INTEGRATED ENVIRONMENTAL DECISION-MAKING IN THE 21st CENTURY

PEER REVIEW DRAFT

May 3, 1999

REPORT FROM THE EPA SCIENCE ADVISORY BOARD'S INTEGRATED RISK PROJECT

U.S. ENVIRONMENTAL PROTECTION AGENCY
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INTEGRATED ENVIRONMENTAL DECISION-MAKING IN THE 21st CENTURY

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PART I THE FRAMEWORK

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CHAPTER 1. PROPOSED FRAMEWORK FOR INTEGRATED ENVIRONMENTAL DECISION-MAKING

1.1 Integrated Environmental Decision-Making

1.1.1 The Call for Integrated Decision-Making

Environmental decision-making has progressed over the years with the accretion of experience and knowledge, becoming ever more subtle, inclusive, and powerful. Despite criticisms leveled against them, and despite inherent limitations, previous environmental decisions and policies have spurred significant environmental and health progress. First, the decisions have addressed a multitude of complicated environmental problems. Second, they have offered broad perspectives on how we formulate and support environmental research. Decision-makers currently draw upon an eclectic mixture of tools and information to inform their decisions: ecological and human health risk assessment; benefit/cost and cost-effectiveness models; expanded risk communication and public participation; and measures for monitoring the results of the decisions themselves. Although these tools provide essential inputs for decision-making, they have typically been applied unevenly and to relatively narrow issues. In the area of human health risks, for example, assessments often have been framed around single stressors or classes of stressors in relatively specific exposure situations. The deficiencies of such highly focused assessments are increasingly apparent; they sidestep the complexities, interrelationships, and subtleties of environmental problems as they actually confront us. Thus, a number of recent studies have urged the Environmental Protection Agency (EPA) to begin to address environmental issues in a more integrated way (e.g., NAPA, 1995; Presidential/Congressional Commission on Risk Assessment and Risk Management, 1997).

Much of the fragmentation in EPA's approach to the control of environmental problems has its roots in the statutory framework that guides the work of the Agency. From its formation in 1970, the EPA has been given responsibility for implementing a number of environmental statutes that mandate targeted actions to control specific pollutants in specific media (e.g., Clean Air Act language regarding particulates in air) or

specific routes of exposure (e.g., Safe Drinking Water Act language regarding priority pollutants in drinking water). The focus on assessing and controlling chemical contaminants pollutant by pollutant in single media has resulted in an evolving collection of federal laws and regulatory requirements that is neither systematic nor comprehensive. Nonetheless, these laws have been largely successful in controlling many of the targeted pollutants and have provided a strong national underpinning for an effective environmental protection program comprised of federal, state, and local controls.

Yet, despite these successes, there is a growing consensus, both within and outside the Agency, that a more integrated approach to environmental management is needed. Prioritizing and managing risks pollutant by pollutant and medium by medium can be both inefficient at reducing the major burdens of environmental impacts on human health and ecosystems and costly in the face of today's shrinking budgets. Of still greater concern is the possibility that such a fragmented approach may cause us to overlook significant environmental problems while we busy ourselves with comparatively minor issues that contribute little to the overall protection of human health and ecosystems. Further, in some instances, current statutes and regulations prevent the Agency from considering all relevant risk, benefit/cost, or other information. A 1995 report from the National Academy of Public Administration (NAPA) pointed out that there are "no established criteria that the Agency might use to set priorities that cut across statutory lines" and called on Congress and the Agency to give serious thought to an "integrated statute that would provide multi-media decision-making authority to the Agency" (NAPA, 1995). The SAB views the issue of statutory integration as a policy discussion and outside the bounds of the present study. Even within the current statutory framework, however, there are numerous opportunities for a more holistic assessment of risks and risk management options, and more inclusive decision-making approaches.

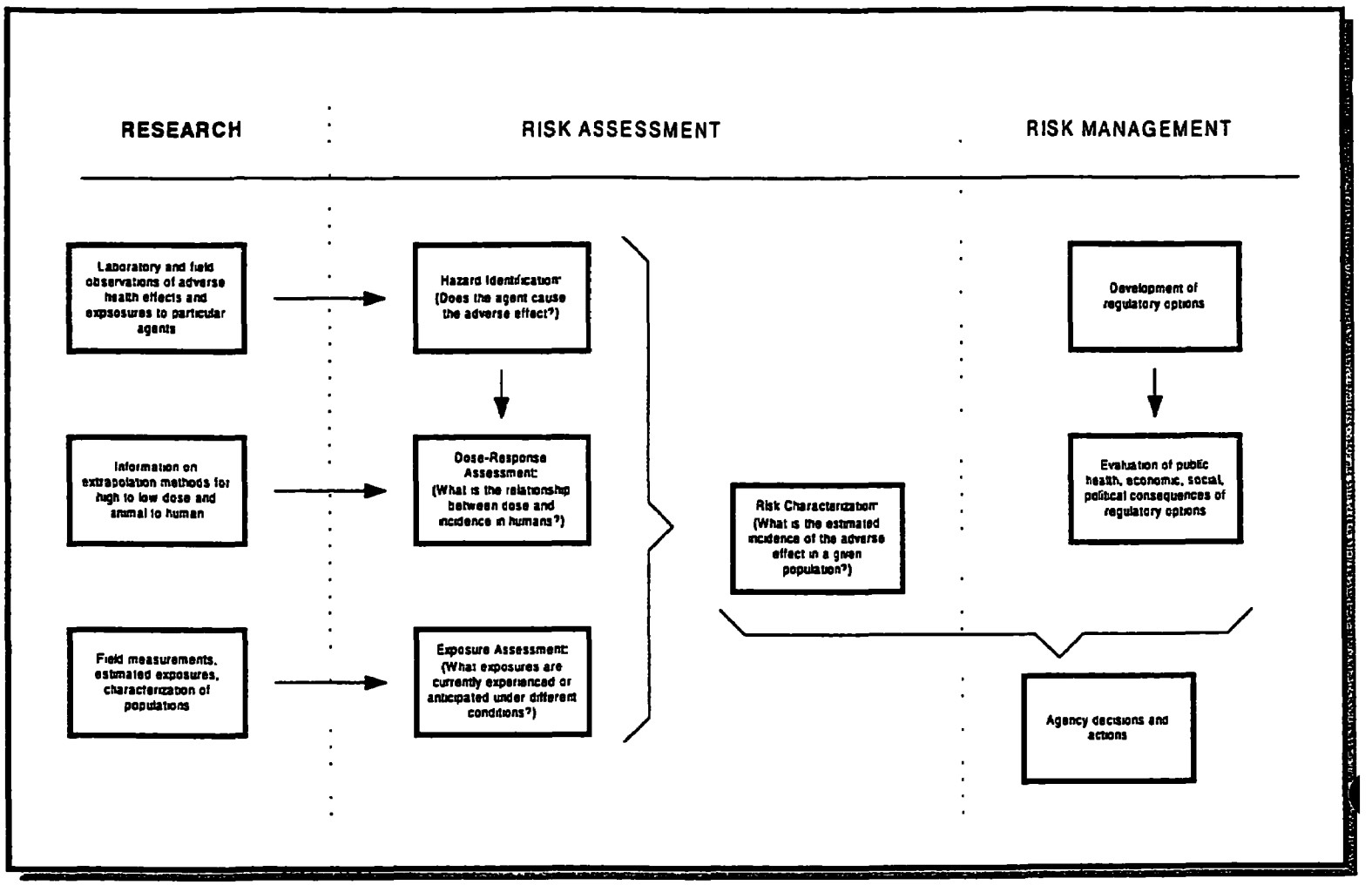
The call for integrated assessment of risks has included the need to consider multiple sources, multiple routes of exposure, and multiple human health endpoints (e.g., cancer, genetic effects, and developmental and reproductive toxicity) (see, for example, NRC, 1994), and aggregate risks posed by multiple agents or stressors (e.g., endocrine disruptors or mixtures of polycyclic aromatic hydrocarbons). Although many Agency risk assessments still focus on a single agent (e.g., lead, 1,3-butadiene), there

clearly has been an evolution in risk assessment methods towards more realistic, multiple source, multiple pathway, multiple agent assessments.

1.1.2 Signs of Progress

In the 1983 publication entitled *Risk Assessment in the Federal Government: Managing the Process* (NRC, 1983), commonly referred to as the "Red Book", an NRC panel laid out the elements of risk assessment and risk management using terminology that came to be the standard. These concepts were adopted by EPA Administrator Ruckleshaus in 1984 and have formed the basis for much of the Agency's action to this day. In summary, the NRC committee described the four steps of risk assessment as hazard identification, dose-response assessment, exposure assessment, and risk characterization, which was defined as "the estimated incidence of the adverse effect in a given population" (NRC, 1983) (Figure 1-1). In addition, the NRC committee stressed the scientific basis for risk assessment and the need for both quantitative and qualitative expressions of risk. Risk management was viewed as "a decision-making process that entails consideration of political, social, economic, and engineering information with risk-related information to develop, analyze, and compare regulatory options and to select the appropriate regulatory response..." (NRC, 1983).

The Red Book was extremely useful in articulating the risk assessment process and its relationship to risk management. The paradigm was expressed, however, in terms of single agents and single health effects in humans. Since that time, the Agency has developed risk assessment guidelines to address a number of endpoints (i.e., cancer, reproductive and developmental toxicity, and neurotoxicity), as well as exposure assessment, which is a component of the risk assessment model. In addition, the Agency has taken steps to consider more integrated exposure scenarios, e.g., multi-route exposures to mixtures of chemical agents associated with Superfund sites (U.S. EPA, 1989) or combustor emissions (EPA, 1990; 1993a), and, in response to the Food Quality Protection Act, to consider multiple pesticides and multiple routes of exposure in assessing children's risk.



Elements of risk assessment and risk management.

Fig. 1-1

From:

National Research Council, Commission on Life Sciences, Committee on the Institutional Means for Assessment of Risks to Public Health. 1983. Risk Assessment in the Federal Government: Managing the Process. National Academy Press. Washington, D.C. 191 pp.

The Agency has also made significant progress in adapting the Red Book paradigm to ecological risk assessment. In 1992, the Agency released its *Framework for Ecological Risk Assessment* (U.S. EPA, 1992) which used the term "characterization of ecological effects" to include both hazard identification and exposure assessment. The Framework also added an explicit Problem Formulation phase prior to the analysis of exposure and effects to emphasize the importance of articulating the problem and a plan for analyzing and characterizing risk prior to conducting specific risk analyses. The resulting framework contained three phases: Problem Formulation, Analysis, and Risk Characterization (Figure 1-2). An expanded discussion of ecological risk assessment principles and approaches was subsequently provided by the Agency in final *Guidelines for Ecological Risk Assessment* (U.S. EPA, 1998). The guidelines note that "although ecological risk assessments provide critical information to risk managers, they are only part of the environmental decision-making process" (U.S. EPA, 1998). In addition to assessing the relationship between a particular stressor and a particular effect, the ecorisk guidelines set the stage for considering multiple effects (including cascading effects) associated with a single stressor or source, as well as multiple causes of an observed effect or change in ecological condition. The Agency has already applied the ecological risk assessment paradigm, including the development of a conceptual model relating various stressors and effects, to five watershed cases (for discussion, see SAB, 1997).

Three additional Agency developments merit brief mention here: a) guidance and support for comparative risk analysis; b) extra-statutory approaches to environmental protection; and c) guidance on planning and scoping for cumulative risk assessment.

First, Comparative Risk Analysis (CRA) has been defined by the Agency as "both an analytical process and a set of methods used to systematically measure, compare, and rank environmental problems" (U.S. EPA, 1993b). The Agency, in its *Unfinished*

Fig. 1-2

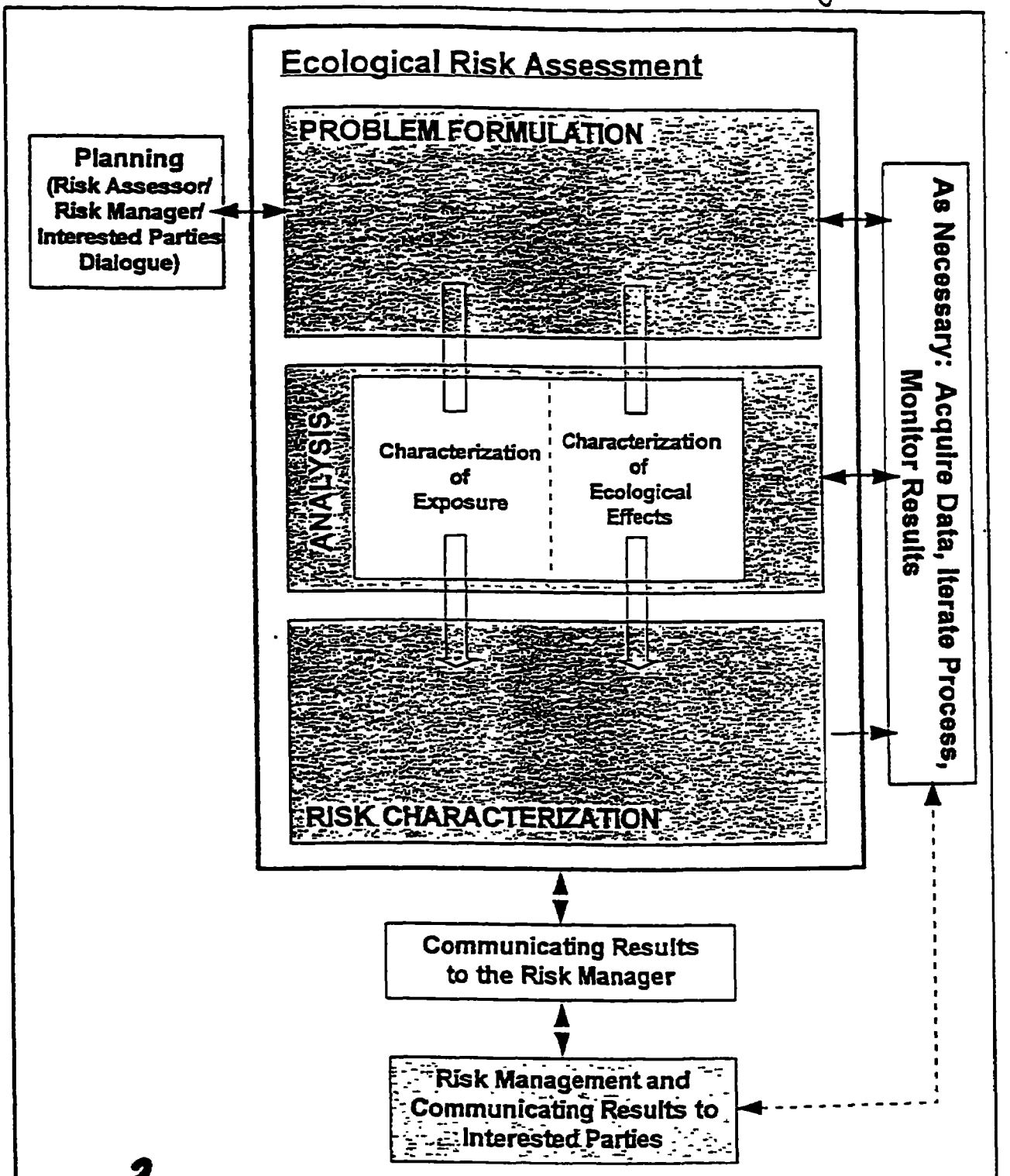


Figure 1-2. The framework for ecological risk assessment (modified from U.S. EPA, 1992a).

Business report (U.S. EPA, 1987), and the SAB, in *Reducing Risk* (SAB, 1990), engaged in comparative risk analyses. In its 1990 report, the Board concluded that it was possible, on a scientific basis, to distinguish between large risks and small risks using a set of technical criteria. In the years that followed, the Agency promoted the wide use of CRA as a process for setting priorities by integrating multiple stressors and multiple types of risks within whole regions such as cities, states or even the nation itself. Comparative risk analysis is intended principally as a policy-development and broad resource-allocation tool. In contrast to *Unfinished Business* and *Reducing Risk*, however, state and local-level comparative risk analyses have highlighted the role of the public and stakeholder groups, in addition to the scientific/technical community, in defining risk priorities. Support for broader inclusion of public values in decision-making is a theme that has been echoed by a number of recent reports (e.g., NRC, 1996; Presidential/ Congressional Commission on Risk Assessment and Risk Management, 1997).

Second, during the 1990s, the Agency has experimented with a number of approaches to re-inventing environmental protection, including greater use of community-based decision-making, voluntary cross-media emissions reductions (the 33/50 project; reference), integrated environmental agreements with states (National Environmental Performance Partnership agreements), and voluntary regulatory reform efforts with an array of stakeholders (e.g., Common Sense Initiative, Project XL).

Third, the Agency has recently issued guidance directing program offices to "consider a broader scope that integrates multiple sources, effects, pathways, stressors, and populations for cumulative risk analyses in all cases for which relevant data are available" (U.S. EPA, 1997). The cumulative risk guidance also notes that on-going Agency efforts to involve stakeholders "will provide the solid basis for engaging interested and affected parties in risk assessment and risk management issues"

The SAB has not reviewed the role and adequacy of the science being used in the planning and evaluation of these activities, including CRA. These endeavors are an indication of movement in the Agency toward more integrated and inclusive methods of environmental decision-making. These experiments, however, do not yet represent the mainstream of EPA's efforts.

1.2 Scope of the Project

It is in this atmosphere that the SAB undertook the task of revisiting its 1990 report, *Reducing Risk*, to update and extend the thinking about how science can best inform the decision-making process. The Charge to the SAB from the Agency included requests to:

a) update the risk rankings in *Reducing Risk* using explicit scientific criteria and the judgments of SAB panel members;
b) identify risk reduction opportunities and strategies;
c) identify uncertainties and data quality issues associated with the risk rankings;
d) assess costs and benefits of risk reduction options; and
e) propose a new framework for assessing the value of ecosystems.

The initial charge also included a request that the SAB explore techniques and criteria for identifying emerging risks. However, the SAB concluded that its recent report, *Beyond the Horizon: Using Foresight to Protect the Environmental Future* (SAB, 1995) provided criteria and suggestions germane to this charge question and so did not elaborate further on the future risks as part of the Integrated Risk Project.

After careful consideration of the Charge and discussion with Deputy Administrator Fred Hansen, the SAB concluded that it could best assist the Agency by investigating approaches for accomplishing these goals and considering the inter-relationship of these tasks in the broader decision-making context.

The project, known as the Integrated Risk Project, was guided by a Steering Committee and five specialized Subcommittees working over several years. The Subcommittees and their respective charges were as follows:

a) **The Steering Committee (SC)**, chaired by Dr. Genevieve Matanoski, set the overall direction for the project by defining scope and timetables. The SC met periodically over the course of the project to: (1) assess the progress and direction of the subcommittees; (2) ensure that the results could be integrated into a comprehensive decision process for identifying current and future environmental risks; and (3) review options for reducing

risks in a holistic context. The SC's efforts were designed specifically to illustrate the relationship among the various factors influencing risk management decisions (e.g., technical assessment of the risks and risk reduction options, economic considerations, equity considerations, and so forth).

b) **The Ecological Risks Subcommittee (ERS)**, chaired by Dr. Mark Harwell, was charged to assess and rank risks to ecosystems at the national scale, as well as to suggest ways in which the risk ranking methodology could be applied at smaller geographical scales (e.g., regional, state, or local). The group was also asked to explore commonalities and differences with the Human Exposure and Health Subcommittee (HEHS) methodology with the aim of integrating the two ranking schemes.

c) **The Human Exposure and Health Subcommittee (HEHS)**, co-chaired by Drs. Joan Daisey and Bernard Weiss, was charged to develop a methodology for assessing and ranking risks to human health, to consider ways in which an integrated risk ranking could be produced that includes both cancer and non-cancer risks, and to test the methodology for a limited set of environmentally mediated health issues. The Subcommittee was also asked to explore commonalities and differences with the ERS methodology.

d) **The Risk Reduction Options Subcommittee (RROS)**, co-chaired by Dr. Wayne Kachel and Ms. Marcia Williams, was charged with developing a methodology for selecting an optimal set of risk reduction options with due regard for the human health and ecological risks (defined in terms of risks associated with environmental stressors, locations, or exposure/transport media). Because of the time constraints on the project, the RROS was asked by the SC only to illustrate the methodology for one or more example problems in lieu of addressing the wider range of risks considered by the ERS and the HEHS.

e) **The Economic Analysis Subcommittee (EAS)**, chaired by Dr. Paul Portney, was charged with assessing current methods for estimating costs and benefits (either physical or monetary) associated both with the implementation of risk reduction strategies and with allowing risks to go unaddressed. The EAS was also asked to consider those aspects of the "net benefits" equation that cannot easily be monetized.

f) **The Valuation Subcommittee (VS)**, co-chaired by Drs. Alan Maki and Milton Russell, was charged to consider a new framework for assessing the value of ecosystems to humans, including ecological services and environmentally mediated health and quality of life values. The work of the VS was intended to provide a wider societal view of risk and risk reduction options than that derived from science-based risk assessments and current methods of economic analysis.

Over the course of the project, the IRP SC and subcommittees held over 25 public meetings and teleconference calls. Although most of these meetings were held in Washington, D.C., public sessions were also held in Berkeley and San Francisco, CA; Atlanta, GA; New Orleans, LA; and Baltimore, MD.

The conceptual model that emerged—the framework for Integrated Environmental Decision-making (IED)—is one that emphasizes the dialogue and interaction between risk assessors, risk reduction options analysts (e.g., engineers, economists, and environmental law/policy experts), decision-makers, and the public. This report describes the IED framework for making integrated decisions, and recommends needed improvements to the tools and techniques required to both implement and evaluate IED.

Throughout the study, the Steering Committee used stressors as a common focus to link the components of the framework because stressors are the actual physical, chemical, or biological changes that affect ecological or human systems, and thus, will be the focus of risk reduction actions. However, the IED framework provides a multi-dimensional approach to problem definition and options analysis, one that recognizes the linkages between stressors, sources, pathways of exposure, and adverse effects endpoints. It is this multi-dimensional approach that can lead to the

most effective overall reduction of risk.

The SAB, as a body of scientists, engineers, and economists, is best suited to describing the scientific and technical analyses that should inform decision-making. The project focused, therefore, on methods for assessing and comparing multiple risks to human health or the environment; an approach for designing a risk reduction program; qualitative and quantitative benefit/cost analyses; and considerations in the design of performance evaluation systems. With regard to those aspects of the IED process that rely on the application of public values (e.g., the selection of specific risk goals and tradeoffs, and decision criteria), the SAB cannot represent the public. However, the report does discuss the importance of public values to decision-making and suggests ways in which EPA, and other decision-making bodies, can utilize deliberative processes to engage stakeholders and the public in aspects of environmental decision-making.

This report represents the SAB's broadest view of environmental decision-making to date. The proposed framework is designed as a flexible guide that can be used to address environmental problems of different size, scope, and location. The effort to develop the IED has required the SAB to venture into risk management areas that do not usually arise in most of its work. However, while methods for reaching decisions about regulatory matters or policies are part of the IED framework, the SAB offers no recommendations directly impinging on specific decisions or policies.

1.3 A Proposed Framework for Integrated Environmental Decision-Making

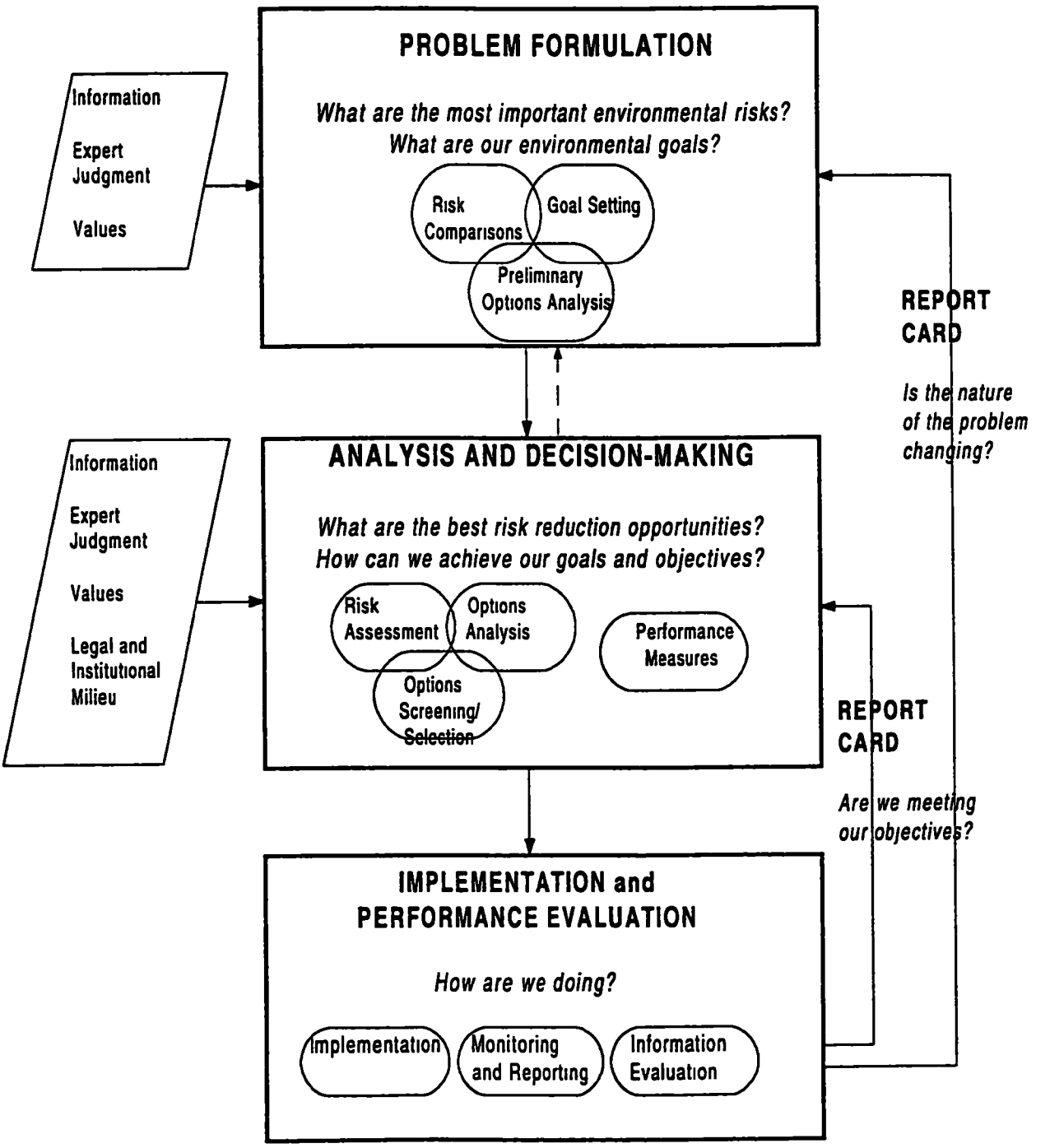
1.3.1 Overview

In order to encourage a more integrated approach to environmental protection, the SAB proposes a framework for decision-making based on the recognition that requires in-depth analysis of projected risk reduction under possible management scenarios and selection of a preferred scenario, based on criteria such as feasibility, cost-effectiveness, seriousness of the risks addressed, and equity. The final phase of IED—Implementation and Performance Evaluation—is one in which the implementation of risk reduction measures occurs and environmental results are monitored and evaluated. Performance Evaluation provides critical feedback so that management approaches can be fine-tuned and the extent and nature of remaining risks, and the means for reducing those risks if necessary, can be re-evaluated.

The complete IED framework is an integrative scheme for making decisions where many different variables, often interacting across physical, regulatory, and organizational boundaries, can be considered simultaneously rather than in isolation by the many types of participants. It allows for: a) the consideration of related clusters of risks; b) the development of multiple risk reduction options; c) the definition of markers for evaluating progress toward specific environmental goals; and d) consideration of public preferences and values throughout the process.

Although the IED process requires the involvement of a broad spectrum of participants (e.g., scientists, engineers, economists, decision-makers, and the public), the different groups have unique roles to play. In other words, the framework does not imply that "everyone must be involved in everything all the time." For example, just as scientists cannot provide the perspective of the general public, members of the general public cannot do the job of scientists. Decision-makers, after considering the various sorts of information (data, views, and judgments) generated by the problem formulation and analysis, must make the decision. The important message of the IED framework is that the various groups involved must maintain effective communication with each other throughout the process so that each can most effectively do its job within the overall context.

Figure 1-3. Integrated Environmental Decision-Making Framework



As illustrated in Figure 1-3, the IED framework is intended to answer a series of straightforward questions. What are the most important environmental risks? What are our environmental goals? What are the best risk reduction opportunities? How can we achieve our goals and objectives? How will we know whether or not we are meeting our goals? What modifications in our approach are needed to improve environmental results? Finding answers to these fundamental questions requires application of scientific and technical assessment and analysis techniques, as well as political, policy, and values-driven choices. The following sections describe the IED process, including the use of the various analytical methods.

1.3.2 Phase I: Problem Formulation

The initial phase of the IED framework is Problem Formulation, in which agreement should be reached among all participants—risk assessors, risk managers, and interested and affected parties—about what needs to done by whom and why, if not how. As shown in Figure 1-3, Phase I includes primarily three related tasks: Risk Comparisons, Goal Setting, and Preliminary Options Analysis. In this initial Phase, the discussions are at the level of planning, scoping, and screening, rather than the detailed analyses conducted in Phase II.

Figure 1-4. Risk Comparisons

The term Risk Comparison is used in this report to denote the characterization and ranking of risks posed by environmental stressors, where an environmental stressor is any physical, chemical, or biological change or agent that could affect ecological or human health systems. In the case of a single stressor, this analysis consists essentially of risk assessment approaches that have already been established for human health risks (NRC, 1983) and for ecological risks (EPA 1992, 1998). The information resulting from this component is a characterization of the nature and magnitude of the risks posed by the stressor and characterization of the systems at risk and elements of systems that may be exposed to the stressor.

In the multi-stressor case, it is important to consider a wide range of environmental risks simultaneously so that the seriousness of risks can be characterized relative to one another. This may include comparison of risks across different stressors affecting human health, ecosystems, and/or quality of life. Risk comparisons may be done by comparing quantitative estimates of risks where that is possible, or by qualitative evaluation of risks against some set of criteria. While quantitative risk comparisons are ideal, qualitative comparisons may be best suited when comparing non-commensurate risks (e.g., risks to humans compared to risks to ecosystems). Chapters 2 and 3 describe approaches to risk comparisons designed to utilize the best available scientific information and expert judgment to categorize risks (e.g., high, medium, and low) and make transparent the influence of any value judgments or tradeoffs. The two approaches can be used to rank risks within, but not across, the categories of human health or ecological risks. This is because the ranking factors, while analogous, differ in specifics for the two types of risks.

The IED framework also allows for the explicit assessment and comparison of risks to quality of life (QOL), often defined as potential non-health impacts on humans from environmental change. Examples of risks often included in this category are aesthetic, economic, and equity impacts, as well as effects on peace of mind, cultural or community identity, and recreational opportunities. Although some of these QOL risks can be assessed via analysis of ecological risks or benefit/cost, the SAB did not specifically propose a method for ranking QOL risks because the selection of QOL ranking criteria is largely a value-driven, rather than scientific, process and as such is more appropriately conducted by a broad group of public or stakeholder representatives rather than by a technical panel. However, the EPA (1993) has developed a guidebook for assessing quality of life risks that provides a starting point for such assessments. In most of the state comparative risk projects, risks to human welfare or quality of life have been considered by a separate, non-technical subcommittee that developed criteria and produced a ranked list of QOL risks.

The SAB also does not propose a formal process for merging the ecological and human health risk issues into a single ranked list because such an activity appropriately would require consideration of many non-technical issues, including political acceptability and societal values. However, the development of a single prioritized list of risks of concern, including risks to ecosystems, human health, and quality of life, can be developed with public/values deliberation and is an appropriate activity during Problem Formulation.

Risk comparison methods (see Figure 1-4) are used to identify the sets of risks to ecological, human health, and/or quality of life systems that will be the subject of detailed consideration in Phase II. Preliminary analysis of risk reduction options is important in Phase I in order to formulate the problem in terms that will be amenable to the greatest overall risk reduction. Problem Formulation requires considerable dialogue between the participants comparing risks and those involved with preliminary identification of available options. This dialogue is designed to focus the problem and

to ensure consistency between the stressors and risks being considered and the risk reduction opportunities that may be available.

Planning, scoping, and screening -- including selection of endpoints -- also requires explicit input of societal values and stakeholder participation. For example, while some of the ecological endpoints that are selected in the problem formulation phase may be chosen strictly because of the value attached to their ecological role, there are also ecological endpoints that will be chosen because of their particular significance to society. Examples of ecological endpoints in this latter category include both economically important species and endangered species. Similarly, human health endpoints would likely include both risks to the general population and those relevant to specific subsets of the population, such as children or the elderly, because of societal concerns about their protection.

There is also an important role for decision-makers in this phase; for example, decision-makers will be involved in helping to identify the important environmental problems to be considered, identifying the sets of at-risk systems to address, and identifying the specific ecological or human health endpoints to select. During Problem Formulation, decision-makers also need to identify clearly the range of potential decisions and management options, examine economic, political, or other constraints on the options to be considered, and to characterize the scope and time frame for IED implementation.

In summary, in Phase I the participants—scientists, decision-makers, and interested and affected parties—seek agreement through deliberative-analytic dialogue (in the meaning of NRC, 1996) on such issues as:

a) the goals for the exercise, including environmental goals to be achieved;
b) which environmental problems/stressors/systems will be included and which will not;
c) effects of concern;
d) the spatial, temporal, and organizational dimensions of the problem;
e) relevant data, models and analyses;
f) possible approaches to data analysis;
g) scoping of the uncertainties involved;

h) research needed to significantly reduce critical uncertainties;
i) a first-cut on the range of options available to reduce risks;
j) the endpoints upon which the health of the ecological or societal systems ultimately will be judged; and
k) the type of factors that will be considered when reaching a decision.

The intent of Phase I is to have an open, yet structured, exchange of information, concerns, opinions, and values that will help to address the kinds of issues listed above. Chapters 2, 3, and 6 describe approaches by which some of the more technical aspects of this information might be developed, and Chapter 5 describes deliberative processes that might be used to set environmental goals and incorporate values deliberation in Problem Formulation. The Agency's experience with some of its stakeholder processes provides insights on constructive interaction for the purposes of planning and scoping.

1.3.3 Phase II: Analysis and Decision-Making

Phase II is that portion of the decision-making process in which most of the traditional "work" is done. Whereas in Phase I the participants formulate the problem using screening level information gained from risk comparisons, goal setting, and preliminary options analysis, in Phase II the technical specialists employ similar approaches, but with greater specificity and data requirements, to develop the in-depth technical information that helps the risk managers to reach a final decision.

In practice, the analysts take the information and general directions gained from Problem Formulation and generate more detailed, more fully supported risk assessments and risk reduction options. In the context of the IED framework, options analysis requires consideration of risk reduction opportunities with regard to their technical feasibility, overall aggregate risk reduction to be obtained (e.g., reductions in "target" risks and collateral reduction in all affected risks), full economic consequences of various risk reduction scenarios, and so forth (see Figure 1-5). Decision-makers may also request analysis of potential options with regard to sustainability, equity, and other potential

Figure 1-5. Options Analysis, Screening, and Selection

This component of the IED framework is focused on identifying the best risk reduction opportunities and is applicable to cases of either a single stressor or for multiple stressors. In its simplest form—the form that has generally been used historically at the Agency— options analysis involves examination of multiple risk reduction options to address a highly ranked risk so as to identify the option(s) that will be most cost-effective in reducing that particular risk. The IED framework expands this analysis to emphasize that ancillary reductions of other risks should also be assessed and factored into the decision process. For example, control of fly-ash emissions to reduce mercury emissions to the environment may not only reduce mercury but may reduce emissions of particulates and possibly other pollutants as well. In an IED approach, these ancillary benefits should be considered explicitly when selecting a management option for a single stressor.

In the single stressor case, the approach described in Chapter 6 for designing a risk reduction program identifies possible actions that could be taken to reduce either the stressor or its effects on the system(s) at risk. An important aspect of this analysis is to examine a broader array of potential options than might typically have been done in the past. Criteria are developed in consultation with decision-makers to screen potential options, aggregate or disaggregate options, and, through an iterative process, converge on a set of options that analysis indicates would optimally reduce the risk. Chapters 4 and 5 provide more detailed guidance on assessing the economic and societal consequences of various options, an important aspect of options analysis.

While this simple form of integrated options analysis can yield a broader view of the benefits after an option or set of options has been selected, a more powerful feature of this integrated analysis results from its application during Problem Formulation, prior to selection of risk reduction options. In such an application, scientists should examine appropriate methods by which one can combine subsets of the ranked risks in order to investigate management options that could impact on the combined risks. The basis for such combinations of risk might include features such as common sources or pathways through the environment. The critical link between the environmental risks in the subset would then be that they are all affected by a single risk reduction option/strategy. While it may not be possible to group all risks of concern on the basis of their technical attributes, a scientific analysis of the risks may well reveal commonalities that indicate which risks will be affected by the same risk reduction option. The likely effect of this integrated view is that the option(s) selected to reduce a group of risks might differ from that which would be selected to reduce the top ranked risk, if it were to be considered in isolation.

When applied to the multi-stressor situation, the IED framework calls for an expanded analysis of risk reduction options so as to identify those options that may simultaneously reduce, directly or indirectly, risks posed by more than one stressor. The goal in this case is to maximize the reduction of the total aggregate risk from multiple stressors, rather than to maximize the reduction of risks posed by any single stressor. This approach, requiring as it may the simultaneous consideration of risks from quite different types of stressors, has not yet been fully utilized, and will not be trivial to implement. Nevertheless, the SAB believes that its development and implementation offer tremendous potential for improving environmental health overall.

decision criteria. Methods for assessing the economic and societal consequences of potential actions are described in Chapters 4 and 5. A detailed discussion of a process for designing a risk reduction program is contained in Chapter 6.

The Analysis portion of Phase II is generally more "analytic" than "deliberative" (NRC, 1996) although a continued level of interaction between the participants in the overall process (scientists, risk managers, and interested and affected parties) is important. Phase II is also more resource-intensive than Phase I since it can involve consideration of more options (e.g., various groupings of related stressors, as well as different risk reduction options) and more detailed analyses (e.g., the effects of different risk reduction options on multiple stressors or groups of stressors, as well as the inclusion of cost-effectiveness factors).

In the Decision-Making portion of Phase II, the Agency or other decision-makers utilize outputs from the analyses of risk and risk reduction options, consider widely-held public values, the views of stakeholders, and legal and institutional constraints, and, ultimately, make environmental decisions. Clearly, this is not a totally scientific process. However, the best science should inform and contribute to decision-making. This can be accomplished, for example, by making explicit a) the implications of the chosen management option(s) to the health of ecological or human systems; b) the economic costs and benefits associated with the option; and c) the societal values that are affected by the decision, including both values relating to economic efficiency and values relating to sustainability and equity.

Another important role for scientific and technical analysis is to make clear the uncertainties associated with estimates of risk, the estimates of risk reduction that may be achieved by different management options, and the economic assessments of various risk management scenarios. The IED approach does not eliminate the uncertainties associated with making decisions. However, by encouraging an open and comprehensive examination of environmental problems in an integrated fashion, the IED framework should lead to a more clear identification of the nature and extent of the uncertainties associated with the available information. In any event, environmental decision-making must proceed in the presence of uncertainties, and nothing in the proposed framework should be construed as precluding environmental decisions because uncertainties remain.

1.3.4 Phase III: Implementation and Performance Evaluation

Phase III of the IED framework consists of implementation of the chosen risk reduction options and evaluation over time of the extent to which the risk reduction measures are achieving the desired environmental outcomes. This Phase involves the articulation and execution of the specific actions that must be taken to implement the decision, and the establishment of a process to evaluate performance and results of such action. The specific activities required to implement an environmental decision will depend on the suite of management options selected for any particular problem or set of problems, and thus we do not address this aspect of the IED in any detail. The Agency has considerable experience with many of the risk reduction options described in this report (e.g., adopting best available technology and imposing permit limits) and is gaining valuable new experience with others (e.g., regulatory negotiation and National Environmental Performance Partnerships).

In contrast to implementation, however, the performance evaluation process is fundamentally rooted in science because it is science that can translate the public's overarching goals (e.g., improved health, sustainable ecosystems) into discrete, measurable components. Accordingly, science is essential in deciding what to monitor, i.e., specifying the endpoints of concern for the systems at risk and identifying the specific measures that need to be monitored in order to characterize the status and trends for those selected endpoints with respect to the environmental goals. Further, the scientific issues of spatial and temporal variability, measurement error, time lags, and so on, must be explicitly addressed in order to demonstrate environmental condition and to separate signal from noise. And finally, reference conditions and benchmarks or milestones along the way to the desired system conditions must be defined scientifically so that meaningful and measurable performance criteria for success or failure can be defined.

Figure 1-6. Performance Evaluation/Report Cards

The Government Performance and Results Act (GPRA) created a requirement that, beginning in Fiscal Year 2000, federal agencies report annually on measurable results of their various programs and activities. GPRA has resulted in a flurry of activity aimed at measuring program performance, including an effort funded by the Office of Science and Technology Policy to design a national environmental report card.

In the IED context, an Environmental Report Card is a tool for communicating to multiple audiences the performance of a risk reduction program in measurable terms related to environmental outcomes. At the broadest level, the Report Card should inform all of the IED participants about how well, in general, the ecological or societal system at risk is responding to the actions taken. At a more detailed level, the Report Card should provide scientists with sufficient monitoring data to improve the risk assessments and/or the risk reduction options previously selected. The need for monitoring data to assess performance in this framework emphasizes again the importance attached to EPA's development of monitoring programs which can measure both ecological and human health exposure and outcomes.

The report card should contain specific milestones that can be used to measure progress towards achieving the environmental goal(s) agreed upon by the IED participants. Each of the selected endpoints defined during Problem Formulation should be a part of the report card, as well as the specific measures or indicators that are monitored to characterize those endpoints. The frequency of the reporting should be decided upon with all of the participants and should be commensurate with the nature of the risk and the time frame for system response. Four types of performance measures can be used in evaluating progress: measures of administrative effort, often called process measures (e.g. number of permits issued, number of control technologies installed); measures of stressor levels; measures of exposure; and measures of environmental outcomes (i.e., measures of adverse effects or condition) that report on changes in the state of the systems at risk (e.g. hectares of wetland restored or the number of cancer cases avoided). Decreases in adverse effects or improvements in health or environmental condition are the ultimate basis for evaluating risk reduction programs, and environmental outcome measures (or early markers of the final outcome) are therefore preferable. It is often necessary, however, to supplement outcome measures with shorter term measures (e.g., early markers of the final outcome, as well as process, exposure, and stressor measures) for purposes of program accountability and course correction.

The IED framework includes the use of Environmental Report Cards to document performance and outcomes of risk reduction activities at several levels and for different audiences (see Figure 1-6). As shown in the IED framework, the performance evaluation process contains several important feedbacks associated with the Report Card. One feedback loop is to the Analysis and Decision-making Phase, reporting on how well the selected risk reduction options and strategies are achieving the environmental goals. This feedback loop allows for adaptive management and changes in implementation activities, including the possible need to identify and analyze additional options to further reduce risks. A second important role for Report Card

information is to allow re-examination, as needed, of the initial risk rankings or other aspects of Problem Formulation. As risk reduction options are put into place, for example, particular risks should be reduced, and a reordering of risk rankings may be appropriate. Further, there may be a shift in or redefinition of societal values over time, requiring different sets of environmental goals and, therefore, different environmental decisions.

In summary, the IED framework emphasizes the need to consider performance information at several points in the decision-making process and to review environmental decisions in light of new scientific understanding, shifts in societal values, changes in stakeholder preferences and available resources, and/or responses of the environment to previous decisions. The topic of performance evaluation and report cards is discussed more fully in Chapter 7.

1.4 Nature of the Framework

1.4.1 Major Characteristics

Integrated environmental decisions should exhibit the following characteristics:

a) **Transparency**. The IED framework is designed to promote transparency so that interested parties will be able to follow the process and be aware of the information that was considered in reaching decisions. Transparency is enhanced by the use of clearly articulated goals, analytical methods, and criteria; open deliberative processes; and well-documented decisions.

b) **Flexibility**. The IED framework can be applied in a flexible manner depending on the specific circumstance; i.e., where appropriate, to permit valid short-cuts, to eliminate unnecessary procedures, and so to expedite the process of decision-making and implementation. Factors such as the extent and nature of a problem, the amount and kind of information available, and the information gaps identified will influence the required degree of complexity of approach and level of detail of the analyses.

c) **Dynamic process design**. The technical analyses required to implement the framework should not be conducted in an isolated, stepwise manner. For example, during the Problem Formulation Phase, problem scoping and definition and preliminary analysis of options will affect the development of goals, and vice versa. Some iteration is also required between Problem Formulation and Analysis, since preliminary analyses will often point out missing elements in the problem definition or inconsistencies in goals.

d) **Explicit feedback, interaction, and cooperation**. The IED process requires cooperation and open, continuing communication between scientists, managers, members of the public, and others involved in the different phases of an IED project. Examples of critical feedback processes include the evaluation of performance against specified

environmental goals, which allows course corrections and may improve future decisions, and two-way communication between policy-making bodies and the public.

e) **The use of information from many sources.** The IED framework requires use of concepts and methods originating in many different scientific, technical, and scholarly fields (e.g., physical sciences, public health, environmental engineering, political science, philosophy, and economics), as appropriate for any given case. In addition to science and scientific judgments, inclusion of public values is needed to compare unlike risks, to set acceptable goals, to assess risks to quality of life, to set priorities, and to reach broadly acceptable decisions.

f) **A way of thinking about environmental problems.** Finally, integrated environmental decision-making is not just a series of methodologies, but rather is a way of thinking, in a whole and complete way, about any environmental decision-making case in order to maximize the efficient reduction of aggregate risk to populations or ecological systems. The IED framework should help to focus attention on the multiple aspects of a problem, a broad range of factors that may influence the decision, and important evaluation processes that add value after a decision is implemented. The following section provides a brief synopsis of the opportunities for integration provided by the framework.

1.4.2 Types of Integration

The IED framework requires that information and viewpoints be integrated at multiple points in the decision-making process. Six critical types of integration are summarized in Figure 1-7, and discussed briefly below.

a) Integrated Risk Assessment

Focusing on a single agent, a single medium, and a single outcome to assess risks is not a realistic representation of the way environmental exposures impact on humans and ecosystems. While such an over-simplified view was arguably necessary in the early days of risk assessment, we should now work to develop tools that can determine risks from multiple exposures and multiple outcomes in order to more accurately represent real world situations. As already noted, some early, but significant, progress has been made in this area and the SAB encourages the Agency to continue these efforts. In response to the Food Quality Protection Act of 1996, for example, the Agency has stepped up efforts to consider risks posed by groups of pesticides exhibiting common modes of action, rather than setting risk levels based on each pesticide individually. In-depth exploration of the analytical challenges inherent in integrated or cumulative risk assessment is beyond the scope of this study, but it is an area that will require extensive thought and methodologic development.

Figure 1-7. Types of Integration in the IED Framework

Integrated Risk Assessment:
developing scientific data and analytical methods for determining risks from multiple exposures and multiple outcomes in order to more accurately represent real world situations.

Risk Comparisons:
considering a wide range of environmental risks simultaneously so that the seriousness of risks can be characterized relative to one another.

Integrated Analysis of Management Options:
investigation of options to reduce subsets of ranked risks, rather than considering single risks in isolation, to achieve greater aggregate risk reduction.

Integrated Analysis of Economic Consequences:
identifying the full range of benefits and costs, both monetized and non-monetized, associated with reduction of multiple risks.

Integration of Performance Information:
using performance evaluation measures to devise course-corrections.

Integrating Multiple Disciplines and Points of View
understanding and utilizing information from all concerned parties in the IED process.

b) Risk Comparisons

The second type of integration important for integrated decision-making involves the consideration of a wide range of environmental risks simultaneously so that the seriousness of risks can be characterized relative to one another. Chapters 2 and 3 describe approaches to technical risk comparisons based on the best scientific information and expert judgment. The methods use stressors as a common point of departure to facilitate an understanding of the interconnections between risks to humans and risks to natural systems. Environmental concerns may also be expressed in terms of sources, pathways, or endpoints; these entities are interrelated so that no matter which entity is used as a basis of comparison, scientific information on stressors, sources, pathways, and endpoints will be required.

An important aspect of the IED framework, however, is the explicit notion that technical risk comparisons are not in themselves sufficient to inform an environmental decision or to set environmental priorities *per se*. Other factors that should be considered in setting risk priorities include the availability of management options, opportunities for overall aggregate risk reduction, economic impacts, and public concerns.

c) Integrated Analysis of Management Options

The third type of integration in the IED framework, discussed in Section 1.3.3, occurs during the analysis and selection of risk reduction options. In this process, risk rankings are analyzed with regard to opportunities for risk reduction (i.e., options available to address the risks) in order to determine the best approach to reduce multiple risks simultaneously in the most cost-effective manner. Clustering of risks is useful both during Problem Formulation (to identify a problem set that will maximize cost-effective risk reduction) and Analysis (to identify the full range of risk reduction benefits associated with any set of control options). A risk reduction program designed to address sets of priority risks will likely differ from a program designed to address a single risk, and should include consideration of the effects (positive or negative) of management options on "non-target" risks.

The SAB recognizes that the Agency often faces a mandate (legislative or court-

ordered) to reduce the risks from certain stressors to no more than some maximum level. In these situations, the Agency can use the IED framework to develop risk reduction options that cost-effectively reduce "collateral" risks from other stressors while meeting the mandated risk level for a particular stressor.

The SAB also recognizes that this type of integrated options analysis is a more difficult undertaking than the traditional approach and that such an exercise would require many estimations regarding combined effectiveness of management options, each subject to uncertainties. However, as the Agency gains experience with and develops new tools for such integrated analyses, the quality of the decisions will improve, and the improvement of environmental conditions should become apparent.

Chapter 6 describes a process for identifying, screening, and selecting risk reduction options for environmental problems defined in terms of single or multiple stressors.

d) Integrated Analysis of Economic Consequences

Approaching environmental problems from the standpoint of integrated risks should improve our ability to identify the full range of benefits and costs, both monetized and non-monetized. In theory, economic analysis is an inherently integrated exercise in which all benefits, values, or gains are compared to the costs or losses from any environmental intervention. Many benefit/cost analyses in the past, however, have focused on the control of a single source of a chemical. This narrow focus is largely the result of the way in which the "problem" has been defined. When the approach to risk is broadened to examine the multiple effects of multiple chemicals stemming from a source and the reduction in multiple risks that will ensue from controlling those chemicals, the scope of the benefit/cost analysis will be broadened to include the multiple benefits from the reductions of multiple risks. Chapter 4 describes the basic principles that underlie benefit/cost analysis and Chapter 5 discusses issues surrounding the assessment of difficult-to-quantify benefits.

e) Integration of Performance Information

Many decision-making schemes make provision for later evaluations of the

decisions that have been made. In practice, however, such after-the-fact reviews too often do not occur. Once a decision has been made, there is a natural tendency to move on to new problems, rather than to re-visit old ones. And yet, integration of information on performance and environmental results is a critical feature of any approach that includes principles of adaptive management. Today's environmental problems are often so complex that, no matter how sophisticated the analysis leading to an initial judgment, it is from reviewing the effects of and making adjustments to an earlier decision that a risk manager becomes better equipped to make future decisions.

Performance evaluation and feedback is an integral part of the IED framework; this component of the framework is summarized in section 1.3.4 and described in more detail in Chapter 7.

f) Integrating Multiple Disciplines and Points of View

Each of the five types of integration described above depends upon, to varying degrees, effective interactions between scientists, decision-makers, and interested and affected parties; i.e., the integration of information and understanding of all concerned parties in the IED process. As noted above, the IED framework requires not only the use of information and perspectives from a number of scientific disciplines, but also the inclusion of public values throughout the decision-making process. In practice, this means that, although the in-depth technical tasks are undertaken by experts in discrete, analytic exercises, the experts periodically will be informed and their results reviewed by other IED participants. Although much of the emphasis in this report is on inclusion of perspectives from outside the Agency, coordination and communication within the Agency during the course of the various analyses will be critical as well. In this way, through a series of deliberative and analytic processes (NRC, 1996), all of the participants gain an understanding of the value-based concerns of the others. In fact, the IED calls for recognition, inclusion, and consideration of values throughout the process. It is only through explicit inclusion and integration of the values and perspectives of a diverse set of participants, from within and outside the Agency, that the most acceptable and effective environmental risk reduction will occur. The problems and promises of this important type of integration are discussed in Chapter 5.

1.4.3 Building on Previous Frameworks

From the previous section, the reader can see that the goal of IED is to identify appropriate risk reduction actions to address a mixture of environmental problems (both human health and ecological), so as to reduce total aggregate risk, rather than focusing only on single risks in isolation. Deliberative processes are one means of deciding which mix of risk reduction goals should be pursued. The IED framework explicitly includes a mechanism for evaluation and feedback, so that a strategy of adaptive management can be easily employed. In addition, the framework includes substantive interaction with interested and affected parties throughout the process so that public values are reflected appropriately.

The IED framework proposed by the SAB builds upon several previous frameworks, in particular the risk assessment/risk management model described by the National Research Council (NRC 1983), the ecological risk assessment framework (U.S. EPA 1992), and the risk characterization process described in NRC (1996), which focused on the interaction between analytic and deliberative processes in decision-making. The IED framework moves beyond these earlier efforts in three significant areas:

a) Evaluation of Single Stressors

Although the SAB emphasizes that the Agency should consider multiple stressors in an integrated approach to risk, it recognizes that the IED framework also should enhance the decision-making process for single stressors, of the type historically considered by the Agency (e.g., a drinking water pollutant). When a single stressor is considered, the IED framework should expand the previous approaches by:

(1) Characterizing stress-effects relationships across all systems and populations;
(2) Exploring a broader range of risk reduction options, some of which may be qualitatively new;
(3) Assessing the benefits and costs of each option, including explicit consideration of non-monetary benefits and costs;
(4) Assessing the magnitude and nature of the aggregate risk reduction

associated with each option;
(5) Involving scientists, options analysts, stakeholders, and risk managers collectively at various points throughout the process; and
(6) Establishing a performance evaluation "report card" to characterize the efficacy of the implemented risk reduction option and to signal both the need and opportunity for adapting the original management decision.

b) Evaluation of Multiple Stressors

One of the most important extensions in the IED framework is that, unlike previous frameworks, a primary goal of the IED process is to consider multiple stressors when formulating the problem and developing possible solutions. The IED provides a structured way to begin to explore multiple ecological risks, multiple human health risks, and/or multiple quality of life concerns. In some cases this approach will lead to consideration of combinations or groups of risks; e.g., all organophosphate pesticides, all automobile emissions, or all factors leading to local loss of biodiversity.

Initial steps in the IED process are designed to produce relative rankings of risks to human health, quality of life, and ecosystems, independently. While the SAB had originally planned to integrate the rankings for the various types of risks, the members concluded that such a merged ranking is based on values, rather than on science alone. The SAB's review of risks of concern also emphasizes that health and ecological risk assessments are often qualitatively different. For example, the focus of the assessments is different; that is, the focus of health risk assessment is an individual within a single species, while the focus of even a narrowly drawn ecological risk assessment is entire populations of any/many species. More generally, ecological risk assessments often address the integrated risks to a prescribed region, such as a watershed. This difference is reflected in the different types of stressors of concern; cf., carcinogens for humans vs. habitat fragmentation. However, in those instances where there are common stressors of concern (e.g., chlorinated pesticides, climate change) or where effects of a stressor on an ecological system produce effects on human health or quality of life (e.g., habitat alteration that affects the range and activity of disease vectors and infective parasites, or changes in the abundance of commercially important or endangered species), there is an opportunity for some merging of concerns to take place. An understanding of the relationships among different types of risks/stressors is

critical to Problem Formulation, both in terms of setting environmental priorities and goals, and for designing subsequent risk and risk reduction analyses.

In addition to ranking of risks, which allows scientists to compare the expected impacts of each risk, the IED framework suggests combining risks into logical groupings, e.g., those with a common source or pathway, in order to identify risk reduction opportunities across stressors. In order to be successful, this analysis requires open, publicly-accessible, and frequent dialogue among those who assess and compare risks, those who determine methods for reducing risks, and those who make the final decisions.

In summary, as the IED framework is implemented to address multiple stressors, it should:

(1) Lead to a more realistic ranking of risks to humans and to ecosystems , where some of those risks may be posed by combinations of related stressors;
(2) Lead the Agency to consider in a systematic fashion all of the appropriate factors related to risks in a given circumstance, including aggregate risk, economic factors, and societal values; and
(3) Lead to action that will increase the reduction in aggregate risk posed by a combination of stressors in a given circumstance.

As the Agency gains experience and develops more sophisticated analytical tools, the benefits of IED should become more apparent. At first, it may be easiest simply to examine the total reductions in exposures. More complete evaluations of outcomes are complicated by the need to estimate reductions in several types of risk (e.g., health, ecological, and quality of life risks) resulting from the reductions of multiple stressors affecting multiple receptors. Nonetheless, appropriate methods should be developed to measure the total impact and benefits of risk reduction programs in the future.

c) Considering Environmental Values

The third extension that the IED framework provides is a more explicit

consideration of the analytic/deliberative process described by the NAS Panel on Risk Characterization (NRC, 1996). It is through such a process that societal values intersect with the scientific risk characterization and risk reduction analyses. The proposed IED framework emphasizes the role and timing of stakeholder and decision-maker inputs to the analytic processes. It explores more deeply the valuation of environmental outcomes and risks and the need to include not only the concepts of economic efficiency and willingness-to-pay, but also issues of environmental sustainability and equity. The IED framework recognizes that societal values constitute the milieu in which integrated environmental decision-making occurs forming the basis for societal goals for improved social welfare, improved ecological/health conditions, and long-term sustainability and equity. It is in the realm of social values that the success or failure of environmental decisions will be judged.

1.5 Benefits and Challenges of the Framework

1.5.1 Benefits

A fully implemented IED framework should result in a new level of environmental protection. The IED framework refines and moves beyond the risk ranking exercises contained in the SAB's *Reducing Risk* report. The techniques described in this report for ranking health and ecological risks offer a clear and systematic approach for using the best scientific information and expert judgment and are substantially more transparent than were those in the earlier effort. In addition, the IED framework goes beyond the comparison of risks and risk reduction options to spotlight the as-yet unrealized benefits to be gained from addressing environmental problems in a truly integrated fashion -- including integration of technical information on risk (including information on multiple stressors, sources, and effects), risk reduction options, and economic consequences; integration of values and goals into the decision-making process; and integration of performance information so that needed course corrections can be identified. While the Agency has made some significant first steps in some of these areas, the full benefits are yet to be achieved.

The expected benefits of the IED framework relate directly to the characteristics of the framework described in this chapter. Transparency of the decision-making process should increase acceptance of the outcome by affected parties, flexibility allows the process to be adapted to suit the needs of a particular situation, open and inclusive dialogue should lead to broader consideration of perspectives and options, and so forth. Broadly speaking, then, implementation of the IED framework should result in:

a) Enhanced ability to improve human health and quality of life and to protect the integrity of ecological systems;

b) Improved targeting of resources for risk reduction; and

c) Greater accountability for results;

1.5.2 Challenges

At the same time, the IED framework presents the Agency with significant challenges. The techniques recommended for ranking risks in the IED will need further refinement during their development and application. Likewise, the concept of combining risks into related subsets for risk management purposes offers considerable promise but introduces new uncertainties that must be acknowledged and addressed. The integration of economic concepts, both for setting environmental goals and for evaluating how best to meet those goals, is important but may also be controversial and, therefore, must be introduced clearly and objectively to avoid unwarranted criticism. The consideration of management options from the total aggregate risk vs. single risk perspective, which is a major feature of the framework, could also be controversial, even in the face of its technical soundness. Finally, the concept of evaluating the impact of decisions and reflecting that information in subsequent management actions (i.e., adaptive management) is not new; however, the challenge of using this approach in a consistent and continuing way remains.

In addition to the technical and methodological issues, it is important to recognize that successful implementation of the IED framework will require some adjustments in the manner and degree to which the many participants interact over the course of the decision-making process. Integrated environmental decision-making requires the sharing of information, ideas, approaches, and management deliberations to a degree now seldom practiced among individuals of very different backgrounds. Although this sharing is a positive aspect of the IED framework, it may require significant adaptation on the part of individual policy-makers and institutions. For example, decision-makers will need to interact more extensively with scientific and technical analysts and the public in the course of developing integrated approaches to environmental risks. Likewise, scientific and technical experts will need to recognize the role (and limitations) of science in decision-making, and also to recognize the legitimate role of values in establishing environmental goals and selecting management approaches. This culture change will be assisted by familiarity and experience with the IED process. Experience, combined with discipline, will also be needed to apply the IED framework with discretion to the depth of detail necessary for a particular problem, and no more; i.e., the IED framework should not become a barrier to decision-making.

Additional challenges likely to be encountered when implementing the IED framework include:

a) problems of understanding arising from differences in terminology and outlook imbedded in the different disciplines and backgrounds of the participants in the process;

b) difficulties of using both qualitative and quantitative measures concurrently in the decision process;

c) the need to compare different types of risks (e.g., health, ecological and quality of life risks) within a common decision framework and to discern and define the inter-relationships among risks so as to define common goals across the different risk types; and

d) time-lags between implementation of risk reduction plans and the detection of results and effects, which make the selection of appropriate performance measures particularly important.

Facing and surmounting the challenges, learning how to use the IED framework flexibly with discretion and care, and improving the underlying methodologies over time, will lead to more effective environmental decision-making and to measurable environmental progress. The SAB expects that the proposed framework and the recommended improvements to the component analyses will assist the Agency to develop a more comprehensive, integrated, and transparent process and culture for setting environmental priorities and making and implementing decisions.

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PART II INPUTS TO ENVIRONMENTAL DECISION-MAKING:

RISK COMPARISONS

PART II — INPUTS TO ENVIRONMENTAL DECISION-MAKING: RISK COMPARISONS

Preface

The process of comparing environmental health and ecological risks is a key component of the Problem Formulation Phase of the IED framework. In the early stages of problem formulation, a draft list of stressors of concern will be identified using existing knowledge and expertise of a group of scientists (including human exposure and health experts and/or ecological experts) and/or on the basis of a review of available scientific literature. A comprehensive list of possible stressors is developed to provide a complete set of possible comparisons. The draft list developed by risk scientists would then be discussed with others in the IED process, including the options analysts, decision-makers, and stakeholders. As a result of these discussions, additional stressors might then be added or removed, some groups of related stressors might be aggregated into a single composite stressor (e.g., "waterborne infectious microbes" vs. "*Cryptosporidium*"), and some stressors might be disaggregated into more specific stressors (e.g., "mercury" vs. "heavy metals") for subsequent analysis. Quantitative and qualitative information on the stressors would be assembled into a format, such as a stressor risk data sheet, to facilitate comparisons. Important information would include: stressor intensity; the adverse effects likely to be associated with exposure to the stressor; observed effects in the population or ecosystem; and some assessment of the causal relationships between the environmental stressor and the observed effects. This latter could include assessments of the probable contribution of the stressor to the observed effects of concern, the so-called attributable risk. Major gaps in knowledge should also be identified as part of Problem Formulation and Analysis in order to inform research planning and help set research priorities.

During the design of the IED framework, the SAB participants acknowledged that technical risk rankings, in isolation, offered insufficient guidance for policy decisions. Given the multitude of problems and issues to be addressed, a more comprehensive and systematic framework for analyzing and reducing environmental health, ecological, and quality of life risks appeared necessary. Analysis and comparison of various

environmental health and ecosystem risks, however, remain the fundamental basis of such a framework, and the foundation for defining risk reduction priorities.

As symbolized by the interlocking circles in the IED framework, the assessment and comparison of risks and the definition of environmental goals are inter-related. In other words, scientific information on the nature and extent of various risks influences the relative priority that society places on those risks. In addition, however, non-scientific issues such as dread, previous experiences, and degree to which exposure is voluntary, for example, also influence the relative priority assigned by the public to environmental risks. For this reason, the ultimate risk priorities that emerge during Problem Formulation will be a product of the interaction between the risk comparison and goal setting processes, and (as described in Part IV) the preliminary assessment of risk reduction potential.

As the IED process proceeds from the problem formulation to the analysis phase, the risk assessors develop more detailed risk analyses for the priority stressors, which can be used to help focus the development of options. The stressor comparisons will be used to help organize thinking about potential options for reducing risks and to examine opportunities for reducing several risks through a single risk reduction option. The more detailed information on exposures and risks also provides the basis for estimating the risk reduction achievable through various options, and for determining the specific environmental outcomes that will be used to evaluate performance against the risk reduction goals.

During Problem Formulation, it is important to identify linkages between health, ecological, and quality of life risks and to seek opportunities to address all three types of risks through a risk reduction program. For this reason, it is helpful to define the risk problems using a common dimension; the SAB Subcommittees chose to define risks in terms of stressors.

Stressors of concern may be chemical (e.g., ozone, heavy metals), physical (e.g., radiation, changes in land use), or biological (e.g., infectious agents in drinking water, introduced exotic species). Further, the stressors may cause adverse health and ecological effects directly (through exposures via environmental media) or may act indirectly. For example, freons released into the atmosphere do not directly influence

health, but, instead, reduce the thickness of the protective ozone layer of the atmosphere, consequently increasing the risks of skin cancer. A methodology for comparing environmental risks to quality of life was not developed by the SAB because the selection of QOL ranking criteria is more appropriately done by a broad group of public or stakeholder representatives rather than by a technical panel. However, the SAB recognizes the need to develop and incorporate such a process into integrated environmental decision-making.

Two Subcommittees of the IRP developed risk comparison approaches whose results could be used in the IED process. One approach, which was applied to the comparison of ecological risks, utilizes an expert group to develop and weight risk ranking factors in order to produce the group's consensus judgment of the relative risks associated with various stressors. The second approach, which is discussed using human health risk issues, involves polling experts individually for their professional judgment of relative degree of risk associated with various stressors, while soliciting information on which factors most influenced this judgment and the degree of confidence each expert assigns to his/her assessments. The latter method provides information not only on the "average" or "median" risk ranking, but also on the range of expert opinions. The consensus approach or the individual polling approach could be applied to either human health or ecological risk comparisons using stressors and rating factors specific to each group. The methodologies also could be used to elicit stakeholder or public views on risk priorities by expanding the composition of the surveyed or empaneled group.

The Ecological Risk Subcommittee (ERS), as a group of experts, developed a set of decision rules for ranking ecological risks. The Subcommittee identified a set of stressors of ecological importance and established a systematic template for considering each stressor with respect to issues of ecological importance (e.g., spatial extent and duration of the stress, intensity of the ecological effect, recoverability once the stressor is removed, and other issues of special ecological significance). The group then developed a process to characterize the co-occurrence of stressor intensity and ecological consequences for an at-risk ecosystem. For each identified ecological factor of importance, a multiplicative value was assigned that would adjust the risk level up or down for each individual stressor. Once all the factors were considered, the initial risk assignment was adjusted by each factor, leading to a summary risk value for each

stressor. Stressors were then placed in narrative risk categories on the basis of the numerical risk scores, which derived directly from the application of the ecologically based criteria.

An important advantage of the ERS method is the transparency of the factors and their assigned relative weights used for deriving the risk rankings. The dialogue among the convened experts helps to ensure that there is a common understanding of terminology and of the specific criteria for assigning ranking factors, a process that is very difficult to accomplish when polling individuals. A potential disadvantage of this approach is that the quantitative scores calculated for the risk rankings could be misused by attributing greater precision to the numbers than the methodology warrants.

The Human Exposure and Health Subcommittee (HEHS) adopted a process based on the use of the World Wide Web to solicit risk comparisons from a large number of experts in fields related to human health risks. One reason for the choice of such a methodology was the extreme breadth of disciplines pertaining to environmental health, e.g., epidemiology, exposure analysis, and toxicology. The respondents are provided with a list of stressors, baseline information regarding these stressors and their associated outcomes, and a consistent template to provide their comparisons of risks to human health attributable to exposures to the stressors. As a part of the survey, a set of about ten factors that might relate to the risk comparison is provided, and the respondent is asked to indicate those influencing her/his rating for each stressor.

An important advantage of the HEHS methodology is that it provides information on the variability in expert opinion about the relative degree of risk associated with various stressors, as well as on the confidence of the experts in their risk comparisons. A disadvantage of the methodology is that the opportunity for interaction and information-sharing among the expert participants is lost, and the effort required to prepare a common set of information for all risk areas to be compared is substantial.

Table PII-1. Correspondence of Human Health and Ecological Risk Comparison Factors

Human Health Risk Factors	Ecological Risk Factors
Size of population affected	Proportion of resource at risk
Particular subpopulations at risk	Distribution of "hot spots"
Severity and persistence of health effects	Recovery potential, Species depletion
Persistence in the environment and/or human body	Duration of stress-effects co-occurrence
Percent of attributable incidence	
Potential future risk	
	Secondary stress induction
	Special ecological significance

Despite the apparent differences in the ERS and HEHS approaches to risk comparisons, the factors influencing the relative ranking of risks share many commonalities (Table PII-1). The primary difference in the two approaches lies in the manner in which the decision rules are derived. The relationship between the methodologies is illustrated in Figure PII-1. Following from left to right in this figure, the initial comprehensive list of stressors is identified and information concerning stressor-effect relationships is collected. This information is provided to a group of experts, either constituted as an expert panel (as in the ERS approach, depicted by the feedback arrow from the Risk Comparison box to the Decision Rules box) or as individual experts polled separately (as in the HEHS approach). In the HEHS methodology, the risk comparisons are derived directly from the experts. The experts are also asked to indicate the factors that influenced their risk ratings for each stressor. A multivariate regression analysis could be used to relate the risk ratings to the various factors that the experts indicate influenced those ratings. The factor weights from the regression analysis then define the Decision Rules. In the ERS approach, the experts develop the Decision Rules directly, then apply them to the stressor database to derive the Risk Comparisons.

One clear advantage of illustrating the two approaches in the context of a unified framework is the provision for verification of the ratings generated using either approach. In the case of the ecological risk comparisons, a larger group of individual experts could be polled explicitly with respect to the ecological risk adjustment factors, and a regression analysis on that database could be compared with the Decision Rules developed by the expert panel. Consistency of the weighting factors would provide an increased confidence in the overall risk comparisons. Correspondingly, following the HEHS process, a subsequent stage might be to convene a panel of human health experts with sufficient diversity of perspectives and assign them the task of weighing the information generated by the surveyed experts then formulating and applying explicit decision rules to arrive at a list of health risk ratings.

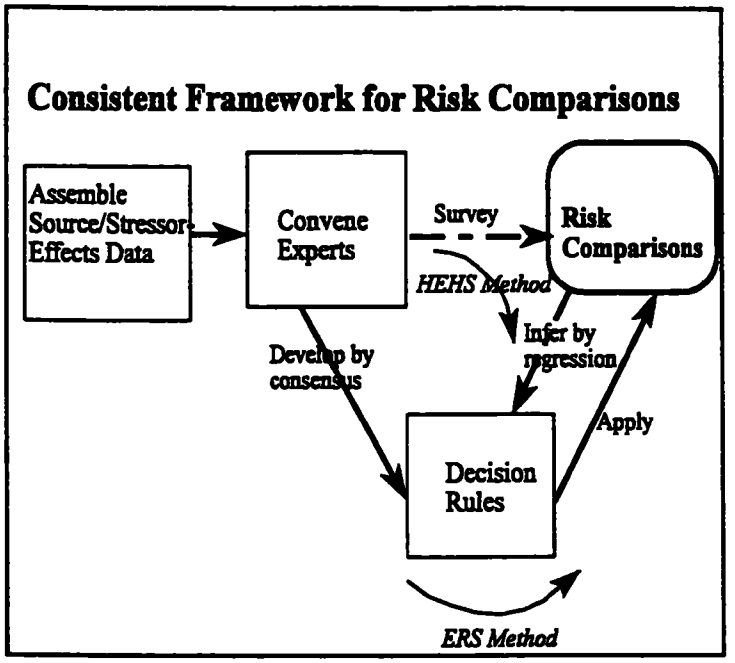

Figure PII-1.

The two subcommittees also considered the possibility of merging human and ecological risks into a single science-based ranking, but realized that the comparison of health risks to ecological risks was not possible on an objective scientific basis because it is fundamentally based upon considerations of societal values. Nevertheless, the use of analogous methods for developing technical risk comparisons for both human health risks and ecological risks, as described in the following chapters, should facilitate the subsequent setting of societal values-based risk priorities during Problem Formulation.

An additional contribution of the risk comparison methods proposed by the two subcommittees lies in their transparent use of decision rules for comparing risks. The

explicit statement of decision rules, and the relative weights or importance assigned by the participants to various considerations, makes clear what assumptions, scientific judgments, and values went into the final results. In this way, users of the approaches can see how the results would change if the judgments about decision rules were to change.

Chapters 2 and 3 describe in detail the approaches developed by the ERS and HEHS, respectively, for comparing multiple environmental risks.

CHAPTER 2: ECOLOGICAL RISKS

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CHAPTER 2. ECOLOGICAL RISKS

2.1 Background

During the 1980s, the U.S. Environmental Protection Agency (EPA) initiated extramural research to develop new approaches to ecotoxicology and to conducting ecological risk assessments (Barnthouse and Suter, 1986; Levin et al., 1989; Harwell and Harwell, 1989; Kelly and Harwell, 1990). The initial development of a methodology to compare and rank ecological risks began in 1986 with the cross-agency *Unfinished Business* Project (EPA 1987a, b). As part of that project, EPA convened an ecological risk panel to develop and apply a methodology to rank the risks to ecological systems of 31 listed "environmental problem areas." The methodology and results of the ecological panel were reported in Harwell and Kelly (1986) and in EPA (1987b). The ecological panel decided that the list of environmental problem areas that had been developed by EPA was influenced more by programmatic considerations than by what an ecological system might experience. Consequently, the panel developed a comprehensive list of environmental stressors, defined as those physical, chemical, or biological changes that might affect an ecological system. Many of the stressors were related directly to the list of environmental problems, but other stressors were identified that were not captured by the list of environmental problems. The ecological risk panel also developed a list of 16 ecological system types, defined so that differences in exposures and/or responses to the stressors could be identified.

In 1989, as part of the EPA Science Advisory Board's Reducing Risk Project, the SAB established a subcommittee on ecological and welfare risks, the latter addressing environmentally mediated factors affecting the quality of human life. That SAB ecological subcommittee adopted and further refined the methodology of the previous *Unfinished Business* panel and developed an updated set of relative ecological risk rankings for the same set of environmental stressors (SAB, 1990a). The summary of the SAB Reducing Risk project and reports of the other subcommittees are contained in the report entitled, *Reducing Risk: Setting Priorities and Strategies for Environmental Protection* (SAB, 1990b).

In *Reducing Risk*, the SAB recommended that EPA enhance the consideration of ecological risks to a level comparable to human health risks and develop guidance for implementing ecological risk assessments. In response, the Agency established a program in the Risk Assessment Forum to develop a new framework for conducting ecological risk assessments. Two major advances were necessary: a) while chemical ecological stressors are analogous to the chemicals considered in human health risk assessments, the domain of ecological risk assessment was broadened also to include biological and physical insults that may cause significant ecological effects; and b) while human health risk assessments address only one species and often only a single endpoint (human cancers), assessment of ecological risks almost always requires considering multiple endpoints because of pervasive differences across species, ecological systems, stressors, and organizational hierarchy (i.e., from species to landscape and larger scales of ecosystems). Consequently, based on two EPA-sponsored workshops (Harwell and Gentile, 1992; Fava et al., 1992), EPA issued a new ecological risk assessment framework (EPA, 1992) which expanded on the NRC (1983) Red Book risk assessment paradigm to include the full suite of natural and anthropogenic stressors (i.e., not just chemicals) that affect the environment at population, community, ecosystem, and landscape levels. The EPA ecological risk assessment framework calls for identification of the at-risk components of ecological systems, selection of ecological endpoints, development of a conceptual model that describes the ecological system and its stress-response relationships, mutual characterization of the stress regime and ecological effects in terms of the selected endpoints, and integration into an overall assessment of ecological risk with explicit accounting of uncertainties. Since issuing the framework, the Agency has issued ecological risk assessment guidelines (EPA/630/R-95/002B) that expand on and modify the framework document, and it has issued guidance on cumulative risk assessment (U.S. EPA, 1997) that further builds on the ecological risk framework. Neither the ecological risk assessment framework nor EPA's proposed ecological and cumulative risk assessment guidelines, however, explicitly describe the methodology needed to rank the relative importance of various ecological risks. Developing an improved methodology for the relative ranking of ecological risks is the focus of this chapter.

2.2 Objectives and Approach

As part of the SAB Integrated Risk Project (IRP), the Ecological Risks Subcommittee (ERS) was formed to address relative risks to ecological systems. The objectives of the ERS were to:

a) examine and refine the methodology for assessing relative ecological risks;

b) develop criteria to rank ecological risks at the national level;

c) apply those criteria to perform a national-level relative ranking of ecological risks;

d) develop a methodology for implementing relative ecological risk assessments at regional or local scales; and

e) assist in integrating ecological risk assessments into a broader conceptual framework that includes ecological, human health, and welfare risks.

To accomplish these objectives, the ERS carefully considered the relative risk methodologies developed in *Unfinished Business* and *Reducing Risk* as well as the more recent ecological risk assessment framework. The Subcommittee concluded that a new methodological framework for evaluating relative ecological risks was needed that expands upon the previous methodologies but adds transparency and clarity and is adaptable to multiple scales of analysis. As detailed in the following sections, the new relative ecological risk methodology focuses primarily on the stressor-effects model to assess ecological risks, but it can also be adapted to apportion the relative contributions of various stressors to an observed effect in the environment. As a part of the methodology development process, the ERS updated the list of environmental stressors and ecological systems of concern, and used a matrix approach to capture information about stressor-effect relationships so that both quantitative and qualitative assignment of risk rankings could be accomplished. The ERS ecological risk ranking methodology was applied at the national scale, and the necessary modifications were identified to apply the methodology to regional, state, or local levels. Close dialogue

with the IRP Human Exposure and Health Subcommittee (HEHS), charged with assessing relative human health risks, has resulted in a consistent framework for both types of environmental risks and allows explicit consideration of the linkages between ecological risks and human health risks.

2.3 ERS Ecological Risk Ranking Methodology

2.3.1 Overview and Rationale

The basic approach proposed by the ERS for ranking the relative significance of ecological risks follows the ecological risk assessment framework (U.S. EPA, 1992), particularly with regard to addressing all types of ecological stressors (physical, chemical, and biological) affecting the important attributes (endpoints) of ecological systems. This approach requires consideration of the two fundamental components of ecological risks for each stressor: a stress or exposure regime, and a response or ecological effects regime. The stressors, exposure patterns, ecological endpoints, and ecological effects are aggregated as appropriate for the scale of the analysis, recognizing that the goal of comparative ecological risk assessments is to characterize the dominant relationships between environmental stressors and ecological effects. For example, in the national-scale comparative risk assessment presented here, ecological endpoints and effects generally were characterized only at the ecosystem and landscape levels, even though ecological effects from a stressor may be observed at the population and community levels as well. Similarly, ecosystem-specific differences in response were generally noted, but the Subcommittee chose to focus on those types of ecological systems considered to be most at risk.

The first step in the comparative ecological risk assessment is to determine the potential ecological importance of each stressor at the ecosystem level or landscape level for ecological systems at risk. This ecological risk characterization is based on a stress-effects profile, i.e., a graphical representation of a stressor's exposure regime with its ecological effects for each at-risk ecosystem type. The stressor exposure is categorized into high, medium, and low levels, normalized to actual exposures in the real world, and effects distributions associated with each stressor levels are estimated. The ecological effects are assigned high, medium, and low intensity categories, based on criteria of the ecological significance of the effect (e.g., high-level effects reflect

major disruptions to the fundamental structure or processes of an ecosystem). The next step in the methodology is to transform these generic ecosystem- or landscape-level stress-effect relationships into a relative ranking of risk at a specific scale (e.g., regional, national, or global) by applying a series of numerical modification factors that relate to the ecological significance of the effect. The resulting modified numerical value constitutes the risk score for a stressor at the specified scale of interest. The relative risk posed by the stressors can then be determined by comparing scores across the stressors and by grouping these scores into qualitative risk categories (e.g., very high, high, medium, and low relative risks). This categorization reflects the semi-quantitative nature of the procedure and identifies clusters of stressors with similar levels of risks. Thus, the primary import of the aggregated rankings is their relative value, i.e., what stressors have relatively the same level of ecological risk, what set of stressors is much higher in risk than others, and so forth. Details of the risk ranking methodology and its application by the ERS to the national scale are described in the remainder of this chapter.

2.3.2 Ecological Risk Characterization

2.3.2.1 Selection and Aggregation of Stressors and Ecosystems

The ecological risk ranking process begins by identifying the stressors that potentially pose environmental risks and the set of ecosystem types for consideration of the risks. For this project, lists of stressors identified by the ERS and by the HEHS were merged. Table 2-1 notes those stressors, or groups of stressors, that may affect ecological and/or human health endpoints. Identifying a common set of stressors facilitates direct comparison of risks to humans and ecosystems. For the ecological risk ranking process, the stressors were characterized at the level of detail sufficient to distinguish between different types of stressor-effect relationships, but not so detailed that an unmanageable number of stressors had to be evaluated. For example, pesticides were evaluated as a class of stressors, rather than on a chemical-by-chemical basis. Some general categories of stressors were disaggregated where appropriate; for example, the stressor class of habitat alteration was divided into specific types of habitat alterations (e.g., habitat fragmentation, habitat conversion).

Table 2-1: Environmental Stressors of Concern

Stressor	Ecological Risk	Human Health Risk
I. CHEMICAL STRESSORS		
Criteria Air Pollutants		
CO	NA	✓
NO_x	✓	✓
O_3	✓	✓
SO_2	✓	✓
Airborne Particulates	*	✓
Toxic Inorganics		
Hg	✓	✓
Other Heavy Metals	✓	(✓)
Other Toxic Inorganics (As, Se, B)	✓	(✓)
Asbestos	NA	✓
Persistent Toxic Organics	✓	(✓)
Endocrine Disruptors	✓	✓
Pesticides, Herbicides	✓	✓
Environmental Tobacco Smoke	NA	✓
Volatile Organics	NA	(✓)
Toxic Anions		
Nitrate, nitrite	NA	✓
Acid Deposition	✓	NA
Nutrients	✓	NA
Radionuclides		
Radon	NA	✓
Others (e.g., Cs, Sr)	✓	✓
Oil/Fuel Leaks, Spills, and Use	✓	✓
Dissolved Oxygen/BOD	✓	NA
Acid Mine Drainage	✓	NA
Contaminated Ground Water	✓	NA
II. BIOLOGICAL STRESSORS		
Bioaerosols/Allergens	NA	(✓)
Human Disease Agents		
Airborne Viruses, Infectious Agents	NA	✓
Waterborne Infectious Microbes	NA	✓
Non-human Disease/Pest Outbreaks	✓	NA
Introduced Species	✓	✓
Genetically Engineered Organisms	✓	✓
III. PHYSICAL STRESSORS		
Climate Change (global, regional, and micro-climate)	✓	✓
Noise	✓	✓
Light Pollution	✓	✓
Habitat Alteration		?
Hydrologic Alteration	✓	
Habitat Fragmentation	✓	
Habitat Conversion	✓	
Physical Habitat Disruption	✓	
Turbidity and Sedimentation	✓	
Altered Fire Regime	✓	
Altered Salinity Regime	✓	
Thermal Pollution	✓	NA
Harvesting Living Resources		?
Coastal	✓	
Freshwater	✓	
UV-B	✓	✓
EMF	NA	✓

Several factors have to be considered in selecting the appropriate level of aggregation or disaggregation of stressors, including the exposure and effects data that are available, differential impacts on the ecological system types of interest, potential management options, and so forth. There are no simple rules for this classification process. For example, the Subcommittee examined alternative ways to aggregate chemicals, ranging from classifications based upon mode-of-action of toxicity to structural characteristics, uses of the chemical, or nature of the ecological effects. No single scheme was considered ideal for the purposes of this risk ranking activity. The classifications that the Subcommittee selected for ecological stressors were based primarily on the source of the chemical (e.g., criteria air pollutants), the physical and chemical nature of the chemical (e.g., nutrients, radionuclides), or, in the case of persistent toxic organics, the usage of the chemical (e.g., pesticides). A more detailed ecological risk ranking across a more disaggregated set of chemical stressors could be done following the same methodology as proposed here but using a different classification scheme; that is, the list the Subcommittee developed (Table 2-1) is not presented as the only or even the best way to classify chemical stressors. In fact, the appropriate level of stressor aggregation for relative human health risk assessment often differed, as noted in Table 2-1.

The set of ecological system types that were selected by ERS (Table 2-2) were modified from the set of ecological system types used in the *Unfinished Business* and *Reducing Risk* projects. The intent of this classification was to capture the range of ecological systems that occur in the U.S., but without having an unnecessarily large set of ecosystem types to consider. The specific set of ecological systems to use for risk rankings at less than national scales may be considerably different; for example, a risk ranking done on the Southeastern U.S. would not need to have the category of semi-arid and arid ecosystems, but it might have several different categories that better refine forested ecosystems (e.g., coastal hardwood forests and swamps, pinelands, oak-hickory upland forests, etc.).

Table 2-2. Types of Ecological Systems

Forests
Lakes
Rivers and Streams
Wetlands
Grasslands
Agroecosystems (Rangeland and Cropland)
Deserts and And Systems
Tundra
Estuarine and Near-Coastal Ecosystems
Marine Ecosystems

2.3.2.2 Development of Ecological Risk Profiles

The foundation of the comparative risk analysis is a set of individual ecological risk profiles (see Appendix A) that summarize for each listed ecological stressor the nature of the stressor and its potential ecological effects; define what is and what is not included within the stressor category; discuss how reliable and certain the information base is; and provide information on any other factors that should be considered when the relative rankings are assigned. Because these risk profiles are modular, they can be revised as new information is developed, thereby facilitating a reevaluation of the risk score for that stressor. In short, the risk profile provides not only the information used to develop the quantitative risk score (discussed below), but also a summary of the level of understanding about a stressor, references for that understanding, and a description of the uncertainties associated with it.

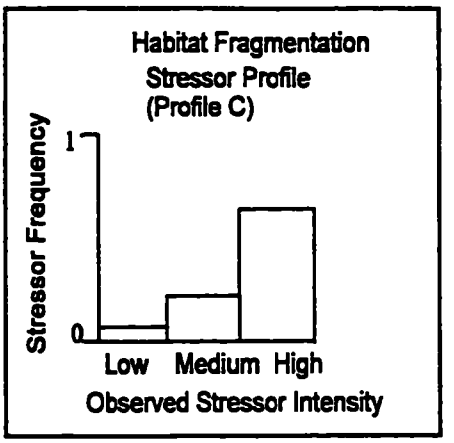


Figure 2-1a: Habitat Fragmentation Stressor Profile

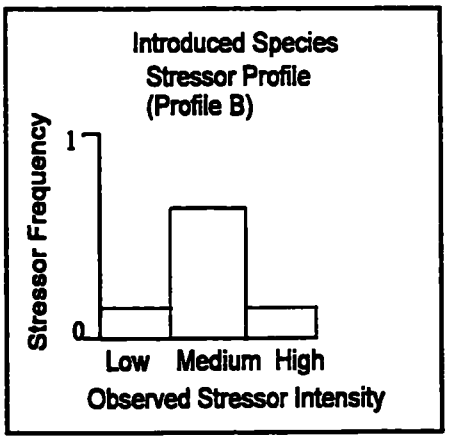


Figure 2-1b: Introduced Species Stressor Profile

An important component of each ecological risk profile is the stress-effects profile, a visual tool for illustrating the relationship between the stressor and effect regimes for each stressor at the ecosystem or landscape level. In essence, the stress-effects profile illustrates the significance of the adverse effects on at-risk ecosystems that are expected to occur as a result of current exposures to the stressor.

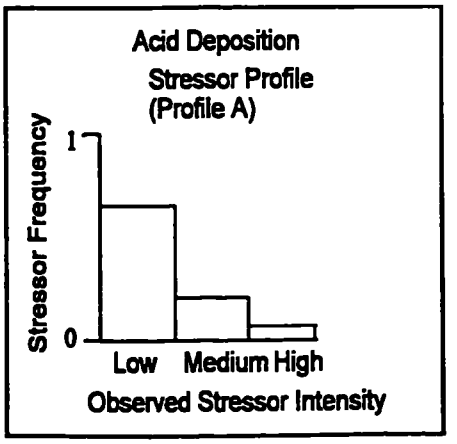


Figure 2-1c: Acid Deposition Stressor Profile

The first element of the stress-effects profile is the stressor profile, i.e., a frequency distribution that depicts the observed stressor intensity affecting the ecological systems at risk in the region of interest. The at-risk ecosystems may differ for each stressor;

e.g., all aquatic and terrestrial systems in the U.S. are the at-risk ecosystems for the stressors of habitat fragmentation (Fig. 2-1a) and introduced species (Fig. 2-1b), but only certain aquatic and forest systems are at risk from the stressor of acid deposition (Fig. 2-1c). The x-axis of the stressor frequency distribution is normalized to a relative scale of low, medium, and high intensity of occurrence of the stressor, based on an estimated actual distribution of the stressor in the environment. Thus, a high level of a stressor would relate to the highest levels found in the environment, not necessarily at the highest levels that have been tested in the laboratory. This procedure allows the ecological risk rankings to be based on relevant levels of the stressor, and it allows identification of the risk reduction that would be achieved if the stressor distribution in the environment were reduced. The y-axis of the stressor profile reflects the relative frequency of low, medium, and high occurrences of the stressor as experienced by the at-risk ecological systems, normalized on a scale of 0 to 1. Thus, if the ecosystem at risk is, for example, estuaries, this curve would illustrate the relative frequency or percentage of all estuaries that are exposed to high stressor intensities, the percentage that are exposed to medium stressor levels, and the percentage that are exposed to low stressor levels.

In practice, the ERS concluded that there are only a limited number of types of environmental stressor frequency distributions, such as log-normal, normal, and skewed. Thus, in order to simplify the ecological risk ranking process, a template of four types of stressor-profiles was developed (Fig. 2-2), and each stressor was assigned to one of these stressor-profile types to represent its frequency distribution. For example, stressor Profile A depicts the most common situation, in which there is the highest frequency of low levels of exposures, and lowest frequency for high levels of exposures; this would be typical, for example, for exposures of a chemical in the

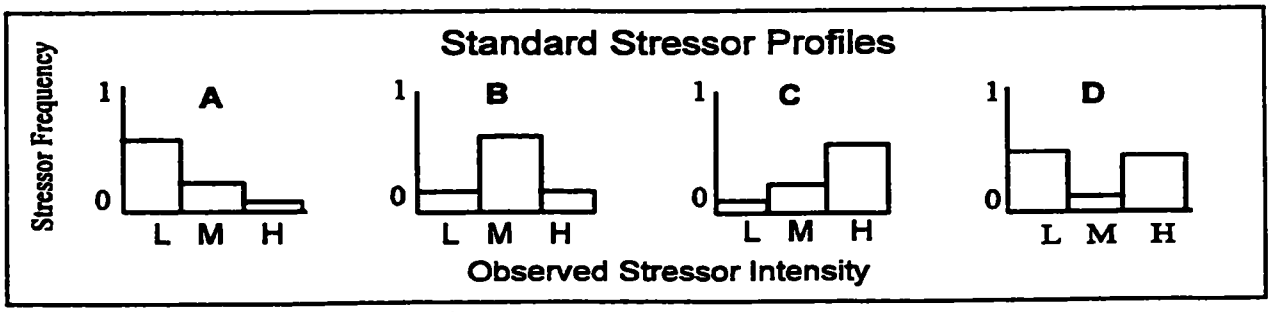


Figure 2-2: Standard Stressor Profiles

environment. Profile C represents the opposite situation, in which there is highest frequency of high-level of exposure; a case in point would be habitat fragmentation, in which most landscapes have become highly fragmented and only a relatively few have low fragmentation. Note that for a more detailed relative risk assessment, the basic template of profiles developed by the ERS could be replaced by the best available information on actual stressor distribution profiles. Also, note that the frequency distributions for a particular stressor are location- and scale-specific; for example, the frequency distribution of acid precipitation is different in the Northeastern U.S. than in the Southeast.

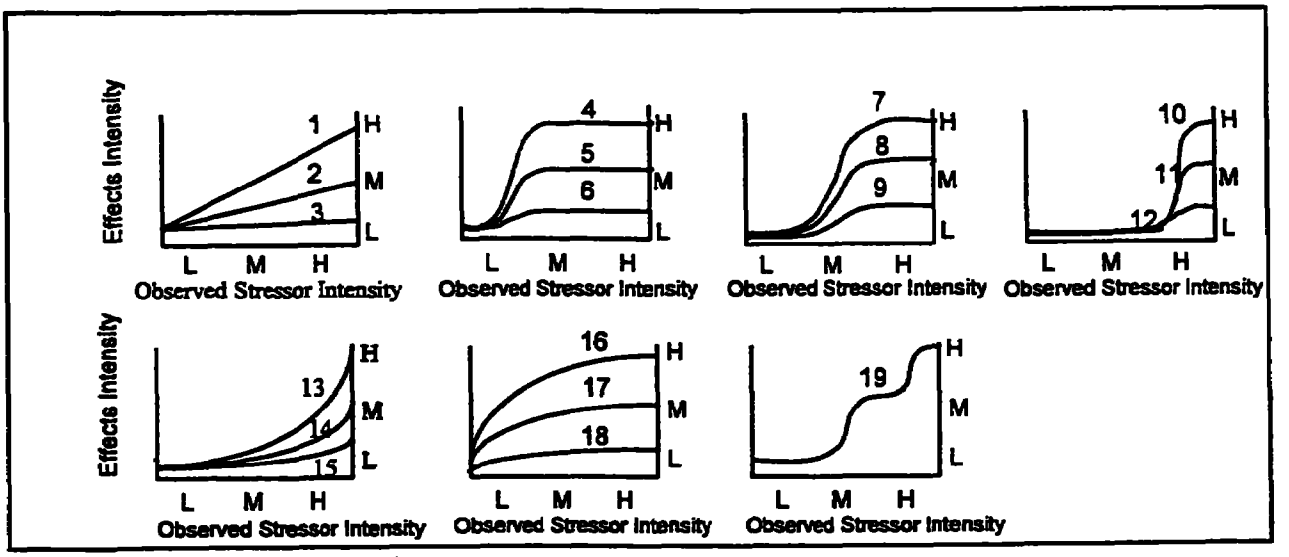


Figure 2-3: Standard Effects Curves

The other half of the stress-effects profile is the effects profile, i.e., the frequency distribution of ecological effects associated with each of the stressor levels. That is, the effects profile shows the intensity of adverse ecological effects that would result from exposure to high, medium, or low levels of a particular stressor. Again, a standard set of effects profiles was developed that the ERS believes represents most environmental situations (Fig. 2-3), and one profile was assigned by ERS to each stressor for the national-scale ranking. For example, Profile 4 illustrates a threshold situation, in which a medium-level stressor causes high-level effects, compared with Profile 10, in which

only exposure to a high-level stressor causes a high-level ecological effect.

Effects are evaluated for the ecological systems considered to be most susceptible to the stressor. The intent is to focus on the most important ecological endpoints for those ecological systems most at risk. For the purposes of the ERS national ranking, the ecological endpoints were constrained to include ecosystem- and landscape-level effects on the structure, function, and/or composition of the system. Examples include an ecological process endpoint such as decomposition rates, a community composition endpoint such as species richness, and a trophic structure endpoint, such as the health of a critical or habitat-creating species. Because of the focus on national risks, ERS did not address other potential ecological endpoints, such as the population levels of individual species. However, for ecological risk ranking conducted on local or regional scales, population-level or other ecological endpoints might supplement the ecosystem- and landscape-level endpoints.

Example effects profiles are shown in Fig. 2-4. Fig. 2-4a illustrates the case of habitat fragmentation, for which Profile 2 was selected to reflect that even high-levels of intensity of habitat fragmentation only cause medium-levels of ecological effects. This is in contrast to the greater potential ecological effect that could result from high levels of introduced species (Fig. 2-4b, Profile 1). For the acid deposition example (Fig. 2-4c), two profiles are shown, one for the more sensitive unbuffered lakes category (Profile 7), the other for forests (Profile 15), although both effects profiles relate to the same stressor profile. Again, in more detailed

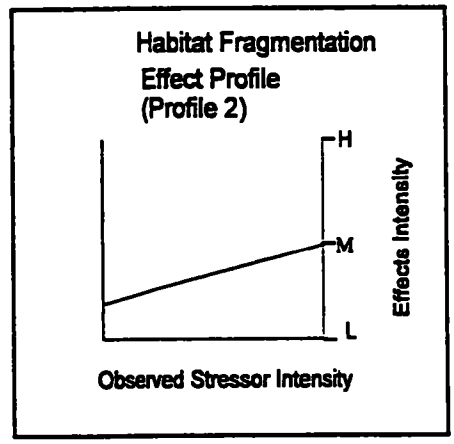


Figure 2-4a. Habitat Fragmentation Profile

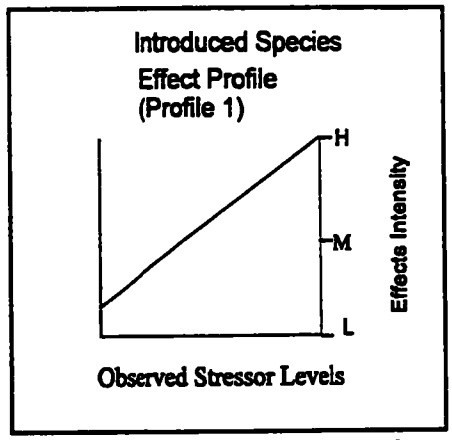


Figure 2-4b. Introduced Species Effects Profile

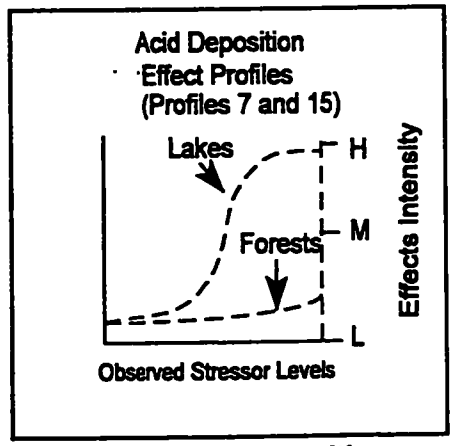


Figure 2-4c. Acid Deposition Effects Profiles

relative risk ranking assessments, a suite of stressor-effects profiles could be developed for each and every ecosystem type of importance in the area of concern, or even for each and every specific ecological endpoint affected by the stressor. While that is beyond the scope of the present application to national relative risk ranking by the ERS, it does illustrate the flexibility of the ERS methodology for specific risk ranking applications.

The combined stressor-effects regime, then, is depicted by overlaying the effects profile (i.e., the distribution of the intensity of effects caused by varying levels of the stressor) onto the stressor profile (i.e., the actual frequency distribution of stressor in the environment of concern) (Fig. 2-5). In this combined stressor-effects profile, the effects profile y-axis is scaled to reflect the intensity of the ecological effect (on a scale of high, medium, and low) associated with the intensities of the stressor that occur in the environment. The stressor profile y-axis is scaled to the frequency of occurrence (0-1) of the stressor in the specific environment of concern. The x-axis, however, is identical for the two profiles, i.e., reflecting low, medium, and high levels of stressor intensity. Thus, even a high-intensity stressor exposure (defined in terms of the levels actually experienced in the environment) might result in only a low-level ecological effect (e.g., acid deposition effects on forest ecosystems shown in Fig. 2-5c).

In general, the high ecological effects category was reserved for those ecological changes that are very significant, involving major changes to the

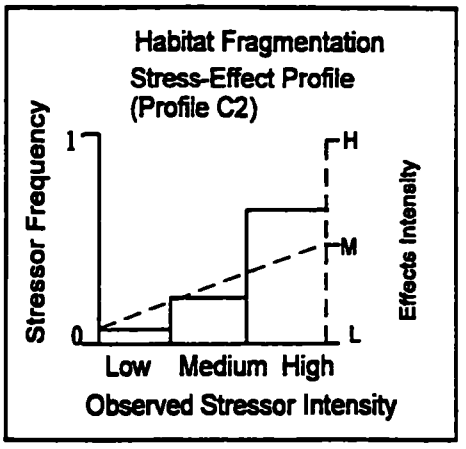


Figure 2-5a. Habitat Fragmentation Stress-Effect Profile

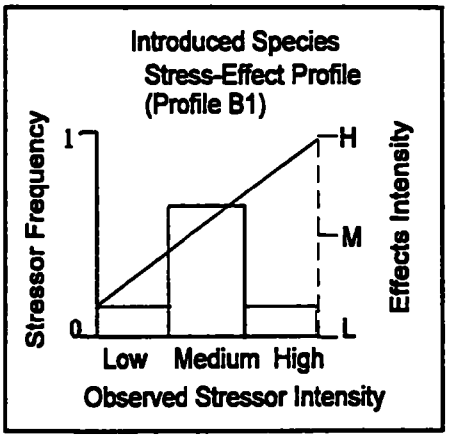


Figure 2-5b. Introduced Species Stress-Effect Profile

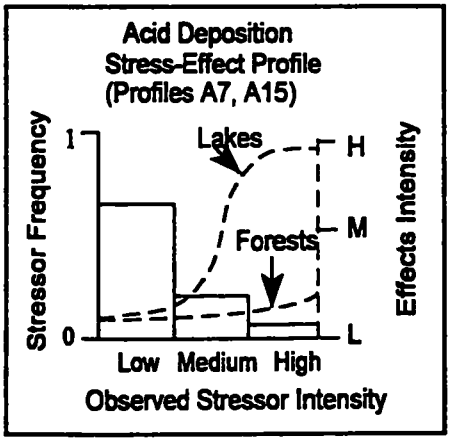


Figure 2-5c. Acid Deposition Stress-Effect Profile

structure, composition, and/or function of the system; an example is a major physical disturbance that removes the major structural components of an ecological system. Medium-level ecological effects were considered to be significant changes in the ecosystem's structure, composition, and/or function; for example, the replacement of one or more native species in the community by introduced exotic species would constitute a medium-level effect. Low-level ecological changes were those that, while detectable (i.e., non-zero), do not entail significant changes to either the structure, composition, or function of the system; an example would be the shift in the salinity isopleth near the freshwater outflow from a canal. The frequency of the no (or *de minimus*) level of ecological effects was not assessed by ERS; this category applies to any ecological changes that would be non-detectable. For example, in the stress-effects profile for acid deposition (Fig. 2-5c), the largest fraction of affected lakes are judged to have low-level ecological consequences (i.e., detectable but not highly significant changes), but there may be a larger set of lakes that do not experience any detectable effects at all and therefore effects would be judged to be insignificant for the ecosystem.

As with the stressor regime, the effects profile can be modified if and when additional data are acquired. Also, the use of multiple effects profiles, such as in Fig. 2-4c, allows the ready identification of those ecological effects that might differ across different ecosystem types, thereby highlighting differential ecological sensitivities and risks. In principle, such a distinction could also be made within a particular ecosystem type, showing differential effects for various ecological endpoints for that ecosystem. This approach would be particularly useful for localized comparative risk assessments.

It should be noted that there are insufficient data on many types of ecological responses to stressors, and much of the available information excludes major constituents of ecological systems (e.g., microbes and amphibians). The Subcommittee recommends using the best available information for characterizing ecological effects from stressors and, if there are quite different responses among different components of ecological systems, to focus on the more sensitive or more ecologically important components. Further, as new information is acquired on stress-response relationships, the database upon which the ecological risk profiles and resulting relative risk rankings are based should be updated and new rankings done periodically. Note that different stress-effects profiles might be appropriate for

conducting this risk ranking exercise at regional or local scales. That is, at a regional scale the frequency distribution of the stressor (stress profile) may well differ from the distribution at the national scale. On the other hand, since the effects profile is based on the responses of a particular ecosystem type to the stressor, effects profiles are not different for risk rankings at different scales. This is an important distinction, as it means that an effects profile database can continually be updated as more information becomes available, and that same database could be used for different risk ranking exercises, but the stressor distribution database must be developed for each specific area of concern, e.g., a particular region or the entire nation.

2.3.3 Development of Relative Ranking of Ecological Risks

Although the stress-effects profiles described in the previous section provide an initial, visual characterization of the ecological risks for vulnerable ecological systems, the ERS concluded that in order to characterize relative ecological risks, it is necessary to convert each profile into a quantitative score. The Subcommittee developed such a methodology that goes well beyond the relative risk ranking approaches of *Unfinished Business* and *Reducing Risk*. The ERS methodology reflects the proportion of the ecological resource at-risk to a particular stressor and adjusts the risk level assigned to each stressor in the context of the specific scale of concern (e.g., regional, or national). In order to produce a national ranking of relative ecological risks, the ERS developed several specific risk modification factors that should be considered in making this risk adjustment. These multiplicative factors, and the rationale for selecting their specific numerical values, are discussed below and summarized in Table 2-2. For ecological risk rankings done at other than the national scale, different factors, or perhaps different numerical values for the same factors, might be needed.

2.3.3.1 Multiplicative Factors Used to Assign Stressor Risk Values

The multiplicative factors proposed for the national-scale ecological risk ranking were developed by expert judgment, expressed through consensus of the Subcommittee. The intent was to establish the weights to apply to various scale-specific considerations that would lead to a higher or lower assignment of a risk score and would more accurately, and more transparently, represent the ecological risk of each stressor vis á vis the other stressors. In essence, each multiplicative factor

addresses the question of what is there about this particular stressor that makes it more or less an ecological risk compared to another stressor.

At the outset, the ERS assigned a numerical value to each of the three levels of ecological effects as defined above: High = 4, Medium = 2, and Low = 1. The calculated risk score for a stressor is quantified by applying the appropriate multiplicative factors to each of three possible effects intensities (H, M, L), then summing the results, as discussed in more detail below. Since the initial effects intensity scores were based on a factor-of-two basis (i.e., high = 2x medium = 4x low), the multiplicative factors were scaled so that a factor that warranted a jump in the major risk category (e.g., from medium to high) would be assigned a multiplicative value of 2. For example, the proportion of the resource experiencing a high level of stress from harvesting of coastal fisheries populations was considered so extensive that a jump of the risk value one full category was assigned (Table 2-3), whereas the proportion of ecological systems experiencing a medium level of effects from UV-B exposure was considered so low that a factor of 0.5 was assigned, resulting in one full drop in risk category.

Less significant factors were scaled relative to this two-fold standard. For example, a multiplicative factor of 1.1 means that the risk value would be increased by a small increment, and a string of about 7 such increments would be needed among the various ranking factors to equal one full category jump (i.e., 1.17). The Subcommittee sought consistency across the factors, so that two factors assigned the same multiplicative value were considered to be relatively equivalent in terms of the risk ranking. Thus, a recovery potential requiring a centuries time scale for recovery once the stressor was removed (multiplicative factor 1.25) was considered equivalent to a medium level of species depletion in terms of ecological significance to the ranking (Table 2-3).

While these multiplicative factors represent the best judgment of the Subcommittee, other considerations might result in somewhat different assigned values. The opportunity to implement the integrated ecological-human health risk ranking framework through an expert opinion survey (discussed in Chapter 3) could provide a verification of the weights assigned by the Subcommittee for the national-scale ecological risk ranking. Again, the transparency of the ERS methodology

readily allows examination of the assumptions and the values assigned by the expert judgment process. If it were determined that the weighting factors needed adjustments, it would be a simple process to revise the multiplicative values and recalculate the risk rankings without having to redesign the relative risk ranking methodology.

Once the risk ranking weighting factors are established, their application to the stressor-effects profiles is straightforward: First, each level of ecological effect is considered separately to calculate partial risk scores. The first partial risk score for a stressor focuses on the high level of ecological effects (initial value of 4). The proportion of resources affected at the high intensity of effects (if any) is assessed, based on the stress-effects profiles (Fig. 2-5). This is followed by each of the other weighting factors described below, all multiplied together to develop a partial risk score reflecting the contribution to the total risk ranking score from the high intensity of the stressor. If the stress-effects profile indicates that no ecological systems experience the high level of ecological effects from that stressor, even when the stressor is at the high level of intensity (e.g., habitat fragmentation profile [C2] shown in Fig. 2-5a and the acid deposition-forest profile [A15] shown in Fig. 2-5c), then the multiplier of zero will apply for the factor "proportion of resources at risk." Consequently, the assigned partial risk contribution from the high intensity effect level is zero. This illustrates an important reason why the weighting factors are multiplicative rather than additive; i.e., if any factor is assigned a value of 0, then that partial risk score is 0.

Similarly, the second partial risk score is calculated for the medium level of ecological effects caused by the stressor (which begins with an initial assigned value of 2). The third partial risk score is based on the low level of ecological effects caused by the stressor (which begins with an initial assigned value of 1). In these cases, the partial risk scores are calculated by applying the appropriate multiplicative factors in the same manner as described for the high-level partial risk score.

Finally, the three partial risk scores (for the H, M, and L levels of effects) are summed to result in the final total risk ranking score for the specific stressor, calculated for the specific scale of concern (in our case, the national scale). The details of the partial and total risk ranking scores developed by ERS are shown in a matrix form in Table 2-4.

Table 2-3. Multiplicative Factors for Deriving National Risk Rankings

Factor (Assessed at H/M/L Effect Level)	Multiplier
1. Proportion of Resource At Risk	
>60 percent	2
30-60 percent	1.5
10-30 percent	1
1-10 percent	.5
<1 percent	.25
effect level (H/M/L) does not occur	0
2. Existence of Hot Spots	
Nationally distributed	1.5
Regional	1.25
Local or None	1
3. Recovery Potential	
Irreversible or >centunes	2
Centunes	1.25
Decades	1
< Decades	.75
4. Duration of Stress-Effects	
≥centunes	1.25
Decades	1.1
<Decades	1
5. Secondary Stress Induction	
Two or more secondary stressors	1.1
< Two secondary stressors	1
6. Species Depletion Potential	
High	1.5
Medium	1.25
Low or None	1
7. Special Ecological Significance	
High	1.25
Medium	1.1
Low or None	1

The following sections describe each multiplicative factor and its assigned weighting values for national-scale rankings.

a) **Proportion of Resource at Risk** - The first entry into the risk scoring process characterizes the proportion of the ecological resource that experiences effects at the specified exposure intensity. This information is the same as shown graphically in the stress-effects profile for each stressor. The question asked is what proportion of the ecological systems at risk in the region of concern for the ranking (e.g., the proportion of all lakes in the nation) is exposed to the stressor such that the high level of ecological effects is realized. If the assigned effects distribution does not show severe ecological effects at high intensities of the stressor, then this proportionality factor is assigned the value of zero. If high-level ecological effects do occur, then the proportion of the ecological systems of concern experiencing high ecological effects is estimated as one of the following categories: <1%, 1-10%, 10-30%, 30-60%, or >60%. Then the multiplicative factor (Table 2-3) is applied based on that proportion category. For example, if more than 60% of the at-risk ecological resources experience the high effect, then the factor would be 2x. On the other hand, if less than 1% experienced that level of effect, then the multiplicative factor assigned would be 0.25x. Similarly, the same question is asked for the proportion of the ecological resource that experiences medium-level effects from the stressor, and finally the proportion at the low-level of effects. For example, in the case of habitat fragmentation, shown in Fig. 2-5a, no ecological systems experience high-level ecological effects, even at high levels of fragmentation, so the partial risk score for the high-level effects is 0. The medium-level effect results from high levels of habitat fragmentation, which the figure illustrates occurs at more than 60% of the systems, resulting in the initial value for medium effects (2) being multiplied by 2x (for the >60% proportionality factor). Similarly, only low-level effects occurs at medium or at low levels of habitat fragmentation, and those conditions occur 10-30% of the systems, resulting in a multiplicative factor of 1x for the low effects partial risk score.

b) **Distribution of "hot spots"** - This multiplicative factor adjusts the risk score if the stressor is distributed in localized hot spots, with disproportionate ecological effects from what would be predicted from the stressor frequency distribution profile. This approach acknowledges that adverse ecological effects may be significant in highly localized areas and, therefore, warrant national concern even though they do not

affect a large proportion of the resource and thus would not show up in the profile (?). This factor is assigned at the highest level of effect for which ecological effects occur (i.e., at the high level if there are high effects, at the medium level if there are medium but no high effects, and at the low level if there are no medium effects). If no or very few hot spots occur, the multiplication factor is 1.0; if hot spots do occur within a few regions of the nation, the multiplication factor is 1.25; and if they occur in most regions of the nation, the factor is 1.5. For example, a 1.5 factor for hot spots was assigned to the stressor heavy metals, which occur at high concentrations in localized areas across the nation.

c) **Recovery potential** - This multiplicative factor relates to ecosystem resilience, i.e., the hypothetical, relative potential of the affected ecosystem to recover if the stressor were to be removed, even if such removal does not appear realistic given physical, societal, or economic constraints. The factor reflects the fact that effects that are irreversible or very long-lasting (e.g., over geological time) are more significant ecologically than those effects that can be reversed quickly. For example, the recovery factor for nutrient effects on estuaries was assigned a value of 0.75 since the ERS estimated that once the nutrient inputs were removed, the system would recover within a decade.

d) **Duration of the stress-effect** - This multiplicative factor takes into account the time into the future that the stress could be expected to occur, so that an adverse effect that is expected to last a long time would be given a higher risk assignment than one that is of short duration. For example, hydrologic modification was assigned the duration factor of 1.25, in part because major changes in streambeds, such as from dams, are expected to persist for centuries.

e) **Secondary stress induction** - This multiplicative factor relates to the number and strength of interactions between the selected stressor and other stressors on the list, i.e., if the stressor induces or predisposes the occurrence of secondary stressors. The purpose of this multiplicative factor is to note those stressors that create cascading effects through the induction of other stressors. If two or more significant interactions exist, the score is multiplied by a factor of 1.1; otherwise, the factor is 1.0. For

example, hydrologic alteration would be expected to cause increases in the nutrient, sedimentation, salinity, and other stressors, so its secondary stress induction factor is 1.1.

f) **Species depletion** - This multiplicative factor relates to the potential for the stressor to result in depletion of species, from local loss of a population (extirpation) up to global extinction of a species. The species depletion factor is included because of the importance of the loss of species and the reduction in biodiversity that has occurred during the past century, significantly affecting the structure, composition, and function of many ecological systems. It also has particular importance to society in that some species are of special concern (e.g., threatened or endangered species). The Subcommittee defined high, medium, and low-level species depletion effects as follows:

> High-level effects: extinction or extirpation of many species, resulting in loss of species diversity or richness at one or more ecological systems; an example of high-level species depletion occurs for high intensities of habitat conversion, such as when grasslands are converted to agricultural systems.
>
> Medium-level effects: extinction or extirpation of a number of ecologically and/or societally important species, but not necessarily resulting in overall decreased species diversity; an example of medium-level species depletion is when pesticides occurring at high intensities cause the loss of many species of insects or other invertebrates in an ecological system adjacent to an agricultural field.
>
> Low-level effects: extinction or extirpation of one to a few species; an example is the loss of a single species because of a pest or disease outbreak.

g) **Special ecological significance** - This multiplicative factor was developed to capture any special significance of the ecological effects from the stressor on the affected ecosystems and/or ecological endpoints. One such modification captures the exceptional importance of certain ecological attributes, i.e., ecosystems or effects that are disproportionate in importance relative to their spatial extent or frequency of occurrence. For example, effects that are particularly consequential to coastal wetlands as a group, or a stressor that could eliminate an entire class of ecological system, such as vernal ponds, or that has broad-based effects on a group of organisms (e.g., all

reptiles) would trigger the special ecological significance multiplier. This multiplier was reserved for ecological risk considerations not captured by the other factors and was only infrequently invoked.

2.3.3.2 Sample Calculation for the Stressor Pesticides

As discussed above, the quantitative step in the ecological risk ranking methodology is to apply the multiplicative adjustment factors to the initial scores for each of the high, medium, and low levels of effects occurrence. To demonstrate this process, consider the calculation of the total risk ranking score for the stressor pesticides (see Tables 2-3 and 2-4). For the high-level effects partial risk score, the Subcommittee concluded that the high intensity of ecological effects (initial value 4) occur only in 1-10% of ecosystems (thus, the multiplicative factor is 0.5); hot spots are distributed nationally (multiplicative factor is 1.5); recovery would be expected to occur in decades (multiplicative factor is 1.0); the duration of stressor-effects would likely be decadal (multiplicative factor is 1.1); there are no secondary induced stressors (multiplicative factor is 1.0); species depletion is highly significant (multiplicative factor is 1.25); and there are disproportionate effects on higher trophic-levels and critical species (multiplicative factor is1.1). Thus, the application of multiplicative factors results in a modified partial risk score of 4.54 for the high-intensity effect. Similarly, calculations were done for the medium-intensity partial risk score (product = 1.10) and for the low-intensity partial risk score (product = 1.10). These three partial risk calculations were then summed to produce a total risk ranking score of 6.74. To complete the risk ranking process, this total risk score for pesticides is compared to the total risk scores calculated for all the other stressors of concern.

2.3.4 Sources of Uncertainty

There are many sources of uncertainty inherent in relative ranking of risks. Some causes of uncertainty are reducible through further research and better scientific information, such as having an improved understanding of stress-effect relationships for a specific stressor affecting a specific ecological endpoint. Other sources of uncertainty are essentially irreducible, such as those caused by the intrinsic complexities and variability of natural ecosystems. Uncertainty is only important here, however, if it significantly changes the relative assignment of risk estimates across the stressors.

Examples of the sources of uncertainty in the ERS national ecological risk ranking process include:

a) aggregation of stressors for the analysis into a class of stressor, i.e., grouping of different specific stressors with different exposure or effects characteristics - an example is the category "persistent toxic organics";

b) aggregation across specific ecological system types, i.e., inclusion of different specific ecosystems with different effects characteristics;

c) aggregation across specific ecological endpoints;

d) lack of information on the exposure regime;

e) lack of information on ecological effects;

f) interactions among multiple stressors; and

g) composition and expertise of the members of the ranking panel.

Clearly, the ERS process of developing a national-scale ranking was limited by the availability of information and time for the exercise, factors that could be mitigated by a concerted effort within EPA or another institution to acquire and analyze the full set of available information on each stressor. A major advantage of a comparative risk ranking process is that many of the associated uncertainties do not make a difference in the ranking, whereas a process of doing an assignment of absolute risk values would be much more subject to uncertainties and much less reproducible. Moreover, when the risk scores are converted to qualitative characterizations of risk, as opposed to numerical risk values, the importance of uncertainties vis á vis the overall ranking is even further reduced. In practice, the Subcommittee found that the quantitative ecological risk scores for each stressor tended to clump into distinct ranges, allowing separation of the stressors into qualitative categories of risk. In most cases, uncertainties in specific multiplication factors applied to a stressor did not cause a reassignment of the stressor from one major, qualitative category to another. This significantly enhances the confidence that can be placed on the relative rankings. As

noted in the next section, however, in some cases uncertainties may be so great that risk associated with a particular stressor (e.g., endocrine disrupting chemicals and genetically engineered organisms) cannot be ranked.

2.4 National-Scale Ecological Risk Ranking

2.4.1 Results of the ERS National-Scale Ecological Risk Ranking

In order to apply the ecological risk ranking methodology to the national scale of relevance, the ERS developed ecological stressor-effect profiles (Appendix 2A) for 32 of the stressors listed in Table 2-1 and scored each stressor using the partial and total ecological risk scores procedures described above. The results of the ERS national-scale risk ranking process are summarized in Table 2-4. The table shows for each stressor the primary ecosystem type considered by ERS to be at risk. The stressor-effect profile curves are next listed to indicate the stressor and effects distributions that the Subcommittee considered was appropriate for that stressor at the national scale. Then, for each effect intensity level, the partial risk scores are calculated by taking the intensity value and multiplying by the number assigned at each cell along the row, resulting in the partial scores at the right of the table. These three partial scores are added and shown in the "sum score" column. The next entry in the table is the ERS's judgment as to whether the trend for risk from the stressor is increasing, decreasing, or remaining the same nationally; for example, the risks from metals were judged to be declining, but the risks from climate change were judged to be increasing. Additionally, ERS identified the potential for surprises in its assignment of risk levels; for example, ERS judged there to be little likelihood of surprises for nutrient additions to the environment, but a high potential for surprises from introduced exotic species. Note that the latter two columns (trends and surprises) were added to characterize further the Subcommittee's judgment of the stressor, but were not incorporated into the relative risk rankings per se. Finally, it should be repeated that: a) ERS assigned the various values based on the assumption that current environmental laws and regulations continue to be enforced; b) these relative rankings apply only at the national scale and might be quite different for regional or local scales; c) only ecosystem- and landscape-level endpoints were considered; d) these rankings were based on the understanding of the ERS members and did not entail analysis of extensive data or models; and e) limitations in the available data may in some cases

Table 2-4: Summary of Ecological Risk Ranking Scores

stressor	ecosystem type at risk	stress-effects curve	effects intensity	proportion of resource	hot spots	recovery duration	secondary stress induction	species depletion	eco signif	Partial SCORE	SUM SCORE	trends	potential for surprises
NOx	terrestrial	A15	1	0.25	1	0.75	1	1	1	0 19	0.19	↓	L
			2	0						0 00			
			4	0						0 00			
tropospheric ozone	forests, shrubs and grasslands	A10	1	0 5	1	0 75	1 1	1	1	0 41	1.65	=	L
			2	0.25	1	0 75	1.1	1	1	0.41			
			4	0.25	1	0 75	1 1	1	1	0 83			
tropospheric ozone	agric	A7	1	1	1	0.75	1	1	1	0.75	2.40	=	L
			2	0.5	1	0.75	1.1	1	1	0.83			
			4	0 25	1	0.75	1.1	1	1	0.83			
SO2	terrestrial	A12	1	0.25	1	1	1.1	1	1	0 28	0.28	↓	L
			2	0						0 00			
			4	0						0 00			
Hg	freshwater and esturaries	A12	1	0.25	1.25	1	1 1	1	1	0 34	0.34	?	L/H?
			2	0						0 00			
			4	0						0 00			
other heavy metals	freshwater and esturaries	A13	1	0 25	1	1	1.1	1	1	0 28	2.48	↓	H
			2	0 25	1	1	1 1	1	1	0 55			
			4	0.25	1.5	1	1.1	1	1	1 65			
toxic inorganics (As, Se, B)	freshwater, esturaries and agriculture	A10	1	0.25	1	1	1 1	1	1	0 28	2.34	=	H
			2	0 25	1	1	1.1	1	1	0.55			
			4	0.25	1 25	1	1 1	1	1 1	1.51			
persistent toxic organics	aquatic and terrestrial	A10	1	0 5	1	1	1.1	1	1	0.55	2.75	↓	H
			2	0.25	1	1	1.1	1	1	0.55			
			4	0.25	1.5	1	1.1	1	1	1 65			
pesticides	aquatic and terrestrial	A7	1	1	1	1	1.1	1	1	1 10	6.74	↑	H
			2	0 5	1	1	1 1	1	1	1.10			
			4	0.5	1.5	1	1.1	1	1.25	1.1	4.54		

Table 2-4 (continued)

stressor	ecosystem type at risk	stress-effects curve	effects intensity	proportion of resource	hot spots	recovery duration	secondary stress induction	species depletion	eco signif	Partial SCORE	SUM SCORE	trends	potential for surprises
acid deposition	lakes	A7	1	1	1	1	1	1	1	1.00	3.31	↓	L
			2	0.5	1	1	1.1	1	1	1.10			
			4	0.25	1	1	1.1	1 1	1	1	1.21		
acid deposition	forest	A15	1	0.25	1	1 25	1.1	1 1	1	1	0 38	0.38	↓	L
			2	0							0.00		
			4	0							0 00		
nutrients	freshwater and esturaries	B19	1	1.5	1	0 75	1.1	1	1	1	1.24	4.25	=	L
			2	1	1	0 75	1 1	1	1	1	1.65			
			4	0.25	1 5	0 75	1.1	1 1	1	1	1.36			
radionuclides	aquatic and terrestrial	A3	1	0.25	1	1	1	1	1	1	0 25	0.25	=	L
			2	0							0 00			
			4	0							0.00			
oil spills	freshwater, esturaries and tundra	A11	1	0.25	1	1	1	1	1	1	0 25	0.88	=	L
			2	0.25	1	1	1	1	1	1 25	0.63			
			4	0							0.00			
DO/BOD	freshwater and esturaries	B19	1	0.5	1	0 75	1	1	1	1	0.38	2.49	↓	L
			2	0 5	1	0 75	1	1	1	1	0.75			
			4	0.25	1.5	0 75	1.1	1.1	1	1	1 36			
acid mine drainage	freshwater	A7	1	0.25	1	0 75	1 1	1	1	1	0.21	1.65	↓	L
			2	0.25	1	0.75	1 1	1	1	1	0 41			
			4	0.25	1 25	0 75	1 1	1	1	1	1 03			
contaminated groundwater	freshwater and esturaries	A12	1	0 25	1	0 75	1.25	1	1	1	0 23	0.23	?	L
			2	0							0.00			
			4	0							0 00			
disease/pest outbreaks	aquatic and terrestrial	A7	1	0.5	1	1	1	1	1	1	0 50	4.80	=	H
			2	0.5	1	1	1	1	1	1	1 00			
			4	0.5	1.5	1	1	1 1	1	1	3.30			

Table 2-4 (continued)

stressor	ecosystem type at risk	stress-effects curve	effects intensity	proportion of resource	hot spots	recovery duration	secondary stress induction	species depletion	eco signif	Partial SCORE	SUM SCORE	trends	potential for surprises
introduced exotic species	aquatic and terrestrial	B1	1	1	1	1	1	1	1	1.00	13.94	↑	H
			2	1 5	1	1	1 25	1	1.25	1	4.69		
			4	1	1	1	1.25	1.1	1.5	1	8 25		
climate change	aquatic and terrestrial	C7	1	1	1	1	1.25	1	1	1	1.25	14.31	↑	H
			2	1	1	1	1 25	1.1	1	1	2.75		
			4	1.5	1	1	1.25	1.1	1.25	1	10 31		
noise	aquatic and terrestrial (fauna only)	A12	1	0 25	1	1	1	1	1	1	0.25	0.25	↑	L
			2	0							0 00		
			4	0							0 00		
light	aquatic and terrestrial (fauna only)	A12	1	0 25	1	1	1	1	1	1	0 25	0.25	↑	H
			2	0							0 00		
			4	0							0 00		
hydrologic alteration	streams, wetlands, and estuaries	D7	1	1 5	1	0 75	1.25	1	1	1	1 41	20.14	=	L
			2	0.5	1	1	1.25	1.1	1 25	1	1.72			
			4	1.5	1	1.25	1.25	1.1	1.5	1.1	17.02			
habitat fragmentation	aquatic and terrestrial	C2	1	1	1	1	1.25	1	1	1	1 25	8.13	↑	L
			2	2	1	1	1 25	1.1	1.25	1	6 88			
			4	0							0 00		
habitat conversion	aquatic and terrestrial	C7	1	0							0 00	15.81	↑	L
			2	1	1	1	1.25	1.1	1.25	1	3 44			
			4	1.5	1	1	1.25	1.1	1 5	1	12.38			
physical habitat disruption	aquatic and terrrestrial	A1	1	1	1	1	1.1	1	1	1	1 10	4.13	↑	L
			2	0 5	1	1	1.1	1.1	1	1	1.21			
			4	0 25	1 5	1	1 1	1.1	1	1	1 82			
turbidity sedimentation	aquatic and wetland	B1	1	1	1	0.75	1.1	1	1	1	0 83	8.39	=	L
			2	1.5	1	0.75	1 1	1 1	1	1	2 72			
			4	1	1	1	1.1	1.1	1	1	4 84			

Table 2-4 (continued)

stressor	ecosystem type at risk	stress-effects curve	effects intensity	proportion of resource	hot spots	recovery duration	secondary stress induction	species depletion	eco signif	Partial SCORE	SUM SCORE	trends	potential for surprises
altered fire regime	terrestrial and wetlands	A13	1	0.5	1	1	1.1	1	1	1	0 55	2.97	↓	L
			2	0.5	1	1	1.1	1.1	1	1	1.21			
			4	0.25	1	1	1 1	1.1	1	1	1 21			
altered salinity regime	freshwater, esturaries and wetlands	A7	1	1	1	1	1.1	1	1	1	1.10	2.98	↑	L
			2	0 25	1	1	1.1	1	1	1	0 55			
			4	0.25	1	1	1.1	1.1	1	1.1	1 33			
harvesting	freshwater populations at-risk	B8	1	1	1	0.75	1 1	1	1	1	0 83	2.48	↑	L
			2	1	1	0.75	1.1	1	1	1	1.65			
			4	0							0 00			
UV-B	aquatic and terrestrial including ocean	A2	1	1	1	0.75	1 25	1	1	1	0.94	1.88	↑	H
			2	0.5	1	0.75	1 25	1	1	1	0.94			
			4	0							0.00			

influence the risk rankings; for example, some chemicals may rank low for ecological risks because there are few data on ecological effects other than for the aquatic and pesticide endpoints required by law. Nevertheless, the Subcommittee is reasonably confident that the overall ecological risk rankings accurately reflect the relative risks to ecological systems in the U.S. from anthropogenic stressors.

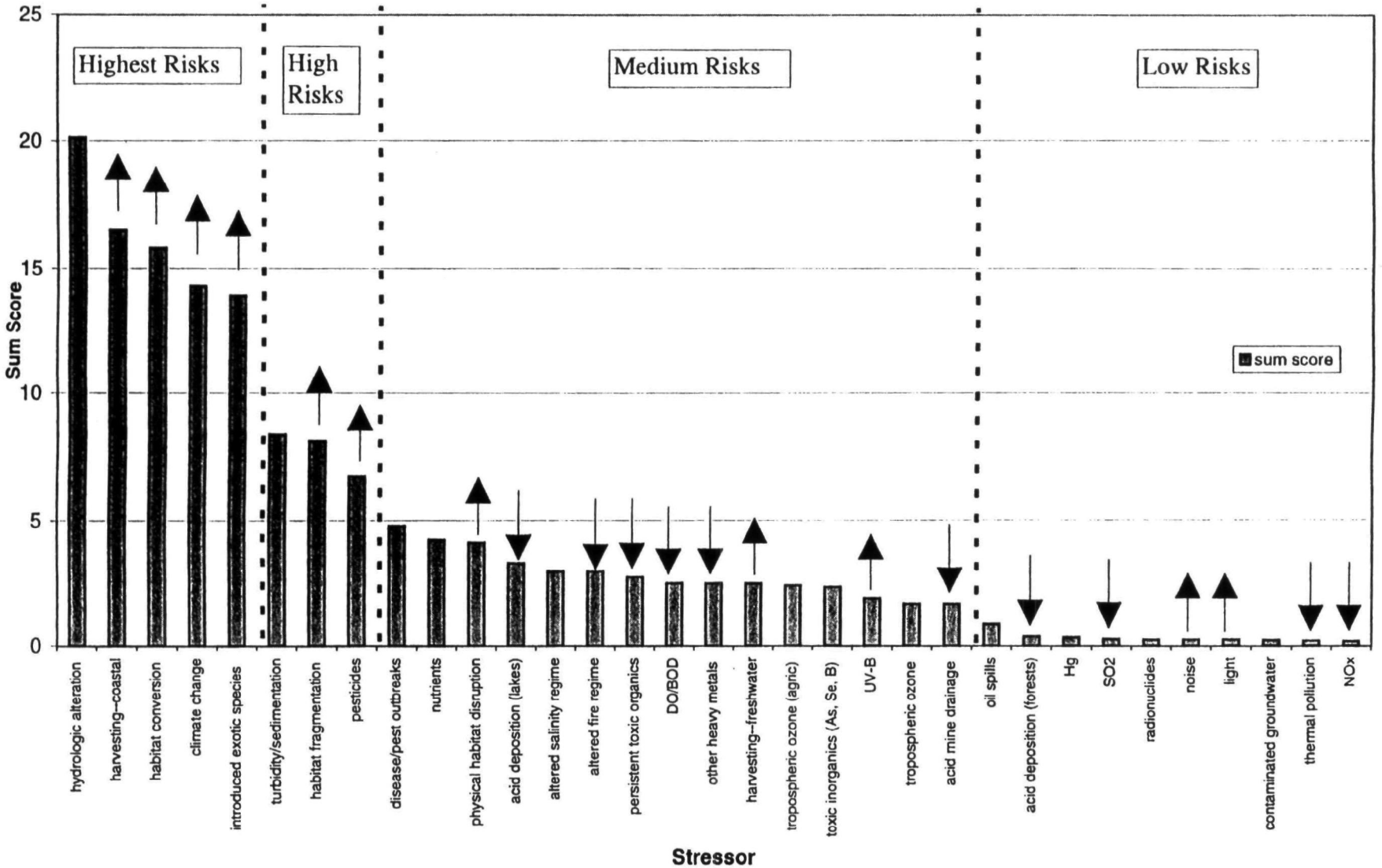
2.4.2 Synthesis and Conclusions

The results from the relative ranking of national ecological risks derived by the ERS using the stressor-based methodology are shown in Table 2-5 and Figure 2-7. The process for assigning the stressors into qualitative risk categories was to rank order the stressor total risk scores and identify, where possible, distinct breaks between groups of stressors. Inspection of Figure 2-7 shows the clear discontinuities that were used to group the stressors into the four qualitative ecological risk categories (highest risks, high risks, medium risks, and low risks). Note that the division between medium and low risks was less distinct and, therefore, somewhat more arbitrary than the distinction between the highest and high categories, and the distinction between the high and medium categories. Other aggregations or dividing lines could be assigned to these numerical rankings, but the important consideration is that there is a clear distinction across the stressors, so

Table 2-5: Relative Ranking of Ecological Risks at the National Scale

HIGHEST ECOLOGICAL RISKS
hydrologic alterations
harvesting marine living resources
habitat conversion
climate change
introduction of exotic species
HIGH ECOLOGICAL RISKS
turbidity/sedimentation
habitat fragmentation
pesticides
MEDIUM ECOLOGICAL RISKS
disease/pest outbreaks
nutrient additions
physical habitat disruption
acid deposition (lakes)
altered fire regime
altered salinity regime
persistent toxic organics
heavy metals other than Hg
DO/BOD
harvesting freshwater living resources
tropospheric ozone
toxic inorganics
UV-B
acid mine drainage
LOW ECOLOGICAL RISKS
oil spills
acid deposition (forests)
Hg
SO_2
radionuclides
noise pollution
light pollution
groundwater contamination
thermal pollution
NO_x
UNKNOWN BUT POTENTIALLY IMPORTANT RISKS
endocrine disruptors
genetically engineered organisms

Figure 2-7. Ecological Risk Ranking Scores



that, for example, the assessment of the ecological risks from hydrologic alteration was clearly quite different from the assessment of ecological risks from oil spills. The Subcommittee believes that its assignments of relative risks into the qualitative categories in Figure 2-7 is a reasonable interpretation of the data.

Further, the Subcommittee believes this categorization is relatively robust, as demonstrated by an exercise ERS conducted of substitution of alternate plausible values for weighting factors assigned to specific stressors, i.e., a type of sensitivity analysis. In all cases the Subcommittee examined, to shift significantly the position of a particular stressor relative to the other stressors would require changes in the multiplicative factors that do not seem warranted based on the information available. Obviously, as new or more complete data become available, or as the exercise is applied to specific regions or locales, then there may be shifts in the relative risk rankings.

Consistent and at least semi-quantitative relationships are maintained in these final, total risk categorizations: The initial entry value of 4.0 that was assigned to high-level ecosystem effects has now been modified so that all total risk scores above 5.0 are assigned to the high risk category, and all total risk scores above 10 are in the highest ecological risk category. Similarly, from the initial value of 2.0 for medium-level effects, the total risk scores in the range of 1.0 to 5.0 now fall into the medium ecological risk category. And anything less than a total ecological risk score of 1.0 was considered to be in the low ecological risk category.

At the national scale, the highest level of ecological risk was assigned by ERS to two classes of habitat alteration (hydrologic modification and habitat conversion), as well as to climate change, introduced exotic species, and over-exploitation of living marine resources through harvesting. Grouped at the high level of ecological risk are turbidity/sedimentation, habitat fragmentation, and pesticides. The medium ecological risk category includes disease and pest outbreaks, nutrients, physical habitat disruption, acid deposition (lakes), and altered fire or salinity regimes. Note that two stressor categories (endocrine disrupters and genetically engineered organisms) did not have calculated risk scores and were not assigned to a narrative risk category because the Subcommittee believed the potential for ecological risks may be very high but is currently largely unknown. The Subcommittee, however, felt that it was important to

distinguish between unknown, but potentially important, risks and low risks. Further research and experience with these stressors may lead at some point in the future to a confident assignment of these to a risk category.

The results shown in Table 2-5 indicate that the greatest ecological risks in the U.S. relate predominantly to physical and biological stressors, not to chemical stressors. That conclusion is consistent with the relative ecological risk ranking of the *Unfinished Business* and *Reducing Risk* projects (EPA 1990a, 1990b; Harwell et al., 1992).

It should be recalled that the risk ranking process done by ERS focused on the national scale of concern and explicitly considers that current environmental laws and regulations will continue to be enforced. At less than national scales, the ecological risk rankings might be different; for example, at a local level the most important risk might be the acid drainage from a mine entering a valuable freshwater stream. The national scale of ranking is meant to look more broadly, and such local-scale risks only become nationally important if captured by the criteria for scaling-up; that is, acid mine drainage would have had a higher national-scale ranking if it occurred at many locations all across the nation.

Another noteworthy conclusion is that many environmental problems that have a high public perception of risks and/or have a high allocation of resources for environmental protection and management may, in fact, constitute relatively lower ecological risk as nationally ranked. Included in this are ecological risks from radionuclides, oil spills, heavy metals, toxic organic and inorganic chemicals, and SO_2 and NO_x air pollution. Again, this conclusion is consistent with *Reducing Risk*, and comparisons of its ecological risk rankings with a poll of public perceptions of ecological risks (Harwell et al., 1992).

There are at least two possible reasons for this discrepancy: The first plausible reason is the success that the environmental regulatory system has had over the past several decades The ERS risk ranking was predicated on the assumption that existing environmental laws and regulations would remain in effect and continue to be enforced; the same risk ranking done in the 1960s might have given quite different results. In that case, risks that have been reduced may not yet be perceived as having been reduced,

illustrating a time-lag in the perception of environmental successes. Second, there is an anthropocentric focus of concern about environmental risk agents, i.e., historically there has been primary attention to human health effects rather than to ecological effects from environmental stressors, so the public has much better understanding of those risks. Both of these discrepancies can be mitigated by increased education of the public on the scientific bases of risk ranking.

The results of the SAB ERS relative ecological risk rankings have considerable implications for the ability of EPA and other regulatory and resource management agencies to reduce risks to ecological systems. If society desires to restore, protect, and sustain the environment with high ecological quality, then the remaining highest priority risks to ecological resources must be a focus of attention. At present, the highest ranked risks are not well regulated by EPA. In addition, examination of the risk rankings in conjunction with expected trends in the associated stressors (Fig. 2-7) highlights the fact that, in the Subcommittee's assessment, many of the top-ranked ecological risks are the very ones that are predicted to experience an upward trend.

2.5 An Effects-Backwards Methodology for Risk Rankings

In the ecological risk ranking procedure described above, the major ecological effects caused by a stressor or group of stressors were analyzed. The logical progression began with the stressor, then proceeded to rank the resulting ecological effects. It is often useful, however, to apportion the major, multiple causes of a particular adverse ecological effect, i.e., to work backwards from an observed effect to estimate the relative contribution to that ecological effect from each of several stressors. In this section, we briefly present the outline of a methodology for beginning with an effect and determining the relative contribution of various stressors to the effect at a national, regional, local, or other scale of interest.

The first step in an effects-backwards relative risk assessment is to identify and characterize the ecological effect, including the spatial and temporal scale of interest and the resources or habitat types affected. For example, the causes of species depletion, defined as loss of native species within the exposed ecosystems for all types of ecosystems nationwide, might be analyzed. A smaller-scale example could be a reduction in forest ecosystem productivity for a particular watershed, which might be

caused by more than one stressor.

The second step in the procedure would be to identify all of the major stressors for which a cause-and-effect linkage could be established or reasonably hypothesized. For this purpose, the stressor list presented in Table 2-1 could be used as a starting point. Depending upon the particular ecological effect being analyzed, groups of stressors could be aggregated to simplify the analysis. Logical aggregations might consist of stressors likely to occur in the same locations or groups of similar classes (such as chemicals) whose independent effects may be small but whose cumulative effects may be significant.

Once the stressor list is developed, a stress-effect profile, similar to those described earlier, could be used to establish the relative strength of each stressor-effect relationship. If desired, a screening-level analysis could be performed by considering only those stressors that exhibit a strong relationship to the effect. Note that in most cases, the stress-effect profiles generated in this context will not be the same as those developed for the stressor-based analysis because in the latter case the effects profile represents an aggregation of multiple ecosystem (or landscape-level) effects, rather than the occurrence of a particular effect. This is because the adverse effects on structure, composition, and function of one or more ecosystem types or landscapes were lumped to streamline the national analysis.

In order to complete the effects-backwards stress-effect profile, it is important explicitly to define high, medium, and low-level effects. In many cases, this step will not be trivial because it entails quantifying or otherwise describing the total effect in order to normalize the y-axis and thereby derive the high, medium, and low effect ranges. In the species depletion example, therefore, the number of species reported extirpated or at significant risk of extirpation within the next decade in all types of terrestrial and aquatic systems could be estimated to provide the 100% line in the normalization. High, medium, and low levels of effect could then be defined accordingly. Alternatively, a rough measure of species at-risk or already affected might be developed for a screening-level analysis; for example:

high-level effects:	the extinction or extirpation of a number of different species in different types of ecological systems

	located in many areas;
medium-level effects:	the extinction or extirpation of a few species, species in only a few ecological systems, or species only in a few localized areas; and
low-level effects:	extinction or extirpation of only a few species in a few ecological systems in selected locations.

In the forest ecosystem productivity example, data may be available to assess the decrease in forest productivity, defined here as the decrease in the amount of energy stored by the forest ecosystem, using measurements of tree diameter, height, volume, basal area, or mean annual increments of growth. High, medium, and low levels of effect would then be normalized to this total.

Using the information in this stress-effect profile, the next step in the effects-backwards relative risk methodology is to assign an ecosystem- or landscape-level partial score and modify that score with appropriate multiplicative factors as in the stressor-based methodology. Once a numerical score for each significant stressor is developed, the stressor-effect scores can be summed. The proportion of each stressor's partial score relative to the total score provides a measure of the relative importance of the stressor in producing the effect of concern. Note that the particular multiplicative factors may well be different from those presented in the national risk ranking methodology above, because of the characteristics of the particular effect or scale of interest. For example, for species depletion at the national scale, the spatial extent, hot spot, and ecological significance modification factors might remain the same. Other factors might need revision: for example, secondary stress induction might be redefined to refer only to an effect on a keystone species, and duration and species depletion potential would not be relevant. Additional factors, such as secondary effects induction, could be added.

While the Subcommittee did not further develop these considerations into a more detailed methodology nor apply an effects-backwards methodology for any environmental problem, we suggest the Agency consider instituting research to complete the development and application of an effects-backward methodology to

apportion causes to effects that are manifested in the environment.

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Appendix 2A: Ecological Risk Profiles

CHAPTER 3. HEALTH RISK RATING METHODOLOGY

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CHAPTER 3. HEALTH RISK RATING METHODOLOGY

3.1 Introduction

3.1.1 Background

Adverse health effects in the U.S. population arise from a wide variety of individual causes and their combinations. EPA is broadly mandated by Congress to address those causes that are environmentally-mediated, i.e., to reduce risks from environmental stressors or conditions that impair health.

As noted in Chapter 1, EPA initiated its first effort to assign relative risk ratings to environmental problems in 1986 to improve allocation of Agency resources. This effort was reported in *Unfinished Business* (EPA, 1987). EPA's Science Advisory Board (SAB) expanded upon this work in the report *Reducing Risk: Setting Priorities and Strategies for Environmental Protection* (SAB, 1990). For that report, the Environmental Health Effects Subcommittee conceptualized health risks as a multidimensional matrix whose axes comprised variables such as exposure sources, exposure routes, health endpoints, and stressors. It noted that the practical translation of this conceptual structure could be achieved by a relational database design allowing independent ratings of the different variables across various dimensions. At the end of the rating process, each of its elements could then be evaluated independently to achieve a global view of where the major problems resided and where resources could most profitably be directed. This perspective, in essence, has been adopted as a guide to the selection of risk reduction options, a topic discussed at length in Chapter 6.

3.1.2 Approach

To meet the requirements of the IRP, the Human Exposure and Health Subcommittee (HEHS) took as its objective the development of a methodology for relative rating of the human health risks from exposures to environmental stressors. The HEHS also viewed the matrix structure as a useful conceptual plan, but, as in 1990, was compelled to consider the pragmatic question of its translation into practice. It concluded that any method developed by the Subcommittee should offer, in so far as

possible, certain key attributes. Specifically, the method should be:

a) as simple as possible to understand, implement, and use;
b) capable of gathering judgments from many respondents quickly and inexpensively;
c) responsive to the technical knowledge of individual experts and able to capture some of the reasoning of the experts;
d) responsive to the concerns of other interested groups; and
e) suitable for analysis of correlations among various respondents.

The method also should explicitly utilize the best existing scientific information on environmental health risks and should be a scientifically based and transparent methodology.

Because evaluation of sources and source categories was to be a focus of the Risk Reduction Options Subcommittee and because coordination of ecological and human health risks was seen as one of the crucial aims of the current exercise, HEHS adopted the Ecological Risks Subcommittee's strategy of structuring its methodology around environmental stressors, i.e., chemical, biological, and physical agents released into the environment by or modified through human activities. Although many comparative risk projects have been structured around the ranking of problems, such as drinking water contamination or urban air pollution, stressors provide a more homogeneous basis for rating adverse health effects. Furthermore, reducing human environmental health risks eventually requires reductions of exposure to environmental agents or stressors, largely by controlling sources. The analysis of stressors can provide a starting point for the development of policy options.

3.1.3 Role and Significance of Expert Opinion in Risk Analysis

Expert opinion plays a special role in the architecture of risk assessment, especially when data are incomplete, contradictory, or multidimensional (Morgan and Henrion, 1990). One method for incorporating expert opinion relies upon a small group of respondents whose implicit or direct aim is to provide a consensus judgment. The Delphi method, described in many publications (Adler and Ziglio, 1996; Linstone and Turoff, 1975), exemplifies such an approach. In striving for consensus, such a group

exercise may deliberately suppress significant disagreement among members of the expert panel. The extent of disagreement, however, is a crucial piece of information. Presenting only a consensus position is equivalent, in some respects, to presenting experimental data only as arithmetic means unaccompanied by any measures of variability. As noted by Morgan and Keith (1995), "...when uncertainty is high because of fundamentally different views about underlying...processes, a consensus summary may not best serve policy analysis needs. An alternative approach, widely used in applied Bayesian decision analysis, formalizes and quantifies the judgment of individual experts through expert solicitation."

A high degree of variability in risk judgments among experts chosen for their specialized knowledge may indicate deficiencies in the data upon which judgments are based, or disagreements in how experts interpret the data. Moreover, an aspect of interpretation often ignored is the degree of confidence the judges place in their assigned ratings, a judgment vector, so to speak, that largely reflects the breadth and depth of the available data. It may also reflect the complexity of such data; a rich data set may, paradoxically, point in several directions simultaneously.

Any approach or methodology aimed at health risk assessment must deal simultaneously with both current questions and future projections. For most currently perceived problems, the primary impediments to establishing a risk ranking or rating reside in the nature of the available data. Although a sparse data base is an obvious source of uncertainty, it is by no means the sole one. For some agents or conditions, even extensive data does not guarantee a shrinkage of uncertainty. Different observers may interpret the same information from different points of view. A process that takes account of both variations in data availability and interpretation is required to conform to current standards of risk characterization such as those described in *Understanding Risk* (NRC, 1996).

Another feature that should be incorporated into a health risk rating methodology is geographic flexibility. Although the IRP has focused primarily on national concerns, particularly in its choice of stressors, the risk rating process should be able to accommodate state and local concerns as well. Without some uniformity in how ratings are achieved, consistency between levels of geographic integration will be difficult to achieve.

A third aspect of desirable methodological flexibility is the capacity to adapt to changes in currently perceived or mandated stressors, exposure standards, or public concerns. Barriers to flexibility include factors ranging from the inability to convene expert rating panels to the formidable difficulties of publishing and distributing new printed documents addressing a particular risk.

The rating process itself, beyond logistics, should also be able to define its limits. An assignment of low risk, for example, must be presented in context. For example, although experts might assign a low overall rating to a specific stressor, such a rating might have to be modified for a specific subpopulation, or when the available information is tentative.

3.2. The Environmental Health Risk Rating Methodology

3.2.1 Nature of the Methodology

The HEHS developed a highly simplified system for polling and characterizing expert judgements of the relative risks of a set of stressors. In the method developed by HEHS, each expert, or other respondent, is asked to rate the health risks of environmental stressors, from "very low" to "very high." Information is provided on the effects of, and population exposures to, each stressor, although the respondent is instructed to draw on his or her own knowledge and scientific judgment to interpret the information provided. The respondent is also asked to identify the determinants of his or her rating. Such determinants might include severity of effect, the size of the affected population, risks to subpopulations, or other factors.

The final results of the rating exercise include not only the average rating of each stressor, but also the distributions of these ratings, and information on the chief considerations upon which different respondents based their ratings. From these results, one can see not only the average rating for each stressor, but one can also identify those stressors for which there is substantial disagreement and variation in ratings.

To implement this approach, HEHS constructed a World Wide Web site designed to collect two types of information: a) ratings of health risks, from a list of

environmental stressors, provided by a sample of respondents not constrained to attain consensus, and b) ratings of the confidence these respondents place in their ratings. The Web site design aimed for the five fundamental properties listed in the previous section.

Another desirable property was the ability to identify the backgrounds of the respondents and to correlate them with their responses. Although this method was designed for the elicitation of expert judgements, an additional advantage of a web-based process is that it can also be used by stakeholders, members of a specific community or interest group, or members of the general public.

The Entry Page for the Web Survey (Appendix 3A) explains the purpose of the survey and presents the instructions. The respondent then begins the process by registering (Figure 3-1). The respondent enters his or her name, affiliation, and e-mail address. To help ensure the integrity of the process, passwords would be issued to restrict access. In addition, however, each respondent, in transmitting the ratings, also automatically provides the Internet address from which they are sent. The respondent then proceeds to the rating page, shown in Figure 3-2. The respondent chooses a stressor for rating by a mouse click on the stressor name. Table 3-1 gives the current stressor list.

Once a stressor is selected, the respondent enters his or her ratings of Risk Rating and Confidence, which range from Very High to Very Low, on the appropriate buttons. The rater can also, by selecting the Information button, access the appropriate Stressor Risk Characterization Data Sheet; risk data sheets on the web site (Appendix 3C) summarize key available scientific information on each of the stressors, e.g., information on exposure route, populations exposed, identified health effects, and so forth. Categories of information that should be presented in the risk data sheets are described further in Section 3.2.3. In rating the stressors, respondents can access the risk data sheet for each stressor and use all, some, or none of this information plus their own expert knowledge to arrive at a risk rating.

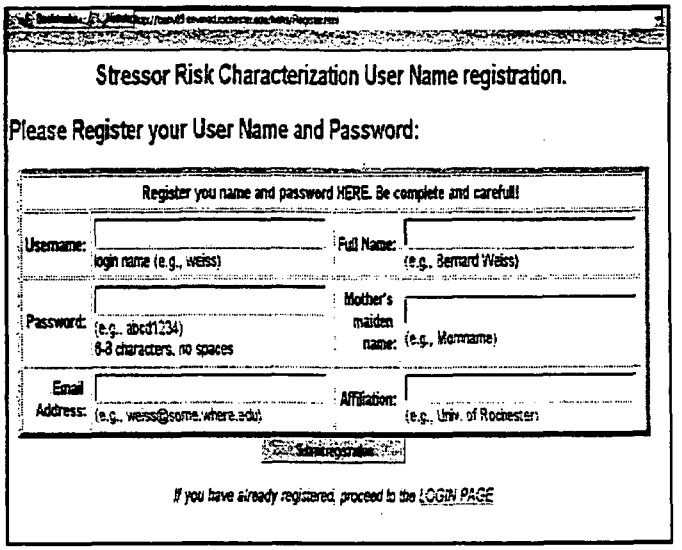

Figure 3-1. Registration form for the risk survey. Only valid registrants are permitted to submit ratings.

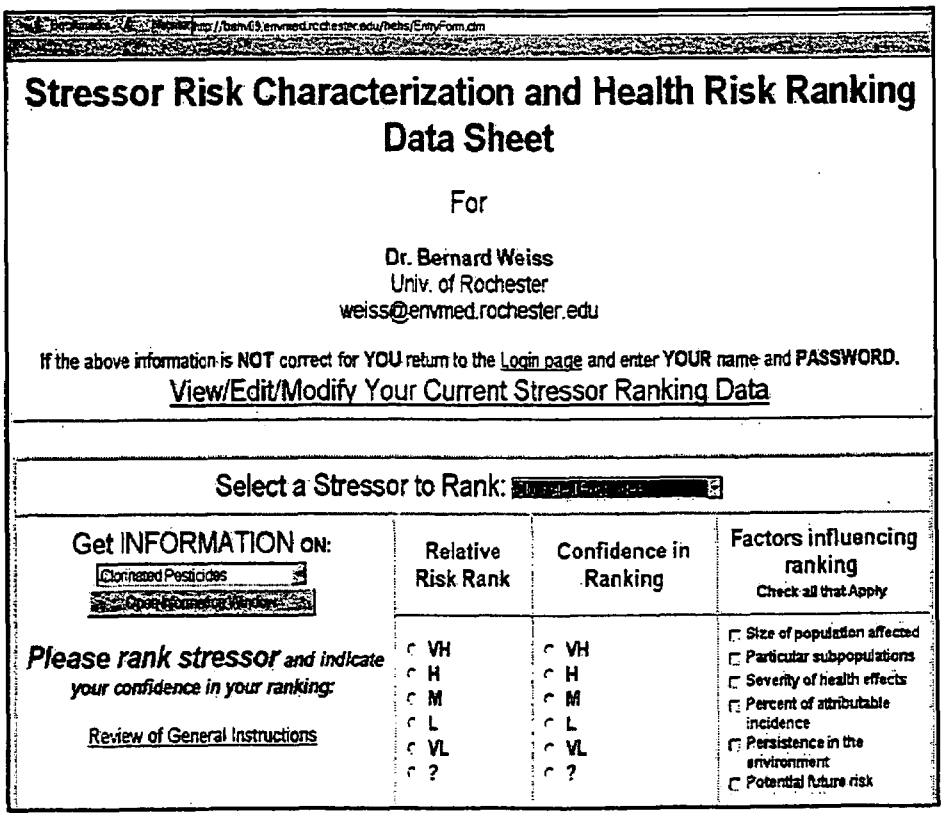

Figure 3-2. Risk rating form. The respondent selects a stressor, then accesses the data sheet (Appendix 2) by selecting the Information Window.

A Review of the General Instructions can also be selected with a click.

Lastly, the respondent checks the appropriate boxes to indicate the factors influencing his or her rating. One or more factors may be selected. A description of how these factors may be interpreted is also given in the Instructions (Appendix 3B). A respondent may also indicate other reasons for the rating and enter a specific comment. Provision for such entries appears at the bottom of the form and is shown in Figure 3-3.

The respondent then proceeds systematically through the list of stressors. The rater can review his or her ratings and accept or modify them, as shown in Figure 3-4. Finally, once all of the stressors have been rated and accepted by the respondent, the results are sent via the Internet for analysis with the ratings submitted by other respondents.

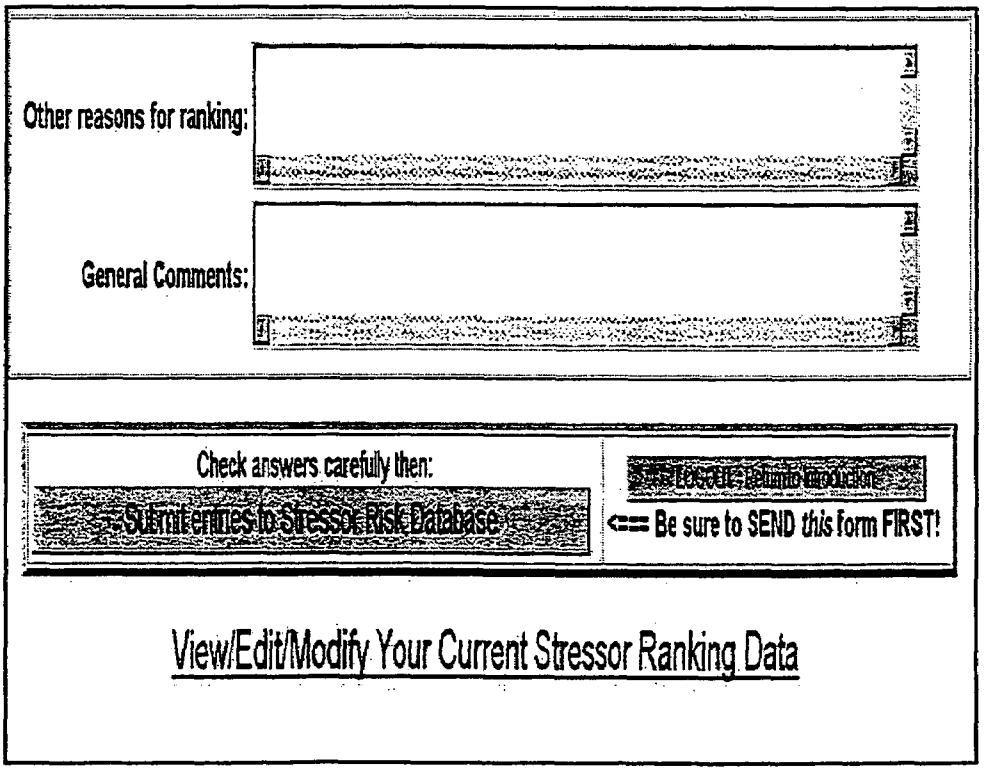

Figure 3-3. Fields for comments and other reasons for assigned ratings in the rating form (Figure 3-2) and buttons for submission to respondent's database.

U.S. EPA Headquarters Library
1200 Pennsylvania Avenue, NW
Washington, DC 20460

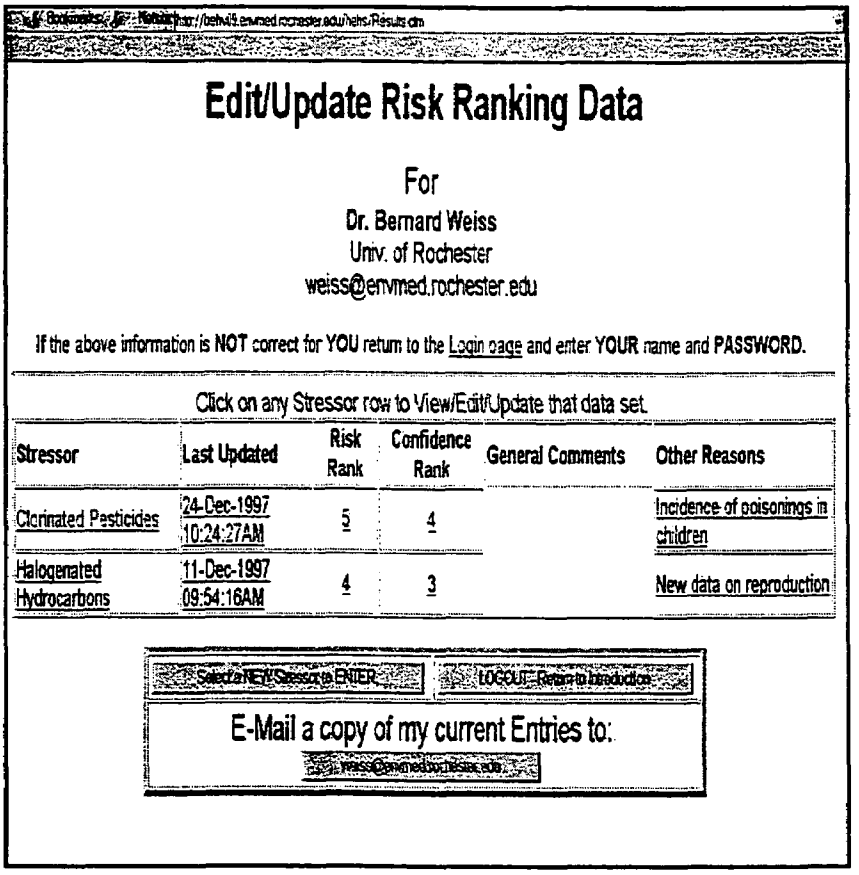

Figure 3-4. Selection summary and revision form. This feature allows the respondent to review and modify previous selections.

3.2.2 Scope of Stressor List

A multitude of physical, chemical, and biological stressors are associated with environmental health risks. For the organic compounds alone one can easily enumerate thousands of compounds distributed among air, water, soil, and food. Clearly, the Risk Rating Methodology could be overwhelmed by undertaking to evaluate too large a list of stressors, which would leave the process paralyzed at the stage of acquisition and development of information data sheets. Such a broad evaluation would come at the expense of a more thorough examination of those stressors offering an established reason for potential or actual concern. The proposed list of stressors to be rated might include the Criteria Pollutants (air, water), Hazardous Waste Priority

Chemicals, Persistent Organic Pollutants, contaminants regulated by emissions or effluent standards, radionuclides regulated by EPA and the Department of Energy, common indoor pollutants, pesticides, metals, indirect stressors (e.g., ozone depleting chemicals), and emerging issues (e.g., hormone-disrupting chemicals).

From a large list of stressors, EPA needs to dedicate the resources to reduce the stressors to be rated to a manageable number, probably no more than 100 at most. This might be done by aggregating some of the stressors on the basis of criteria such as commonality of toxic targets, endpoints, and sources. Since EPA resources will be limited, especially at the beginning, the Agency might wish to begin with a deliberately limited list of stressors, but one whose members intersect a number of groupings. For example, HEHS developed a short list of stressors with which to test and illustrate the methodology (Table 3-1).

This list, with categories that are not always mutually exclusive, is presented as an example and should be modified as needed based on review and analysis of broader lists of pollutants and other stressors noted above. The approach should be iterative, with stressors added and deleted over time based upon new information about exposures, health risks, and stakeholder concerns.

Table 3-1. List of Current Stressors Selected for Risk Ratings

Criteria Air Pollutants
Carbon Monoxide
Ozone
PM-10

Heavy Metals
Arsenic
Lead
Mercury

Persistent Organic Chemicals
Dioxins, furans, PCBs
Hormone disrupting chemicals
Pesticides, Herbicides

Environmental Tobacco Smoke

Polycyclic Aromatic Hydrocarbons

VOCs (Air Toxics)
Benzene
Tetrachloroethylene
Formaldehyde

Bioaerosols
House dust mites allergen and fungi

Physical Agents
Radon
EMF

Indirect Stressors
CO_2 and other greenhouse gases
Freons and other ozone layer depleting chemicals

3.2.3 Stressor Risk Characterization Data Sheets

The Stressor Risk Characterization Data Sheets provide key information useful in rating risks. This summary information should be based primarily on studies reported in the peer reviewed scientific literature and presented succinctly. Peer-reviewed comprehensive risk assessments developed by EPA, such as the those for the criteria air pollutants, or those developed for environmental tobacco smoke (EPA, 1992), radon (EPA, 1992), dioxins (EPA, 1995), can provide key conclusions and information for inclusion in the risk data sheets. For stressors not yet subjected to a comprehensive risk assessment, including a review and analysis of the literature, the information included in the data sheet should be based primarily on peer-reviewed papers reported in the scientific literature that vary in experimental designs and protocols. Risk data sheets to be used in the rating process should themselves be subjected to peer review before use and should include references to the sources of information. An example of a Stressor Risk Characterization Data Sheet is shown in Table 3-2, and the full collection of risk data sheets prepared by HEHS is presented in Appendix 3C. The following kinds of information should be included for each risk data sheet, if available:

a) **Exposure routes and pathways pertaining to the stressor**. Route of exposure might be inhalation, ingestion, dermal or some combination, depending upon the particular stressor. Pathway of exposure should provide some information on how the stressor travels from the source to the human receptor. For example, radon is advectively transported from soil gases into homes and its decay progeny are inhaled. The dioxins are generally emitted from combustion sources, transported through the environment and accumulated in food.

b) **Population exposed to the stressor**, e.g., total U.S. population or a special subpopulation(s). For many stressors, e.g., criteria pollutants, virtually all of the U.S. population is exposed to some degree. An example of an exposed sub-population of concern would be subsistence fishermen and their families, who may have a very high exposure to PCBs, dioxins or methylmercury.

c) **Average dose** to (or exposure concentration) for the exposed population.

Ideally, the distribution of exposures for the population would be presented here. However, this information is currently available only for a limited number of stressors, e.g., radon. Information on high end exposures in the exposed population should also be described, if possible.

d) **Relevant animal toxicological data**, e.g., NOAELs or LOAELs if available. It is generally accepted that an agent that produces an adverse effect in experimental animal studies will potentially pose a hazard to humans following sufficient exposure. In many instances, animal toxicology data can provide supporting evidence of adverse health effects in exposed humans. In the absence of adequate data on the toxicity of an agent in humans, we assume that effects in animal species are probably indicative of effects in humans. This is a default assumption, and is abandoned only when all relevant information on aspects of differential absorption, distribution, metabolism, and toxicokinetics between species have been considered and indicate that effects observed in animals will not occur in humans. Minor adverse effects will not be discounted unless there is evidence that they are not relevant to humans.

e) **Health effects from environmental exposures.** Information on the observed or potential adverse health effects for human populations at current environmental exposure levels should be summarized. Well-designed and conducted epidemiological studies of various sub-populations, such as highly exposed workers or the general population (e.g., PM-10, ozone), which have been peer reviewed, provide good evidence that exposed populations are at risk of adverse health effects.

f) **Biological half life in humans**, if known.

g) **Sizes of populations experiencing specific adverse health effects** due to current exposures to the stressor. This information will not be available for every stressor.

h) **Severity and persistence of adverse health effects**. For example, mortality, morbidity, and reversibility of endpoint.

i) **Occupational exposure limit(s)**. (Optional) Although Threshold Limit Values and Permissible Exposure Limits developed for the protection of healthy workers in industry are much higher than would be appropriate for the general population, which includes children and susceptible sub-populations such as asthmatics, these limits are often very close to exposures at which adverse health effects have been observed in human populations. They provide a useful reference point for comparison to average and high-end environmental exposures.

j) **Environmental half-life**. Environmentally persistent stressors, such as dioxins and freons, can accumulate in various environmental compartments and may be of greater concern than those which degrade rapidly under environmental conditions.

k) **Comments**. Any additional information which might be of use in rating the relative risk of a particular stressor should be included here.

Although it will not be possible to provide these kinds of data for every stressor, the data sheets should provide a space for each. Gaps in the data sheets will enable the experts (and the Agency) to easily identify critical gaps in information. In addition, it might be desirable to include an estimate of the fraction of disease incidence attributable to exposures to the environmental stressor. A stressor for which a substantial fraction of the total annual disease incidence is attributable to exposures to the stressor might be rated somewhat higher than one which contributed little to overall incidence. There are, however, some methodological pitfalls in making such estimates, e.g., double counting across two stressors.

Table 3-2. Stressor Risk Characterization Data Sheet for PM-10

STRESSOR	PM-10 (Airborne Particulate Matter, ≤10 μm MMAD)
Exposure Routes and Pathways Considered	Inhalation of outdoor air (PM in outdoor air infiltrates into indoor environments)
Population Exposed	All of U.S. population
Average (Potential)* Dose for Exposed Populations	• ~ 11 m^3/day X (9 - 34 μg/m^3) = 99 μg/day to 374 μg/day inhaled, depending upon where one lives • Doses to patients with chronic obstructive pulmonary disease (COPD) may be 3-times greater than for healthy adults (U.S. EPA, 1996)
Animal Toxicological Data	
Health Effects from Environmental Exposures	• Increased death rates in elderly from cardiopulmonary disease • Increased hospital admissions for COPD (emphysema, bronchitis) patients • Increased risk of acute respiratory disease • Decreased lung function in children and asthmatics
Size of Populations (estimated) experiencing health effects:	• ~60,000 premature deaths in the elderly annually from cardiopulmonary disease (American Journal of Respiratory and Critical Care Medicine, 1995) • Relative risk values range from about 1.02 to 1.08. Although RR is not very high, exposed population is so large than incidences of health effects large.
Severity and Persistence of Adverse Health Effects:	• Premature death • Hospitalizations due to exacerbation of COPD • Decreased lung function in children and asthmatic is irreversible effect
Estimated Fraction of Disease Incidence Attributable to Environmental Exposures (if Known):	PM is estimated to account for about 60,000 premature cardiovascular deaths annually; this is about 8% of the annual cardiovascular deaths in the U.S.
Other Comments	

REFERENCES:

U.S. EPA. 1996. Air Quality Criteria for Particulate Matter. EPA/600/P-95/001cF, April 1996.

U.S. EPA. 1992. Technical Support Document for the 1992 Citizen's Guide to Radon. U.S. Environmental Protection Agency.

3.2.4 Rating of Confidence in Relative Risk Rating of a Stressor

Respondents are asked to indicate their level of confidence in the risk rating given for each stressor. The confidence ratings are Very High, High, Medium, Low and Very Low. This rating can include both the respondent's judgment of the state of existing knowledge as well as the weight accorded his or her individual expertise about a particular stressor. The Committee considered separate ratings for these items but, after much discussion, decided to include only a single confidence rating in the interest of maintaining a simple methodology. If a respondent believes the state of scientific knowledge to be inadequate for a rating, this information can be supplied in the comments section.

3.2.5 Factors Influencing Relative Risk Rating

For each stressor rated, the respondent is also asked to indicate the major factors which influenced his or her judgment of relative risk by checking the appropriate boxes. These factors are:

a) **Size of the population affected.** A high risk rating might be based on evidence that many members of the population are experiencing adverse health effects from environmental exposures to the stressor, while a low risk rating based on size of population would indicate that only a small fraction of the population is being exposed at present.

b) **Particular sub-populations at risk.** A check for this factor in combination with a high risk rating for the stressor might arise if some small sub-population(s) are subjected to very high exposures and risks.

c) **Severity and persistence of health effects.** A high risk rating might be given for a stressor based on health effects that are life-threatening, life shortening, and/or irreversible. A low risk rating might be assigned to a stressor because the most significant adverse health effect is transient and apparently reversible.

d) **Percentage of attributable incidence.** A respondent might assign a high risk rating to a stressor because exposure to the environmental stressor accounts for a significant percentage of the total annual incidence of a given disease or health endpoint. For example, based on the estimated annual number of lung cancers from radon exposures (EPA report on radon, 1992), it can be estimated that about 1-2 % of the annual incidence of lung cancer in the U.S. is attributable to radon exposures.

e) **Persistence in the environment.** A respondent might rate a stressor as very high or high because it is not rapidly degraded in the environment but tends to accumulate and, because of this property, poses an environmental health risk.

f) **Potential future risk.** A respondent might rate a particular stressor as very high or high because the stressor is very likely to pose a significant environmental health risk in the future if no actions are taken to reduce anticipated exposures arising from identified sources. The respondent might also check this factor as influencing his/her low risk rating because he/she judges that no increased health risk is likely to arise in the future.

g) **Other.** The respondent is asked to check this if some major factor, other than those provided, influenced his or her rating. The respondent is also asked to identify this other factor(s).

h) **Comments.** Experts may have important knowledge that has not yet become widely known. The "Comments" box, as well as the "Other" category, provide means to alert EPA to this knowledge.

3.2.6 Survey Design

Implementation of an expert survey requires a specific description of the population to be surveyed and how the sample is drawn. Even the Delphi process, or any other process that relies on a select committee to make judgments or issue ratings is, sometimes implicitly, a hostage to sample selection procedures and possible bias. One virtue of the Web design is the enlarged possibilities it presents for explicit

samples. For example, a risk rating exercise might draw from the total SAB membership; or, for some purposes, it might sample from the membership of the Inhalation Specialty Section of the Society of Toxicology or the International Society for Exposure Analysis; or, its sampling population might come from a specific community. Because expert groups may be viewed as inherently biased, the ability of EPA to use an unbiased survey technique should prove useful.

3.3 Analysis and Reporting of Relative Risk Rating Survey Data

A leading purpose of the Web survey instrument is to evaluate the diversity of rater estimates. The importance of rater diversity is illustrated in Table 3-3, which shows some of the patterns of rater judgments that would yield equivalent average (mean) risk ratings.

Table 3-3. Distributions Of Risk Ratings For a Stressor That Would Yield an Average Rating of "Medium"

Very High	High	Medium	Low	Very Low
oo	oo	oo	oo	oo
o	o	oooooo	o	o
		oooooooooo		
ooo	oo		oo	ooo
oooo	o		o	oooo

Patterns of rater responses convey important messages. A stressor whose ratings are concentrated at the extremes reflects a high degree of polarization among the population sampled. Similarly, the degree of consensus among raters on the dimension of confidence also reflects the characteristics of the sample population and its interpretation of current information. Figure 3-5 depicts four synthesized examples of the kinds of patterns such plots might produce (based on 12 fictional respondents). As discussed by the Subcommittee, they convey data potentially helpful to the formulation of risk options. For example, plot (C) might characterize a phenomenon such as global warming whose possible health impacts might be widespread and

severe but that currently lacks enough supporting data to elicit much confidence in that evaluation.

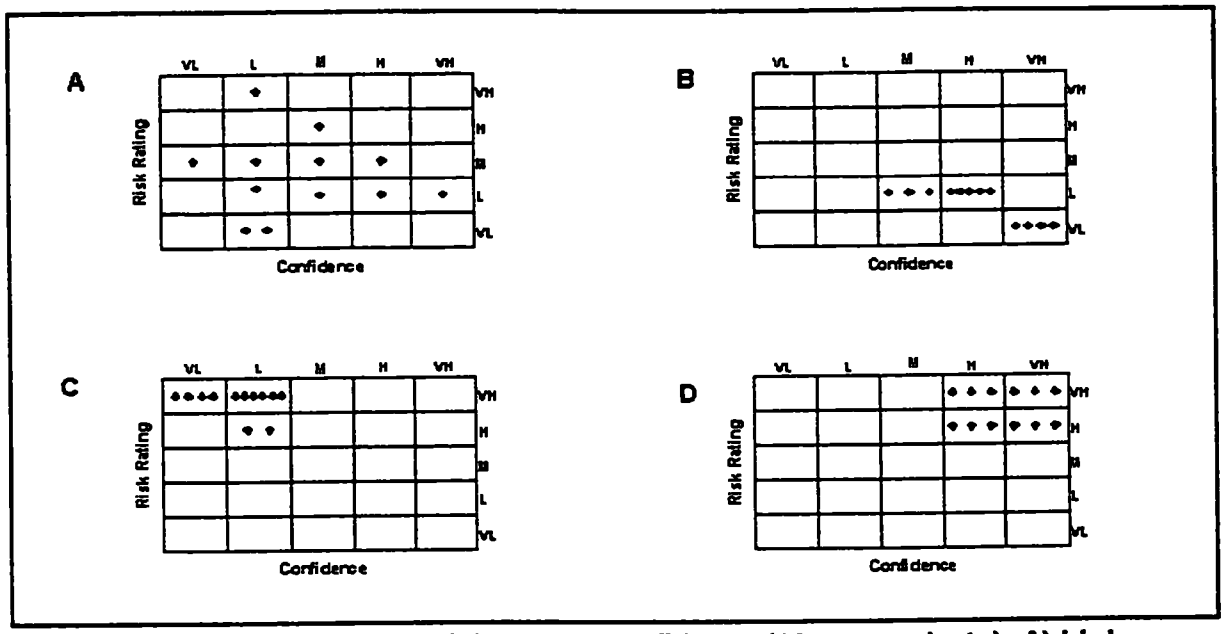

Figure 3-5. Synthetic plots of risk versus confidence (12 respondents): A) high variability; B) low risk, mid to high confidence; C) high risk, low confidence; D) high risk, high confidence

The Subcommittee views distributions of respondents' risk ratings and confidence in those ratings as the key pieces of information provided by the rating exercise. In many instances, these raw data may prove sufficient for conveying the relative degree of risk and confidence associated with each rated stressor. However, the Agency may apply data reduction techniques to these data; for example, many readers of EPA publications will be comfortable with means and standard deviations, which can be expressed by converting rater responses into numerical scores. When the distributions are clearly asymmetrical, Box Plots showing medians and 10th, 25th, 75th and 90th percentiles will prove more informative. Other kinds of transformations are also available, but are constrained by the limitation that the ratings are ordinal, not ratio scales. The Agency could also use non-numerical, visual presentations of the results of the survey that would provide a different way of looking at the results, and might entail fewer underlying assumptions than do numerical data reduction techniques.

In essence, the Agency is encouraged to develop a set of rules governing how the various combinations of risk/confidence ratings (25 in the current scheme) are ordered. These are subtle judgments, and might be assigned to an expert panel for resolution once the survey data have been gathered.

A second analysis of the Web data will be based on rater responses to the prompt requesting the factors upon which the ratings were based (Figure 3-2). These data provide considerable amounts of information, but are not readily transformed into a simple score. However, they could be tallied as distributions, and when tallied, would illuminate the risk ratings and provide the Agency with potential initiatives for remediation or regulation, and specific goals for health research.

One major advantage of a Web survey is the ease of modification. The original design sought to include sources, pathways (exposure route), and endpoints as well as the basis for selection. In this first version, respondents were asked to rank primary, secondary, and tertiary relationships for all variables. For example, several different sources might be responsible for environmental exposure to a particular stressor (e.g., both transportation and point sources for air pollutants), or a stressor might traverse different exposure routes (e.g., ingestion and inhalation for arsenic), or might affect more than one endpoint (e.g., brain development and blood pressure for lead). Although HEHS deemed this ranking form too complex for its current purposes, it illustrates the flexibility afforded by electronic survey techniques. Some of the modifications that might be incorporated in future versions, at the cost of increased complexity, include provision for rater choices of endpoint, default hazard identifications, default q1* and RfD values, graphical data, and the submission of the rater's own quantitative risk figures. Building in such modifications, however, would constitute a major software design project.

3.4 Implications of Ratings

The Relative Risk Rating Survey will provide information from a variety of experts and stakeholders that can yield ratings for a stressor(s) that reflects the category of risk, and assignment of the confidence that the responders to the survey made about the rating. Currently, assignment of a stressor to a priority category has typically been formulated by a single number or classification. Separating risk and reliability estimates

provides a regulator, policy maker, scientist, or other interested party, with the types of information necessary to begin establishing a more meaningful priority for the particular stressor. Based upon the risk and confidence ratings, actions or potential responses could range from no additional actions at the time, to new research, or to implementation of new exposure and risk reduction strategies. These should be considered, at the outset, as the initial priorities, and be revisited periodically to retain or to modify the initial risk assignment. At the same time, new stressors can be added to the list based upon new environmental release or formation, exposure, or health data.

3.5 A Fuzzy Logic Approach

An important element in the process of soliciting probabilistic judgments about risks from experts is obtaining their views of the uncertainties bounding their estimates. Morgan and Keith (1995) describe how they tried to obtain both mean and standard deviation judgments about climate change from a small group of authorities and Morgan and Henrion (1990) describe several ways by which such information might be presented graphically. The survey technique described in this chapter deals with uncertainty by eliciting the judge's evaluation of the confidence he or she places in the health risk rating. A low degree of confidence corresponds, in this instance, to wide uncertainty and a high degree to narrow uncertainty.

Different combinations of risk and confidence ratings, as depicted in Figure 3-5, can help illuminate which stressors deserve closer examination or which might be assigned high or low priorities. Modifications of the survey protocol could be designed, however, to procure indices of uncertainty more directly. Fuzzy logic approaches could prove useful for doing so.

Fuzzy logic is the term invoked by Lotfi Zadeh, the originator of the discipline, to designate a tactic, in essence, for dealing with uncertainty. As noted by Morgan and Henrion (1990), even experts find it more comfortable to deal with uncertain probabilities by applying verbal labels such as "likely" and "unlikely" rather than by issuing numerical estimates. Fuzzy logic, rather than representing uncertainty in the conventional sense, as in soliciting estimates of variance, deals directly with ambiguities such as those inherent in the terms above. Conventional logic treats every propositon as either true or false. Conventional mathematical definitions of set membership also

rely on binary definitions. Fuzzy systems view truth definitions and fuzzy set membership as possessing any value in the range [0.0, 1.0].

Consider a speaker's description of the weather as "hot." It conveys considerable information although the adjective is imprecise and subjective and is a function of the speaker's personal experience and environment. Figure 3-6 is a Fuzzy depiction of "hot." Almost everyone would consider an ambient temperature of 40°C to be hot, so that an ambient temperature that high might be assigned a truth value of, say, 0.90. In Zadeh's terminology, that value corresponds to its membership in the set, "hot." Below that temperature, your definition might depend largely on where you live. Some inhabitants of Rochester, New York, would classify a reading of 25°C as "hot," but residents of Miami might consider such a reading as somewhat cool. At 25°C, membership in the set, "hot," might elicit a truth value of 0.15. In traditional, Boolean, set theory, "hot" would be assigned a *crisp* definition such as greater than 30°C and symbolize a clear dividing line certain to evoke debate between Rochesterians and Floridians.

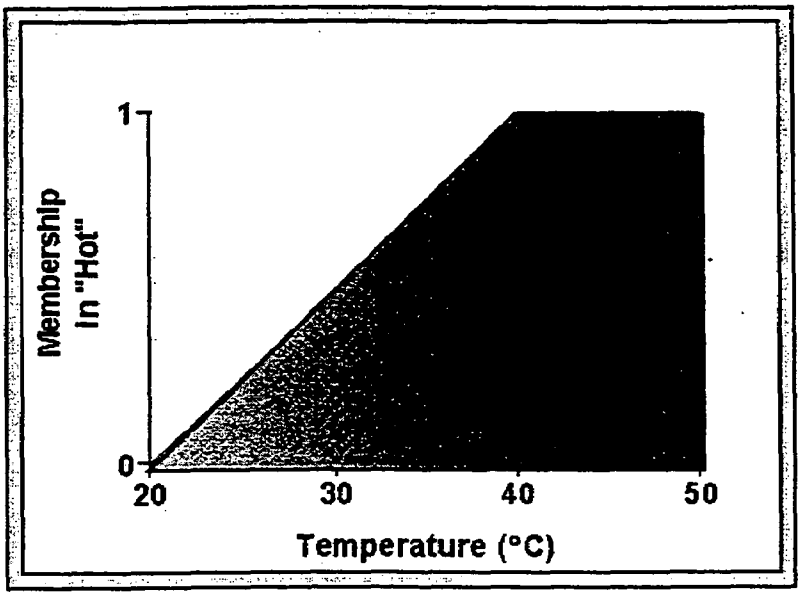

Figure 3-6. Membership in the class, "Hot," is not fixed, but depends upon individual definitions. The shading is not necessary, but is designed to show that, in this instance, the proportion of judges applying the adjective is less dense at the lower temperatures.

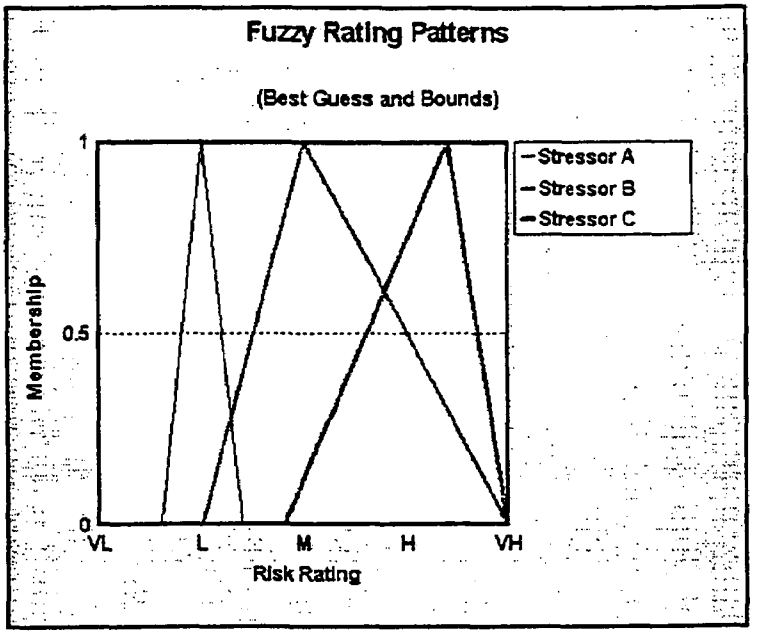

Figure 3-7. The vertical line, situated there by a hypothetical rater, can be said to represent divided membership in the risk classes, "Medium" and "High." In place of a numerical value, the respondent's rating can be interpreted as an estimate of a truth or membership value in the fuzzy set, "Medium" of about 0.7 and a membership value in "High" of 0.3.

One way to display the elements of fuzzy logic classification is shown in Figure 3-7. Here, the usual numerical scale has been translated into verbal labels from Very Low to Very High, in conformity with the categories used earlier. The figure depicts the set (or rating) of "Medium," for example, as extending from Low to High, with maximum membership (or truth) value at Medium and minimum membership values at Low and High. Suppose a judge were asked to locate the point on the abscissa corresponding to his or her best estimate of risk and marked it at the position designated by the vertical line. The position of the line indicates majority membership in "medium," and what might be called minority membership in "high." Roughly translated into percentages, the line coincides with about 70% membership in "medium" and 30% in "high."

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The estimate defined by the vertical line in Figure 3-7 provides no indication of the rater's uncertainty about her judgment. As we noted earlier, a rater's uncertainty can be translated into what statisticians term "confidence limits." Morgan and Keith (1995) secured analogous information from their judges by asking for standard deviation estimates. With a fuzzy logic approach, a similar kind of estimate can be obtained more simply.

Figure 3-8 depicts fuzzy risk ratings of three hypothetical stressors. The triangle labeled "Stressor A" reflects the rater's judgment that this agent presents a Low risk that is unlikely to be much lower or higher. The fuzzy rating for "Stressor B" is centered at Medium, and is asymmetric to reflect the judgment that it could be as small as Low or as great as Very High. The third triangle, centered between High and Very High, shows the rater inclined towards the latter, but also locating the lower bound between Low and Medium.

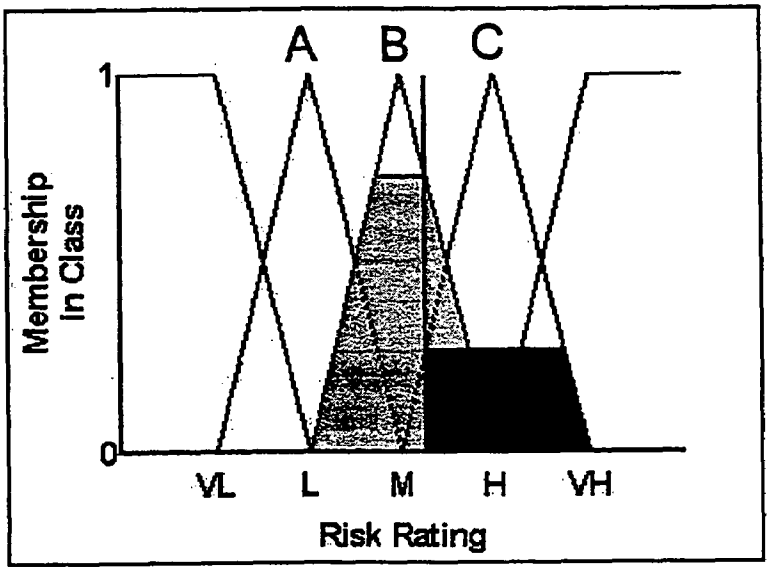

Figure 3-8. Hypothetical ratings for three different stressors. The peak represents the rater's best guess of where along the abscissa the risk belongs; that is, where membership is believed to be 1.0. The lower corners of the triangle are set to reflect the rater's judgment of the lower and upper bounds of risk. To summarize the judgments of many raters about a particular stressor, the abscissa locations are defined by a scale, say, from 0 to 100 (Figure 9) and the heights summed at each integer position from 0 to 100.

An example of the kind of visual display that can be used to elicit such judgments from raters is shown in Figure 3-9. The rater views three horizontal bars, each equipped with a slider that can be manipulated by a computer mouse. As the slider is moved, its corresponding value on the abscissa, which ranges from 0 to 100, appears above the bar. The uppermost bar is used to set the location of the rater's best guess about the risk posed by the particular stressor. The left bar allows the rater to set her estimated lower bound, defined as, "Unlikely to be any lower than this." The right bar sets the location of the estimated upper bound, defined as, "Unlikely to be any higher than this." Respondents tested so far find it relatively easy to manipulate the sliders. The triangles plotted as a result may be asymmetric, as shown in Figure 3-7. For summarizing the responses of a specified sample of raters, the abscissa is divided into, say, 100 units, and the summed ordinate values then calculated.

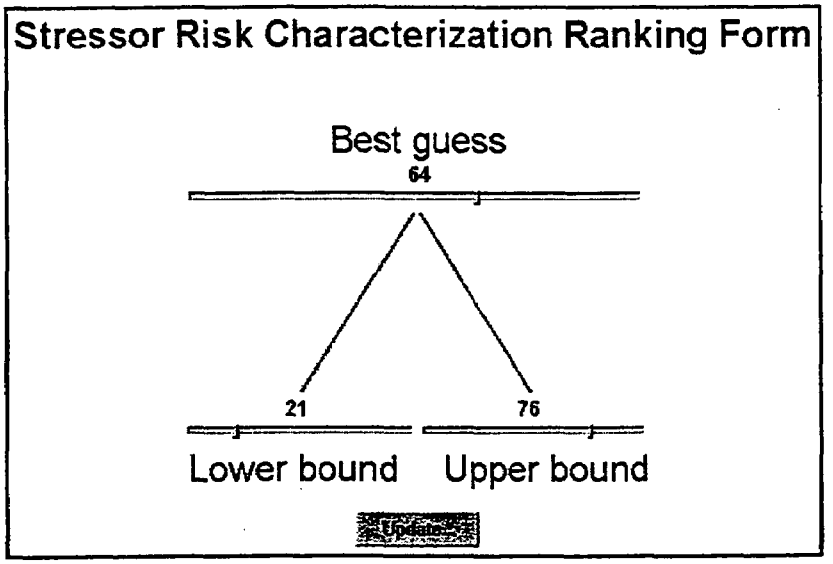

Figure 3-9. Depiction of a visual display presented to a rater asked to indicate where she would position her best guess of the health risk posed by a specific stressor and where she would place the lower and upper bounds of her estimate. With the computer mouse, the respondent locates the slider at the appropriate numerical index. For this display, the scale runs from 0 to 100, but a second parallel scale, could show the corresponding verbal labels from, say, Very Low to Very High. These numerical positions can be converted into the kind of visual display shown in Figure 3-8.

Fuzzy logic can also be applied to the criteria upon which the raters have based their risk estimates. For any particular stressor, they could be asked to also submit their degree of reliance on these criteria. That is, they could be asked to supply the weight accorded them, from Very Low to Very High, in making their risk judgment. For example, the rating for a particular stressor might have been based largely upon the size of the population affected while another stressor rating might have been based largely on outcome severity, and its impact on a particular subpopulation such as children. The weights given these criteria and the associated uncertainty about their relevance can be translated into the kind of responses depicted in Figure 3-8.

3.6 Extensions and Refinements of the Methodology

The Health Risk Rating Methodology was developed by the Human Exposure and Health Subcommittee to provide a unique tool for assigning relative risk to selected stressors. Its design was governed by three determinants that are seen as lacking in many risk evaluation exercises. First, it explicitly recognizes uncertainties by requesting raters to provide estimates of their confidence in the available data and in their own grasp of this information. Second, it offers the Agency a mechanism with which to secure risk ratings from designated populations not limited by the constraints of time and geography imposed by the usual committee deliberations. Third, it permits improved estimates of rater variability, a key policy index that often is not available or is explicitly concealed by focusing on consensus.

For this prototype exercise, the selection of stressors and the characterizations of each stressor in the data sheets were based upon the experiences and professional judgment of the members of the Subcommittee. Because of the limited resources available to it, the Subcommittee was unable to extensively test and refine the methodology. In practice, the EPA would likely convene a working group and identify resources for conducting the described activities.

These resources will provide the basis for developing an implementation plan for the proposed rating methodology. This must include: a) selection of the stressors to submit for risk rating at the present time, b) development of a sampling program for expert selection and completion of data sheets, c) statistical methods for combining the risks and the confidence to obtain a weighed priority, d) methods that link the data sheets to basic resources or sources of information or exposure, effects, and other variables, e) approaches for peer review of data sheets prior to implementation of a expert survey, f) activities to promote the availability and use of the Stressor Risk Characterization Data Sheets, ratings and prioritizations of individual stressors by stakeholders, e.g., on the Internet, and g) identification of a timetable for implementation of the project, and completion of respondents on the initial data sets and newly identified stressors.

3.7 Summary and Conclusions

a) Risk ratings can inform environmental decision-making, but in and of themselves cannot be the sole basis of these decisions.

b) Risk ratings are not purely scientific since value judgments concerning the relative importance of a number of factors must be used to arrive at a rating for each stressor. Furthermore, when information on a particular stressor is limited, judgments must be made about the uncertainties and the degree of protectiveness that is appropriate.

c) Human health risk ratings, however, can be scientifically based, justifiable, well-defined, and transparent.

d) Risk ratings are traditionally the product of a committee instructed to reach a consensus. Elicitation of the opinions of many individual experts, through a formal process, can provide an alternative approach to rating environmental health risks, especially when data are incomplete, contradictory, or multidimensional. This approach also provides the means to formally address uncertainties in the ratings, and to elicit information on the factors that influenced the ratings.

e) Development of a single, merged ecological and human health risk list of ratings requires value judgments. Ecological and health risk ratings can be developed within a consistent conceptual framework, with many commonalities in the factors used to rate risks.

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Appendix 3A. Health Risk Assessment Introduction

Thank you for agreeing to participate in this survey. Its purpose is to inquire about how you rate the health risks of a variety of environmental stressors. It also asks about the degree of confidence you place in your ratings or in the available scientific data. We are also interested in the reasons for your risk ratings. This information will be incorporated into a report for EPA's Integrated Risk Project, an undertaking of the Science Advisory Board.

Here is how it works. At the bottom of this page, you select **Go to the Register Form.** An entry form will appear. To register using the **Registration Form**, enter the information requested and choose a password, as indicated. You can also practice using the system by entering *test* and *test* for the username and password, respectively. When you select **Login**, the **Stressor Risk Characterization and Health Risk Rating Data Sheet** will appear. On the same page, you can select **Review of General Instructions**, which describes your task. It also contains descriptions of some of the criteria (**Factors**) you may have used to assign your ratings. After reviewing the instructions, you **Exit** the page (or click on **Go to Stressor Risk Characterization Data Sheet**) and proceed to rate the stressors. When you choose **Select a Stressor to Rate,** a list of stressors will appear. Once you have made a selection, you are asked to make a rating of Health Risk from Very High to Very Low. Your will also mark your confidence in your rating. "Confidence" is a way of dealing with the largely subjective nature of such ratings, and permits us to join combinations of risk and confidence ratings for the assignments of priorities or ratings. For each of the stressors, we have tried to assemble the most relevant information in a highly compressed format. You access this information by selecting **Open Information Window**. After you have reviewed this information, you return to the rating form either by selecting **Close Information Window** (if you have the rating form open concurrently) or by exiting the information page.

Once you have rated the first stressor, you then go on to the next stressor in the list. If you do not wish to rate a particular sstressor either because of lack of information or because of the nature of the stressor, there is no reason for you to do so; that is important information in itself. We have provided opportunities for you to offer additional reasons for your ratings and for general comments.

We have confined this demonstration to a limited number of stressors. The forms can be expanded. Try it so we can see how it works and how we might want to proceed.

When you have finished making your selections, check your answers and then push the "Send Results" button. That will mail the answers to me (Bernie Weiss). For technical details or problems e-mail **Geoff Inglis** (inglis@envmed.rochester.edu) .

You may send additional comments on these pages to **Bernie Weiss.** Your comments will be posted for the group to read on the **"comments page"**

Go to the Register Form

If you are already registered:

Go to the Login Form

Appendix 3B. Instructions

For each environmental stressor, please indicate:

Your expert judgment of the relative environmental health risk that it poses to the U.S. population, either current or potential future risk. Assume that all current controls and regulations remain in place for the rating. Please rate as:
Very High (VH), High (H), Medium (M), Low (L) and Very Low (VL) or (?).

We have provided some information on each stressor in a "Risk Data Sheet." For the relative risk rating, please use your expert judgment, and all, some, or none of the information provided on the risk data sheet.
If you feel that you cannot assign a relative risk rating to any given stressor, for any reason, please indicate this with a " ?" in the entry for rank.
Your level of confidence in your rating.
Very High (VH), High (H), Medium (M), Low (L) and Very Low (VL) or (?).
Your confidence in the relative health risk rank that you assign to a given stressor can include both your judgment of the state of the existing scientific knowledge about the health risks of the stressor plus a judgment of your own expertise regarding any particular stressor.

Please indicate also (by checking) the major factor or factors that influenced your relative risk rating. For example, one particular factor might have been decisive. Alternatively, a collection of several factors might have determined your rating.
Check all factors that apply.

Size of population affected : A high risk rating might be based on evidence that many members of the population are experiencing adverse health effects due to environmental exposures to the stressor. Conversely, a low risk rating might be based on evidence that few, if any, members of the population are experiencing adverse health effects as a result of exposure to the stressor.

Particular subpopulations : Although most of the population may not be exposed to levels of the stressor great enough to cause adverse health effects, a rating of high risk might arise if some smaller sub-populations are subjected to very high exposures and risks; or, the stressor is might be viewed as acting on an especially susceptible subpopulation. A rating of low risk might arise if little

evidence points to high exposures or risks for any subpopulations or if extreme sensitivity has not been identified in any subpopulations.

Severity of health effects : A high risk rating might be based on the presence of pronounced, irreversible, or life-threatening adverse health effects irrespective of the size of population affected. In parallel, a low risk rating might be based on mild, transient health consequences arising from exposure to the stressor especially if they occur in a limited number of individuals.

Percent of attributable incidence : A high risk rating might arise if a significant, though small, proportion of the total U.S. incidence of a health effect is seen to arise from exposures to the environmental stressor; e.g., it is estimated that about 1-2% of the annual lung cancer incidence in the U.S. is due to radon. In contrast, a low risk rating might be based on lack of evidence that a significant proportion of the incidence or prevalence of an identified health effect can reasonably be attributed to the stressor.

Persistence in the environment : A high risk rating might follow if the stressor is believed not to degrade rapidly in the environment but, instead, tends to accumulate over time in one or more environmental compartments. A low risk rating might follow if the stressor is believed to degrade relatively rapidly in the environment.

Potential future risk : A high risk rating might be based on predictions of a significant potential for environmental health risks in the future if no actions are taken to reduce anticipated exposures arising from identified sources. A low risk rating might be based on the assumption that no rise in or relatively minor increases in exposures to the stressor are likely to be seen in the future.

Other: Please describe briefly any other reason for your ratings.

Comments : Provide, if you wish, comments further amplifying or describing the reasons for your choice of ratings.

To Summarize : Please be sure that you have responded to all three questions:

- Your relative risk rating
- Your confidence in your rating
- The factors determining your responses

Go to Stressor Risk Introduction page.

Go to Stressor Risk Characterization Ranking Sheet.

1

2 # Appendix 3C. Risk Characterization Data Sheets

PART III INPUTS TO ENVIRONMENTAL DECISION-MAKING: ECONOMICS AND VALUATION

PART III—INPUTS TO ENVIRONMENTAL DECISION-MAKING: ECONOMICS AND VALUATION

Preface

Integrated environmental decision-making at its heart is about how people, acting as individuals or through their government, can make themselves best off through actions that affect the environment directly and indirectly. Judging which among alternative actions will produce the most "well being" requires information not only on risks, discussed in the previous chapters, but also on the definitions of "well being", i.e., goals, and the relationships among goals and among alternative choices or actions. Just as the IED framework requires that we look at the full range of risks, it is also necessary to consider the full economic consequences of decisions of whether and how to address certain environmental risks or sets of risks. An understanding of the tradeoffs implied by these choices is crucial both during *Problem Formulation* and *Analysis and Decision-Making.*

The Economic Analysis Subcommittee (EAS) was given the task of describing the ways in which benefit/cost analysis can be used to frame choices about different uses of societal (public and private) resources. In addition, at the request of the Deputy Administrator and in recognition of the fact that some environmental changes are difficult to monetize and value, a complementary subcommittee, the Valuation Subcommittee, was formed to focus explicitly on improved methods for assessing ecosystem values and incorporating that information into decision-making. The VS was a multi-disciplinary group that included ecologists, economists, and other social scientists. The work of these subcommittees is contained in the following two chapters, which taken together provide a framework within which choices about the environment and its protection can be understood and implemented to best effect.

As discussed in previous chapters, environmental decision- making requires an understanding of the physical and biological relationships between stressors--substances and conditions that can cause harm--and organisms or ecological systems of concern, including the relationship between alternative levels of these stressors (or of

substitutes for them) and their effects. This is the province of science, and with other factors included, is known by the term "risk assessment."

By itself, however, such information is inadequate to guide decisions. The reason is that there are a near-infinite number of substances and conditions that can and do impose harm, and it is impossible and unwise to act to eliminate or even to reduce them all. The ability to lessen the risks they pose is limited by the resources of labor, capital, knowledge (technology), and physical endowment that is devoted to the task. And these same resources are those on which people depend to provide the other things desired and indeed necessary for life, including food, shelter, fiber, manufactured goods, amenities, and investment to provide for future populations and for increases in ability to satisfy wants in the future. A critical task during Problem Formulation, therefore, is to choose which of the risks (potentials for harm) or combinations of risks are to be acted upon, and to what degree; i.e., to set environmental or risk reduction goals. Intimately related, of course, is the choice of the most effective and efficient mechanisms and instruments to be used in actually accomplishing the task; these aspects of options selection are discussed in Part IV.

The premise our democratic society has selected as the fundamental building block for such choices is that they should be guided by the individual goals and desires of people as expressed directly and in and through their governmental and other institutions. Logic then suggests that the task of the decision-maker is to maximize the attainment of these goals and desires with respect to both environment and all the other things that people want. These include, of course, things material and non-material, and also people's goals for future generations and in fulfilling their stewardship responsibilities for natural systems. Put directly, then, it follows that the task of environmental protection activities is to provide--for now and for the future--the healthiest, safest, most ecologically secure set of conditions that the American people are willing to pay for (in terms of other things they must give up to get them).

Chapters 4 and 5 provide a framework to guide decision-makers to deliver on this task.

Chapter 4 adopts the commonsense framework for decision-making of comparing the advantages (benefits) and disadvantages (costs) of the range of plausible actions presented by a situation. It first addresses the difficult question of how to sort out the true benefits and costs of actions designed to reduce risks and improve the environment, and in the process identifies the many ways in which unsophisticated efforts to assess economic consequences both underestimate and overestimate some elements of benefits and costs, and totally ignore others. It then discusses techniques for measuring and eliciting the benefits and costs of environmental actions, and describes the requirements for and difficulties and uncertainties of such techniques.

Chapter 4 demonstrates that the development of benefit/cost measures in terms useful to decision makers is not a simple task. It requires disciplined treatment of a number of factors. For one thing, the treatment of a stream of benefits and costs as they are realized over time is crucially important. The chapter concludes that discounting of those streams in a consistent way to show current decision-makers the full implications (present and future) of alternative choices is essential to provide meaningful information. It also recognizes that this discounting process can be controversial, and that it cannot be undertaken blindly.

Another matter of concern in presenting benefit/cost information is how to treat the fact that while analysis can demonstrate which risk management scenario provides the greatest net benefits, it cannot take account of the fact that the distribution of the costs and benefits is not uniform. Sometimes the costs are borne by some people and the benefits gained by others. More frequently, some may lose a little and others a lot, while at the same time others are benefitting disproportionately or not at all. Complicating all of this is the fact that individuals affected by the environmental action are unequally endowed in wealth and income. This influences both the benefits and costs as they enter the evaluation framework, and the sense of fairness that pervade the results. Chapter 4 recognizes that these distributional issues cannot be handled within the framework of benefit/cost analysis, but it asserts that this makes them no less important in the choice of actions to be taken. Consequently, distributional issues are seen as matters on which the analysis can shed light, but which must be taken into account outside the analysis at the final decision stage.

Chapter 4 conveys a mixture of confidence and humility. The framework of

benefits and costs is seen as a consistent, coherent, transparent, robust tool that provides decision-makers with a firm basis on which to proceed. When well done, the products of specific benefit/cost analyses are seen as important inputs for decisions, sufficient to the task to be relied upon with confidence most of the time, always limited by data and sometimes by methodology, and sometimes sufficiently uncertain as to provide only indicative information—which nonetheless can be useful.

As with risk assessment and comparison, however, even the best and most complete benefit/cost analyses are also shown to be inadequate in themselves to yield "answers." Other factors always are present and must be taken into account. In essence, integrated environmental decision-making requires consideration of the broad implications of actions taken to improve the environment across stressors and across options to relieve them. The framework outlined in Chapter 4 provides a methodological approach to such decision-making.

Chapter 5 is a vital complement to Chapter 4. It addresses the important issue of how the benefits (and sometimes costs) of changes in ecological outcomes can be properly determined and incorporated into the decision process. By their nature, changes in ecological conditions are often not easily observed in quantifiable terms that can be rendered in the monetary units that most often are used to compare possible outcomes in a benefit/cost analysis. For this reason, it is often thought that the benefits received from ecological protection are under-estimated, and consequently, that ecological systems are under-protected as compared to other goals and desires of people.

Chapter 5 examines the issue of ecological valuation in detail. In doing so, it starts with an examination of ecological values and concludes that in principle they are not different from other values, and that they need not enter into the decision process in any unique way. At the same time, however, measuring and incorporating the values ascribed to anthropogenic changes in ecological conditions do present serious difficulties that require that special care be taken.

For one thing, citizens doing the valuing often have insufficient knowledge of how changes in ecological factors affect the things they care about. They need expert scientific assistance in making the important connections. For another, many of the

benefits from ecological systems are subtle and do not enter into conscious consideration in the normal course of events, as do market goods and services. For these, special techniques for eliciting preferences are required. In addition, some of the values ascribed to ecological outcomes arise in a social context and may not surface from commonly used individual preference measures. Equity, sustainability, and stewardship are often cited as examples of such values.

Most importantly, because ecological services often do not enter into markets or enter incompletely, it is difficult if not impossible to provide monetized measures of the benefits they deliver. Often it is not possible to even determine quantitative measures for differences in outcomes. Qualitative measures of benefits and costs, therefore, must be arrived at and then incorporated into decision processes. In short, the valuation of ecological costs and benefits is prone to error because elements valued by people may be omitted or incorrectly specified and because measurement is inherently more difficult than with goods and services for which market and market-like measures are available.

Chapter 5 offers suggestions to practitioners of benefit/cost analysis and decision-makers that will improve the incorporation of ecological matters into decisions. It goes further, however, in making two recommendations. The first is that there be an expanded use of deliberative processes to assure that all relevant elements are included in decisions, and that they are valued properly. While deliberative processes are seen to have an important role, Chapter 5 also notes that such processes should be used discriminately and be tailored to the situation. When this is done, deliberative processes will not only generally lead to outcomes that are more satisfactory and robust, but also will not cause inordinate delay and indeed may even speed the delivery of ecological protection because they lessen post-decision controversy.

The other key recommendation of Chapter 5 is for further research on and experimentation with additional approaches to valuing benefits of environmental systems. Existing approaches are seen to be inadequate in their treatment of such values as fairness and sustainability. They also have difficulty in incorporating the systemic benefits of such matters as biodiversity, and are incomplete in their treatment of dynamic responses to change. More holistic approaches that take account of the web of interactions involved in decisions surrounding ecological systems and their

interactions with other production and consumption activities would be helpful.

The overall message from Chapters 4 and 5 is that integrated environmental decision-making requires a framework within which the decision-maker can meld the results of science and the goals and values of the people served and formulate acceptable decisions. The further message of these chapters is that such a framework exists, but that its use requires skill and artistry, and indeed that in some respects the framework itself remains a work in progress.

CHAPTER 4. BENEFIT/COST ANALYSIS FOR INTEGRATED RISK DECISIONS

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CHAPTER 4. BENEFIT/COST ANALYSIS FOR INTEGRATED RISK DECISIONS

4.1 Introduction

No society, no matter how wealthy, has available the resources to "solve" every problem that confronts it. Schools will never be as good, nor food, clothing, shelter and access to medical care as abundant as we would all like. Similarly, no society--modern or otherwise -- will ever be free of all environmentally mediated risks to human health and ecosystems. Thus, it is of great importance to be able to identify the most serious threats to health and the environment, and also to target certain of them for possible mitigation.

Identifying what we might call the total risks associated with air and water pollution, solid and hazardous waste disposal, drinking water contamination and other activities is largely the province of risk assessment. Determining these risks requires us to ascertain which pollutants are extant in the ambient environment, the route(s) through which humans or ecosystems are exposed to these pollutants and the duration of these exposures, and the resulting consequences. At a minimum, this requires the skills of engineers and chemists; atmospheric, terrestrial, and aquatic scientists; toxicologists, epidemiologists and biostatisticians, and clinical health specialists; and ecologists (including botanists, forest and fisheries scientists and others). Some risks may be described quantitatively, occasionally with some statistical precision. Often, we can do no more than suggest that a particular pollutant may have adverse effects on human populations, or on ecological systems. As described in Part II, assessment and comparison of risks associated with different stressors or activities is an important part of both formulating "the problem" and analyzing possible risk reduction options.

Economics typically concerns itself with the way in which people allocate scarce resources among many competing needs so as to make themselves--broadly speaking--as well off as possible. As such, the possible contributions of economics to Integrated Environmental Decision-making lie largely--though not exclusively-- in deciding which risks it makes sense for society to make incremental changes in

(sometimes called "marginal" changes) once they have been characterized, in deciding to what degree those risks ought to be reduced, and in identifying appropriate policy instruments that can be employed to achieve such risk reductions.

Economics, along with other behavioral sciences, can also help in understanding risks by characterizing the extent to which changes in behavior can effect exposure to stressors. For example, although many people spend time outdoors on a clear day, it would be incorrect to count them as the population at risk on a very polluted day -- for the simple reason that some of those people would alter their behavior on account of the pollution. Similarly, contamination of an aquifer used for drinking water may well increase some people's risk of acute or chronic illness, but may not affect all users if everyone were informed about the risk and most chose instead to drink bottled water. In both of these cases, economics would suggest that there were "damages" associated with the pollution, but these damages can take forms other than just increased risk of illness.

It is very important to note the distinction drawn above between risk posed by a certain exposure level and the possible "changes" in risk that would result from control actions that reduce either stressor or exposure levels, or both. Economic analysis is especially useful in determining which changes in risk level make sense to pursue via policy measures and which do not.

The focus of this chapter is on the role of economic analysis, particularly benefit-cost analysis (or BCA), in helping society to decide which environmental risks to address first and which to leave for a later time. The IED framework recognizes that such analysis should be an important component of decision-making, but not the sole consideration. In fact, there are instances where benefit-cost considerations do not play a direct role in decision-making. For instance, although it would be most unusual, decisions about which risks to address and how to address them could be determined in a national plebiscite in which voters directly established risk management priorities; voters in California often do make decisions in statewide referenda (including decisions on important environmental matters) that are elsewhere made by legislatures, or even by administrative agencies to which legislatures have delegated power. As suggested above, such decisions are more frequently made by elected officials, who sometimes not only specify which environmental risks are to be controlled, but also to what level of

residual risk and using which control technologies or techniques.

Most commonly, however, risk management decisions are made by appointed officials and the civil servants who work for them, acting under statutes that delegate to federal, state or local regulatory agencies the power to establish priorities and determine who must take actions to reduce particular risks. Their decisions can be, and generally are, informed by a variety of types of analysis, of which economic analysis is one. The IED framework is intended to help ensure that the broadest range of analyses are applied in an integrated fashion.

4.2 Fundamental Questions in the Economic Analysis of Risk

Once risks have been characterized and compared, at least two important questions remain. How much, if at all, should each of the risks be reduced, and in what order? How should these reductions be accomplished? Economics offers insights on the answers to both questions. Let us address the second question first, because it is perhaps the easier of the two. Because resources are scarce relative to human wants even in the richest of societies, economists answer the second question -- how shall risk reductions be accomplished? -- quite directly. Generally, their answer is, "In the least expensive way possible." For, by reducing as cheaply as possible the risks associated with indoor radon, heavy metals in aquatic ecosystems, or radioactive contamination in landfills, to pick but three examples, more of society's resources are available for education, national defense, health care or the private goods and services that people desire. So long as we accomplish what we set out to do (in risk reduction), why spend more than we have to?

In fact, there is a "cousin" of BCA -- called cost-effectiveness analysis (CEA)-- that deals explicitly with this question. If, and this is a big "if," the only output of a regulatory program is to reduce cancer incidence, say, CEA would rank cancer prevention programs on the basis of "cost per cancer case avoided." Then the least expensive option would be pursued first, followed by the next most attractive one, and so on until a decision had been made to no longer pursue reductions in cancer risk. The point at which this process stops is the subject of a subsequent section. This approach would maximize the reduction in cancer incidence for the amount of money spent. Other patterns of spending the same amount would prevent fewer cancers.

There are several problems with CEA that bear mention here. First, it only works for "apples vs. apples" comparisons. If one program reduces cancer cases and also improves the visibility of the air, while another reduces cancer and protects aquatic ecosystems, too, they cannot be compared on a simple cost per cancer case avoided basis. Most environmental regulatory programs have multiple outputs (that is, they provide more than one kind of beneficial effect). Thus, CEA is often not of much practical assistance in risk ranking and management. Another problem with CEA was alluded to above. While it is easy to see the common sense in starting with the cheapest control opportunities first, and working our way up the list, CEA provides no obvious "stopping rule" unless Congress, for instance, has limited the size or cost of the cleanup effort. That is, it does not tell us when we have reached a cost per cancer case avoided that is too much from society's standpoint.

Still another problem with CEA is that it treats cases of cancer -- or other endpoints -- as being homogenous, when in fact they may not be. For example, suppose the cheapest way to prevent cancer cases is to pay the cost of smokers' participation in smoking cessation programs that they would not otherwise attend. Some members of society, perhaps many, might object to such a program on the grounds that these risks are borne voluntarily -- at least more voluntarily, say, than those that arise from airborne exposures to other carcinogens. Even though the latter might be more expensive to reduce, society might prefer to address them first. In other words, we care about dimensions of risk other than sheer statistical magnitude. In addition to the voluntary or involuntary nature of the risk, these other dimensions may include such things as the degree of "dread" associated with the risk; for instance, for whatever reason, people seem to fear radiation-induced cancers more than those associated with other causes. Despite this and its other limitations, CEA has been a very useful tool in assisting regulatory officials interested in rationalizing risk reduction programs.

One final word about doing things for least cost. The attractiveness of this idea is the reason why economists and others are often enthusiastic about what have come to be called market-based or economic-incentive approaches to environmental policy. These include such things as taxes on pollution, tradable discharge permits, and deposit-refund schemes, among others. The attraction of these approaches is that they are able not only to provide incentives to meet an overall environmental goal (mercury

emissions are reduced to x tons per year, for instance), but also that the costs of complying with this limit are minimized across society. Furthermore, the burden of identifying the least cost options falls on the private sector, which is especially good at reducing costs. This is not the place to go into the reasons why this is so, though it is worth pointing out that such approaches have been used quite successfully in the United States (for the phase out of leaded gasoline in the 1980's, and for the control of sulfur dioxide in the 1990 amendments to the Clean Air Act) and in a number of countries in Europe and elsewhere around the world.

This brings us to the question of how much reduction we should seek for the various risks that society faces. There are several ways this question can be answered. One approach would be to direct the regulatory authorities to set standards for the risky pollutants so as to provide a margin of safety against adverse health effects - an approach premised on thresholds. This is the directive Congress has given to the EPA in important parts of the Clean Air Act, the Clean Water Act, the Resource Conservation and Recovery Act, and in other environmental statutes as well.

This seemingly attractive approach often founders when the underlying science suggests -- as it often does -- that there is no "safe" level; that is, a threshold concentration cannot be found that guarantees the health of affected individuals or ecosystems. This issue has arisen recently in the context of the revision of the National Ambient Air Quality Standard for ozone, and the establishment of a new standard for particulate matter of 2.5 microns and less. In both cases, the Clean Air Science Advisory Committee has indicated that no clear "bright line" can be found such that concentrations below that level can be regarded as being safe.

There is another problem with the threshold approach. It might sometimes be the case that even if a safe level could be found, society might deem it too expensive to provide, especially if a somewhat less protective level costs substantially less. A safe-standard approach denies the legitimacy of such tradeoffs. Once again, this issue has arisen in the context of air quality standard setting, with the EPA having publicly acknowledged that setting the ozone standard at too low a level could have significant adverse economic repercussions.

Another way to decide the question of how much protection to provide is to tie

the answer to affordability. That is, one could decide that any risk reductions that regulated entities (whether corporations, lower levels of government, or individuals) can "afford" are worth undertaking. This is the approach often taken when the EPA writes discharge standards for new sources under the Clean Air or Clean Water Acts. Indeed, they have been directed to do so by Congress, which has drafted statutes in which "economic achievability" is one of the relevant criteria. While this approach may seem to deal satisfactorily with all economic concerns, this is not so. First, controls on some sources might be desirable even if the source could not afford them -- in some instances, completely shutting down a plant may turn out to be in the best interests of society if there is no other alternative measure, and if that plant discharges substances that poses serious threats to human health and/or the environment.

Second, some controls may be affordable but nevertheless not worth pursuing. This would be the case for relatively trivial risks that would be expensive to ameliorate. Even if the sources of these risks were quite profitable, many people would be uncomfortable spending a lot of money on not-so-serious problems. The problem with requiring successful firms to do more than those that are just breaking even is that it creates exactly the wrong incentives for firms: do well and you will be more heavily regulated, do poorly and we'll take it easy on you. That is neither the right environmental nor the right economic signal to be sending.

There is, of course, a third way to make decisions about which risks to control and by how much. This approach involves an effort to strike a balance between, on the one hand, the good that will be done for human health and the environment when one or more risks are reduced and, on the other, the costs or other adverse consequences associated with taking such actions. The criticisms above of both the threshold and the affordability approaches go right to the heart of this matter.

Benefit-cost analysis is the technique most commonly used to assist in balancing the favorable effects of risk reductions (the benefits) with the adverse consequences (the costs). Underlying BCA is the notion that it may be possible to measure both the good and bad effects of a policy change (a measure to reduce one or more risks in this case) using a common denominator -- money. If so, one can then ascertain whether the gains to the gainers outweigh the losses to the losers, and thus determine, on net, whether society as a whole is made better or worse off as a result of the change. A

whole host of possible risk reduction policies could then be compared on the basis of which ones do the greatest net good and which do little or no net good.

We offer several observations about BCA before moving on to a discussion about how one might value benefits and costs, make comparisons among them, and take into account effects that are felt at different points in time.

First, and perhaps most importantly, most economists think of BCA as no more than a tool to assist in decision making. This is important because most non-economists (and perhaps some economists) believe that BCA must be used as a decision rule. That is, they believe that the use of BCA means that only those policy changes that pass a benefit-cost test should be put in place. To repeat, most economists do not hold this position, although all economists would argue that the information in a well-done BCA can be of great value in helping to make decisions about risk reduction policies (Arrow et al., 1996).

Second, when trying to translate benefits and costs into dollar terms, it is the preference of individuals that count. That is, we look to each individual to determine how to value a given benefit or cost.

This is often a hard pill for non-economists to swallow. Some people reason that if the public knew what they themselves know about a particular problem, surely everyone would ascribe greater benefits to solving the problem than they do currently. And sometimes it is the case that willingness to pay to reduce a particular risk is small solely because affected parties do not understand the consequences. Often, however, they understand the problem quite well, but simply do not care as much about the effects as someone else. It is unclear how one would attempt to value the consequences of a policy change to someone without attempting to determine that person's considered valuation of the change.

Another difficulty with BCA has to do with its reliance on individuals' valuations, however they might be revealed. When willingness to pay is used as the monetary value of a change, as it generally is, the values that individuals reveal are conditioned by their incomes. Thus, an individual will be willing to pay no more each year for a risk reduction than the total amount of money he/she has available after taxes. Accordingly,

the social benefit (as counted in a BCA) of reducing the risk of premature mortality for a wealthy person will generally be greater than doing so for a poor person (because the former will be able to pay more for the same statistical reduction in risk than the latter). This is a problematic aspect of BCA for almost all economists. From time to time, people have attempted to derive weights that might be used to inflate the preferences of the poor so that they count for more in BCA. The problem with this approach is obvious to all -- who shall be delegated the responsibility of assigning the weights, and on what basis should these weights be derived? The final section of this chapter takes up such questions related to the distribution of benefits and costs.

We turn now to a discussion of the benefits of risk reduction policies, and how they can be estimated and expressed in dollar terms.

4.3 The Benefits of Risk Reduction

Economic valuation of environmental resources has been characterized by some other disciplines as being anthropocentric and utilitarian, and inattentive to the intrinsic value of these resources. To some degree this is true. If an environmental change never matters in any way to any human - today or in the future - then it will not, even in principle, show up in any economic valuation or assessment. As Freeman (1997) points out, however, it is important to distinguish between ecosystem functions (e.g., photosynthesis, absorption, and dispersal) and the environmental services produced by ecosystems that are valued by humans. The range of these services is great. They include obvious environmental products such as food or fiber and services such as flood protection, but also include the quality of recreational experiences, the aesthetics of the landscape, and such desires (for whatever reasons) as the protection of marine mammals. Humans may "passively" value environmental services. For example, "existence value" reflects human recognition of the so-called "intrinsic" value of an ecosystem. If an environmental measure produces a change that matters to humans, for whatever reasons, then it is an item to be counted in the assessment, at least in principle. In practice, there are, however, numerous problems. Table 4-1 presents examples of environmental and resource service flows for which value measures might be desired (Freeman, 1993).

Policies for environmental protection often affect the functioning of ecosystems.

The social benefits of this protection depend on the link between these functions and some associated service flow that is valued by humans. In most cases an environmental protection measure will enhance environmental services marginally, and it is this change in services that economists seek to value. Before the benefits of environmental protection can be calculated, therefore, it is necessary to predict the actual consequences on natural systems from failing to engage in protective activities. Good science is essential to competent environmental benefit-cost analysis.

In some cases the physical consequences of pollution are clear-cut and easy to quantify. For example, a single "point source" of water pollution upstream renders cooling water for a downstream factory sufficiently impure that special pre-treatment equipment must be installed before the water can be used. In other cases, however, the connection is far less clear. Deteriorating air quality due to pollutants from a wide range of sources (vehicles, factories, and perhaps even natural vegetation) may reduce local average life expectancy by several months for susceptible individuals. About the most that can be claimed is that the poor air quality changes the probability that any given individual will live to be, say, at least 70 years of age. But so many other behavioral and genetic factors influence the individual's actual life-span that it is impossible to say for sure what would be any individual's actual benefits from reducing air pollution. Often, it is not even scientifically clear what effect reducing pollution from one source (say vehicles) will actually have on overall ambient pollutant levels.

In still other cases, such as the dramatic alteration of ecosystems through destruction of wetlands or through global warming, scientists are uncertain even as to the nature of the outcome. In these cases, economic valuation of potential gains from environmental protection measures falls short. This is not because benefit-cost methods ignore environmental services that do not produce market goods, but because the pathways by which humans are ultimately affected cannot be well articulated.

It is obviously very much easier to value the benefits of environmental protection when the effect is straightforward, as in the first example, but increasingly difficult as the scientific uncertainty about the physical effects increases. Even in the intermediate case, where the environmental service effects of the policy in question can be articulated no more clearly than as a change in average life expectancies or a change in the probability of respiratory disease, the task of ascertaining individuals' values of

these changes is somewhat complicated.

From an economic perspective, the environment can be viewed as a form of natural asset that provides service flows used by people in the production of things like recreation, agricultural output, the assimilation of pollutants, or even an amorphous "good" such as "quality of life." This is analogous to the manner in which real physical capital assets (e.g., factories and equipment) provide service flows used in manufacturing production. Like real physical capital, if the natural environment (as a

TABLE 4-1. Examples of Environmental and Resource Service Flows

- Resource or environmental media
  - (Air, Water, Fishery, Forest)
- Effects
  - Direct impacts on humans
    - Human health (morbidity/mortality associated with pollution)
    - Odor, visibility, visual aesthetic
  - Ecosystem impacts (biological mechanisms)
    - Impacts on the economic productivity of ecological systems
      - Agricultural productivity
      - Forestry
      - Commercial fisheries
    - Other ecosystem impacts
      - Recreational uses of ecosystems-fishing, hunting
      - Ecological diversity, stability
  - Impacts through nonliving systems
    - Materials damage, soiling, production costs
    - Weather, climate
- Economic channels
  - Market values
    - Changes in income to producers
    - Changes in the availability/price for marketed goods/ services to consumers.
  - Non-Market Values, i.e., changes in availability of:
    - Health
    - Environmental amenities
      - Visibility
      - Opportunities for recreation

Adapted from Freeman, 1993, pp. 13-14

productive asset) is allowed to deteriorate, this lessens the flow of services it is capable of providing.

A decision to devote society's scarce resources to environmental protection, however, means that these resources will not be available to be used for other purposes. These other purposes might be more food produced today or greater current manufacturing output, or it might be more investment in productive capital equipment or research to enhance output in the future, all of which are also valuable to society. It is

not surprising that poor countries spend little on environmental protection. Banning DDT, or other harmful but cheap and effective pesticides, has little appeal for people whose current battle to protect themselves from immediate illness and to feed themselves today depends on these chemicals. The concept of valuation of environmental goods and services is couched in terms of society's willingness to make trade-offs between competing uses of limited resources, and by aggregating over individuals' willingness to make these trade-offs.

Economists' tools of valuation were originally developed in a more limited context, one in which policy changes mostly caused changes in an individual's income and/or prices that he faced in the market. Over the last twenty years, however, these ideas have been extended to accommodate changes in the qualities of goods, to public goods that are "shared" by individuals, and to other nonmarket services such as environmental quality and human health. Many environmental goods and services are considered by economists to be public goods because one person's enjoyment of them does not diminish the ability of others to enjoy them also.

We can think of people as having preferences among alternative "bundles" that include both market and nonmarket goods. Some of these nonmarket goods include environmental services. Typically people's preferences allow substitution - if the quantity of one good in a bundle is decreased, the quantity of some other good would have to be increased in order to leave the individual just as well off as before the change. This substitutability between goods establishes the trade-off between pairs of goods that both matter to people. If one of the goods has a monetary value, then this willingness to make a trade-off reveals the monetary value of the other good. It is not necessary that money be the metric for comparison, it is merely convenient.

When the alternative "bundle" includes something dramatic, such as loss of life or loss of a loved one, the individual will be understandably unwilling to make trade-offs with other goods. However, environmental actions rarely, if ever, imply such dramatic outcomes with certainty. An environmental protection measure is more likely to alter slightly the probability of illness or loss of life, much like wearing a seat belt affects the probability of death, and people are observably willing to trade off money and time to alter this probability.

Rarely is it necessary to determine the value of the environment as a whole. The typical challenge is to ascertain the social value derived from maintaining some type of ecosystem function at its current level through pollution prevention or increasing its functions somewhat through more stringent regulations. Failing to implement the policy would compromise the ecosystem's functioning to some extent, thereby lessening the quantity of services the environment is capable of providing for human enjoyment.

When it is possible to observe freely functioning markets for goods whose availability is affected by policy, economists are fairly comfortable with the appropriateness of their standard tools for estimating the benefits or loss of benefits resulting from that policy. If a policy results in higher incomes or lower prices for them, people are generally made better off. Higher incomes or lower prices mean an increased difference between what the individual is willing to pay for a good and the amount that he is required to pay. This difference is called a "surplus," and modern welfare economics recognizes that welfare gains (or losses) as a result of some policy are approximated well by increases (or decreases) in this surplus that result from the policy's implementation.

The same logic still holds despite the absence of standard markets for many environmental services. Standard markets make the task easier because consumers' decisions as to how much of a good to purchase at different market prices helps reveal something about the surplus they gain at any given price. With non-market environmental goods, it is necessary to infer this willingness to trade off dollars for additional quantities of environmental services using other techniques. Environmental economists have developed a repertoire of techniques over the past thirty years that fall roughly into two categories: a) indirect measurement (revealed preference), and b) direct questioning (stated preference). The following is a brief survey.

4.3.1 Revealed Preference

Economists have always preferred to measure trade-offs by the observed decisions of consumers in real markets whenever possible. These are called "revealed preference" methods, because the consumers' actions reveal something about their willingness to trade-off one good for another. In other words, we get to see them put their money where their preferences are. The task is made easier if the researcher can

exploit relationships that might exist between the non-marketed (environmental) good and a good that has a market price. For example, suppose a change in the environmental good only matters to a particular person if he is also a purchaser of some good that has a price (e.g., water quality is important to those who buy fishing rods). Then the environmental good and this market good can be seen to be complements of sorts. The individuals' behavior with respect to the marketed good, when the environmental quality changes, will reveal something about how he values that environmental change. Likewise, if the individual can mitigate the negative consequences of the decline in some environmental service by purchasing some market good (buying bottled water when aquifers are contaminated, for instance), then the environmental good and the market good are substitutes of some sort. We can learn something about the lost benefits of the environmental service from these individuals' actions with regard to this substitute market good.

First, the *Travel Cost Method* (TCM) is perhaps the oldest of these revealed preferences methods for non-market valuation and relies on the complementary relationship. An individual will decide whether to take a trip (generally a recreational trip) to consume an environmental good if the perceived benefits of the visit exceed the costs of the visit. In this case, environmental service might be viewed as a quality characteristic of the trip that matters only if the individual visits the relevant site. By observing how people trade off such things as distance traveled to access sites of better environmental quality, researchers can learn something about the value people place on differences in this quality.

Second, economic methods that model individuals' *decisions to avert or mitigate* the consequences of environmental deterioration may shed light on how people value other types of changes in environmental quality. These techniques are most applicable where the failure to institute a given policy would increase risk of illness or loss of life, although they have other applications as well. For example, if a household installs filter systems or purchases bottled water to avoid contaminants in the groundwater that they drink, then they are attempting to avert or mitigate the consequences of water quality deterioration. By analyzing this type of behavior, economists can at the least establish bounds on the willingness of individuals to pay for improved water quality. In this case, bottled water is a substitute for clean groundwater in altering the risk of illness from drinking water.

Third, in some cases, individuals reveal their preferences for environmental goods in the *housing market.* Realtors know that the price paid for a property varies with the characteristics of that property. Additional bathrooms or bedrooms or a better view will show up in the transaction price of a dwelling, but local environmental characteristics will show up as well. For example, researchers have found that housing prices vary with air quality in many cities. Home-buyers' willingness to pay for this additional environmental quality reveal something about their willingness to trade-off money for this type of environmental improvement. This approach is called the *hedonic property value method* of valuation.

Fourth, *wages* for similar jobs in different locations may vary with the local level of environmental quality. Higher wages may have to be paid to induce workers to take jobs in areas with more pollution, greater health risks, or fewer environmental amenities. In fact, the values used by the EPA and other regulatory agencies for reductions in mortality risk come almost exclusively from studies linking compensation to the riskiness of the job. Through the *hedonic wage method*, it is sometimes possible to attribute differences in wages to differences in the quality of the environment, and thereby indirectly to value variations in environmental quality.

The above methods are well established for measuring the conceptual trade-offs that economists consider as the basis of environmental valuation. However they are applicable only in special cases. For the first method - the travel cost approach -- to be complete, individuals must care about changes in the environmental good only if they visit a particular site. For the second to be complete, (that dealing with complementary purchases), individuals' values for changes in environmental goods must depend solely on their ability to produce some ultimate service which can be produced as well using market goods, but at some increased cost. In all of these methods, individuals must be able to discern the environmental consequences and the researcher must be able to find a marketed good with a special relationship to the environmental good that can be exploited. These conditions hold only in a subset of the cases in which environmental protection measures need to be evaluated.

4.3.2 Stated Preference

Economists generally mistrust trade-off information that has not been observed

(even indirectly) in real markets, but for some changes in environmental goods and services, we simply cannot observe behavior that reveals their values to people. This is particularly true when the value is a passive one. For example, an individual may value a change in an environmental good because he wants to preserve the option of consuming it in the future (option demand) or because he desires to preserve the good for his heirs (bequest demand). Still others envision no current or future personal use by themselves or their heirs, but still wish to protect the good because they believe it should be protected or because they derive satisfaction from knowing it exists in a protected state (existence demand). With no standard market trade-offs to observe, economists often resort to surveys in which they construct hypothetical markets. Value information elicited this way is described as "stated preference" (as opposed to revealed preference) information.

In this approach (often called contingent or hypothetical valuation), survey respondents are presented with a well-defined scenario that requires them, hypothetically, to trade-off something (generally money) for a change in the environmental good or service in question. While rarely asked in such a direct way, researchers seek answers to questions such as: "How much would you be willing to pay to prevent an additional 5% of the population of sea otters from being killed due to a hazardous waste spill?" (Or, alternatively, what compensation would you require in order to willingly accept a reduction in this environmental amenity?) Early attempts to use this method were fraught with problems. As might be imagined, the values that result are susceptible to manipulation through the manner in which the valuation questions are posed. Protocols have been continuously refined, however, and the reliability of these methods is improving. (For further discussion of the problems associated with elicitation of values, see Chapter 5).

An alternative but related method is to infer values from individuals' hypothetical behavior (as opposed to hypothetical payments). This approach can be particularly useful when paying to prevent environmental degradation seems too implausible or subject to bias. Respondents might be asked "If visibility were x miles and travel costs were $y, how many trips would you take to site z?" In yet another approach (contingent choice models), respondents are given descriptions of several scenarios and asked which one they would prefer. One characteristic of each scenario is a level of prices or net income and another is environmental quality, allowing the researcher to infer

something about respondents' willingness to trade dollars for variations in the level of environmental amenities.

The types of methods described briefly here can sometimes be used in conjunction with one another, either to measure different types of values that are generated by a change in an environmental resource or to better estimate one type of value through added information. Whatever methods are used, they are least successful when the protection measure to be valued has little-understood, amorphous, or very long-term consequences. In these cases it is difficult to articulate to individuals the way in which the policy will affect society.

4.4 Costs of Environmental Protection

The task of estimating the costs of specific environmental protection efforts may seem straightforward, compared with the conceptual problems and empirical difficulties associated with estimating environmental protection benefits. To some degree, this is true. In most cases, it is easier to develop cost estimates than benefit estimates to a commensurate degree of precision and reliability. But as we move towards developing more precise and reliable cost estimates, significant conceptual and empirical issues arise.

The economist's notion of cost, or more precisely, "opportunity cost," is linked with-- but distinct from--everyday usage of the word, "cost." In this part of the chapter, we explore what this concept means in the context of environmental protection efforts, and we briefly examine the ways in which it can be made operational through quantitative, empirical analysis.

Conceptually, there are four steps required to appraise the cost of an environmental-protection measure. First, we need to identify the specific policy instrument that is associated with the measure. This is because the same target, such as a given reduction in ambient pollutant concentration, may be achieved at very different total costs with different policy instruments. For example, it is well known that under a variety of circumstances, a market-based policy instrument -- such as an emission tax or a tradeable permit system -- can enable a regulated sector to achieve an ambient target at relatively low aggregate cost, compared with a conventional

standard that has the effect of requiring all sources to adopt the same abatement technology (Baumol and Oates 1988). This example leads to the second conceptual step: identifying the specific actions that sources will take to comply with the statute or regulation as implemented with the given policy instrument. Some of these actions may involve the adoption of a new piece of equipment, but others may involve a change in process. Third, it is necessary to identify the true cost of each action, which -- as we emphasize below -- requires much more than assessing required monetary outlays in an accounting sense. Fourth, it is necessary to aggregate these costs across society and over the relevant time frame.

Economists take cost to be an indication of what must be sacrificed in order to obtain something (through purchase, exchange, or production). The concept of opportunity cost provides a measure of the value of all of the things that must be sacrificed if one or more risks are to be reduced. Opportunity costs typically do not coincide with an accountant's measure of costs, namely monetary outlays. This may simply be because out-of-pocket costs do not capture all of the explicit and implicit costs that have been incurred, such as the cost of time associated with waiting in line for a vehicle inspection. Or it may be because the monetary prices of the resources required to produce a good or service -- in our case, environmental quality -- may themselves provide inaccurate indications of the opportunity costs of those resources, because of failures by markets that result in transaction prices not fully reflecting social values. Likewise, it is important to distinguish between the private costs of some good or service to producers or consumers of that good or service, and the social costs imposed on society as a whole. The cost concept that is relevant for environmental policy analysis refers to overall social opportunity costs. Thus, the costs of environmental protection are essentially the forgone social benefits due to employing scarce resources for environmental protection purposes, instead of putting these resources to their next best use.

Thus, costs and benefits are two sides of the same coin. Environmental benefits are created by taking some environmental policy action, while other (presumably largely non-environmental) benefits are thereby foregone. Hence, in keeping with the definition of benefits established above, the cost of an environmental protection measure may be defined as the (gross) decrease in consumer and producer surpluses[1] associated with the measure and with any price and/or income changes that may result (Cropper and

Oates, 1992).

Environmental protection measures are intended to have desirable consequences, labeled "benefits," and are likely also to have undesirable consequences, labeled "costs," but there are many situations in which there may be some ambiguity regarding whether a specific regulatory consequence should be counted as an increased cost or a decreased benefit. Take, for example, an environmental regulation that provides direct protection of human health (benefits) and requires expenditures of capital and labor (costs) to achieve this purpose. But the positive health impacts of the regulation may have the effect of increasing labor productivity, thereby decreasing the costs of producing other goods and services. Is this a decrease in the costs of the regulation or an increase in its benefits? To avoid this sort of confusion, economists argue for the use of net benefits (benefits minus costs), as opposed to benefit-cost ratios as an evaluative criteria (see below).

With these definitions in mind, we provide in Table 4-2 a view of the costs of environmental protection measures, beginning with the most obvious and moving towards the least direct.[2] First, many policy makers and much of the general public would identify the on-budget costs to government of administering (monitoring and enforcing) environmental laws and regulations as the cost of environmental regulation. Most economic analysts, on the other hand, would identify the capital and operating expenditures associated with regulatory compliance as a substantial portion of the overall costs of regulation, although a considerable share of compliance costs for some regulations fall on governments rather than private firms -- a good example being the regulation of contaminants in drinking water, a Federal regulation, the cost of which is borne primarily by municipal governments. Additional direct costs include legal and other transaction costs, the effects of refocused management attention, and the possibility of disrupted production.

Next, there are what have sometimes been called "negative costs" (in our conceptual framework, non-environmental benefits) of environmental regulation, including the beneficial productivity impacts of a cleaner environment and the potential innovation-stimulating effects of regulation.[3] "General equilibrium" or multi-market effects associated with product substitution, discouraged investment[4], and retarded innovation constitute another important layer of costs[5], as do the transition costs of

TABLE 4-2. Costs of Environmental Regulation

Government Administration of Environmental Statutes and Regulations
Monitoring
Enforcement
Private Sector Compliance Expenditures
Capital
Operating
Other Direct Costs
Legal and Other Transactional
Shifted Management Focus
Disrupted Production
General Equilibrium Effects
Product Substitution
Discouraged Investment
Retarded Innovation
Transition Costs
Temporary Unemployment
Obsolete Capital
Social Impacts
Loss and Change of Jobs for Some Workers
Economic Security Impacts

Source: Jaffe et al., 1995

real-world economies responding over time to regulatory changes. Finally, there are other potential social impacts that are given substantial weight in political forums, including those involving jobs and economic security.

Within the category of direct compliance costs, business expenditures for pollution abatement in the United States represent, on average, about 61 percent of total direct abatement costs (Rutledge and Leonard, 1992); personal consumption abatement, 11 percent; government abatement, 23 percent; government regulation and monitoring, 2 percent; and research and development, 3 percent. These averages must be taken with the appropriate grain of salt, since -- as suggested earlier --

measuring even direct costs is by no means a trivial undertaking. For example, there is significant variation between cost estimates produced by the U.S. Environmental Protection Agency (1990) and the U.S. Department of Commerce (1993) for these activities.

There are a number of potential problems of interpretation associated with any cost data. For example, the questionnaire used by the U.S. Department of Commerce (1993) to collect data for its Pollution Abatement Costs and Expenditures (PACE) survey[6] asked corporate and government officials how capital expenditures compare to what they would have been in the absence of environmental regulations. This creates two problems.[7] The first involves the determination of an appropriate baseline. Absent any regulation, firms might still engage in some -- perhaps a great deal of -- pollution control to limit tort liability, stay on good terms with communities in which they are located, maintain a good environmental image, etc. Should such expenditures be included or excluded in the no-regulation baseline?

Second, when additional capital expenditures are made for end-of-the-pipe abatement equipment, respondents have relatively little difficulty in calculating these expenditures. But when new capital equipment is installed, which has the effect of both reducing emissions and improving the final product or enhancing the efficiency with which it is produced, it is far more difficult to calculate how much of the expenditures are attributable to environmental standards.[8]

In the give-and-take of environmental policy debates, it has frequently been the case that abatement costs of proposed regulations have been over-estimated (Hammitt, 1997). This may be due partly to the adversarial nature of the environmental policy process, but it is also a natural consequence of employing short-term cost analyses that do not take into account potential, future cost savings due to technological change, some of which may be endogenous to the regulatory regime.

As we said at the outset, the task of estimating the costs of environmental protection efforts is relatively straightforward, at least compared with that of estimating environmental protection benefits, but producing high-quality cost estimates still requires careful analysis.

4.5 Comparing Total Benefits and Total Costs

After the streams of benefits and costs associated with a pollution control program have been monetized, benefit-cost analysis involves evaluation according to the discounted sum of the net benefits it provides. This section discusses how net benefits are computed, and then aggregated over time. It also discusses in greater detail what it means for a program to "pass a benefit-cost test."

4.5.1 Calculating Net Benefits

The net benefits of a program in a given year (year "t") are the difference between the total benefits of the program in that year and its total costs in that year. In the case in which most of the costs of the program are incurred in the early years, possibly in the form of capital investment, while the benefits of the program are spread over time, net benefits may initially be negative, but eventually become positive. In a case where the annual costs and benefits of the program are constant over time, net benefits will also be constant.

Two caveats are in order. When net benefits are computed in each year, it is customary to express them in constant dollars (in real terms). That is, any increases in costs and benefits due to inflation are subtracted out, so that costs and benefits in each year are expressed in, for example, 1997 dollars. A second caveat concerns comparing constant dollars at different points in time. When adding the dollar value of net benefits over several years, it is important to realize that people are not indifferent between receiving a dollar in 1997 and a dollar in the year 2007. The dollar received in the year 1997, if invested at 5%, will grow to $\$(1.05)^{10} = \$1.63$, by the year 2007. Equivalently, a dollar received in the year 2007 is worth only $\$1/(1.05)^{10} = 61¢$ today, since this is all one need invest today to obtain $1 in the year 2007. For this reason, economists express all future net benefits in terms of the present year's dollars; that is, they compute the present value of net benefits in each year before aggregating net benefits.

4.5.2 Present Values and Discounting

The present value of a dollar of net benefits occurring 10 years from today is the

amount that must be invested today to yield one dollar in ten years. Generally, the present discounted value of $1 received in t years is $1/(1+r)^t$, where "r" is the annual social rate of discount.

How should the annual social rate of discount ("r") be determined? Many economists would argue that "r" should represent the real rate of return on a riskless capital investment. By real return, we mean the return actually paid minus the rate of inflation. To illustrate, in 1980 the rate paid on a Treasury-Bill was about 11.6%, but the rate of inflation (as measured by the GDP deflator) averaged 9.4%. The real return on government bonds was therefore 2.2%. Since we assume that benefits and costs will be measured in real terms (adjusted for the rate of inflation), discount rates should be measured in real terms also. The reason for using a riskless rate of return (such as the rate of return on Treasury-Bills) is that the rate of return applied to public projects should not include a risk premium. This is because public investments are financed by all members of society, thus spreading the risk of the investment.

There is, however, another way to estimate the social discount rate. This is to say that the costs and benefits in year "t" should be discounted to the present by a consumption discount factor - the rate at which people are willing to trade consumption in year "t" for consumption today. In a world of perfect markets, the consumption discount factor should equal the discount factor $1/(1+r)^t$ where "r" is the return on capital investment. There are, however, cases where the two may differ, and this leads to complications.

For simplicity, suppose that we measure the annual social rate of discount ("r") by the real rate of return on riskless investments. In the U.S., this has ranged historically from 0% to 4%. This suggests that the appropriate social discount rate lies in this interval. Whether a rate of 0% or 4% is used, however, will often make a great deal of difference to the magnitude of the present value of net benefits. A higher discount rate, other things equal, will reduce the net present value of a program. A discount rate of 2% for example, implies that one dollar received in 20 years is worth 67¢ today, whereas it is worth only 46¢ at a discount rate of 4%. Furthermore, the discounted sum of the net benefits of a program will be more sensitive to changes in the discount rate the longer the horizon over which benefits and costs extend and the greater the disparity in the time profiles of benefits and costs.

While the concept of discounting has a sound rationale, it can lead to conclusions that many people, including economists, find unpalatable. Even at a discount rate of 2% per annum, a dollar of benefits received 200 years from now counts for less than 2¢ today. This leads to the conclusion that the present value of net benefits will sometimes be negative for projects whose costs are concentrated in the present but whose benefits stretch far into the future. To avoid rejecting such future-oriented programs, some people have called for not discounting future costs and benefits at all, especially for projects with long horizons. At first, this might seem to be a course of action that would favor future generations. In an important sense, however, it does not. If, by using a zero discount rate, we adopt programs that do not pay off until the distant future, we are implicitly passing up opportunities to invest in other projects that yield higher rates of return. Since these projects may increase the capital available to future generations, it is not clear that we have made them better off by using a zero discount rate.

4.5.3 The Net Benefit Criterion and the Scope of Public Projects

Suppose that we have calculated the present value of the net benefits of an environmental project in each year of the project's life. Adding these together will produce the discounted sum of net benefits of the project.[9] What can be done with this information? As stated at the beginning of this section, positive net benefits imply that, at its current level, the project increases social welfare, in the sense that the gainers could, in principle, reimburse the losers and still have something left over for themselves (see Section 4.5.I). This does not, however, mean that the size of the project is optimal. Figure 4-1 shows how the present value of the marginal benefits and marginal costs of an environmental project vary with the scope of the project. In the illustration, the scope of the project is the reduction in pollutant emissions from a given baseline. The optimal reduction in emissions, according to the diagram, is "A" -- the point where the increase in benefits (or marginal benefits) from increasing the percent reduction, just equal the additional cost of the reduction. Beyond this point, the additional cost of further reductions exceeds the marginal benefits of the reductions.

The total benefits of a given pollutant reduction, however, exceed the total costs of the reduction at program levels that are much larger than optimal. The total benefits of the reduction -- the area under the marginal benefit curve -- equal or exceed the total

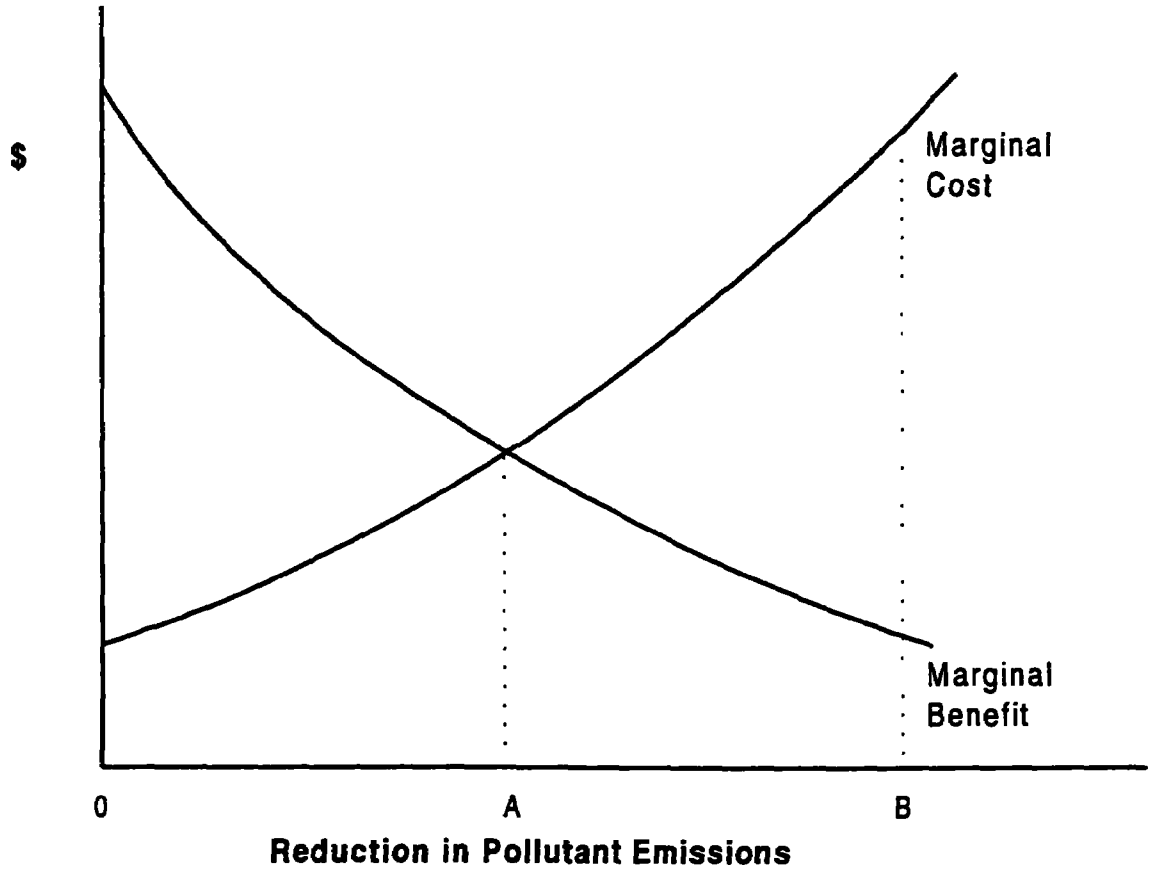

Figure 4-1. Marginal costs and marginal benefits associated with an environmental project.

costs of the reduction -- the area under the marginal cost curve -- for all reductions from 0 to "B". Thus, even programs that propose reductions greater than optimal still pass a benefit-cost test.

The lesson of Figure 4-1 is that a program that passes a benefit-cost test at its current level of implementation may still be suboptimal, in the sense that a similar program at a different level of implementation could yield larger net benefits. A corollary to this result is that a program that consists of many components, for example, a regulation that covers many sources, may, overall, pass the benefit-cost test, even though individual components of the program (regulations on individual sources) may

not pass the test. In the context of IED, one objective of integrated options analysis is to improve net benefits by careful selection of risks to be addressed and risk reduction options to be pursued.

4.5.4 Uncertainty in the Measurement of Benefits and Costs

The state of the art of measurement of the benefit and costs of reducing risks is not sufficiently developed to produce exact measures of economic value. This leads to the question, must policy makers wait for further research to produce exact measures before they can use benefit and cost information as an aid in decision making? If not, how should they interpret the ranges of values that current research has produced?

To counsel waiting for exact measures is equivalent to saying that in many cases value measures should never be used. The state of the art cannot be expected to advance to the point of producing exact values for all kinds of environmental change. This is because of the inherent uncertainty and imprecision in measurement techniques, and because of uncertainties about which models are appropriate in specific circumstances. So how are policy makers to proceed in the face of continued and inherent uncertainty about economic values?

A simple approach with a basic intuitive appeal is to perform sets of calculations with the upper and lower bounds of the range, and perhaps with the midpoint of the range as well. This is the essence of the approach for valuing reductions in mortality risks outlined in the EPA guidelines for performing regulatory impact analyses (U.S. EPA, 1983).

Clearly, if the benefits of a policy calculated with the upper end of the ranges are less than the lower end of the range of estimated costs, the policy is unlikely to be justifiable on economic grounds; and if the benefits calculated with the lower end of the range exceed the upper end of the range of costs, the economic case for the policy is quite strong. This simple-minded approach is a step in the right direction; but it does not make use of all of the relevant information contained in the set of available estimates. This range reflects only the information contained in the two estimates yielding the highest and lowest values; it ignores the information on the quality of these two estimates; and it ignores the information contained in the other estimates that yield

intermediate values.

It is possible to make use of all available estimates and to incorporate judgments about the quality of each of these estimates. This can be done by viewing probabilities as statements about the degree of confidence held about the occurrence of some possible event. The approach involves assigning probabilities to all of the values produced by the available estimates, where a higher probability reflects a greater degree of confidence in that estimate. For example, the assignment of a probability of unity to a particular estimate means we are certain that this study has produced the correct value. Once the probabilities have been assigned, various useful summary statistics can be generated. For example, the expected value of the parameter in question (the probability distribution) can be calculated and used for benefit-cost calculations. The variance of the distribution can be used to determine confidence intervals on the value to be used, thus preserving for policy makers information on the uncertainty about economic values.

4.6 Distributional Considerations

This discussion of benefits and costs, as well as the way the two are compared, glosses over an important point -- one that has been raised increasingly in discussions about which risks the government ought to be addressing. Specifically, BCA is silent about the distributional implications of risk reduction measures. This has given rise to concerns about the absence of environmental equity or environmental justice in the decision making processes of regulatory agencies.

To illustrate this point, suppose hypothetically that emissions from a series of small auto paint shops in Los Angeles were resulting in slight increases in ambient ozone levels in Beverly Hills. This in turn was slightly increasing the risks of eye irritation among wealthy joggers there. Because the latter have very high incomes, the calculated benefit from reducing this risk is very great -- the joggers would be willing to pay substantial amounts to be free of this annoyance. Suppose further that closing the paint shop, and in the process rendering unemployed some of the unskilled people who work there, is the only way to reduce the risk -- control equipment could not be afforded. In this case, the calculated benefits of the risk reduction could be well in excess of the costs, but because all the benefits go to the wealthy while the costs are borne by the

poor, one might be quite uncomfortable with enacting the controls.

In other situations, the situation is reversed. That is, the physical benefits of a risk reduction program would go to a poor population -- say, those who live around a Superfund site -- while the costs of the cleanup would be shared much more broadly and, in general, by those with higher incomes. But because the willingness to pay of those affected by the site is quite low, the benefits as calculated in a BCA would be small relative to the costs. Thus, such a cleanup might fail a benefit-cost test even though it would have what many would regard as favorable distributional consequences. Simply put, BCA counts a dollar's worth of benefit or cost to a wealthy person the same as a dollar's worth of benefit to a pauper. This can result in some efficient programs (programs for which the net benefits are positive) having unattractive features. Economists have long acknowledged that efficiency and equity are not always compatible objectives.

In the introductory section, we mentioned this concern with BCA, and suggested that much thought has been given over the years to the incorporation of distributional weights for BCA that would attempt to incorporate such considerations into determinations of efficiency. We believe this is a poor idea, not because we are unconcerned with the distribution of income, but rather because we see no way to derive any kind of consensus on what the weights should be. It seems better to estimate the benefits and costs as best we can, and also provide as much information as possible about who will gain and who will lose, and let decision makers take distributional considerations into account in whatever way they see fit.

4.7 Conclusions

What are we to conclude from this discussion? Several points bear repeating. First, and perhaps most importantly, knowing the risks that arise from current and future exposures to pollutants in the environment is certainly important, but it is an insufficient basis, on its own, for planning future regulatory or other forms of action. That is, it does not naturally follow that the best thing to do once one knows which risks pose the biggest threats to human health or the environment is to take actions to reduce the biggest risks first. While perhaps superficially counterintuitive, it may make much more sense to attack problems farther down on the list.

This is because a more sensible basis for action is asking which problems most lend themselves to amelioration. That is, we ought to ask where the resources we have available for environmental improvement (be they financial, human, or other) can best be deployed. This in turn requires that we know how much risk reduction we can get and for how much money in each of the risk categories. This is no more than the environmental application of "the biggest bang for the buck" aphorism that originated when analytical thinking was applied to defense spending.

Although economics can provide only parts of the answer to the question, "How can we best spend our environmental dollars?", those are important parts nonetheless. Economics can tell us, for instance, how much it would cost to attempt to reduce average ambient ozone concentrations from 0.10 parts per million to 0.08 parts per million. It can tell us how much it might cost to reduce heavy metal deposition into aquatic ecosystems by 30 percent. And it can tell us what we could expect to spend to reduce pesticide residues on fresh fruits and vegetables.

Economics can do more than inform us about the costs of various types of control actions. It can also illuminate the value that individuals place on these and other types of risk reductions, and how these values compare to the costs of taking actions. This latter role of economics--benefit-cost analysis--is controversial, poorly understood, and often misused. This is no reason, however, to throw the baby out with the proverbial bathwater. If carefully used, with its basic assumptions clearly and openly described, benefit-cost analysis can be an invaluable aid to informed decision making.

If economics is used carefully, it can contribute significantly to the nation's environmental protection efforts. Specifically, along with quantitative risk assessment and other analytical tools, it can help us identify places where we can accomplish a lot for a little, and avoid spending a great deal to do relatively little good. It is harder making decisions in cases where both the benefits and the costs seem large, or those cases where both are on the small side. Here, finer distinctions must be drawn. This means in turn that the assumptions that underlie a benefit-cost assessment may make the difference between a favorable and an unfavorable outcome. So long as those assumptions are made explicit, however, and so long as distributional and other

considerations are also factored into decision making, BCA can be a most valuable input into the formulation of an appropriate environmental policy.

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Endnotes

[1]See definitions of consumer surplus and producer surplus, above, in our discussion of environmental benefits.

[2]For a useful decomposition and analysis of the full costs of environmental regulation, see: Schmalensee (1994).

[3]The notion that environmental regulation can foster economic growth is a controversial one among economists. For a debate about this proposition, see: Porter and van der Linde, 1995; and Palmer et al., 1995.

[4]For example, if a firm chooses to close a plant because of a new regulation (rather than installing expensive control equipment), this would be counted as zero cost in narrow compliance-cost estimates, but it is obviously a real cost.

[5]Note that it is extremely difficult to measure "retarded innovation," because the degree of innovation that would have occurred in the absence of the environmental-protection measure is essentially unobservable.

[6]For over twenty years, the PACE survey provided a unique source of information on pollution control costs, since it was the only annual, representative survey to cover all of U.S. manufacturing. In 1996, the collection of data was terminated for budgetary reasons. Given that the annual costs of the survey were approximately $1 million, this decision itself ought to be questioned on benefit-cost grounds. In April 1999, EPA announced that it would reinstitute the annual PACE survey with the Bureau of the Census.

[7]For detailed discussion of environmental compliance cost measurement problems, see U.S. Congressional Budget Office (1985).

[8]An irony of the movement of environmental policy towards more reliance on performance standards (both uniform and market-based) and less on end-of-pipe technological standards is that it is becoming increasingly difficult to measure even the narrowest notion of pollution abatement costs (Jaffe et al., 1995).

[9] Formally, if bt denotes benefits in year t and ct costs, the discounted sum of net benefits over a horizon of T years is given by:

$$\sum_{t=0}^{T} (bt-ct)/(1+r)t$$

CHAPTER 5. ASSESSING THE VALUE OF NATURAL RESOURCES

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CHAPTER 5. ASSESSING THE VALUE OF NATURAL RESOURCES

OVERVIEW

In its 1990 report, *Reducing Risk: Setting Priorities and Strategies for Environmental Protection*, the SAB recommended that the Agency "develop improved analytical methods to value natural resources and to account for long-term environmental effects in its economic analyses" because "traditional methods of economic analysis tend to undervalue ecological resources and fail to treat adequately questions of intergenerational equity." During 1996, Deputy Administrator Fred Hansen urged the SAB to address the need for improved methods of measuring environmental benefits.

The Valuation Subcommittee was established as a multidisciplinary group to act on this request. The specific charge was for the Subcommittee to define better the full range of relevant questions that must be considered in ecological and human health valuation. It was further to identify conditions where existing economic methodologies seem not to address or monetize adequately ecological endpoints that may be important contributors to the value society places on ecological resources. The Subcommittee was challenged to use its broad diversity of expertise and perspective to scope out a more complete framework for valuation and to identify the types of methodological developments or research needed to implement the framework. Although it was requested to do so in the charge, the Subcommittee was not able fully to consider valuation of human health issues, and the Subcommittee's conclusions may not apply directly to human health.

The group discussed the charge and worked on its response in public meetings on two separate occasions. Also, in a three-day workshop during April, 1997 the concept of environmental valuation was discussed from five perspectives: 1) the environmental management decision context for valuation; 2) the nature of value and values; 3) the economic concept of value; 4) the importance of deliberative processes to valuation; and 5) additional approaches to environmental valuation.

Many stakeholders feel that existing economic analysis methods undervalue ecological resources because they miss important issues. The Subcommittee agrees that valuation of ecological services is a complex process and a still-evolving field of research and that improvements in both approach and practice are needed. This report discusses the components of the process that should be considered in valuation efforts. It concludes that there is no single approach to valuation that can be used in all situations. The Subcommittee's work confirms that an interdisciplinary perspective is required to meet the challenges and complexities of valuation.

One of the basic premises of welfare economics is that economic values are based on individuals' preferences and that people know their preferences. Where individuals are ignorant of the roles of ecological functions in contributing to valued service flows, it may be necessary to use experts' knowledge of the functioning of environmental systems as an input in the valuation process. In principle, the economic valuation framework can be utilized to define and measure the economic values of changes in the functions and services of environmental systems that affect individuals' welfare either directly or indirectly. However, this framework may be difficult to implement in practice where the relationship between the function and the service flow to individuals is indirect or subtle. Economic approaches to valuation are not mechanisms for producing "the answer" since they may be incomplete, may include some elements which are difficult or impossible to estimate, and may employ preference elicitation processes that are incomplete.

Not all benefits or costs can be easily quantified, much less translated into dollar terms. There is a need for qualitative methods to help decision makers understand the hard-to-define values that are important in finding solutions to complex ecosystem problems. When integrating the results from different methods, care must be taken to assure that quantitative factors do not dominate important qualitative factors. To be most useful, valuation issues and approaches should be made as explicit as possible and should involve assembling the appropriate available information, clearly stating the assumptions and uncertainties, and ensuring that the application of methods is transparent. These analyses are best used to inform but not dictate, decisions related to environmental protection policies, programs, and research.

There is much uncertainty in our knowledge and understanding of many of the factors necessary in the environmental decision-making process. Uncertainty will persist, yet we cannot wait for certainty in our analytical inputs to make decisions. The process of adaptive management, discussed in this report, is attractive because it allows decisions to be made and management to proceed in the face of uncertainty. In doing so, it allows for feedback from experience gained during implementation of environmental management practices, the development of new knowledge by the research community, and for revisiting and revising past decisions on the basis of the new insights provided by this research and experience.

General themes that emerged during the Subcommittee's discussions included:

1) For decision-making purposes in a governmental context, ecological valuation is an anthropocentric exercise (people's wishes count; there is no external set of values waiting to be discovered for application to decision-making).

2) The value of anything reflects its contribution toward the achievement of some goal. The process of valuation cannot be separated from the need to reach agreement on goals.

3) Environmental valuation requires a diverse and interdisciplinary process involving interaction and deliberation among scientists, decision makers, and other stakeholders to identify goals and to define endpoints to characterize those goals.

4) Existing economic approaches, broadly considered, are consistent and coherent frameworks for valuation because they organize a system of trade-offs. However, they are not mechanisms for producing "the answer" because they may omit trans-economic values that may be important, may include some elements that are difficult or impossible to estimate, and may employ preference elicitation processes that are incomplete.

5) An expanded, rich, and complex process using multiple approaches is required to fully encompass ecological valuation.

The Subcommittee's work confirms the challenges and complexities of environmental valuation exercises, and the resultant environmental management actions based on those values. The Subcommittee recommends that expanded, rich, and complex processes be employed to fully characterize environmental values. This process will involve interaction and deliberation among scientists, decision makers, and other stakeholders in order to identify goals, define endpoints to characterize those goals, and to implement approaches to achieve those goals. This process must be one of on-going dialogue and adjustment and it should consider: 1) why people care for the things they do--their preferences; 2) the appropriate use of deliberative processes to elicit preferences and the rationale for them; 3) economic valuation frameworks to define and measure the economic value of changes in environmental systems functions and services affecting individual welfare; and 4) presentation of available physical or other quantitative measures, or qualitative descriptions of the effects of alternative actions when costs and benefits are not fully captured by monetary measures. The Subcommittee recognizes that environmental valuation remains a craft embedded in political processes.

5.1 Introduction

5.1.1 Background

In its 1990 report, *Reducing Risk: Setting Priorities and Strategies for Environmental Protection*, the U.S. Environmental Protection Agency's (EPA) Science Advisory Board (SAB) recommended that the Agency "develop improved analytical methods to value natural resources and to account for long-term environmental effects in its economic analyses." This recommendation was based on the view that "traditional methods of economic analysis tend to undervalue ecological resources and fail to treat adequately questions of intergenerational equity." In February 1996, Deputy Administrator Fred Hansen met with the SAB Executive Committee and urged the SAB to address the need for improved methods of measuring environmental benefits as part of its project to update *Reducing Risk*.

In response, the Executive Committee agreed to establish a Valuation Subcommittee as part of the Integrated Risk Project. The Valuation Subcommittee's deliberations were intended to complement those of the Economic Analysis

Subcommittee so that the final Integrated Risk Project report would include information relevant to the economic assessment of the benefits and costs of various risk reduction options, as well as recommendations for improving future assessments of the value of ecological resources to society.

5.1.2 Objectives and Approach

The Valuation Subcommittee was charged to define better the full range of relevant questions that must be considered in ecological and human health valuation. The prevailing feeling was that existing economic methodologies, which are used to quantify the monetary value of certain ecological resources, do not address or monetize adequately important ecological endpoints (ecological processes, environmental structures, and many other characteristics) that may be important contributors to the value society places on ecological resources (e.g., intergenerational issues, stewardship issues, sustainability options, and time scale issues). Similarly, many environmentally mediated quality of life issues of great importance to people and society are difficult to quantify (e.g., aesthetic, culture, religion, security, equity). Consequently, a new look at the overall approach to valuation was requested: one that considered the utility of current economic methodologies as a point-of-departure, but which went beyond these and identified other approaches to supplement them and provide a more comprehensive basis for valuing ecological systems and their relationships to human health and quality of life. The Subcommittee was challenged to use its broad diversity of expertise and perspective to scope out a more complete framework for valuation and to identify the types of methodological developments or research needed to implement the framework.

In order to meet this charge, a diverse group was assembled with individuals from disciplines such as public policy, economics, ecology, philosophy, communications, psychology and sociology. Institutional affiliations represented in the group were also diverse; for example, members came from academia, industry, independent research institutions, and consulting firms. Some members were formerly associated with government organizations.

The Subcommittee discussed the charge and planned for its response in public meetings on two separate occasions. These meetings led to a three-day workshop in

Baltimore, Maryland during April, 1997. At this workshop, the members were assigned to one of five panels and led Subcommittee discussions about valuation from five perspectives. The discussions and recommendations resulting from these activities are the basis for this chapter.

The Subcommittee discussions fell into the following broad areas: 1) the environmental management decision context for valuation; 2) the nature of value and values; 3) the economic concept of value; 4) the importance of deliberative processes for eliciting values and the goals on which they are predicated; and 5) additional approaches to environmental valuation. Although requested to do so in the charge, the Subcommittee was not able to consider valuation of human health issues. Consequently the Subcommittee's conclusions may not apply directly to human health.

The focus of Subcommittee activity was the consideration of alternatives and enhancements to economic analysis techniques that would allow the Agency to characterize the benefits and costs of environmental protection actions more comprehensively and accurately and to ensure that environmental decision-making takes account of the values that people attach to ecological systems, and system functions and components.

5.2 Valuation and the Decision Context

The need to conduct environmental valuation exercises arises because people, as they exercise their prerogative of free choice, sometimes carry out activities that result in changes to environmental systems that can be detrimental. For some of these situations, society has established laws to control the behavior that leads to the detrimental effects. Government agents are entrusted with the responsibility and authority to evaluate the detrimental changes that meet or exceed triggering-criteria contained in these laws, and to decide whether action is needed to prevent or control these changes. These decisions require, in turn, a basis for comparing alternative outcomes, and this leads to the need to place a value on the ecological changes envisioned.

5.2.1 The Valuation Process in Regulation

When a statutory criterion is triggered or a problem is otherwise identified, a chain of activity is initiated that results in a decision on the need for, and if appropriate, the type of intervention. As discussed in previous chapters, the IED framework provides a way of considering information (both data and professional judgments) on risks, control options and practices, economic impacts, and social and quality of life issues. Equally important is the consideration of the value ascribed to alternative outcomes. Methods to estimate such values are a critical component of the IED framework.

As discussed in Chapter 4, value is routinely assessed during the environmental management decision-making process using economic analysis techniques. The usual approach is to apply economic analyses to estimate the costs and benefits of each of a suite of potential environmental management options. The costs and benefits are then compared to determine the net benefit associated with each option, and this information, along with other decision criteria specified by the decision-maker, serves as the basis for deciding whether to take action, and if so, on the most suitable action to implement.

Figure 5-1 reflects a prototypical relationship between increasing degrees of environmental stressor control and the societal costs and benefits that might be associated with a specific control option. The relationship demonstrated in the figure suggests that at first, toward the origin, the initial marginal costs of control are minimal but gradually increase because a higher degree of effort is needed to bring about increasing reduction in the stressor. At the same time, the associated marginal benefits are high with the initial reduction in the stressor but decline with additional reduction. The key point shown by the figure is that typical economic analysis techniques, which compare costs and benefits of control options (usually in dollar terms) reveal a point in the stressor reduction process at which the associated costs of additional control are not "worth it" because the benefits gained are less than those costs. Further reductions in the stressor will bring benefits, but these are won only at costs (expenditures of resources with alternative uses) that on net, in the judgment of the party doing the evaluation, make the overall outcome worse. In a public policy context, the government decision-maker stands as the agent for the people as a whole and acts on the

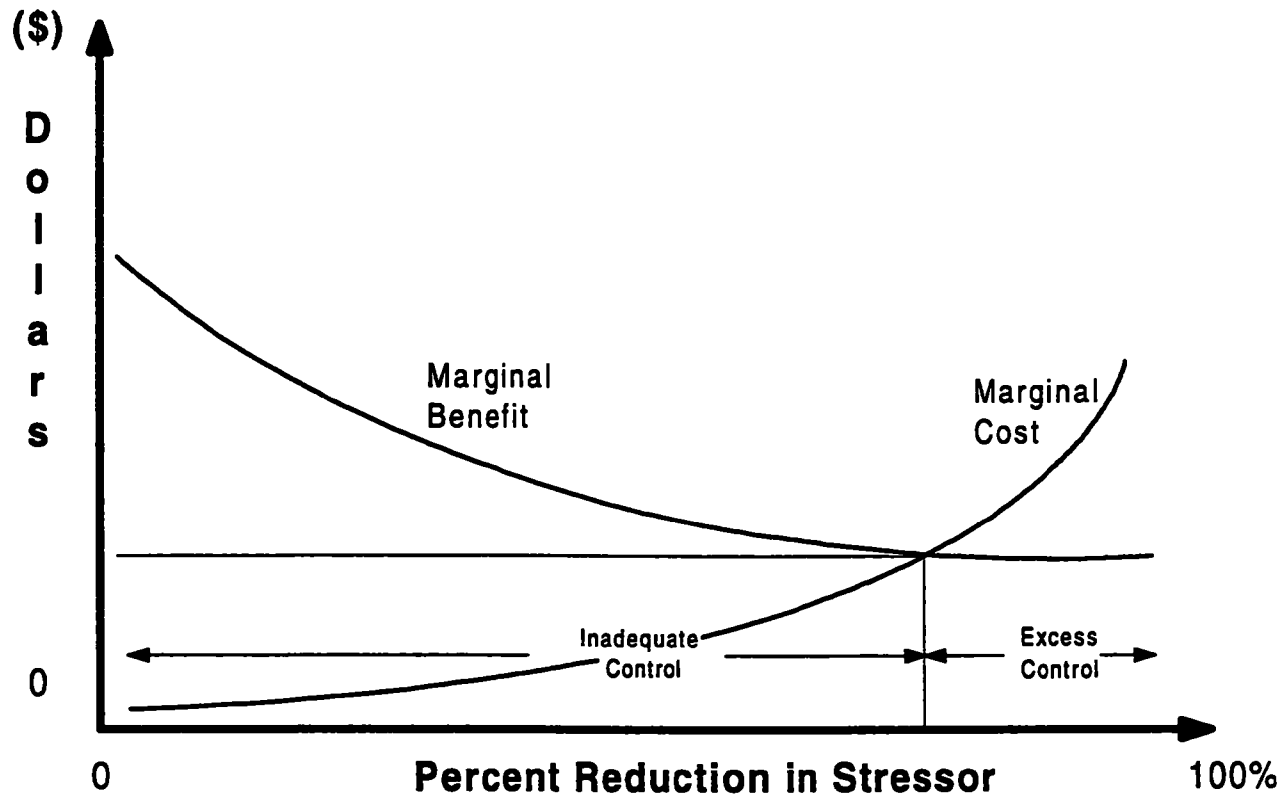

Figure 5-1. Typical relationship of marginal costs and marginal benefits associated with a proposed environmental control option.

valuations that they place on stressor reduction and on the associated use of resources to accomplish it. An example is the distribution of the heavy metal, lead in the terrestrial environment at a hazardous waste site. Initially high concentrations of lead pose significant environmental risks which can be sharply reduced during a site clean up program. These initial reductions in lead concentrations can be achieved at relatively low costs; however, a point of excess control can be reached where the costs of additional lead removal exceed the marginal benefits and associated reduction in risks realized.

While accepting the stylized approach to trade-offs embodied in this diagram, the Subcommittee noted that its effectiveness in practice as a guide to action depended on

the completeness and correctness of the elements included in the calculation of the prototypical curves. As illustrated in the discussion below, with respect to completeness, the Subcommittee found that in practice some elements valued by people could be systematically omitted, others could be poorly quantified, and others might evolve only in a social context and not surface from commonly used individual preference measures. With respect to correctness, the Subcommittee noted that selection of the elements to be valued was determined by the social goals presumed to be relevant, and that those goals were a complex mixture that included matters sometimes overlooked such as fairness, long-term sustainability, and stewardship. Further, the Subcommittee noted that in practice, estimation of the values that individuals placed on these elements is subject to error, and that aggregation of such values involves methodological uncertainties. These considerations formed the basis for Subcommittee discussions of enhancements to economic analysis techniques to allow a more comprehensive and accurate characterization of environmental values. The results of the discussions are captured in the sections that follow.

5.2.2 The Decision Context

Many environmental regulations have been positively received by the public and these actions have produced noticeable improvements in the quality of our air, land, and waters. In addition, the regulation of certain chemical products and residuals have brought improvements in human health and the state of certain wild species. However, some environmental decisions have met with concern and conflict and have led to an erosion of trust in governmental institutions. Common criticisms of environmental decisions are that they are yielding inadequate returns for the resources used, often focus on perceived, rather than actual, risks, and define problems too narrowly (e.g., see Sexton, 1996). In addition, there is a growing sense that decision-making should be more open, allowing greater understanding and participation by stakeholders and the public.

In considering the source of controversy in environmental management decision-making, Brown (1997) suggested that "...disputes about environmental policy, while often seeming to be about the facts, are--at least as much--about an underlying value framework that legitimates a particular decision." Dietz et al. (1989) noted that participants in risk policy debates differ in their perceptions about the role of facts and

values in such conflicts. The norms and goals of society, in effect, provide the dominant contextual grounding for environmental decision-making.

The values that people place on certain health and environmental states, conceptions of their rights and obligations, and outcomes associated with environmental perturbations and responses, reflect their norms and social objectives. The values underlying these debates and decision-making processes are not always clear. These values may be complex and in competition with other values, including not only the values of other people, but even values held by the same person. Because value frameworks are so diverse and complex, there is a need to refine methodologies for eliciting and comparing values quantitatively and qualitatively, both interpersonally and individually.

"Values" as used in the two preceding paragraphs is a term which describes the set of underlying factors that, taken together, cause people to hold the opinions that they hold and to make the choices that they make when presented with real situations. Unfortunately, the English language also uses the term "value" and its derivatives in an operational sense as a descriptor of the "worth" of outcomes, measured in terms of what would be willingly sacrificed in exchange. Thus the confusion in terms between a person "valuing" health in the abstract, the first definition of the term, and a person placing an inferred "value" of so many dollars, but no more, on avoiding a day of limited activity and discomfort due to a cold. The exchange of economic "value" is therefore derived from the vector of all the underlying abstract "values" held by the individual in concert with the situation presented. When it comes to making environmental (and other) decisions, "value" is used in an operational sense as a measure of what one outcome is worth in comparison with alternatives. This is the sense in which values are reflected in, for example, benefit/cost analysis. It is with this meaning that "value" is used in most appearances in this Chapter. Other uses of the term also occur, however, and are made clear by the context.

The Valuation Subcommittee discussed the issues of scientific uncertainty, competing values, and the completeness of individual valuation approaches. It concluded that all of the approaches used to develop summary information on risks, costs, and benefits for decision-makers reflect to some degree the disciplinary background and values held by those individuals who conduct the assessments. This

highlights the importance of knowing who participates in data collection and evaluation and the methods they use to do so. It further suggests the importance of making sure that individuals and groups having a stake in the outcome of decision-making participate in the process of describing and valuing changes in environmental systems, and in estimating the costs and benefits of intervention to control or redress these changes. Some form of interaction and deliberation among agencies and among those who hold direct or indirect interests in decisions is often recommended as a way of identifying and addressing the disparate values held by individual stakeholders (NRC, 1996; Risk Commission Report, 1997). The Subcommittee has endorsed this recommendation.

Valuation and value judgments are important in all phases of the IED framework (see Figure 5-2). Initial definition of values, and the goals to which they give rise, is required in the early problem formulation phase to help define the scope of the problem, its context, and the breadth of evaluation required in the analysis phase. Value judgments are also made during the analysis and characterization of risks. During the Analysis and Decision-making Phase, valuation input is again needed as alternative management or control options (e.g., technologies and practices) are evaluated to help identify and measure the relative costs and benefits of different degrees of risk reduction and of alternative ways of accomplishing it.

Values considered during environmental management decision-making are those associated with changes to both human dominated (e.g., farms, forests) and natural environmental systems. Substantial variation exists within natural ecosystems and to be socially significant, a change caused by a stressor must exceed the natural variability of the environmental system and involve impacts affecting significant social values. However, since no precise standards exist for measuring stressor reduction costs and associated societal benefits at the system level, it is inevitable that the decision on which values to weigh most heavily will require deliberation and negotiation. Therefore, it is clear that the articulation of these values, and the resultant regulatory actions based on them, must necessarily involve deliberation including scientists, the public, and policy makers in a process of on-going dialogue and adjustment.

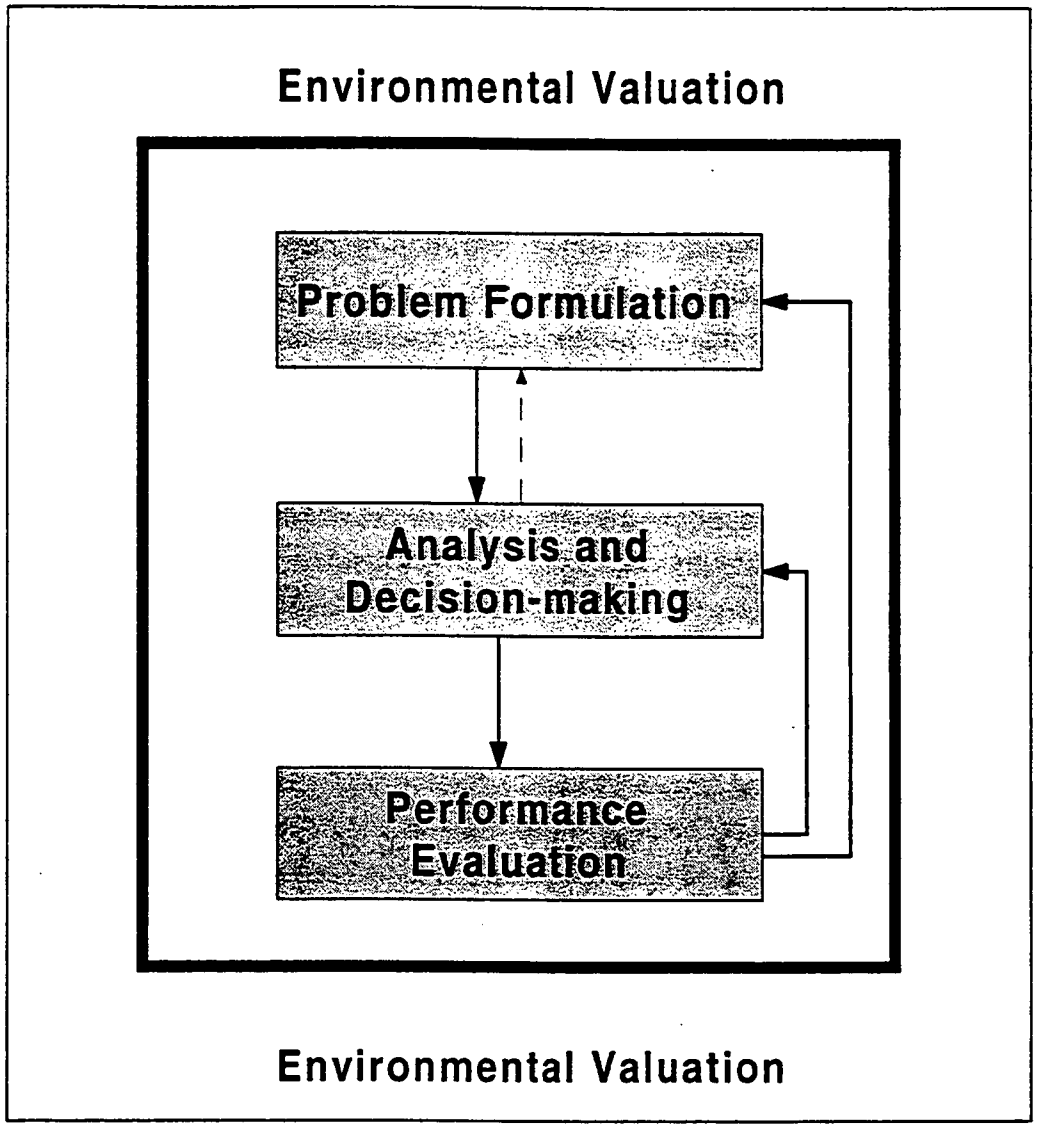

Figure 5-2. Valuation in the IED Process

Although current valuation techniques can deal satisfactorily with some environmental system changes (e.g., the impact of air pollution on managed agricultural systems), they may deal inadequately or not at all with others. When one thinks of considering the benefits of environmental systems, it is important to recognize that some things can be quantified and some not. Of those that can be quantified, some can be monetized and some not. For example, it is possible to quantify concern with biodiversity without being able to monetize it, or otherwise express it in units that can be compared to, say, changes in unemployment. Therefore, environmental system

valuation efforts must include both quantitative and qualitative perspectives in an integrated approach to capture all components of things valued.

Quantitative methods include those approaches designed to measure human attitudes, preferences (i.e., choices when presented with alternatives, desires, operational proximate goals) and values. As discussed in the previous chapter, these methods include: 1) direct measures of preferences and values using survey methods; 2) inferring preferences and values from observed behavior and choices; 3) inferring preferences and values from choice experiments; and 4) using estimates of values of similar commodities that were derived from using one of the above methods, or what economists refer to as benefits transfer.

Since not all environmental costs and benefits can be easily quantified for use in benefit-cost analyses, there is a need for qualitative methods to help decision-makers understand the hard-to-define values that are important in finding solutions to complex environmental system problems. Qualitative methods must involve at least the detailed narrative description of effects in complete terms such that comparisons with other ecological attributes can be made. Although this qualitative information cannot be directly compared to quantitative information in common terms, qualitative information should be clearly described and it should be readily available to assist during decision-making (e.g., see Florida Risk-Based Priority Council, 1996).

This information is used to guide decisions. At times, the path forward seems clear and uncomplicated, and decisions are straight forward. At other times, however, uncertainties exist — in how the interventions chosen will work, how the systems to be altered will react, and in how the outcomes will be valued. The process of adaptive management has developed as an innovative technique to assist in environmental decision-making in the face of such uncertainty. Rather than waiting until complete data are available that allow participants to understand all aspects of the at-risk system, adaptive management allows action to proceed on the basis of the best understanding initially available. The response to action is monitored, and the information gained is used to modify or design the next stage in the program. In adaptive management there is direct feedback between science and management such that policy decisions can make use of the best available scientific information in a sequential fashion. This idea has been well-developed in many contexts for nearly three decades (Campbell, 1969)

and has been recently articulated around environmental problems by Lee (1993). As the IED framework implies, valuation is not a one-time activity, it is an iterative part of the risk management process responding to the development of new information over time. That is, changes in the perception of the value of actions toward reaching goals is part of adaptive management as well.

* * * * *

In the following sections we discuss four topics relevant to environmental valuation: 1) the nature of values; 2) the traditional economic valuation framework; 3) the importance of the deliberative process to valuation; and 4) alternatives for the determination of environmental system values. Each of these sections includes a discussion and provides some recommendations for the consideration of valuation issues during environmental decision-making.

5.3 The Nature of Values

5.3.1 Introduction

Controversies surrounding environmental decisions are often expressed in terms of disagreements about values. These controversies are difficult to resolve in part because the word "values" is used in different ways and with different meanings. "Values" can refer to different goals of environmental decisions or to different means for achieving those goals. But "values" can also be used to express concerns that are not about goals or outcomes at all. Such concerns can be about the appropriateness of different procedures for reaching decisions, the expressive functions of different actions, or other things. People sometimes use the word "values" to refer to the strength of preferences for different outcomes, and it may be possible to interpret these values quantitatively, in monetary terms, without distortion. But people also use "values" to refer to more qualitative and difficult-to-quantify aspects of environmental outcomes. The challenge for the IED concept, then, it to include appropriate means for both interpreting values and incorporating them into the decision-making process.

Valuation exercises should be designed to explore the nature, meaning, and importance of these different environmental values. They should help to reveal what

people value, why, and (at least for some values) how much. They should challenge those people to think critically and clearly about their value judgments and the reasons for making them. And they should provide satisfactory ways of bringing together different kinds of values or concerns, while taking account of the costs and benefits of different alternatives, in ways that help parties reach agreement on values, where this is possible, or agreement on decisions where disagreements about values persist. The section that follows explores some aspects of the nature and meaning of environmental values, and it examines a few of the complex factors that must be understood and reconciled in the process of environmental decision-making.

5.3.2 Values

a) EPA should base its decisions on human interests and values. In doing so, it is important that EPA realize that people care for things for different reasons and in different ways. Valuation must be sensitive to these features of human values.

Philosophers have debated whether moral values are fundamentally anthropocentric (i.e., whether values refer to or are based solely on human needs and interests) or whether some values may be understood independently of human nature. EPA must base its decisions on human interests and an understanding of the reasons why people care for the things they do, and for this reason it must focus on human values. At the same time, it is important to realize that people care for things for different reasons and in different ways. Some things are valued for themselves, or as ends, and some things instrumentally or as means for the satisfaction of other ends. People care about some things simply because they desire them or because they contribute to their happiness. But other things have value because people judge them to be good or important in their own right, and therefore care about these things because they believe them to be valuable.

The valuation of environmental resources must be sensitive to the basic character of human values. Most people value the aspects of nature and environmental systems both instrumentally and as ends. They want to protect the environment as a thing to be valued in itself, as well as managing it for the other

services it provides. Environmental management involves multiple goals, which differ in kind as well as in intensity. The decision problems of environmental management are thus multi-attribute problems. They involve bringing together different kinds of values that may conflict or be incompatible in a number of ways. Values from all these sources are involved in forming preferences.

b) Preferences need to be examined. Some preferences may be more important than others in ways that are not captured by measurements of strength of preference alone. Preferences should be elicited in ways that reveal which preferences individuals endorse in light of information and the reasons that bear on them.

Preferences refer to a person's ordering or ranking of alternative states, or in the economic sense, alternative bundles of market and non-market goods and services. Preferences are the basis for the choices that people make; and preferences may be revealed by these choices. As described in the previous chapter, economists have developed two categories of techniques for assessing values: indirect measurement (revealed preference) and direct questioning (expressed preference). The discussion and recommendations that follow describe a supplementary approach, wherein stakeholders are directly involved in the deliberative process to collectively define ecological values.

EPA needs to examine the preferences people express for different alternatives and their reasons for holding those preferences. People can be asked to reveal their preferences, but the process often should not end there. Some preferences may be ill-considered; some may be based on erroneous or insufficient information; and some may be inconsistent with some other deeply held values. And some preferences may be less important to some individuals than they are to others--for clear and convincing reasons. The process of revealing preferences needs to explore and examine these reasons, and it should lead people not only to express their preferences but to endorse the preferences they have expressed after they have been examined in the light of information and reasons that bear on them. Further, the process can lead others to better appreciate reasons they may not have considered or to formulate opposition to some expressed preferences that may form the basis for rational deliberation and consensus-building.

EPA's objective, therefore, should not be simply to elicit preferences but to try to uncover or construct critically examined preferences. The process for doing this often must be deliberative and not be simply limited to gathering and assembling preferences.

c) Some important preferences for protecting and managing environmental resources are constructed in the elicitation process, so it is important to be sensitive to the dependence of preferences on the elicitation method and the way the alternatives are framed. This fact has some important consequences: 1) the assessor is necessarily more an active participant than a neutral observer in this process; 2) people need to be provided tools to help them think clearly about their preferences; and 3) the valuation process should be guided by some operating principles, e.g., questions should not be posed in only one way.

In many cases, values are revealed by the choices people make in their daily lives. It might be said that the values underlying these choices were formed by the individual's life experiences. In other cases, the values may not be pre-formed or revealed in the choices people have already made, and they must be constructed during the valuation process. The preferences expressed in the context of a valuation of changes in environmental resources are often constructed at the time the preferences are elicited and cannot be separated from the elicitation process. This fact does not negate the importance of the information gained from preferences constructed in this way. People should continue to be asked to express the values they hold, however they are formed. But we must be extremely sensitive to the dependence of preferences on the process of elicitation and the way the alternatives are framed.

One consequence of the constructed values perspective is the need to be sensitive to the reasons and causes of the preferences that are expressed. A second consequence is the need to understand the sensitivity of expressed values to methodological factors. A third consequence is an appreciation that the assessor is not a neutral elicitor but necessarily an active participant in the process of revealing preferences.

Decisions that are made by the individual being questioned in the course of a valuation exercise will include important values and factors that are not independent of how information is framed and how preferences are measured. EPA needs to go beyond some traditionally accepted economic approaches to the valuation of natural resources. Simply providing respondents with information may not yield expressions of coherent and consistent values. People need to be provided with tools to help them think about their values. Examples of tools that might help people indicate which of their desires or preferences they would regard as important, and which might be merely fleeting, are to be found in multi-attribute value analysis and various forms of structured deliberation processes that encourage people to think and reflect upon what they are saying, and to consider whether they really mean what they say.

Finally, the constructive nature of valuation processes should lead EPA to exercise caution and to take care, where possible, to pose a question in more than one way. EPA should also be explicitly aware of framing and elicitation effects in order to avoid the biases that they create. Approaches should be developed to attempt to decrease biases from framing, anchoring, etc. EPA might also consider sponsoring research to determine how accurate direct expressions of preference might be, under the best deliberative conditions, as indicators of citizens' true environmental values.

d) Because of the qualitative dimensions of values, the valuation process must emphasize deliberative as contrasted to mechanistic, predetermined components. Valuation must allow people to express the different qualities and kinds of values they regard as important, and it should challenge them to think clearly and usefully about these values.

Structuring valuation processes to emphasize deliberative as contrasted to mechanistic or algorithmic ways should help EPA to achieve another important goal, which is to be sensitive to the different qualities and characteristics of values that people often regard as important. Especially in the areas of environmental protection, many people regard some values as "protected" in ways that lead them to resist conceiving what is important to them as an exchangeable commodity. Also, many people think there are some important social values involved that cannot be reduced to an aggregation of individual preferences.

The valuation process should allow room for people to express values of these sorts. Again, however, even these preferences are not simply to be accepted at face value. These values, too, need to be challenged and explored. A sensitivity to the different characters of values, as well as to the need to explain and defend them with reason, should become part of the deliberative process.

The nature of values and the choices they support are important topics of economics, but these subjects have also been studied in depth by other social science disciplines, especially psychology and sociology, and by philosophy. The participation of these other disciplines in the design of valuation instruments/surveys should be encouraged.

e) The valuation process should be transparent and explicit. Any valuation process is subject to possible abuse. Rather than trying to deny this fact, EPA should guard against possible abuse or bias by making the process explicit and open to review.

Any valuation process is subject to possible abuse, and that is certainly true for the kind of more deliberative and constructive process that we are recommending be considered along with more traditional approaches. Given the nature of environmental values, the agent or elicitor of those values is involved in the process in a way that cannot be detached. Rather than deny that fact, EPA should aim instead at protecting against the potential for abuse in other ways. These involve making as much of the process explicit and open as possible, so that it will be transparent to participants and observers alike.

Eliciting values is a difficult process in that there are no hard and fast formulas that guarantee success. However, there are better and worse ways of doing these things, and it is important to try to become aware of the lessons that have been learned by able practitioners and to support a culture in the EPA that will encourage Agency officials to develop and use the requisite skills.

* * * * *

The next section discusses the concept of economic value, economic value and benefit-cost analysis, economic valuation of the functions and services of environmental systems, and some issues and problems associated with economic valuation.

5.4 The Economic Valuation Framework

5.4.1 The Concept of Economic Value

a) The economic value of an environmental change is the increase in human economic well-being or welfare that it produces. It is measured by the amount of some other good (usually money, for convenience) that can be taken away from those individuals affected by the change without reducing their welfare below the pre-change levels.

The instrumental value of anything stems from its ability to contribute to the achievement of some goal (Costanza and Folke, 1997, p. 49). One kind of value is economic value. The economic value of a thing depends on its contribution to the economic well-being of people. This economic concept of value has its foundation in neoclassical welfare economics. Each individual's welfare depends not only on that individual's consumption of private goods purchased in markets and of goods and services provided by the government, but also on the quantities and qualities of nonmarket goods and service flows each receives from the environment. Thus the basis for valuing changes in the flows of goods and services from the environment is their effects on human welfare. This anthropocentric focus of economic valuation does not preclude individuals from having concerns for the survival and well-being of other species. Individuals can value the survival of other species not only because of the uses they make of them (for example, for food and recreation) but also because of an altruistic or ethical concern. The latter can be the source of nonuse, passive use, or existence values.

The economic value of a good, that is, its contribution to the welfare of an individual, is measured by how much of some other good is required to make up for its loss. The economic value of a good could in principle be measured by the quantity of any other good that can substitute for its loss. If the substitute good has a market price,

then value can be expressed in terms of the money required to purchase the necessary quantity of the substitute good. Since in principle any other market good can be a substitute for the good in question, the economic value of the good can be defined as the amount of money or purchasing power required to substitute for the loss of a unit of the good, that is, to restore the individual to his/her original level of welfare.

b) Economic values can be revealed by the choices that people make as they trade off one good thing for another.

Since the source of economic value is the preferences of individuals, the choices that people make reveal something about these preferences and values. A variety of methods have been developed to infer the values placed on environmental goods and services from the choices that people make. The trade-offs that people make as they choose less of one good and substitute more of another good reveal something about the values people place on these goods. An example is the trade-off between money and air quality that people consciously or unconsciously make because houses in clean air areas have higher prices, other things held equal, as has been demonstrated in a number of studies. For further discussion of economic valuation methods, see, for example, Braden and Kolstad (1991), Freeman (1993), and Chapter 4.

5.4.2 Economic Value and Benefit-Cost Analysis

a) The economic values of changes in environmental systems should be included in benefit-cost analyses of environmental policies. Such benefit-cost analyses will be useful inputs into public policy decision-making.

The purpose of economic valuation of the services of environmental systems is to provide decision-makers with information that will help them make choices about policies toward specific environmental systems in the face of scarcity and opportunity cost. In most cases the question is not in the form of "either/or," but about how far to go. For example, the question usually would not be whether to protect an environmental system or do nothing, but instead what level or degree of protection or other management intervention should be attempted. Information on the values of environmental systems should take a form that is helpful in these cases. This means

that the relevant question is the value of the change in the services of environmental systems that an action would produce, not the total value of all services (Toman, 1997).

If society wishes to make the most (in terms of individuals' well-being) of its endowment of resources, it should compare the values of what its members receive from any environmental change or use of a resource (that is, the benefit of the change) with the values of what its members give up because of the change (that is, the costs). If the sum of all benefits exceeds the sum of all costs, the proposed change passes the "easy" benefit-cost test. When there is more than one proposal or project or degree of change under consideration, only one option will pass the more restrictive benefit-cost test and yield the highest aggregate net benefits. A society could choose to make public policy choices solely on the basis of comparisons of aggregate benefits and costs (maximizing net benefits). However, there may be other things besides economic benefits and costs that a society might want to consider when making policy choices (see, e.g., Arrow et al., 1996).

5.4.3 Economic Valuation of the Functions and Services of Environmental Systems.

a) The economic value of a function or service of an environmental system can take the form of a direct use value (the value of a service provided to people), indirect use value (when a function indirectly supports a service used by people), or nonuse or existence value (when an individual values an environmental system even though he/she does not make any direct or indirect use of it).

The functions of environmental systems include photosynthesis, decomposition, nutrient recycling, and so forth. The services of environmental systems are the materials and other services that they provide that enhance human welfare and are therefore valued by people. Examples of service flows include wood and fiber from forests, and amenities such as scenic vistas and wildlife observation. The values of these services are often called direct values (Brown, 1990) or direct use values (Goulder and Kennedy, 1997) to distinguish them from the nonuse or existence values mentioned above and the indirect use values discussed below.

The functions of an ecological system can also have an economic value if they help to support some service flow to people. If they do support a valuable service flow, the contribution made by these functions is an indirect use value (Brown, 1990; Perrings et al., 1995; Goulder and Kennedy, 1997). The indirect use value of a function can be derived from the change in the value of the service flow that it supports. For example, if an increase in the rate of photosynthesis in an ecological system results in an increase in the flow of economically valuable food or fiber, the economic value of the increase in photosynthesis is the increase in the value of the food or fiber it supports. What is required to measure this indirect use value is knowledge of the link between the function of the ecological system and the economically valuable service that it supports.

Sometimes the connection between a function of an environmental system and an economically valuable service flow may be quite direct, as in the case of photosynthesis producing useful plant material. But in many cases, the connection can be indirect and quite subtle. For example, photosynthesis by wild flowers may help to support a population of wild bees that also pollinate commercially valuable fruits. The basic point is that the economic theory of value accommodates changes in the functions of ecological systems that affect the well-being of individuals indirectly. However, these values can enter into the decision process only if the links between functions and services are known.

The functions of environmental systems may also have intrinsic worth or value in the eyes of some. This intrinsic value may be a form of nonuse value. Such nonuse values have the same standing as direct and indirect use values because they reflect the preferences of individuals. While quantification of nonuse values is difficult, they are of no lesser importance for this reason.

5.4.4 Issues and Problems

a) Aggregation of Individual's Values — Benefit-cost analyses are based on the summation of individuals' positive (benefits) and negative (costs) values. This simple summation raises issues of equity or fairness. For this and other reasons, most economists suggest using aggregate net benefits as only one of several possible considerations in public policy decision-making.

As discussed in Chapter 4, the simple summing of individuals' values to calculate net benefits is not without its problems. In summary, assessment of net benefits offers no guidance on the desirability of the resulting distribution of benefits and costs (i.e., some projects with positive net benefits may produce benefits for some people but at the expense of reducing the well-being of others). Another problem is that the economic values of individuals reflect not only their preferences, but also their economic circumstances, especially their wealth or income. One who judges the present distribution of income or wealth to be inequitable has reason also to question the economic values that emerge from that distribution as a basis for making public policy decisions.

For reasons such as these, many writers advocate using aggregate net benefits as only one input into the decision process and allowing consideration of other factors, such as distributional equity, as well (e.g., Arrow et al., 1996) .

b) Preferences, Values, and Knowledge — One of the basic premises of welfare economics is that economic values are based on individuals' preferences and that people know their preferences. Where individuals are ignorant of the roles of ecological functions in contributing to valued service flows, it may be necessary to use experts' knowledge of the functioning of environmental systems as an input in the valuation process.

A basic premise of welfare economics is that economic values are based on individuals' preferences and that people know their preferences. This premise is problematic when it comes to applying welfare economics to the valuation of the services and functions of environmental systems because an individual might act as if he/she placed no value on a function if the individual were ignorant of its role in contributing to a valued service flow from the ecological system (Dasgupta, 1990; Goulder and Kennedy, 1997). In the practice of environmental system valuation, there are ways to work around the ignorance of individuals about ecological systems. Value measures can be based on ecologists' knowledge of the relationship between the functions of ecological systems and valued services that they provide. Many examples exist in the environmental valuation literature where technical knowledge of physical and biological relationships coming from experts is incorporated into the process by

which individuals' preferences are determined. For example, even though people may not know the relationship between concentrations of air pollutants and the incidence of respiratory disease, expert knowledge of this relationship can be combined with peoples' revealed values for avoiding disease to calculate the benefits of improved air quality. However, when values are elicited through stated preference methods, the provision of information by experts is more problematical, as is discussed in section 5.3.2 above.

c) Replacement Cost as a Measure of Value — The cost of replacing an ecological function with a human engineered system can be used as a measure of the value of the natural system, but only if substantial conditions are met.

Some authors have used estimates of the cost of replacing a function of an ecological system with a human engineered system as a measure of the economic value of the function itself. For example, Gosselink et al. (1974) used an estimate of the cost of a tertiary sewage treatment system as the economic value of the nutrient removal function of a wetland. Replacement cost can be a valid measure of economic value only if three conditions are met (see, for example, Shabman and Batie, 1978): 1) the human-engineered system provides functions that are equivalent in quality and magnitude to the natural function; 2) the human-engineered system is the least cost alternative way of performing this function; and 3) individuals in aggregate would in fact be willing to incur these costs if the natural function were no longer available.

5.4.5 Conclusions

In principle, the economic valuation framework can be utilized to define and measure the economic values of changes in the functions and services of ecological systems that affect individuals' welfare either directly or indirectly. In principle, the framework can be used to formulate the choices required to incorporate a variety of factors into social decisions. This framework may be difficult to implement in practice where the relationship between the function and the service flow to individuals is indirect or subtle. Further, effective application of the framework requires that all relevant matters be included, which sometimes suggests the use of processes including deliberation, discussed immediately below, to assure that this happens. There is now a

growing number of examples of application of the economic framework to environmental valuation (see, e.g., Perrings et al., 1995; Freeman, 1997).

* * * * *

The next section discusses the need for deliberation in eliciting values. This reflects the Subcommittee's interest in finding ways to determine people's values regarding specific environmental problems, in the face of the shortcomings that are suggested as being associated with traditional environmental valuation exercises.

5.5 The Importance of Deliberative Processes to Valuation

5.5.1 Introduction

Deliberation among interested parties, though not required in all decision-making cases, is often useful for eliciting information on important goals, on "best judgment" on certain relationships, on measures for judging goal attainment, and on value tradeoffs that must be made in decision-making and implementation. A variety of types of deliberation are available, depending upon the sufficiency of knowledge available to address the problem at hand, and the extent of agreement that might exist on the types of values involved in the issue.

Appropriate types of deliberation can help EPA select suitable analyses and reach decisions that incorporate broadly acceptable tradeoffs of efficiency, fairness, and sustainability. A substantial literature related to environmental decision-making, including a number of recent reports (NRC, 1996; Presidential Commission of Risk Assessment and Risk Management, 1997), stress the importance of deliberation to Agency decision-making.

By deliberation we mean..."any formal or informal processes for communication and for raising and collectively solving problems" (NRC, 1996; p. 73). Deliberation is seen as the process of bringing together those people whose expertise or perspective is useful in order to craft management and policy options. Deliberation can examine findings, interpretations, and/or economic and non-economic values in light of each other to achieve greater clarity and to construct and consider management options.

Thus, deliberative processes differ from but supplement analytic processes which use "systematic application of specific theories and methods" (NRC, 1996; p. 215) such as those of science, social science, and economics. Some types of deliberation primarily involve disciplinary experts, while others require involvement of stakeholders who bring disparate perspectives and knowledge to the table. In some cases, stakeholders may be disciplinary experts who may have different interpretations of the state of knowledge than experts in the Agency. Deliberative processes need not be reserved for interactions with non-federal stakeholders, but may also be used to facilitate intra-Agency or interagency discussions on an issue.

Some of the reasons given in the NRC and Presidential Commission reports for using deliberative processes include:

Clarifying and potentially advancing resolution of issues of fairness. Issues such as distributional equity (Who benefits?) and procedural equity (Who decides?) can be raised explicitly and viewed in the specific context in which an environmental decision needs to be made (see, e.g., Vaughan, 1995). (When we use the word "fairness" we are referring to both distributional and procedural equity, two issues that have been concerns of the Environmental Justice movement.)

Informing multi-dimensional tradeoffs among efficiency, fairness, environmental sustainability, and other concerns. In situations that involve conflicts about such tradeoffs, deliberation can explore potential tradeoffs and craft various options that might reduce the amount of conflict (see, e.g., NRC, 1996).

Increasing credibility. By discussing various perspectives, the credibility of information and Agency decision-making can be enhanced, although agreement with Agency process does not necessarily result in agreement with Agency decisions (see, e.g., Rosener, 1981; Mazmanian and Nienabur, 1979).

Informing priorities for research. Studies suggest that more data are not necessarily better for organizational decision-making. Individuals and organizations have limits on the amount of information they can assimilate. In

addition, some information may be peripheral to the key dimensions of the decisions to be made and do little to clarify or resolve conflicts (Dietz, 1988). Deliberation can help determine the research that is most likely to be key to decision-making and distinguish it from research that may not reduce uncertainty sufficiently to inform the decision at hand (NRC, 1996). Thus deliberation can conserve resources.

5.5.2 Aspects and Recommendations

a) We recommend that the context (characteristics, scope, and implications) of an Agency decision be taken into account in the selection of deliberative processes.

This concept is expressed by our typology of deliberative processes with stakeholders and experts (Figure 5-3). The type of deliberation, the selection of specific deliberative techniques (e.g., scientific meeting, citizen advisory panel, citizen jury, informal meeting, etc.), and the selection of participants all vary by the situation (English et al., 1993).

There is no one-size-fits-all approach. The Agency should consider two questions when determining the type of deliberation that is most appropriate:

I. To what extent is the agreement on values (e.g., fairness, sustainability, efficiency, etc.) and on appropriate tradeoffs among them sufficient to reach a decision?

The relative importance of efficiency, fairness, sustainability and other concerns may vary among experts and stakeholders. When Agency decisions require tradeoffs among and along these dimensions and are likely to lead to conflict, agencies are often forced to make judgments that cannot be based solely on knowledge. The selection of an appropriate type of deliberation will depend on where the extent of agreement falls on the continuum between high and low.

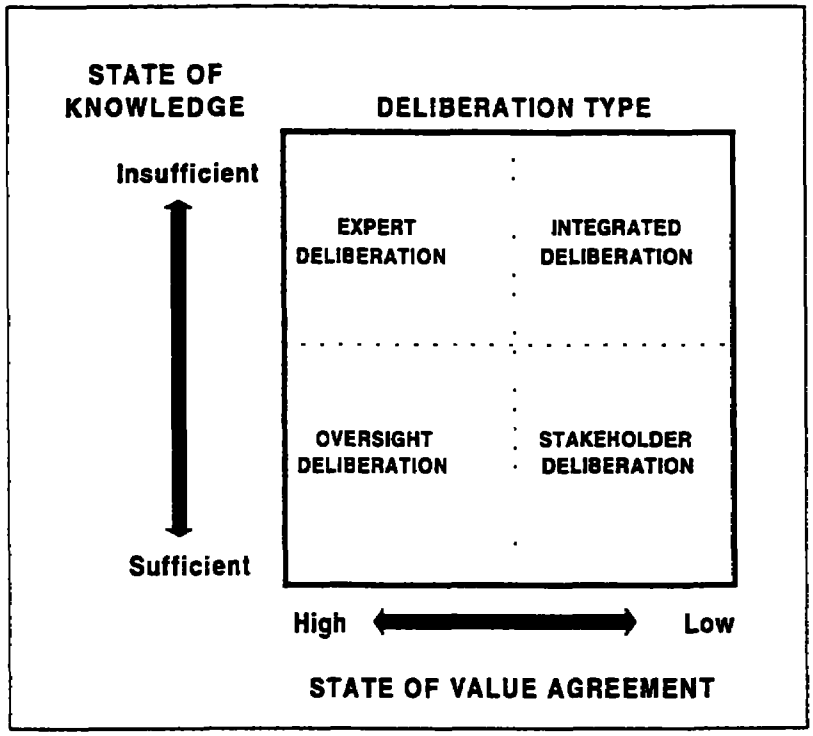

Figure 5-3. Typology of Deliberation Processes With Stakeholders and Experts (adapted from Chess, Dietz, and Shannon, 1998)

ii. To what extent is the state of knowledge sufficient to address the problem at hand?

By knowledge we mean information and understanding from the biological and physical sciences, engineering, economics, and the other social sciences. The answer to this question depends on the extent of knowledge about information critical to making a particular decision. In many situations, knowledge from environmental sciences, such as information about environmental system processes and the nature of potential threats, may play a decisive role. In others, knowledge about economic costs and benefits may be decisive. Social science knowledge may also be decisive, for example by providing an understanding of communities where demographics, ethnicity, or racial composition must

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be given serious consideration. Agency diagnosis of a situation, and the form of deliberation needed, will depend, in part, on its assessment of the extent to which available information is adequate for making a decision. When the state of knowledge is insufficient or controversial, or when there is lack of agreement about the state of knowledge, more extensive deliberation will be needed.

The Subcommittee appreciates that Agency decisions may be constrained by regulations, resources, and court decisions. These limitations should be made clear to participants in any deliberative process. Yet, situations in which agencies have no latitude are rare (Pflugh and Shannon, 1990). Without reflection on the above two questions, agencies may fail to appreciate the likelihood that a decision, made without appropriate deliberation, might become hopelessly stuck in environmental gridlock.

b) When agreement about values (economic and non-economic) is high and the state of knowledge (relevant science, economics, and social science) is sufficient and/or non-controversial, Agency decision-making is likely to be routine. Deliberation will only be needed periodically, if at all, for oversight (Oversight Deliberation).

Most Agency decisions are routine administrative ones that conform to existing regulations and policies. Such decisions may include non-controversial permitting, changes in labeling, and minor shifts in administrative procedures. In such situations, oversight deliberation, the periodic conferring of experts to assess a program and potential modifications, is appropriate. However, if conflict develops around multi-dimensional tradeoffs or the state of knowledge, the type of deliberation will need to move toward another quadrant.

c) When agreement about values is low, but the state of knowledge is sufficient and/or non-controversial, Agency decision-making will require multi-dimensional tradeoffs based on knowledge. Stakeholder Deliberation is needed.

In such situations, the state of knowledge is sufficient to inform multi-dimensional tradeoffs, but there is little agreement about which tradeoffs to make. Because the

conflict is usually based fundamentally on values, not knowledge, the deliberation can involve primarily stakeholders who will evaluate tradeoffs in light of their priorities. Stakeholders, informed by available knowledge, can craft options with varying tradeoffs. Experts may provide information about the potential impacts of various options, but they need not be as extensively involved as in expert deliberation, as noted in the following section.

d) When agreement about values is high, and the state of knowledge is insufficient and/or controversial, Agency decision-making is likely to be experimental and iterative. Expert Deliberation is needed.

In such situations, making decisions is difficult primarily because of the state of knowledge. For example, there may be limited knowledge about the impact of human management on a particular environmental system. Expert deliberation — on-going conferring among experts (often from different disciplines) — can be needed to develop appropriate monitoring processes and to interpret results. Based on the results, expert deliberation may result in recommendations for changes in management of the environmental system; i.e., adaptive management with expert deliberation at intervals determined by the nature of the experiment. For example, monitoring of the impact of reducing water flow to an environmental system may require experts to confer at regular intervals to review monitoring data and determine if the water flow should be changed. However, if value-based conflict arises over the results of such iterative decision-making, the situation will require integrated deliberation, involving experts and outside stakeholders working together to make multi-dimensional tradeoffs on the basis of limited knowledge.

e) When agreement about values is low and the state of knowledge is insufficient and/or controversial, Agency decision-making is likely to require multi-dimensional tradeoffs based on insufficient knowledge. Integrated Deliberation involving both experts and outside stakeholders is needed.

These decisions are usually the most difficult for agencies because there is little confidence in the state of knowledge about the impacts of tradeoffs on economic efficiency, fairness, sustainability and other concerns. In such situations integrated

deliberation may be needed. By integrated deliberation we mean on-going interaction among experts and stakeholders during problem formulation, collection of information, and development of options (NRC, 1996; Commission on Risk Assessment and Risk Management, 1997). Integrated deliberation may also be needed during implementation and performance evaluation, and may take the form of adaptive management with stakeholders and experts reviewing the results and suggesting iterative changes. The nature of integrated deliberation depends on the situation, but, in general, the greater the conflict (or potential conflict), the more extensive the deliberation needed.

f) EPA can identify stakeholders by asking the following questions (adapted from Chess and Hance, 1994):

-Who has information and expertise that might be helpful?
-Who has been involved in similar decisions before?
-Who has wanted to be involved in similar decisions before?
-Who may be affected by the decision?
-Who may reasonably be angered if not included?

The process for selecting stakeholders should be perceived as fair, lead to a broad representation of stakeholders, and should ensure that representatives of stakeholder groups are perceived by the groups as acceptable representatives (NRC, 1996; English, 1993).

Stakeholder involvement processes need not preclude involvement by a greater number of participants. In fact, in conflicted situations it may be useful to complement a stakeholder involvement process with survey data (see, e.g., Kathlene and Martin, 1991). Researchers are also exploring forms of deliberative processes such as citizen juries, which involve random selection of citizens (Brown et al., 1995; Crosby et al., 1986).

g) **Involving stakeholders before extensive development of Agency proposals may generate constructive options and trust. Integrated and stakeholder deliberation should include stakeholder involvement in problem formulation and options generation.**

Early involvement facilitates agencies' ability to define problems and craft solutions that may meet satisfactorily the goals of both agencies and stakeholders. When participation is on-going, the Agency can identify problems before they become crises (Thomas, 1995). As the Government Accounting Office (1994) has suggested in the context of Superfund, earlier outreach can improve participatory efforts and minimize dissatisfaction.

h) **We recommend that the existing literature on group processes, ongoing research on deliberation, experienced practitioners, and stakeholder input be used to provide guidance on the selection of deliberative methods.**

Just as selection of the type of deliberation is determined by the situation, the selection of deliberative methods is dependent on the context (English, 1993). There is no single guide that can cover all situations, and care must be taken in designing a process so it is appropriate to the problem being addressed. As with all policy analysis tools, high quality results depend on careful application of the methods that are grounded in both theory and practical experience. The sources of guidance for selection of deliberative processes include:

Existing Literature. There is a substantial and sophisticated body of research on small group processes and a smaller but important body of research on deliberative processes. Key entry points to this literature include Mazmanian and Nienaber, 1979; Rosener, 1981; Fiorino, 1990; Dietz, 1994; Renn et al., 1995; NRC, 1996; and U.S. Department of Energy, 1996.

Ongoing Research. Increasing effectiveness in the use of deliberation will require ongoing research on deliberative processes and their use in policy contexts. Some research of this type is proceeding, including a few

projects funded by the recent EPA/National Science Foundation joint initiative. But compared to other policy analysis methods, very little has been invested in research to support deliberation (NRC, 1996).

Experienced Practitioners. Like many methods of policy analysis, the translation of theory and empirical research into practice is a craft that depends on practitioners who are both knowledgeable and experienced. Such practitioners can also be an important source of systematic information. However, deliberation rarely can simply be turned over to a consultant. Both integrated deliberation and stakeholder deliberation require Agency involvement.

Stakeholder Input: Those participating in a deliberation are a vital source of information on how to proceed. Further, systematic post-hoc studies of stakeholder perceptions are critical to building cumulative knowledge (see, e.g., Balch and Sutton, 1995; and NRC, 1996). For further examples of such studies, see also "Existing Literature" above.

i) EPA should build additional institutional capacity for deliberation.

Effective decision-making depends upon institutional capacity appropriate to the challenges posed by the situations we have described. That institutional capacity is inadequate, both in the larger community and in EPA, and the Agency should move to develop it in both arenas. Research and training support, for example, would deepen the resources available to EPA, while greater use of deliberative processes would call forth additional interest in the outside community. EPA can utilize external resources to make more effective use of deliberation, but it also needs to increase its internal capacity by adding expertise and making other organizational changes such as:

Expertise. Deliberative processes depend on experienced, trained practitioners and knowledge and skills provided by those with familiarity with social science research in areas such as psychology, sociology, anthropology, political science, management, and economics. EPA may need to hire personnel with this expertise.

Organizational Structure. Currently, community relations, public information, and other personnel serve as technicians (carrying out instructions) rather than being part of organizational decision-making (Grunig, 1992). Thus, key decisions may be made without the presence of a manager with expertise in stakeholder values and concerns. We suggest that those with expertise in deliberation should serve not only as technicians, but as both mid-level and senior managers, who can assist the Agency to develop appropriate deliberative processes and to make critical decisions that involve conflicts over values.

Organizational Learning. To improve deliberative processes, agencies must learn from experience (Chess et al., 1995). Mechanisms are needed to collect, analyze, and interpret data on deliberative processes. Just as records of environmental monitoring or risk assessments help to improve those practices, learning about deliberation will require record keeping and means to institutionalize what is learned.

j) Because environmental systems are arranged in hierarchies, effective deliberation requires institutional arrangements that reflect this structure, with appropriate authority, institutional capacity, and information available at local, regional, and national levels.

Such institutional arrangements allow citizens who are directly affected by tradeoffs among fairness, economic efficiency, and sustainability to deliberate over their resolution and to bring to bear critical information about local conditions, particularly those involving spatial and temporal distribution of resources.

Relationships among institutions at the same level and with those at other levels should be relatively flexible, allowing stakeholders and experts the leeway to match institutional capacities with the specific ecological problem. To enable community-based institutions to participate in making multi-dimensional tradeoffs, care should be taken to avoid severe imbalances among different levels of institutions.

When community-based institutions have already developed management of resources and made multi-dimensional tradeoffs, other levels of government should

recognize the local management systems that are in place and, to the extent possible, work with rather than supplant those systems (see, e.g., Schlager, 1994). While community institutions can draw on local knowledge about distribution of resources over space and time, they are often at a disadvantage in involving additional scientific expertise. Conversely, higher level institutions have greater capacity and resources to draw on scientific expertise but may be at a disadvantage in their appreciation of local knowledge about the functioning of an environmental system or of a resource within it. Thus, successful resolution of sustainability problems and multi-dimensional problems may be more likely if scientific capacity of local institutions is increased and higher level institutions draw on local knowledge.

k) Deliberation should be protected from undue manipulation by special interests.

Conflicts among interest groups vying for power is inherent in democracy. Deliberative processes can be protected from undue manipulation through: a) open and transparent processes, b) rules and mechanisms to ensure due process; c) sufficiently broad participation so that deliberation is not dominated by special interests, d) structured group processes that reduce the possibility of manipulation, and e) expertise of those leading the processes so that manipulation is seen quickly and constrained.

* * * * *

The following section presents information on additional approaches to valuing benefits of environmental systems. The additional approaches are discussed in response to the need to improve this ability to value environmental benefits in ways that address the limitations of traditional economic analysis techniques. The following considerations, and others, need to be the subject of continuing research as society's needs to address environmental issues become increasingly more complicated and exceed the adequacy of current methodologies.

5.6 Additional Approaches to Valuation of Environmental Systems

5.6.1 Introduction

"Valuation ultimately refers to the contribution of an item to meeting a specific goal. A baseball player is valuable to the extent he contributes to the goal of the team's winning. In ecology, a gene is valuable to the extent it contributes to the goal of survival of the individuals possessing it and their progeny. In conventional economics, a commodity is valuable to the extent it contributes to the goal of individual welfare as assessed by willingness to pay. The point is that one cannot state a value without stating the goal being served. Goals are not necessarily independent, i.e. increasing activity toward one goal may result in tradeoffs which decrease other goals" (Costanza and Folke, 1997)

5.6.2 Findings

a) **The value of something stems from its ability to contribute to the achievement of some goal.**

b) **Fairness, sustainability, and other values, as well as economic efficiency, are goals that should be recognized and explicitly addressed in environmental resource decision-making. Such goals can be compatible and mutually reinforcing; i.e., sustainability could be enhanced by improving the efficiency of resource use and reducing ecological stressors, and fairness can enhance acceptance of and compliance with environmental decisions.**

Conventional economic value is based on the goal of maximizing individual economic welfare, which includes all the things people want including environmental quality and sustainability of ecological systems. But explicit recognition of such non-private goals, and of their importance, is crucial in environmental valuation. For example, sustainability is critical to future generations. It is important to properly ascribe value to outcomes that contribute to achieving sustainability, social equity, or other ends that may be deemed important, as well as the value placed on the goals of producing things people want directly. This broadening is particularly important if the

goals are potentially in conflict. The narrow view of environmental decision-making as a choice of one or another action, each representing widely differing value-driven viewpoints, outcomes, benefits, and costs, is rarely the case. Rather, environmental decision-making is an activity that involves trade-offs among a number of goals that are the product of different value choices. Decisions must reflect some combination of these disparate values. Goals that may be important in environmental decision-making include those relating to aesthetics, services of nature, recreation, consumption, culture, biodiversity, and human health. Because of the variety of values-driven goals that enter the decision-making process, there is a need for a number of ways to conduct valuation exercises that are used to inform decision-makers.

c) Ecological systems provide diverse benefits to society, including marketed goods and services, nonmarket direct and indirect use values, and nonuse or existence values. Ecological services have values that can be identified and at least partially quantified.

d) Appropriately characterizing the societal value of the full range of environmental services requires approaches in addition to traditional individual preference techniques. Such techniques would inform the democratic processes that can be used to determine and then implement policies which enhance the flow of environmental services that would not be provided by the operation of market and quasi-market fulfillment of individual choice.

e) An important example of an element that may not be adequately reflected in valuation exercises is biodiversity. Biodiversity is positively related to ecological services and is declining principally because of anthropogenic- induced land use changes and other habitat alteration. Biodiversity has benefits, and its loss is a cost that should be factored into decision-making.

Biodiversity is one of the important attributes of environmental systems which clearly emerges when aggregates of species and populations are considered. The term biodiversity is commonly used to describe the number and types of organisms in a

given area, as well as the variability (genetic and phenotypic) within populations of organisms. In addition, biodiversity includes "the ecological roles that these species play, the way that composition of species changes as we move across a region, and the groupings of species (ecosystems) that occur in particular areas (such as grassland or forest), together with the processes and interactions that take place within and between these systems. It also covers the diversity of ecosystems in landscapes, of landscapes in biomes, and of biomes on the planet." (Heywood et al., 1995).

Because biodiversity reflects the complex interactions among species in an area, and between living organisms and the physical environment, it is reasonable to suppose that the services provided by a specific environmental system are affected by the various levels of diversity contained within the system. Therefore, when conducting valuation exercises, it is important to consider the biodiversity of the system.

According to Heywood et al. (1995), "The values placed on biodiversity are strongly linked to the human influences on it and their underlying social and economic driving forces. They are also dependent on some degree of knowledge of the scientific role of particular elements or processes of biodiversity in the functioning of our ecosystems and societies.... [W]hile it is undoubtedly true that the multiple values of biodiversity are not adequately captured in its market value, if we want to commit and prioritize resources to its conservation and sustainable use, then applying economic measures to its evaluation is unavoidable."

It is likely that the environmental values most often included in economic analysis are those associated with environmental services flows having direct market values (e.g., timber, pharmaceuticals, fish, wildlife, etc.). However, the importance of biodiversity to the provision of these, and many other service flows, has not yet been adequately recognized or explored. Therefore, we need to identify critical functions and to focus research efforts on these so that important measures reflective of the variety of factors associated with biodiversity can be completely reflected in the analysis of benefits from environmental management.

f) Environmental systems are organized in the form of nested hierarchies, having spatial, temporal, and organizational dimensions. For this reason, decision-making authority should be organized in a

way that mirrors or parallels the natural systems. A major benefit would be reduced information costs and analytical complexity.

There is a need to manage events in the human/environmental system at a range of different scales (Wilson and Dickie, 1995). In the management of natural resources, the spatial scale inherent in the regulatory apparatus often influences the nature of the questions that are addressed. For example, marine fisheries are managed primarily on a regional scale (by the U.S. regional fishery management councils), and this management has tended to focus on questions of how many fish to harvest rather than on ecological factors such as habitat and population structure that occur at smaller scales. In addition, large scale management oversimplifies local phenomena that are important to regulatory goals; ecological interactions are complex and the outcomes of human intervention are often uncertain. Because region or ecosystem-specific differences can lead to large transaction costs in centralized management schemes, an allocation of decision-making authority toward a decentralized hierarchical regulatory system can bring timely and accurate information to bear on both the regulatory and valuation problems.

With respect to valuation, the ideal is to exercise decision-making authority at the scale where individual incentives are most closely aligned with societal objectives. Put another way, individuals, acting in their own self interest, may over-exploit open access resources (often referred to as the "tragedy of the commons"--Hardin, 1968). Individuals acting as parts of groups or communities may adopt norms or rules of behavior that serve to avoid such over-exploitation (Ostrum, 1990; Schlager et al., 1994).

Another benefit of decentralized management is to improve the equity and credibility of decision-making processes. These are important characteristics that can improve the potential for stakeholder "buy-in" and compliance with decisions.

g) Ecosystem management is a promising new approach for making choices in the face of the uncertainties associated with the risks and benefits of management decisions, especially in the areas of fairness and environmental sustainability.

Many of the more straightforward environmental problems that have been the focus of the last few decades of environmental protection in the U.S. (e.g., air and water pollution by toxic chemicals) have been addressed in EPA's and other agencies' environmental decisions and policies, resulting in improved environmental conditions. However, the human activities that now cause higher risks for ecological systems are more physical than chemical, involving such stressors as habitat alteration and hydrologic modifications that cause effects at regional scales. The next generation of advances in environmental protection, then, must deal with regional-scale interactions of humans with natural ecological systems. Ecosystem management (also termed community-based environmental protection [CBEP] by EPA) is an emerging concept that addresses such regional issues and seeks to develop mutually dependent sustainability for human and ecological systems (U.S. MAB, 1994; Christensen et al., 1996; Harwell et al., 1996).

Ecosystem management uses a deliberative process for society to make explicit choices about what level of sustainable ecological quality is desired at each specific location within a region. This deliberative process requires extensive and recursive interactions among scientists, decision-makers, and stakeholders to identify spatially explicit environmental goals, define the ecological and societal endpoints that characterize those goals, and implement management approaches to achieve the goals. This requires an approach to governance that is commensurate with the spatial, temporal, and organizational hierarchies of the ecological systems. Further, ecosystem management involves making decisions with explicit consideration of the uncertainties associated with environmental risks and benefits.

Sustainability is a central desired tenet of ecosystem management. Consequently, the time horizon for decisions is by necessity inter-generational in human terms and, thus, issues of sustainability and inter-generational equity (i.e.,the distribution of ecological services over generations of humans) are intimately linked. Since ecosystem management is goal-driven, it is essential that science properly inform the societal goal-setting process by defining what is sustainable and what constraints the natural systems impose on management options. Just as importantly, however, it is up to society, not scientists alone, to set the sustainability goals. Once these goals are established, then science can help by translating the societal goals into specific ecological, hydrological, and societal conditions and identifying the specific measures

that need to be monitored in order to evaluate how well the goals are or are not being met. The comparison of how ecological systems respond to decisions and policies with specified performance criteria constitutes the critical element needed for adaptive management, which is a key component of ecosystem management. Consequently, for regional-scale environmental protection, ecosystem management constitutes a driving force both linking societal values with scientific understanding and guiding the environmental decision-making and adaptive management processes, forming the context in which the science and society, analytic/deliberative process develops (NAS, 1996).

h) Scientific analyses and models that explain the principles of operation and the interconnections among socioeconomic and ecological systems are important to environmental management. Not all environmental problems require the same level of scientific analysis and deliberation. Complex issues may need to emphasize deliberation and use judgment based on qualitative analyses.

i) During deliberations and presentations to the Subcommittee, several interesting directions emerged that, with further research, may improve our ability to measure some of the emergent values that environmental management approaches, and environmental systems in general, confer on the economic system. Results of such research should help to determine how best to align the regulatory process with the opportunities for minimum cost compliance. Research is also needed to determine how dynamic adjustments to regulatory interventions can be incorporated better into the process of estimating the prospective costs and benefits of possible actions so that realistic information is given to decision-makers. For example, research may be fruitful in the following areas:

1) The relationship between biodiversity and environmental service flows of importance to humans:

As discussed above, it seems a good working hypothesis that

biodiversity of an ecosystem is related to the level of goods and services of value to humans from that system.

2) Systems analysis of the relationship between economy and environment:

A number of recent empirical studies in the United States have shown that states with better environmental conditions generally have better economic conditions. For example, Meyer (1992) demonstrated a positive statistical correlation between higher state environmental rankings and better job conditions, e.g., lower unemployment and higher productivity. Similarly, Templet and Farber (1994) demonstrated that surrogate measures of environmental risk (e.g., the ratio of chemical releases, as reported by the Toxic Release Inventory (TRI) data base, to jobs created in the discharging industrial sector) are inversely related to measures of economic welfare (e.g., unemployment rates and disposable income per capita), and Cannon (1993) showed that higher economic growth rates prevail where environmental rankings are higher. Hall (1994) developed state rankings based on economic and environmental conditions that turned out to be remarkably similar and correlated well.

These results do not establish the fact or direction of causation. While these results suggest that the economy could be enhanced if the environment is improved, it is also possible that citizens in higher income states have a higher demand for environmental quality and convince their governments to deliver it. It also may be that the initial endowment of natural and man-made resources in some states led to both economic and environmental conditions being better.

A positive correlation between broad, general measures of environmental conditions and measures of economic production is frequent in circumstances where rampant industrial growth has occurred without consideration of the damage done by residuals, a condition that is familiar in some developing countries, and indeed, occurred historically in the United States. When such externality conditions are presented in

specific situations (i.e., industrial water discharges damaging a fishery), existing regulatory and tort remedies are readily available. The issue is, however, whether more subtle and general levels of environmental stress cause effects such that actions can be taken which would improve both the environment and economic production. Research on this issue could be designed to determine whether such circumstances exist, and if so, what remedies are available that would improve environmental conditions at a cost that did not exceed the improvement of economic production and the increased value ascribed to the environmental improvement itself.

The complementarity interpretation can be understood in relation to a simple model (Figure 5-4) which places the economic system within the environmental system and reliant on it to provide energy, materials and information and to accept waste. If the environmental system is negatively impacted by disruption or residuals, it may contribute less of these services to the economic system and, as a consequence, the economy may become less productive and contribute less to public welfare. In this view, the economy could be enhanced if the environment is improved and vice versa.

3) Biophysical measures as a surrogate for the value of ecological systems:

One alternative method for estimating ecological values assumes a biophysical basis for value (e.g., Costanza, 1980; Cleveland et al., 1984; Costanza et al., 1989). This theory suggests that in the long run, humans come to value things according to how costly they are to produce, and that this cost is ultimately a function of how organized the goods are relative to their environment. To organize a complex structure takes energy, both directly in the form of fuel and indirectly in the form of other organized structures like factories. For example, a car is a much more organized structure than a lump of iron ore, and therefore, it takes much energy (directly and indirectly) to organize iron ore into a car. The amount of solar energy required to grow forests can therefore serve as a measure

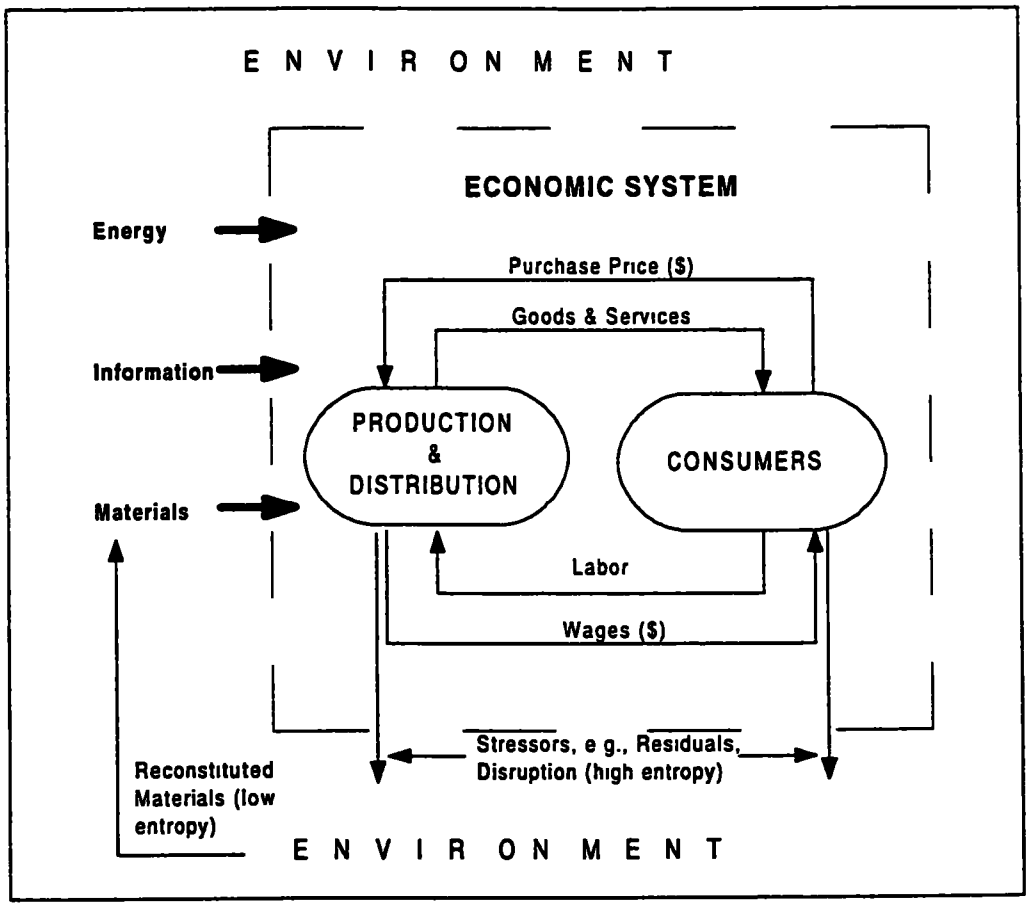

Figure 5-4. The Relationship of Economy to Environment (modified from Templet, 1995)

of their energy cost, their organization, and hence, according to this theory, their value. Various methods have been suggested in the literature for estimating these direct and indirect energy costs ranging from methods based on input-output analysis (Costanza, 1980; Cleveland et al., 1984; Costanza et al., 1989; Ulanowicz, 1986; Mageau et al., 1995) to methods based on embodied energy (or "emergy") analysis (Odum, 1996). For a comparison of input-output based and "emergy" based analysis, see Brown and Herendeen (1996). Of these, the input-output approach, borrowed from economics, appears to offer the most promise

for estimating direct and indirect energy costs.

4) The efficiencies and environmental benefits that may result from the implementation of industrial ecology principles:

If Figure 5-4 is treated as the basis for a simple model, inputs of energy, information and materials are seen to yield outputs from the economic system in the form of goods, services, and residuals. Improved processes for converting inputs into desired outputs can yield lower energy use and/or lower production of residuals. Both of these results tend to reduce environmental stresses. Recognition of these interactions has led to a focus on "industrial ecology," which emphasizes a holistic approach to organizing production processes as a substitute for the piecemeal, sub-process decisions which often ignore potential beneficial connections across the enterprise and even outside of it. Recycling of residuals into useful products, use of waste heat from some operations as input into others, changes in production processes to reduce residuals or to yield residuals that are more easily recycled or used, and other means of more nearly "closing the loop" have been implemented. The implication of all this is that the relationship between energy and other inputs and outputs, including residuals that stress the environment, is not immutable. Hence, environmental policy actions that penalize the production of residuals need not lead to proportionate reductions in output of desired goods. Instead, in reaction to regulatory requirements, the system may, and most likely will, adjust at multiple points and these adjustments will lower costs (compared to a response that just dealt with the residual "at the end of the pipe") and may have other beneficial effects. Therefore, regulatory instruments should encourage private sector decision-makers to look for means to lower harmful residuals while leaving them with maximum flexibility in how this is accomplished.

Recent experience with industrial ecology suggests that *a priori* estimates of the social costs of reducing residuals that may harm the environment are often excessive. As the tenets of industrial ecology become more widespread and widely adopted, the promise is that

improved environmental quality can be bought at a lower cost in lost goods and services production than previously thought. This suggests that estimates of the costs of reducing harmful residuals should incorporate realistic opportunities for systemic adjustment. Further, estimates of benefits from such action should be holistic as well. For example, they should take into account any expected reductions in environmental harm from input or process changes that would result from the regulatory action.

The conclusion is that the optimal level of environmental quality may be underestimated if too narrow and static a picture of response to regulatory pressures is taken as the basis for regulatory decisions. Consequently, opportunities for increases in social welfare may be lost. Research on the efficiencies and cost-savings that may result from the implementation of industrial ecology principles would improve our ability to estimate accurately net benefits for different management scenarios.

In principle, the economic valuation framework can be used to formulate the choices required to incorporate ecological and other factors into social decisions. However, traditional economic analysis techniques have limitations of methodology and completeness. This section has identified some of these limitations and sketched some of the research directions, that if successful, may relax them. A key theme of this section is that provision of a rich and diverse set of information to decision makers can contribute to improved decision making.

5.7 Summary and Conclusions

Because many scientists, decision-makers, and other stakeholders feel that economic analysis methods undervalue environmental resources, the EPA Science Advisory Board established a Subcommittee as part of its Integrated Risk Project to explore ways to advance the dialogue that would lead to a more complete framework for estimating the benefits of environmental management actions. In meeting its charge, the Valuation Subcommittee considered: 1) the environmental management decision context for valuation; 2) the nature of value and values; 3) the economic concept of value; 4) the importance of deliberative processes to valuation; and 5)

additional approaches to environmental valuation. This chapter is the result of the Subcommittee's discussions on this issue. The Subcommittees conclusions for each of the areas considered suggest the following:

Context is critical to valuation. Important contextual issues include: 1) the type of environmental management action being considered; 2) societal norms and objectives that underlie the action being considered; 3) the spatial and temporal scope of the issue; 4) the identity of stakeholders; and 5) the range of technological and behavioral options available to control the problem.

The norms, principles, and objectives of society represent its values, provide the reason for addressing environmental problems, and provide guidance in identifying the goals of environmental management actions. The benefit of an action reflects its contribution toward the achievement of those goals. Determining benefits cannot be separated from the need to reach agreement on goals.

Fairness, sustainability, and other values, as well as economic efficiency, form goals that should be recognized and explicitly addressed in environmental resource decision-making.

For decision-making in a governmental context, environmental valuation is recognized as an anthropocentric exercise. People care for things for different reasons and in different ways. Things are valued in themselves (as ends) and instrumentally (as a means for the satisfaction of other ends). People care about some things because they desire them or because they contribute to their happiness, and other things have value because people judge them to be good or important in their own right.

Some analysis techniques reveal people's preferences for one or another thing through empirical study. Other techniques attempt to elicit preferences through interaction with individuals. Careful and unbiased interactions with individuals permit us to learn the qualitative dimensions of their value statements.

A basic premise of welfare economics is that economic values are based on individuals' preferences and that people know their preferences. Where individuals are ignorant of the roles of ecological functions in contributing to valued service flows, it may be necessary to use experts' knowledge of the functioning of environmental systems as an input in the valuation process. In principle, the economic valuation framework can be utilized to define and measure the economic values of changes in the functions and services of ecological systems that affect individuals' welfare either directly or indirectly. However, this framework may be difficult to implement in practice where the relationship between the function and the service flow to individuals is indirect or subtle.

Economic approaches, while consistent and coherent frameworks for valuation, are not mechanisms for producing "the answer" since they may omit trans-economic values that may be important, may include some elements that are difficult or impossible to estimate, and may employ preference elicitation processes that are incomplete.

Not all benefits or costs can be easily quantified. There is a need for qualitative methods to provide valuation measures for decision-makers to use in solving complex ecosystem problems. Care must be taken to assure that quantitative factors do not dominate important qualitative factors.

To be most useful, valuation approaches, methods, and information should be made as explicit as possible. These analyses are best used to inform, but not dictate, decisions related to environmental protection policies, programs, and research.

Deliberation among interested parties, though not required in all or perhaps most decision-making situations, is often useful for eliciting information on important goals, on "best judgment" on uncertain relationships, on measures for judging goal attainment, and on value tradeoffs that must be made in decision-making and implementation. Depending upon the sufficiency of knowledge available to address the problem at hand, and the extent of agreement that might exist on values involved in the issue, one of the following

types of deliberation may be useful during decision-making: oversight deliberation, stakeholder deliberation, expert deliberation, or integrated deliberation.

The relationship between energy and other inputs and outputs (including residuals that stress the environment) is not immutable. The optimal level of environmental quality may be underestimated if too narrow and static a picture of response to regulatory pressures is taken as the basis for regulatory decisions. Consequently, opportunities for increases in social welfare may be lost. Research is needed to determine how best to align the regulatory process with the opportunities for minimum cost compliance. It is also needed to determine how dynamic adjustments can be incorporated better into the process of estimating the prospective costs and benefits of possible actions.

Decision-making must proceed in a timely manner. Even though there is much uncertainty associated with the factors important to decision-making, decision-makers cannot wait for certainty to make and implement decisions. A process of adaptive management can allow decisions to be made and implemented in the face of uncertainty. Knowledge gained from implementation experience and research provides feedback with which to revisit and revise past decisions.

It is important that the total U.S. investment in environmental protection be applied effectively and efficiently. Rather than simply address stressors in order of some scientifically derived risk ranking, multiple risks should be considered simultaneously, using both quantitative and qualitative factors, to determine the optimal way to address the full suite of problems that are in need of attention.

The Subcommittee's work confirms the challenges and complexities of environmental valuation exercises and the environmental management actions based on those values. The Subcommittee recommends that expanded, rich, and complex processes be employed to characterize environmental values more fully. These processes should involve interaction and deliberation among scientists, decision makers, and other stakeholders in order to identify goals, define endpoints to characterize those goals, clarify important but uncertain relationships, and to implement

approaches to achieve those goals. Processes should include on-going dialogue and adjustment, and they should consider: 1) why people care for the things they do; 2) the appropriate use of deliberative processes to elicit preferences and the rationale for them; 3) economic valuation frameworks to define and measure the economic value of changes in environmental systems functions and services affecting individual welfare; and 4) presentation of available physical or other quantitative measures, or qualitative descriptions of the effects of alternative actions when costs and benefits are not fully captured by monetary measures. The Subcommittee recognizes that environmental valuation remains a craft embedded in political processes and that much additional research is needed in all areas that are important to estimating the benefits and costs of environmental management action.

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PART IV INPUTS TO ENVIRONMENTAL DECISION-MAKING:

RISK REDUCTION APPROACHES

PART IV—INPUTS TO ENVIRONMENTAL DECISION-MAKING: RISK REDUCTION APPROACHES

Preface

Previous chapters have discussed approaches for assessing and comparing environmental risks and the implications of various risk reduction goals and choices on economic or other measures of welfare. These analyses are important tasks in the *Problem Formulation* and *Analysis and Decision-making* phases of the IED framework. An additional set of analyses required by the framework, however, relates to the overall design and selection of risk management scenarios to address "the problem." This latter set of analyses, including identification and selection of specific risk management approaches, is the focus of the following chapter. The IED framework calls for preliminary consideration of risk reduction options during Problem Formulation, but the most in-depth consideration of options occurs during Phase II when the environmental problem, or set of problems, and associated environmental goals, have been defined. Options are analyzed with regard to their potential to reduce single and multiple risks of concern, associated costs, sustainability, equity, and other criteria specified by the IED participants.

The Risk Reduction Options Subcommittee (RROS) was composed of individuals with broad experience in the various engineering and non-engineering approaches to risk reduction so that a wide range of management tools would be considered. The group was charged with developing a process for identifying the most effective risk management approaches for a variety of types of risk problems that might confront a decision-maker. To meet this charge, the RROS formed subgroups to consider options analysis from three different perspectives: 1) for a stressor-based problem (e.g., ozone); 2) for a geographically based problem (e.g., risks associated with an urban area); and 3) for a media-based problem (e.g., contaminated groundwater).

The resulting 10-step method for identifying, screening, and selecting risk reduction options is the subject of Chapter 6, and is summarized in Figure 6-1. The ten steps are:

Step 1: Define the problem

Step 2: Develop background information
Step 3: Identify the spectrum of risk reduction options
Step 4: Establish screening criteria
Step 5: Screen potential risk reduction options
Step 6: Evaluate remaining risk reduction options
Step 7: Refine the options
Step 8: Select an option
Step 9: Document the process
Step 10: Quantify option effectiveness

In relation to the IED framework, Steps 1 through 4 are primarily the province of the Problem Formulation Phase, whereas Steps 5 through 9 relate primarily to the Analysis and Decision-making Phase. As noted in the IED framework, however, some iteration is inevitable between Phases I and II. The final step, Step 10, occurs during the Implementation and Performance Evaluation Phase of the IED framework.

The first step in the 10-step method is to define the problem. The chapter articulates the importance of a clear problem statement, including specific goals for what/whose risk is to be reduced and by how much. Clear environmental goals, with explicit statement of the relationships and the potential tradeoffs among goals, are the foundation from which objectives for the risk reduction program can be derived. In addition to information on the nature of the risks (e.g., location, severity, specific subpopulations or endpoints affected, and exposure media), the chapter discusses other apects of Problem Formulation that are particularly critical for the design of a risk reduction program. For example, the desired state or goal must be stated in measurable terms and methods for measuring improvements toward the goal (i.e., indirect measures such as stressor levels or environmental outcome measures such as reduction in a specific adverse health effect) must be specified. It is also essential to identify likely constraints on possible solutions (e.g., budget, time, jurisdictional authority, and legality) that will help establish screening criteria for the risk reduction options.

The chapter then goes on to describe the remaining 9 steps in the methodology, from development of background information and identification and screening of potential risk reduction options, to selection of an option that best meets the screening

and selection criteria and documentation of the decision. Various decision analysis methods are discussed briefly, including benefit/cost analysis (described more fully in Chapter 4), matrix qualitative ranking, and multi-attribute decision procedures. The last step in the RROS method, quantifying the effectiveness of the risk reduction program, is analogous to the *Performance Evaluation* phase of the IED framework. The topic of performance evaluation and report cards is discussed further in Part V of this report.

In addition to describing a comprehensive decision process for selecting risk reduction options, the chapter places special emphasis on the types of environmental problems best or least suited to different risk reduction tools.

Although the chapter often refers to "the problem solver," integrated environmental decision-making requires that many types of participants communicate and contribute to problem formulation and solution. In addition, much of the chapter is written around the assumption that the problem and the constraints surrounding its solution have already been specified and that the task at hand is to choose among approaches for risk reduction. It is worth emphasizing, however, that the IED framework recognizes the important role of preliminary options analysis during Problem Formulation, so that problems are not just defined on the basis of risk and our interest in reducing risk, but also take into account our ability to reduce risk within likely constraints.

The methodology for developing and selecting risk reduction options is discussed primarily in terms of reduction of a particular risk, e.g., one associated with a particular chemical stressor or source. The chapter notes that although it is often best to address risks with a combination of risk reduction tools, this is often not done because of inadequate information on the multiple sources of a stressor and their relative contribution to total risk. Extending the methodology to an integrated problem set containing risks from multiple stressors (e.g., those experienced by a particular community) will further increase the complexity of the analysis. Thus, it will be important to aggregate or disaggregate the problem set (e.g., using "root cause" or "common source/common pathway" analysis) so that analysis of risk reduction options is more manageable. The chapter also notes that the complexity of the analysis, including the screening and selection of options, increases greatly with an increasing number of criteria against which options are to be "optimized."

In summary, when applying the 10-step process to an integrated problem set, consisting of multiple stressors, sources, or endpoints, the following issues arise:

1) **Aggregation** is helpful in identifying multiple stressors that may have a common set of risk reduction options with the objective of selecting a set of options that will provide the most risk reduction for the set of risks being analyzed. Practically speaking, it will not be possible to optimize risk reduction over all stressors of concern considered at once. For this reason, screening and aggregation of stressors into manageable subsets should be driven by analysis of common aspects of stressors, roots causes, and/or activities.

2) In order to compare risk reduction across sets of options and sets of risks, and to evaluate risk reduction per unit cost, it is critical to have a **common measurement of risk or common denominator for all risks**. In many cases, comparisons of risks and risk reduction under different scenarios will involve unlike risks (e.g., cancer risk in humans vs. chronic health effects vs. effects on wildlife populations), even where those risks have a common "root cause" (e.g., a single stressor or source).

3) **Uncertainty** of the analysis is likely to increase as a broader set of options is considered for more than one stressor. Sources of this uncertainty include the relative contribution of different stressors/sources to the total aggregate risk; the effectiveness of combined options for reducing aggregate risk; and the benefit/cost, equity, or other tradeoffs involved in addressing groups of risks. Some types of options are easier to implement when uncertainty is high (e.g., communication and education and environmental management systems, rather than regulations/enforcement); the extent of uncertainty associated with a decision may affect the balance of options selected.

CHAPTER 6. RISK REDUCTION OPTIONS

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CHAPTER 6. RISK REDUCTION OPTIONS

6.1 Introduction and Approach

The focus of this chapter is on the identification and selection of risk reduction options to solve an environmental problem or set of problems. The methodologies and thinking in this chapter are a product of the Risk Reduction Options Subcommittee (RROS) and are articulated in the 10-step decision process that forms the backbone of the discussion.

EPA, state, and local regulators have embraced the concept of risk-based environmental decision-making, including the need to prioritize environmental problem-solving efforts. Nonetheless, the RROS recognized that regulatory decision-makers face many constraints in determining risk priorities and in selecting risk management options, including constraints imposed by implementing statutes and by Congress. The Subcommittee defined its goal as helping regulatory officials maximize the risk reduction achieved for any fixed amount of resources, whether those resources are public or private. This chapter, therefore, suggests a methodology for achieving improved risk reduction outcomes, rather than recommending risk reduction solutions for specific environmental problems.

The goal of the RROS was to develop a methodology that would: (1) identify an appropriate set of options for any given human health or environmental problem, (2) create a reproducible and documented decision process where risk reduction decisions are optimized based on explicit decision-maker policy choices, and (3) produce risk reduction decisions transparent to interested parties.

At the first meeting, the RROS explored different approaches to risk reduction available to regulatory or community decision-makers. The approaches fall into one of the following seven categories which may not be completely independent.

a) Communication and education approaches, including information or data dissemination to consumers and communities, options that provide technical

assistance and technology transfer to emission producers, and options that provide information to government(s);

b) Enforcement approaches, including government-implemented options to obtain improved compliance with existing regulations as well as options that involve non-government parties in enforcement activities;

c) Conventional and innovative engineering approaches, including options where technology for achieving pollution prevention, waste minimization, or end of pipe control is communicated, encouraged, or mandated;

d) International and intergovernmental cooperation, including both formal and informal cooperation with other U.S. and non-U.S. government agencies and non-government organizations;

e) Environmental management systems, including options that focus on communication, regulation, and/or market incentives for improved management systems which will reduce risks/liabilities and improve compliance. Management systems cover the areas of setting performance expectations, defining roles and responsibilities, identifying and managing risks, ensuring communication and training, developing implementing procedures, and assessing/measuring performance.

f) Market incentives, including a broad range of options that can be implemented independently or in combination with regulation, communications, or enforcement options. These options can include marketable permits, deposit/refund systems, fees and taxes, subsidies and tax credits, differential regulatory requirements based on performance, government procurement approaches, etc.; and

g) Regulation including mandated technologies, performance levels, risk levels, specific product composition, specific activity requirements, specific information disclosure requirements, and so forth.

Knowledgeable experts in each of the seven areas were invited to share their experience with these potential risk reduction tools. Each expert was asked to address where the option has been used, what lessons have been learned about where it works well and where it doesn't, existing obstacles to its use, biggest advantages and disadvantages to the approach, the future directions for the approach, and risk areas that could optimally benefit from the approach. These presentations and opportunities for interaction provided Subcommittee members with a common base of information for methodology development.

In order to pursue methodology development, the RROS began by dividing into three subgroups. Each subgroup selected a different approach to a human health/environmental problem. One subgroup selected ozone, a stressor-based approach. Another subgroup selected Elizabeth, New Jersey, an urban geographic-based or location based approach. The third group selected groundwater, a media-based approach. All three groups had the same goal; to develop a risk reduction decision methodology that policy-makers could utilize and interested parties would find transparent. By selecting examples of three very different types of problem sets, it was anticipated that a universal methodology that could work for any human health/environmental problem would result.

Once the three subgroups completed their work, a single common ten-step methodology was jointly developed by the full RROS. That methodology was tested against a fourth environmental area of concern, namely, "How can EPA best implement the tolerance-setting requirements laid out in the Food Quality Protection Act of 1995?"

The overall risk reduction methodology devised by the RROS contains ten steps, summarized in Figure 6-1, and discussed in detail in subsequent sections. The ten steps are:

1. Define the problem,
2. Develop background information,
3. Identify the spectrum of risk reduction options,
4. Establish screening and prioritization criteria,
5. Screen and prioritize potential risk reduction options,
6. Evaluate the remaining risk reduction options,
7. Optimize the options,
8. Select an option,
9. Document the process, and
10. Quantify option effectiveness.

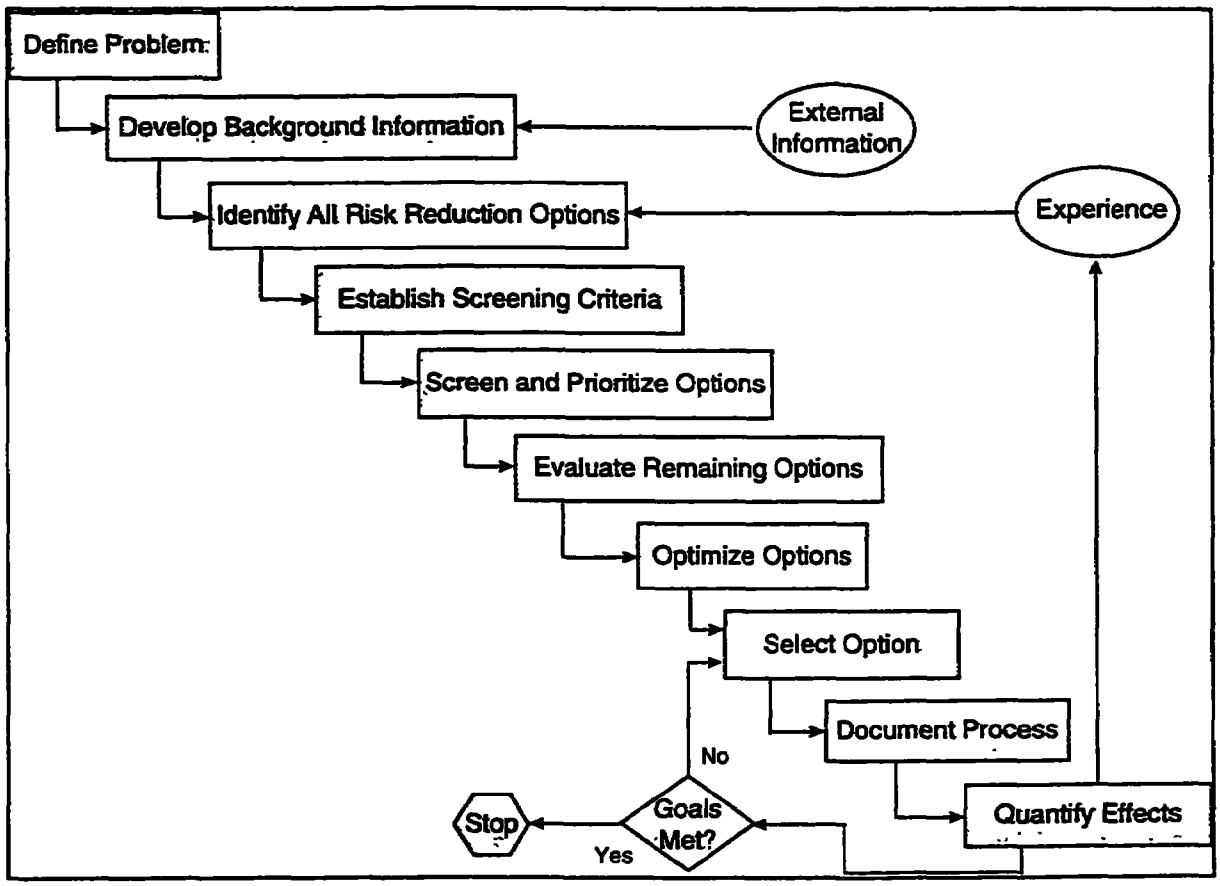

Figure 6-1. Risk Reduction Option Selection Methodology

The RROS methodology has multiple applications. First, if a decision-maker is faced with a specified environmental area of concern, he/she can apply the methodology to obtain the best risk reduction outcome for that problem. A single area of environmental concern may be expressed in terms of: a) individual stressors, such as heavy metals or particulates; b) activities, such as energy use or transportation; c) specific industries; or d) geographic areas, such as the Gulf of Mexico or the greater Chicago area. The RROS methodology provides a formal approach for evaluating

various risk reduction options so that the decision-maker can select an outcome that maximally achieves goals while considering existing constraints.

Another application of the methodology is to select a single risk reduction outcome that will maximize cost-effective risk reduction for a number of different environmental areas of concern. For example, the decision-maker may want to identify which type of risk reduction outcome would maximize risk reduction for a set of stressors. Or, the decision-maker may want to identify which type of risk reduction option would maximize risk reduction for certain activities, such as energy utilization, transportation, and agriculture.

A third application of the methodology is to compare the potential risk reduction outcomes for a several different environmental areas of concern and select the one risk reduction option that maximally achieves goals while considering constraints.

The comparison of risk reduction options is contingent upon the ability to quantify, or measure, risk. Qualitative risk evaluations *do not* work well in this methodology.

6.2 Define the Problem

Before attempting to solve a problem it is necessary to develop a clear definition of the problem. People often minimize the importance of defining a problem for which they seek a solution. They desire to get right to problem solving, without first making sure that they understand completely and precisely what needs to be corrected or improved. In the risk reduction context, failure to adequately define the problem can result in a sub-optimal solution or possibly a solution that does not work at all.

The assumption is that at this stage the problem has been identified. The challenge is simply to choose among various approaches to reducing risk for a specific problem. It would be easy, therefore, to jump directly into problem solving or to a solution.

This section describes the need for a clear articulation of the problem. It also discusses the four components of a clear problem statement, including the use of dis-aggregation (breaking the problem apart), the need for stakeholder input, and the need for coordination among and within governments.

Today's environmental problems are extremely complex and interrelated. For example, exposure to lead may come from many sources, some of which are interdependent. Inhalation of volatilized lead in air, ingestion of lead-contaminated drinking water, and ingestion by children of paint chips containing lead are but a few of the sources of lead exposure. Exposure to pesticides presents a similar range of sources: workplace exposure experienced by manufacturing workers, formulators and applicators; incidental exposures in lawns, golf courses or vegetable gardens; exposure to grocery produce; and consumption of groundwater contaminated with pesticides.

Given this complexity, pursuit of a solution without understanding precisely what needs improvement will be ineffective. For example, does one seek to reduce the lead burden overall, or is the interest in reducing (at least) the lead exposure of children, who are believed to be more susceptible to its adverse effects? Understanding these issues allows one to focus on the issues that will yield the optimum results. It will allow the identification of the root cause of the problem so that solutions can be tailored accordingly.

Problem definition is also essential in understanding and articulating the objectives, or improvement goals, for the risk reduction program. These aspects of the issue must be clearly stated as part of the problem definition as well. It is a tautology to say that the over-arching objective of any risk reduction project is to reduce risk and leave it at that. In any program designed to reduce a risk, or set of risks, there will be complementary and competing objectives. All must be articulated, and competing goals prioritized, before one can design a program to achieve them.

The program objectives must also be measurable. For example, an objective for a lead reduction program might be: reduce the blood lead level in children in the U.S. to a level at which adverse effects are not likely to occur. Likewise, a consumer

risk reduction program for residual pesticides might seek to reduce the level of residual pesticides to a level that minimizes adverse effects based upon science's current definition of what level is acceptable.

Without a clear understanding of the objective and the process for measuring progress, it will be impossible to determine whether the program is effective. In other words, these factors are crucial to defining success. Congress, environmental groups and others frequently berate EPA for failing to show positive results from its programs. This stems, in large part, from a lack of adequate criteria for defining program success, or, in the case of some programs, using the wrong measurement criteria. The problem-solver must clearly understand the objectives of the program if he/she hopes to design a successful solution. Without this, developing a solution becomes a game of "pin the tail on the donkey," in which the donkey's location is not known until the tail (the solution) has been pinned on and the blindfold removed.

The overall objective of the problem definition phase is to develop a specific, narrowly focused statement of what is wrong with the current state, and to provide a clear understanding of the desired state, or goal. The objective of this phase is to move from a version of reality that defies resolution to a version along the pathway to success.

To accomplish this, one needs to proceed through the six-step process set forth below.

a) Identify the elements of the problem context.

b) Understand and, if possible quantify, the magnitude of the harm.

c) Identify the goal of the risk reduction program or desired state after implementation.

d) Identify any constraints, or hard limitations on the solution.

e) Define the affected population.

f) Narrow the problem to its essence, or core elements

The first step is to identify the elements or variables of a problem context. This might also be characterized as understanding the nature of the problem. What is it that needs to be improved? Is it reducing environmental harm, improving health, or improving visibility, for example.

The second step involves an attempt to quantify, or at least understand qualitatively, the magnitude of the harm. In addition to these fundamental questions, the problem-solver must understand the characteristics of the problem context. Is the problem site-specific, such as a waste site; or is it broad-based, such as nitrate runoff into estuaries. What are the sources of the risk and the nature of those sources? Do they involve multiple environmental media, or are they limited to a particular environmental media? Are they large quantity, discrete sources or small quantity, diffuse sources? What are the relationships, if any, among these variables? Answers to these questions will significantly affect the structure of an effective risk reduction program. For example, large quantity, discrete sources are inherently more manageable through direct regulatory approaches than are small diffuse sources.

Third, while gathering information about the nature of the problem, one should also gather information about the desired outcome. Specifically, by looking beyond the solution to the desired state, the goal of the risk reduction program can be identified. This understanding must come early in the process if the problem is to be defined properly. In addition, to be able to measure progress toward the goal, the desired state must be articulated in measurable terms.

The fourth step is to identify and articulate any constraints on the solution. These are necessary to establish the screening criteria for risk reduction options. Some likely solution options constraints include: budget, time, jurisdictional authority of the implementing agency and the legality of the proposed solution. For example, if it is desired to achieve the risk reduction within 5 years, that should be identified as a constraint so that solutions that will take more than 5 years will be eliminated in the screening stage. Jurisdictional authority is one screening criterion that must be used cautiously. One of the key recommendations of this section is that governmental

agencies coordinate and work more closely with one another. Therefore, one would not want to screen out options simply because multiple agencies must be involved. There are, however, circumstances in which Congress will not have delegated adequate authority to any agency to implement a particular solution. In those cases, it will be necessary to limit the risk reduction options considered to those within the scope of?? [DFO note: some text missing here?]

The legality of the proposed solution could become an issue in a variety of ways. The proposed solution might not be capable of receiving the necessary licenses or permits, or there might be antitrust concerns, particularly in the context of market-based solutions.

The fifth step in the process is to define the affected population. For risks that affect human health, this requires a clear understanding of the receptor population. Are the risks imposed on a population in a specific geographical area, near a waste site, for example? Or, do the risks cross geographical boundaries, such as in the case of pesticides? For risks that only affect the environment, what is the affected ecosystem? Who are the users of the resource?

Properly defining the problem requires divergent thinking, to insure that all of the variables are explored. This requires solicitation of inputs from diverse stakeholders and experts, including the affected population and the users of the natural resource, other governments (multi-national and state) and other agencies, and the potentially regulated community.

One must balance the desire for broad stakeholder involvement with the recognition that there is an associated process efficiency cost. Allowing for public and other inputs to any process necessarily takes time. For problems to which there is a need or desire to move quickly, it may be necessary to allow limited, or in some cases no, public input. For example, cleaning up an oil spill, or addressing an imminent and substantial endangerment under the Superfund Program, requires expeditious action and therefore may not allow for public input because a lack of action may exacerbate the problem. Even in the case of problems for which longer-term solutions are appropriate, the cost in time and resources must be considered in determining how

extensive a stakeholder input process to devise. In general, however, appropriate stakeholder input strengthens the overall decision-making process. Important considerations for selecting the type and extent of deliberation for a particular decision are discussed more fully in Chapter 5.

Beyond simple stakeholder input, it is critically important that the agency leading the risk reduction effort coordinates with and involves other agencies that may have overlapping or related jurisdiction over the problem. It is so often the case today that one federal agency is not fully coordinated with others when developing programs to solve problems. When that happens, at best sub-optimal solutions are the outcome. At worst, the program is a failure. Therefore, in the course of defining the problem, the lead agency should consult at multiple points in the process with any and all relevant agencies. This should include not only federal agencies, but state agencies, other national government agencies and international bodies as well.

After developing a broad array of elements of the problem, the problem solver must proceed to the sixth step: identifying those elements that constitute the essence of the problem--*the real problem*. At this stage, it is necessary to *narrow* the problem definition so that it becomes one of manageable dimension and to ask if it is effective to deal with many contributing problems, or if they should be separated into multiple problems. Moreover, one must consider whether the problem should be addressed at a higher level.

This presents squarely the issue of aggregation and dis-aggregation of problems, terms describing the process of combining common or hierarchical problems, and, alternatively, separating them into smaller parts. Aggregation requires that one deal with multiple problems, usually at a higher level. This may be appropriate, particularly if the root cause(s) of the problems are common or related. Alternatively, it may be better to dis-aggregate problems to better understand the root cause(s) of each problem more precisely. Dis-aggregation may also be necessary to create a problem that is resolvable, as opposed to simply a jumble of confusing issues. Problem aggregation and dis-aggregation may need to be an iterative process. The problem-solver may find that after he/she defines the problem and begins preliminary work on solutions, the problem needs to be dis-aggregated. This

takes him/her back to the problem definition stage, where he/she must formulate a workable problem definition once again.

Problem definition may require the problem solver to circle through the 6-step process repeatedly. The result, however, will be an optimum problem definition.

The output of this process should be a clear, focused statement of the problem that contains the following elements. First, it should contain a clear statement of the current situation. This should also contain a clear articulation of the problem, or what is wrong with the current state. Second, it should contain an explanation of the desired state. This must be stated in measurable terms. If possible, the problem definition should also describe the method or methods for measuring improvements toward the desired state.

6.3 Develop Background Information

It is prudent to consider the larger problem in which the risk reduction opportunity is imbedded before thinking about information needed to consider the merits of a specific risk reduction option. For example, if the risk reduction option under consideration is subsidizing the construction of radon resistant homes, the process should begin with an analysis of the radon exposure problem. What has been done to date with the problem within the Agency or within other governmental and non-governmental organizations? Why hasn't the problem been solved or at least substantially reduced? If previous attempts have been made to reduce the risk, how much did different management options contribute toward meeting the goals? Did these options take longer than anticipated to accomplish the task? How much did scientific uncertainty contribute to undermining management options? If it is a new problem, why has it only recently come to attention? What are the key elements of the problem that need to be addressed before a solution can be devised? What might be learned about the problem in the near future that may change the acceptability of a risk reduction option?

After considering these contextual questions, the next action is to consider what specific information about toxins, engineering, economics, geography, psychology,

law, and other subjects are necessary to start a process that will lead to assessing the utility of a specific policy or implementation option. Does this information exist? If the answer is yes, who has the data? How can it be obtained, and in what forms? Is it reliable? If it is not highly reliable because of the way it was gathered or recorded, and/or if the scientific basis supporting it is shaky, are there available time and financial resources to gather high quality data? Or are there other data that can substitute for the best data? Can professional experience or the judgment of informed members of the public substitute for what is uncertain, or at least reduce the level of uncertainty in the data?

At the end of the process, one should know:

a) How the problem came to exist,

b) How it has changed over time and across space,

c) Who has taken policy or implementation actions to date and what their results have been,

d) What are the alternative risk reduction options,

e) Where the most promising places in the casual sequence to introduce risk reduction options exist,

f) What the opportunity is for a multi-stage hybrid risk reduction option, and

g) What forces are likely to constrain each potential risk reduction option.

There are two general categories of information needed to screen options.

Risk Management Information
Spatial and temporal scale of the problem.
Risk reduction objective.
List of identified options.
Screening factors and constraints.
Decision-maker(s) identified and committed to the project
Prospective implementer of selected options.

Options Performance Information
Chance of success of an option within a given set of constraints.
Chance of success of composite options within a given constraint (including synergistic effects)

Information on risk management scenarios would be collected during the stages that precede screening. This information is integrated into the work done on problem definition, development of key background information, and identification of a full range of risk reduction options.

Options performance information is required to estimate whether an option is likely to be successful within a given set of constraints. Such a determination can be based on one or more of the following information bases; expert opinion, previous experience, and predictive methods including mathematical modeling. Using a site-specific case as an example, one may consider various treatment options (nested in the engineering category of options) for reducing contaminant migration source terms at a contaminated site. As determined by the kinetics of physicochemical/biological phenomena and other implementation factors that apply to a given technology, a minimum remediation time is usually required. Thus, the use of quantitative methods to assess whether or not the time for implementing the technology meets the time requirement stated in the risk reduction objective would be useful. Quantitative methods for such assessments can be deterministic or probabilistic, but they can be structured to estimate the reliability of an option.

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6.4 Identify the Spectrum of Risk Reduction Options

The protections delivered by the U.S. environmental management system derive largely from technology-based or harm-based emission limitations although the system also relies, to a lesser extent, on other tools such as market incentives and taxation. The past five years have seen a contentious debate about the structure and composition of this system. Critics of the system charge that the tools commonly used for emission limitations, "command and control regulations," are inefficient from the standpoint of delivering benefits in an optimally cost-effective way. Specifically, the system has been charged with:

a) focusing too much on end-of-the pipe;

b) ignoring the fact that some dischargers can reduce pollution more cheaply than others and that some areas have more serious pollution problems than others;

c) blocking opportunities available to achieve emission limitations using cheaper alternatives for compliance; and

d) imposing more stringent controls on new plants than on existing ones, a dichotomy that creates incentives to keep older, less efficient, higher polluting plants in service (Glicksman and Chapman, 199?).

"Command and control" approaches have also been faulted with causing a serious misallocation of resources, in that the regulations may be directed at trivial risks while more significant risks sometimes remain unaddressed (FN85 in Glicksman?).

Critics of conventional regulatory approaches have generally argued for the substitution of economic incentives, such as marketable permits, effluent taxes or charges, or subsidies, to deliver more efficient pollution control (citatation in Glicksman, ??). Although these economic tools have been used occasionally, in the past EPA has been charged with being generally biased against new approaches,

such as market based approaches, that could conceivably achieve better reductions more cost effectively (Report on Strategic Options Subcommittee, Appendix C, RR, 1990).

There is little dispute that there are severe inherent limits in the extent of protection that the existing regulatory structure can deliver. The large number of processes and substances that must be regulated, the burden of proof to justify proposed regulations, the information requirements to design the regulations, the time and cost of issuing and enforcing permits all conspire against solving the still daunting environmental problems that we face with conventional approaches. Many are concerned that the system leads to excessive delay in addressing important environmental hazards.

To date, the debate about delivering environmental protection has focused nearly exclusively on the strengths and weaknesses of specific risk reduction tools, such as technology-based standards or economic incentive systems. It has not focused on the sometimes sharp differences in the nature of specific environmental problems to be solved using these tools. These differences among types of problems, once properly considered, suggest that no single risk reduction tool is superior across the board in addressing all types of environmental risks. No tool so frequently holds superior promise that the regulator should assume it can be applied to the problem at hand without considerable analysis. From this perspective, then, other approaches must be considered, including market based incentives, non-technology-based standards, more public disclosure, or less "command and control" -- without references to the types of problems remaining for the Agency to address -- do not engage the debate where the problem really lies, which is in matching specific environmental risks with the best possible risk reduction options.

The Subcommittee firmly believes that each environmental risk should first be examined for certain critical elements prior to selecting a suitable set of risk reduction options. In addition, our review of the application of various risk reduction options reveals that the problems EPA has experienced in effectively controlling some environmental risks may stem more from insufficient information regarding types and number of sources of emissions contributing to the problem than from inherent

limitations of the option selected. This inadequate understanding of the sources can result in a sub-optimal or ineffective solution. For example, Amoco Oil Company did a study of its refinery in Yorktown, Virginia that led to a widely publicized critique of the limits of the conventional air emissions regulatory system (reference??). The major finding of the Amoco study was a very large source of fugitive benzene emissions that had been unknown both to the regulators and Amoco prior to the investigation. Without complete understanding of the sources to be addressed at this facility, however, no risk reduction option in EPA's toolbox would effectively address emissions at this plant. Thus, a complete understanding of the sources to be regulated is at least as important, if not more important, than the ultimate selection of the risk reduction option.

Many environmental risks are best addressed with a combination of risk reduction tools. Under such an approach, certain contributors to the total risk might be best controlled under one approach, while other sources contributing to the same risk could be best addressed by another. Because the types and magnitude of contributors to certain risks are so poorly understood, complex, or diverse, the application of a mixture of risk reduction tools to decrease an environmental hazard is uncommon.

This section describes the full range of risk reduction options currently available to achieve environmental and public health protection. Unlike several other reviews (OTA: Environmental Policy Tools, 1995; McGarity 46 Law and Contemp. Probs 159, Summer 1983) addressing this topic, this approach places special emphasis on the types of environmental problems (or elements of problems) best or least suited to a particular risk reduction tool wherever possible. The chapter attempts to identify the core problems where the tool works well, and where the tool is less effective. Table 6-1 at the end of the section summarizes this information.

6.4.1 Communication/Education

Information reporting is a communication/education instrument that requires regulated entities to provide specified types of information to an agency which in turn makes it available to the general public. Typically, the information concerns facility

emissions or operating characteristics, e.g. spill plans, pollution prevention plans, etc. Information reporting (for public use, as opposed to compliance use) is based on the theory that disclosure of polluting activities by regulated entities will raise public concern, which will then lead them to respond to this concern by changing behavior. The scheme takes advantage of a regulated entity's desire to be a good neighbor and responsible corporate citizen, as well as their fear of adverse publicity or possible loss of sales. In addition, because the public's heightened awareness of polluting activities increases the possibility of new regulations, it provides another type of incentive for regulated entities to pursue pollution reduction strategies.

Information reporting provides less direct assurance than many other tools that goals will be met, because it does not mandate overall pollution limits or place an explicit price on pollution. Furthermore, information reporting will change the behavior of different regulated entities differently. Even if a high percentage the regulated community reports, it is hard to know how many will change behavior to reduce pollution and whether there will be some locations with disproportionately few reductions.

Information reporting programs are being used with increasing frequency. They were sparked by the Bhopal, India disaster, which alerted many in the United States to the need to know more about the chemicals used and stored at industrial facilities. Now, the major federal disclosure requirements are found in Section 313 of the Superfund Amendment and Reauthorization Act (SARA), the Toxics Release Inventory (TRI), which calls for certain manufacturing facilities to submit annual reports on the amounts of listed toxics chemicals released (routinely or accidentally) into the environment. At the state level, California has created Proposition 65, California's safe Drinking Water and Toxic Enforcement Act. It requires public warning of the potential cancer or reproductive effects of 542 listed chemicals either emitted or present in products. This policy has resulted in the installation of warning signs throughout the State of California, including in grocery stores and restaurants. There remain questions as to whether all the signage has truly reduced exposure of Californians to toxins.

Information reporting is most likely to be effective in situations where:

It is possible to convey the information in a simple and accurate manner;

The regulatory target is a product where consumer preferences are driven by environmental concerns, as opposed to exclusively price, quality, performance, and convenience or other resource(s) for purchasing the product;

The link between the product and environmental damage is clear;

The risk, though relatively easy to mitigate, is not on a scale to have been prioritized by government programs; or

Hazards are not due to multiple exposures or toxic hot spots, or where hazard is due to combined factors.

Technical assistance, a communication/education approach, consists of government programs designed to educate private entities and the public to make better environmental choices. It seeks to achieve environmental goals by increasing the understanding of pollution problems and potential solutions. It may take the form of manuals and guidance, training programs and materials, information clearinghouses, hot lines, facility evaluations, and technology R&D (OTA, p. 138). Most technical assistance services are provided at no, or minimal, cost to the user, and participation in programs is typically voluntary.

Some technical assistance programs have been developed in response to congressional mandates, while EPA and other agencies have initiated others. Section 507 of the Clean Air Act, for example, requires states to establish Small Business Stationary Source Technical and Environmental Compliance Assistance Programs. Others have been initiated by EPA or other agencies to help implement mandated environmental programs. For example, section 319 of the Clean Water Act calls for states to manage diffuse non-point sources of water pollution. EPA and the U.S. Dept. of Agriculture have developed extensive guidance documents describing best management practices that non-point sources might use to control their pollution. Other programs do not respond directly to statutory mandates but are derived from the general objective of improving environmental quality. For example, the EPA Green

Lights Program conducts energy audits and makes specific recommendations for energy efficiency in exchange for an agreement from participants to install the recommended lighting systems.

Technical assistance is most effective where polluting entities are not knowledgeable about the environmental consequences of their actions or alternatives available to improve their current environmental practices.

6.4.2 Enforcement

Liability provisions require that polluters pay for the damage they cause by requiring compensation for environmental damage. It is enforced two ways: by common-law theories such as negligence or nuisance, or by statute, particularly the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). For either type of liability, a successful claim typically requires an established causal link between the harm and the pollution, which has been traced back to its source.

Liability differs from regulation in that it engages the responsible parties after-the-fact. Liability rules establish the price of environmental damage, while regulations seek to control the activities that create such damage. As such, liability is a crude instrument for affecting behavior; by itself it cannot ensure that the appropriate amount of care is exercised in all circumstances (Shavell, 1984, in Percival, p. 134). Despite this, the specter of potential liability is widely thought to encourage pollution prevention and responsible waste management ethics, because the dollar amounts involved can be huge, much larger than the costs of preventive behavior in the first instance. As of September 1994, for example, Exxon had already spent $3.4 billion to clean up the spill and settle suits from the Prince William Sound oil spill.

Liability also provides incentives for environmental auditing and other self-appraisals, in order to gauge the potential financial exposure and correct problems before they grow (OTA, p. 125). As a policy tool, liability is particularly attractive when a private actor is in a better position than the government to assess the risks of its activity and to determine the level of care to exercise. On the other

hand, regulation is more efficient than liability for preventing accidents caused by those likely to escape responsibility for their actions; this factor usually weighs in favor of regulation as a means for deterring environmental damage because of the difficulty of proving causation under common law.

Several statutes provide liability schemes to address environmental problems. CERCLA provides for strict retroactive liability for cleanup of hazardous waste dumps. The Oil Pollution Act provides strict liability for natural resource damages, and third party damages caused by petroleum spills. The Clean Water Act makes responsible parties liable for cleanup costs for a spill of hazardous substances into surface waters, capping liability at $50 million unless the discharge was the result of negligence or willful misconduct.

Liability is an effective incentive for environmentally beneficial behavior only to the degree that the decision-maker is concerned that he/she will be held liable. Difficulties involved in proving a causal link between a particular action and damage to a particular plaintiff has limited the effectiveness of this tool for addressing many environmental hazards. Liability works best where:

The contamination is traceable back to one source (harm that is widely dispersed or difficult to trace will not spark adequate incentive to reduce risks);

There is sufficient information on the extent of harm that is caused by a substance or activity to establish a causal link between the harm and the pollution;

The impacts of the contamination are sufficiently concentrated to make a claim worthwhile to the injured party; or

The party responsible for the damage has the capacity to provide compensation for the full amount of harm the actions produce. (If an activity can cause more damage than the actor is capable of repaying, fear of liability will not provide sufficient incentive for private investment in an efficient level of precautions.)

If liability insurance is available (as a matter of fact, it is often required to ensure that compensation will be available for victims of environmental damage), the existence of this insurance can reduce responsible party incentive to prevent damage in the first instance.

6.4.3 Engineering

Engineered solutions are often the first ones considered. The propensity to jump to this conclusion and the proven versatility of engineered solutions has led to the proliferation of end-of-pipe treatment technologies. Only the recent development of pollution prevention strategies has lessened their popularity. Engineered solutions are often very costly and take years to implement.

6.4.4 International and Intergovernmental Cooperation

International and intergovernmental cooperation lends itself to those issues that have large geographical impact. Global warming is an excellent example whereby the cooperation of many world organizations is required to control the emission of greenhouse gases that eventually leads to global warming.

6.4.5 Management Systems

Management systems integrate options to create effective solutions. It starts with establishing organizational principles and ends with the attainment of corporate goals. This option is gaining favor as entities realize that it establishes the basis for getting the most from existing resources.

6.4.6 Market Incentives

Under tradable emissions , a market incentive approach, the government first sets a level of aggregate emissions over a specified time period, consistent with environmental goals by issuing only the number of permits corresponding to that level. The total allowable emissions are then allocated to individual sources through government-issued permits. Unlike conventional permit systems, however, each

regulated entity can buy and sell permits from others. An entity might choose to do so if the relative costs of emissions control make it less expensive to buy (or profitable to sell) a permit from another entity. In theory, trading would continue until the cost of controlling another pound of pollution is the same for all entities and is equal to the cost of a permit.

The most well known applications of emissions trading include: Sulfur dioxide (SO_2) reduction for acid rain control and oxides of nitrogen (NO_x) reduction for urban ozone in Los Angeles. It was considered but rejected to control volatile organic carbon (compounds?) (VOC) emissions (ozone precursors) in the Los Angeles basin. Trading has also been proposed under the NPDES system for effluent discharge trades, in particular for phosphorus reductions in the Dillon Reservoir (Colorado) and the Tar-Pamlico River Basin (North Carolina) and for biochemical oxygen demand (BOD) reduction in the Fox River (Wisconsin). These watershed trading programs are under-utilized to date because there are not disproportionate costs across sources associated with the reductions. Wetland mitigation banking, where agencies authorize distribution of wetlands in return for promise of future enhancement of other wetlands, is also another trading venue. A lack of certainty that the appropriate regulatory agency will approve a "trade" is thought to be the "rate limiter" to trading.

The reductions must be quantifiable to an acceptable level of certainty. This generally means that the regulatory agency must have the capacity to measure pollutant levels in question at the source or as they apply across the system. This includes measuring the baseline pollution level and changes from baseline to allow a source to generate tradable credits. Because establishing actual emission reductions for mobile sources is more difficult than for most stationary industrial sources, it is less clear that emissions trading schemes will achieve intended results where mobile sources are involved.

There must be a large number of sources with significant variations in control costs. If everyone's cost to control emissions is the same, then emission credits will not be a bigger bargain than reducing emissions at the facility. The pollution problem must not involve hot spots. Otherwise, the trading system may result in inadvertently disproportionate exposures. The program must be evasion proof. Actions that

produce credits must be enforceable. The program is especially valuable where expected control costs are very high; this will enhance interest in a more cost-effective approach. The emissions should have a well-defined relationship to the problem being solved.

With pollution charges, a regulated entity is required to pay a set dollar amount for each unit of pollution emitted or disposed. Sources are free to choose whether to emit pollution and pay the charge, or to pay for the installation of controls to reduce the charge. Pollution charges do not set a limit on emissions or production. However, if they are set high enough, and if they vary according to the amount of pollution produced, they can provide significant financial incentives to reduce or eliminate environmentally harmful behavior.

Although charges have commonly been levied against sources as a revenue-raising measure, such as with permit fees, pollution charges set at a level sufficient to change behavior have not been used extensively in the United States. Two exceptions are solid waste management and the charge that was levied on chlorofluorocarbons (CFCs) during their phaseout. Under typical solid waste schemes, companies pay charges that rise as waste volume rises, while most households pay flat fees unrelated to the amount of waste generated. Volume-based charges have been applied to household waste in about 100 jurisdictions across the country as well, however, and they are believed to have been successful in reducing volumes of waste generated in many of these instances (OTA, p. 123).

During the CFC phaseout, users were required to pay a charge per pound of CFC, multiplied by an ozone-depleting factor. The charge was increased in subsequent years. CFC production decreased much more rapidly than originally anticipated, and many attribute this rapid decline to the effectiveness of the CFC tax (OTA, p. 122). Because the tax was used in conjunction with a mandatory phase-out, it is difficult to ascertain the relative importance of each tool in achieving the overall success of this program. Clearly, however, a combined approach of charges and phase-out can be effective in reducing environmentally harmful behavior. It is important to note that the CFC taxes collected by the U.S. government went to general revenue and were directly allocated to environmental programs/improvements.

Subsidies are policy instruments that provide financial assistance to entice entities to change their behavior or help entities having difficulty complying with requirements. Generally, subsidies take the form of grants or low- no-interest loans to municipalities and public entities and preferential tax treatment for private entities. The Clean Water Act, for example, had a long-standing construction grant program that enabled municipalities to comply with Clean Water Act requirements for wastewater treatment plants. This grant program was phased out by the 1987 amendments to the Act and replaced with a state revolving loan fund. The Clean Air Act also authorizes several grant programs. For example, section 105 provides for grants to state and local governments to implement air pollution control programs.

For many years, private companies have been allowed to take accelerated depreciation on investments aimed at reducing water pollution.

6.4.7 Regulation

Harm-based standards are established on the basis of what is required to achieve health or environmental protection goals. EPA typically establishes these standards by determining the amount of the pollutant in the ambient environment that will meet a health or environmental goal set by Congress. This determination involves scientific judgments as to the extent to which different concentrations of the pollutant cause harm. After the Agency establishes an acceptable concentration, a mathematical model is used to calculate an overall allowable pollution load for the region with which this acceptable concentration will not be exceeded. EPA or the state then apportions an acceptable pollutant concentration or loading among the individual sources that it has identified. Harm-based standards are expressed in facility permits as emission rates for the source (mass per unit time period), as a concentration of pollutant in a source's discharge, or as a percentage reduction in emissions from a source.

As a practical matter, EPA's ability to craft a policy that meets a harm-based goal is contingent on a thorough understanding of the sources contributing to the pollution load from the facility or region. Often, the Agency has insufficient knowledge of the number, types, and magnitude of contributors to the problem. Bubble emission

and integrated permitting initiatives have been developed to overcome the lack of knowledge in this area by providing flexibility to regulated entities to alter control levels for various sources within a facility as long as overall limitations are achieved. (The 3M integrated permit for its Hutchinson, Minnesota facility, for example, allows the company to shift emission controls among sources within the facility as long as aggregate VOC control levels are satisfied.) However, neither bubbling not integrated permitting have been widely used to date, again because of the inadequate knowledge of sources to be "bubbled" or because of monitoring and accountability difficulties.

Harm-based standards are notoriously difficult and time-consuming to set because of analytical uncertainties and gaps in available data about both the scientific basis for concern and the demography of the sources contributing to the problem. Uncertainties inherent in predicting the effects of different patterns and levels of environmental releases also pose big problems. Harm based standards also require extensive data on existing ambient pollutant concentrations and health effects, which often are not available.

Nonetheless, harm-based standards, along with technology/design standards, are the most heavily used environmental policy tools in the U.S. Often, the two are used together. The Clean Water Act provides harm-based standards for "water quality limited streams", where industrial sources must comply with stricter pollution control than would otherwise be necessary in order to achieve improved water quality in a highly negatively impacted area. The Clean Air Act National Ambient Air Quality Standards (NAAQS), also exemplify harm-based standards.

In contrast to harm-based standards, technology-based standards derive from the level of control technology Congress expects pollution sources to implement, such as "reasonably available control technology." Technology based standards can be viewed as the opposite of health-based standards, because instead of asking what is needed to protect health or the environment, they ask what is possible to do (Percival, p. 146). Most commonly, these standards are set as a performance level for the regulatory target to meet, without specifying how this should be done (performance standards). They also can specify how a certain facility or piece of machinery should

be designed and engineered (design standards). EPA develops both design and performance standards, although some of its performance standards become de facto design standards. This can occur where EPA's performance regulation is established on a specific technology and, in responding to the target, regulated entities decide the safest course to compliance is to install a specific design technology. EPA's New Source Performance Standards under the Clean Air Act and its various effluent limitations under the Clean Water Act are examples of performance standards, but they are often criticized for limiting acceptable technologies in practice to those that EPA has used as a basis for developing the performance standards (Percival, p. 150).

Disadvantages technology-based standards are that they: a) may cause the expenditure of more or less resources than necessary, b) assume state of technology is knowable, and c) discourage innovation.

Fixed emissions standards based on current assessments of technology are widely faulted for the absence of incentives for further innovation. Laws that force industry to develop innovative solutions by adopting standards more stringent than those attainable by current-available technology overcome this important limitation.

Both the Clean Water Act and the Clean Air Act call for technology-forcing regulations. The effluent limitations in the Clean Water Act, which are based on "best available technology economically achievable," look to transfer state-of-the-art practices in one industry to another in order to achieve reductions beyond what is currently in practice. The Clean Air Act Amendments? of 1990 contains a broad effort to force technology developed for mobile sources, including better tailpipe controls, cleaner fuels, and cleaner engines than existed at the time of passage of the Act.

"Congress's initial approach to technology-forcing regulation was to engage automakers in a high-stakes game of chicken. Title II of the 1970 Clean Air Act mandated emission standards for automobiles that were "a function of the degree of control required, not the degree of technology available today." The Act required automobile manufacturers to slash vehicle emissions by 90 percent in order to be able to continue selling cars in the United States after a specified deadline, subject to a one-year extension. Not surprisingly, the automobile manufacturers fought hard to convince EPA to extend the deadline on the ground that the necessary technology for reducing vehicle emissions was not yet available. After EPA Administrator William Ruckelshaus shocked the industry by refusing to extend the deadline,

the auto industry convinced a court to order him to reconsider. In May 1973, EPA agreed to relax the standards for two years. Meanwhile, GM had a breakthrough in the development of catalytic converter technology, which promised to be a boon to fuel efficiency and performance but would also meet the emission standard required in the Act. Six weeks after EPA extended the deadline, GM announced that all of its 1975 models would be equipped with the devices. "I came out of the whole exercise with catalytic converters as a technological optimist in what the industry can do," said Walsh, the EPA's top auto pollution expert in the 1970's and now a private consultant. "When you give them a challenge, they meet it." (Percival, p. 168-171)

Disadvantages of technology-forcing regulations are that they a) may compel too much monies to be spent in a narrow area and b) require a tremendous amount of information.

A product ban either prohibits or restricts the manufacture, distribution, or use of a substance identified as a problem to human health or the environment. It is most seriously considered where product use is sufficiently damaging that zero use is a desirable outcome. Product bans may be imposed prior to the product's sale and use in commerce, through various pre-market product approval programs that seek to prevent excessively risky products from reaching the marketplace. More publicly visible bans concern themselves with substances already on the market, such as lead or asbestos. Whereas the burden of proof lies with the manufacturer to show that a product is safe for pre-market approval, the burden of proof lies with EPA to show that a product should be banned once it is already on the market .

Product bans and restrictions focus on the commodity itself rather than on polluting byproducts from its manufacturing. As a result, they are used primarily where the hazard is the commodity itself. The U.S. has banned lead in gasoline and paint, asbestos, PCBs, CFCs, and numerous pesticides, including DDT, Aldrin, and Dieldrin.

Product bans are most effective where a single substance or activity is causing a particular environmental problem. They are not effective where the environmental problem has multiple or complex causes, such as low dissolved oxygen concentrations in urban rivers.

A product ban is best used where all of the uses of a product pose unacceptable risks. Otherwise, considerable analytical resources must be set aside to enforce a narrower ban or prevent unauthorized use.

Product bans have superb potential to induce technological innovation by stimulating rapid research on substitutes. However, they are risky when the consequences of a failure to develop an alternative are very high (such as, if we were to contemplate the banning of the internal combustion engine to address urban smog). Their success hinges on the development of genuinely safer alternative products; otherwise the substitute will pose its own hazards.

With a challenge regulation, responsibility for solving a problem is shifted from government decision-makers to the sources themselves. Under this approach, the government establishes targets necessary to solve an environmental/public health problem, with a timetable for implementation. The targets are defined for multiple sources, at the industry sector rather than the individual facility level. These sources are given the collective responsibility for designing and implementing a program that meets the targets. The government specifies an alternative program or sanction that will be triggered if progress towards the targets is not achieved.

Challenge regulations have not yet been extensively adopted in any country and have not been used to date in the United States. EPA's voluntary 33/50 Program (reference?), however, is an example of a pilot program to test this approach. The most widespread use abroad has been in establishing producer responsibility for various forms of wastes to encourage source reduction and recycling. In the German Green Dot program, the government established a regulatory approach outlining industry's obligations to take back packaging from customers. However, it then gave industries the opportunity to establish an alternative recycling program of their own for meeting the targeted rates.

Since industry designs the implementation plan, this tool is not well suited to problems with localized concerns about the effects of the pollution. Thus, it should not be used to reduce pollutants or solve problems for which exposures vary widely across locations.

The challenge is unlikely to be met if it is difficult to fairly allocate responsibility across the target group. Competition among firms may make it difficult to develop an implementation plan.

Some identified options may not be based on technology. They could be policy options. Performance data are rather sparse on non-technological options, and models constructed to estimate very high uncertainties and controversies may characterize their reliabilities within a given constraint. Issues related to the performance of a variety of both technological and policy-level options have been discussed by Davies and Chou (1992), Levin (1990), IINEPA (1996), Norman and Keenan (1996), C and EN (1996), CHEM NEWS (1996), OTA (1995), USEPA (1990) and UN (1993). Performance information on composite options (i.e., ...) is very sparse.

Table 6-1. Conditions Under Which Options Are Most and Least Effective

Option Category	Most Effective	Least Effective
• Communication/Education		
– Info. Reporting	• Regulated entity is concerned • Info. easily conveyed • Consumer pref. Driven by env. Concerns • Clear cause and effect • Risk not prioritized by gov't. • Single source	Entity doesn't value public image
– Tech. Assist.	• Regulated entity is not knowledgeable	
• Enforcement	• Single source • Clear cause and effect • Significant impact • Regulated entity has "deep pockets"	
• Engineering		Non-materiel issues
• Int'l & Intergov't. Cooperation	Broad impact issues	
• Management Systems		Weak organizational structure
• Market Incentives		
– Tradable Emissions	• High control cost • Multiple sources w/ variable control capability • Quantifiable emissions • Fixed sources • Enforceable	
– Pollution Charges		
– Subsidies		
• Regulation		
– Standards		
+ Harm Based	• Well understood sources	• Analytical uncertainties • Data gaps
+ Design		
+ Tech. Forcing		
– Product Bans	• Single substance is cause • All uses have unacceptable outcome	• Multiple sources • Few or no alternatives
– Challenge Regulations		• Localized Concerns • Varying Exposures

6.5 Establish Screening and Prioritization Criteria

The simplest way to evaluate risk reduction options would be to reduce the consequences of specific options to a single measure of effectiveness. (In fact, this is precisely what a classic cost-benefit analysis of a regulatory option seeks to do.) However, it is not easy to achieve consensus on reducing to a single measure the wide range of views and legitimate perspectives that come to bear in formulating risk policy. An alternate approach is to reduce the consequences of a risk reduction option to performance on as small a set of criteria as possible. The Subcommittee concluded that five basic criteria could be used to evaluate risk reduction options: environmental effectiveness, cost, equity, workability, and flexibility. With regard to the first two criteria, we have concluded that it is more transparent to separately consider environmental benefits and environmental costs, rather than to combine them into a single measure of net benefits. Ideally, performance in each of these groupings should be summarized along a single dimension, such as lives-saved or dollars-of-cost.

Environmental effectiveness encompasses the variety of criteria that have to do with the environmental impacts/physical benefits. For the most part, the issue here is to what extent does a regulation accomplish the environmental protection goals that originally motivated the regulation? The focus in on the meaningful environmental consequences of a regulation, rather than simply meeting some arbitrary ambient concentration goal. Another consideration in this category is the likelihood that the environmental goals will be met. As an example, the NAAQS and associated State Implementation Plans for urban ozone would suggest that environmental goals are well served by this regulatory approach. In reality, however, in many cities the goals are often not met. The economic version of this criterion (discussed in Chapter 4) is efficiency, defined as that level of risk reduction which maximizes the surplus of benefits over costs. Therefore, an aggregate measure of environmental benefits is desirable to help compare the performance along this dimension with performance on other evaluative criteria.

Cost is the second grouping and includes a number of factors. Of primary interest are the direct and indirect costs associated with a regulation. ~~excluding the monetized~~

~~value of the environmental benefits that may result from the regulation (which are accounted for under the previous category?).~~

Equity deals with the distribution of costs and benefits from the regulation; i.e., who bears the cost and who benefits. This might involve the distribution of costs and benefits to economic groups (e.g., rich vs. poor), ethnic groups (e.g., blacks vs. whites), or different regions of the country (e.g., Midwest vs. west). Clearly any regulation that involves concentrated costs and benefits will have very different political characteristics than a regulation with more dispersed costs and benefits.

Workability deals with how easy it might be to initiate and implement a regulation. Some regulations have a great deal of administrative burden. Some regulatory approaches are so new as to make their success highly uncertain. Some regulatory approaches are very familiar to regulators and polluters so as to be very workable.

Flexibility refers to how easy a regulation is to change when new information emerges or to adapt to new circumstances.

There are two purposes for the evaluative criteria discussed here. One is to help fine-tune a given option. For instance, if an emission charge has been proposed, there are many choices as to exactly how it will be implemented, what pollutants will be included, what the level of the charge will be, and what kinds of enforcement will be included. Using the five criteria, the option in question can be adjusted, tailored, and honed to do as well as possible against each. This results in an option that performs as well as possible.

The second purpose for the five evaluative criteria is to compare several different approaches to solving a particular environmental problem. Each approach must be fine-tuned. The only question remaining is which approach is the top candidate for implementation.

In order to facilitate the comparison of options, it is important to summarize performance for the criteria along as few dimensions as possible. This may be most

easily done for cost, using a dollar metric. It can also be done for the other criteria, though perhaps not so easily. For instance, the performance of an option on equity grounds could be measured on a scale of 1 to 10, with guidelines to govern the use of particular ratings. On environmental effectiveness, results could be summarized in terms of statistical lives saved or the willingness-to-pay of the population to achieve the level of environmental performance, relative to some baseline.

A more extensive discussion of each of the five evaluative criteria follows. This discussion is not intended to cover all aspects of each but to give an idea of the breadth of issues that are embodied in the criteria.

Environmental effectiveness summarizes how well a risk reduction option performs in terms of its environmental outcomes, which obviously is an important aspect. It is preferable to define environmental performance in physical or health terms (lives saved, species protected), rather than in terms of administrative process measures (e.g., the likelihood of meeting a regulatory target such as no more than three violations of an ambient standard in a year). Outcome measures are not always possible, however, since the physical effects may be difficult to quantify.

Consider the following issues with respect to environmental effectiveness:

a) How does expected performance compare to environmental goals? If the goal is to reduce morbidity and mortality by a certain amount, has this goal been achieved? If the goal is to reduce stressor concentrations by a certain amount, has this goal been achieved? Often there will be no explicit or quantitative environmental goal. One may simply wish to reduce pollutant levels as much as possible (or justified). In this case, there is no explicit end-point.

b) What are the quantifiable physical gains from the regulation? For example, what is the reduction in morbidity and mortality? What are the goals in recreation opportunities (with cleaner rivers, for example)? What are the maintenance goals from a cleaner ambient environment? What is the nature of

species preservation or ecosystem health? The point is not to measure gain in arbitrary units such as how much BOD reduction has been achieved, but in more meaningful terms (if possible).

c) What is the value of the environmental gains? It is sometimes, though not always, possible to quantify the physical gains. This should be in a single metric such as dollars or the number of statistical lives saved and should summarize the diverse environmental consequences of the regulation.

d) What is the probability of success in obtaining the expected environmental gains? Some risk reduction options are more proven than others.

e) Can performance be monitored? One must be able to conclude whether aggregate environmental goals are being met. It is not always easy to monitor. For instance, VOC emissions from small businesses may be very difficult to monitor.

f) When can results be expected? Is the regulation designed for immediate results or will it take a long time to be effective?

g) What are the associated risks? For instance, regulations to reduce sulfur dioxide emissions may contribute to accelerating global warming.

h) Is pollution prevented rather than abated? Although the environmental effects may appear to be the same, it is more desirable to reduce the generation of pollution through process change rather than simply appending an abatement technology (end-of-pipe treatment).

The criterion of cost includes all of the non-environmental costs and benefits of a risk reduction option. The specific way in which costs are measured is discussed in Chapter 4. Questions relevant to this criterion include:

a) What are the direct costs of implementing the risk reduction option? This would include the costs incurred by a polluter in reducing emissions, which might be costs of abatement equipment or costs from adopting more costly processes. Also included would be administrative costs -- costs associated with applying for permits, participating in review procedures, and costs associated with uncertainty of what might emerge from the regulatory process. A third direct cost would be the consumer losses associated with higher product prices (because of higher production costs). It is also important to include the regulatory costs, incurred by the EPA. These would include the costs of formulating and administering the regulation as well as enforcement and monitoring costs.

b) Are costs of residual damage reflected in product prices? In order for consumers to receive an adequate signal regarding how damaging the production of goods might be, the price of those goods should reflect all environmental costs, including the costs of abatement but also including the environmental damage associated with residual pollution. If consumers see product prices that are excessively low, this will generate too much consumption of the polluting product and consequent inefficiencies.

c) Are there incentives to innovate? Some options provide little incentive to be creative and innovate. Often, the option spells out what to do and innovation only results in tightening the requirements. It is important that an option provide incentives for innovation by polluters and by equipment vendors. Often this requires that all or part of the gains from the innovation accrue to the innovator.

d) How much regulatory commitment is involved with the option? If there is little commitment, does that increase costs substantially? When a regulator commits to a risk reduction option, and cannot retreat, then polluters make least cost investments. Without commitment, polluters will take more flexible, but also more costly, approaches to meeting regulations. A lack of commitment on the part of regulators can only increase costs, often with no associated environmental gain.

Equity: Problems arise when any identifiable group disproportionately bears the costs of a selected option. One might also be concerned with benefits falling disproportionally on particular groups; there seem to be fewer political problems when small groups benefit from a governmental action than when small groups bear costs (substantiating reference?).

Basically, what is at issue here is the distribution consequence of a risk reduction option. Several issues arise:

a) What is the incidence of costs? Do any groups disproportionally bear the costs? For example, if a benefit is distributed equally in a population and the cost is applied equally to each individual, lower income families will pay a higher percentage for equal benefit.

b) What is the incidence of benefits? This is a very similar issue as the incidence of costs, although perhaps not as important. Nevertheless, it is important to identify any groups in the population that may disproportionally gain the benefits.

c) Do benefits and costs accrue to the same group? If one group disproportionally bears costs but also reaps benefits, there is less concern than if different groups bear the costs than receive benefits.

d) Are the regulations confiscatory? Polluters may be more opposed to a marketable permits system where permits are auctioned off initially than a system with permits freely distributed. Any regulation that is viewed as excessively confiscatory is more likely to be opposed in the political process.

e) Are benefits and costs concentrated or diffused? The more diffuse the costs and benefits, the less likely it is that specific groups will emerge to oppose the regulation in the political process.

Some risk reduction options work better than others. Workability addresses how easy it will be to implement an option.

a) How large is the experience base with a particular risk reduction option? If an option is a variant of one that has been used many times before, then both the EPA and the polluter will have experience in making it work. In contrast, a new option may be more difficult to implement and work out the problems.

b) How much information does the government need in order to implement the option?? The primary criticism of command-and-control regulation is that the EPA requires an unreasonable amount of information.

c) What are the demands on government? An option that has less demands on government is to be preferred to one that has heavy demands on government, all other things being equal.

d) Is the public involved in the decision process? The more the public is involved as a risk reduction option is implemented, the less likely it will be that unexpected problems develop. Often when the public is excluded, factors that would have been brought to light through public hearings end up causing significant problems during implementation.

e) Are enforcement and monitoring easy? Clearly some regulations are easier to enforce than others. Product bans are straightforward, provided leakage from unregulated areas is not a problem. If there is significant reliance on unobserved actions by polluters (e.g., maintenance), then it will be difficult to ensure that these actions have been taken.

f) Are the financial burdens on government excessive? Some options are cheaper than others to administer. The larger the burden on government, the more difficult to sustain.

g) Are options viewed as excessively intrusive? Some options are politically unpopular because they are viewed as too intrusive. For instance, proposals to use intelligent vehicle systems to automatically charge for road use have been met with privacy objections. Some citizens do not like the idea of their movements being centrally monitored.

No matter how well an option is designed, changes will undoubtedly be needed making flexibility important. It is important to recognize the ease with which an option can be modified to tailor to specific conditions. Several issues emerge.

a) Is the option adaptable to new information? If new information requires that Congress re-draft the appropriate legislation, the option will be more difficult to modify than if changes are automatic. The more readily an option can adapt to changed circumstances, the better it will perform.

b) Does the option promote innovation and diffusion of innovation? This issue was mentioned in the cost-effectiveness grouping. It is important that an option be flexible enough to allow innovation (this is not always the case). It is even more important that the option reward innovation. Innovation is a primary means for increasing environmental effectiveness while reducing costs.

6.6 Screen and Prioritize Potential Risk Reduction Options

Options screening is the process by which a broad range of risk reduction options is reduced to a set of unranked options for more detailed comparative analyses. An option can be single or composite. Screened-in options are those that can be implemented to meet the risk reduction objective within all constraints specified in the problem formulation. A constraint is a specified magnitude of a screening factor. For example, if the budget limitation is $1,000,000, then budget is a screening factor, and the constraint is $1,000,000.

An important utility of the screening stage is that it enables the assessment of a wide array of identified options. The probabilities of option targeting and arbitrary

elimination of options are minimized. Option targeting is a common problem in remedy selection. For example, engineering remedies are more likely to be selected as a remedy for a contaminated site than communication options, regardless of their relative attributes. The screening process provides a systematic method for determining whether each identified option meets specified performance criteria. Depending on the constraints, both categories of risk reduction options may be screened-in for more detailed comparative analysis during subsequent stages of the options selection process.

The application of multi-criteria, multi-objective analyses to a large number of identified options may unduly complicate the options selection process. The screening step provides an opportunity for the identification of potentially unfeasible options, thereby eliminating these options from further consideration. The number of options suitable for comparative analyses in the post-screening stages of the options selection process is then reduced to a more manageable level. In essence, data requirements on options screened-in are proportional to the number of options. The generation of a prioritized (and reduced) list of options for comparative analyses reduces the costs and time required for acquiring relevant data. For example, if the direct financial cost constraint specified for reducing ecosystem risk due to contaminated sediments at a site is $1,000,000 and an effective treatment technology can not be implemented under $3,000,000, that option would be screened out as written. This would eliminate the need to acquire additional data that are required to compare it with other options in subsequent analysis. It should be noted that this option may be restructured and re-screened later in the process.

The screening process satisfies the need to give assurance to stakeholders that a narrow set of options was not pre-targeted. Transparency is an important issue, especially in cases where the implementation of an option is associated with shared risks and costs. Transparency is particularly important when different parties within the impacted communities prefer different options, some of which are rejected for not meeting specified selection criteria. In the selection of a site for a waste management facility, the potential impacts on the surrounding communities vary both spatially and temporally. Consequently, communities proximal to selected sites may feel unfairly treated if the site selection process is not transparent. Because the issue of

environmental equity has gained importance within the past decade, several systematic techniques for site screening have been developed (Wright et al., 1993; Smith, 1996; Frannis, 1993; Knight, 1985; Schwartz et al., 1989; Zeiss and Paddon, 1992). The screening process provides the opportunity for decision-makers to show that no party or segment of the community had been pre-selected to bear a disproportionate hardship.

Some options that are individually undesirable may exhibit increased utility in combination with other options. The screening process provides the decision-maker(s) with the first opportunity to identify options that could be potentially combined.

The specificity of an option is an important determinant of the uncertainties associated with the potential performance of the option in risk reduction. Diffuse and imprecise options have high uncertainties. A distinction should be made between a composite option that comprises distinct options and a generically stated option. The generic option can be regarded as a category of options. Specific options within the category may screen in while others may screen out. Thus, the performance uncertainty is higher for the option category than for any of the options within it. As an example, Table 6-2 shows various option categories and specific options that are nested within them.

Table 6-2. Examples of option categories and specific options (option categories have greater performance uncertainties than specific options).

OPTION CATEGORIES	EXAMPLES OF SPECIFIC OPTIONS
ENGINEERING	• Site cleanup • Waste containment • Technology development
COMMUNICATION/EDUCATION	• Community outreach • Formal education • Technical guidance • Research support
REGULATION	• Control of production process • Banning of products • Facility siting controls • Facility operational controls
ENFORCEMENT	• Litigation • Fines
MARKET INCENTIVES	• Tax relief • Relaxation of controls • Subsidies • Direct purchase of friendly products
INTERNATIONAL/ INTER-GOVERNMENTAL COOPERATION	• International treaties • International standards (ISO) • Tariffs on goods • Foreign aid/loan
ENVIRONMENTAL MANAGEMENT SYSTEMS	Application of technology and techniques to mitigate risks, and direct use communities for operations and planning

If an identified option is merely described as "Engineering," high uncertainty is associated with its effectiveness as a risk reduction option. If, however, a specific engineering measure is identified, the expected performance can be stated with a higher level of confidence. A specific option such as site cleanup, can be further

desegregated into specific cleanup technologies: electrokinetics, pump-and-treat, incineration, etc. Generically stated options should be refined when possible.

Another important factor is the relaxation or tightening of constraints. As discussed in the next section, the amount of money or other resources allocated in the budget for use on a risk reduction project is a constraint. Some potentially effective options may screen out because they cannot be implemented within the budget (constraint). If identified options are to be left unmodified, one alternative is to relax the constraint by increasing the budget resources. The greater the gap in the professional hierarchy between the project implementer and the budget allocators, the more difficult it is for the risk reduction project implementer to relax a budgetary constraint (i.e., it may not be within the implementer's authority to do so).

The objective of the risk reduction project needs to be defined as explicitly as possible. The objective is a condition or an acceptable level of service at which any of the identified options is considered potentially effective. The objective should not be confused with a constraint. Examples of objectives are:

a) Reduction of particulate matter in the ambient air in Washington, DC by 30%;

b) Reduction of metals concentration in River Brown to Drinking Water Levels; or

c) Reduction of cancer risk due to radon in U.S. homes by 50%.

The operational rule for options screening is that if an option is deemed impossible to implement within limits specified on critical factors, that option is eliminated from further analysis unless it is refined or combined with other options to satisfy the limits set by the project constraints. These critical factors are herein called screening factors. Examples of screening factors are:

a) Budget,

b) Time,

c) Jurisdictional Authority, and

d) Legality.

If a minimum or maximum level allowed on each screening factor is specified, then this threshold level is considered as a constraint. For example, a regulatory agency may seek to reduce particulate matter in the ambient air in Washington, DC by 30% with a maximum budget of $13 million within the next 3 years. For this risk reduction objective, budget and time are screening factors. More specifically, for any option to screen in, it must be implementable within budget and time constraints of $15 million and 3 years, respectively.

Essentially, a "yes" or "no" answer is required, and each screening factor is taken one at a time. It may also be possible to estimate the level of confidence associated with the "yes" or "no" answer. For example, members of a city council may state: "We are 80% confident that increasing parking fees in a central business district of Washington, DC will reduce air pollution by 40% within a 3-year period, because car pooling or greater use of the metro system will occur at the new parking fee level." Thus, "increase in parking fees" would screen in as an air pollution risk reduction option at the level of confidence of 80%. The confidence level estimated is an expression of uncertainty. It is worth noting that an increase in parking fees is not necessarily the best option or set of options. Such a determination is not the focus of the options screening stage of analysis. Rather, the purpose of this effort is to determine how many options or combinations are available for implementation within the constraints established.

Risk reduction objectives, screening factors and associated constraints have a hierarchical but interwoven structure. The decision-maker on constraints depends on the scenario. The scenario could be general or site-specific with respect to the spatial coverage of the desired results of actions associated with risk mitigation objectives. An example of a general scenario is an effort to implement measures that are

intended to reduce skin cancer in the United States. The problem may not be confined to a particular location with respect to causes and/or effects. An example of a site-specific scenario is groundwater contamination beneath a gasoline storage tank. The relative level of involvement of the decision-maker as a function of the scope of the scenario is illustrated in Figure 6-2.

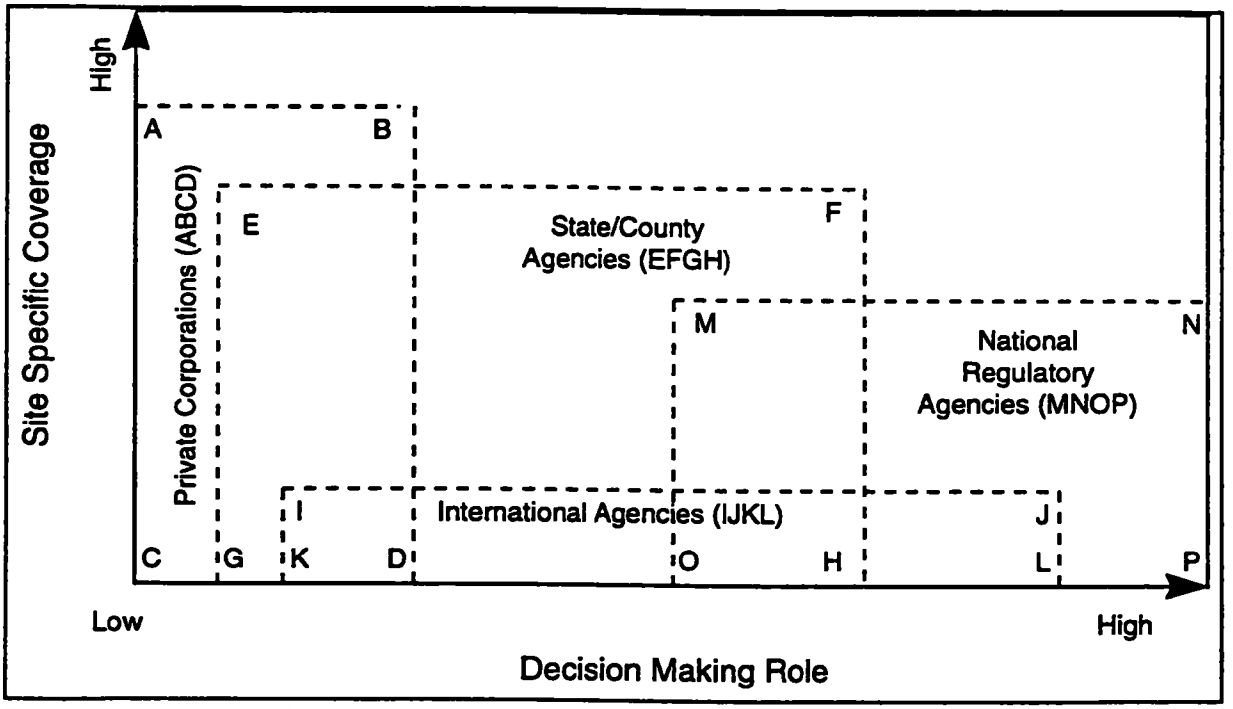

Figure 6-2. Decision-Makers on Constraints on Risk Reduction Objectives Under Various Scenarios

National regulatory agencies such as the EPA tend to be involved as decision-makers, mostly in general-coverage scenarios. In some cases, they may also be involved in risk reduction programs at a site. Private companies are involved as decision-makers almost exclusively at specific sites. Generally, the decision-maker specifies the overriding constraints. Sometimes, the decision-maker may not be the eventual implementer of some of the options that have high screen-in potential. The

roles of regulatory agencies and private companies in constraint specification is illustrated by the following examples.

a) A regulatory agency identifies constraints for its risk reduction policy development approaches.

b) Regulatory agencies usually specify constraints (generally time and performance level) associated with a requirement that the private sector implement risk reduction projects.

c) A private company specifies constraints for various screening factors that it has selected for evaluation of internally generated options for solving an environmental issue.

It is important to note that options screening involves the use of one screening (evaluation) factor at a time. Constraint levels on other screening factors are held constant. An option is considered as being fully screened after it has been evaluated with respect to each of the set of screening factors. Screened-in options are those that are likely to meet the specified risk reduction objective under the constraint levels specified on each of the screening factors. During screening an effort is not made to rank the option(s) relative to others. In contrast, subsequent analysis (broad analysis and optimization) of screened-in options involve explicit ranking. Furthermore, pooled or aggregate ratings or scores (with or without weights) are not used to express the utility of each option as in the screening stage. The sequential steps in the options screening methodology are illustrated in Figure 6-3.

USEPA Headquarters Library
1200 Pennsylvania Avenue NW
Washington DC 20460

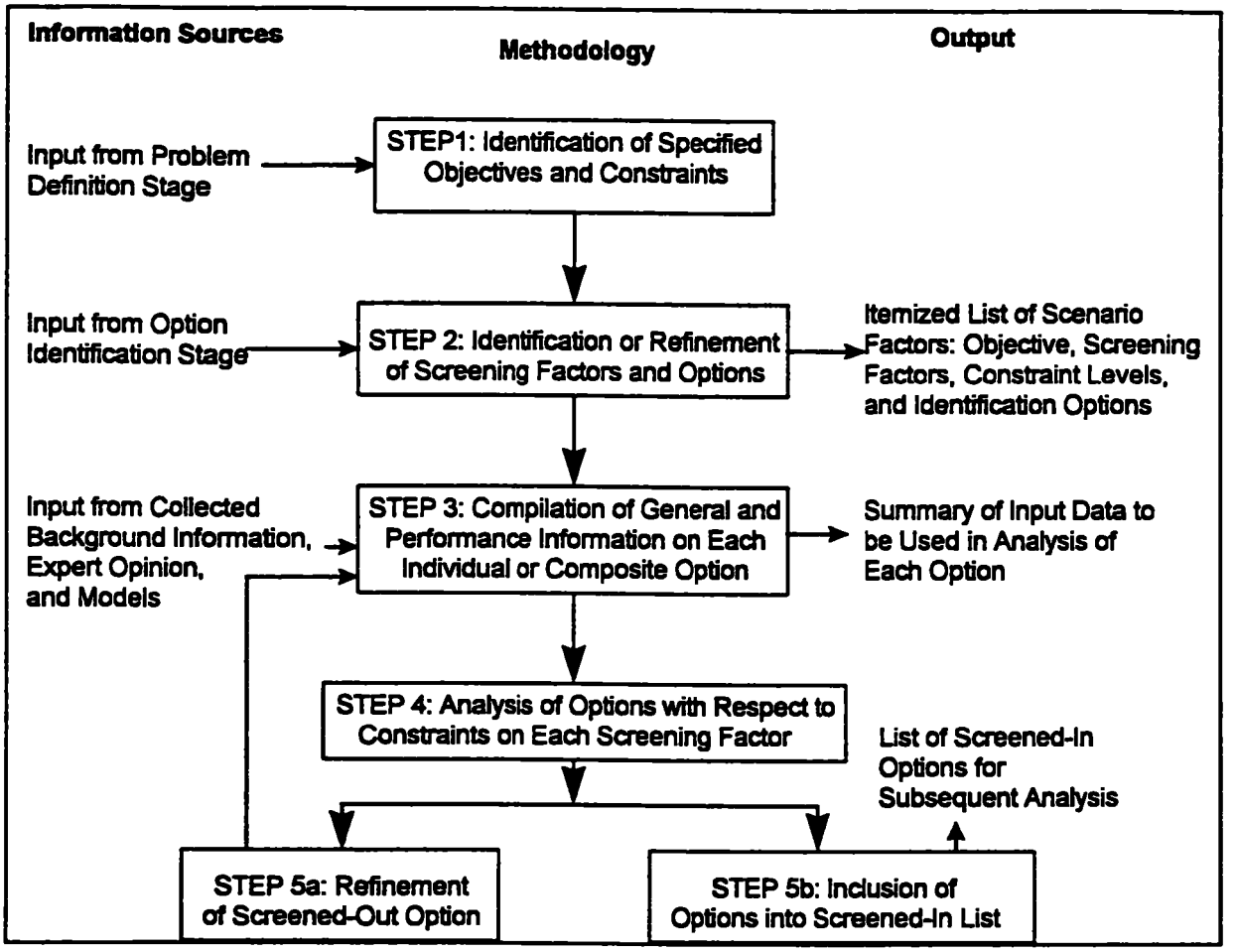

Figure 6-3. A Screening Methodology for Risk Reduction Options

6.7 Evaluate the Remaining Risk Reduction Options

The primary focus of this effort is to develop a common base of information for comparing and selecting among the various sets of options, however different they may be. For example, this might include comparing the use of consumer education programs on pesticide residues, to the imposition of an environmental charge on certain pesticide uses, to combining use restrictions with a trading program, all as

options for reducing the consumption of pesticide residues in food. A secondary, but extremely important, purpose is to continue to lay the groundwork for explaining the selection process to the wide range of interested parties.

The first concern is to be clear about the inputs to the assessment process. In addition, data will be necessary on:

a) the past or current conditions relevant to understanding and controlling the identified risks (patterns of pesticide use, likely alternatives, patterns of food consumption);

b) the performance of individual risk reduction options in other, hopefully similar, circumstances (consumer labeling, liability schemes); and

c) the characteristics of each option relevant to the evaluation criteria.

Some of this data will have been generated on the way to this point in the process.

The most problematic set of inputs, however, are the estimates of the future results of implementing any particular risk reduction option. For example, estimating the reduction in exposure levels resulting from a food labeling program is extremely difficult. No matter how much data on past experience has been collected, these estimates will reflect a significant degree of subjective judgment by the estimating parties. This judgment will reflect their past experience, as well as their approach to the available data.

As such, it is important to identify clearly the people who are providing the estimates, hence, making these judgment calls. They can include:

a) the parties selecting the priority risks to be addressed;

b) outside experts on the characteristics of those risks or individual risk reduction options (including those from other program offices in EPA or other regulatory agencies);

c) the assessors themselves; and

d) other interested or affected parties (e.g., businesses, workers, consumers, and environmental groups).

The likely accuracy of their estimates on any particular factor (along with their approach to the underlying assumptions) should be the focus when assigning tasks among these different parties.

Since application of all the assessment criteria requires that estimates of performance be made, all include a degree of subjectivity. That fact tends to be obscured, however, when an estimate is expressed in numbers, rather than in words. Many people have a tendency to accord greater weight and certainty to quantified data (whether justified or not). The task for decision-makers is to recognize and account for this effect during the evaluation process.

The effort to be consistent across evaluation criteria is further complicated by the fact that some measures more readily lend themselves to quantified expression. Examples include: expected reductions in pesticide use; expected reductions in pesticide residues; and the costs of applying alternative methods of pest control.

At the same time, all of the evaluation criteria can draw from quantitative data. Population statistics are a key part of equity considerations (in what regions are the pesticides of concern used and by whom; how do patterns of food consumption vary across different population groups). Results of prior experience with particular risk reduction options includes percentage changes in baseline conditions (such as through the lead phase-out program under the Clean Air Act) and the levels of incentives created in the past (such as the barriers to innovation which can arise from too rigid an application of technology based controls).

In fact, all of the measures and results of the assessment process can be expressed in either a quantitative or qualitative manner. For example, the result of an assessment of environmental effectiveness for any particular option can be expressed either as a number (1,2 or 3) or a word (low, medium or high). Results can also be aggregated across criteria for any particular option in either qualitative or quantitative terms. It should be remembered that, even though numbers can be applied, the accuracy of the analysis is the same as using low, medium, and high. However, the general belief is that using numbers makes the decision more accurate or quantitative. Avoid falling into the "analysis trap."

Choosing a method of expression requires consideration of a number of competing factors, most of them going to the question of how important to the decision-maker is consistency in the options' scores across particular criteria. Converting all of the estimates into the same scale of numbers, or words, helps the comparison of different options across the evaluation criteria, but may obscure important bits of data relevant to an option's score on any particular criterion (such as hiding a specific estimate of the costs to be borne by particular populations in an overall score of "high" or "3" on the equity criterion). Similarly, aggregating a particular option's scores across all criteria make for easier comparisons among risk reduction options, but obscure the relative strengths and weaknesses of any particular option (an overall score of "medium" or "2" might result from either scores of "medium" on all criteria, or a mix of only "lows" and "highs" across the criteria).

However expressed, the basis for the results of individual assessments should be explained. The goal in doing so is to illuminate the policy choices and assumptions that are behind the analysis.

The analysis of risk reduction options includes considerable amounts of uncertainty. This is true in both the data that are available and the estimates that are made. Data may be lacking because they have not been or cannot be collected (actual pesticide residues in food consumed in all different regions of the country, with any seasonal variations). Theoretically, policy makers operating under time or cost constraints (experience of other regulators with trading programs) may not have reviewed available data. Data may be sought on processes that are inherently

variable and incapable of accurate measurement or prediction (actual levels of illness resulting from chronic exposure to pesticide residues in the U.S.).

At the same time, the manner in which these uncertainties are expressed affects peoples' decision-making in unusual ways. As with quantified expression, the more certainty that people believe is associated with a piece of data or an estimate, the more weight they give to it, whether it deserves that weighting or not.

Decision-makers have several options for addressing these uncertainties. Least productive is to abandon the assessment process for particular risk reduction options just because the relevant inputs are uncertain. Another approach is to identify the data needed, along with the time and costs necessary to obtain them, before deciding how best to proceed.

The most useful approach is explicitly to reflect the uncertainty in the assessment process. Any predictive process faces differing levels of uncertainty. Policy action often needs to be taken in the face of those uncertainties. To the extent policy makers can be explicit about the level and type of uncertainty in the different parts of their decision-making process, their judgments can be understood and evaluated directly by concerned parties. This is true not only for those criteria expressed in a quantitative manner, but for each criterion and the assessment as a whole.

Having collected the information and considered the issues described above, the process of analyzing particular risk reduction options against the evaluation criteria is a relatively straightforward process of applying the decision-maker's best judgment. The estimated measurements developed for any particular option are reviewed and scores determined for each criterion. For example, an option would have a high score for cost-effectiveness if its direct implementation costs were low and a high level of risk reduction was expected to be achieved.

One question that does arise is who should conduct the assessment. While the information collection process and the explanation of the results should reach the

widest possible audiences, optimally, the assessment process itself should be conducted by the ultimate decision-maker(s). Such an approach means that less specific expertise is brought to bear on evaluating any particular criterion than would be the case if a group of specially retained experts were asked to undertake the assessment and report their results. It has the advantages, however, of providing greater consistency to the review across criteria and options, as well as making the policy judgments clearer and more readily explained.

Two major products result from completion of the assessments for each risk reduction option. One is a series of scores across the evaluation criteria for any particular group of options. As discussed above, these scores can be expressed in qualitative or quantitative terms. The level of uncertainty associated with the score for any particular criterion should be considered and reflected appropriately.

A major issue here is whether to aggregate the separate scores on different criteria into a single, overall score for each risk reduction option. The major advantage of doing so is to make comparison of the results easier across different options. For example, if one option has an aggregate score of "high" and another an aggregate score of "medium", one could arguably have a basis for choosing the first over the second without further analysis.

The major disadvantage of an aggregated score is that the total hides the relative strengths and weaknesses of particular options on particular criteria. For example, the aggregated score of "high" for one option may include a lower score on equity or flexibility than the aggregated score of "medium" for another option. Assigning different weights to different criteria as part of the aggregation process can make some adjustment. For some policy makers, this will not be sufficient and the entire matrix should be used to compare the strengths and weaknesses of different options.

The second major product of the assessment process is the identification of issues that should be fed back into earlier steps in the methodology, including: problem definition; data collection; risk reduction options identification; as well as the

choice, design and measures of the evaluation criteria. Within the time available, the use of such feed-back loops will improve considerably the ultimate outcome.

The final step in the assessment of risk reduction options is to document the process used. What were the inputs and how were they assembled? What measures were used and how were they evaluated/assessed? How were scores assigned and by whom?

With the baseline of assessment results in hand for a number of different risk reduction options, the decision-maker can then move to the next steps of optimizing the remaining option sets and selecting one or more for implementation.

6.8 Optimize the Options

Any given set of risk reduction options is but some small subset of the virtually infinite set of possible options. For this reason, we propose that any subset that has passed a preliminary screening and back-of-the-envelope evaluation should be refined further before full scale option evaluation is undertaken. Thus risk reduction option refinement begins with a set of options that have been at least cursorily evaluated on multiple criteria and screened on one or more criteria. Some of the options considered have already been redefined in this process before this stage.

The goal of this step in the analysis is to improve good alternatives, with the subsidiary objectives of improving and reducing the option set under consideration. Given that the options have been evaluated on a set of criteria, ideally refinement will improve their ratings on each of these criteria.

The simplest way to view refinement is to think of improving a risk reduction option relative to a single criterion. For example, if quality adjusted life years (QALYs) lost were the criterion and the number of QALYs lost could be reduced without changing other attributes of the option, the option that resulted in fewest QALYs lost would be preferred, ceteris paribus. When the criteria are quantifiable and it is easy to agree on appropriate measures, such refinements could, in theory, be treated as

optimization problems. One could think of the task as that of maximizing risk reduction subject to constraints imposed by other criteria.

The choice of criteria will obviously determine what changes are made in options. Less obviously, how criteria are interpreted and considered will also influence how options are refined. Some criteria will be candidates for maximization. Risk reduction, flexibility, and technological innovation are examples of these. Others, such as cost, will be candidates for minimization. Depending on how criteria are formulated, assessment of an option relative to a criterion may be qualitative or quantitative. It is assumed that most will be subject at least to some kind of ranking. When this is not the case, it is likely that criteria incorporate or reflect multiple values that might usefully be discussed separately (see Table/Figure??--list of proposed criteria, including explicit consideration of competing risks: ref, OTA report)

Treating risk reduction option refinement as an optimization problem introduces several problems, however, because many criteria that might be used are not easily quantifiable on comparable scales. For example, flexibility and technological innovation could be difficult to measure on a scale that all interested and affected parties would agree is comparable to any risk reduction measure chosen (e.g., QALYs). Measuring such attributes on subjective scales is a possibility (e.g., asking to what extent the interested and affected parties (Commission on Risk Assessment and Risk Management, 199?) agree or disagree that the option promotes technological innovation). Such an approach would introduce respondent variability that could be a basis for discussing further option refinements, but might be difficult to predict and use in any consistent approach to optimization.

Not only will evaluation of an option on any given criterion pose a challenge, but predictions of changes on multiple criteria subject to specific refinements in an option or option set are likely to be highly uncertain and difficult to quantify.

Optimization generally assumes that a single, summary, quantitative measure can be used to capture and represent the overall value or effect of the option. As discussed elsewhere, we do not recommend such summary measures, for procedural reasons among others. They involve establishing specific tradeoffs between criteria,

and, by representing the results as a summary measure, can conceal changes vis-à-vis specific criteria. Hence optimization may be a useful way of conceptualizing partial evaluations of options (e.g., relative to cost, or time), but we do not recommend it as a way of thinking about the entire refinement process.

A wide range of policy tools can be used to define an option, including various forms of incentives, education, regulation, enforcement (add to list) and mixes thereof. Because intervening at many points in a hazardous process can reduce risks, very different options may achieve similar risk reduction goals. For example, education about use of emergency medical services or alternatives may achieve as much risk reduction as education about proper use of specific protection devices (e.g., with application of pesticides). Banning a given product may achieve the same risk reduction as providing incentives for development of alternative products. In such cases, however, the options' other attributes may differ considerably (e.g., flexibility, or cost).

If those involved in refining options find at this point that they are having trouble determining whether a change improves an option, it may be worthwhile for the interested and affected parties to return to a discussion of the values at stake (Keeney, Value Focused Thinking) and consider redefining the criteria used to evaluate options. For this reason, it is likely to be important that interested and affected parties are represented in the option refinement stage as well.

Expertise and creativity are key components of structuring good solutions to difficult policy problems. Expertise is only acquired through many years of experience and training, and is likely to change the way people think about the task at hand (Anderson; Van Lehn refs). Such differences often create communication problems between experts and non-experts involved, which should be addressed explicitly. However, systematic consideration of a variety of ways of improving options, along with good communications between decision-makers with expertise in different areas (including interested and affected parties other than EPA), should foster creativity.

Options can be refined by redefinition in at least the following ways:

Aggregation - combining options,
a) Increase the scope of a proposed action (e.g. the actor, time-frame, resources allocated, kinds of tools used, or intervention points in the hazardous process)
b) Combine complementary options to create option sets - with due consideration to what exactly is to be done, as well as how much of each action

Dis-aggregation - changing or eliminating parts of option sets, and
a) Reduce the scope of a proposed action (e.g. the actor, time-frame, resources allocated, kinds of tools used, or intervention points in the hazardous process)
b) Eliminate parts of option sets

Some combination of aggregation and dis-aggregation

Each of these approaches raises further questions, such as what the optimal scope of a proposed action is, or what mix of actions and in what proportions is likely to be optimal or best on the most criteria. In any proposed change, risk tradeoffs should be reconsidered explicitly, as they are likely to change as well.

As mentioned above, how criteria are considered is also likely to influence option refinement. Such influences are not likely to be obvious, but may be serious. For example, criteria that are easier to measure may be weighted more heavily implicitly (i.e., receive more attention, be considered more thoroughly) simply because they are easier to evaluate and discuss. Similarly, if a criterion is usually considered under a given set of constraints that set of constraints may be imposed implicitly.

Other kinds of cognitive traps are discussed at length in the judgment and decision-making literature (Kahneman and Tversky refs; Plous; Arkes and Hammond). Some examples:

a) Context effects generally (e.g., framing effects) - an option described in terms of losses may be evaluated differently than the same option described in terms of gains;

b) Use of heuristics such as availability (a risk that comes more easily to mind may be considered riskier than a comparable risk that is not as "available");

c) Prominence - people may prioritize or weight an option implicitly based on the dimension that is most important to them; and

d) Uncertainty - people may weigh less ambiguous attributes of an option more heavily, in implicit tradeoffs, because of an inherent preference for avoiding ambiguity (Hsee ref).

Knowing when to stop refining an option set is as important as at least attempting to refine the option set originally considered. Several guidelines for stopping rules are proposed, both in terms of the refinement process, as well as in terms of its original goals.

Process rules could include stopping once each option has been examined at least once; considered input from participants with differing perspectives (all interested and affected parties); and/or examined historical precedents, approaches used in other countries, and approaches used for other risks (by analogy).

Achievement-oriented stopping rules will be focused on whether the resulting options comprise a manageable or workable set. This might be determined by considering resource limitations, by undertaking back-of-the-envelope value of information analyses, or by achieving some set that can easily be screened to a predetermined size.

This step in the risk reduction option ranking procedure is a quality check. While elements of the refinement process discussed above are likely to have been incorporated in earlier phases of the procedure, including this step explicitly helps

ensure that those creating the options and ranking them will not become locked into an unnecessarily restricted way of thinking about options. The goal of this step is to improve the options being considered, by explicitly considering whether they could be redefined to better meet the criteria. To keep this step manageable, specific stopping rules are recommended.

6.9 Select an Option

The decision-maker is now faced with selecting the "best" risk reduction option(s) from a set of options or group of options that have been screened and optimized to address the defined environmental problem. Thus, the goal of this step in the process is to actually select the risk reduction option for the defined problem. As with other components of the process, this selection process should be as transparent as possible to those outside the selection process. In many ways, it is even more important for this selection process to be made transparent since this activity will ultimately result in the risk reduction option that will be carried forth. The decision-maker will ultimately have to make the decision based upon the current policies of the Agency but it is a basic tenant of this effort that any decision be fully delineated and documented, including any factors taken into account and how they are weighted relative to one another. This section discusses **decision analysis methods** that are appropriate for use in the selection of a risk reduction option by being rigorous enough to provide the necessary transparency to the selection process, by being robust enough to deal with the uncertainties of complex environmental risk reduction option selection, and generally being compatible with environmental risk reduction option selection.

In previous steps, there has been a "down selection" of risk reduction options and potentially some changes or aggregation were made to the original options in order to ensure that the best mix of options or groups of options are being considered. Now the optimized, aggregated, and screened list of risk reduction options will be further evaluated in an attempt to select the best option or set of options. This selection among the various risk reduction options will be based upon a comparative evaluation of the options versus a set of criteria. Using the evaluation of the alternative options relative to these criteria as input, one or more decision analysis methods,

described below, will be used to make the final selection of the risk reduction option or set of options. As previously noted, the evaluation criteria span the range of issues appropriate for consideration when making an environmental decision from those dealing with technical performance and cost factors to those dealing with social factors. As was the case in the screening steps, the goal is to quantify the score or value for each evaluation criteria for each risk reduction option. The degree to which the scores of each evaluation criteria can be quantified and with what degree of certainty, will influence the type of decision analysis method used, as discussed below.

There are numerous decision analysis methods that can be used for assessing the alternative risk reduction options and several good literature sources that deal with the multi-attribute decision analysis. The selection of the appropriate decision analysis procedures will depend on many factors including the degree of quantitation that can be brought to each of the evaluation criteria and the need for a transparent, well delineated, selection process. The general types of decision analysis methods that could be used for this analysis range from those that are most qualitative to the those that are most quantitative:

a) Holistic,

b) Cost Benefit,

c) Matrix qualitative (High to Low) Ranking,

d) Decision Factor Analysis (numerical rating factors with weighting on the factors), and

e) Optimization (multi-attribute decision procedures).

In the end, the decision-maker will likely choose a hybrid of these processes in making the decision on the best mix of risk reduction options in order to take advantage of the special features of the different decision analysis tools for the

particular environmental problem. In the following paragraphs we discuss the basis of each of these approaches and the potential utility of the decision approach for the types of selection encountered relative environmental risk reduction options.

The holistic decision process refers to the completely (or almost completely) qualitative procedure of making the decision among the various risk reduction options based upon intuition, professional judgment and certain known facts about the various evaluation criteria. This procedure can be used for consideration of the other criteria when one dominant criteria such as environmental performance has been quantified more rigorously. This approach suffers from the fact that it does not adequately allow delineation of the decision process due to the use of personal judgment.

Cost benefit analysis is a decision analysis procedure that can be used to evaluate the relative costs and benefits of the risk reduction options. This procedure, for example, would allow a comparison of the costs for each pound of pollutant removed by the various risk reduction options. If each of the evaluation criteria could be cast as a monetary cost or benefit to society then a comprehensive quantitative evaluation of costs and benefits could be made. For complex environmental problems with numerous risk reduction options and a large number of criteria, this approach is limited by the large informational requirement.

Matrix comparisons involve the development of a matrix of the assessment of each of the evaluation criteria for each risk reduction option. A quantitative (numerical value) or a qualitative (high, medium, low) ranking is developed for each criteria for each risk reduction option. The decision-maker then must choose the risk reduction option to implement that provides the best balance in simultaneously meeting all of the criteria. It is of paramount importance that the decision-maker delineate the tradeoffs that were made in the final decision and provide the rationale for the final selection given the different criteria considered to be important. Some potential decision rules that are useful in the final selection are provided in the following section.

Decision factor analysis is similar to the matrix approach but differs in that each of the evaluation criteria is weighted and scored with a numerical value during the evaluation of each risk reduction option. In this approach, each option is evaluated

either qualitatively or quantitatively relative to each evaluation criteria and given a quantitative numerical score. The weight placed on each criterion reflects the relative importance of this criteria to the decision-maker for this particular environmental problem. The selected risk reduction option is thus the option with the highest cumulative score given the numerical score and weighting for each evaluation criteria. The advantage of this approach is that it clearly delineates the factors used in the evaluation and the weighting applied to the criteria. The disadvantage of this approach is the need to develop quantitative scores that discriminate between the options that may be difficult to quantify for complex environmental problems and complex criteria such as social equity. Furthermore, not all preferences are consistent with this approach (e.g. lexicographic preferences).

Optimization refers to the decision-making process that attempts to quantify the decision into a formulated relationship between an objective function that is to be maximized (or minimized) relative to a series of constraints (or criteria). The objective function can be a function of cost, benefits, and detriments and can account for tradeoffs among these by incorporating relative weighting factors and functional relationships. There are a number of methods for determining the optimum set of decision variables and commercial software is available to carry out the procedures. While this procedure is the most quantitative approach it is often not suitable for complex environmental problems due to the difficulty in establishing functional relationships between decision variables especially with qualitative factors and due to the nonlinear relationships that often exists among the various factors.

We conducted a number of example evaluations involving the actual selection of risk reduction options for a series of hypothetical environmental problems. The environmental problems were oriented around different contextual viewpoints, specifically environmental stressor (e.g., VOC emissions as related to urban ozone), the media (water, air, solid waste) context and location (place based) context. In these examples, the selection criteria included both factors that were quantifiable and those that were unlikely to be quantifiable for the set of different risk reduction options. The qualitative nature of some of the inputs to the decision, tends to move the selection to be of a subjective nature. However, a purely holistic approach does not provide the appropriate degree of delineation and therefore transparency.

In order to accomplish this selection for an environmental problem with a wide array of potential solutions, it may be necessary, due to time and resource constraints, to further narrow the field with the qualitative matrix approach in order to generate a manageable set of highly ranked options and then conduct a more rigorous analysis of the criteria for each of the highly ranked options. The combination of techniques represents a hybrid of the selection processes discussed in the previous sections but may improve the overall selection process efficiency and ultimate result.

The final selection of the optimal risk reduction option or set of options can be made by applying different decision rules. The use of decision rules is particularly important when the matrix approach is used for the selection process due to the complexity of the particular environmental problem under analysis. For example in the special case called "dominance" where one single risk reduction option is highest ranked relative to every criteria it would clearly be selected. Other possible decision rules include using a lexicographic rule (just looking at a single criterion), selecting the option(s) with the highest ranking on a specific criterion, or excluding options with the lowest ranking on any one criterion. The policy decision maker could also select the option(s) whose lowest ranking on any criterion is higher than the lowest rankings for other options, or use some model to combine rankings on individual criteria into one or more scores (as discussed above). Again, the exact application of the decision rule(s) can not be a priori specified. The policy decision maker must, however, document any decision rule employed and provide the rationale for that decision rule in the context of the particular environmental risk reduction options considered. It must be possible to follow the thought process through the decision-making to the final selection of the risk reduction option or set of options.

One of the greatest challenges in selecting a risk reduction option, or set of options, in addressing a complex environmental problem is dealing with the uncertainties in the performance of the options relative to the criteria. In order for the selection process to be transparent to those on the outside of the process, the process must document the uncertainties in the evaluation of each criteria and deal with the uncertainties relative to the final selection in an open manner. As discussed above, the presence of uncertainties for more complex problems and options will tend

to push the decision analysis process to more qualitative methods such as matrix approaches with low, medium, and high rankings as opposed to decision factor or optimization approaches with quantitative measurement. In any case, it is important to deal with the uncertainties within the selection process framework.

There are several potential methods of dealing with the impacts of uncertainties in the scores of the evaluation criteria. One method is to have a separate criteria for each risk reduction option that scores the overall uncertainty of the performance of the option relative to all of the other evaluation criteria. A risk reduction option with the same overall score relative to the other criteria but a high degree of uncertainty would be less favorable relative to those with higher degrees of uncertainty but equivalent scores on other criteria. It is important not to reject a better option solely on the basis of the certainty of the evaluation criteria. Nonetheless, uncertainty could be used as one more criterion in the evaluation of the overall acceptability of an option versus other options.

A second approach is to attempt to quantify the uncertainty range relative to each criteria. Given this range of criteria scores, a sensitivity analysis of uncertainties for the highest ranked risk reduction options can be undertaken to determine if the rank order of the risk reduction options changes over the range of uncertainty assumptions. This is essentially a form of sensitivity analysis on the options. The most comprehensive method is to conduct the process in an iterative manner by collecting new data for those parameters that are so sensitive as to change the selection of risk reduction options. This last method can also have a significant impact on time and costs due to the resources required to collect the additional data.

Ideally the selection process will lead to a single option or single set of options that are considered to be "best" by the decision-maker and the decision process, including criteria and criteria weighting, will be clearly delineated for those outside the process. Under certain situations there may no clear winner, but rather a set of options may be very close with no clear discriminators among the evaluation criteria. In this case, it may be necessary to analyze other criteria or further quantify special criteria to help delineate the better case.

6.10 Document the Process

The final decision should be transparent, i.e., the motivation and criteria used in reaching the decision should be obvious. Therefore, it is important to document not just the final decision, but also the process leading to the decision.

Both the process and the documentation need to contain sufficient detail to substantiate decisions, but should not be so detailed as to be unusable. Therefore, we recommend simple documentation, with tables emphasizing the breadth of options considered, and the values and judgments used in evaluating those options, and ultimately in selecting the best option. The process of making judgments is inherently complicated, especially in a public forum, because of the need to elicit different values and input from a broad spectrum of people. Thus, the need to document, and be consciously aware of biases and decisions when making evaluations.

6.11 Quantify Option Effectiveness

Measurement is a critical component in risk reduction evaluation. The bottom line in implementation of risk reduction options is to determine whether or not the goal is achieved. Once implemented, it is critical that the effectiveness of the risk reduction option or set of options be quantified. If we cannot measure the success of our performance, than we probably cannot determine whether or not we are achieving our goal. It is critical that both what we measure and how we measure are built in before we begin implementing selected risk reduction options.

The following are the four basic approaches to measurement:

Direct - Measures that directly quantify the current status against a set goal or goals for improvement (also referred to as **outcome measures**).

Indirect - Measures a component that has a direct impact on the specific area for improvement, but does not specifically provide quantitative measurement of the specific improvement goal (including **administrative process** or **stressor measures**).

Leading Indicator - Measurement of a specific parameter that provides an indication of the probable success to achieve a goal.

Lagging Indicator - Measures after the fact a specific parameter that may directly or indirectly measure the performance against the improvement goal or goals.

To better understand these four approaches, the discussion below describes measurement approaches using two examples. The first example discusses a typical set of risk reduction options to achieve a reduction in cases of childhood asthma (the goal). The second example describes a set of risk reduction options to improve the biodiversity in a receiving stream. We selected these examples to describe measurement approaches in both public health and ecosystem environmental problems.

For this example, the goal of a specific set of risk reduction options is to reduce the number of childhood asthma cases. Therefore, measurement of the number of childhood asthma cases on an annual basis is a direct measurement of the goal. This is also a lagging indicator because the cases have already occurred and, therefore, the measurement system using this indicator tracks the result from the output of the system as it currently exists.

For purposes of this example, we will assume that there is a correlation between air quality and childhood asthma cases. Therefore, air quality can be monitored within the study area that provides an indirect measurement of the goal. The assumption is that improvements in air quality will result in reductions in the number of childhood asthma cases. Therefore, measurements of air quality indirectly predict improvements in the number of childhood asthma cases and also represent a leading indicator. Expanding the community's rideshare program is one of the mechanisms that could be implemented to assist in improving air quality. Therefore, the measurement of the rideshare program (decreases in single vehicle travel miles) represents an indirect indictor on air quality and is also a leading indicator for air quality.

As can be seen from this example, a number of parameters can be quantitatively measured to help define the system and predict whether the risk reduction option(s) will achieve the overall goal. It is important to understand the interrelationship of what is being measured and how it relates to the specific improvement goal.

A typical goal for an aquatic ecosystem is to improve both the quantity and diversity of the biota in a stream, river, or lake. Measurement of biota diversity is a direct measurement and a lagging indicator. A number of techniques have been described in the literature to perform biodiversity measurements. Clearly, the water quality has direct impact on the biota diversity. Therefore, measuring dissolved oxygen, temperature, salinity, and other water quality parameters are critical in defining the water environment in which the biota live. These measurements of water quality parameters are indirect measurements of biota diversity and are leading indicators. Improvements in the water quality parameters typically lead to improvements in biota. A secondary indirect measurement can be reduction in combined sewer overflow to the receiving stream. Typically, combined sewer overflow adds pollutants to the receiving stream and negatively impacts water quality, and subsequently negatively impacts biodiversity. Projects can be implemented to reduce combined sewer overflow and the reductions in the quantity of sewer overflow as a function of time can be measured, which represent a leading indicator of water quality.

Reduction in the quantity of combined sewer overflow is just one example of a risk reduction option that could be implemented to improve biodiversity. Clearly, water quality is one of the key controlling conditions within the ecosystem impacting biodiversity. Therefore, other risk reduction options that improve water quality also need to be monitored. Those risk reduction options that have the greatest impact on controlling the key water quality parameters must be monitored to determine their impact on the ability to achieve the overall goal.

In addition to water quality, the physical conditions (including water velocity, low flow water volume, and bottom substrate) within the aquatic ecosystem also play a major role in determining biodiversity. The purpose of this discussion is to alert the

reader to the fact that ecosystems are complex and that there are a number of parameters and conditions that impact biodiversity. It is, therefore, critical to determine the key controlling parameters that impact the specific goal to be achieved and to quantitatively monitor the risk reduction options which are implemented to improve those conditions in the eco-system parameters which lead to achieving the goal.

In order to evaluate the effectiveness of the risk reduction options implemented, it is necessary to establish a monitoring system/program that provides sufficient data to predict performance and effectiveness. Measurement program plans should describe:

a) Location and types of samples to be collected;

b) Frequency of sampling;

c) Analytical methods;

d) Quality control; and

e) Management review.

The location, type, and frequency of sampling is critical to ensure that sufficient data are collected to predict the performance of a specific risk reduction option or of the system as a whole. In the previous example of the childhood asthma cases, it may be beneficial to collect the number of cases on a monthly basis to see if there is any variation in frequency with time of year. Since there is frequently a direct correlation with air quality and there are seasonal variations in air quality, one would expect seasonal variations in the number of childhood asthma cases. If the number of cases is only collected on an annual basis, the evidence of seasonal variability would not be available. Collecting the data on a daily basis would be too frequent and provide little added benefit over collection on a monthly basis. In the case of the aquatic ecosystem, it is critical to perform the biodiversity analysis at the same time

each year to eliminate seasonal variations. For example, in a river ecosystem, the biodiversity is very different in the summer from the winter. For biodiversity, it is important to compare data from the same time of year. However, water quality data should be collected year-round to understand whether there are seasonal variation which may have an impact upon the biota. For example, spring runoff events may transport pollutants into the receiving body of water that may subsequently impact the biodiversity.

Data collection is extremely resource-intensive. It is, therefore, critical to select the number of locations, appropriate monitoring parameters, and sampling frequency to optimize the cost impact to the project. Sampling needs to be frequent enough to be representative of the study program. Where appropriate, the frequency should be sufficient to perform proper statistical analysis.

Most samples that are collected of physical systems, such as water quality samples from a receiving stream, will require laboratory analysis. The analytical methods employed should be consistent with the nature of the samples and with the most recently approved method of analysis. For example, the analytical method for preparing stream water samples for measuring metals is different from the analytical procedures used to prepare samples for analysis of stream sediment materials. There are a number of existing references that specify current and approved procedures, including Standard Methods for the Examination of Water and Wastewater and procedures published by American Standards for Testing and Materials (ASTM).

In all cases, sampling programs require proper quality control. This includes the collection of field blanks, trip blanks and laboratory blanks and spikes to ensure that there has been no unknown contaminant added as a result of the sample collection methodology, transportation or the analytical procedure. Depending on the nature of the samples and the type of analysis being performed, there are general recommended procedures on quality control. It is recommended that before a sampling program be implemented, a formal quality control program be written and approved.

Management review of the data is critical to ensure that sampling and analytical methodologies are properly followed, quality control procedures fully implemented, and performance tracked. As required, the sampling program should be modified to ensure efficient and cost effective sampling and analysis are performed. Frequently, sampling programs are modified over time to focus on specific areas of interest and reduce the sampling frequency and analysis for parameters that show little variation.

To improve the overall implementation of risk reduction options, it is critical that the performance be communicated to the public. The monitoring data should be assembled in a simple to understand format and presented periodically at public hearings/meetings. This is one of the best methods to ensure acceptance of the program being implemented and to receive feedback from interested parties on the performance of the program.

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PART V IMPLEMENTATION AND PERFORMANCE EVALUATION

PREFACE

CHAPTER 7. PERFORMANCE EVALUATION: THE USE OF ENVIRONMENTAL DECISION REPORT CARDS

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CHAPTER 7. PERFORMANCE EVALUATION: THE DESIGN AND USE OF ENVIRONMENTAL REPORT CARDS

7.1 Introduction

The lack of a widely accepted and commonly used method for evaluating the effectiveness of governmental (including environmental) programs represents one of the great challenges facing agencies such as the EPA as the millennium draws to a close. With the passage of the Government Performance and Results Act (GPRA) in 1993, the Congress has mandated that an agency must do much more to answer the bottom-line question "How are we doing?" if it expects continued, let alone expanded, support. In the context of EPA, the answer to this question is even more important now than in the past because of the growing number of complex environmental issues, the evolution of institutional roles from narrowly focused mandates to broader public demand for high quality protection of human health and ecological integrity, and the intensified scrutiny of the cost of environmental programs.

In the same way that GPRA demands accountability for performance at a government agency-wide level, the IED framework seeks accountability for performance at the level of individual integrated environmental decisions. In the former case, the question is "What is the state of the environment, and what is the impact of Agency actions on that state and the trends in the environment?" In the latter case, the question is "What is the state of that portion of the environment that a specific management decision was designed to affect, and what information is available that might suggest adaptive management options that could obtain more desirable results more efficiently?"

The term "environmental report card" has been used to refer to the evaluations that are applicable to both the more global GPRA-like and the more specific IED-like situations. For example, recently several groups have recommended that a "report card" be developed for national environmental programs (e.g., Enterprise for the Environment, 1997; Vice President Al Gore, in his direction to the National Science and Technology Council, year??). In many respects the Annual Reports from the Council on Environmental Quality served as prototypes for a national environmental report card.

In addition, several regional initiatives have generated report cards, including reports of the U.S. Forest Service and Bureau of Land Management in conjunction with the Northwest Forest Plan, the Chesapeake Bay Program's annual State-of-the-Bay publication, the State of the Great Lakes report (Environment Canada and EPA, 1997), the State of Boston Harbor report, and others (Young, Harwell to provide cites). Each of these reports provides a periodic update of governmental agency activities and, to a limited extent, related environmental improvements.

In contrast to these environmental report cards that assess the environment more broadly, the IED uses the term "environmental decision report card (EDRC)" to refer to an evaluation of specific decisions.

As depicted in the framework diagram (Figure 1-3), the EDRC is the formal mechanism for feedback and course correction. Such feedback on the extent and distribution of environmental effects is helpful in determining whether the relative seriousness of risks was accurately characterized in the first place and whether specific risks have changed as a result of an implemented risk reduction program. Report cards should also provide the basis for evaluating the performance of specific risk reduction programs or decisions, as judged against decision criteria such as efficacy at reducing aggregate risk, cost, equity, and time required to achieve risk reduction goals. Therefore, environmental report cards should provide the information needed to a) identify opportunities for course correction and adaptive management, i.e., modification of risk reduction approaches in light of performance information or new information on risks; and b) assign specific accountability--to individuals, programs, or organizations--for environmental results. A reporting system that focuses on environmental outcomes associated with individual decisions will provide greater focus and discipline by explicitly expressing the relationship between an environmental investment (e.g., money, information, skills, and time) and the results achieved.

The enhanced accountability provided by a well-designed report card mechanism is particularly important in the context of the greater flexibility espoused in the IED framework for both government agencies and private interests to approach problems with innovative solutions. As the system of integrated environmental decision-making evolves, with its focus on reducing aggregate risk through flexible approaches to risk reduction, reporting on the rate of achievement of particular single-

stressor standards (or other individual characteristics affecting environmental health) will fail to tell the whole story. In its stead, there needs to be a mechanism to gauge the collective impact of sets of risk management approaches on actual outcomes.

A well-designed and appropriately implemented outcomes reporting system also can engender long-term public support for environmental programs. By providing periodic objective reports that link decisions to outcomes on the health of ecosystems, human health, and quality of life — in other words, information about the things people really care about — rather than specific chemical risks whose impacts may not be obviously relevant to most people (e.g., mercury or 1,3-butadiene), the reporting system has the capacity to generate a strong sense of ownership and investment on the part of the public. (For a more detailed discussion of the benefits from performance-based reporting systems, see Enterprise for the Environment, 1997.)

While interest in performance evaluation is growing, EDRCs and the broader environmental report cards will be only as useful as the information contained in them. In the following sections, we discuss various types of performance measures and some issues associated with their use in environmental reporting systems.

7.2 Types of Performance Measures

With the growing interest in performance evaluation, a number of closely related terms have emerged to refer to different performance measures; i.e., those measures that are used in constructing the report card. For the sake of clarity, the Committee defines the following types of performance measures:

a) **Process Measures**, i.e., measures of administrative effort or program actions that are presumed to result in environmental or health improvements (e.g., numbers of permits issued, number of enforcement cases pursued, number of contaminated sites cleaned up to standards).

b) **Stressor Measures (levels):** measures (levels) of stressors in the environment that are used to determine attainment or non-attainment of desired reductions in stressor levels (e.g., total emissions of a pollutant, levels of dissolved oxygen or turbidity levels in a stream, and density of roads in a

watershed).

c) **Exposure Measures:** measures of the co-occurrence or contact between an individual or population and environmental stressor(s) over a defined time period. The term exposure is traditionally associated with chemical stressors (e.g., stressor levels in food, concentrations of stressors in tissues, time-activity measures, and total exposure to a contaminant via all routes), whereas the term co-occurrence is often used as a broader term applicable to chemical, physical, and biological stressors. Exposure measures are more directly related to effects than ambient levels of stressors in environmental media because they address direct contact with the stressor. These measures can also be more readily linked to risk management decisions than effects measures since causes of adverse effects are often multi-factorial.

d) **Effects Measures:** measures of human and/or ecological effects, the changes in which can be used to assess the impact of an environmental risk reduction program (e.g., asthma rates, cancer rates, acres of wetlands gained or lost, local extinctions of important species). Another application of effects measures is for **condition assessment**, in which a suite of effects measures are evaluated and reported in combination to characterize the health or condition of an entire population or ecosystem. Condition assessment provides a baseline against which to evaluate the success of specific environmental decisions or multiple decisions impacting a population or geographic region. In addition, reporting on condition provides feedback on whether the problem formulation has correctly identified the set of stressors associated with the environmental or health consequences of concern.

The Committee considers measures of effects or condition to be **environmental outcome measures.** We note that this definition differs from the Agency's definition of "environmental outcomes," which also includes measures of stressor levels (EPA Strategic Plan, 1997). We recommend, however, that stressor measures be kept distinct from environmental outcome measures because a) changes (increases or decreases) in stressor levels do not necessarily translate into changes (increases or decreases) in risk, and b) the public's environmental goals are typically in terms of desired states of health or condition, rather than desired stressor levels.

The different types of measures can be thought of in terms of a spectrum of measures ranging from least directly to most directly related to actual environmental outcomes, which are of primary interest to the public. In many cases, the spectrum of measures will also correspond to a spectrum of response times, ranging from shorter term to longer term measures. For example, it is relatively quick and easy to document the number of permits issued in a year, but it is more difficult and time-consuming to determine the actual impact of that process measure on the condition of the environment. Figure 7-1 illustrates this spectrum and relates the Committee's terminology to other commonly used terms for environmental performance measures.

Figure 7-1. Spectrum of Performance Measures

			Environmental Outcomes
Process Measures	**Stressor Measures**	**Exposure Measures**	**Effects/Condition Measures**
activity measures[1] *output measures*[2] *response indicators*[3] *"beans"*	*pressure indicators*[3] *release measures* *emission measures*	*co-occurrence*[4]	*adverse effects*[5] *health outcomes*[6] *state indicators*[3] *ecological indicators*[7]

Least directly related to Environmental Outcomes — Most directly related to Environmental Outcomes

[1]EPA Strategic Plan; [2]GPRA; [3]OECD Framework; [4]Chapter 2; [5]Ecorisk Guidelines; [6]Understanding Risk; [7]EMAP

Examples of the various types of performance measures and their use in concert to evaluate the success of an IED project is illustrated in Figure 7-2 using a watershed restoration example.

Figure 7-2. Evaluating the Success of an Integrated Environmental Decision: A Watershed Example

To illustrate the use of each of the types of performance measures to report on a specific Integrated Environmental Decision, consider a watershed management or restoration program. In this example, the ultimate **goal** of the program is to maintain or re-establish an ecosystem that supports a diversity of habitat types along with their resident communities of plants and animals, supports essential ecological functions, and is self-sustaining. Here, the Integrated Environmental Decision will consist of a set of interrelated actions, many of which will be designed to address multiple stressors in order to achieve a reduction in the aggregate risk from those stressors. Other IED program actions will focus on restoring damage from past stressors (such as the restoration of riparian zones damaged by livestock operations in order to decrease sedimentation downstream, provide shade and cooler temperatures for aquatic species, and provide additional nutrients to the system from dropping leaves).

The evaluation criteria used to judge overall IED program results will include measures of habitat quality (such as length of intact corridors of natural riparian vegetation), water quality and temperature, hydrology that mimics natural variations, the extent of connectivity between floodplains and the river, nutrient balance, presence of sustainable native populations, and the like. Taken together, these **effects measures** effectively describe the **condition** of the watershed, i.e., whether the watershed, in fact, can sustainably support native populations and their habitats and maintain ecological functions -- and they therefore report on the aggregate results of the IED program.

Each of these effects measures can also be used as to evaluate the success of individual actions within the IED program. Following the example above, measures of length of intact riparian corridor, water temperature, and nutrient balance would be used to assess the success of specific actions taken to restore riparian zones.

The time frame required to see changes in the effects measures will vary. Some, such as population levels of short-lived species of interest, may be detected after only one year of a management regime that alters pollutant inputs and water releases to the system. Other environmental responses may take more than ten years to be detectable.

In addition to the effects measures and condition assessment, direct reporting on the decreased pressure from various stressors will be useful. In the example above, such **stressor measures** might include the decrease in the rate of new riparian damage and increases in the release of cool water from dams in the summer. Stressor measures relating to other actions that are also part of the IED program might include reductions in ambient pollution levels, decreases in the number of unscreened pumps that injure fish, and the restoration of periodic flood flows.

Finally, **process measures** will provide insight into the level of effort expended and provide a shorter-term indicator that the program is proceeding as planned. Examples of process measures might include conservation easement contracts signed for the management of riparian corridors, changes in the regulations governing water releases from dams, and numbers of water pollution permits updated with new effluent limits.

Although the focus of this chapter is on evaluating decisions on the basis of environmental outcomes, integrated environmental decision-making may seek to achieve other goals such as environmental justice, quality of life, inter-generational equity, or public "right to know." In such cases, it will be important to define performance measures to evaluate these other categories of outcomes, referred to in a recent NRC report (1996) as social, economic, and ethical outcomes.

7.3 Improved Report Cards

7.3.1 Employing a Broader Range of Measures

In order to provide a meaningful assessment of environmental program performance, it is critical that EDRCs provide accurate information not only about process measures (such as how many industrial units now have permits and are meeting air pollution requirements, or how much money has been spent controlling water pollution), but also on environmental and health outcomes. Moreover, the specific measures selected should include those that are related to desired states of human health or ecological condition (such as the proportion of at-risk children that are free from pollutant-related respiratory diseases, or the maintenance of native biodiversity in a stream).

In cases where outcome measures are not available, or not measurable in the short term, the selection of the most appropriate measures of impact of an environmental decision should be influenced by the strength of the association between the measure and the desired environmental outcome. If, for example, exposure to a chemical is identified as the causative agent for a particular disease, and exposure is closely correlated with ambient concentrations of a stressor, then a stressor measure may be an adequate predictor of resultant health effects. In other cases (e.g., persistent bioaccumulative toxics), environmental concentrations in a particular medium may not be related clearly to human dose or exposure. In such cases, direct measures of exposure (e.g., fish tissue contaminant levels) or of the incidence rate of adverse effects in populations at risk (e.g., subsistence fishers or fish-eating wildlife) may be needed to evaluate progress toward the desired environmental outcome.

The SAB urges the Agency to place increased emphasis on outcome measures, developing new ones where required, because the ultimate goal of integrated environmental decision-making is the actual reduction of risks and adverse effects. We recognize that environmental outcomes, whether measures of health, ecological condition, or quality of life, may not be observable over short time frames, so that process, stressor, and exposure measures will continue to be important in environmental decision-making and management. Outcome measures, however, are an important means of verifying the theoretical basis for the control or abatement

options chosen. In other words, if the postulated relationship between stressors and effects is inaccurate, then stressor or exposure reductions might not produce the expected improvements in adverse effects or condition. A robust reporting system will include a mix of process, stressor, exposure, and outcome measures.

Even the Agency's traditional list of process measures will need to be expanded in order to evaluate the performance of some of the new approaches to environmental management. In recent years, the Agency has added more tools to the environmental toolbox, bringing to bear such new approaches as economic incentives, negotiated agreements, and the like (see Chapter 6). Therefore, the collection of data to evaluate these new approaches should evolve as well. There are several important inputs currently missing from most reporting systems that would provide valuable information for EDRCs that assess how well environmental programs have worked and what changes or adjustment might be made. In the case of a marketable permit system, for example, systematic reporting about the number of transactions, the average price per unit of the traded commodity, the number of market participants, the net reduction in pollution output, and average marginal cost per unit of pollution reduced should be compared with initial predictions. To the extent that a risk management program involves more than one tool, the entire panoply of actions should be reviewed for efficacy, cost, and other measures of performance. Separate information should also be provided on the "non-quantitative" inputs to decision-making, including the effects of the risk reduction program on sustainability and equity.

While all of these parameters are often reported somewhere, seldom are they reported in the same place, in terms understandable to the public, and in a format that provides useful input into the next iteration of decision-making on a particular issue. This feedback loop is essential to the functioning of the proposed IED framework.

In summary, over the years the Agency has developed a traditional set of systems to track administrative process measures and stressor measures. However, in addition to these measures, the IED framework emphasizes the need to strengthen reporting by a) expanding the range of measures used to evaluate the performance of new (non-traditional) environmental management approaches, and b) shifting from the traditional focus on process and stressor measures to a more balanced suite of measures, including measures of exposure and environmental outcomes.

7.3.2 Reporting on Environmental Outcomes

7.3.2.1 Environmental Health

Setting up an outcomes-based reporting system requires an understanding of the chain of events linking human activities to stressor levels to effects of concern (whether health, ecological, or quality of life); hence the need for process, stressor, exposure and especially effects measures in an EDRC. The expected response time for various measures along this chain of events will vary. Thus, the time frame for monitoring and reporting on suites of measures needs to be established based on the time expected for changes to occur.

In looking at current reporting systems, it is clear that the nation has much more fully developed systems for gathering and analyzing information on outcome measures for human health than it does for ecological condition. The Department of Health and Human Services (HHS) – the second largest Department in the federal government -- oversees an extensive effort to fund and coordinate information collection and analysis at local, state, and national levels, which provides input on measures of effects and condition for human health. The Department has introduced a number of useful approaches in this regard. For example, the HHS effort to establish national human health goals—*Healthy People 2000*—and to track progress towards these goals (HHS, 1990; 1995) using outcome measures has served as a model for the Agency's own effort to articulate environmental goals (EPA, 1996). In addition to HHS, there are many other groups -- at various scales -- that also periodically release studies of health effects/trends in different geographic areas; e.g, World Resources Institute, World Watch, and the World Health Organization.

Reporting on Quality of Life

Human welfare and quality of life (QOL), defined in terms of non-health related impacts from environmental exposures or ecological change, is inextricably linked with both ecological condition and human health issues. Progress toward achieving QOL goals can and should be measured in an integrated performance evaluation system. In some cases, QOL measures will relate to exposures to chemical agents (e.g., nuisance odors), and thus might be included in an overall report on health. Other indicators of QOL will derive from the consequences of a change in ecological condition (e.g., changes in biological diversity, availability of recreationally important species), and might be reported in concert with measures or indicators of ecological effects. Yet other QOL measures may relate to economic consequences of environmental exposures or ecological changes (e.g., changes in property values, job impacts of changes in the availability of harvestable natural resources).

However, current systems often provide fragmented health information, such as data on only the most serious outcomes; e.g., deaths from chronic obstructive pulmonary disease, rather than total incidence of respiratory problems. Many important health outcomes, such as reproductive health or neurologic health of children, are not well captured by these systems. In many cases, environmental data are collected without consideration of how they might be used to understand the role of the environment in reported health outcomes or how the data might be used to attribute risk from exposures to environmental stressors. As a result, these data are of limited utility in assessing the relative contribution of environmentally mediated factors relative to other major drivers, including nutrition, education/lifestyle, economic status, and genetics.

We recommend, therefore, that the Agency work with HHS and other governmental and non-governmental organizations to develop an assessment that examines the impact of exposures to environmental stressors on human health, with a particular -- but not exclusive -- focus on those stressors that fall within the Agency's sphere of influence. Emphasis should be placed on measuring total or aggregate exposures to various stressors, i.e., by all routes of exposure — inhalation, ingestion, and dermal contact. Such an "environmental health" report card could serve as a component of a larger "human health report card" released by some of these larger organizations which could eventually attribute health outcomes to specific environmental risks or combinations of such risks. Any system that is developed should be systematically scanned for changes that may indicate problems. Reviews at present are episodic and so may not recognize early changes in health indicators.

Efforts to report through EDRCs on the effectiveness of environmental protection programs in reducing human health risks would benefit from the development of a conceptual framework describing the relationship between goals, interventions, and desired outcomes. Evaluating the effectiveness of an environmental intervention on human systems will require consideration of the same factors that are used to determine risk; namely the measure of a health effect (outcome), the measure of exposure to identify potential stressors of concern, and the establishment of human dose. The health outcome would be the endpoint of choice for evaluation. However, when an effect takes many years to occur (e.g., cancer), measures of early steps along the path to the outcome or ~~even~~ exposure measures should be used for evaluating the

effectiveness of interventions.

At the same time, we need to be aware that implementation of an EDRC reporting system for human health risk reduction will be hampered, in many cases, by a lack of data in key areas; e.g., exposures, differential susceptibilities among subpopulations, and incidence of non-fatal diseases. It is important to recognize that there are critical gaps in currently available data (discussed further in section 7.4). At the same time, these gaps should not limit efforts to collect the data that do exist to evaluate specific programs today or constrain the consideration of desirable performance measures for which data might be collected in the future.

Even where data on environmental health are available, the challenge of relating information on exposures and effects remains. Current data often are not sufficient to link exposures in populations to health outcomes. Occupational/workplace exposures are more easily dealt with in this regard than residential exposures. Geographic mapping systems are a potential tool for linking residences to exposures. However, the usefulness of mapping approaches is still dependent on the precision possible for assessing exposures by exact location. Obtaining such precision is frequently a difficult task in rural and suburban areas. However, efforts are underway to improve the recording of location for both exposures and, in some cases, outcome data, which will improve the ability to match and model exposure and outcome data in the future.

To further complicate matters, the susceptibility of populations is known to vary, and science is moving rapidly to determine what constitutes these susceptibility characteristics. At present, we can judge susceptibility by general characteristics such as age, race, and sex, but even these have not been used extensively to judge the differences in risks. An important challenge lying ahead is to develop methods and reporting systems that use traditional exposure information, population characteristics, and rapidly expanding data on biomarkers of susceptibility and exposure to expand our ability to be more specific in identifying subsets of the general population that are at risk of disease.

7.3.2.2 Ecological Health

The focus of the IED framework is on improving the environment by making individual -- yet integrated -- environmental decisions and tracking their impact on human and ecological health through appropriate measures consolidated and reported in an EDRC. In this regard, discussions of EDRCs for ecological health and human health are quite similar. However, in the case of ecological health, the particular outcomes of concern are less obvious, certainly to the public. Whereas a human health outcome of reduced childhood asthma is easily grasped and enjoys a broad consensus as a desired goal, some ecological outcomes (e.g., increased biodiversity) may be more esoteric. Thus, the Agency, working with the scientific community, should provide leadership in defining specific ecological outcome measures that can be used to quantify progress toward the widely held goal of protecting ecological integrity.

This challenge of clearly articulating ecological outcomes should be confronted early, in the Problem Formulation Phase of the IED. Through open and frank discussion among assessors, managers, stakeholders, and the public, common agreement should be reached on the desired outcomes and the risk reduction options of choice. In addition, the selection of ecological outcome measures should be guided by a conceptual model, also developed in the Problem Formulation Phase, that relates expected changes in specific environmental outcomes to anticipated changes in stressors resulting from specific management approaches or interventions. The Problem Formulation Phase should also settle on the measures or indicators of those changes that will be monitored and reported. Generalized conceptual models of various ecosystem types and landscape units may be used to provide a scientific foundation for the conceptual framework.

While easy to give, in the past this advice has been difficult to follow. But there is reason to believe that much more is possible today than was possible just a few years ago. For instance, the idea of a conceptual model is working its way into the mainstream (Ref: EPA Eco RA GLs). The notion of appropriate monitoring endpoints is being integrated into ecological risk management plans (Ref: South Florida). The concept of adaptive management to address ecological problems is becoming more commonplace (Ref: SETAC on adaptive management). The practice of stakeholder involvement has gained wide acceptance (Ref: Comm on RA/RM)

Therefore, the Agency should vigorously pursue the development of ecological EDRCs to evaluate the performance of its integrated environmental decisions. At the same time, the Agency should be aware of and support -- by contribution to and utilization of the results of -- efforts to generate report cards to assess the overall condition of the nation's environment. These efforts can incorporate the results of individual EPA risk reduction actions in the process of addressing the broader questions of ecological health; e.g., "Are our landscapes and ecosystems functionally and structurally intact, supporting a broad range of native animal and plant communities, or not?" The National Science and Technology Council recently concluded that comprehensive assessment of the nation's natural resources will require an integrated national strategy for environmental monitoring and research (CENR, 1997). An ongoing project by the Heinz Center, funded by the White House's Office of Science and Technology Policy, is intended to produce a national environmental report card, integrating environmental program results across the federal government, by the year 2001 (Heinz Center, 1998).

The choice of appropriate ecological outcome measures should be guided by a conceptual model that relates specific environmental goals and sub-goals to management approaches or interventions. The conceptual model should, in turn, link expected changes in stressor levels to expected changes in human or ecological condition associated with specific risk reduction actions. In order to assess ecological condition, it will also be important to select measures of ecological attributes for which cause-and-effect relationships with stressors (at least those being managed) have not been postulated. The model should also indicate the measures or indicators of that change that will be monitored and reported. In addition, it is important to evaluate comprehensively the essential characteristics of landscapes and ecosystems across levels of hierarchy and time scales in a way that will allow aggregation and disaggregation of information for a variety of geographical boundaries. In order to provide a scientific foundation for the conceptual framework, it would be useful to employ generalized conceptual models of various ecosystem types and landscape units.

7.3.3 Additional Design Considerations

Environmental Decision Report Cards (EDRCs) should exhibit the following general characteristics:

a) **Transparent:** First and foremost, each EDRC should be derived in a clear and methodical fashion from objective measurements. Whenever expert judgment is required to aggregate components of the report card (such as aggregating disease information on separate subpopulations or aggregating information regarding the quality of different habitats within an ecosystem), the method of aggregation should be laid out explicitly. If algorithms are used to combine different types of measurements, the values assigned to the elements in the algorithm should be documented and justified. Ideally, EDRCs should include the results of sensitivity analysis and uncertainty analysis. Clear documentation of the algorithms used also helps to assure continuity over time, so that report card results from one decade can be fairly compared to the next since for some outcomes, such comparisons will need to be made over many years, even decades or generations.

b) **Relate closely to environmental goals:** Measures included in the report card should be directly related to the specific goals of the risk reduction program. In other words, the design of the report card should allow one to understand how changes in performance measures relate to the attainment of environmental goals and vice versa. Although different versions of the EDRC (with differing amounts of technical detail and aggregation) will be needed by the various audiences and users of performance information, the underlying scientific framework must be consistent. Ideally, the logic upon which the EDRC is based should be both systematic and hierarchical so that the information can be a) aggregated-- at least qualitatively--to a national or international scale, and b) disaggregated to meet regional and local needs in other report card exercises.

c) **Build a continuing historical record:** The EDRC should set forth a frame of reference for each important ecological, human health, or quality of life goal, indicating its past and current status, desired long-term condition, and shorter term milestones. To be truly effective, the EDRCs must be maintained and compared from time to time in order to (1) determine the trend, (2) become aware of new factors that might be

impacting on the goal, and (3) learn the lessons that adaptive management has to teach.

d) **Provide an integrated picture**: The report should be organized to convey a more integrated picture of health (e.g., a healthy terrestrial ecosystem), rather than just progress toward stressor/pollutant-specific goals; e.g., a target effluent level. In commenting on the draft EPA document, *Environmental Goals for America* (EPA, 1996), the SAB noted that the structure of the draft report was not ideal for this purpose because it was "centered on the current program office- or media-specific issues" (SAB, 1997).

When the IED framework is applied to multiple environmental risks, the performance measures selected during Phase II and monitored during Phase III may be tied to specific risk reduction tools. Indeed, designers and implementers of risk reduction programs require information on the relative effectiveness of selected risk reduction approaches in meeting risk reduction targets, as assessed against a number of decision criteria (e.g., timeliness, cost-effectiveness, and equity). Such reassessment requires that risk reduction be attributed to specific risk reduction tools, to the extent possible. At some point, however, it will also be important that performance measures be reported in combination in an integrated report card so that overall progress toward achieving a goal can be assessed.

e) Credible: An additional consideration in the design of report cards is the question of who will generate them. It is valuable, from the standpoint of learning, for those implementing the risk reduction measures (whether in the public or private sector) to generate such reports. It is possible, however, that information will be gathered and reported by some group not directly involved in the implementation. Such a scheme has the advantage of independent reporting but reduces the opportunity for learning by those implementing the plans. A third possibility is for the report cards to be prepared by those implementing the plan (self-reporting), combined with an outside audit program by auditors not involved in the implementation. This latter combination has the advantage that those implementing the plans learn as they produce their reports; they also keep their eyes on the yardsticks set up for measurement as they implement plans; at the same time, the problems that can frequently arise with self-reporting schemes are counteracted by the periodic audits.

7.4 Implications for Existing Monitoring Systems

Decisions on the design of a report card system have implications for information collection systems, both those that track administrative processes and those that collect environmental data (e.g., ambient monitoring programs). In selecting performance measures, there will often be a gap between what it would be desirable to know and what it may be feasible to know. This gap may exist as a result of several factors, including the limited state of knowledge about the relationship between stressors and effects, the costs involved in obtaining certain types of information, and the willingness of affected people to provide information. Nonetheless, it is important that careful consideration of the most desirable performance measures, including those based on an established chain from process and stressor measures to exposure and outcome measures, influence the types of information that are actually collected.

Existing monitoring systems, both those geared toward environmental condition and those intended to assess human health and welfare, provide invaluable information, particularly regarding the implementation of current legal mandates. Compliance monitoring and ambient air and water quality monitoring networks, for example, provide significant process as well as some outcome data. The EDRCs and overall report cards designed to answer broader questions, however, will likely require changes to existing monitoring systems.

Ecological Health Outcomes: Assessments of ecological integrity, for example, require information not only on water quality, air quality, soil quality, water flow, and populations of certain species, which are commonly monitored today, but also measurements of biological community structure, the presence of successional states, diversity of habitat types across the landscape, connectivity, altered topography, hydrodynamic patterns, and so forth. This latter set of parameters is not typically monitored by EPA, but has received greater attention by other federal agencies (USGS, USFS). This fact emphasizes the importance of strengthening and maintaining the collaboration among the many agencies that conduct monitoring, an effort begun several years ago under the auspices of the Committee on Environment and Natural Resources (CENR, 1997). Linking monitoring efforts into a coherent framework will improve the quality and robustness of the EDRCs.

Human Health Outcomes: As noted above, much of the data, both on environmental exposures and on health outcomes, needed to evaluate the effectiveness of a risk reduction program are not currently being collected. The measurement of disease for the total population exists only for diseases recorded on death certificates. Mortality data do not provide information on the incidence or the occurrence of disease, but only death from a specific disease. The incidence of cancer is now being recorded in many states (ref CDC Cancer Registry Program), but the cases cannot be tied to exposures because of the long latency of the disease. Still more important, many health outcomes today are associated with environmental exposures but records are not routinely kept on these diseases for the general population. For example, asthma episodes, behavioral changes in children, and minor neurologic deficits are known to be important effects of various exposures, but there is no complete database regarding these outcomes that could serve to link these effects to specific exposures. Testing for respiratory function, intelligence, hearing, and vision is done on many segments of the population, but the information resources are fragmented and have not been used to evaluate the effects of environmental change. The available data linking environmental exposures to health outcomes have come from special studies on limited and often selected populations. Further, these outcomes are usually evaluated using surveys designed to detect trends, rather than to look for potential causes. For many problems, only certain subpopulations are at risk, either because of their susceptibility or special lifestyles that may increase exposures. Little information is available specifically on these subsets to allow for evaluation.

Exposure: The databases on human exposure have often been limited by infrequent national sampling sources, e.g., the Center for Disease Control's National Health and Human Nutrition Examination Survey (NHANES), and the Total Exposure Assessment Methodology (TEAM) study (EPA, 1987). States have far more data available but these data often are not in electronic format and available for easy use. Air and water monitoring data are only indirect measures of exposure and reflect only a limited number of monitoring stations. In some situations, only non-compliance data are recorded. This practice limits the usefulness of the information for modeling exposures of individuals over space and time. Research to develop a better understanding of the relationships between centrally located monitoring sites and the actual human exposures could make some of these data more useful. Ultimately,

however, more direct monitoring of human exposures will be required to demonstrate the effectiveness of various risk management strategies.

Measurements of pollutants in environmental media are less intrusive than human exposure measurements. However, the knowledge needed to link exposure of individuals to measures of the pollutant in various environmental media has not been well developed. For example, we have monitored outdoor air for many years. However, to use such data for exposure assessment, we must have information on time-activity patterns and the penetration of outdoor air pollutants into buildings since most of the population spends about 90% of the time indoors. In addition, exposures to indoor pollutant sources (e.g., environmental tobacco smoke) will add to those from outdoor air, and will also contribute to adverse health effects. Similarly, measurements of a pollutant in food supplies may not capture changes, both losses and gains, that occur during food preparation in homes and restaurants. Ultimately, considerably more measurement of exposures will be required, as well as measurement of changes in exposure over time, in order to meet the challenges of performance evaluation.

In short, the concept of report cards will challenge assessors to make the most out of the data that are available and, perhaps, to re-define and re-design the systems in place to gather environmental data. The goal is to answer questions that people want to have answered.

7.5 Summary and Recommendations

The extent of environmental data collected (e.g., on contaminant levels in various environmental media, on tissue burdens, on human exposures, and disease) is expansive and holds the promise of improved measures to generate effective Environmental Decision Report Cards (EDRCs). At the same time, in order to achieve this promise, the current information reporting systems would benefit from additional development and, in some cases, restructuring. An improved performance evaluation system will be characterized by clear articulation of objectives during the Problem Formulation (Phase I), careful selection of performance measures in Analysis and Decision-Making (Phase II), and refinement, integration, and possible expansion of existing data collection efforts in Implementation and Performance Evaluation (Phase III).

In closing, the SAB would like to direct the Agency's attention to the promise and challenge of environmental reporting as discussed in this report. That is, reporting in the context of the IED framework would move the Agency further along the spectrum of measures illustrated in Figure 7-1, increasing the attention to observable environmental outcomes from specific management programs. In addition, the challenge of combining information from a number of integrated decisions into an overall environmental report card remains. With respect to this next generation of performance evaluation, the SAB offers some initial observations:

a) Implementation of a more integrated form of environmental decision-making will require an improved environmental report card system that provides reliable and accurate information about the issues people care about: how healthy are the nation's citizens? is the integrity of our ecological systems intact? is the quality of life getting better or worse? In order to answer these questions effectively, government agencies and the public need to develop a reporting system that focuses on the desired states of human health and ecological condition, rather than program-specific objectives.

b) Such a system should build on the report cards devised to evaluate specific integrated decisions, along with information from other sources, to report on the overall human health and ecological condition.

c) In this regard, the Agency, in concert with others in the National Science and Technology Council, can play a leading role in designing the improved reporting system by developing a conceptual framework that: (1) defines ecological integrity and desired human health results in terms of measurements that can be used at the local, state and national levels, and (2) provides a systematic and transparent methodology for aggregating these measurements into an overall picture of health and integrity.

d) Research on selected report card components, such as methods of accurately measuring certain ecological characteristics at the landscape scale, methods of identifying susceptible human subpopulations at risk for disease, and methods for combining these into more aggregated measures of performance will be required and should proceed concurrently with the development of an overall report card framework.

e) Specifically, current data collection and analysis systems should be upgraded

so that information can be assembled in a format that relates directly to the conceptual framework, as well as the EDRC, and that lends itself to aggregation along a variety of geographical boundaries.

f) The Agency should provide a results-based environmental report card regularly to the public and to policymakers, and use such a report card — along with specific EDRCs — as a tool to support re-evaluation of program implementation within the Agency.

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PART VI CONCLUSIONS AND RECOMMENDATIONS

CHAPTER 8: CONCLUSIONS AND RECOMMENDATIONS

Over the course of the Integrated Risk Project, the Steering Committee and the project subcommittees came to a number of conclusions about the need for and benefits of improved methods of assessing and managing aggregate risks, broadening benefit/cost analyses to include difficult-to-monetize values, explicitly considering the role of values in decision-making, and so forth. The preceding chapters include suggestions and recommendations for specific improvements in the methodologies and approaches that underlie the IED framework. In addition, however, the Steering Committee came to a number of broad conclusions as a result of the discussions and thinking that led to the proposed IED framework itself. These overall conclusions and recommendations, which are the focus of a companion document, *Integrated Environmental Decision-making in the 21st Century: Summary Recommendations*, are summarized below:

> **Recommendation 1: EPA should accelerate the transition to integrated, outcomes-based environmental protection and apply the an integrated environmental decision-making framework in selected cases, while maintaining the safeguards afforded by the current system.**

Previous environmental management approaches, both regulatory and non-regulatory, have resulted in substantial improvements in human health and ecological condition, particularly in regard to chemical risks. While maintaining the current regulatory and management structure, the IED framework offers an approach for going beyond current protections by taking a truly integrated look at risks, opportunities for risk reduction, and the consequences of addressing (or failing to address) those risks. The focus of this next generation of environmental decision-making should be on environmental results; that is, on demonstrable outcomes (improvements) in the environment resulting from integrated action. In order to further incorporate integrated environmental decision-making into EPA programs, the Agency should select a few test cases where the IED framework can be applied explicitly. Such test cases offer the best means for demonstrating the potential of the IED framework, identifying needed improvements in its underlying methodologies, and realizing its benefits with real-world problems.

Recommendation 2: Because science plays a critical role in protecting the environment, EPA should commit the resources necessary to expand the scientific foundation for integrated decision-making and outcomes-based environmental management.

An important theme of the Integrated Risk Project is the need to integrate scientific and technical information on risks and risk reduction opportunities with information on societal values, aspirations, and goals. The call for greater inclusion of multiple disciplines and points of view, however, must not obscure the fact that science and scientific methods are critical to sound environmental decision-making. Science has a unique and critical role to play in protection of the environment. It is through scientific investigations that many environmental problems are first discovered (e.g., the depletion of stratospheric ozone, hypotheses about environmental endocrine disruptors). Science also is instrumental in developing, testing, and evaluating risk reduction options, and social sciences offer techniques for assessing societal preferences and wants. Implementation of the IED framework, particularly its application to multiple-stressor problems, will in many cases require new science, both empirical and theoretical. In order to gain the greatest benefit from the IED approach, the Agency will need to invest in research designed to support integrated risk assessment and management.

Recommendation 3: EPA should apply and encourage the broader use of risk comparison methodologies, such as those described in this document, that clearly identify how scientific information and judgment are incorporated into risk comparisons.

Scientific information on risk, such as quantified risk assessments and scientifically demonstrated relationships between stressors and effects, provides the essential basis for making objective risk comparisons. However, desired scientific information often is incomplete or absent, and scientists have to use their best professional judgment to bridge important gaps in the data. In other cases, scientific information and analysis, by themselves, are not sufficient for comparing risks; e.g., when comparing human healths risks vs. ecosystem risks or cancer risk in adults vs. neurologic risk to children. In these instances, it is public values that come into play in making the comparisons. Thus, methodologies used to compare environmental risks--

whether human health risks, ecosystem risks, or both--should incorporate not only the most up-to-date scientific information, but also should identify explicitly where professional judgment and values have influenced the results. The SAB has developed two prototype methodologies for risk comparisons that meet these criteria. These approaches should be points of departure for the Agency as it seeks to develop and use science-based methodologies for comparing relative environmental risks.

Recommendation 4: EPA should use a broader range of risk reduction options in combination to manage environmental risks.

In 1990, the SAB recommended in *Reducing Risk* that the nation make greater use of all the tools, including market forces, information, and product specifications, available to reduce risk. Now, almost a decade later, many of those tools are being used to a greater extent than ever before. The challenge for EPA is not only to expand the use of those various tools, but also to use them in creative, coordinated ways to reduce multiple risks to multiple receptors in communities and ecosystems across the country. The SAB has developed a risk reduction options methodology that can be applied fruitfully to single stressors. However, the methodology should be especially useful for identifying multi-dimensional strategies to control complex environmental problems involving many sources, stressors, and receptors.

Recommendation 5: When evaluating risk reduction options, EPA should weigh the full range of advantages and disadvantages, both those measured in dollars as costs and benefits and those for which there may not be a comprehensive dollar measure, such as sustainability and equity.

The valuation framework that underlies environmental decisions is the simple formulation of whether the gains that accrue from protective actions are worth what is given up to attain them. There is a subsidiary question: would other possible actions be preferable? This question highlights the importance of taking all effects, including long-term effects, into account. It also raises the sometimes hidden issue of how people value different aspects of the environment. Some of society's environmental values can be measured directly in monetizable terms, and others can be inferred and translated into monetizable terms with some confidence. But other things that people value, such as sustainability and equity, often may be expressed only qualitatively, yet are of no less importance for that reason. As a part of this project, the SAB has

described the applicability and limitations of the benefit/cost framework, and suggested areas where new approaches to characterizing values are essential. The SAB also has attempted to define better the full range of relevant questions that must be considered in ecological valuation.

Recommendation 6: EPA should make fuller use of the scientific methods available to characterize public values and incorporate those values into goal-setting and decision-making.

Community and national values have been and will continue to be a primary driver of community-level and national efforts to protect the environment. However, values usually are not weighed transparently in the decision-making process. Rather, they are usually implicit in the judgments made by decision-makers. Thus, they influence decisions in ways that are not clear to, or reviewable by, the public.

Because public values undoubtedly help shape environmental decisions, it is important to understand and document their role in and influence on decision-making. It is also important to elicit public values systematically, differentiate values from technical information as a part of decision-making, and include their effects on decisions as part of the public record. Individual and community values should be solicited systematically by social scientists and other appropriately trained individuals. The deliberative processes that are used in arriving at decisions should involve professionals trained in fields like consensus-building and dispute resolution. EPA should make more extensive use of existing expertise in the areas of behavioral science and decision logic so that a more complete representation of community values is incorporated into the Agency's decisions.

Recommendation 7: EPA should identify, collect, and disseminate scientifically-based environmental metrics organized in new ways to support a more integrated approach to managing environmental risk.

The transition to and effectiveness of integrated, outcomes-based environmental protection will depend to a large extent on the availability and utilization of appropriate information in the area of exposure, human health, ecological health, and quality of life. Current information collection efforts, however, often are insufficient, inadequately organized, or focused on inappropriate endpoints. In the area of human health, for

example, data on non-fatal outcomes of environmental exposures, such as asthma, are not being collected except in a rudimentary manner or at certain sites. Only mortality information is collected systematically at all locations and can be related to some limited information on selected exposures. With regard to ecological data, more comprehensive information is needed on the current status of ecosystems such as wetlands, lakes, forests, and grasslands. This information should include the extent to which each ecosystem is exposed to and affected by non-chemical stressors such as habitat conversion, habitat fragmentation, and invasions of exotic species.

The challenge facing EPA and the nation is not only one of collecting the right environmental data, but of finding new ways to manage and use that data. Working with other federal and states agencies, EPA should take a leadership role in: identifying critical environmental data gaps, including data on exposures and health and ecological outcomes; integrating the largely fragmented data collection efforts already underway; and disseminating integrated environmental information to decision-makers and the public.

Recommendation 8: EPA should develop a system of "report cards" to organize and disseminate information on the status of ecological and human health and the quality of life in order to assess the effectiveness of its environmental decisions and to guide future environmental management.

One of the most valuable uses of environmental data is to measure the results of the actions society takes to reduce environmental risk. However, even if such data did exist, through a vigorous implementation of Recommendation 7, there would still be a need to develop widely accepted and commonly used methods for evaluating a) the state of our environment and b) the success of national environmental protection efforts -- where success is measured in terms of demonstrable outcomes in the health of humans and ecosystems. Regarding the state of the environment, EPA should work with federal and state entities to develop indicators of ecological and human health conditions in our country. Such a environmental report card would be readily comprehensible to the public and policy makers alike, serving to galvanize them on action toward specific, measurable goals. The concept of a report card should also be applied to evaluating the impact and effectiveness of individual decisions made under the IED methodology; i.e., an environmental decision report card (EDRC).

Reporting on measures of progress towards ecological, human health, and quality of life goals will provide a number of benefits. First, the exercise of defining performance measures and reporting on them will bring more focus and discipline to the Agency by expressing the relationship between investments (measured in time, money, or information) and environmental results. Second, shorter term measures of progress (including process measures or measures of stressor reductions) will be useful for accountability and course correction. Third, longer term measures of progress (such as improvements in overall human and ecosystem health and quality of life) will be most helpful in determining whether goals are being met, whether further actions are needed to control well-recognized stressors, or whether new actions are needed to control new stressors.

To strengthen their credibility and utility, environmental report cards should contain information derived from objective measurements, be transparent and clearly documented, and provide integrated information on progress towards multiple inter-related environmental goals.

<u>Recommendation 9</u>: EPA should expand and develop new collaborative working relationships with other federal and non-federal governmental agencies and others who also will be involved in integrated environmental decision-making.

Inherent in the IED framework is the idea that problem formulation and decision-making must match in scale and location. In some cases, decisions may be most effective when local or state governments play the primary role. In others, coordinated action across several levels of government or among a number of state and/or local governments will be required. In still others, the current emphasis on centralized decision-making may be the preferred approach. In any event, integrated thinking about environmental problems will tend to drive decision-making to the agency or level of government where decisions are most appropriately made. As a consequence, EPA's role will evolve to one in which the depth of control gives way to broader involvement in partnership with other agencies. EPA will continue to be responsible for implementing and enforcing federal environmental laws and statutes, conducting environmental research and development, and conducting stressor-specific risk assessments. At the same time, the Agency can exert national leadership to bring

together appropriate agencies and stakeholders to explore integrated approaches to environmental problems.

Recommendation 10: EPA should aggressively explore options for reducing risks from significant stressors that currently are addressed inadequately by the nation's environmental institutions.

Over the course of the Integrated Risk Project, it became clear to the SAB that a number of important human health and ecological risks are not being addressed adequately by the nation's environmental institutions. In many cases this is because risk management responsibility is not clearly assigned to any one government entity, or is scattered over many agencies and/or levels or government. This fragmented approach results in uncoordinated and incomplete efforts to identify cause-and-effect linkages and to manage those risks. With regard to ecological risks, the SAB has concluded that many of the highest ranking risks (e.g., hydrologic alterations, harvesting of living marine resources, habitat conversion, climate change, and introduction of exotic species) are associated with physical and biological, rather than chemical, stressors, which do not fall clearly within the purview of any single federal agency. Important human health risks that remain unaddressed include those for which the environmental exposure link is suspected but not certain (e.g., asthma, brain cancer, and non-Hodgkins lymphoma) and risks associated with susceptible and/or compromised human populations.

To control many of these inadequately addressed risks will require the kind of integrated decision-making envisioned in this report. It will also required a new kind of integrated institutional leadership. When EPA determines that serious risks are not being addressed effectively by existing environmental institutions or decision-making systems, the Agency has a responsibility to inform the public about those risks and bring together the appropriate federal, state, and local agencies to address them.