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QUANTITATIVE RISK ANALYSIS FOR TOXIC CHEMICALS IN THE ENVIRONMENT

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Summary

An overview and basic framework is presented for quantification of human health risks associated with toxic chemicals in the environment. The presentation is designed for program managers who require an introduction to the principles of risk analysis and an understanding of the current state-of-the-art. The basic terminology is explained, the methodological components are reviewed, and a series of procedures is discussed for estimating ambient concentrations, effect potency, and human exposure and risk levels for a chemical substance. Practical considerations are discussed, including the uncertainties introduced by data gaps and modelling assumptions. Finally, a simple numerical illustration of the calculation procedures is presented.

1. Overview of environmental risk analysis

Quantitative analysis of human health risks has become increasingly important in the U.S. and other industrialized countries as a means of supporting decisions that affect public health. Risk analysis has frequently been used to evaluate the impacts of new products or new technologies and to support the development of government regulations. In the field of environmental protection, risk analysis has become a useful tool for judging both the degree of risk associated with chemical pollutants and for selecting control strategies that can reduce these risks to an acceptable level. Despite the many uncertainties that can arise, risk analysis is the only available means of quantifying the benefits of environmental control, and will continue to be applied and refined as further experience is gained with toxic chemicals.

The usual purpose of an environmental risk analysis is to quantify the degree of human exposure to one or more pollutants, and to estimate the potential adverse effects of such exposure. The scope of the analysis can vary considerably depending on the intended use of the results. Some of the factors that must be considered in defining the boundaries of the analysis are the following:

- geographic scale, which may be global, national, regional or local;
- pollutant sources, which may include industrial, residential, commercial or natural sources, including both point and nonpoint releases;

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- environmental pathways, which may include air, surface water, soil, groundwater, and the biotic food chain;
- exposure routes, which may include ingestion, inhalation and dermal absorption;
- adverse effects, which may include acute or chronic human health effects, ranging from mild reversible effects to mortality — economic and environmental effects are not considered here;
- time frame of the assessment, which may be retrospective, current, or prospective.

The output of a risk analysis will generally be an estimate of the fraction of the exposed populations that are expected to experience adverse health effects (or equivalently, the probability that a random individual will be affected). Due to the statistical nature of toxic effects, particularly at low doses, it is rare that a definitive statement can be made about whether or not an effect will occur. Nevertheless, the results of a probabilistic risk analysis can be used in a variety of ways to guide policy making and priority setting. Possible applications of risk analysis include:

- comparison of risk levels among chemicals or comparison against other risks, such as automobile transport;
- comparison of the impact of different routes or pathways of exposure such as inhalation versus ingestion;
- comparison of source-proximate (near-field) subpopulations to the rest of the population (far-field), in the case of point source pollution;
- identification of the sources responsible for the greatest contribution to risk, both in terms of population affected and magnitude of individual risk;
- identification of sensitive subpopulations or high-risk areas in the region being studied;
- estimation of risk reduction or redistribution under various intervention scenarios.

Although risk analysis is a complex endeavor, not easily summarized, this paper attempts to describe a general framework for performing risk analysis. In Section 2 the important components of risk are explicitly defined and related within a conceptual hierarchy. Section 3 discusses some of the practical issues that arise in implementing a risk analysis, and a numerical illustration of such an analysis is provided in Section 4. Finally, the existing literature on risk analysis is surveyed, and additional references are provided for those who require in-depth methodological information. The intent of this paper is to provide a concise introductory guide to human health risk analysis, focusing particularly on chronic effects, at a level of detail more appropriate for managers than for analysts and practitioners.

2. A framework for risk analysis

2.1 Introduction

Risk may be defined as the potential for negative consequences of an

event or activity. In the context of assessment of risk from environmental pollutants, the event or activity is the release of a pollutant into, and its subsequent traverse through, the environment such that humans and other biota are exposed, and the negative consequences are any adverse effects on the exposed populations. Thus, if a pollutant is believed to be harmful and if it is present in the environment, there is certainly a potential for exposure and subsequent harm; that is, some risk exists. The purpose of risk analysis is to go beyond such a qualitative statement of potential risk, by estimating or measuring this potential.

Although the nature of adverse effects may be well understood, the key difficulty in risk estimation lies in determining the probability that adverse effects will occur. The probability is comprised of two factors:

- The likelihood that groups of humans will be exposed to various levels of the pollutants.
- The likelihood that exposed humans will experience adverse effects.

These two factors correspond to the two major branches of investigation in risk analysis — exposure and effects.

Analyzing the probability of adverse effects of different pollutants will present different types of problems, depending upon pollutant properties and effects. For a highly persistent substance that is present in the human diet and known to have long-term effects, the main challenge lies in estimating the likelihood of adverse effects based upon observed exposure levels. On the other hand, for a substance that is degraded rapidly and appears only in scattered locations, but is known to be an acute toxicant, the focus should be on estimating the likelihood of exposure. Therefore, a risk analysis framework must be flexible enough to encompass these and a multitude of other situations.

For a population of susceptible humans, risk may be expressed in several ways. One can state the probabilities that certain fractions of the population will be adversely affected (e.g., 5% chance that 9/10 will be affected, 20% chance that 1/3 will be affected). This sort of quantitative estimate is usually difficult to achieve. Alternatively, one can state the expected number that may be affected, allowing a certain margin for error to reflect uncertainties in the underlying data (e.g., 200,000 ± 50,000). Finally, one can give an order-of-magnitude estimate that has no real measure of confidence attached to it (e.g., at most 5% will be affected). Each of these ways of expressing the degree of risk can be more detailed in terms of types of effects, e.g., the chance of a specific disease, premature death, extent of disability, etc.

Hence, risk estimates may be classified into three types, corresponding to decreasing level of precision with which the population at risk and the degree of risk can be characterized:

- probability distribution,
- numerical interval, and
- order of magnitude.

The level of precision of a risk estimate cannot exceed the precision of the exposure and effects data from which it is obtained. In cases where probabilistic risk estimates cannot be obtained, it may be possible to develop a range or numerical interval of risks. In other cases, lack of data may preclude any process other than the most general or comparative estimate of risk.

2.2 Terminology

In the practice of human health risk assessment, there are a number of terms which are frequently used and occasionally misunderstood. For purposes of consistency and clarity, the following definitions are suggested, and will be utilized throughout this paper:

Risk: the probability of an adverse outcome; thus an individual's risk of liver cancer is the probability that he will be afflicted during his lifetime. *Cumulative risk*, or population risk, is the number of cases of a specific effect expected in a given population. *Excess risk* is the increase in probability of an effect associated with a specific cause (e.g., exposure to a toxic substance).

Hazard: an event or situation that may present a threat to human health and safety. A hazard can be described in qualitative terms, whereas the risk associated with that hazard can be expressed in quantitative terms (e.g., cigarette smoking is a hazard).

Exposure: human inhalation, ingestion, or dermal absorption of a chemical substance in an environmental, occupational, dietary, or other context. The exposure *route* or *pathway* is the specific mechanism whereby the exposure occurs (e.g., drinking of contaminated groundwater). The quantity of a chemical inhaled, ingested, or absorbed is called the *intake*. *Whole-body* exposure is often distinguished from *target-organ* exposure; the latter depends on pharmacokinetic factors such as metabolism and excretion. Another important distinction is between *acute* (i.e., short-term) and *chronic* (i.e., long-term) exposure, which usually have different effects.

Dose: the quantity or concentration of chemical intake in either an experimental or real-life situation. For chronic exposures, the *dose rate* is the rate of intake per time unit.

Toxicity: the capability of a chemical substance to induce adverse effects in exposed humans. (For example, *carcinogenicity* is the capability of a chemical to induce cancer. More specific classes of toxicity include teratogenicity, mutagenicity, leukemogenicity, etc.) The degree of a chemical's toxicity relative to other substances is called its *potency*. The lower the dose required for a particular toxic effect, the higher the potency.

Susceptibility: the vulnerability of humans to toxic effects when exposed to chemical agents. Susceptibility is usually variable in human populations, so that the effective potency of a chemical will differ among exposed individuals.

2.3 Conceptual framework

The elements of risk analysis defined above may be logically related

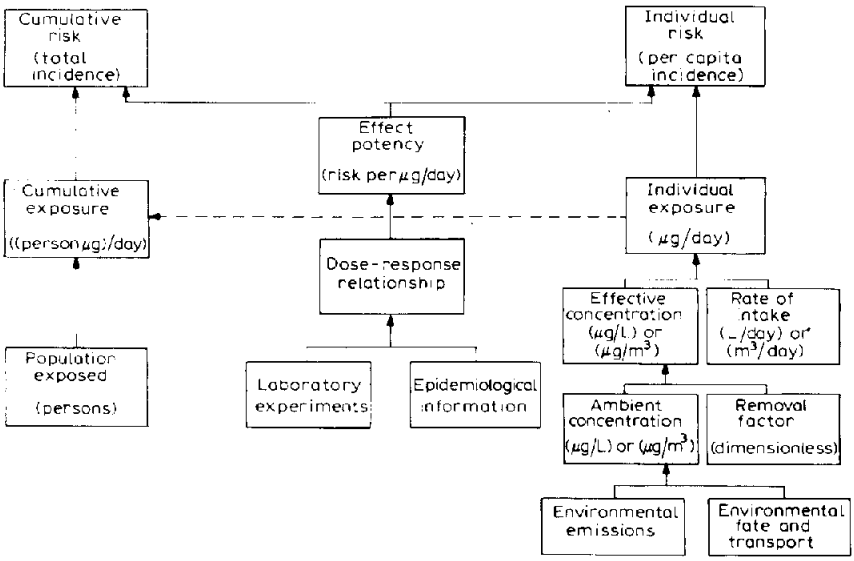

Fig. 1. Risk analytic framework for chronic human effects of environmental pollutants.

according to a hierarchical framework, as depicted in Fig. 1. This framework shows how various items of information are usually combined to produce a quantitative assessment of human risk, expressed in either individual or cumulative terms. The focus of this scheme is upon chronic effects in the case of environmental exposures, though it can easily accommodate occupational, dietary, or household exposures. However, dealing with acute effects requires a substantively different approach.* The most commonly used units of measurement are also displayed for each data element.

There are two key items in the framework that are critical to the quantification of risk, namely the effect potency and the effective concentration. Methods for estimating these items are discussed in Sections 3.2 and 3.3, respectively. The *effect potency* measures the increase in human risk as a function of dose rate, and in the simplest case can be expressed as a linear "unit risk" coefficient. The *effective concentration* is used to determine the dose rate, and represents the level at which the chemical intake occurs in the medium of interest. For example, if formaldehyde is present in inhaled air at an effective concentration of 10 $\mu g/m^3$ and a person typically inhales 20 m^3/day, then the individual whole-body exposure is about 200 μg/day or 0.2 mg/day. If the unit risk of cancer is 0.0003 per mg/day then the individual's excess lifetime risk is $0.2 \times 0.0003 = 6 \times 10^{-5}$; this result represents the estimated increase over normal cancer risk due to formaldehyde exposure.

*Since acute effect thresholds are generally known, the key analytic issue in risk assessment is quantifying the likelihood of exposure to various dose levels.

The framework in Fig. 1 also permits the calculation of cumulative risk when the size of the exposed population is known. For example, if the number of people exposed at the level computed above is two million, then the cumulative exposure is 4×10^5 (person mg)/day and the cumulative risk turns out to be 120. Put another way, the estimated excess incidence of cancer in the exposed population is 120 cases out of two million people over their lifetime. Of course, there are differences in susceptibility depending upon age, sex, and other factors, so that this type of crude population average assumes a homogeneous population with identical background factors. The cumulative risk is sometimes expressed as an annual rate of increased cancer; for example, if the average lifetime of the population were 60 years, then the excess risk would be 120/60 = 2 additional cancers per year due to formaldehyde exposure.

The above calculations are deceptively simple, in that they mask a large set of technical issues which must be addressed in a formal risk assessment. For example, the level of exposure is unlikely to be uniform for all individuals, due to differences in occupational and domestic activities and locations. Similarly, the susceptibility of individuals may vary across different subgroups. It is usually necessary to identify subpopulations within the total population exposed, and to separately analyze the exposure and risk levels for each subpopulation. Another important issue is the quantification of uncertainty, which is discussed in greater depth in Section 3. It is generally impossible to develop precise risk estimates due to uncertainty both in the available data and in the models which are used to calculate effect potency and effective concentration. Therefore, the results of a risk analysis should be expressed as a credible range of risk, which may span several orders of magnitude in the case of extreme uncertainty.

3. Practical considerations

3.1 Introduction

The conceptual framework presented in Section 2 may readily be translated into a set of procedures for performing a risk analysis. Once the pollutants, exposure pathways and population of concern have been identified, the analyst will normally proceed in the following sequence:

1. Estimate the effect potency
2. Estimate the ambient concentration in the environment
3. Estimate the effective concentration to which humans are exposed
4. Calculate the individual exposure
5. Calculate the cumulative exposure for the population (if desired)
6. Calculate the individual or cumulative risk

Each of these steps presents a host of difficulties in terms of both data restrictions and complexity of biological and environmental mechanisms. While these difficulties can be overcome by using simplifying assumptions, subjective judgements, and mathematical models, such techniques inevitably

introduce uncertainties into the analysis. In particular, steps 1 and 2 present the greatest challenge, and are dealt with separately in subsequent sections. Step 3 involves the conversion of an ambient environmental concentration to an effective concentration; that is, the actual concentration to which human subpopulations will be exposed. This is accomplished through the use of a *removal factor*, which represents reduction in concentration due to intervening factors such as drinking water treatment or air filtration in buildings. Steps 4 through 6 are computationaly straightforward and were illustrated in Section 2.3.

For other sorts of toxic chemical exposure, the framework in Fig. 1 may be readily modified to show the differences in analytic methodology. For example, dietary exposure may be addressed by letting effective concentration represent the contamination level in foodstuffs. Alternatively, if bioconcentration via the food chain is quantifiable, then the removal factor can be replaced by a bioconcentration factor which is multiplied by the ambient concentration; this is often done for edible aquatic organisms. Dermal absorption may be computed by estimating the rate of percutaneous intake, although it is generally much lower than ingestion or inhalation intake. Many other variations are possible on this general scheme, provided that they preserve the fundamental distinction between exposure and effects. However, in some cases the available data on effects are inadequate for quantifying potency, so that the risk analysis must be constrained to exposure estimation with qualitative statements about potential risk.

3.2 Estimation of effect potency

The basis of any potency estimate for a specific substance is a set of data, obtained experimentally or through field observation, describing the effects of that substance upon a population of organisms. Of course, human data are preferable for estimation of human risk, but epidemiological studies suffer from difficulties in quantifying exposures, as well as from a host of confounding factors, such as genetic and lifestyle variations. For these reasons risk analyses are more commonly performed using laboratory data for one or more species of mammals. Dose—response measurements are usually provided at several dose levels, including a control group which receives no dosage. The scientific quality and reliability of these data are an important consideration in risk assessment. For example, if the exposure route and experimental regimen employed (e.g., intramuscular injection) do not agree with the most likely mode of human exposure, the data must be interpreted cautiously.

In order to extrapolate laboratory animal results to humans, an interspecies dose conversion must be performed. Animals such as rodents have different physical dimensions, rates of intake (ingestion or inhalation), and lifespans from humans, and therefore are expected to respond differently to a specified dose level of any chemical. Estimation of equivalent human doses is usually performed by scaling laboratory doses according to observable spe-

TABLE 1

Dosage conversion chart

Species	Assumptions[a]			Dose	Human equivalent dose				
	Weight (kg)	Rate of ingestion (kg/day)	Rate of respiration (m^3/day)		Total μg/day	Food (ppb)	Contact (μg/L)	Breathing (ng/m^3)	Acute (mg)
Human	70	2	10.7						
Mouse or hamster	0.025	0.003	0.033	1 μg/day	2800				
				1 ppb	8.4	4.2			
				1 μg/L	0.002		1		
				1 ng/m^3	0.09			8.6	
Rat	0.3	0.015	0.14	1 μg/day	233				
				1 ppb	3.5	1.75			
				1 μg/L	0.002		1		
				1 ng/m^3	0.03			3	
Any mammal (acute effects)				1 mg/kg					70

[a]Equivalent human doses are assumed proportional to weight and rate of intake. Rates of respiration are based on minute volume while resting. Source: Handbook of Biological Data, W.S. Spector.

cies differences (see Table 1). Unfortunately, detailed quantitative data on the comparative pharmacokinetics of animals and humans are nonexistent, so that scaling methods remain approximate. In carcinogenic risk extrapolation, it is commonly assumed that the rate of response for mammals is proportional to internal surface area. Although other bases for conversion (e.g., body weight) have been utilized, the surface area method is the most widely accepted. This approach is more conservative, yielding risk estimates about an order of magnitude greater than those derived from scaling by body weight.

When investigating chronic effects, one is usually interested in environmental exposure levels that are far below the typical concentrations needed to produce statistically significant results in laboratory animals. Therefore, it is customary to use dose—response models which extrapolate from the experimental doses to the range of human dosage. (See Tables 2 and 3.) Each model postulates a different shape of dose—response curve at low doses, so that by using several models, a range of uncertainty can be established between the least conservative and the most conservative results. The resulting range of uncertainty will frequently dominate any of the uncertainties generated in the exposure estimation procedures. While the specific mathematical methods of dose—response estimation are beyond the scope of this paper, they are well documented in the references discussed in Section 6.

One important point of controversy in risk extrapolation is the existence of a threshold level for carcinogenic and mutagenic response to a pollutant. Some argue that an organism is able to cope with low doses of a substance through metabolic processes or repair mechanisms, so that harmful effects do not appear until a certain minimum threshold, or "safe dose", is surpassed. Others contend that a carcinogenic substance must be considered

TABLE 2

Carcinogenic response in B6C3F1 mice DEHP in the diet for two years[a]

	Dosage (mg/kg)	Equivalent human dose (mg/day)	Response[b]	Percent
Male mice	0	0	9/50	18
	3000	1800	14/48	29
	6000	3600	19/50	38
Female mice	0	0	0/50	0
	3000	1800	7/50	14
	6000	3600	17/50	34

[a]Source: National Toxicology Program, Carcinogenesis Bioassay of di(2-ethylhexyl) phthalate, DHHS 81-773, 1980.
[b]Hepatocellular carcinoma.

TABLE 3

Probable upper bounds on expected excess lifetime cancers per million population due to DEHP ingestion[a]

	Exposure level (mg/day)				
	0.01	0.1	1	10	100
One-hit model	1	10	100	1000	10,000
Log-probit model	—	0.3	30	1000	20,000
Multi-stage model	0.5	5	50	500	5,000

[a]Source: Arthur D. Little, Inc.

potentially harmful at any dose, and that even a single molecule may initiate a tumor at the cellular level. This is the so-called "one-hit" hypothesis. The question of existence of a threshold has often been circumvented by the approach of selecting an "acceptable" risk level and determining the corresponding acceptable or "virtually safe" dose (VSD). However, from a practical point of view, the behavior of the dose–response curve at low doses is an academic question. Due to realistic restrictions on sample size in animal experiments, it is extremely difficult to statistically reject the no-threshold hypothesis, so that the threshold issue may never be satisfactorily resolved through purely statistical arguments.

It is important to note that for a particular substance the assumption of carcinogenicity to humans may be false, even though it is a proven carcinogen in several animal species. In such a case, the lower bound on the excess risk to humans is effectively zero, in the sense that zero-risk is a possibility which cannot be dismissed. Thus, the risk estimates obtained through dose–

response extrapolation must be regarded as probable upper bounds on the true human risk.

3.3 Estimation of ambient concentration

A crucial step in any environmental risk analysis is to characterize the presence of a selected chemical in the environment. There are three major types of practical information that can be used as a basis for subsequent assessments of exposure and risk. These are monitoring data, environmental emissions, and environmental fate and transport information.

Many government agencies maintain computerized environmental data bases from which average ambient and effluent concentrations in air and water can be established, as well as concentrations in soil, sludges, plants, animals, human tissues, foods and drinking water. Some of the more common problems with these monitoring data include: uncertainties in the chemical analytical procedures used, confidence levels, and detection limits; uncertainties in obtaining representative samples of the environmental media; lack of data on the temporal variations in concentrations at different locations; uncertainties in the chemical or physical forms of the pollutant; and lack of sufficiently detailed and/or extensive data. Despite these limitations, monitoring data can provide an excellent indication of the locations of pollutant releases to the environment, a potential means for assessing exposure of humans and other biota, and a direct means of confirmation of environmental releases and pathways analysis.

Ideally, the ambient concentration of a specific pollutant should be estimated through detailed field monitoring. In practice, however, the difficulty and expense of monitoring is prohibitive unless the area of concern is extremely small. Moreover, for chemicals that have not yet been introduced to market, field data cannot be obtained. As a result, it is often necessary to use information about pollutant emissions, fate, and transport in order to estimate ambient concentrations. In an environmental risk analysis the following types of emissions data are useful:

- Identification of significant pollutant sources within an industry segment or geographic area.
- Identification of the chemical and physical forms of the pollutant.
- Characterization of the environmental loading of the pollutant — quantities, geographic locations, rates, receiving environments.
- Identification of uses and releases of the pollutant leading to direct exposure.
- Achievement of a numerical balance between production and uses or releases, if possible.
- Establishment of confidence or uncertainty in observations or estimates of the releases of the pollutant.

Development of an environmental emission profile requires a systematic identification of sources, estimation of releases, and characterization of the environment into which the releases occur. All types of manufacturing pro-

cesses, transportation, storage, and disposal activities, as well as uses of the pollutant or products in which the pollutant is a contaminant, may be considered. In many cases quantitative data on releases may not be available, so that engineering estimates will have to be made in order to ascertain likely or expected environmental releases.

The environment is not static — pollutants are transported, undergo transformation, accumulate and degrade — so that the environmental distribution of a pollutant is different from that associated directly with environmental releases. Therefore, fate and pathways analyses are frequently used to estimate the actual distribution of a pollutant in the environment in the absence of adequate monitoring data. Specific goals of an environmental fate and pathways analysis can include:

- Definition of environmental compartments of importance.
- Identification of important mechanisms for transport, physical, biological, and chemical change in environmental media and of predominant chemical forms of the pollutant.
- Summarization of transfer and reaction rates, controlling processes and lifetimes of the pollutant in the environment.
- Tracing of pollutant pathways from sources to sinks.
- Estimation of pollutant concentrations in different environmental media and their time dependence.
- Comparison of the results of pathways and fate analysis with monitoring data.
- Provision of quantitative relationships between environmental releases and exposure.

A variety of modeling approaches has been used to estimate pollutant concentrations in exposure media. These range from qualitative estimates extrapolated from case examples or environmental scenarios, simple analytical equilibrium or transport models, to complex multi-media models. For many environmental situations, adequate models do not exist or are just now under development. Furthermore, for new or uncommon chemicals, many of the physical, chemical, and biological properties needed to estimate transformation rates, persistence, and distribution are not available. For example, few models exist to predict adequately the distribution of pollutants released from a landfill into groundwater and surface water. Therefore, it is important that limitations and uncertainties in the model results be clearly identified. While a thorough discussion of the state-of-the-art of fate and transport modeling is beyond the scope of this paper, Section 6 provides a number of useful references.

3.4 Multiple risk factors and multiple effects

One of the most severe problems affecting human health risk analysis is the fact that exposures to toxic substances do not occur in isolation. Each individual is exposed to a host of potential toxic agents over his or her lifetime, including air and water pollutants, food contaminants, cigarette

smoke, industrial chemicals, and both natural and man-made radiation. Therefore, in predicting the additional risk imposed upon a population by a specific substance, one should ideally account for the baseline effect incidence within that population due to all other risk factors, including natural causes. Unfortunately, the relationships between risk factors are not simply additive. Some chemicals are believed to interact in a "synergistic" way, resulting in a higher risk than would be calculated by simply combining their separate potencies. Many other types of interaction are possible, and available data to quantify these interactions are extremely scarce. When modification of the effect potency to account for multiple factors is not feasible, the risk analyst should at the very least identify the possible influence of synergistic or other interactions.

Another issue that should be considered in health risk analysis is the potential for a single chemical substance to have multiple effects. For example, lead is known to have several different types of chronic effects, including neurobehavorial, hematological, and renal toxicity. It is not unusual for a chemical to have both acute and chronic effects, although the acute effects will generally occur at higher dose levels. In a risk analysis for a large population, provided the response probability is sufficiently low, one can estimate the incidence of the various effects separately, using the appropriate potency or dose—response relationship for each effect. However, the likelihood that a single individual experiences two or more different chronic effects as a result of exposure to a single substance is difficult to predict. It depends greatly on the chemical's pharmacokinetic properties and on the specific effects being considered. In short, independence of effects may be assumed for populations but not for individuals.

4. Numerical illustration

The risk analysis framework presented in Section 2 can readily be demonstrated by a hypothetical example. Assume that the following information is known for a large metropolitan area:

- The city water department serves three million people with treated surface water.
- The chemical 1,2-dichloroethane has been detected in the drinking water treatment plant influent at a level of 5 μg/L.
- The water treatment processes are believed to remove 20% of that chemical (i.e., the fraction passing through to the consumer is 0.8).
- The chemical is a suspected carcinogen with a linear effect potency at low doses estimated to be at most 5×10^{-8} per μg/day, according to the U.S. EPA Carcinogen Assessment Group.

The risk calculations described in Section 3 can readily proceed as follows based on the information given. They are illustrated for convenience in Fig. 2.

Chemical: 1,2-dichloroethane
Medium : drinking water
Effect : cancer

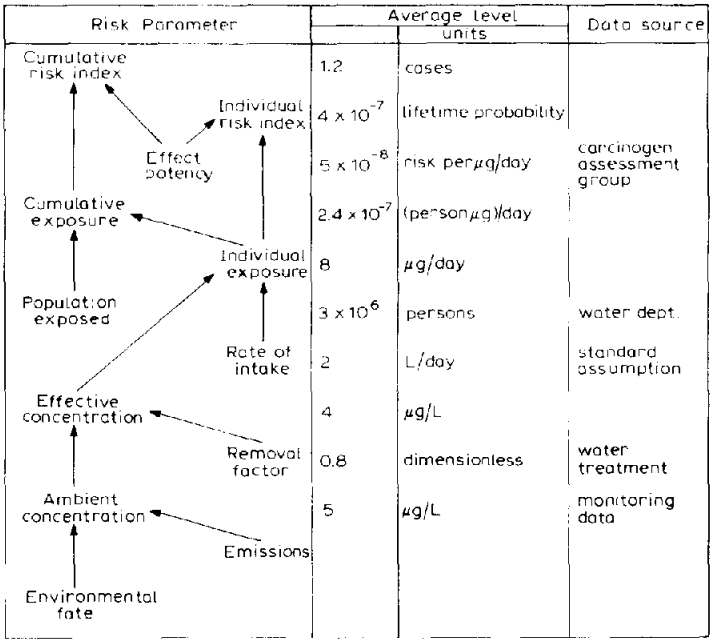

Fig. 2. Risk analysis summary sheet.

Step 1: Potency = 5×10^{-8} per μg/day

Step 2: Ambient concentration = 5 μg/L

Step 3: Effective concentration = $5 \times 0.8 = 4$ μg/L

Step 4: Individual exposure = 4×2 L/day = 8 μg/day

Step 5: Cumulative exposure = $8 \times 3 \times 10^6 = 2.4 \times 10^7$ (person μg)/day

Step 6: Individual risk = $8 \times 5 \times 10^{-8} = 4 \times 10^{-7}$ (lifetime)

Cumulative risk = $2.4 \times 10^7 \times 5 \times 10^{-8} = 1.2$ cases

Thus, the chances of an average individual experiencing an excess cancer, i.e., over and above normal cancer incidence, due to the presence of 1,2-dichloroethane in the water supply is at most 4×10^{-7}, a relatively low risk level. Only 1.2 excess cancer cases are anticipated in the entire exposed population.

This example is deliberately simplified, in that the exposure level is constant for the entire population. Ordinarily, a risk analysis would require a more complicated description of variations in exposure, and would use

sensitivity analysis to examine the implications of uncertainty in the available data. Also, no effort was made to identify the sources of 1,2-dichloroethane or the surface water dispersion patterns. Had the risk estimate been higher, a fate and transport analysis might be necessary to determine appropriate control levels for emission sources. Nevertheless, the basic principles of risk analysis are fully exhibited by this example, and are indeed quite elementary.

5. Acknowledgment

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6. References

Risk analysis is still an emerging field, and many of the techniques described in Section 3 are evolving rapidly. A thorough survey of the literature is beyond the scope of this paper, but a number of reference books are provided below to guide the risk analyst in the utilization of up-to-date methods. In addition, there are hundreds of government reports and journal publications referenced in these books that deal with specific topics such as pathway analysis and dose—response extrapolation.

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