# A Primer on Environmental Risk Analysis

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## ABSTRACT

Throughout history, hazardous material has represented a risk to individuals. The task of regulating risk can be overwhelming. A regulator must interpret and coordinate a wide variety of information from experts and the lay public. To aid the regulator, the loosely coordinated field of environmental risk analysis has evolved.

This paper provides a nontechnical examination of the four steps in the analysis of risks associated with an environmental hazard. The steps are (1) risk assessment--the use of scientific data to quantify risk, (2) risk perception--the manner in which individuals or societies perceive risk, (3) risk valuation--monetary valuation of reductions in risk, and (4) risk management--controlling risk in the "best" interests of societies. Each step is examined in terms of its tasks and possible problems. The paper provides a reference for regulators who must understand both the insights and the limitations of environmental risk analysis.

## Introduction

Hazardous material often represents a risk to individuals and society.[1] The more toxic, flammable, radioactive, or corrosive a substance, the more an individual is exposed, the more public health is at risk. Whether it is contaminated groundwater, carcinogenic by-products of herbicide use, or dioxin exposure, a regulator must manage an ever-growing list of hazardous material.

For example, in recent years society has become increasingly concerned about the quality of groundwater used for human consumption. Concern arises because the primary means for disposing of the estimated annual 34-51 million metric tons of hazardous industrial waste is on landfill sites. Landfill sites often have inadequate safeguards to prevent hazardous substances from seeping into water supplies. The U.S. Environmental Protection Agency estimates that 30,000-50,000 hazardous waste dump sites exist, of which at least 2,000 sites pose imminent risk to public health. Communities are concerned that hazardous material will migrate from new or abandoned waste sites and contaminate drinking water supplies.[2] As community pressure for political action increases, a regulator confronts the unenviable task of collecting and interpreting information about the risk.

The task of regulating risk can be overwhelming. Among other duties, a regulator must coordinate and interpret information on the nature of the risk, assess the scientific accuracy of the information, transform public hysteria into a well-reasoned community dialogue, and determine the economic feasibility of reducing a risk to an acceptable level. Obviously, the regulator cannot do this alone. To aid the regulator, the general field of risk analysis has evolved.

Risk analysis is a loosely knit network of research disciplines (e.g., toxicology, radiology, epidemiology, geology, chemistry, atmospheric sciences, engineering, economics, psychology, philosophy, management science) that collects and interprets information on differing aspects of risk. The technical, economical, psychological, ethical, and policy information is then passed to the regulator. To substantiate a regulatory decision, a regulator must understand both the insights

and the limitations of risk analysis information. The purpose of this paper is to serve as a nontechnical primer on risk analysis applied to hazardous material and human health.[3] The primer is not designed to be exhaustive in detail, but rather a simple, quick reference.

We separate risk analysis into four interrelated steps: risk assessment, risk perception, risk valuation, and risk management.[4] Each step is examined in terms of tasks and potential problems. The four steps of risk analysis may be defined as follows:

1. Risk assessment--the use of science and engineering to quantify the relationship between exposure, the probability of exposure, and the likely influence on public health and safety in terms of morbidity and mortality.
2. Risk perception--the examination of how an individual or a society's subjective probability estimates and other psychological factors (e.g., dread, familiarity) influence choice under risky situations.
3. Risk valuation--the estimation of the economic value of a reduction in risk in order to determine the optimal level of risk and safety in society.[5]
4. Risk management--the attempt by the public and policymakers to regulate or control risk in the "best" interest of society.

By understanding both the insights and the limitations of environmental risk analysis, the regulator can remain flexible while providing effective policy alternatives. The priorities for effective regulation of risk include (1) acknowledging that the assessment of risk is not without difficulty and should be presented as a range of possibilities, (2) reconciling expert and lay perceptions of risk by allowing individuals to scrutinize the regulator's policy recommendations and by providing effective information programs, (3) recognizing the trade-off between cost-effective regulation and other noneconomic goals such as zero-risk policies, and (4) convincing the public that risk management policies are in its best interest. By taking these priorities into consideration the regulator can transform the technical analysis into a humane policy tool through which the regulator can select an acceptable level of risk.

The following four sections of this primer describe the four steps of risk analysis. The final section presents the summary and conclusions.

## Risk Assessment

Risk assessment estimates the likelihood of adverse health consequences from exposure to an environmental hazard (U.S. Federal Register 1986).[6] Risk assessment is an explicit, orderly, and rigorous technique to deal with complex issues in determining whether a hazard exists and the potential adverse effects of the hazard. Although uncertainties exist due to limited data and imperfectly understood dose-response relations, risk assessment categorizes the available evidence so that regulators have better information for environmental risk management.

A complete risk assessment has both qualitative and quantitative components. Hazard identification is the qualitative component in which the inherent adverse effect of a hazard is determined. The qualitative assessment examines the likelihood that a hazard is a human carcinogen, mutagen, or developmental toxin. Note that risk is not actually assessed by hazard identification. Rather, the question to be answered is whether it is correct to infer that adverse effects occurring in one setting can be transferred to another setting (Environs 1986).

Risk is actually assessed in the quantitative component, which consists of three steps: the dose-response estimate, the exposure assessment, and risk characterization (National Academy of Sciences 1983). These steps provide a numerical estimate of the potential adverse health consequences of exposure. Given the uncertainties in the evidence, a numerical estimate of risk is not to be taken as a magical number. Rather, it is to be presented as an estimate that is conditional on assumptions and scientific judgment.

The following subsections consider the qualitative and quantitative components in more detail.

### Hazard Identification

Hazard identification is a qualitative assessment that collects, organizes, and evaluates all the relevant biological and chemical information to determine whether a hazard may be a risk to public health. The goal is to determine whether a hazard warrants the attention of a full-scale quantitative assessment. If a hazard has generated evidence of carcinogenicity, mutagenicity, or developmental toxicity, then the need exists for quantitative risk assessment.

Hazard identification is designed to present a weight-of-evidence ranking on the potential adverse health effects from exposure to a hazard. The weight-of-evidence ranking is derived from two major information sources supplemented with the evaluation of supporting information (U.S. Federal Register 1986). The first major source of information is the evidence derived from long-term animal studies. Hazards that generate evidence of positive adverse effects in such studies are generally considered high risk unless there is substantial contradictory evidence. In he case of carcinogenic hazards, the weight-of-evidence ranking increases given an increase in the number of affected tissue sites, an increase in affected animal species or strains, a decrease in the time to tumor or death, or an increase in the proportion of mutagen to benign tumors. Unless there are biological reasons to consider animal data irrelevant to humans, it is generally assumed that adverse human effects can be inferred from adverse effects on lab animals.

The second major source of information is human studies. The role of epidemiologic studies in providing data for risk assessment has become more prominent than at any time in history (Hoffman 1989). The major advantage of epidemiology is the direct human evidence of health effects. Given adequate characterization of exposed and control groups and proper identification of biases and confounding variables, the weight-of-evidence ranking increases with the number of studies showing comparable results of hazard exposure on different populations. Note that epidemiological studies are capable only of detecting comparatively large increases in relative risk (U.S. Federal Register 1986). Therefore, the nondetection of adverse effects does not rule out potential health risks. To determine the effect, the human information must be examined in relation to the animal data and bearing in mind the following types of supporting information.

First, the chemical-physical properties and routes and patterns of exposure must be evaluated. Hazards that are highly toxic and highly mobile, and that have numerous exposure pathways, are of most concern to the risk assessor. Second, the structure-activity correlations that refute or support predictions of adverse effects should be considered. On the basis of structure-activity relationships, the U.S. Environmental Protection Agency (EPA) has developed toxic equivalency factor methodologies for assessing the risk of untested compounds. Third, the relevant metabolic and pharmacokinetic properties of the hazard should be reviewed. This

includes reviewing the metabolic pathways of exposure, the biological dynamics (absorption, metabolism, distribution, and elimination), and the effects of single exposure and multimedia exposure to hazard (see McKone and Kastenburg 1986). Fourth, the toxicologic effects such as organ damage or suppression of the immune system should be examined to determine the interaction of toxicity and the carcinogenic or mutagenic effects of a hazard. Finally, short-term tests for DNA damage, in vitro transformation, and chromosome observation can provide useful information to characterize risk. Two approaches used in the short-term testing for developmental toxicity include in vivo mammalian screening (tests using a pregnant mammal) and in vitro screening (tests employing subjects other than pregnant mammals) (U.S. Federal Register 1986).

Hazard identification provides the qualitative information necessary for the risk assessor to weight all available evidence to determine whether a quantitative assessment is warranted. By ranking the evidence in terms of uncertainties and assumptions, the decision to further assess the risk can be justified. For example, the EPA ranks the overall risk of carcinogenicity in five weight-of-evidence groups: A--carcinogenic to humans; B--probably carcinogenic to humans; C--possibly carcinogenic to humans; D--not classified as to human carcinogenicity; and E--evidence of noncarcinogenicity for humans. Only hazards in groups A or B are unambiguously regarded as suitable for quantitative risk assessment.

**Dose-Response Estimation**

The first step in quantitative risk assessment is the estimation of the dose-response function. A dose-response function quantifies how an individual's health responds to various levels of exposure. Dose is often considered shorthand for exposure to a hazard, and response is the percentage of the exposed population suffering death or illness. Threshold exposure levels can be determined below which only minimal health risk exists.

There are three primary methods of estimating a dose-response function: clinical human studies, interspecies comparison (toxicological method), and epidemiological studies. The clinical human studies involve exposing human subjects to various ambient levels and determining health effects. Because they use human subjects, these studies are costly and controversial. Since

clinical studies seldom provide sufficient information for a risk assessment, the two principal sources of toxicity data are interspecies comparison and epidemiology.

Interspecies comparison and epidemiological studies both use a detailed, disaggregated research technique, but the methods differ in terms of subjects studied. The interspecies comparison method uses animal bioassays to determine the dose-response function. The method attempts to extrapolate the high-dose responses of animals in order to determine the low-dose responses of humans. The method can be described as follows:

1. Laboratory animals are given high doses of a substance. The health response is then examined and recorded over time. A high dose-response function for the animal is then calculated with the available data. High doses for animals are used despite the fact that humans are usually exposed to low doses because the sample size for a low dose animal experiment would be unmanageable and costly, (see, for example, Nichols 1983).

2. Mathematical models extrapolate the high dose-response functions into low dose-response functions for animals. Since 1980, the extrapolation model used by the EPA for carcinogens is the linearized multistage model. The linearized multistage model estimates the plausible upper bound to the risk, but not necessarily a realistic prediction of risk. Since the true risk is unknown, both the upper and lower bounds should be reported (U.S. Federal Register 1986). Note that when pharmacokinetic data or metabolic data suggest that an alternative low-dose extrapolation model is more appropriate, then the EPA can use the alternative. See Nichols (1983) or Schneiderman (1980) for a description and comparison of the alternative extrapolation models. In the case of extrapolating mutagenicity risk, the EPA concentrates solely on data from in vivo heritable mammalian germ-cell tests, until other tests can be determined to be equivalent (U.S. Federal Register 1986). The animal assays provide data on the frequency of induced mutations resulting in adverse health effects. Human risk is assessed by extrapolating the induced mutation frequency downward to the anticipated level of human exposure.

3. The low dose-response function for animals is extrapolated into low dose-response functions for humans. Two commonly used techniques for extrapolating between species are

conversion based on relative weights (mg per kg body weight per day) or relative surface areas (mg per $m^2$ body surface per day).

The qualitative evidence provided by the hazard identification should be used to guide the dose-response estimation. Evidence suggests that extrapolation between species can be improved by utilizing pharmacokinetic data of the hazard in the animal species and in humans. Incorporating the relevant information on absorption, metabolism, distribution, and elimination of an agent may be helpful in developing a more accurate interspecies dose-response function. One other promising technique for interspecies comparison involves examining relative molecular structures of the species and humans. These more accurate procedures based on knowledge of biological mechanisms may replace the current practices of extrapolating between doses and between species (North and Yosie 1987).

The epidemiological method estimates dose-response functions by observing human health in natural surroundings. Two types of epidemiological studies are cohort studies and case-reference studies. A cohort study compares a group of individuals exposed to the contaminant to a control group. A cohort study can be retrospective or prospective in nature. A case-reference study compares individuals with reference individuals who have a similar environment. Ideally, the individuals differ only in exposure to the contaminant. Information on individual health is obtained through large surveys and from aggregate health statistics. Once the data are collected, epidemiological methods use multivariate econometric models to estimate the dose-response function. The dependent variable, human health, is related to the independent variables of human health determinants. Independent variables include the dose of the contaminant and lifestyle characteristics.[7]

**Exposure Assessment**

The second step in quantitative risk assessment identifies the population at risk and the likelihood of exposure to the hazard. An exposure assessment estimates the magnitude, frequency, duration, and route of exposure (U.S. Federal Register 1986). To illustrate, consider the EPA's suggested outline for a complete exposure assessment (Table 1). We focus on the EPA

Table 1. Suggested outline for an exposure assessment: U.S. Environmental Protection Agency

1. General Information for Each Chemical or Mixture
   a. Identify (e.g., molecular formula)
   b. Chemical and physical properties

2. Sources
   a. Production and distribution
   b. Uses
   c. Disposal
   d. Summary of environmental releases

3. Exposure Pathways and Environmental Fate
   a. Transport and transformation
   b. Identification of principle pathways
   c. Predicting environmental distribution

4. Measured or Estimated Concentrations
   a. Uses of measurements
   b. Estimation of environmental concentration

5. Exposed Populations
   a. Human population (size, characteristics, location, habits)
   b. Nonhuman population (size, characteristics, location, habits)

6. Integrated Exposure Analysis
   a. Calculation of exposure
   b. Human dosimetry and biological measurements
   c. Development of exposure scenarios and profiles
   d. Evaluation of uncertainty

SOURCE: U.S. Federal Register (1986, p. 34046).

given North and Yosie's (1987) observation that the agency has taken a leadership role in pioneering methods of categorizing the available evidence of adverse health effects.

First, the general information for each hazard (chemical or mixture) is collected. This information includes the molecular formula and chemical and physical properties. Second, the points where the hazard enters the environment must be examined. A complete exposure assessment should examine the production, distribution, uses, and disposal of the hazard. Third, the exposure pathways and the environmental fate should be determined. The multimedia transportation or physical transformation of the hazard is important to consider since a hazard's mobility may differ depending on the environment. For example, one must consider how a hazard behaves in air, soil, water, or biological media; its physical reaction to other compounds; the temporal and spatial persistence; and intermedia transfer. In addition, the principle exposure pathways to humans should be determined. It is particularly important to identify the media in which humans are most likely to be exposed to the hazard.

Fourth, concentrations of hazards should be measured to provide input to estimate exposure for all environmental media. The concentration estimates should be consistent with the results of dose-response estimates. Fifth, exposure assessment must determine the exposed populations (human and nonhuman) in terms of size, characteristics, location, and habits.

Sixth, the integrated exposure analysis combines the concentration estimates with the descriptions of exposed populations to determine an exposure profile. A profile provides a summary of information on the size of the exposed population; the routes of exposure; and the duration, frequency, and intensity of exposure. Often it is necessary to develop several subprofiles to separate different exposure scenarios (e.g., occupational, consumer, water, air exposure). The subprofiles are then aggregated to determine total exposure to the hazard.

Finally, an exposure assessment must present the uncertainties associated with the procedures. The lack of complete information on exposed populations precludes estimation of precise exposure distributions. Consequently, the EPA recommends that sample sizes be increased or that confidence intervals and standard goodness-of-fit tests be employed to assess the quality of the data. Regardless of the uncertainty or the technique used to correct it, the risk

assessor should present the evidence weighted by the uncertainty in the estimates. This will ensure an appreciation of the inherent difficulties in estimating exposure to a hazards.

**Risk Characterization**

The last step in the quantitative assessment summarizes hazard identification, dose-response estimation, and exposure assessment to generate a numerical estimate of risk. Risk characterization determines risk thresholds, or "safe" exposure levels, below which further analysis is not necessary. For noncarcinogens, thresholds are bounded by two levels: the lowest exposure at which effects are seen (lowest-observed-effect level, or LOEL) and the highest exposure at which no effects are seen (no-observed-effect level, or NOEL). To ensure safe exposure levels, NOEL is generally used as the threshold measure since it is a more conservative estimate. For carcinogens and most reproductive toxins, however, regulators assume no threshold. Instead, regulators use an acceptable risk level, often selected as $10^{-6}$ or to $10^{-5}$ risk of cancer or other detriment per lifetime.

Safety factors are often used to compensate for scientific uncertainties and data limitations. Two safety factors are generally used for noncarcinogens: a tenfold factor to compensate for sensitive individuals (e.g., children) when extrapolating from studies of "average, healthy" individuals, and a 100-fold factor when extrapolating from long-term animal experiments (Environs 1986). Safety factors are further extended to compensate for studies of short duration or highly experimental studies. Finally, the risk characterization should interpret the numerical estimates to provide the risk manager some information on the limitations and strengths of the evidence.

Risk assessment, then, provides a framework for quantifying risk. With n-fold safety factors, quantified risk estimates generally provide policymakers a conservative safety level to aid decision making. Even with safety factors, however, scientific uncertainties can inhibit a policymaker's confidence in the risk estimate. One problem is that extrapolating health risks to humans from animals is a delicate maneuver. The inaccuracies of extrapolating across species are compounded by differing dose levels, time horizons, and biophysical systems. Because of budget and manageability constraints, animals are artificially exposed to high doses for short durations,

whereas humans normally are exposed to low doses for long durations. Specification errors associated with extrapolating across differing dose levels and time intervals are complicated, often lending to oversimplification. Extrapolation problems are further compounded when symptoms of human health risk develop several decades after exposure. In addition, different species have different thresholds, thereby creating measurement difficulties in susceptibility, sensitivity, and metabolism activation (Schneiderman 1980). Finally, extrapolations outside the range of observations are sensitive to distribution assumptions and tail probabilities (Rosen 1981).

Another major factor of uncertainty is a hazard's mechanisms of transport and transformation in the multimedia environment. Many researchers believe that this uncertainty may equal or exceed the uncertainty in dose-response estimation (see, for example, the volume edited by Cohen 1986). Determining movement across media is difficult, but it is preferred to single-medium approaches that are partial and often counterproductive. Recent advances in multimedia modeling offer a method to systematically organize and evaluate information to provide a comprehensive view of the likelihood of adverse health effects (see, for example, McKone and Kastenberg 1986). As more researchers recognize the importance of the multimedia approach, reducing the uncertainty of transport and transformation will become a major research priority.

Combined, these scientific uncertainties can lead to a wide range of estimates regarding the actual health risk. When the range of confidence for a health risk is too large, it is of questionable use to policymakers. There is, therefore, a tendency for political judgment and informal speculation to displace "factual" evidence in making social decisions.

Two major steps in reducing the uncertainties associated with risk assessment include the use of pharmacokinetics and the use of expert opinion in probabilistic risk assessment. As noted earlier, North and Yosie (1987) suggest that more accurate procedures based on knowledge of biological mechanisms may someday replace the current procedures of scaling doses from lab animals to humans and extrapolating high-dose to low-dose exposures. Also, probability methods exist to assess the judgment of expert opinion on uncertainties regarding health risks (see, for example, Mosleh and Apostolakis 1986 or Mosleh et al. 1988). Given sparse empirical data, the

models allow the risk manager to assess the credibility of the expert opinion. North and Yosie believe that the probability methods provide an attractive alternative to the currently used conservative safety factors or plausible upper-bound estimates for risk.

### Risk Perception

Hazardous material can conjure up images of a fortified storage facility containing sanitized, airtight receptacles, or an abandoned dump site teeming with rusty, leaking barrels of toxic waste. The images induce vividly different perceptions of risk to public health. Yet both images of perceived risk can exist simultaneously in a community, causing considerable disagreement as to whether the risk should be regulated or not. To ensure public safety, a policymaker must act as an arbiter reconciling the divergent perceptions and proposing regulatory solutions. Therefore, a policymaker's understanding of risk perception is fundamental for effective regulation of risk.

Risk perception research examines human responses to risks associated with an environmental hazard. Formally, perceived risk is a function of an individual's subjective probability estimates based on various psychological factors (e.g, dread, familiarity, control), and is represented as a set of bets about uncertain outcomes for a specific choice (Cole and Withey 1982). Originating with experiments attempting to verify the von Neumann-Morgenstern (1944) axioms of expected utility theory, risk perception has become the subject of considerable research in the social sciences and, to a lesser extent, the natural sciences (Mitchell 1984). For a historical survey of risk perception research, see Otway and Thomas (1982) and Slovic et al. (1982a, 1982b, 1985).

Risk perception research has four basic tasks. First, risk perception research examines why some risks are more acceptable than others. Individuals who accept the risk of smoking or driving without seat belts may not accept the risk associated with nearby treatment, storage, and disposal of hazardous material. In a seminal article on risk acceptance, Starr (1969) argued that voluntary risk is more acceptable than involuntary risk. Therefore, any technology that inhibits public "voluntariness" will be less acceptable to society. Recent research has refined Starr's argument by contending that voluntariness, hence acceptance, is actually determined by such

factors as perceived controllability, familiarity, dread, anxiety, regret, time horizons, and spatial dimensions (Rowe 1977; Covello 1984).

Risk perception examines lay persons' perceptions of risky technologies and the determinants of their relative acceptability. Using risk-benefit analysis (Crouch and Wilson 1982), researchers attempt to measure the welfare benefits of risky technologies. The majority of risk acceptance research has been in the area of public perception of low-probability/high-consequence technology such as nuclear power (see, for example, Waller and Covello 1984; Ricci et al. 1984; Schwing and Albers 1980).

Laypersons often will not accept risk if the hazard is perceived as uncontrollable, regardless of expert opinion. For example, Table 2 illustrates the perception gap in risk associated with nuclear power. In the early 1980s, laypersons perceived nuclear power as the number one risk to public safety, while experts ranked it twentieth--below household accidents. Yet regardless of expert opinion, during the late 1970s and 1980s Swedish citizens perceived nuclear power risk as so unacceptable that policymakers finally agreed to phase out the entire industry over the next quarter century.

The second task examines why laypersons' perception of risk differs from the experts' "objective" risk estimates. As noted by Johnson (1989), risk perception is a complex cognitive phenomenon in which individuals filter, transform, and interpret information in light of preconceptions, attitudes, and information-processing abilities. Research indicates a poor calibration between experts' opinions and laypersons' perceptions (see Table 2). Spangler (1982) argues that this perception "gap" can lead to rejection of potentially beneficial technologies (e.g., commercial nuclear power). On the other hand, the "gap" can also lead to the acceptance of risk technologies. Risk perception research attempts to define why and how laypersons overestimate low risks and underestimate high risks before making a choice (see, for example, Fischoff et al., 1981 and Covello 1984). Figure 1 illustrates the perception gap. The 45-degree line represents the case where perceived risk equals actual risk. The line intersecting the 45-degree line shows how individuals seem actually to perceive risk. This gap does not necessarily imply that people

Table 2. Expert and public rankings of risks of dying from various activities and technologies

| | Public | Experts |
|---|---|---|
| Nuclear Power | 1 | 20 |
| Motor Vehicles | 2 | 1 |
| Handgun | 3 | 4 |
| Smoking | 4 | 2 |
| Motorcycles | 5 | 6 |
| Alcoholic Beverages | 6 | 3 |
| General Aviation | 7 | 12 |
| Police Work | 8 | 17 |
| Pesticides | 9 | 8 |
| Surgery | 10 | 5 |
| Fire Fighting | 11 | 18 |
| Large Construction | 12 | 13 |
| Hunting | 13 | 23 |
| Spray Cans | 14 | 26 |
| Mountain Climbing | 15 | 29 |
| Bicycles | 16 | 15 |
| Commercial Aviation | 17 | 16 |
| Electric Power | 18 | 9 |
| Swimming | 19 | 10 |
| Contraceptives | 20 | 11 |
| Skiing | 21 | 30 |
| X-rays | 22 | 7 |
| Football | 23 | 27 |
| Railroads | 24 | 19 |
| Food Preservatives | 25 | 14 |
| Food Coloring | 26 | 21 |
| Power Mowers | 27 | 28 |
| Prescription Antibiotics | 28 | 24 |
| Home Appliances | 29 | 22 |
| Vaccinations | 30 | 25 |

SOURCE: Adapted from Allman (1985, p. 41).

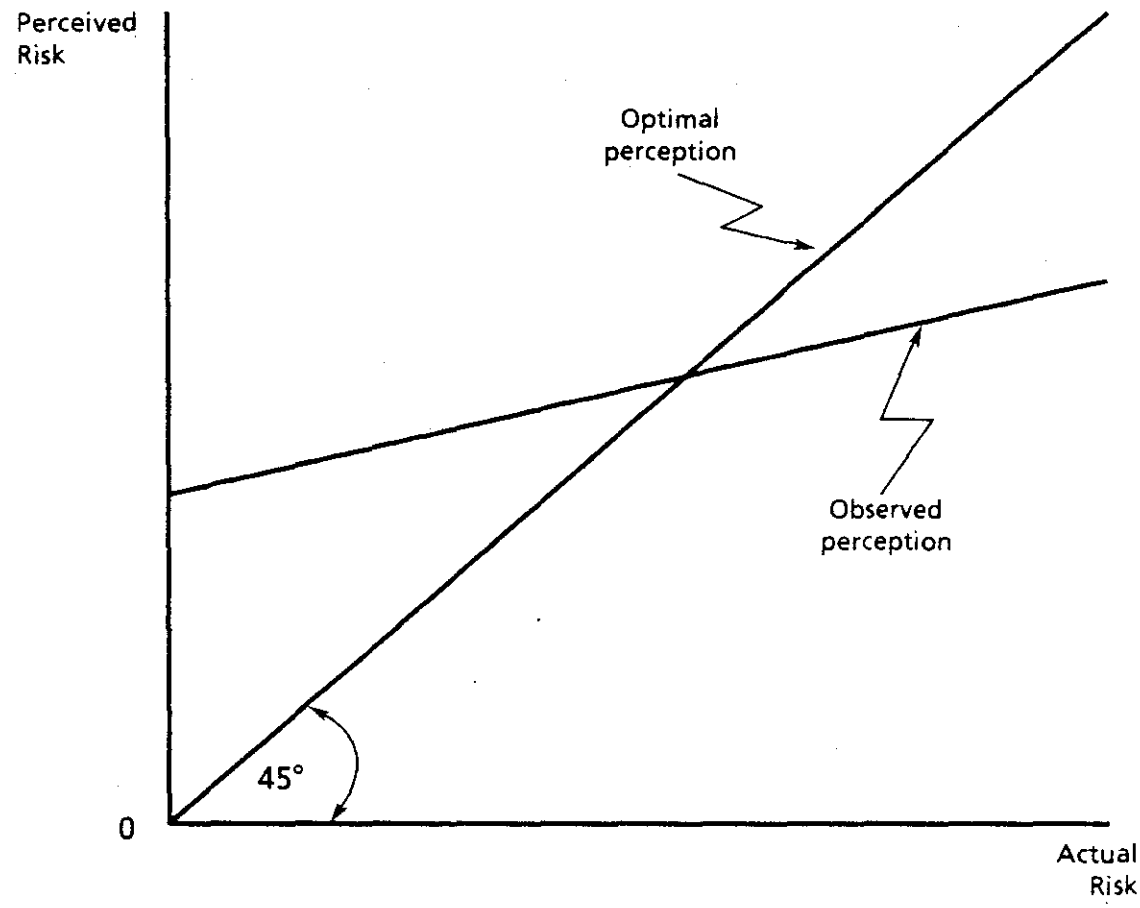


Figure 1. Perceived risk vs. actual risk.

are behaving irrationally. According to Viscusi (1989), it indicates that individuals use prior information, often incorrect, when forming probabilistic beliefs.

The perception gap raises a potential dilemma for regulation. Suppose experts argue that the risks from a certain product are unacceptable, while many individuals perceive the opposite. Does the policymaker ban the product or allow individuals to use their own discretion? The policymaker's dilemma is to balance the trade-off between preserving individual freedom of choice and maintaining public safety. The policymaker may indeed be tempted to step in and regulate the risk in the best interests of society. Such paternalistic action, however, conflicts with our society's commitment to consumer sovereignty; that is, that the individual is best able to judge what is or is not in his or her own best self-interest.

Often one group demands more regulation to ensure public safety, while another demands less regulation to ensure civil liberties. The regulator can act as mediator between the two groups or can develop information mechanisms to reconcile differing perceptions of risk. Regardless, it is a fine line between ensuring public safety and protecting individual freedom.

The "perception gap" raises the issue of ex ante/ex post choice in welfare theory under uncertainty. Standard welfare theory aggregates individual preferences to obtain a social welfare function. Under uncertainty, the social welfare function can be derived in two ways: ex ante and ex post states of the world. An ex ante choice implies deriving a social welfare function by maximizing individual expected utility (using individual perception of probability) and summing the individual benefits. The ex post choice derives a social welfare function by summing individual preferences and maximizing aggregate expected welfare (using policymakers' perception of probability). Hammond (1981) demonstrates that the ex ante and the ex post will be equivalent if (1) all individuals have the same probability, equal to the policymaker, and (2) the social welfare function is a weighted sum of individual utilities.

A social welfare function should respect individual preferences. However, Sandmo (1983) points out that although preferences are usually assumed synonymous with "tastes," the policymaker might not respect individual (mis)perception of risk. The question becomes whose probabilities, the public's or the experts', should be used for welfare evaluation. The

policymaker must decide whether to use individual perceptions (ex ante) or his own perception (ex post) derived from expert opinion. The question amounts to whether the regulator's information is of better quality then the layperson's information, thereby leading to better decisions. The ethical decision of ex ante/ex post choice raises questions regarding the degree of individual freedom in (the misperception of) risky situations and the right of the regulator to impose personal values on private citizens. These ethical questions will continue to be at the forefront in the debate to regulate public health (see Döderlien 1983).

The third task examines information issues in risk perception. For individuals to make educated choices about risk, they need information about the environmental hazard. Obtaining and processing information are costly activities, however. Given that an individual's ability to process information is limited, more information may confuse rather than improve decision making. Studies confirm that a limit exists beyond which additional information does not improve decisions. Magat et al. (1988) demonstrate that hazard warnings can be effective, but the cognitive limitations of individuals are important. Increasing the amount of risk information creates an information processing trade-off. Individuals recall less information due to problems of information overload. Magat et al. (1988) conclude that information such as hazard warnings must be carefully designed to account for an individual's information-processing abilities.

Johnson (1989) has examined the practical and ethical dilemmas confronting regulators charged with protecting public health from environmental risk. Johnson treats the regulator as the provider of information on the nature of the risk. Two alternative information programs are considered: the traditional standard-setting approach of disclosure and the economic-based approach of informed consent. The disclosure approach imposes expert opinion about what policies are most appropriate to protect public health. The approach attempts to minimize the incidence of morbidity and mortality in the population by constructing a uniform threshold of acceptable risk. In contrast, the policy of informed consent reflects the principle of respect for individual autonomy. Given that preferences for risk vary substantially in a population, informed consent limits government's role to disseminating information in a form most useful to the individual. The individual then selects a level of protection based on risk preferences.

As shown by Tversky and Kahneman (1981), however, individual responses to risk are subject to the framing and presentation of information. Like Viscusi (1989), Johnson stresses that regulators should pay as much attention to how they provide the information as they do to what information they provide. Recent work on the information content of pamphlets describing the risks associated with radon have demonstrated that information disclosure based on standard-setting assumptions yields perceptions and behavior significantly different than those based on informed-consent assumptions (see Smith and Johnson 1988 and Sims and Baumann 1983).[8]

The fourth task considers the subjective judgments of experts in risk assessment. Because of the uncertainties involved in risk assessment, natural scientists and technicians must make value judgments. Whittemore (1983) discusses how the supposedly "factual" step of risk assessment is unavoidably influenced by values. She concludes that the U.S. Office of Science and Technology Policy, which describes toxic substance regulations, sets an unrealistic and unattainable goal of separating science (risk assessment) and policy (risk management). She suggests four steps in reconciling the difficulty between facts and values. First, recognize that a clean separation of risk analysis into matters of fact and value is illusory. Value judgments exist for both policymakers and scientists. Second, improve our techniques of quantifying and dealing with uncertainty. New tools are required to quantify and order sources of uncertainty in risk analysis and to order them in terms of importance. Third, the apparent enigmas and inconsistencies of human judgment require more attention to understand the logic behind decision making in the scientific community. The recognition and acknowledgment that facts and values are often inseparable gives human perspective to the technical nature of risk assessment (see also Whyte and Burton 1980). Finally, decision-making procedures need reevaluation because too much regulatory power rests with experts and administrators not directly accountable to the public. Whittemore suggests that a "lay court" of informal representatives from all sectors set regulatory standards. This suggestion might not be feasible, however, given the high cost of coordinating a wide group of divergent regulators.

Both the lay public and experts have subjective perceptions about the risks posed by an environmental hazard. Therefore, to guarantee that the value-driven viewpoint of natural

scientists can be used as a guide to improve public policy, Lowrance (1976) has suggested the following guidelines for scientists: (1) include critical, articulate laymen in the risk assessment group; (2) place on record their source of bias and potential conflicts of interest; (3) disclose in detail the specific bases upon which assessments are made; (4) reveal the degree of certainty with which the various parts of the decision are known; and (5) express findings in clear, jargon-free terms in nontechnical presentations. These guidelines apply for any scientist, natural or social. By recognizing the importance of risk perception in environmental risk analysis, Lowrance's guidelines can lead to more acceptable policy decisions.

## Risk Valuation

A policymaker confronts difficult choices regarding the regulation of environmental hazards. Constrained budgets and increased fiscal accountability prevent the reduction of all risk to all individuals. Deciding which risks to reduce and to what degree requires evaluation of each new or revised regulation. In order to ensure comparability of value across all sectors of the economy, the policymaker must be able to rank regulatory alternatives in terms of a common unit. Arguably, the most common denominator is money, or monetary equivalence. Risk valuation systematically evaluates each regulation by estimating the monetary value (benefits and costs) of a reduction in risk. Essentially, risk valuation is benefit-cost analysis under uncertainty (see Crouch and Wilson 1982).

Risk valuation has been required for any new major regulation in the United States since 1981 (see U.S. Environmental Protection Agency 1987). Prior to 1981, regulations frequently required risk to be reduced to the lowest possible level, often without regard to economic efficiency. To curb this practice, President Reagan signed Executive Order 12291, which mandates that only regulations where the economic benefits exceed costs will be enacted.

Valuing the costs and benefits of reduced risk is formidable and often controversial. While the cost of regulating hazardous substances, which includes such items as handling, storage, containment, transport, and chemical transformation, can be measured with some degree of accuracy, the benefits are elusive and difficult to quantify. Problems arise because goods associated with reduced risk (e.g., life and limb) are not bought and sold on the auction block.

These goods rarely if ever enter a private market, and they remain unpriced by collective agency action.

For example, valuing risk implicitly requires valuing human morbidity or mortality. The loaded term "value of life" raises more than a few eyebrows. Ethical and moral beliefs often force a person to balk at the idea. The economic value of life is quantified explicitly and implicitly everyday, however. Whenever a policy is accepted or rejected, whenever the status quo remains, life is implicitly valued. For example, a North Carolina hospital recently refused to spend $150 per health care worker for an inoculation against hepatitis B. Given the workers' odds of catching the disease, the hospital has implicitly placed a relatively low value on life. Nothing is lost by explicitly examining the economic value of life, or more to the point, the value of reduced statistical risk.

Given the fundamental economic problem of determining an optimal level of risk and safety in society, risk valuation has developed two general approaches to measuring the economic benefits of reduced risk: the human capital and willingness-to-pay approaches. Each approach is designed to determine an individual's preference for risk reduction.

**Human Capital Approach**

The human capital approach values risk reductions by examining an individual's lifetime earnings and activities. The value of a risk reduction is the gain in future earning and consumption. The value of saving a life is often calculated as what the individual contributes to society through the net present value of future earnings and consumption. The human capital approach has an advantage in that it is actuarial; that is, it uses full age-specific accounting to evaluate risk reductions. A major drawback of the approach is that it assigns lower values to the lives of women and minorities, and zero value to retired individuals. The approach also lacks justification based on traditional economic welfare theory.[9] For this reason, economists have rejected the human capital method for the willingness-to-pay approach (see Linnerooth 1979 or Zeckhauser and Shepard 1984).

**Willingness-to-Pay Approach**

The willingness-to-pay approach has been advocated by economists because it has a firm grounding in traditional welfare theory. As pointed out by Mishan (1971, p. 705), "...there is more to be said for rough estimates of a precise concept than precise estimates of economically irrelevant concepts." The willingness-to-pay (WTP) approach estimates the value of a statistical life by estimating the change in welfare given a change in the risk of injury or death. The welfare change equals the maximum an individual would be willing to pay to reduce risk or the minimum compensation an individual would be willing to accept for an increase in risk. The welfare measure is then summed across all individuals to provide an estimated value of a statistical life. To illustrate, consider the following example. Suppose 100,000 individuals were each willing to pay $20 to reduce risk from three deaths per 100,000 lives to one death per 100,000 lives. The total willingness to pay, then, is two million dollars and the value per statistical life is one million dollars given two lives are saved.

There are three general approaches to the WTP method: the hedonic wage-risk model, the contingent valuation method, and the averting behavior (or consumer market) method.

Hedonic Wage-Risk Model. Hedonic wage-risk models are based on hedonic price theory (Rosen 1974). This theory views the commodity (wage rate) as consisting of various attributes (e.g., job safety, occupation, location, environment of work). A worker will adjust the job such that the marginal willingness to pay for each attribute equals the marginal contribution of each attribute to the wage rate. The value of job safety (risk) is the marginal willingness to pay for the attribute "job safety." There should exist a trade-off between the wage rate and job safety.

The workers' view on risk is compared to the employer's view on risk. Theoretically, the employer will increase job safety until its marginal cost equals the marginal reduction in worker wage rates. The market equilibrium between workers and employers determines the risk premium. The risk premium is the extra monetary compensation for risky jobs. The hedonic wage-rate model then calculates the market clearing wage-risk function. At that point, the worker's marginal valuation for a marginal change in risk is determined, other job attributes held constant.

Violette and Chestnut's (1983) review of the early (1974-1983) empirical results of the hedonic wage-risk model indicates that value-of-statistical-life estimates fall in two ranges: $450,000-$720,000 and $4,000,000-$8,500,000 (in 1986 dollars). Fisher et al.'s (1989) review of the new wage-risk studies (1982-1988) indicates the value of a statistical life between $900,000 and $6,800,000. The results reinforce the high range estimates of the early studies.

The degree to which these empirical estimates will be used in environmental risk regulation depends on the acceptance of the hedonic wage-risk model. The hedonic wage-rate model may not be accepted by policymakers for several reasons. First, the underlying assumptions of the hedonic model may not be realistic. In particular, the assumptions that workers have full knowledge of the risks associated with the job and can change jobs without cost may not be acceptable simplifications of reality. Second, risks associated with jobs may have little correlation with risks associated with environmental hazards such as groundwater contamination. Third, the sample population in the hedonic models represents only a segment of the total population. This segment is the employed labor force. Therefore, many elements of the total population are under-represented (e.g., children, elders).

Contingent Valuation Method. The contingent valuation method estimates the value of life and limb by constructing an auction in which safety or risk can be bought or sold. By establishing a hypothetical market containing features of naturally occurring markets, the method attempts to reveal an individual's implicit price for a risk reduction. Through careful construction of understandable demand-revealing institutions, values are determined through a survey or interview. The challenge is to make the hypothetical market realistic and relevant to the individual. A well-structured contingent valuation experiment allows individuals to solve their own income-risk trade-off. It requires the trade-off to be defined so that the individuals will interpret the problem identically and as intended by the researcher. The method can elicit both the willingness to pay for reduced risk and the willingness to accept compensation for increased risk.

The contingent valuation method has been advocated as a viable tool for benefit estimation. Its advantages include flexibility, inexpensive data collection, and the ability to create markets

where none exist (see Brookshire and Crocker 1981 or Durden and Shogren 1988). The method allows the researcher to manipulate the market structure and institutions for the policy question at hand. Quantity and quality dimensions such as temporal context, spatial dimensions, and property right entitlements can be defined to reflect specific policy requirements. The method is not restricted to policy questions involving on-the-job risks as is the hedonic wage-risk model. A broader spectrum can be examined (e.g., risk from groundwater contamination).

Although a carefully designed contingent valuation experiment can provide information on individual trade-offs between safety and income, the contingent valuation method is also subject to a number of critical biases. One complaint is the hypothetical nature of contingent valuation experiments. The method rests on what individuals say rather than what they do. Critics claim that "hypothetical questions generate hypothetical answers." Other problems include the potential to misrepresent true preferences for the good, an individual's sensitivity to question framing (Tversky and Kahnemann 1981), inability to value low-probability events, unfamiliarity with risk commodity, and responses based on attitude, not behavior (see Smith et al. 1985).

Fisher et al. (1989) review two recent contingent valuation experiments in an attempt to determine the value of changes in the risk of death. Given advances in methodology, the recent work is arguably more reliable. The judgmental best estimate of the value of a statistical life was approximately $2,800,000-$3,000,000 for both studies (in 1986 dollars). The range of values is consistent with the high-range estimates of the hedonic wage-risk model, thereby dampening the complaints of its critics.

Averting Behavior Method. The averting behavior (or consumer market) method estimates willingness to pay for risk reductions based on an individual's actual revealed preferences for durable and nondurable self-protection mechanisms. Examples include smoke detectors, seat belts, medicine, and behavioral reactions to smoking. Risk is assumed endogenous, such that the method directly accounts for behavioral responses to changes in risk. The revealed preference for self-protection allows estimation of the economic benefits of reduced risk. The current estimates of the value of a statistical life range from $460,000 to $610,000 (in 1986 dollars) (Fisher et al. 1989). The averting behavior method has major limitations in providing credible value-of-

statistical-life estimates. As of this date, the research is limited to single studies of a particular consumer self-protection market. Repetition by different researchers with different data sets will be required before the method is accepted as a viable measure of the value of a statistical life.

In deciding which risk valuation method to use, a policymaker must recognize the crucial trade-off between cost of implementation and the type of benefits estimated. The human capital approach is relatively inexpensive, but the benefits estimated are not based on traditional economic theory. The contingent valuation and averting behavior methods are more expensive since they require primary data collection. However, both methods estimate willingness to pay for risk reduction based on preference revelation theory. As in most circumstances, acquiring detailed and precise information is a costly activity, and policymakers must decide how detailed benefit estimation need be.

Based on the existing evidence from the hedonic wage-risk models, the contingent valuation method, and the averting behavior experiments, Fisher et al. (1989) argue that the most defensible range for the value of a statistical life is from $1,600,000 to $8,500,000 (in 1986 dollars). The lower bound of $1,600,000 is supported by the most recent wage-risk models and contingent valuation experiments. Evidence exists that the early finding of a lower value of life was downwardly biased due to the assumptions underlying each of the analyses. Fisher et al. are less confident in the upper bound as the maximum value of risk. The appropriateness of the upper bound may be conditional on the specific risk in question. The upper value of reduced risk may increase depending on whether the risk is voluntary or involuntary, whether it afflicts a family member rather than oneself, or whether death occurs only after a long illness.

Given the expanding research in risk valuation, more detailed and sophisticated experiments will arise. Therefore, Fisher et al. suggest that the $1,600,000-$8,500,000 range can best be seen as an interim range. The interim range, however, can still provide useful information in the regulation of risk.

**Risk Management**

Environmental hazards such as the improper disposal of hazardous material present a long-term risk to environmental quality and human health. A regulator must act as a risk manager and decide what to do once risk has been determined to exist by integrating risk assessment with psychological, economic, and political factors (Ruckelshaus 1984). Risk management policies are complicated by numerous factors: scientific complexity and uncertainty, political and economic pressure from special interest groups, financial abilities to clean up disposal sites, jurisdictional disputes, unresolved liability, and variations in local, state, and federal policy goals. These complications have led to the enactment of a number of federal laws related to hazardous substances (see Table 3).

Although actual implementation may vary, successful strategies for dealing with problems resulting from past disposal practices and for regulating future risks should consider the following: frameworks for regulation, equity versus efficiency criteria, determination of financial liability, and the promotion of public trust.

First, in order to guide public health decisions, the risk manager must utilize a regulatory framework or an incentive system. Lave (1984) has outlined seven frameworks for regulating safety and health risks:[10]

1. Technology-based standards--Standards are a centralized process of setting permissible levels of an environmental contaminant. The incentive is that emitters who violate the level are liable to pay a fine or tax. Alternative types of standards include uniform limits on total emissions per day or year, emission per ton of input used in a production process, and type of equipment used in production, requiring that it be the best available control technology. The advantage of standards is that costs and benefits do not need to be estimated, only technology-based engineering decisions that construct a uniform threshold of acceptable risk. No attention is generally paid to cost-effectiveness.

2. Market regulation or charges--Charges are a type of decentralized regulation that strives for economic efficiency; that is, minimizing costs to society. Charges include such incentive-based mechanisms as taxes, tradeable emission permits, and subsidies (see, for example, Schelling

Table 3. Federal regulation of hazardous substances

| Legislation | Enactment Date | Legislation | Enactment Date |
|---|---|---|---|
| Food, Drug, and Cosmetics Act | 1906 | Safe Drinking Water Act | 1974 |
| Federal Insecticide, Fungicide, and Rodenticide Act | 1948 | Resource Conservation and Recovery Act | 1976 |
| Atomic Energy Act | 1954 | Toxic Substances Control Act | 1976 |
| Federal Hazardous Substances Act | 1960 | Federal Mine Safety and Health Act | 1977 |
| Poultry Products Inspections Act | 1968 | Comprehensive Environmental Response, Compensation, and Liability Act | 1981 |
| Occupational Safety and Health Act | 1970 | Hazardous and Solid Waste Amendments | 1984 |
| Poison Prevention Packaging Act | 1970 | Asbestos School Hazard Abatement Act | 1984 |
| Clean Air Act | 1970 | Superfund Reauthorization Act | 1986 |
| Hazardous Materials Transportation Act | 1972 | Safe Drinking Water Act Amendments | 1986 |
| Clean Water Act | 1972 | Water Quality Act | 1987 |
| Marine Protection, Research and Sanctuaries Act | 1972 | Lead Contamination Control Act | 1988 |
| Consumer Product Safety Act | 1972 | Medical Waste Trucking Act | 1988 |
| Lead-Based Paint Poison: Prevention Act | 1973 | Indoor Radon Abatement Act | 1988 |

1983 or Hahn 1989). For example, a charge system can limit emissions of a contaminant by auctioning either feedstock, generation, or disposal permits (see Hahn 1988). The permits would state the permissible emission level. It is then up to the emitters to find the least costly technology to satisfy the permit or to purchase permits from low-waste producers.

Market regulation also includes the risk communication programs discussed in the section on risk perception. Viscusi's (1989) work on hazard warnings as a regulatory alternative has generated convincing evidence of the efficacy of information to correct perceived market failures. The major benefit of information programs is that individuals are allowed to make informed choices based on preferences toward risk rather than being forced to accept uniform government bans or regulation. As Lave (1984) notes, the use of market regulation puts faith in consumer judgment. The risk manager must be sure that the information consumers have will result in more accurate private decisions regarding risk.

3. Cost effectiveness--Risk management takes into consideration the public's preferences and perceptions of risk. Lave (1984) argues that for equitable or efficient regulation, the public must be represented on questions of risk. This can be accomplished through open meetings, interviews, and votes. Health and safety objectives are set by risk management and the public in an open session. The best method for achieving the goals is identified. Cost effectiveness attempts to find the least costly method to achieve the goals. A major advantage of cost effectiveness is that it does not require an explicit estimate on the value of life. The method maximizes lives saved given a fixed budget where assumptions of values are built directly into the model.

4. Cost-benefit analysis--Cost-benefit analysis can be used as a tool to measure the economic efficiency of a regulation. Cost-benefit analysis attempts to measure the costs associated with the risk regulation and the subsequent welfare benefits from a risk reduction. The costs of differing policy alternatives are then compared with their benefits to determine if and to what extent the risk will be reduced. The value of life must be explicitly considered in a cost-benefit analysis for reduced risk.

Along with the value-of-life estimates there are many controversial aspects to cost-benefit analysis, including distribution of wealth and the rate of discount. The goal of cost-benefit is to maximize economic efficiency and make the resulting "pie" as large as possible. Little, if any,concern is given to equity and distributional issues of how the "pie" will be divided. Lave (1984) notes, however, that none of the frameworks really deals well with this issue.

The choice of the appropriate discount rate has been debated by economists for decades. The higher the discount rate, the less future benefits and costs weigh in the analysis. Currently, Executive Order 12291 requires a 10 percent discount rate unless otherwise justified. To justify a lower rate implies acceptance of a policy that transfers resources from today to future generations. The opportunity cost of such a transfer requires trading off solutions to today's problems for future consumption by descendants. On the other hand, a higher rate may imply that the future costs of hazardous material are irrelevant to today's decisions. Discounting will continue to be a controversial topic as political pressure for fiscal accountability continues to increase.

5. Zero risk--Regulation such as the Delaney Clause of the Food, Drug, and Cosmetic Act, which prohibits the presence of any known carcinogen as a food additive in processed food, is a zero risk regulation. As science becomes increasingly adept at measuring small amounts of trace chemicals that are potential carcinogens, the zero risk approach is becoming increasingly restrictive (see Crouch and Wilson 1981). The costs associated with approaching zero risk often increase at such a rapid rate that it becomes economically inefficient to meet such a stringent requirement. An alternative goal is for society to define an acceptable level of risk that can be attained with efficiency by current technology.

6. Risk-benefit--This method involves a direct comparison of the trade-off between risk and dollar benefits. As discussed in the section on risk analysis, economists have devoted considerable effort to determining the implicit dollar evaluation of reduced risk. Risk-benefit analysis is less formal and quantitative than cost-benefit analysis. It is a flexible framework that broadens the scope of decision making. Lave (1984), however, believes that its flexibility is often distracting to regulators who usually require a precise framework to justify policy decisions.

7. Risk-risk--This is a direct comparison of substituting one risk for another. Lave (1984) identifies two types of risk-risk frameworks: direct and indirect. The direct framework requires estimation of the trade-off between consumer health risks and substances that offer a direct health benefit. The health benefits of drugs, exercise, and diet, for example, fit into this framework. In contrast, the indirect framework allows for a wider range of risks, including occupational risks. As Viscusi et al. (1988) note, the benefit of the risk-risk framework is that regulators can convert health outcomes into fatality risk equivalents, which might allow more meaningful comparisons than a risk-dollar trade-off.

Since all risks cannot be reduced for all individuals, a second element of risk management is justification of the regulation by either economic efficiency or equity criteria. An efficiency criterion identifies the technology or institution for achieving the desired risk reduction and determines the least costly method to achieve the reduction. An efficiency criterion recognizes that goals must be realistic and assumes that safety cannot be pursued without regard to cost. Risk-benefit, market regulation, and cost-benefit analyses are examples of regulatory frameworks that use the efficiency criteria. In contrast, the equity criterion distributes the burden of the risk based on a subjective measure to weight individual welfare. Risk can be evenly distributed among the population or can be progressively or regressively distributed based on, say, wealth. Zero risk and technology-based standards are examples of frameworks with equity as the decision criterion. The efficiency and equity criteria cannot be accomplished simultaneously, however, without costly redistribution of wealth. There is a regulatory trade-off between efficiently reducing risk and equitable risk sharing. The risk manager and the public must decide how much of each criterion to achieve (see Payne and Brough 1981).

The third task of risk management is to decide who will bear the burden of the regulation; that is, whether the emitters or the receptors must pay for the removal of or protection from the environmental hazard. To assist in determining the burden, in 1981 Congress passed the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), which established Superfund. Superfund ($8.5 billion) is funded by feedstock taxes and is used to pay cleanup costs of an abandoned disposal site if it is considered dangerous to either human health

or the environment. The act required the EPA to select a national priorities list of at least 400 sites, with the top priority of each state listed in the first 100 sites (Yandle 1988). As of 1988, the average cleanup cost of a Superfund site is between $21 million and $30 million, a considerable amount more than the projected costs of $8.1 million per site.
Superfund has the right to recover costs from the owner or the operator of the site, or from anyone responsible for transporting hazardous materials to the site, if identifiable. The potentially responsible party is liable for cleanup costs and health damages. The government does not have to show negligence and can request triple the costs of action. Strict liability and joint and several liability among many defendants have created a tendency to go for the firms with the deepest pockets. The costs of litigation have been estimated at $8 billion, 55 percent of the estimated direct cleanup costs of the National Priority List sites (Rich 1985). Rich (1985) notes that if two-thirds to three-fourths of the Superfund litigation costs could be eliminated, then 300 to 400 additional sites could be cleaned up below the project costs of current program.

Since the government has not been willing to settle until the identity of all major contributors and most minor contributors is well established, insurance and reinsurance markets have drastically retreated from the environmental liability market (Faron 1985). With insurers curtailing liability coverage, litigation costs will continue to increase, diverting valuable resources from future cleanup operations. If the goal is to minimize the social costs of a realized environmental hazard, then it will be necessary to reexamine the incentives behind Superfund, since it is doubtful that the program will ever produce sufficient benefits to justify its costs (see Yandle 1988).

A final task of risk management is to convince the public that proposed or active policies are actually in its best interests. William Ruckelshaus (1984) argues that risk management can be flexible and effective only if the public believes regulators are acting in the public interest. Ruckelshaus identifies four principles for better discussion about risk between the public and government.

First, risk assessments must be presented as distributions of estimates. Realistic estimates are needed to demonstrate a range of possibilities. There should be no magical estimates of risk

susceptible to random manipulation. Second, allow the public to see and scrutinize the underlying assumptions of risk analysis. Ruckelshaus advocates an honest, hands-on-the-table approach to risk analysis. Third, risk reduction is the main concern of the risk manager. Risk management should communicate that cost-benefit analysis is not the only value of importance. Fourth, the public and the risk manager should understand the limits of quantification. Just because a value is not included in a cost-benefit analysis does not mean the value is not important. Risk management should try to make risk analysis more human to the lay public.

Given that individuals misperceive probability and risk, both elected policymakers and appointed regulators often serve as self-appointed spokesmen for public wants. The tendency is for policymakers to attempt to protect society from the consequences of their own misperception of risk. The wide range of risk assessments and risk valuations tends to cause informal speculation and conservatism among regulators. Given the long time horizons associated with hazards and risk, policymakers can confidently defend the position that it is better to err on the side of safety. Yet given also our society's notion of consumer sovereignty, the policymaker may not be justified in substituting a personal perception of risk for those of individuals. In terms of overall costs and benefits and in terms of maintaining individual autonomy, the acquisition and dissemination of risk information may prove to be the most efficient alternative to the traditional regulatory approach of technology-based standards.

## Summary

The continuing conflict between risky technology programs and health constraints will increase the demand for risk analysis as a tool in social decision making. In addition, as science becomes increasingly adept at detecting environmental hazards, society will have an ever-increasing list available for risk analysis. This paper presented a brief overview of the process of risk analysis in which four basic steps of risk analysis were identified and examined.

The four steps of risk analysis are (1) risk assessment--technical research attempting to quantifiably measure risks, (2) risk perception--research attempting to examine how individuals perceive risks, (3) risk valuation--research attempting to measure the value associated with

reductions in risk, and (4) risk management--the process of regulating risk in the "best" interests of society.

Each step was examined in terms of its tasks, possible methods, and underlying problems. The combination and interaction of the four steps yields the complex whole of risk analysis. Byconsidering both the insights and the limitations of environmental risk analysis, a regulator can determine an acceptable and economically feasible level of risk. As individuals we must recognize the current limits of knowledge and require more information on potential risks and the regulatory alternatives. As a society we must be willing to pay for this information.

## NOTES

1. Defining risk is controversial. Risk has been defined as synonymous with hazard, a historical phenomenon (Whyte and Burton 1980), the probability of an event (Smith et al. 1985), uncertainty plus damages (Kaplan and Garrick 1981), and as a political act (Fischhoff et al. 1984).

   Kaplan and Garrick (1981) argue that risk should be defined as a set of "triplets." The three aspects are scenario description, probability of scenario occurring, and damage resulting from occurrence of scenario. The scenario description distinguishes between low-probability/high-consequence events that would be equated in the "probability times severity" definition. Historically, the field of economics has defined "risk" as the objective probability of an event occurring (Knight 1921). A subjective probability was defined as "uncertainty." The current economic literature often treats risk and uncertainty as synonymous (for example, Machina 1983). Risk has also been defined as variance, the random variable, and an event conditional on a known or unknown state of the world. However, the most standard definition of risk is the probability times the severity of suffering from an event (Whyte and Burton 1980, Lowrance 1980). That is the definition used in this paper.

2. Hazardous materials include industrial disposal of heavy metals, toxic organic compounds, hydrocarbons, agricultural nitrates, bacterial contamination from human activity, and acid deposition percolation. The other major sources of groundwater contamination include sludge lagoons or pits, disposal or injection wells, septic tanks or sewers, land spreading of agricultural chemicals or irrigation, and underground storage tanks (Burmaster 1982). Groundwater contaminants can be categorized as being of three general types: chemical, biological, and radioactive. The severity of the contamination type is measured by the toxicity and mobility of the substance. Obviously, contaminating substances that are highly toxic and highly mobile are of most concern (see Jackson 1982 or Schecter 1985).

3. As a reviewer correctly noted, risk analysis is not restricted to hazardous material and human health. Natural hazards such as floods or hurricanes also impose risks to individuals. The reader is referred to Kates (1978) and Whyte and Burton (1980) for discussions of risks associated with natural hazards.

4. As one explores the literature on risk analysis, one is immediately confronted with the lack of a universal terminology. There is risk assessment, risk quantification, risk evaluation, risk identification, risk estimation, risk determination, and so on. The terminology used in this paper corresponds to that of the National Academy of Sciences (1983) for the steps risk assessment and risk management. I have identified the steps of risk perception and valuation to stress the important contributions made by the social sciences in improving our understanding of risk.

5. The optimal level of risk, as defined by economists, exists when the marginal benefits equal the marginal costs of reduced risk. Economic optimality is generally based on an efficiency criterion, rarely on issues of equity.

6. The material in this section draws heavily from the discussion in the U.S. Federal Register (1986) on risk assessment in the U.S. Environmental Protection Agency.

7. Hoffman (1989) has identified four major problem areas in epidemiological studies: the assessment of the exposure response sequences, quantification of exposure, recognition of biases and confounding variables, and quality and validity of data.

8. Viscusi (1989) reviews the institutional content of hazard warning programs for federally proposed alcohol beverage warning legislation and food cancer warnings in the State of California. Viscusi argues the language of the hazard warnings maximizes political interests rather than advancing the primary objective of informing consumers and enabling them to make better decisions. By ignoring fundamental economic and psychological concepts of decision making under risk, the currently proposed warnings do not convey the information necessary for consumers to make sound choices regarding risks and precautions.

9. Welfare theory is based on the idea that individuals maximize their utility of a good given a constraint. The utility is the ordinal representation of preferences for risk or any other commodity.

10. Actually, Lave (1984) identifies an eighth framework--regulatory budget. However, due to the many ambiguities and uncertainties associated with the framework, it is considered by Lave to be an intellectually incoherent framework. It will not be reviewed.

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