exposed population, EP.

$$\mathrm{EC} = (\mathrm{EP})(R) \qquad 45.3$$

Under the EPA approach, the *carcinogen potency factor* (CPF) is the slope of the dose-response curve at very low exposures. The CPF is also called the *potency factor* or *slope factor* and has dimensions of $(\mathrm{mg/kg{\cdot}day})^{-1}$. The CPF is pathway (route) specific. The CPF is obtained by extrapolation from the high doses typically used in toxicological studies.

The CPF is the probability of risk produced by lifetime exposure to 1.0 mg/kg·day of the known or potential human carcinogen. Thus, the slope factor can be multiplied by the long-term daily intake (*chronic daily intake*, CDI) to obtain the lifetime probability of risk, R, for daily doses other than 1.0 mg/kg·day. For less than lifetime exposure, the *exposure duration* must be used to calculate the total intake, which must be divided by the *averaging duration* of 70 years for carcinogens. For noncarcinogens, the averaging duration is the same as the exposure duration (See Eq. 11.75).

$$R = (\mathrm{SF})(\mathrm{CDI}) \qquad 45.4$$

Table 45.3 *EPA Standard Values for Intake Calculations*

parameter	standard value
average body weight, adult	70 kg
average body weight, child	10 kg
daily water ingestion, adult	2 L
daily water ingestion, child	1 L
daily air breathed, adult	20 m^3
daily air breathed, child	5 m^3
daily fish consumed, adult	6.5 g
lifetime exposure period	70 yr

Example 45.3

A village with a stable population of 50,000 has a water supply that has been contaminated with benzene (C_6H_6) from a leaking underground storage tank. The leak occurred during the 20 yr before it was removed. The estimated average concentration of benzene during this period of leaking was 50 μg/L. It is expected that it will take another 10 yr before the benzene will be below detectable levels. The average estimated concentration of benzene during this period is 20 μg/L. The slope factor for benzene by the oral route is 2.0×10^{-2} $(\mathrm{mg/kg{\cdot}day})^{-1}$. Assume a 70 yr lifespan, 70 kg adults, 10 kg children, 2 L/day adult water consumption, 1 L/day child water consumption, and 10% children in the village. If the acceptable risks of additional cancer deaths due to benzene in the water are 1 adult and 0.5 child, can the water supply be used for the next 10 yr or should the village abandon the supply?

Solution

step 1: Find the chronic daily intake (CDI) for adults and children for each time period.

Given a total lifespan (AT) of 70 yr and an exposure factor (EF) of 1, use Table 45.3 and Eq. 11.75.

$$\mathrm{CDI} = \frac{(E)(\mathrm{CR})(\mathrm{EF})(\mathrm{ED})}{(\mathrm{BW})(\mathrm{AT})}$$

	BW (kg)	CR (L/day)	ED (yr)	E (mg/L)	CDI (mg/kg·day)
adults	70	2	10	20×10^{-3}	8.16×10^{-4}
			20	50×10^{-3}	4.08×10^{-4}
children	10	1	10	20×10^{-3}	2.86×10^{-4}
			20	50×10^{-3}	1.43×10^{-3}

step 2: Find the probable risk of additional cancers, R, for adults and children for each time period.

Given a slope factor for orally ingested benzene of 2.0×10^{-2} $(\mathrm{mg/kg{\cdot}day})^{-1}$, use Eq. 45.4.

$$R = (\mathrm{SF})(\mathrm{CDI})$$

Equation 45.4 yields the following probabilities of excess cancer risk.

	ED (yr)	CDI (mg/kg·day)	R
adults	10	8.16×10^{-5}	1.63×10^{-6}
	20	4.08×10^{-4}	8.16×10^{-6}
children	10	2.86×10^{-4}	5.72×10^{-6}
	20	1.43×10^{-3}	2.86×10^{-5}

Step 3: Find the total excess cancers for adults and children for each time period.

From Eq. 45.3,

$$EC = (EP)(R)$$

	EP	ED (yr)	R	EC
adults	45,000	10	1.63×10^{-6}	0.0734
		20	8.16×10^{-6}	0.367
children	5000	10	5.72×10^{-6}	0.0286
		20	2.86×10^{-5}	0.143

The total adult excess cancer risk is

$$\begin{aligned} EC_{a,\text{total}} &= EC_{a,10} + EC_{a,20} \\ &= 0.0734 + 0.367 \\ &= 0.440 \end{aligned}$$

Similarly, the total children excess cancer risk is given by

$$\begin{aligned} EC_{c,\text{total}} &= EC_{c,10} + EC_{c,20} \\ &= 0.0286 + 0.143 \\ &= 0.172 \end{aligned}$$

Because the total number of excess cancers for adults and children is less than 1.0 and 0.5, respectively, the water supply can be used.

Bioconcentration Factors

Besides direct human exposure to toxicants through water ingestion, inhalation, and skin contact, the EPA also has developed *bioconcentration factors* (BCF) so that the human intake from consumption of fish and other foods can be determined. Bioconcentration factors have been developed for many toxicants and provide a relationship between the toxicant concentration in the tissue of the organism (e.g., fish) and the concentration in the medium (e.g., water). The concentration in the organism equals the product of the BCF and the concentration in the medium. Not all chemicals or other substances will bioaccumulate, and the BCF pertains to a specific organism, such as fish.

$$C_{org} = (BCF)(C_w) \qquad 45.5$$

Selected bioconcentration factors (BCF) for selected chemicals in fish are given in Table 45.4. The substances are arranged in descending order of BCFs to illustrate the substances that have a high potential to bioaccumulate in fish. These substances are of great importance when the oral pathway is present in a particular situation.

Table 45.4 *Selected Bioconcentration Factors for Fish*

substance	BCF (L/kg)
polychlorinated biphenyls	100 000
4,4′ DDT	54 000
DDE	51 000
heptachlor	15 700
chlordane	14 000
toxaphene	13 100
mercury	5500
2,3,7,8 tetrachloro dibenzo-p-dioxin (TCDD)	5000
dieldrin	4760
copper	200
cadmium	81
lead	49
zinc	47
arsenic	44
tetrachloroethylene	31
aldrin	28
carbon tetrachloride	19
chromium	16
chlorobenzene	10
benzene	5.2
chloroform	3.75
vinyl chloride	1.17
antimony	1

The BCF factors can be applied to determine the total dose to humans who ingest fish from water contaminated with toxicants that bioaccumulate. This dose would be added to the dose received from drinking the contaminated water.

Example 45.4

The town of Central has a water supply from the Middle River that was discovered to be contaminated with 0.03 μg/L of heptachlor (carcinogen class B2) for 5 yr. People in the town have continually enjoyed the large trout from the Middle River and consume twice the EPA standard factor for fish consumption. The slope factor for heptachlor is 4.5 $(mg/kg \cdot d)^{-1}$. What is the

risk of excess cancer over the lifetime of an adult if the heptachlor contamination is removed this year?

Solution

Determine the standard factors. From Tables 45.3 and 45.4,

$$\begin{aligned}\text{fish consumption} &= (6.5\ \text{g/day})(2) = 13\ \text{g/day}\\ \text{CR} &= 2\ \text{L/day}\\ \text{ED} &= 5\ \text{yr}\\ \text{BW} &= 70\ \text{kg}\\ \text{BCF} &= 15\,700\ \text{L/kg}\\ C_{\text{heptachlor}} &= 0.03\ \mu\text{g/L}\\ \text{SF}_{\text{heptachlor}} &= 4.5\ (\text{mg/kg·day})^{-1}\end{aligned}$$

Calculate the CDI for heptachlor in water.

From Eq. 11.75,

$$\begin{aligned}\text{CDI}_w &= \frac{(C_{\text{heptachlor}})(\text{CR})(\text{EF})(\text{ED})}{(\text{BW})(\text{AT})}\\ &= \frac{\left(0.03\times10^{-6}\ \frac{\text{g}}{\text{L}}\right)\left(2\ \frac{\text{L}}{\text{day}}\right)(1)(5\ \text{yr})\left(10^3\ \frac{\text{mg}}{\text{g}}\right)}{(70\ \text{kg})(70\ \text{yr})}\\ &= 6.12\times10^{-8}\ \text{mg/kg·day}\end{aligned}$$

Calculate the concentration of heptachlor in fish and determine its CDI.

From Eq. 45.5,

$$\begin{aligned}C_{\text{heptaclor/fish}} &= (\text{BCF})(C_{\text{heptachlor}})\\ &= \left(15\,700\ \frac{\text{L}}{\text{kg}}\right)\left(0.03\times10^{-6}\ \frac{\text{g}}{\text{L}}\right)\\ &\quad\times\left(10^3\ \frac{\text{mg}}{\text{g}}\right)\\ &= 0.471\ \text{mg/kg}\end{aligned}$$

From Eq. 11.75,

$$\begin{aligned}\text{CDI}_{\text{fish}} &= \frac{\left(0.471\ \frac{\text{mg}}{\text{kg}}\right)\left(13\ \frac{\text{g}}{\text{day}}\right)(1)(5\ \text{yr})\left(\frac{1\ \text{kg}}{10^3\ \text{g}}\right)}{(70\ \text{kg})(70\ \text{yr})}\\ &= 6.25\times10^{-6}\ \text{mg/kg·day}\end{aligned}$$

The total CDI is

$$\begin{aligned}\text{CDI}_{\text{total}} &= \text{CDI}_w + \text{CDI}_{\text{fish}}\\ &= 6.12\times10^{-8}\ \frac{\text{mg}}{\text{kg·day}}\\ &\quad+6.25\times10^{-6}\ \frac{\text{mg}}{\text{kg·day}}\\ &= 6.31\times10^{-6}\ \text{mg/kg·day}\end{aligned}$$

From Eq. 45.4, the excess cancer risk is

$$\begin{aligned}R &= (\text{SF}_{\text{heptachlor}})(\text{CDI}_{\text{total}})\\ &= \left(4.5\ \frac{\text{kg·day}}{\text{mg}}\right)\left(6.31\times10^{-6}\ \frac{\text{mg}}{\text{kg·day}}\right)\\ &= 28.4\times10^{-6}\end{aligned}$$

The total risk of excess cancer is 28.4×10^{-6}.

ACGIH Methods

The American Conference of Governmental Industrial Hygienists (ACGIH) uses methods for determining the safe human dose that are somewhat different from the EPA methods previously described.

Threshold Limit Values

First, the ACGIH method uses *threshold limit values* (TLV) determined for both noncarcinogens and carcinogens. The TLVs are the concentrations in air that workers could be repeatedly exposed to on a daily basis without adverse health effects. The term TLV-TWA means the maximum time-weighted average concentration that all workers may be exposed to during an 8 hr day and 40 hr week. The TLV-TWA is for the inhalation route of exposure.

ACGIH also determines *short-term exposure limits* (TLV-STEL) for airborne toxicants, the recommended concentrations that workers may be exposed to for short periods during the workday without suffering certain adverse health effects (irritation, chronic tissue damage, and narcosis). The TLV-STEL is the TWA concentration in air that should not be exceeded for more than 15 min of the workday. The TLV-STEL should not occur more than four times daily, and there should be at least 60 min between successive STEL exposures. In such cases, the excursions may exceed three times the TLV-TWA for no more than a total of 30 min during the workday, but shall not exceed five times the TLV-TWA under any circumstances. In all cases, the TLV-TWA may not be exceeded. Short-term exposure limits have not been established by ACGIH for some toxicants.

ACGIH also publishes *ceiling threshold limit values* (TLV-C) that should not be exceeded at any time during the workday. If instantaneous sampling is infeasible, the sampling period for the TLV-C can be up to 15 min in duration. Also, the TLV-TWA should not be exceeded.

For mixtures of substances, the *equivalent exposure* over 8 hours is the sum of the individual exposures.

$$E = \frac{1}{8}\sum_{i=1}^{n} C_i T_i \qquad 45.6$$

The *hazard ratio* is the concentration of the contaminant divided by the exposure limit of the contaminant.