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Guidelines for Preparing Economic Analyses

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U.S. Environmental Protection Agency

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Preface

The U.S. Environmental Protection Agency (EPA), Regulatory Policy Council oversees regulatory planning by the Agency. In this capacity, the Council establishes analytical procedures on risk management issues to ensure that high quality and consistent practices are followed in accordance with Federal and Agency regulatory procedures. In early 1996, the conduct and consistency of economic analyses prepared in support of regulatory actions were identified as areas in need of updated and more specific guidance than was presently available in the Agency. The Regulatory Policy Council assembled a group economists and policy analysts serving in offices throughout the EPA to serve on an Economic Consistency Workgroup. The Council instructed the Workgroup to develop a series of issue papers to assess important economic analytical issues facing the Agency. After discussing the results of the findings in the issue papers, the Council charged the Workgroup with preparing a guidance document to assist in the preparation of economic analyses used for regulatory development and policy evaluation conducted by the Agency.

The *Guidelines for Preparing Economic Analyses* (or *EA Guidelines*) is part of a continuing effort by the EPA to develop improved guidance on the preparation and use of sound science in support of the decision making process. The *EA Guidelines* provide guidance on analyzing the benefits, costs, and economic impacts of regulations and policies. The document draws from several previously published sources, including existing economic guidelines materials prepared by the EPA in the mid-1980s, other Agency economic analyses and handbooks, and materials prepared by the Office of Management and Budget in support of Executive Order 12866 on regulatory planning and analysis. It seeks to incorporate recent theoretical, empirical, and modeling advances in environmental economics, drawing upon the considerable body of scholarly literature.

In an effort to ensure the *EA Guidelines* presents sound, scientific information consistent with mainstream practices in environmental economics, the Agency's Science Advisory Board (SAB) was charged with undertaking an extensive peer review of the document. The review was performed by the SAB's Environmental Economics Advisory Committee (EEAC), comprising leading U.S. environmental economists affiliated with major colleges, universities and economic research institutions. The EEAC provided substantial input on the content and organization of the document, reviewing the materials for accuracy in both economic theory and practice. In their final review report to the Agency (included as Appendix A in this document), the SAB concluded that the *EA Guidelines* receive an overall rating of "excellent," saying it "succeed(s) in reflecting methods and practices that enjoy widespread acceptance in the environmental economics profession."

Constant advances in theoretical and empirical research in the field of environmental economics will require that the Agency reexamine the *EA Guidelines* on a continual basis. The Agency will again enlist experts in the field of environmental economics and engage in an open review of the scientific basis of the document when it is reevaluated in the future.

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Acronyms and Abbreviations

AC	annualized costs
BAT	best available technology
BCA	benefit-cost analysis
BPT	best practicable technology
CA	conjoint analysis
CE	certainty equivalent
CEQ	Council on Environmental Quality
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CFC	chlorofluorocarbons
CFR	Code of Federal Regulations
CGE	computable general equilibrium
COI	cost of illness
CPI	Consumer Price Index
CR	contingent ranking
CV	contingent valuation
DALY	disability-adjusted life year
DOE	Department of Energy
DOT	Department of Transportation
DWL	dead weight loss
EA	economic analysis
EBIT	earnings before interest and taxes
EEAC	Environmental Economics Advisory Committee
EIA	economic impact analysis
EO	Executive Order
EPA	Environmental Protection Agency
FINDS	Facility Index Data System
FTE	full-time equivalent employment
GDP	gross domestic product
I/O	input-output
IPCC	Intergovernmental Panel on Climate Change
LP	linear programming
MR	marginal revenue
MPC	marginal private costs
MSC	marginal social costs
MSD	marginal social damages
NAICS	North American Industrial Classification System
NB	net benefits

NEPA	National Environmental Policy Act
NESHAP	National Emission Standard for Hazardous Air Pollutant
NFV	net future value
NOAA	National Oceanic and Atmospheric Administration
OCC	opportunity cost of capital
OECD	Organization for Economic Cooperation and Development
OLS	ordinary least squares
OMB	Office of Management and Budget
OSHA	Occupational Safety and Health Administration
PRA	Paperwork Reduction Act
POTW	publicly-owned (wastewater) treatment work
PVC	present value of costs
QALY	quality-adjusted life year
RAPIDS	Rule and Policy Information Development System
RFA	Regulatory Flexibility Act
RIA	regulatory impact analysis
RUM	random utility model
SAB	Science Advisory Board
SAM	social accounting matrix
S&P	Standard & Poors
SBA	Small Business Administration
SBREFA	Small Business Regulatory Enforcement Fairness Act
SIC	Standard Industrial Classification
TAMM	Timber Assessment Market Model
TSLS	two-stage least squares
UMRA	Unfunded Mandates Reform Act
USC	United States Code
VSL	value of statistical life
VSLY	value of statistical life-year
WTA	willingness to accept
WTP	willingness to pay

Acknowledgments

The preparation of the *Guidelines for Preparing Economic Analyses* (or *EA Guidelines*) was managed under the direction of the Regulatory Policy Council, chaired by the Deputy Administrator of EPA. Initial work on the *EA Guidelines* began in 1996 under the direction of Deputy Administrator, Fred J. Hansen during his tenure at the Agency, and concluded in 2000 under Deputy Administrator, W. Michael McCabe.

The principal manager of the Economic Consistency Workgroup was Al McGartland, Director, National Center for Environmental Economics. The Economic Consistency Workgroup consists of staff economists and policy analysts from across the Agency, who contributed to the development and review of the materials contained in the *EA Guidelines*. The primary writers/editors for the *EA Guidelines* were Chris Dockins and Brett Snyder. Additional persons responsible for final preparation of the text include: Liwayway Adkins, Kathleen Bell, Jennifer Bowen, Jared Creason, Richard Garbaccio, Richard Iovanna, Robin Jenkins, Elizabeth McClelland, Nicole Owens, Michael Podolsky, Keith Sargent, Nathalie Simon, Lanelle Wiggins and Melonie Williams.

Representatives from EPA's many different program offices contributed to development and review of the *EA Guidelines* as members of the Economic Consistency Workgroup. Persons playing more active roles in the review process included: Ghulam Ali, Rebecca K. Allen, Paul Balserak, Allen Basala, John B. Bennett, Paul Borst, Nick Bouwes, Jennifer Bowen, Ed Chu, Matthew Clark, Gary Cole, Jim DeMocker, Mark Eads, Ron Evans, Sandy Evelanko, Glenn Farber, John Faulkner, Leslye Fraser, Arthur Grube, Patricia Hall, Chris Herman, Bryan Hubbell, Barnes Johnson, Robert Lee, Jonathan Libber, Mahesh Podar, John R. Powers, Elliot Rosenberg, Jean Schumann, Ann Watkins, William Wheeler, David Widawsky and Jan Young.

In addition to EPA staff, a number of contractors developed key materials used as technical assistance in preparation of the *EA Guidelines* and supporting documents. They include Frank Arnold and Francis Sussman of ICF Consulting, Inc.; Lisa Robinson, Marla Markowski, Jim Neumann and Robert Unsworth, of Industrial Economics; Leland Deck, Ellen Post and Penny Schafer of Abt Associates; Robert Raucher, Brian Hurd, Lauraine Chesnut, David Mills, Greg Pitts, Tom Ottom of Stratus Consulting; Richard Bishop of University of Wisconsin-Madison; Michael Hanemann of University of California-Berkeley; David Burmaster of Alceon Corporation; and Anna Alberini of University of Colorado-Boulder.

The Science Advisory Board (SAB) review of the *EA Guidelines* was supervised by Donald G. Barnes, Director of the SAB Staff and by Dr. Joan M. Daisy, Chair of the SAB. Thomas Miller was the Designated Federal Official for the SAB-Environmental Economics Advisory Committee (EEAC) charged with conducting the peer review of the *EA Guidelines*. Dorothy Clark provided administrative management support to the SAB-EEAC.

The SAB-EEAC was chaired by Dr. Robert Stavins of the John F. Kennedy School of Government, Harvard University. SAB-EEAC members serving on the committee during the review of the *EA Guidelines* included: Dr. Nancy Bockstael of University of Maryland-College Park, Dr. Dallas Burtraw of Resources for the Future, Dr. Trudy Cameron of University of California-Los Angeles, Dr. Maureen Cropper of the World Bank, Dr. Herman Daly of University of Maryland-College Park, Dr. A. Myrick Freeman of Bowdoin College, Dr. Lawrence Goulder of Stanford University, Dr. Dale Jorgenson of Harvard University, Dr. Paul Joskow of Massachusetts Institute of Technology, Dr. Catherine Kling of Iowa State University, Dr. Charles Kolstad of University of California-Santa Barbara, Dr. Richard Revesz of New York University School of Law, Dr. Jason Shogren of University of Wyoming, Dr. Hilary Sigman of Rutgers University, Dr. Richard Schmalensee of Massachusetts Institute of Technology, and Dr. V. Kip Viscusi of Harvard University.

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Acknowledgments

Chapter 1: Introduction

Background to the Guidelines for Performing Economic Analyses

In December of 1983, the U.S. Environmental Protection Agency issued its *Guidelines for Performing Regulatory Impact Analysis*[1] *(RIA Guidelines)*.

Since their promulgation, the original RIA Guidelines have remained largely unaltered, experiencing only a few modifications and additions to specific sections during the 1980s.

Much has changed since 1983, however, so EPA has prepared these revised and updated *Guidelines for Preparing Economic Analyses (EA Guidelines)*. The revised *EA Guidelines* reflect the evolution of environmental policy making and economic analysis over the past decade and a half.

Recent years have seen an expansion of the universe of economic and social issues that are potentially affected by environmental policies. In 1983, the content of the analyses required for RIAs was driven mostly by Executive Order 12291, which directed federal agencies to assess the costs, benefits, and economic impacts of their rules, and established a formal review process by the Office of Management and Budget (OMB). This process and its goals were reaffirmed in 1992 with the issuance of Executive Order 12866 on regulatory planning and review. OMB subsequently released the document *Economic Analysis of Federal Regulations Under Executive Order 12866*[2] (or *Best Practices*), which served to illustrate specific techniques and issues concerning the conduct of economic analysis in support of EO 12866. More recently, OMB released the document *Guidelines to Standardize Measures of Costs and Benefits and the Format of Accounting Statements*[3] (or *OMB Guidelines*) which currently serves as guidelines to federal agencies on economic analysis.

In addition to requirements to prepare economic analyses set forth by Executive Order, economic assessments are also called for under various administrative statutes. For example, agencies are explicitly directed to examine whether their policies impose new "unfunded mandates" on state, local, and tribal governments, and to review economic impacts on small businesses, governments, and nonprofit enterprises under the Unfunded Mandates Reform Act of 1995 (P.L. 104-4) and the Regulatory Flexibility Act, as amended by the Small Business Regulatory Enforcement Fairness Act (5 U.S.C. 601-612).

Policy makers have also extended the scope of relevant effects to be considered beyond these mandatory

[1] U.S. Environmental Protection Agency, *Guidelines for Performing Regulatory Impact Analyses*. EPA-230-84-003, December 1983. Reprinted with Appendices in March 1991.

[2] U.S. Office of Management and Budget, *Economic Analysis of Federal Regulations Under Executive Order 12866*, January 11, 1996. This "*Best Practices*" document can be found at the U.S. White House, Office of Management and Budget website: http://www.whitehouse.gov/OMB/inforeg/riaguide.html under the section titled "Regulatory Policy" (accessed 8/28/2000).

[3] U.S. Office of Management and Budget, M-00-08 *Guidelines to Standardize Measures of Costs and Benefits and the Format of Accounting Statements*, March 22, 2000. The *OMB Guidelines* serves to implement Section 638(c) of the 1999 Omnibus Consolidated and Emergency Supplemental Appropriations Act and Section 628(c) of the Fiscal Year 2000 Treasury and General Government Appropriations Act. They require OMB to issue guidelines to help agencies estimate the benefits and costs of federal regulations and paperwork and summarize the results of the associated analysis. The *OMB Guidelines* can be found at the U.S. White House, Office of Management and Budget website: http://www.whitehouse.gov/OMB/memoranda/index.html under the section titled "Selected Memorandum to Heads of Federal Departments and Agencies" (accessed 8/28/2000).

assessments. For example, the Pollution Prevention Act was passed in 1990 and the Agency has undertaken new initiatives that explored voluntary, non-regulatory approaches to address past and potential future pollution sources. Economic assessment of these types of actions can provide useful information on the economic efficiency of allocating society's resources in these ways.

The *EA Guidelines* have been updated to keep pace with the evolving emphases policy makers place on different economic and social concerns affected by environmental policies. Underlying this exercise is the recognition that a thorough and careful economic analysis is an important component in designing sound environmental policies. Preparing high quality economic analyses can greatly enhance the effectiveness of environmental policies by providing policy makers with the ability to systematically assess the consequences of regulatory and non-regulatory actions. An economic analysis can describe the implications of policy alternatives not just for economic efficiency, but for the magnitude and distribution of an array of impacts. Economic analyses also serve as a mechanism for organizing information carefully. Thus, even when data are insufficient to support particular types of economic analyses, the conceptual scoping exercise may provide useful insights.

The *RIA Guidelines* focused appropriately not only on what was required for assessing costs, benefits, and economic impacts of policies, but also on the basic technical procedures for doing so. Over the past 15 years, however, economic science has developed new techniques for benefits estimation, different economic models for assessing costs and other effects, and greatly expanded data sources and related guidance materials. These are all reflected in this document.

As a result of these modifications and updates, the new *EA Guidelines* will continue to serve, as always, to ensure that the EPA's economic analyses are prepared to inform its policy making processes and satisfy OMB's requirements for regulatory review. The new *EA Guidelines* also seek to establish an interactive policy development process between analysts and decision makers through an expanded set of cost, benefit, economic impacts, and equity effects assessments, an up-to-date encapsulation of environmental economics theory and practice, and an enhanced emphasis on practical applications.

The Scope of the EA Guidelines

The focus of the *EA Guidelines* is on the economic analyses typically conducted for environmental policies using regulatory or non-regulatory management strategies. Other guidance documents exist for related analyses, some of which are inputs to economic assessments. No attempt is made here to summarize these other guidance materials. Instead, their existence and content are noted in the appropriate sections. The *EA Guidelines* follow generally the outline of OMB's *Best Practices* and the *OMB Guidelines*, except insofar as these guidelines embody assessment principles and policy advice developed recently by EPA for its economic analyses.

As with the previous *RIA Guidelines*, the presentation of economic concepts and applications in this document assumes the reader has some background in microeconomics as applied to environmental and natural resource policies. Thus, to fully understand and apply the approaches and recommendations presented in the *EA Guidelines* readers should be familiar with basic applied microeconomic analysis, the concepts and measurement of consumer and producer surplus, and the economic foundations of benefit-cost evaluation. Persons lacking these skills, but seeking to better understand economics, will require an alternative presentation of the materials contained in this document. Supplemental written material will be prepared to accompany this document, including training materials developed to reach a wider audience of individuals responsible for using the types of economic tools and information described here.

The *EA Guidelines* are designed to provide assistance to analysts in the economic analysis of environmental policies, but they do not provide a rigid blueprint or a "cookbook" for all policy assessments. The most productive and illuminating approaches for particular situations will depend on a variety of case-specific factors and will require professional judgment to apply. The *EA Guidelines* should be viewed as a summary of analytical methodologies, empirical techniques, and data sources that can assist in performing economic analyses of environmental policies. When drawing upon these resources, there is no substitute for reviewing the original source materials.

In all cases, the *EA Guidelines* recommend adhering to the following general principles as stated by OMB (EO 12866, Introduction):

> "'Analysis of the risks, benefits, and costs associated with regulation must be guided by the principles of full disclosure and transparency. Data, models, inferences, and assumptions should be identified and evaluated explicitly, together with adequate justifications of choices made, and assessments of the effects of these choices on the analysis. The existence of plausible alternative models or assumptions, and their implications, should be identified. In the absence of adequate valid data, properly identified assumptions are necessary for conducting an assessment."
>
> "Analysis of the risks, benefits, and costs associated with regulation inevitably also involves uncertainties and requires informed professional judgments. There should be balance between thoroughness of analysis and practical limits to the agency's capacity to carry out analysis. The amount of analysis (whether scientific, statistical, or economic) that a particular issue requires depends on the need for more thorough analysis because of the importance and complexity of the issue, the need for expedition, the nature of the statutory language and the extent of statutory discretion, and the sensitivity of net benefits to the choice of regulatory alternatives.'"

Thus, economic analyses should always acknowledge and characterize important uncertainties that arise throughout the analysis. Economic analyses should clearly state the judgments and decisions associated with these uncertainties and should identify the implications of these choices. When assumptions are necessary in order to carry out the analysis, the reasons for those assumptions must be stated explicitly and clearly. Further, economic analyses of environmental policies should be flexible enough to be tailored to the specific circumstances of a particular policy, and to incorporate new information and advances in the theory and practice of environmental policy analysis.

Organization of the EA Guidelines

The remainder of this document is organized into nine main chapters as follows:

- Chapter 2: Statutory and Executive Order Requirements for Conducting Economic Analyses reviews the major statutes and other directives mandating certain assessments of the consequences of policy actions;
- Chapter 3: Statement of Need for the Proposal provides guidance on procedures and analyses for clearly identifying the environmental problem to be addressed and for justifying federal intervention to correct it;
- Chapter 4: Regulatory and Non-Regulatory Approaches to Consider discusses the variety of regulatory and non-regulatory approaches analysts and policy makers ought to consider in developing strategies for environmental improvement;
- Chapter 5: Overview of Economic Analysis of Environmental Policy provides a theoretical overview of environmental economic analyses, as well as guidance concerning baseline specification and the treatment of uncertainty;
- Chapter 6: Analysis of Social Discounting presents a review of discounting procedures and provides guidance on social discounting in conventional contexts and over very long time horizons;
- Chapter 7: Analyzing Benefits provides guidance for assessing the benefits of environmental policies including various techniques of valuing risk-reduction and other benefits;
- Chapter 8: Analyzing Social Costs presents the basic theoretical approach for assessing the social costs of environmental policies and describes how this can be applied in practice;
- Chapter 9: Distributional Analyses provides guidance for performing a variety of different assessments of the economic impacts and equity effects of environmental policies; and

- Chapter 10: Using Economic Analyses in Decision Making concludes the main body of the *EA Guidelines* with suggestions for evaluating different policy approaches and options, and for presenting the quantified and unquantified results of the various economic analyses to policy makers.

Chapter 2: Statutory and Executive Order Requirements for Conducting Economic Analyses

Policy makers need information on the benefits, costs, and other effects of alternative options for addressing a particular environmental problem in order to make sound policy decisions. In addition, various statutes specifically require economic analyses of policy actions. General mandates may also direct agencies to conduct specific types of economic analyses. In some cases, agencies have established their own requirements for certain types of assessments of their policies. This chapter discusses specific requirements that apply to all of EPA's programs.[1]

OMB's basic requirements for regulatory review, including their *Best Practices* and *OMB Guidelines* documents, have helped to shape EPA's methodological and empirical approaches for conducting economic analyses. Several new mandates to conduct specific economic assessments of environmental policies have also recently been enacted. Many of the mandates that introduce economic analyses requirements of policies are briefly reviewed here.[2] In each case, citations for the relevant mandates or statutes and references to applicable EPA guidelines are provided.[3]

- **Executive Order 12866, "Regulatory Planning and Review"** requires analysis of benefits and costs for all significant regulatory actions. The Regulatory Working Group has prepared general guidance for complying with the requirements of EO 12866.[4] EO 12866 requires a statement of the need for the proposed action, examination of alternative approaches, and analysis of social benefits and costs. Chapters 3 through 8 of this document describe methods for meeting these requirements. EO 12866 also states that the distributional and equity effects of a rule should be considered. Chapter 9 describes methods for analyzing and assessing these effects.

- **The Unfunded Mandates Reform Act of 1995 (P.L. 104-4)** directs agencies to assess the effects of federal regulatory actions on state, local, and tribal governments, and the private sector. Agencies are to obtain meaningful input from state, local, and tribal governments for rules containing "significant federal intergovernmental mandates." These are federal mandates which may result in the expenditure by state,

[1] EPA personnel seeking information on EPA's policies and guidelines applicable to rule development can be found at the following EPA Intranet website http://intranet.epa.gov/rapids (accessed 8/18/2000, internal EPA document). Many of the citations included in this section can be found at this site. Note, this website and other additional websites referenced in this document are located on EPA's Intranet website and are limited to use by EPA personnel. When cited in this document, EPA Intranet websites will be labeled as "internal EPA document."

[2] Statutory provisions that require economic analysis but that apply only to specific EPA programs are not described here. However, analysts should carefully consider the relevant program-specific statutory requirements when designing and conducting economic analyses, recognizing that these requirements may mandate specific economic analyses.

[3] More information on some of these program-specific mandates can be found in Chapter 9 of this document.

[4] U.S. Office of Management and Budget, "Memorandum for Members of the Regulatory Working Group: Economic Analysis of Federal Regulations Under Executive Order No. 12866," January 11, 1996. The guidance also addresses the requirements of the Unfunded Mandates Reform Act and the Regulatory Flexibility Act.

local, and tribal governments, in the aggregate, or by the private sector, of $100 million or more in any one year.[5] UMRA also directs agencies to assess the effects of federal regulatory actions that will have a significant or unique effect on small governments. OMB has provided general guidance on complying with UMRA.[6]

- **Executive Order 13132, "Federalism"** requires consultation with affected state and local governments on rules that have federalism implications—that is regulations and policy statements "that have substantial direct effects on states (and local governments), on the relationship between the national government and the states, or on the distribution of power and responsibilities among the various levels of government." EO 13132 also imposes additional consultation obligations on agencies if they promulgate regulations with federalism implications that either: (1) impose substantial direct compliance costs on state and local governments not required by statute and do not provide funds to cover these costs, or (2) preempt state or local laws.[7]

- **The Regulatory Flexibility Act of 1980 (5 U.S.C. 610-612) (RFA)**, as amended by **The Small Business Regulatory Enforcement Fairness Act of 1996 (P.L. 96-354) (SBREFA)** requires that federal agencies determine if a regulation will have a significant economic impact on a substantial number of small entities (including small businesses, governments, and non-profit organizations.) If a regulation will have such an impact, agencies must prepare a Regulatory Flexibility Analysis and comply with a number of procedural requirements to solicit and consider flexible regulatory options that minimize adverse economic impacts on small entities. EPA has prepared Revised Guidance on complying with the RFA and SBREFA requirements.[8] Chapter 1 of that document provides guidance on the analytical requirements, including thresholds for determining "significant impact," "substantial number," and "small entities," and recommended quantitative measures for evaluating economic impacts on small entities.

- **Executive Order 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations"** requires federal agencies to identify and address, as appropriate, disproportionately high and adverse human health or environmental effects of its programs, policies, and activities on minority populations and low income populations. EO 12898 also requires the same consideration for Native American programs. EPA and the Council on Environmental Quality (CEQ) have prepared guidance for addressing environmental justice concerns in the context of NEPA requirements.[9] These materials provide definitions of key phrases in the Executive Order, which draw on draft guidance prepared by an interagency task force.[10]

- **Executive Order 13045, "Protection of Children from Environmental Health Risks and Safety Risks"** requires agencies to evaluate the health or safety effects of planned regulations on children. For economically significant rules that are subject to EO 13045, agencies are required to explain why the planned regulation is preferable to other potentially

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[5] U.S. Environmental Protection Agency, *EPA Guidance - Unfunded Mandates Reform Act of 1995, Interim Guidance*, March 23, 1995.

[6] U.S. Office of Management and Budget, "Guidance for Implementing Title II of S.1." Memorandum from Sally Katzen, Administrator, Office of Information and Regulatory Affairs, March 31, 1995.

[7] U.S. Environmental Protection Agency, *Interim Guidance on Executive Order 13132: Federalism*, February 2000.

[8] U.S. Environmental Protection Agency, *EPA Revised Interim Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act*, March 29, 1999.

[9] For more information see U.S. Environmental Protection Agency, *Interim Final Guidance for Incorporating Environmental Justice Concerns in EPA's NEPA Compliance Analyses*, Office of Federal Activities, April 1998, and Council on Environmental Quality, *Guidance for Addressing Environmental Justice under the National Environmental Policy Act (NEPA)*, March 1998.

[10] Interagency Working Group on Environmental Justice, *Final Guidance for Federal Agencies on Key Terms in Executive Order 12898*, August 8, 1995.

effective and reasonably feasible alternatives considered by the agency. EPA has prepared guidance on compliance with EO 13045.[11] Materials in Chapter 9 provide suggestions for the types of questions analysts could ask to characterize risks to children, and refers analysts to the various EPA guidance documents on risk assessment for information on analytic methodologies. While EO 13045 primarily addresses risk rather than economic analyses, economic analyses may be needed to determine whether EO 13045 requirements apply to a specific rule.

- **Executive Order 13084, "Consultation and Coordination with Indian Tribal Governments"** requires agencies to recognize the unique legal relationship with Indian tribal governments set forth in the Constitution and other treaties and documents. The order seeks to establish a regular and meaningful consultation and collaboration with Indian tribal governments in the development of regulations, imposition of unfunded mandates, and process for seeking waivers from federal requirements. The order seeks to encourage cooperation of tribal governments in development of regulations that significantly or uniquely affect their communities, including use of consensual mechanisms and negotiated rulemaking.

[11] U.S. Environmental Protection Agency, *EPA Rule Writer's Guide to Executive Order 13045: Guidance for Considering Risks to Children During the Establishment of Public Health-Related and Risk-Related Standards*, Interim Final Guidance, April 30, 1998.

Chapter 3: Statement of Need for the Proposal

3.1 Introduction

An appropriate point of departure for economic analyses of an environmental policy is a clear statement of the need for policy action. Key components of this discussion include an examination of the nature of the pollution problem to be addressed, an analysis of the reasons existing legal and other institutions have failed to correct the problem, and a justification for federal intervention instead of other alternatives. Statutory and judicial requirements that mandate the promulgation of particular policies or the evaluation of specific effects are also key factors in motivating certain analyses and policy actions. In some instances statutes prohibit the use of certain types of analyses in policy making. In these cases, the guidance presented in this document should be applied selectively to be consistent with such mandates.

3.2 Problem Definition

The initial problem definition discussion should briefly review the nature of the environmental problem to be addressed. The following considerations are often relevant:

- primary pollutants causing the problem and their magnitude;
- media through which exposures or damages take place;
- private and public sector sources responsible for creating the problem;
- human exposures involved and the health effects due to those exposures;
- non-human resources affected and the harm that results;
- expected evolution of the pollution problem over the time horizon of the analysis;
- current control and mitigation techniques; and
- the amount or proportion (or both) of the environmental problem likely to be corrected by federal action.

3.3 Reasons for Market or Institutional Failure

Following this concise problem definition summary should be an examination of the reasons why the market and other public and private sector institutions have failed to correct the problem. This component should be viewed as a key part of the process of environmental policy development because the underlying failure itself often suggests the most appropriate remedy for the problem.

Four categories of "market failure" are discussed in OMB's *Best Practices* in the sections titled externalities, market power, natural monopoly, and information asymmetry. For environmental conditions, externalities are the most likely causes of the failure of private and public sector institutions to correct pollution damages. However, information asymmetries and even pre-existing government-induced distortions can also be responsible for these problems.

Externalities can occur for many reasons. Transactions costs, for example, can make it difficult for injured parties to use legal or other means to

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cause polluters to internalize the damages they cause. A similar result can occur when property rights to the media or resources harmed are held in common or are poorly defined. Externalities can also arise because tracing the causal connections from activities that pose environmental risks to the resulting damages can be very difficult and often involve long time periods.

A comprehensive examination of the market's failure to address a specific environmental problem involves more than a statement that harms exist. Economic analyses should explore, for example, why transactions costs are high or why property rights are difficult to assign clearly. Similar analyses are appropriate for situations in which other factors are responsible for the failure of the market or other public and private sector institutions to address environmental problems.

3.4 Need for Federal Action

The final component of this initial statement of the need for the proposal is an analysis of why a federal remedy is necessary instead of actions by private and other public sector entities, such as the judicial system and state and local governments. Federal involvement is often required by pollution that crosses jurisdictional boundaries, by international environmental problems, and by statutory and other authorities. Economic analyses should make clear the basis for federal involvement by comparing it with the performance of a variety of realistic alternatives that rely on other institutions and arrangements. This discussion should also verify that the proposed action is within the relevant statutory authorities and that the results of the policy will be preferable to no action. Aspects of the regulations being proposed and promulgated that are not discretionary, but are dictated by statutory requirements, should be identified, as this may have an influence on the development of the economic analysis and presentation of the results.[1]

[1] The reader is also referred to Executive Order 13132 on "Federalism" for the introductory statements regarding principles of federalism and the section describing the doctrines of preemption.

Chapter 4: Regulatory and Non-Regulatory Approaches to Consider

4.1 Introduction

Once the need for federal policy action to address an environmental problem has been established, economic analyses should define and evaluate a range of possible regulatory and non-regulatory approaches. Many different approaches may help achieve efficient environmental protection. It is largely the analyst's responsibility to consider and characterize these approaches and then to present feasible alternatives for decision makers to consider early in the policy making process. The analyst should also be cognizant of constraints that may be placed on the use of non-regulatory approaches for addressing a specific environmental problem. Market-oriented options, for example, may not be consistent with statutory mandates and the best response to an environmental problem might require action outside the authority of the relevant statute.

This chapter briefly describes several of these approaches, but it does not attempt to detail the relative merits of putting them into practice for particular EPA policy initiatives. The goal here is to introduce several of the terms and concepts to analysts and to provide references that describe the conceptual foundations of each approach.[1] For some approaches, this chapter provides references on existing applications to environmental regulatory programs. Four general types of approaches are described below. The chapter concludes with some notes on fine-tuning policy approaches.

4.2 Traditional Design-Based Command and Control

Design-based command and control regulations have a long history in environmental policy, generally taking the form of specifying certain technologies or designs. These regulations usually impose the same requirements on all sources, although new and existing sources as groups are frequently subject to different standards.[2] An advantage of the approach is its relative ease of compliance monitoring and enforcement. Nonetheless, command and control regulations may be less cost-effective than other approaches, meaning that the same environmental protection might be achieved at a lower cost or more environmental protection might be secured for the same cost. Also, command and control regulations may not readily accommodate or encourage technological innovation or may fail to provide incentives to reduce pollution beyond what would be undertaken to comply with the standard.[3]

4.3 Performance-Oriented Approaches

Rather than mandating a particular technology for compliance, *performance-based standards* specify a source's maximum allowable level of pollution and

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[1] Baumol and Oates (1993), particularly Chapters 10-14, is a useful general reference on the economic foundations of many of these approaches.

[2] For a discussion on this subject and ways these types of programs lead to this result, see Helfand (1992).

[3] For some theoretical analyses of this point, see Malueg (1989), Milliman and Prince (1989), and Jung et al. (1996). A recent review of empirical literature can be found in Jaffe and Stavins (1995).

then allows the source to meet this target in whatever manner it chooses (e.g., the least costly and most flexible manner available). This approach has the advantage of allowing sources to effectively tailor pollution control requirements to their particular circumstances and encourages and accommodates technological innovation. Often, performance-based standards provide the opportunity to achieve the same goals more cost-effectively than command and control approaches. However, these approaches may place additional burdens on monitoring to ensure compliance and do not introduce incentives to reduce emissions or hazard levels beyond prescribed requirements.

4.4 Market-Oriented Approaches

A wide variety of methods for environmental protection fall under the general classification of *market-oriented approaches*. In one manner or another, each of these makes use of private sector incentives, information, and decision making in the pursuit of environmental improvement. Market-oriented approaches can differ from more traditional regulatory methods with regard to their economic efficiency and distribution of benefits and costs within the economy. These approaches include, for example:

- taxes, fees, or charges;
- subsidies;
- marketable permit systems;
- deposit-refund systems;
- offsets and bubbling;
- insurance/financial assurance requirements;
- liability rules; and
- information provision.

Some aspects to consider when choosing among these approaches as potential regulatory options are briefly described below.[4]

4.4.1 Descriptions of Market-Based Approaches

Taxes, fees, charges, and subsidies generally "price" pollution and leave decisions about the level of emissions to each source. For example, emissions of a toxic substance might be subject to an environmental charge based on the damages these emissions cause. Sources would individually decide how much to control these emissions based on the costs of the control and the magnitude of the charge. Taxes, fees, and charges have some highly desirable theoretic properties, including encouraging pollution control activities. However, they also sometimes impose substantially different burdens on pollution sources than do other approaches. One example is the potential liability that taxes, fees, and charges impose for residual pollution, which other approaches allow without charge. Issues surrounding the use of these approaches concern the collection of revenues and the distribution of economic "rents" from these programs, including deciding who should collect these fees (e.g., government or private sector) and what to do with revenues raised by these mechanisms (e.g., reduce other types of taxes on the regulated entities or redistribute the funds to finance other public services).

Marketable permit systems provide environmental improvements similar to those provided by taxes, fees, and charges. They function differently, however, in that the marketable permits approach sets the total quantity of emissions, while taxes, fees, and charges set the effective "price" of emitting pollutants.[5] If the permits are auctioned or otherwise sold to pollution sources, the

[4] This document does not go into the level of detail necessary to fully describe and provide a means of evaluating the relative merits of different regulatory and non-regulatory approaches. Instead, there is a growing literature on applied market-oriented approaches for environmental protection that should be reviewed prior to considering these regulatory approaches. For example, Anderson and Lohof (1997) and Stavins (1998a, 1998b) provide recent compilations of information on the theory behind and empirical use of economic incentives systems applied to environmental protection. Additional sources for details on incentive systems include Moore (1989), Tietenberg (1985, 1992), EPA (1991), OECD (1989, 1991), and proceedings published under the "Project 88" forum sponsored by the Center for Science and International Affairs, Harvard University (Stavins (1988, 1991)). These sources, and the references they contain, should be consulted for additional information concerning the design, operation, and performance of many of these instruments.

[5] The U.S. Acid Rain Program established under Title IV of the 1990 Clean Air Act Amendments is a good example of a marketable permit program. For recent economic analyses of this program see Joskow et al (1998) and Stavins (1998c). For more information on the program itself visit EPA's Acid Rain website at http://www.epa.gov/acidrain (accessed 8/28/00).

distributional consequences of this approach are similar to those experienced when using taxes, fees, and charges. However, if new entrants must obtain permits from existing sources, then the distributional consequences of permit systems will differ from those likely to arise after the introduction of technology-based standards. The potential to establish a barrier to entry on the basis of limiting quantities (e.g., if "grandfathering" of current emission sources is part of the program) can affect the eventual distribution of revenues, expenses, and "rents" within the economy. The ultimate distribution of "rents" under these programs can be an important feature of market-based approaches and, therefore, should be considered when comparing these with more traditional regulatory approaches.

Deposit-refund systems are like specialized forms of taxes. The deposit operates as a tax and the refund serves as an offsetting subsidy. Many good examples of deposit-refund systems exist, most of which are geared toward reducing litter and increasing the recycling rates of certain components of municipal solid waste.[6] Perhaps the most prominent examples are those programs associated with newspapers, plastic, and glass bottles.

Offsets and bubbling allow restricted forms of emissions trading across or within sources. This approach has seen widespread use, mostly in controlling air pollution in non-attainment areas. An offset, for example, would allow a new source of emissions in an airshed to negotiate with an existing source to secure reduction in the latter's emissions. This reduction would then be used to accommodate the emissions from the new source. Bubbling can allow a facility to consider all sources of emissions of a particular pollutant within the facility in achieving an overall target level of emission control or environmental improvement.

Insurance and financial assurance arrangements generally require those engaged in environmentally risky activities to ensure, typically through a third party, that sufficient resources will be available to remedy future damages. This arrangement harnesses the financial incentives of private sector companies to promote and maintain environmentally safer practices. An example of this approach to environmental protection is the financial assurance requirements related to closure and post-closure care for hazardous waste treatment, storage, and disposal facilities.

Liability rules are legal tools that allow victims (or the government) to force polluters that cause damages to pay for those damages after they occur.[7] They are typically applied to infrequent events such as cleanup of hazardous waste sites under CERCLA or cleanup after oil spills under the Oil Pollution Control Act. There are a variety of types of liability rules and in some situations these rules can mimic the desirable properties of taxes. However, this is not the case in all situations and even in those specific cases proper functioning of liability rules depends on a legal system which may not perfectly implement the rules.

Finally, **information provision** operates by ensuring that production and consumption decisions are adequately informed about the environmental and human health consequences of certain choices. In some cases, shifts in these decisions can encourage environmentally benign activities and discourage environmentally detrimental ones. The Toxics Release Inventory, consumer-based programs on the risks of radon in homes, and pesticide labeling programs are examples of efforts by EPA to implement information-based policy approaches.

4.4.2 Selecting Market-Oriented Approaches

The most appropriate market-oriented regulatory approach depends on a wide variety of factors, such as the nature of the market failure, the specific circumstances of the pollution problem, and the ultimate goals of policy makers.[8] The choice between taxes (or fees and charges) and marketable permits, for example, rests theoretically on such matters as the degree of uncertainty surrounding the estimated benefits and costs of pollution control as

[6] For example, Arnold (1995) analyses the merits of a deposit-refund system in a case study focusing on enhancing used-oil recycling and Sigman (1995) reviews policy options to address lead recycling.

[7] See Segerson (1995) for a discussion of the various types of liability rules, the efficiency properties of each type of rule, and an extensive bibliography.

[8] Helpful references that discuss aspects to consider when comparing among different approaches include EPA (1980), Hahn (1990), Hahn and Stavins (1992), and OECD (1994a, 1994b).

well as how marginal benefits and costs change with the stringency of the pollution control target. This choice also depends on distributional considerations and the extent to which policy makers are willing to allow the market to determine exact outcomes. Marketable permits, for example, set the total level of pollution control, but the market determines which sources reduce emissions and to what extent. Taxes, however, leave both the extent of control by individual sources and the total level of control to market determination.

Consideration should also be given to potential differences among economic instruments that have implications for the revenues collected under alternative mechanisms. The opportunities to direct collected resources at reductions in other inefficiencies introduced in markets that have consequences for economic welfare will affect the assessment of market-oriented approaches.[9]

The use of a particular market-oriented approach is often suggested directly by the cause of the pollution problem and constraints on the efficacy of other traditional policy instruments. For example, subsidies and deposit-refund systems place some enforcement burden on the regulated entities. This feature makes these approaches attractive if large numbers of small pollution sources exist and attempts to prohibit their actions are likely to fail due to risk of widespread noncompliance and costly enforcement. A positive incentive in these cases can solve both the original market failure and the enforcement problem.

Offsets and bubbles tend to be more appropriate when policy makers seek to help sources reduce compliance costs, while still attaining the environmental improvement embodied in a more traditional standards-based, source-by-source approach. Similarly, insurance and financial assurance mechanisms are useful instruments to supplement existing standards and rules when there is a significant risk that sources of future pollution might be incapable of financing the required pollution control or damage mitigation.

Finally, information remedies are often suggested when a market has failed to provide information and policy makers believe that private and public sector decisionmakers will act to address an environmental problem once the information has been disseminated. Voluntary approaches are closely related to information remedies and are most useful when they bring to bear the market's knowledge and innovation efforts on a particular environmental problem, and when direct standards-based methods would be very time-consuming and costly to develop.

4.5 Non-Regulatory Approaches

In addition to regulatory approaches, EPA has pursued a number of non-regulatory initiatives that rely heavily on *voluntary approaches* to achieve improvements in emissions controls and management of environmental hazards. Much of the foundation for these initiatives rests with the concepts underlying a "Pollution Prevention" approach to environmental management choices. In the Pollution Prevention Act of 1990, Congress established as a national policy that:

- pollution should be prevented or reduced at the source whenever feasible;
- pollution that cannot be prevented should be recycled in an environmentally safe manner whenever feasible;
- pollution that cannot be prevented or recycled should be treated in an environmentally safe manner whenever feasible; and
- disposal or other release into the environment should be employed as a last resort and should be conducted in an environmentally safe manner.

Working directly with a broad array of institutions that participate in decisions affecting the environment (e.g., consumers, regulatory agencies, industry), an effort is made to reach "common sense" understanding of the benefits and costs of management strategies that prevent damages from occurring, versus strategies aimed at reacting to the consequences of realized environmental hazards. Furthermore, some preventive measures can be instituted without establishing a regulatory program, but instead through a facilitated process of identifying problems and

[9] For useful references on the emerging issues concerning the uses of revenues from pollution charges (e.g., applying environmental tax revenues so as to reduce other taxes and fees in the economy) and ways to analyze these policies, see Bovenberg and de Moojii (1994), Goulder (1996), Bovenberg and Goulder 1996), Goulder et. al. (1997), and Jorgenson (1998a, 1998b).

solutions. This can involve sharing information and experiences among participants on the use of procedures, practices, or processes that reduce or eliminate the generation of pollutants and waste at the source. Examples within the manufacturing sector include developing and distributing information on input substitution or modification, product reformulation, process modifications, improved housekeeping, and on-site closed-loop recycling. Further, pollution prevention includes other practices that reduce or eliminate pollutants through the protection of material resources by conservation and increased efficiency in the uses of raw materials, energy, water, or other resources.

Examples of voluntary programs include: (1) the 33/50 toxic substances program under which many companies have established voluntary targets for reducing the use of various toxic chemicals, (2) the "ENERGY STAR" energy efficiency labeling program, and (3) the "Design for the Environment" program. The last of these programs seeks to form voluntary partnerships with industry and other stakeholders in order to develop environmentally safer alternatives to existing products and processes that prevent the need to cleanup pollution created as by-products in manufacturing processes. Much of the literature developed to document these changes can be found in public policy and industrial ecology literature sources.[10]

4.6 Fine-Tuning Policy Approaches

In addition to considering a wide variety of possible approaches for environmental protection, analysts and policy makers should also examine other characteristics of regulatory or non-regulatory policies that affect their costs and effectiveness. For example, evaluating benefits, costs, and other effects at different levels of stringency for a given policy can help to determine settings that provide the greatest net benefits to society. Similarly, tailoring pollution control requirements to account for geographical differences in environmental effects and source differences in pollution control costs will tend to achieve greater environmental protection at lower costs. Finally, phasing in policies over time to allow new requirements to be embedded in new investments can often substantially reduce a policy's costs while sacrificing relatively few of its benefits, especially when large-scale premature retirement of capital equipment can be avoided.

Constraints, such as statutory provisions, can limit the number of available regulatory and non-regulatory approaches for addressing a specific environmental problem. Market-oriented options, for example, may not be consistent with statutory mandates and the best response to an environmental problem might require action outside the authority of the relevant statute. Nevertheless, the strategy that best informs policy makers is generally one that adopts an expansive view of a problem's possible solutions and then provides cogent and detailed economic analysis of their benefits, costs, and other effects.

[10] For more illustrations of ongoing programs and policies, the following websites offer useful information: http://www.epa.gov/opei/ (accessed 8/28/2000) and http://www.epa.gov/p2/ (accessed 8/28/2000).

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Chapter 5: Overview of Economic Analysis of Environmental Policy

5.1 Introduction

This chapter provides a brief overview of several different analyses and assessments that are normally conducted in the course of evaluating environmental policies. It also presents background and guidance on several cross-cutting methodological topics. The suggestions in this chapter, and throughout this document, are not intended to be rigid rules to be applied uniformly for each and every economic analysis. Instead, they are intended to produce a consistent, well-reasoned, and transparent process for framing economic analyses regardless of the specific characteristics and features of any given policy.

The next section outlines a conceptual perspective for economic analysis and identifies the component assessments that together form an economic analysis in practice. This section also defines certain terms that are used throughout this and the remaining chapters of the *EA Guidelines*. The remaining sections of this chapter explore some common methodological elements that are shared by virtually all economic analyses of environmental policies. The third section of this chapter addresses the choice of analytic baseline and the fourth discusses predicting responses to new policies. Treatment of uncertainty is addressed in the fifth section and the final section addresses some emerging analytical issues. Each section first reviews the nature of the methodological topic and its impact on the economic analyses, and then provides general guidelines for incorporating or addressing associated issues in practice.

5.2 Economic Framework and Definition of Terms

A Conceptual Perspective for Economic Analysis

The conceptually appropriate framework for assessing all the impacts of an environmental regulation is an economic model of general equilibrium. The starting point of such a model is to define the allocation of resources and interrelationships for an entire economy with all its diverse components (households, firms, government). Potential regulatory alternatives are then modeled as economic changes that move the economy from a state of equilibrium absent the regulation to a new state of equilibrium with the regulation in effect. The differences between the old and new states—measured as changes in prices, quantities produced and consumed, income and other economic quantities—can be used to characterize the net welfare changes for each affected group identified in the model.

Analysts can rely on different outputs and conclusions from the general equilibrium framework to assess issues of both *efficiency* and *distribution*. At EPA these issues often take the form of three distinct questions:

- Is it theoretically possible for the "gainers" from the policy to fully compensate the "losers" and still remain better off?
- Who are the gainers and losers from the policy and associated economic changes?
- And how did a particular group—especially a group that may be considered to be disadvantaged—fare as a result of the policy change?

Chapter 5

The first question is directed at the measurement of efficiency, and is based on the *Potential Pareto criterion*. This criterion is the foundation of benefit-cost analysis, requiring that a policy's net benefits to society be positive. Measuring net benefits by summing all of the welfare changes for all groups provides an answer to this question.

The last two questions are related to the distributional consequences of the policy. Because a general equilibrium framework provides for the ability to estimate welfare changes for particular groups, these questions can be pursued using the same approach taken to answer the efficiency question, provided that the general equilibrium model is developed at an appropriate level of disaggregation.

Practical Compromises: Benefit Cost Analysis, Economic Impacts Analysis, and Equity Assessments

Although a general equilibrium framework can, in principle, provide the information needed to address all three questions, in practice analysts have limited access to the tools and resources needed to adopt a general equilibrium approach[1]. More often, EPA must resort to assembling a set of different models to address issues of efficiency and distribution separately. However, the limitations on employing general equilibrium models have greatly diminished in recent years with advances in the theory, tools, and data needed to use the approach. Chapter 8 contains additional information on general equilibrium models.

The *EA Guidelines* follow more traditional practices and adopt conventional labels to distinguish models or approaches used to answer questions on the efficiency and distribution of environmental regulations. For purposes of this document, the presentation separates the concepts and approaches into the following three general categories:

- the examination of net social benefits using a *benefit-cost analysis* (BCA);
- the examination of gainers and losers using an *economic impacts analysis* (EIA); and
- the examination of particular sub-populations, especially those considered to be disadvantaged, using an *equity assessment*.

This division is necessary not only because of data and resource limitations, but because analysts often lack models that are sufficiently comprehensive to address all of these dimensions concurrently. Within a BCA, for example, EPA is generally unable to measure benefits with the same models used for estimating costs, necessitating separate treatment of costs and benefits. Further, when estimating social costs there are cases in which some direct expenditures can be identified, but data and models are unavailable to track the "ripple" effects of these expenditures through the economy. For most practical applications, therefore, a complete economic analysis comprises a benefit-cost analysis, an economic impacts analysis, and an equity assessment.

Benefit-cost analysis evaluates the favorable effects of policy actions and the associated opportunity costs of those actions. The favorable effects are defined as benefits and the opportunities foregone define economic costs. While conceptually symmetric, benefits and costs must often be evaluated separately due to practical considerations. Analysts may even organize the analysis of benefits differently from the analysis of costs, but they should be aware of the conceptual relationship between the two. Using estimates of health and other risk-reduction effects provided by risk assessors, benefits analyses apply a variety of economic methodologies to estimate the value of anticipated health improvements and other sources of environmental benefits. Social cost analyses attempt to estimate the total welfare costs, net of any transfers, imposed by environmental policies. In most instances, these costs are measured by higher costs of consumption goods for consumers and lower earnings for producers and other factors of production. Some of the findings of a social cost analysis are inputs for benefits analyses, such as predicted changes in the outputs of goods associated with a pollution problem.

The assumptions and modeling framework developed for the BCA, constrain and limit the estimation techniques used to examine gainers and losers (in an EIA) or to examine impacts on disadvantaged sub-populations (in an

[1] The general equilibrium framework will at least capture all "market" benefits and costs, but may not include non-market benefits, such as those associated with existence value. In practice, models of general equilibrium may also be unable to analyze relatively small sectors of the economy. For more on general equilibrium analysis see Chapter 8, section 4.5.

equity assessment). To estimate these two categories of impacts we rely on a multiplicity of estimation techniques. The constraints faced by these analyses as well as details regarding estimation techniques are given by Chapter 9.

5.3 Baseline Specification

An economic analysis of a policy or regulation compares "the world with the policy or regulation" (the policy scenario) with "the world absent the policy or regulation" (the baseline scenario). Impacts of policies or regulations are measured by the resulting differences between these two scenarios. Measured differences may include changes in pollutant emissions and ambient concentrations, changes in usage or production of toxic substances, and incidence rates for adverse health effects associated with exposure to pollutants.

Specification of baseline conditions can have profound influence on the measurement and interpretation of analytic results. The complexity of the regulatory and policy-making stipulations may not yield a clear-cut decision on the specification of baseline conditions. The honesty and integrity of the analysis depend on the ability of the analyst to provide well-defined and defensible choices in the selection and estimation of baseline conditions. Analysts uncertain about the selection of baseline conditions are advised to review the guiding principles listed below. In the development of the rule, the analyst is responsible for raising questions about baseline definitions early within the regulatory development process, and should receive the views of enforcement and general counsel staff. Doing so can facilitate the consistent treatment of this issue in EPA analyses.

5.3.1 Guiding Principles for Baseline Specification

Baseline specification can be thought of as having two steps—selection and quantification. The first step is to select a baseline that is appropriate to the question the analysis is intended to address. The second step is to estimate the values of the relevant factors in the selected baseline scenario. Several guiding principles to assist in the treatment of baselines in an analysis are listed below. Though they exhibit a common sense approach to the issue, the analyst is advised to provide explicit statements within the analysis on each point. Failure to do so may result in a confusing analytic presentation, inefficient use of time and resources, and misinterpretation of the economic results.

- **Clearly state the question the analysis is addressing.** The type of regulatory question facing an analyst will affect the selection of the baseline in an analysis. A baseline definition appropriate to many analyses will be "reality in the absence of the regulation." However, to ensure provisions contained in statutes or policies precipitating the regulatory action are appropriately addressed, it is useful to assume full compliance with regulatory requirements in most cases.[2] Clearly stating the questions to be answered by the analysis will help not only in choosing an appropriate baseline, but also in communicating this information to persons using the results of the analysis.

- **Clearly identify all aspects of the baseline conditions that are uncertain and all assumptions made in specifying the baseline.** If the analyst had complete information about current values and perfect foresight about the future, the appropriate baseline conditions could be characterized with certainty. This, of course, is never the case. Current values of factors are often uncertain, and future values of factors are always uncertain. Estimates of uncertain factors should be based on actual data, to the extent possible. Uncertainties underlying the baseline conditions should be treated as other types of uncertainties are handled throughout the analysis. If, in the face of uncertainty, assumptions about baseline components are made, these should be the most realistic assumptions possible. For example, where reliable projections of future economic activity and

[2] Analysts should refer later sections of these guidelines (Section 5.3.2) and other cited EPA documents prepared in support of implementing these statutes, for more detailed guidance on the treatment of baseline definitions and compliance assumptions used for economic analyses required under these statutes. Much of the information on EPA's policies and guidelines applicable to rule development can be found at the following EPA Intranet website http://intranet.epa.gov/rapids (accessed 8/02/2000, internal EPA document).

demographics are available, this information should be accounted for in defining the baseline. All assumptions should be clearly stated, with particular attention given to situations calling for more than one baseline to be included in the analysis.

- **Be consistent throughout the analysis in the use of baselines.** The same baseline should be carried through for all components of the analysis. For example, the comparison of costs and benefits in a benefit-cost analysis should draw upon estimates derived using the same baseline, so that the calculation of net economic benefits yields a meaningful economic measure. Likewise, when comparing and ranking alternative regulatory options, the same baseline should be used for all options under consideration. When use of more than one baseline scenario is warranted, the analyst must avoid the mistake of combining analytic results obtained from different baseline scenarios. To limit confusion on this point, if multiple baseline scenarios are included in an analysis, presentations of economic information should clearly describe and refer to which baseline scenario is being used.

- **Determine the appropriate level of effort for baseline specification.** Every analysis is limited by finite resources. Analytical efforts should be concentrated on those components of the baseline that are most important to the analysis. If several components of the baseline are uncertain, the analysis should concentrate its limited resources on refining the estimates of those components that have the greatest effect on interpretation of the results.

- **Clearly state the "starting point" of baseline and policy scenarios.** A starting point of an analysis is the point in time at which the comparison between the baseline and policy scenarios begins. This is conceptually a point in time when the two scenarios are believed to diverge. For example, one approach is to organize the analysis presuming that the policy scenario conditions diverge from those in the baseline at the time an enforceable requirement becomes effective. Another convenient approach is to set the starting point to be coincident with promulgation of the final rule. These dates may be appropriate to use, as they are clearly defined under monitored administrative procedures, or represent deadlines that compliance progress can be measured against.

  However, where behavioral changes are motivated by the expected outcome of the regulatory process, the actual timing of the formal issuance of an enforceable requirement should not be used to define differences between the baseline and policy scenarios. Earlier starting points, such as the date authorizing legislation is signed into law, the date the rule is first published in a Notice of Proposed Rule Making, or other regulatory development process milestones, may be supported if divergence from the baseline occurs due to anticipation of promulgation. In some instances, parties anticipating the outcome of a regulatory initiative may change their economic behavior, including spending resources to meet expected emission or hazard reductions prior to the compliance deadline set by enforceable requirements. The same issues arise in the treatment of non-regulatory programs, in which voluntary or negotiated environmental goals may be established, leading parties to take steps to achieve these goals at rates different from those expected in the absence of the program. In these cases, it may be appropriate to include these costs and benefits into the analysis of the policy action, and not subsume these into the baseline scenario. The dynamic aspects of market and consumer behavior, and the many motivations leading to change, can make it more difficult to attribute economic costs and benefits to specific regulations. Looking at the sensitivity of the outcome of the analysis to these conditions and assumptions will be useful.

- **Let the duration of important effects of a policy dictate the structure of the analysis and baseline.** To consider how the benefits of a proposed policy compare with the costs of the policy, the analyst will assemble estimates of the present discounted values of the total costs and benefits attributable to the policy. How one defines the baseline is particularly important in situations in which the accrual of costs and/or benefits do not coincide due to lagged effects, or occur over an extended period of time. For example, the human health benefits of a policy that reduces leachate from landfills may not be manifest for many years, if the potential for human exposure through contaminated groundwater may occur decades after closure of the landfill. In theory, then, the longer the time frame, the more likely the analysis will depict all the benefits and costs of the policy that are expected to occur. However, forecasts of economic, demographic, and technological trends necessary for baseline specification must also span the entire period of the analysis. Because the reliability of many forecasts diminishes into the future, the analyst must balance the advantages of structuring the analysis to include a longer time span against the disadvantages of the decreasing reliability of the analytic results.

Defining the baseline and policy scenarios will often require information and assumptions on trends in behavior, and how these trends may be affected by regulatory management options. For example, the analyst may observe trends in economic activity or pollution control technologies that occur for reasons other than direct environmental regulations. For example, as the purchasing power of consumer income increases over time, demand for different commodities can change. Demand for some commodities may grow at rates faster than the rate of change in income, while demand for other goods may decrease. Therefore, where these trends are highly uncertain or are expected to have significant influence on the evaluation of regulatory alternatives (including a "no-regulatory control" alternative), the analyst should clearly explain and identify their choices in the analysis.

Lastly, in some cases the benefits of a policy will be expected to increase over time. Some analyses must therefore look far enough into the future to assure that benefits are not substantially underestimated. For example, suppose a policy that would greatly reduce greenhouse gas emissions were being proposed. In the baseline scenario, the level of greenhouse gases in the atmosphere would steadily increase over time, with a corresponding increase in expected human health and welfare and ecological changes. A benefit-cost analysis limited to the first decade after initiation of the policy would be likely to distort the relationship of benefits and costs associated with the policy. In this case, the conflict between the need to consider a long time frame and the decreasing reliability of forecasting far into the future may be substantial. In most cases, primary considerations in determining the time horizon of the analysis will be the time span of the physical effects that drive the benefits estimates, and capital investment cycles associated with environmental expenditures.

5.3.2 Compliance Rate Issues and Baseline Specification

One aspect of baseline specification that is particularly complex, and for which assumptions are typically necessary, is that of compliance rates. The treatment of compliance in the baseline scenario can significantly affect the results of the analysis. Therefore, it is important to be clear to persons using the analysis how assumptions about compliance behavior are incorporated into the analysis, and how sensitive the results are to the handling of compliance rates.

It can be challenging to clearly demonstrate the economic effects attributable to a new regulation or policy, while avoiding the potential for double-counting of benefits, costs, and impacts associated with separate existing regulations. To aid in preparation of the economic analysis and presentation of results, it is common to establish baseline conditions so that the affected regulated entities are in full compliance with other separate existing regulations. Assuming full compliance with existing regulations will enable the analysis to focus on the incremental economic effects of the new rule or policy, the results of which are used to evaluate the predicted economic changes. This information also meets the requirements contained in many of the statutes and administrative orders that use economic information as evidence that further steps need to be taken to address the effects on regulated parties (described in Chapters 2 and 9).

Defining the baseline in this fashion may pose some challenges to the analyst, since current observed or reported economic behavior may represent the consequences of either under-compliance or over-compliance with existing regulations. For example, it is possible to observe over-compliance by regulated entities with enforceable standards. One can find industries whose current effluent discharge concentrations for regulated pollutants are measured below concentrations legally required by existing effluent guideline regulations. On the other hand, evidence for under-compliance is evident in the resources devoted by EPA and other state and local regulatory agencies to enforce rules through orders, fines, and negotiated settlements. As a result, it will be important that the analysis separates the changes associated with a new regulation from actions taken to meet existing requirements. This is of particular importance if actions taken to meet existing requirements are coincident with, but not caused by, changes introduced by the new regulation.[3]

For some types of analyses it is sensible to establish a baseline of "current practice" (i.e., what is believed to be the actual degree of compliance, rather than assumed full compliance). For example, when a new action under review is intended to address or "fix-up" compliance problems associated with existing policies, information on current practices belongs in the baseline. Otherwise, defining the baseline in a manner that disregards this behavior will obscure the value of investigating whether further or alternative regulatory actions are necessary (e.g., as was the case in a review of banning lead from gasoline, which was precipitated in part by the noncompliance of consumers misfueling their non-leaded gasoline automobiles (EPA, 1985)). For a deregulatory rule (e.g., a rule designed to address potential changes in or clarify definitions of regulatory performance that frees entities from enforceable requirements contained in an existing rule), it may be sensible to perform the analysis using both a full compliance and "current practices" baseline. A full compliance scenario in this instance introduces some added complications to the analysis, but it may be important to report on the economic effects of failing to take the deregulatory action.

In cases of over-compliance with existing policies, or actions already taken in the economic interests of the affected parties, current practices can be used to define baseline conditions unless these practices are expected to change or are highly uncertain in ways that are directly associated with the rule being analyzed. For example, observed over-compliance by a regulated entity may be the result of choices it has made to anticipate forthcoming more stringent federal regulatory requirements. If there should be a decision not to follow through with the anticipated federal regulation, the analysis will need to establish whether the current observed over-compliance behavior by the regulated entity may be curtailed to meet existing (i.e., relatively less stringent) requirements. If the regulated entity in this example is expected to continue to over-comply despite the absence of the more stringent regulation, then the policy scenario should not contain the costs and benefits attributable to this behavior, and it is appropriate to account for them in the baseline scenario that describes the "world without the regulation." However, if the regulated entity will relax its pollution control practices to meet current requirements after the stricter regulation fails to emerge, then the costs and benefits of the over-compliance behavior should be attributed to the policy scenario. In these situations, it may be useful to consider performing the analysis with alternative baseline scenarios, and demonstrate the potential economic consequences of different assumptions associated with the expected changes in this type of behavior.

Analysts may also elect to incorporate predicted differences in compliance rates within policy options considered for new rules, in cases where compliance behavior is known to vary systematically with the regulatory options being considered (e.g., if the expected compliance rate with a rule may differ if entities are regulated using economic incentives as compared with prescribed control technologies).

Despite the above possible complexities, it is prudent for most analyses of regulations to develop baseline and policy scenarios that assume full compliance with existing and newly enacted regulations. One rationale for adopting these assumptions is that the analytic results will provide information on the unique role the action under consideration is expected to have on the economy, which may be required under the authorizing statute, or administrative laws and policies. As a practical matter, noncompliant behavior will need to be known, estimable, and occurring at rates that can affect the evaluation of policy options before totally rejecting assumptions of "full

[3] For example, assigning costs between an existing and new regulation could be further complicated, if, as a result of under-compliance with the existing regulation, the estimated "joint" cost of meeting both regulations differs from the summed marginal costs of first meeting the existing regulation, followed by implementing the new regulation. The same concern equally applies to the attribution of benefits and economic impacts to each regulation. Under these circumstances, the analyst should seek further directions provided by the authorizing legislation for the regulation, or instructions contained in other operative laws and policies.

compliance" for existing and new policies. In the end, assumptions on compliance behavior for current and new requirements should be clearly presented in the description of the analytic approach and assumption used for the analysis. Care should be taken to describe the importance of these assumptions when comparing regulatory options for which social costs and benefits, and economic impacts have been estimated.

5.3.3 Multiple Rules or Regulations and Baseline Specification

If conditions exist where there are no other relevant regulations, specifying a baseline is not complicated by questions of whether other regulations are being implemented and, if so, which regulations are responsible for environmental improvements and can "take credit" for reductions in risks. That is, there is no need to be concerned with which environmental improvements are in the baseline. Nor is it necessary to try to determine how these other regulations affect market conditions that directly influence the costs or the benefits associated with the policy of interest.

But actual conditions in the regulation of environmental risks are much more complex, and it is an unusual case where the above holds true. There are many regulatory agencies (i.e., federal, state, local) affecting environmental behavior, and several forms of consumer and industrial behavior are regulated by agencies whose agendas can overlap with EPA's (e.g., OSHA, DOT, DOE). Absent an orderly sequence of events that allows attributing changes in behavior to a unique regulatory source, in practice, there is no non-arbitrary way to allocate the costs and the benefits of a package of overlapping policies to each individual policy. Whether any one of these policies is "in the baseline" of the benefit-cost analysis of another policy is, to a large degree, a matter of choice. There is no theoretically correct order for conducting a sequential analysis of multiple overlapping policies that are promulgated simultaneously.

An idealized approach would attempt to analyze all of the policies together when assessing the total costs and benefits resulting from the package of policies. However, this kind of comprehensive analysis is usually not feasible. A practical alternative may be to consider the actual or statutory timing of the promulgation and/or implementation of the policies, and use this to establish a sequence with which to analyze related rules. But even when the temporal order of policies makes it clearer which policies are "in the baseline" and which are not, different depictions of the timing and impacts of pre-existing or overlapping policies can still have a substantial effect on the outcome of a benefit-cost analysis. An example of this, offered by Arnold (1995), concerns regulations designed to reduce the production of chlorofluorocarbons (CFCs) and other ozone-depleting substances. In this case, the impacts of multiple regulations on production decisions were not separable or independent of the order of their issuance, so that the costs and benefits of requirements estimated for each regulation were dependent on which preexisting rules were considered binding in the analysis. A similar illustration concerning hazardous waste regulations is also provided by Arnold (1995), wherein an assessment of the costs and benefits associated with several regulations is performed, demonstrating that the result of evaluating each individual regulation varies significantly depending on which of the other regulations are included in the baseline.

Therefore, the best practice is to be clear as to the baseline selected for the analysis, and to present a justification for making this choice. This can include providing information on the status of other regulatory actions that may have some effect on the baseline, and conducting sensitivity analyses that test for the implications of including or omitting other regulations. Some regulatory actions have attempted to directly link rules together that affect the same industrial category (e.g., the pulp and paper effluent guidelines and NESHAP rules (EPA, 1997)). While statutory and judicial deadlines may inhibit the linking of rules that fall on the same regulated entities (e.g., UMRA and RFA require analyses be performed for each rule), coordination between rulemaking groups is advocated in EPA's regulatory development process, and sharing of data, models, and joint decisions on analytic approaches is strongly recommended.

5.3.4 Summary

The specification of the baseline for an economic analysis can have a profound influence on the outcome of the analysis. The estimated costs and/or benefits of a proposed policy can change by an order of magnitude under different baseline assumptions. Careful thought in specifying the baseline is therefore crucial to a defensible analysis.

The first step is to be clear about the question being asked and therefore what baseline the analyst would like to specify. The second step is to characterize that baseline as well as possible within the constraints of the analysis. This involves determining which baseline parameters are most important to the analysis, assessing the advisability of expending resources to improve the estimates of those parameters, and making reasonable assumptions when necessary. In all cases, assumptions and uncertainties should be clearly stated as part of the analysis, along with a discussion of how alternative, plausible assumptions would be likely to affect the outcome of the analysis. Within the resources available, sensitivity analysis and uncertainty analysis are valuable tools for illustrating the potential impacts of assumptions made and quantifying, to the degree possible, the extent of the uncertainty underlying the specified baseline. Finally, the estimation of the costs and benefits attributable to individual policies in a package of policies is a problem for which there simply is no "correct answer."

Many factors will affect the configuration of the baseline in EPA's economic analyses. This means that even though analytical choices are well-constructed and logical, the consequences of these differences may frustrate efforts to attain comparability of baselines across different regulatory activities. Still, in any effort to evaluate regulatory options and assess benefits, costs, and economic impacts attributable to an individual rule, the analysis should be internally consistent in its definition and use of baseline assumptions. This is imperative when more than one baseline scenario is introduced, since this provides more possibilities to erroneously compare costs and benefits across different baselines. A decision to include multiple baselines into an analysis can result in a complex set of modeling choices, and an abundance of analytic results to interpret and communicate to decision makers. Therefore, analysts are advised to seek clear direction from management about baseline definitions early during the development of a rule.

5.4 Predicting Responses to a New Environmental Policy

It is impossible to measure an environmental policy's costs and benefits without a clear characterization of actions taken in response to the policy. Some policies are prescriptive in specifying what actions are required—for example, mandating the use of a specific type of pollution control equipment. It can be difficult, however, to predict responses to less-direct performance standards, such as bans on the production or use of certain products or processes, and market-based incentive programs. Analysts should make explicit all assumptions about responses, and should consider plausible alternative compliance options. Alternatively, when the number of conceivable options is essentially infinite, the analysis should at least span the range of possibilities. Cost-effectiveness analysis can often be used to identify and map out dominant regulatory options and responses. When it is not possible to characterize compliance responses with a high degree of certainty, the analysis should include a description of the likely direction of bias in the estimates—whether costs and benefits are over- or understated—if this is known.

Predicting responses starts with a comprehensive list of possible response options. These may include the use of different compliance technologies (if the technology is not specified by the policy itself) or waste management methods; changes in operations to avoid or reduce the need for new controls or the utilization of materials whose use is restricted by a policy (including various types of pollution prevention); shutting down a production line or plant to avoid the investments required to achieve compliance; or even noncompliance.[4] Typically, parties affected by a policy are assumed to choose the compliance option that minimizes their costs. In some cases, however, it may be reasonable to select a more costly option as the most likely response. Sometimes a higher

[4] As in the case of baseline specification, most analyses will assume full compliance by all entities that continue to operate. For some policies that present significant enforcement challenges, or for options that differ in ways that are likely to affect compliance rates, it may be useful to calculate how costs and benefits compare when using estimates of compliance rates less than 100 percent.

cost option may significantly reduce future legal liabilities, or achieve compliance with other rules being implemented at either the same time, or those expected to be promulgated in the future. However, the additional costs of compliance responses in excess of least-cost strategy costs should be attributed to these other causes.

Estimating responses is often the most difficult for pollution prevention policies because these options are generally more site- and process-specific than end-of-pipe control technologies. Predicting the costs and environmental effects of pollution prevention policies may require detailed information on industrial processes. As a result, the costs of a pollution prevention policy may be overstated and the benefits either over- or understated (depending on the nature of the process changes involved). Nevertheless, economic analyses should at least include qualitative discussion of potential pollution prevention responses and their effects on costs and benefits.

Predicting reductions in output (e.g., production line or plant closures) in response to a policy requires analysis of market characteristics that determine the allocation of cost increases among directly affected entities and their suppliers, customers, and competitors. This subject is discussed in the economic impact analysis section of Chapter 9.

5.5 Analyzing and Presenting Uncertainty

This section contains guidance on dealing with uncertainty in regulatory economic analyses, focusing on characterizing the precision of estimated economic outcomes such as net benefits. It provides specific recommendations for describing and presenting problems arising from uncertainty, and suggestions for carrying out sensitivity analyses.

This section concludes with a discussion of the welfare considerations related to risk and uncertainty.[5] These considerations are largely distinct from those associated with characterizing precision. The use of certainty equivalents for addressing these problems is addressed briefly, but detailed treatment is beyond the scope of this discussion.[6] Issues related to differences in risk perceptions and the provision of information are described, and the role of quasi-option values in decisions characterized by irreversible consequences is addressed briefly.

5.5.1 Guiding Principles for Uncertainty Analysis

Uncertainty is inherent in economic analyses, particularly those associated with environmental benefits for which there are no existing markets. The issue for the analyst is not how to avoid uncertainty, but how to account for it and present useful conclusions to those making policy decisions. Treatment of uncertainty, therefore, should be considered part of the communication process between analysts and policy makers.

Transparency and clarity of presentation are the guiding principles for assessing and describing uncertainty in economic analyses. Although the extent to which uncertainty is treated and presented will vary according to the specific needs of the economic analysis, some general minimum requirements apply to most economic analyses. In assessing and presenting uncertainty the analyst should, if feasible:

- present outcomes or conclusions based on expected or most plausible values;
- provide descriptions of all known key assumptions, biases, and omissions;
- perform sensitivity analysis on key assumptions; and
- justify the assumptions used in the sensitivity analysis.

The outcome of the initial assessment of uncertainty may be sufficient to support the policy decisions. If, however, the implications of uncertainty are not adequately captured in the initial assessment then a more sophisticated

[5] Stemming from definitions given in Knight (1921) economists have distinguished risk and uncertainty according to how well one can characterize the probabilities associated with potential outcomes. Risk applies to situations or circumstances in which a probability distribution is known or assumed, while uncertainty applies to cases where knowledge of probabilities is absent. Note that the economic definitions for these terms may differ from those used in other disciplines.

[6] Several other issues associated with uncertainty are also beyond the scope of this brief discussion, including verification, validation, and plausibility checks. Analysts will need to consult other sources for additional information on these topics.

analysis should be undertaken. The need for additional analysis should be clearly stated, along with a description of the other methods used for assessing uncertainty. These methods include decision trees, Delphi-type methods[7], and meta-analysis. Probabilistic methods, including Monte Carlo analysis, can be particularly useful because they explicitly characterize analytical uncertainty and variability. However, these methods can be difficult to implement, often requiring more data than are available to the analyst.[8]

Confidence intervals are generally useful to describe the uncertainty associated with particular variables. When data are available to estimate confidence intervals they can serve to characterize the precision of estimates and to bound the values used in sensitivity analysis.

5.5.2 Performing Sensitivity Analysis

Most analytical base cases, or primary analyses, generally do not address uncertainty and present expected or most plausible outcomes. Regardless of the basis for the primary analysis, point estimates alone do not provide policy makers with information about the full range of potential outcomes. Additional information is needed if the decision-maker is to have a more complete view of the potential impacts of the policy alternatives. It is always useful to see how net benefit estimates or other outputs of the economic analysis change with assumptions about input parameters. Sensitivity analysis provides a systematic method for making these determinations. Keeping in mind some basic principles can enhance sensitivity analysis.

- **Focus on key variables.** For most applied economic analyses, a full sensitivity analysis that includes every variable is not feasible. Instead the analyst must limit the sensitivity analysis to those input parameters that are considered to be key or particularly important. In determining which parameters are key, the analyst should carefully consider both the range of possible values for input parameters and each one's functional relationship to the output of analysis. The analyst should specify a plausible range of values for each key variable, including the rationale for the range of values tested.

- **Present the results clearly.** Results of the sensitivity analysis should be presented clearly and accompanied with descriptive text. The most common approach to this sort of partial sensitivity analysis is to estimate the change in net benefits (for a benefit-cost analysis) or other economic outcome while varying a single parameter, leaving other parameters at their base value. A more complete analysis will present the marginal changes in the economic outcome as the input parameter takes on progressively higher or lower values. Varying two parameters simultaneously can often provide a richer picture of the implications of base values and the robustness of the analysis. Analysts should consider using graphs to present these combined sensitivity analyses by plotting one parameter on the x-axis, the economic outcome on the y-axis, and treating the second parameter as a shift variable.[9]

- **Identify switch points.** "Switch point" values for key input parameters can be very informative, especially in benefit-cost analyses. Switch points are defined as those conditions at which the recommended policy decision changes (e.g., when the estimation of net benefits changes sign). While switch points are not tests of confidence in the statistical sense, they can help provide decision-makers with an understanding of how robust the analysis is.

- **Assess the need for more detailed analysis.** Finally, sensitivity analyses can also be useful as a screening device to determine where more extensive treatment of uncertainty may be needed. In some cases the plausible range of values for the parameter may be narrowed with further research or data gathering, or the analyst may be able to better characterize the parameter's uncertainty. If several parameters

[7] There a number of such techniques, but all of these methods focus on the use of eliciting and combining expert judgment to inform analysis. See Chapter 7 of Morgan and Henrion (1990) for more detail on the use of these methods.

[8] Morgan and Henrion (1990) is a useful general reference that includes descriptions of many methods to assess uncertainty.

[9] When the analysis contains many highly uncertain variables, presentation may be facilitated by noting the uncertainty of each in footnotes and carrying through the central analysis using best point estimates.

appear to have a large impact on the results of the analysis then a more sophisticated treatment of uncertainty may be necessary.

5.5.3 Welfare Considerations Related to Uncertainty and Risk

So far this discussion has focused upon uncertainty as it must be accommodated by the analyst charged with performing an economic assessment and the decision-maker who receives this information. A separate but related issue is how individuals affected by environmental policies respond to uncertainty in outcomes and imperfect information. These responses may have an impact on how individuals respond to policy alternatives and how they value policy outcomes. Some of these considerations are noted here, but this treatment is not detailed or exhaustive. It is important to note that analytical precision and welfare effects are distinct concepts. Certainty equivalents, for example, address welfare effects and are appropriate for assessing efficiency in a benefit-cost analysis, but they do not assess analytical precision or mitigate the usefulness of sensitivity analyses and bounding cases.

- **Risk attitudes and certainty equivalents:** Individuals and other entities are generally not neutral when faced with situations of uncertainty or risk. In most cases related to environment and health they are considered to be *risk averse*, favoring a certain outcome to one that is uncertain even if the expected value of the risky outcome is equal to the value of the certain one. The theoretically preferred manner of incorporating risk attitudes is to use *certainty equivalents*, sometimes termed certain monetary equivalents. Certainty equivalents are defined as the minimum amount that an individual would be willing to accept with certainty instead of facing the uncertain outcomes.[10]

  While certainty equivalents have theoretic appeal, they are difficult to put into practice for economic analyses of environmental policies. Estimation of certainty equivalents requires detailed knowledge of (or assumptions about) risk preferences, and analysts are unlikely to have these data. To estimate certainty equivalents one must also be able to assign probabilities to the set of potential outcomes. It is often very difficult or impossible to make these assignments.

- **Lay and expert risk perceptions:** Lay perceptions of risk may differ significantly from scientific assessments of the same risk, and an extensive literature has developed on the topic.[11] Because individuals respond according to their own risk perceptions, it is important for the analyst to be attentive to situations where there is an obvious divergence in these two measures. In such cases, analysts should consider evaluating policy options under both sets of information, clearly stating the basis for economic value estimates used or developed in their analysis. Because providing information to the public may reduce differences between lay and expert perceptions of risk, and may allay public concerns, analysts should consider including these strategies in their analysis of potential policy options.

- **Provision of information:** Some policy actions focus on providing information on risks to health and welfare. Inasmuch as this information allows consumers to make better decisions regarding their households' welfare there is an economic benefit to providing this information. Revealed preference benefit analyses, however, can make new information appear to have a net negative effect on household welfare because households may undertake new (and costly) activities in response. An appropriate framework for evaluating the benefits of information provision under these circumstances is to assess the costs of sub-optimal household decisions under the less-complete information.[12] Analysts should carefully consider these issues when they evaluate policies that focus on information provision.

[10] Some researchers have suggested risk-adjusted discount rates as an alternative device for incorporating information on the uncertainty future benefits and costs. Most economists now conclude, however, that the discount rate should *not* be adjusted to account for uncertainty in benefit-cost analysis.

[11] Useful general sources include Slovic (1987) and Fischhoff et al. (1978).

[12] Foster and Just (1989) describes this approach more fully, demonstrating that compensating surplus is an appropriate measure of willingness-to-pay under these conditions. The authors illustrate this with an empirical application to food safety.

- **Quasi-option value:** Another relevant issue in decision-making under uncertainty is that of quasi-option value as identified by Arrow and Fisher (1974). Some environmental policies involve irreversible decisions that must be made in the face of uncertainty. If information that reduces this uncertainty can be expected to develop over time, then there is a positive "quasi-option" value to waiting until this information is available. In this case, the value originates from possessing the option to hold off on making the decision until uncertainties are resolved. Generally, it is difficult to quantitatively include quasi-option values in an economic analysis of environmental policy, but the concept is useful and may be highlighted qualitatively if circumstances warrant. For more on this issue see Freeman (1984, 1993), Fisher and Hanemann (1987), and Cochrane and Cutler (1990).

5.6 Emerging Cross-Cutting Issues

Many other cross-cutting issues are not detailed in the *EA Guidelines*. Some of these issues are difficult or impossible to incorporate fully into economic analyses at this time, but may become either more important or more tractable as the economic literature develops. Although the relevance of these considerations depends on the specifics of the policy being considered, analysts may want to at least consider these issues qualitatively. Three emerging issues are identified here: tax interaction effects, the pace of exogenous technological change, and the effects of regulation on innovation.

- **Tax Interaction effects:** Although evaluations of environmental policies typically assume a first-best regulatory setting, preexisting taxes such as those on labor income create a second-best setting. This difference can affect the estimated costs of policy actions. Recent advances in applied general equilibrium analysis have led to generally replicable qualitative results. These studies indicate that ignoring these effects may result in underestimating the cost of compliance, from a social perspective.[13] However, the magnitude of the effect—and perhaps the direction—will vary across policies. Although the analytical emphasis on this issue has been on estimating costs, benefits analysis can conceivably suffer a similar bias.

- **Pace of exogenous technological change:** Economic analysis of environmental policies may be affected by the pace of exogenous technological change. In principal, accounting for this can either increase or decrease marginal and total abatement costs, depending on the direction of change. Generally, however, the expectation is that accounting for exogenous technological change would decrease estimated abatement costs. Recent analyses have indicated that even for mature technologies the magnitude of this effect can be large.[14]

- **Regulation and innovation:** More extensive research is being developed to examine the impact of various regulatory approaches on firms' research and development decisions for abatement technology.[15] As suggested in the descriptions of alternative regulatory approaches in Chapter 4, this impact may be positive or negative depending on the regulatory approach and setting. Generally, economists expect that incentive-based instruments will provide greater incentive for cost-reducing innovations than will command and control regulatory approaches. Policies that provide information to firms and consumers may also affect technological innovation. Chapter 8 provides some additional information on the subject of regulation, innovation, and the implications for estimating social costs.

[13] For an example see Bovenberg and Goulder (1997).

[14] See, for example, Ellerman and Montero (1998) and Carlson et al (1998).

[15] See, for example, Milliman and Prince (1989) and Biglaiser and Horowitz (1995).

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Chapter 6: Analysis of Social Discounting

6.1 Introduction

The costs and benefits of many environmental policies are frequently paid and received at different points over the course of sometimes long time horizons. As a result, benefit-cost and related analyses that are key components of EPA's policy development and evaluation process must describe future effects in terms that help present day policy makers choose appropriate approaches for environmental protection.

One common method for doing so is called discounting, which is the process whereby the values of future effects are adjusted to render them comparable to the values placed on current consumption, costs, and benefits, reflecting the fact that a given amount of future consumption is worth less than the same amount of consumption today. Time discounting is accomplished by multiplying the future values of a policy's effects by discount factors that reflect both the amount of time between the present and the point at which these events occur and the degree to which current consumption is more highly valued than future consumption.

Despite the relative simplicity of the discounting concept, choosing a discount rate has been one of the most contentious and controversial aspects of EPA's economic analyses of environmental policies. While there are several plausible explanations for why discounting in environmental policy evaluation has been unsettled for many years, the most important is that the theoretical and applied economics literature on discounting from a social perspective is voluminous and technically complex. This makes it difficult to distill precise advice on appropriate discounting procedures for policy analysis. Moreover, in some cases the economics literature by itself does not yield simple and robust discounting rules for practical applications because making such important social decisions requires inputs from disciplines other than economics.

Nonetheless, it is important to consider the uncertainties surrounding social discounting in the broader context of applied economic analysis. Benefit-cost analysis is not a precise tool that yields firm numerical results, rather, it is a general framework for more carefully accounting for the potential and varied effects of government programs. Some of these effects can be quantified, whereas others can only be assessed qualitatively. Some may be relatively certain, whereas others may be quite speculative.

The imprecision connected with assessing benefits and costs suggests that the controversy surrounding the discount rate, in many circumstances, may have more theoretical than practical significance. For example, the effects on net benefits of alternative assumptions made for measuring and valuing uncertain effects of environmental policies can overwhelm the effects of changes in the discount rate. Additionally, for some government projects, benefits and costs may have similar time profiles, or benefits may so outweigh costs (or vice versa), that changes in the discount rate will not influence the policy implications of the analysis.

This review of the basics of social discounting begins in Section 6.2 with a discussion of some general considerations in social discounting. In Section 6.3, various discounting procedures for environmental policy assessment are presented and evaluated. This detailed discussion is divided into social discounting as applied in intra-generational contexts, where very long time horizons involving multiple generations do not apply, and discounting for inter-generational

circumstances involving long time horizons and unborn generations. EPA guidance for intra-generational social discounting is presented in Section 6.3.1.5. EPA guidance on inter-generational social discounting follows in Section 6.3.2.4. Finally, discounting and related procedures for situations in which some effects are not monetized are addressed in Section 6.4.

6.2 General Considerations in Social Discounting

This section reviews a few basic concepts and considerations central to understanding the role and importance of discounting in public policy evaluation. The focus is mainly on describing social discounting and on distinguishing discounting per se from other aspects of measuring and summarizing the costs, benefits, impacts, and other consequences of environmental policies. It also discusses the circumstances in which discounting has a large impact on the net social benefits of an environmental policy.

6.2.1 Social and Private Discounting

Discounting in public policy evaluation is normally referred to as *social discounting* or *discounting using the social rate of interest.* The process itself—applying discount factors to future flows of costs, benefits, and other consequences of environmental and other policies—is mechanically the same as the discounting process in private individuals' economic and financial calculations. What makes it "social" discounting is that it is being applied in the context of evaluating a policy's effects from an overall social perspective. Clearly, private and social perspectives can yield very different conclusions concerning, for example, the cost of engaging in an activity that also generates environmental harms.

Whether social discounting also departs significantly from private discounting, however, is less clear. Some approaches to social discounting suggest that the procedures and rates should be the same as those used in private sector discounting. Other perspectives, however, suggest that social discounting is a very different process than a single individual's discounting. In any event, at a minimum, the term "social discounting" refers to the broad society-as-a-whole point of view embodied in benefit-cost and other analyses of public policies. Whether it also connotes procedures and rates different from private discounting is a central question explored in this chapter.

6.2.2 Methods for Summarizing Present and Future Costs and Benefits

Most applications of social discounting in environmental policy evaluation involve translating future values into present ones. The conceptual foundation of discounting is based on the fact that present consumption is valued differently from future consumption. Discounting renders costs and benefits that occur in different time periods comparable by stating them all in present day terms. The resulting net present value is at least one measure of social value that might be used in evaluating environmental policies.

6.2.2.1 Net Present Value

In formal terms, the net present value of a projected stream of current and future benefits and costs is found by multiplying the benefits and costs in each year by a time-dependent weight, d_t, and adding all of the weighted values as follows:

$$NPV = NB_0 + d_1 NB_1 + d_2 NB_2 + \dots + d_n NB_n$$

NB_t is the net difference between benefits and costs (B_t-C_t) that accrue at the end of period, t, and the discounting weights are given by:

$$d_t = 1/(1+r)^t$$

where r is the discount rate and n is the final period in the future in which the policy's effects are felt.

To account for inflation, either real or nominal values may be used, as long as they are used consistently. In other words, nominal costs and benefits require nominal discount rates, and real costs and benefits require real discount rates. Moreover, consistent decision making requires that the same discount rate be used for both benefits and costs. Otherwise, any policy can be justified by choosing a sufficiently low discount rate for benefits, by choosing a sufficiently high discount rates for costs, or by choosing a sufficiently long time horizon.

It is important to be explicit about how time periods are designated and when, within each time period, costs and benefits accrue. Typically, time periods are years, but alternative time periods may prove desirable or necessary if costs or benefits accrue at irregular or non-annual intervals. The preceding formula assumes that $t=0$ designates the beginning of the first period. Therefore, C_0 represents startup costs such as capital costs that occur immediately upon implementation of the regulation. The formula further assumes that no additional costs are incurred until the end of the first year of regulatory compliance.[1] Benefits, if any, also accrue at the end of each time period. Therefore, the following diagram illustrates how net benefits (measured in dollars) are distributed over time.

TIME ➞

Year t	0	1	2	3	4 ...	n
\$	NB_0	NB_1	NB_2	NB_3	NB_4 ...	NB_n

6.2.2.2 Annualized Values

In addition to net present value, there are other procedures for rendering costs and benefits that occur in more than one time period comparable. One method is to annualize the costs and benefits over the duration of the policy. For example, in the absence of discount rates ($r=0$), a regulation that costs \$100,000 at the end of the first year, \$200,000 at the end of the second year, and \$300,000 at the end of the third year can be said to cost \$200,000 a year in annualized costs over the three year period. Comparing annualized costs to annualized benefits is equivalent to comparing the present values of costs and benefits.

Costs and benefits each may be annualized separately by using a two-step procedure. To annualize the costs, for example, the present value of costs is calculated using the NPV formula in Section 6.2.2.1, except that the stream of costs alone, not the net benefits, is used in the calculation. This present value is then annualized (as in calculating mortgage payments) according to the following formula:

$$AC = PVC \times \frac{r \times (1+r)^n}{(1+r)^n - 1}$$

where,

AC = annualized cost accrued at the end of each of n periods;

PVC = present value of costs;

r = the discount rate per period; and

n = the duration of the policy.

Note that the annualized cost is the amount one would have to pay at the end of each period t to add up to the same cost in present value terms as the stream of costs being annualized. There is no initial cost at $t=0$ in this annualization. Such an initial cost can be incorporated into the annualization using the slightly different formula:

$$AC = PVC \times \frac{r \times (1+r)^n}{(1+r)^{(n+1)} - 1}$$

This approach is also useful when analyzing non-monetized benefits, such as reductions in emissions or reductions in health risks, *when benefits are constant over time*. The average cost-effectiveness of a policy can be calculated by dividing the annualized cost by the annual benefit to produce measures of program effectiveness, such as the cost per ton of emissions avoided.

6.2.2.3 Net Future Value

Finally, there is yet another way of rendering costs and benefits that occur in more than one time period comparable. Instead of discounting all future values back to the present, it is possible to accumulate them forward to some future time period—for example, to the end of the last year of the policy's effects, n. Here, the net benefit test is whether the accumulated net future value (NFV) is positive.

$$NFV = d_0NB_0 + d_1NB_1 + d_2NB_2 + ... + d_{n-1}NB_{n-1} + NB_n$$

[1] See EPA (1995) for an example in which operating and monitoring costs were assumed to be spread out evenly throughout each year of compliance. While the exponential function above is the most accurate way of modeling the relationship between present value and a continuous stream of benefits and costs, simple adjustments to the equations above can sometimes adapt them for use under alternative assumptions about how dollar flows are distributed over time.

NB_t is the net difference between benefits and costs (B_t-C_t) that accrue in year t and the accumulation weights, d_t, are given by

$$d_t = (1+r)^{(n-t)}$$

where r is the discount rate.

Each of these methods employs an interest rate to translate values through or across time, so the methods are not really different ways to determine the benefits and costs of a policy. Instead, they are different ways to express and compare costs and benefits that occur in multiple time periods on a consistent basis. Discounting places all costs and benefits in the present time period, annualization spreads them smoothly through time, and accumulation states them all in the future. But each procedure uses the same discount (interest) rate, so they are different ways to describe the same underlying phenomenon.

Depending on the circumstances, one method might have advantages over the others. For example, annualizing the costs of two machines with different service lifetimes might reveal that the one with the higher total cost actually has a lower annual cost because of its longer lifetime. Similarly, discounting to the present is likely to be the most informative procedure when analyzing a policy that requires an immediate investment and offers a stream of highly variable future benefits.

In general, however, these are alternative ways of translating costs and benefits through time using an interest rate. Therefore, the analysis, discussion, and conclusions presented in this chapter apply to all methods of translating costs, benefits, and effects through time, even though the focus is mostly on discounting.

6.2.3 Sensitivity of Present Value Estimates to the Discount Rate

The impact of discounting streams of costs and benefits in public policy evaluation is sometimes large and sometimes not, depending on the circumstances. When all effects occur in the same period, discounting may be unnecessary or superfluous: net benefits are positive or negative regardless of the discount rate used or the procedure for translating them through time. Similarly, when costs and benefits of a policy are largely constant over the relevant time frame, discounting costs and benefits will produce the same conclusion concerning the policy as would examination of a single year's costs and benefits. Of course, higher discount rates will reduce the present value of any future cost or benefit. But if costs and benefits of a policy occur simultaneously and if their relative values do not change over time, whether the net present value of such a policy is positive does not depend on the discount rate.

Discounting can substantially affect the present value net benefits estimates for public policies when there is a significant difference in the timing of costs and benefits. For example, if the costs of a policy are incurred today, they are not discounted at all. But if the benefits will occur 30 years from now, the present value of the benefits, and, hence, the net present value of the policy's effects, depends critically on the discount rate used.

Suppose the cost of some environmental policy that is incurred entirely in the present is $1 billion and that after 30 years a benefit results that is estimated to be worth $5 billion in the future. Without discounting, a policy that offers benefits five times its cost appears to be a very worthwhile social investment. Discounting the $5 billion future benefits, however, can radically alter the economic assessment of the net present value of the policy. Five billion dollars, 30 years in the future, discounted at one percent is $3.71 billion, at three percent it is worth $2.06 billion, at seven percent it is worth $657 million, and at 10 percent it is worth only $287 million. In this case, the range of discount rates generates over an order of magnitude difference in the present value of benefits. And, longer time horizons will produce even more dramatic effects on a policy's net present value. Hence, the choice of the discount rate largely determines whether this policy is considered, at least on economic efficiency grounds, to offer society positive or negative net benefits.

Thus, for government projects and policies that require large initial outlays or that have long delays before benefits are realized, the selection of the discount rate can be a major factor in determining whether the net present value is positive. Many of EPA's policies fit these profiles. Large investments by public or private parties are usually required early on, whereas the benefits of those investments either accrue for many years thereafter, such as improvements in health and environmental quality, or will not begin for many years, such as reductions in the contamination of environmental systems from hazardous waste, landfill facilities, and the protection of the earth's atmosphere and climate.

6.2.4 Distinguishing Discounting from Other Procedures

Discounting is only one of several components that are necessary in order to produce comparable estimates of a policy's costs and benefits that accrue over more than one time period. Discounting is a technique for translating values from one time period to another in order to express the values of a policy's consequences in consistent terms. It is not, however, a method for actually determining the future values of future costs and benefits. Two considerations related to determining these future values—projecting future values based on present ones and accounting for risk—are closely related to discounting.

6.2.4.1 Future Values of Costs and Benefits

The future value of one of an environmental policy's effects may hinge critically on the assumed rate of growth of wealth over time. There may also be a connection between increasing wealth and the discount rate for expressing future values in present day terms. Nevertheless, the process of determining the values of future costs and benefits and then translating them into present terms are two conceptually distinct procedures.

It is generally appropriate to conduct each of these tasks separately. And, it is prudent to avoid attempting to "correct" for errors in one procedure by "adjusting" the other. For example, it is technically possible to use a current valuation for a future benefit whose future value is expected to increase, but then reduce the discount rate to reflect that assumed rising valuation through time. Nevertheless, this is usually unwise because the values of other consequences of a policy might not follow the same rate of increase over time. Thus, these might be over- or under-corrected by the adjusted discount rate. The only way to avoid that result would be to use a different adjusted discount rate for each cost and benefit stream, which is generally inappropriate.

6.2.4.2 Risk and the Social Discount Rate

The relationship between risk and the rate of return on assets has been an important subject in modern finance. Risk considerations also have played a role in the controversy surrounding the selection of an appropriate discount rate for benefit-cost analysis. For example, one recommendation is that public projects with risky or uncertain future costs and benefits should be discounted at a higher rate reflecting those risks, just as it is in the private sector.

The concept of risk is often interpreted narrowly as being measured by the variability or range of possible outcomes of a project. Greater variation implies more risk according to this view. But the notion of risk should be conceptualized more broadly. Rather than being taken in isolation, the risk of a project is measured by its effect on the variability in outcomes of the entire portfolio of assets. In general, the degree of risk associated with an asset is measured in terms of the covariance of its returns with those of the portfolio of assets to which it is added.[2] When viewed from this broader perspective, most environmental projects are either riskless or reduce risk. This is because most environmental projects have benefits and costs that are widely dispersed and that are uncorrelated or negatively correlated with future measured income and other aspects of economic welfare.

Nevertheless, the costs and benefits of some environmental policies can be risky in this broader sense. In these cases, it is commonly argued that the discount rate should be adjusted upward by a risk premium to value future uncertain returns. However, this is generally not the correct procedure because it requires the discount rate to reflect both the risk of future returns as well as the length of time until they materialize. That is, if the goal is to reduce the present value of a project's returns to reflect their risk, the same decrease in present value will be

[2] An assumption underlying this analysis is that the asset being acquired is a very small fraction of the portfolio of assets already held. If this assumption is violated, the variability in returns of the asset can directly affect the variability of returns to the entire portfolio. The potential costs and benefits of environmental policies generally are spread among large numbers of people, however, which satisfies the condition that the asset acquired be a small portion of the portfolio of assets already held.

produced by a smaller increase in the discount rate the longer the delay until the returns are received.[3]

Economic theory suggests using two different "instruments" to accomplish the two different goals. One such procedure to account for risk is to value a project's uncertain returns using the *certain monetary equivalent* or *certainty equivalent*. As discussed in Chapter 5 of this document, this is the amount risk averse individuals would be willing to pay with certainty for the risky prospect. The certainty equivalent should then be discounted using the rate of interest individuals use to discount other perfectly certain flows.

Hence, to properly account for risk in benefit-cost analyses, the first step should be to evaluate whether a project is actually risky from the broader perspective of society's larger portfolio of assets. Many government policies are not risky at all, so that their expected values of costs and benefits can be discounted directly using a risk-free interest rate. For projects that offer truly risky prospects, however, certain monetary equivalents for these returns should be derived and then discounted to the present using a risk-free rate. The discount rate should not be "adjusted" to account for risky costs and benefits.

6.3 Approaches to Social Discounting

This review of the basics of social discounting and the new EPA guidance on the subject begins with discounting in conventional or *intra-generational* contexts, where very long time horizons involving multiple generations do not apply. Next, approaches for *inter-generational* social discounting, involving very long time horizons and unborn generations, are presented and evaluated.

The main purpose of this discussion is to provide a broad overview of the extensive literature on social discounting in order to distill from it practical guidelines for environmental policy evaluation. It is not, however, a detailed review of the literature on discounting in public project evaluation, which is vast in scope and volume. Excellent sources for summaries of the social discounting literature are Lind (1982a), Lind (1982b), Lind (1990), Lind (1994), Lyon (1990), Lyon (1994), Kolb and Scheraga (1990), Scheraga (1990), IPCC (1996), Pearce and Turner (1990), and Pearce and Ulph (1994).

6.3.1 Intra-Generational Social Discounting

This section explores social discounting in conventional or intra-generational contexts, specifically those in which very-long-time-horizon issues are not important features. Most of the traditional discounting literature focuses on these circumstances. Intra-generational contexts may well have decades-long time frames, but they do not explicitly confront the extremely long time horizons and impacts on unborn generations that are central to the extensions of social discounting research into climate change, nuclear waste disposal, and other such policy issues. The division of the problem into intra-generational and inter-generational social discounting helps to understand the substantially different contribution economic approaches can offer in each area.

The discussion begins with a brief review of the analytical foundations of conventional social discounting. It next outlines the major social discounting approaches suggested in the literature. The section concludes with a review of the concrete conclusions and advice offered by the traditional discounting literature and the new guidance developed by EPA for use in social discounting in intra-generational contexts.

6.3.1.1 Analytical Foundation of Intra-Generational Social Discounting

Conventional social discounting is rooted firmly in the view that the government is acting on behalf of its citizens in undertaking public projects and promulgating environmental and other policies. Therefore, benefit-cost analysis of these actions should seek to estimate the costs and benefits experienced by all of the affected parties, and in so

[3] Note that if discount rates are adjusted to incorporate risk, the adjustment is not always upward. Risky benefits are worth less than their expected value, so an upward adjustment of the discount rate will reduce the present value. But uncertain costs would require a downward adjustment of the discount rate to increase the present value to reflect the fact that risk averse individuals would pay more than the expected value of the costs to avoid bearing the uncertain prospect.

doing determine whether, in aggregate, the gainers under a policy would be able to compensate the losers.

This foundation for social discounting has an important implication for the choice of a social discounting method. Just as consumer sovereignty dictates that the government should incorporate the specific values that particular individuals place on outcomes that affect them in assessing its actions, the government should also discount future costs and benefits in the same way that the affected individuals do. Strict adherence to the principles of consumer sovereignty is necessary in order to determine how much each person would agree he or she is made better or worse off by a given policy in present value terms.

The analytical and ethical foundation of the intra-generational social discounting literature thus rests on the traditional test of a "potential" Pareto improvement in social welfare—whether gainers could compensate the losers. This framework fundamentally casts the consequences of government policies in terms of collections of individuals contemplating changes in their own consumption (broadly defined) over time. Thus, social discounting in this context should seek to mimic the discounting practices of the affected individuals.

The Paretian economics point of view, however, is not the only ethical perspective possible in this context. As discussed in the section on inter-generational discounting, another approach is to cast the problem in terms of maximizing a social welfare function that includes utilities of present and future individuals and is maximized according to an alternative set of objectives and constraints. While alternative social welfare functions could apply to intra-generational circumstances, it is generally confined to inter-generational contexts. Hence, although there is nothing inherent in a short-time-horizon policy that dictates that only the Paretian perspective is appropriate for intra-generational situations, this is the most commonly accepted point of departure in the social discounting literature for these circumstances. It is also worth considering the two very distinct foundations for social discounting separately because their implications for determining the social discount rate are quite different. The Paretian economic approach suggests that the social discount rate is to be found by examining the preferences of affected parties, while the discount rate under alternative social welfare functions is not necessarily based on the preferences of existing individuals.[4]

6.3.1.2 Fundamental Procedures for Intra-Generational Social Discounting

Given the reasonably precise and circumscribed objective of social discounting as described above, the volume of literature on the topic is surprisingly diverse and complex. This section briefly reviews the major approaches suggested in the literature and evaluates their implications for practical social discounting in environmental policy assessments. The section concludes with a summary of recommended practices for social discounting in intra-generational contexts.

Consumption Rate of Interest Approach for Social Discounting

The economic literature begins by pointing out that under a variety of restrictive assumptions—no taxes, no risk, perfect capital markets—the task of discounting effects experienced by individuals would be straightforward. Analysts should simply use the observable market rate of interest that underlies the intertemporal consumption allocation decisions of those same individuals. The rate at which individuals are willing to exchange consumption over time is normally referred to as the "consumption rate of interest."

The simplifying assumptions (especially the absence of taxes on investment returns) imply that the consumption rate of interest equals the market interest rate, which also equals the rate of return on private sector investments. In this case, individuals discount future consumption at the market rate of interest, which is also the rate at which consumption can be translated through time via private sector investments. Hence, if the government seeks to value costs and benefits in present day terms in the same way as the affected individuals, it also should discount using the market rate of interest.

One of the simplifying assumptions underlying this result—that the consumption rate of investment at which consumers discount future consumption equals the rate of

[4] This concept is also treated in Chapter 10 in the discussion of ways to jointly consider efficiency and equity within a single analytical framework.

return on private sector investment—probably does not hold in practice. Taxes on private sector investment returns can cause the *social rate* at which consumption can be traded through time (the pre-tax rate of return) to exceed the rate at which *individuals* can trade consumption over time (the post-tax consumption rate of interest).

For example, suppose the market rate of interest, net of inflation, is five percent, and that taxes on capital income amount to 40 percent of the net return. In this case, private investments will yield five percent, of which two percent is paid in taxes to the government, with individuals receiving the remaining three percent. From a social perspective, consumption can be traded from the present to the future at a rate of five percent. But individuals effectively trade consumption through time at a rate of three percent because they owe taxes on investment earnings. As a result, the consumption rate of interest is three percent, which is substantially less than the five percent social rate of return on private sector investments (also known as the social opportunity cost of private capital).

Over several decades, a very large body of economic literature developed, analyzing the implications for social discounting of divergences between the consumption rate of interest and the social rate of return on private sector investment. The dominant approaches in this literature are briefly outlined here.

Consumption Rate of Interest-Shadow Price of Capital: The Traditional View

One approach that enjoys widespread support among economists recommends that social discounting in intragenerational contexts should use the consumption rate of interest to discount future costs and benefits that have been valued in terms of future consumption. Intuitively, this procedure makes sense because the government is assumed to be valuing future consequences of its policies just as the affected citizens would. If individuals discount future consumption (and the costs and benefits of a public policy) using the consumption rate of interest, then so should the government. So, the social rate of discount should equal the consumption rate of interest.

But, if the costs of financing a public project or the costs of regulatory compliance displace private investments, society loses the total pre-tax returns from those foregone investments. Private capital investments might be displaced if, for example, public projects are financed with government debt and the supply of investment capital is relatively fixed. This is the "closed economy" condition. In this case, discounting costs and benefits using the consumption rate of interest (the post-tax rate of interest) does not seem to capture the fact that society loses the higher, social (pre-tax) rate of return on foregone investments.

Under the consumption rate of interest-shadow price of capital approach for social discounting, the social value of displacing private capital investments is taken into account prior to discounting. Under this approach, when a public project displaces private sector investments, the correct method for measuring the social costs and benefits requires an adjustment of the estimated costs (and perhaps benefits as well) prior to discounting using the consumption rate of interest. This adjustment factor is referred to as the "shadow price of capital."[5]

The shadow (social) price (value) of private capital is intended to capture the fact that a unit of private capital produces a stream of social returns at a rate greater than the rate at which they are discounted by individuals. If the social rate of discount is the consumption rate of interest, then the social value of a \$1 private sector investment will be greater than \$1. The investment produces a rate of return for its owners equal to the post-tax consumption rate of interest, plus a stream of tax revenues (considered to be consumption) for the government.

To illustrate this simply, suppose that the consumption rate of interest is three percent, that the pre-tax rate of return on private investments is five percent, that the net-of-tax earnings from these investments are consumed in each period, and that the investment exists in perpetuity (amortization payments from the gross returns of the investment are devoted to preserving the value of the capital intact). A \$1 private investment with these characteristics will produce a stream of private consumption of \$0.03 per year and tax revenues of \$0.02 per year. Discounting the private post-tax stream of consumption at the three percent consumption rate of interest yields a present value of \$1. Discounting the stream of tax revenues at the same rate yields a present value of about \$0.67. The social value

[5] Lind (1982a) remains the seminal source for this approach in the social discounting literature.

of this $1 private investment—the shadow price of capital—is thus $1.67, substantially greater than the $1 private value that individuals place on it.

Therefore, if financing a public project displaces private investments, this "consumption rate of interest-shadow price of capital" approach suggests adjusting the project's costs upward by the shadow price of capital and then discounting all costs and benefits using a social rate of discount equal to the consumption rate of interest. To apply this approach, the first step is to determine whether private investment flows will be altered by a policy. Typically, project costs are thought to displace private capital, at least in part, although project benefits could encourage additional private sector investments. Next, all of the altered private investment flows (positive and negative) are multiplied by the shadow price of capital to convert them into consumption-equivalent units. All flows of consumption and consumption-equivalents are then discounted using the consumption rate of interest.

A simple example of this method is as follows. Suppose the pre-tax rate of return from private investments is five percent and the post-tax rate is three percent, with the difference attributable to taxation of capital income. Assume as well that increases in government debt displace private investments dollar-for-dollar, and that increased taxes reduce individuals' current consumption also on a one-for-one basis.[6] Finally, assume that the $1 current cost of a public project is financed 75 percent with government debt and 25 percent with current taxes and that this project produces a benefit 40 years from now that is estimated to be worth $5 in the future.

Using the consumption rate of interest-shadow price of capital approach, first multiply 75 percent of the $1 current cost (which is the amount of displaced private investment) by the shadow price of capital (assume this is the $1.67 figure from above). This yields $1.2525, to which is added the $0.25 amount by which the project's costs displace current consumption. The total social cost is therefore $1.5025. This results in a net social present value of about $0.03, which is the present value of the future $5 benefit discounted at the three percent consumption rate of interest ($1.5328) minus the $1.5025 social cost.

Thus, under the consumption rate of interest-shadow price of capital approach, costs are adjusted upward to reflect the higher social costs of displacing private investments, but discounting for time itself is accomplished using the consumption rate of interest—consistent with how individuals trade and value consumption over time.

Variants of this approach exist. For example, the Kolb-Scheraga (1988) approach recommends annualizing capital expenditures using the pre-tax rate and then discounting all cost and benefits using the consumption rate of interest.

Other Social Discounting Approaches

Other approaches for social discounting in the literature have been recommended on and off over the years. These alternatives focus on different methods than the shadow price of capital approach for evaluating policies that displace private sector investments. However, the procedures these approaches use will not generally produce a correct estimate of the social present value of a policy's costs and benefits. Some of these other methods for social discounting are reviewed and evaluated below.

Weighted average of pre- and post-tax rates of return: A major alternative approach for addressing the divergence between the higher social rate of return on private investments and lower consumption rate of interest is to set the discount rate for public projects equal to a weighted average of the two. The weights would equal the proportions of project financing that displace private investment and consumption, respectively. Intuitively, this approach would set an overall project discount according to the amount lost by displacing consumption (using the lower consumption rate of interest) and the amount lost by displacing investments (using the higher social rate of return on private capital).

For example, suppose the social rate of return from private investments is five percent and the consumption rate of interest is three percent, as above. Suppose further that 75 percent of a public project's costs are financed using

[6] The assumption that additional government borrowing crowds out private investment dollar-for-dollar is not critical to the example. If crowding out is less than dollar-for-dollar, then the 75 percent of the project's cost that is financed by additional debt would be further divided into the proportion of that percentage of the cost that displaces private investment, which should then be adjusted using the shadow price of capital, and the remainder of the cost, which is drawn from consumption and therefore does not need to be adjusted.

government debt, with the remaining 25 percent of the costs raised through taxation. Finally, assume that government debt crowds out private investment on a dollar-for-dollar basis and that increased taxes reduce individuals' current consumption also on a one-for-one basis. The weighted average approach then suggests that the social rate of discount should be 75 percent of five percent plus 25 percent of three percent, or four and a half percent. If the proportions of the project's financing from each revenue source were reversed, however, the weighted average discount rate would instead be 25 percent of five percent plus 75 percent of three percent, or three and a half percent.

This approach has enjoyed considerable popularity over the years, and is probably acceptable for similarly timed cost and benefit flows.[7] As presented above, however, it is technically incorrect and can produce net present value results substantially different from the correct result (where "correct" is defined by the consumption rate of interest-shadow price of capital approach). The problem with the simple weighted average approach is that it seeks to accomplish two tasks using the social discount rate—pure time discounting and adjusting for the displacement of private investments that yield pre-tax social returns higher than the consumption rate of interest.

In general, the "synthesized" discount rate based on the social rate of return from private investments and the consumption rate of interest that accomplishes both objectives—and so arrives the correct present value—depends on the timing of the cost and benefits flows. A simple weighted average based only on project cost components will not in general produce the correct result.

To understand this, consider how the weighted average discount rate approach performs for the simple numerical example discussed above. Assume that the social rate of return on private investments is five percent, that the consumption rate of interest is three percent, that increases in government debt displace private investments dollar-for-dollar, that the $1 current cost of a public project is financed 75 percent with government debt and 25 percent with current taxes, and that the project produces a benefit 40 years from now that is estimated to be worth $5 in the future.

The weighted average social discount rate approach would suggest discounting the future benefit at a four and a half percent rate (0.75 times five percent plus 0.25 times three percent). This produces an estimated net social present value of -$0.14, which is the present value of the future $5 benefit discounted using a four and a half percent rate ($0.86) minus the current year $1 cost. In this case, the weighted average social discount rate approach suggests that the project's net social present value is negative. But earlier, the consumption rate of interest-shadow price of capital approach was applied to exactly this scenario, concluding that the net social present value is positive.

The problem with the weighted average approach is that its method for accounting for the higher social cost of displaced private investments is to "over discount" the benefits. But the amount of "over discounting" necessary in this example to adjust for the actual social costs of the project's costs depends on the time profile of the benefit stream—the farther in the future the benefits occur, the less "over discounting" is needed. The source of the project's financing is therefore insufficient to define a single rate of social discount that will produce correct net social present value results for any given policy.

Accordingly, to derive the weighted average discount rate that will produce the correct net present value requires that the consumption rate of interest-shadow price of capital method be used first to compute the net present value. The discount rate that produces this correct present value based on discounting costs and benefits, but not adjusting for the shadow price of capital, can then be calculated. There seems to be little purpose to this exercise because it requires the net present value of a policy to be computed using accurate procedures first before the adjusted discount rate can be derived.

Opportunity cost of capital: Another approach for social discounting argues that the government should not invest (or compel investment through its policies) in any project that offers a rate of return less than the social rate of return on private investments. Stated another way, because the citizens collectively enjoy the benefits of all

[7] Lind (1982b) provides a clear exposition of the weighted average approach for estimating the social discount rate. The large literature on this topic, spanning the 1960s through the early 1980s, has been summarized well by Lind and others.

public and private investments, welfare will be higher overall if the government invests in projects with the highest rates of return.[8]

Critics of this social investment rule argue that the government cannot realistically tax citizens and then invest in private sector projects. Therefore, the issue is not what "could" be done with the funds, but rather what "would" be done with them. Thus, if the government obtains funds for a project through taxation and this displaces only private consumption, then relative to consuming the resources today, welfare is increased as long as the project generates future benefits that exceed those costs when discounted at the consumption rate of interest. Of course, it remains true that welfare would be further increased if the funds were devoted to an even more valuable project.

A closely related opportunity cost-based observation is that the government faces a menu of projects and, for whatever reason, is not able to undertake all projects that have positive net social benefits when computed using a social rate of discount equal to the consumption rate of interest. In this event, the opportunity costs of funding one program are the benefits of other programs not funded. Proponents of this view typically conclude that the "hurdle" discount rate for a particular project should be equal to the rate of return offered by other projects foregone.

Regardless of the particular point of departure, the central point of the opportunity cost strand of the social discounting literature is valid. Social welfare will be improved if the government invests in projects that have higher values than if it invests in lower value ones. Hence, if the net present value of benefits of all courses of action are examined using the consumption rate of interest and the set with the highest net benefits are pursued, social welfare will be higher than otherwise.[9] So stated, this advice is correct.

However, it does not follow that rates of return offered by alternative private or public projects define the level of the social discount rate. An alternative project might produce large benefits over the future and thus offer a large "rate of return." But if individuals discount these future benefits using the consumption rate of interest, the correct way to describe this project is that it offers substantial present value net benefits. In general, the opportunity cost argument is not about the social discount rate *per se*, but about correctly and consistently examining the social values of all alternatives. As was the case for the shadow price of capital, an alternative project with a high rate of return will have a high social net present value. But this does not imply that its rate of return should become the social rate of discount to be used for pure time discounting for other projects.

Consumption Rate of Interest-Shadow Price of Capital: The New View

Over the years, the consumption rate of interest-shadow price of capital approach to social discounting has gained increasingly wide acceptance among economists. Recently, however, a key assumption in that analysis has been questioned—the assumption that the economy is "closed" to foreign capital flows—and an alternative hypothesis concerning government crowding out of private investment has been put forward.[10] According to this new view, earlier analyses implicitly assumed that capital flows into the nation were either nonexistent or very insensitive to interest rates, a "closed economy" assumption. Empirical evidence suggests, however, that international capital flows are quite large and very sensitive to interest rate changes. In this case, the supply of investment funds to the U.S. equity and debt markets is likely to be highly elastic (the "open economy" condition) and, thus, private capital displacement is much less important than it was previously thought to be.

Under this new view, it is inappropriate to assume that financing a public project through borrowing will result in dollar-for-dollar crowding out of private investment. If, instead, financing public projects results in no crowding out of private investment, then no adjustments using the shadow price of capital are necessary. Benefits and costs should be discounted using the consumption rate of interest alone. However, the literature to date does not adequately support the assumption of zero crowding out. It is more likely that there exists some degree of private capital

[8] Many authors cite high opportunity costs of public investments. Among these are Birdsall and Steer (1993), Schelling (1995), and Lyon (1994). On the technical issue of rates of return vs. net present values, see Lind (1990) and Cowen and Parfit (1992).

[9] Clearly, such an approach cannot be followed when a particular action is mandated.

[10] See Lind (1990) for this revision of the consumption rate of interest-shadow price of capital approach.

displacement within the spectrum between zero and dollar-for-dollar displacement. The degree of crowding out will depend on the magnitude of the policy or program being analyzed. Unfortunately, while the shadow price of capital adjustment requires an assessment of the proportion of costs that displace investment,[11] the literature provides little empirical evidence as to the relationship between project size and capital displacement.[12]

6.3.1.3 Applying the Consumption Rate of Interest Approach to Environmental Policies

The extension of the consumption rate of interest-shadow price of capital approach to the case of an economy "open" to substantial capital flows is relatively recent. And, as is true for most of the discounting literature, virtually all of the discussion focuses on public project financing, rather than on environmental policies that largely mandate that private parties undertake certain actions or expenditures in pursuit of social objectives. Finally, while it is intuitive to argue that private investments are not displaced by either additional government borrowing or mandatory private investments for environmental protection, it is often the gross gains and losses of the affected parties in the economy that are the focus of economic impact analyses. How the change in the assumption concerning the availability of investment funds to the economy translates into these gross gains and losses is critical for conducting accurate environmental policy assessments.

For all of these reasons, it is worth clarifying the capital displacement and adjustment issue for environmental policies that mandate capital investments in the context of both the "open" and "closed" economy assumptions regarding capital flows.

Environmentally-Mandated Private Investments in a "Closed" Economy

To focus closely and exclusively on the shadow price of capital adjustment issue, some simplifying assumptions are helpful. Assume that there is no risk and uncertainty, that all firms and the government borrow at the interest rate i, that taxes on investment income are levied on all sources of such income at a rate of t, and that the resulting post-tax interest rate, r $(=i\times(1\text{-}t))$, is the rate at which individuals discount future consumption.[13] Further, assume that the net-of-tax returns from all investments are consumed in each year (to assist in making this illustration as simple as possible). Assume, finally, that the supply of investment funds is perfectly inelastic with respect to their price, the interest rate.

Consider, first, a public project that costs \$1, is financed through taxes on labor and other factors of production (but not capital), and offers future environmental benefits. Assuming that increased current taxation only reduces consumption, the cost of the project is this amount of reduced current consumption. Future benefits, once valued in terms of future consumption, can be discounted to the present using a social rate of discount equal to the consumption rate of interest. For the remainder of this discussion, the benefits side of the calculations will be ignored to focus on the cost calculation considerations.

Now, consider exactly the same project, but assume that it is financed only through government borrowing, which crowds out an equal amount of private sector investment. To calculate the costs of this project financing, it is helpful to analyze the impacts on the different entities affected. First, the private sector investors who lend \$1 to the government instead of to private firms are indifferent. They receive the interest rate i from either source and, therefore, continue to receive a stream of returns net-of-tax equal to $\$1\times r$.

Next, consider the government, which can be thought of as representing the interests of citizens in future years. The foregone private investments would have generated a stream of tax revenues of $\$1\times t\times i$ each year, which is lost. But the increased public debt is taxable, so the government regains this $\$1\times t\times i$ each year and the streams of gained and lost tax revenues offset each other. Nevertheless, the government must service this new debt

[11] See footnote 5 for a description of this adjustment.

[12] See Lind (1990) for a summary of the empirical literature in this area.

[13] The relevant tax rate t is the effective marginal tax rate. It is difficult to determine this rate in the aggregate with any reasonable degree of accuracy.

by raising future taxes each year by the amount $\$1 \times i$ (assuming, for simplicity, that the debt is a perpetuity).

As a result, the cost of financing this public project through government debt in a closed economy context is a stream of decreased consumption experienced in the future of $\$1 \times i$ per year forever. The present value of this stream of foregone consumption computed using the consumption rate of interest, r, exceeds \$1. This is the essence of the shadow price of capital adjustment rationale. The value of i exceeds r in this example because of the tax wedge between the social (pre-tax) rate of return on investments and the (post-tax) consumption rate of interest. This is the equivalent of observing that a taxable investment yields a private return of r per year to the investor and a "return" of $t \times i$ to the government in the form of tax revenues.

Assume now that the relevant investment is a private sector capital project that must be undertaken in order to comply with an environmental policy. To estimate the social costs of this requirement under the closed economy assumption, two polar cases are useful to examine: no cost shifting to consumers or other factors of production and full cost shifting to consumers through higher product prices.

In the case of no cost shifting to consumers, the owners of the firms required to make these investments either must obtain debt or equity funds or reduce their other investment and lending activities, to comply. Wherever the required funds originate, two facts are clear. One is that other taxable investments of \$1 will not be undertaken. The second is that, because the price of the products or services into which this environmental investment flows does not rise, the mandated investment will produce no "return" for their owners or for the government in the form of future tax revenues. The result is that the owners of the affected firms lose a stream of investment income, $\$1 \times r$, and the government loses a stream of tax revenues of $\$1 \times t \times i$, because of the displaced private investment. But since $r = i \times (1-t)$, this adds up to a stream of costs of $\$1 \times i$ per year. Once again, this is essentially the shadow price of capital adjustment.

Now assume that the cost of the mandated environmental investment is shifted to consumers through higher prices, which rise by enough to provide the full social pre-tax return of i. In this case, the owners of the firms required to make these investments are indifferent. Similarly, the government is indifferent—it still receives a stream of tax revenues from the \$1 investment. Here, however, it is consumers of the affected product or service who are not indifferent. In fact, the product price increases they face are precisely enough to provide the $\$1 \times i$ pre-tax social return on the mandated investment. Here again, this is essentially the shadow price of capital adjustment.

Environmentally-Mandated Private Investments in an "Open" Economy

The central difference between the closed and open economy contexts concerns the conditions of supply of investment funds. In the closed economy case, the amount of these funds is fixed, so the total available for all projects, private and public, is constant. Hence, the key to analyzing that case lies in tracing the implications of altering the composition of the investments undertaken with and without a new public project or a new environmental policy mandating private investments.

In the open economy context, however, what is fixed is not the supply of investment funds, but the price at which they may be obtained. In this case, all investments worth undertaking without a new public project or a new environmental policy requiring investments will still be worth undertaking with those new policies—so that there will be no impact on capital availability and the level of private sector investments. This suggests that measuring the costs of these policies in this open economy context may be slightly different than in the closed economy case.

Purely tax-financed public projects are not discussed here because the results for that case do not depend on the assumption concerning the supply of capital. For debt-financed public projects, however, the results under the closed and open economy assumptions are very different. In this open economy case, the government's increased \$1 of borrowing does not change the level of U.S. private sector investment. Hence, the government must service the debt at a cost of $\$1 \times i$ per year, but also gains from that $\$1 \times i \times t$ of tax revenues from these new taxable interest payments.[14] The net cost is only $\$1 \times i \times (1\text{-}t)$, which is the stream of future reduced consumption citizens will experience as the net cost of the new public project. But because $i \times (1\text{-}t) = r$, this stream of reduced consumption is equal to $\$1 \times r$. Discounted at the consumption rate of interest, r, the present value cost is $1. This is the rationale for not using the shadow price of capital adjustment.

To analyze the implications of the open economy assumption for mandated private investments, the no- and full-cost pass-through polar cases continue to be helpful. In the case of no-cost pass-through, the results are very simple. The owners of the firms required to make the investments to comply with an environmental policy will obtain the necessary funds either from their own resources that would have been invested elsewhere, or from other sources, and undertake the required investments. Because the price of the services or products subject to the new policy do not rise to compensate for these costs, no return to these owners or to the government in the form of tax revenues will result. But, because the supply of investment funds to the economy is perfectly elastic, no other private sector investments will be foregone.

The result in this case is that the owners of the entities required to make these investments will lose a stream of private investment returns of $\$1 \times r$ (their net-of-tax return on production investments) if the mandatory investment causes them to reduce investment elsewhere. Alternatively, the owners of the affected firms may increase their demands for investment funds in the market and continue with their pre-policy investment plans. Nevertheless, because i is constant, all investment projects that were profitable before the policy is imposed will still be profitable and these investments will be undertaken as if the policy did not exist. Hence, the government loses no tax revenue as a result and no shadow price of capital adjustment is appropriate here.

Finally, if the costs of the mandated private investments are fully passed through to consumers, the owners of the affected firms are now indifferent. The government and the consumers of the relevant services or products, however, are not. First, the consumers of the affected sector's output face price increases equivalent to $\$1 \times i$, which is the amount necessary to fully recoup the full pre-tax social return on the invested capital. But the government gains a stream of tax revenues associated with this mandated investment, amounting to $\$1 \times i \times t$ per year. Again, all other investments are still undertaken because of the assumption regarding the supply of investment funds.

As a result, the net cost to society is the price increase borne by consumers, equal to $\$1 \times i$ per year, minus the increase government tax revenues—which represents future reduced taxation—of $\$1 \times i \times t$ per year, for a net cost of only $\$1 \times i(1\text{-}t) = \$1 \times r$. Thus, the shadow price of capital adjustment is not necessary here. But, note that the cost increase for the firm and its consumers is measured by the pre-tax amount per year, $\$1 \times i$, not the net social cost of $\$1 \times r$ per year. The former is the relevant measure for modeling private sector "economic impacts" and for assessing the gross gains and losses of a policy, while the latter represents the social perspective.

6.3.1.4 Summary of Advice from the Economics Literature

The vast majority of the traditional social discounting literature has focused on exploring the implications for public project evaluation of a few, probably very important, departures from the idealized no-other-distortions simplified economy for which unambiguous social discounting recommendations can be made. Yet, in the development of that literature, many matters have been addressed and are considered by many contributors to this literature to be somewhat settled, some of which are discussed above, and others not (largely because they are not directly social discounting issues). In particular, for intra-generational social discounting:

- There is reasonable agreement that the social rate of discount ought to reflect the private rates of consumption discount of the citizens affected.
- If social and private returns from private investments are different, then adjustments should be introduced

[14] The taxability of the interest payments on the increased amount of government debt in an open economy context is complex because of the international nature of capital markets. Generally speaking, taxes are owed on interest earnings from government obligations to the country that pays the interest, although there are exceptions to this rule. Hence, if U.S. citizens increase their lending to the U.S. government, the interest earnings would clearly be taxable. If foreign investors purchase the increased U.S. government debt, normally these interest payments are taxable as U.S. income.

to reflect this when and if policies alter private investment flows.

- Uncertainty and risk should largely be addressed through appropriate valuation of costs and benefits (e.g., certain monetary equivalents) rather than through modifications of the discount rate.

- Changes in the values of environmental goods and other such factors should likewise be reflected in direct cost and benefit measurements, not through adjustments to the social discount rate.

- Irreversibility of consequences is an option value concept and requires separate treatment in benefit-cost analyses, but it does not provide a reason to adjust the discount rate.

- Opportunity costs of other public and private uses for funds should be considered in evaluating the desirability of undertaking a particular public investment or policy. That a project offers a positive present value of net benefits when discounted using the consumption rate of interest does not by itself imply that the policy should be undertaken.

These conclusions demonstrate the significant progress made in the theoretical social discounting literature, especially regarding the implications of divergences between social and private rates of returns on investments. However, exactly what numerical rate of interest to use for social discounting in practical policy evaluations remains somewhat unsettled.

Moreover, some recent literature questions some of the most basic premises underlying the conventional social discounting analysis. For example, recent studies of individuals' financial and other decision making suggest that even a single person may appear to value and discount different actions, goods, and wealth components differently. This "mental accounts" or "self-control" approach suggests that individuals may well evaluate some aspects of the future quite differently from other consequences. The discount rate an individual might apply to a given future benefit or cost, as a result, may not be observable from market prices, interest rates, or other phenomena. This may be especially the case if the future consequences in question are not tradable commodities. Some recent evidence from experimental economics also indicates that discount rates appear to be lower the larger the magnitude of the underlying effect being valued, higher for gains than for losses, and tend to decline as the length of time to the event increases.[15]

Despite all of these limitations, practical economic analyses must use social discounting to assist in evaluating environmental policies. Hence, even limited guidance is helpful in developing recommendations for practical analyses. What is offered in the empirical literature for choosing a social discount rate focuses on estimating the consumption rate of interest at which individuals translate consumption through time with reasonable certainty.

For this, historical rates of return, post-tax and after inflation, on "safe" assets, such as U.S. Treasury securities, are normally used, typically resulting in rates in the range of one to three percent.[16] Some studies have expanded this portfolio to include other bonds, stocks, and even housing and this generally raises the range of rates slightly. It should be noted that these rates are *ex post* rates of return, not anticipated, and they are somewhat sensitive to the time periods selected and the classes of assets considered.[17] A recent study of the social discount rate for the United Kingdom places the consumption rate of interest at two to four percent, with the balance of the evidence pointing toward the lower end of the range.[18]

[15] Shefrin and Thaler (1988) and Thaler (1985) are central sources for the mental accounts idea and Lowenstein and Thaler (1989) report numerous examples of various inconsistencies and other aspects of individual intertemporal choices.

[16] Estimates of the consumption rate of interest (an individual's marginal rate of time preference) could be based on either after-tax lending or borrowing rates. Because individuals may be in different marginal tax brackets, have different levels of assets, and have different opportunities to borrow and invest, the type of interest rate that best reflects marginal time preference will differ among individuals. Additionally, individuals routinely are observed to have several different types of savings, each possibly yielding different returns, while simultaneously borrowing at different rates of interest. Thus, discerning an average marginal rate of time preference from observed interest rates is very difficult. However, the fact that, on net, individuals generally accumulate assets over their working lives suggests that the after-tax returns on savings instruments generally available to the public will provide a reasonable estimate of the consumption rate of interest.

[17] Ibbotson and Sinquefield (1984 and annual updates) provide historical rates of return for various assets and for different holding periods.

[18] Lind (1982b) offers some empirical estimates of the consumption rate of interest. Pearce and Ulph (1994) provide the estimates of the consumption rate of interest for the United Kingdom. Lyon (1994) provides estimates of the shadow price of capital under a variety of assumptions.

Finally, for the shadow price of capital, even less concrete empirical guidance is available. This parameter depends on the consumption rate of interest, the gross-of-tax rate of return on private investment, and the rate of consumption out of net investment returns, among other factors. Depending on the magnitudes of these factors, shadow prices from close to one to three, 20, 100, and infinity can result. Lyon (1990) has an excellent review of how to calculate the shadow price of capital and possible settings for the various parameters that determine its magnitude. Moreover, the shadow price of capital adjustment will require an assessment of the proportion of project costs that displace private investment. Whether or not this adjustment is necessary appears to depend largely on whether the economy in question is assumed to be open or closed and on the magnitude of the intervention or program considered relative to the flow of investment capital from abroad.[19]

6.3.1.5 Guidance for Intra-Generational Social Discounting

For economic analyses of intra-generational policies analysts should apply the consumption rate of interest approach. There should be no adjustments using the shadow price of capital unless there are strong reasons to believe that a particular policy will affect the level of U.S. private sector investment.[20] Based on historical rates of return on relatively risk-free investments, adjusted for taxes and inflation, a consumption rate of interest measured at two to three percent is justified.

OMB's own guidance on discounting[21] currently recommends discounting using a rate of seven percent, an estimate of the average real pre-tax rate of return generated by private sector investments. EPA economic analyses therefore should provide estimates of the present values of costs and benefits using both a two to three percent rate and OMB's guidance on discounting. In some cases, a sensitivity analysis at discount rates within this range may provide useful information to decision makers.

In addition, all analyses should present the undiscounted streams of benefits and costs. This is not equivalent to calculating a present value using a discount rate of zero. In other words, the flow of benefits and costs should be displayed rather than a summation of values.

6.3.2 Inter-Generational Social Discounting

This section focuses on social discounting in the context of policies with very long time horizons involving multiple generations. Policies with potential inter-generational impacts include global climate change, radioactive waste disposal, groundwater pollution, and biodiversity. Because of potentially large or catastrophic impacts on unborn generations and because policies with very long time horizons often involve high costs imposed by current generations, there is less agreement in the literature on the appropriate approach to discounting over very long time horizons. This section attempts to present a balanced discussion of alternative points of view. The discussion first focuses on how the point of departure for inter-generational discounting differs in some very fundamental ways from that of intra-generational social discounting. Next, various approaches for deciding whether and how to discount when evaluating inter-generational policies are reviewed. Finally, the section concludes by summarizing the advice offered by the economics literature and EPA's new guidelines for inter-generational social discounting.

[19] Studies suggesting that increased U.S. government borrowing does not crowd out U.S. private investment generally examine the impact of changes in the level of government borrowing on interest rates. The lack of a significant positive correlation of government borrowing and interest rates is the foundation of this conclusion. Because changes in yearly U.S. government borrowing during the past several decades have been in the many billions of dollars, it is reasonable to conclude that EPA programs and policies costing a fraction of these amounts are not likely to result in significant crowding out of U.S. private investments.

[20] As the estimation of the shadow price of capital can be a costly exercise, analysts should use a value-of-information approach to determine whether it is worthwhile to pursue a quantitative assessment of the effects of private capital displacement. Should a quantitative assessment be undertaken, the analysis should include a sensitivity analysis of alternative assumptions regarding the degree of crowding out.

[21] OMB Circular A-94, "Guidelines and Discount Rates for Benefit-Cost Analysis of Federal Programs," October 29, 1992 (note: see updates to cost-effectiveness rates—most recent released in January 2000, http://www.whitehouse.gov/OMB/circulars/a094/a094.html (accessed 8/28/2000)) and U.S. Office of Management and Budget, *Guidelines to Standardize Measures of Costs and Benefits and the Format of Accounting Statements,* report M-00-08, March 22, 2000,.http://www.whitehouse.gov/media/pdf/m00-08.pdf (accessed 8/28/2000).

6.3.2.1 Analytical Foundation of Inter-Generational Social Discounting

One obvious problem with long-time-horizon policies is that many of the people affected are not alive. Hence, while the preferences of each affected individual are knowable (if probably unknown in practice) for intra-generational social discounting problems, they are essentially unknowable for those involving future generations not yet born. This is not always a severe problem for practical policymaking, especially when policies impose relatively modest costs and benefits, or when the benefits begin immediately, or in the not too distant future. And most of the time, it suffices to assume that future generations will have preferences much like those of present generations.

The more serious challenge posed by long-time-horizon situations arises primarily when costs and benefits of an action or inaction are very large and are distributed asymmetrically over vast expanses of time. Here, the crux of the problem is that future generations are not present to participate in making the relevant social choices. Instead, these decisions will be made only by existing generations. Social discounting in these cases can no longer be thought of as a process of consulting the preferences of all affected parties concerning their valuation today of effects they will experience in different time periods.

Moreover, compounding interest over very long time horizons can have profound impacts on the inter-generational distribution of welfare. An extremely large cost far enough in the future has essentially zero present value when discounted at even a low rate. But a modest sum invested today at the same low interest rate also can grow to a staggering amount given enough time. Therefore, mechanically discounting very large distant future effects of a policy without thinking carefully about the ethical implications is not advised.

6.3.2.2 Perspectives on Inter-Generational Social Discounting

The social discounting literature contains many different perspectives on social discounting in inter-generational contexts. This section briefly describes the major approaches and their theoretical motivations. The focus in this discussion is on the social discount rate itself, so such other issues as the shadow price of capital adjustments, while clearly still relevant under certain assumptions, are kept in the background.

Social Welfare Planner Approach

One popular recommendation is that social discounting for inter-generational policies should be based upon methods economists have used for many years in optimal growth analyses. In these models, the policy maker is understood to be maximizing the utilities of all present and future generations using a well-defined social welfare function.[22]

In optimal growth models, the social rate of discount generally equals the sum of two factors. One is a discount rate for pure time preference, which measures the degree to which the social planner favors the utility of current and near future members of society over that of individuals in the more distant future. The other is an adjustment reflecting the fact that the marginal utility of consumption will decline over time as consumption per capita increases (equal to the elasticity of marginal utility multiplied by the rate of increase of consumption over time).

If the world actually corresponded to the theoretical construct of an optimal growth model and there were no taxes or other distortions, the social discount rate as defined in these models would be equal to the market interest rate. And, the market rate of interest, in turn, would also be equal to the social rate of return on private investments and the consumption rate of interest. But because the world contains many distortions, and is not likely to conform to the conditions that characterize optimal growth models, the social rate of discount is not observable in the economy.

Recent practical applications of this approach to very-long-time-horizon analyses have therefore attempted to estimate the social discount rate by constructing it from its components. Most assume that the rate of pure time discount is zero, adhering to the ethical precept that the policy maker ought not to inherently favor present generations' consumption over that of future generations. For the other component of the social discount rate,

[22] Key literature on this topic includes Arrow et al. (1996), Lind (1994), Schelling (1995), Solow (1992), Manne (1994), Toth (1994), Sen (1982), Dasgupta (1982), and Pearce and Ulph (1994).

hypothetical, but perhaps plausible, estimates of the elasticity of marginal utility and the rate of growth of consumption over time are introduced. The product of these two factors is the implied social discount rate. This computational procedure in essence derives an implied social rate of discount under the assumption that future generations will be richer than current generations, so that the marginal utility of consumption is projected to fall over time. Rates developed using this technique generally range from one-half percent to three percent.[23]

Optimal growth modeling, however, is only one strand of the substantial body of research and writing on intertemporal social welfare maximization and optimal growth. This literature extends from the economics and ethics of inter-personal and inter-generational wealth distribution, to the more specific environment-growth issues raised in the "sustainability" literature, and even to the appropriate form of the social welfare function, e.g., utilitarianism or Rawls' maxi-min criterion.

Clearly, economics alone cannot provide definitive guidance for selecting the "correct" social welfare function or the social rate of time preference. Nevertheless, economics can offer a few insights concerning the implications and consequences of alternative choices and some advice on the appropriate and consistent use of the social welfare function approach as a policy evaluation tool.

Approaches Based on Existing Individuals' Preferences

The major alternative to the social welfare planner approach for inter-generational discounting is to rely on the preferences of current individuals for an appropriate discount rate. At its core, this perspective rejects the view that the problem is one of balancing the interests of all humans who will live now and in the future. Instead, according to this perspective, it is fundamentally about individuals alive today allocating their scarce resources to competing ends, one of which happens to be the welfare of future generations. Several specific approaches fall into this category.

- **Consumption rate of interest/infinitely-lived individuals:** Although not popular in theoretical terms, in practice it is common to adopt the approach of simply making no great distinction between inter-generational and intra-generational social discounting. Models of infinitely-lived individuals, for example, suggest the consumption rate of interest as the social discount rate. But the assumption that people live forever is contrary to the fact that individuals actually do not live long enough to experience distant future consequences of a policy and to report today the present values they place on those effects. As such, models of infinitely-lived individuals essentially ignore the fundamental problem posed in evaluations of policies that affect distant future generations.

- **Inter-generational discounting vs. time discounting:** Another suggestion for social discounting in inter-generational contexts is to examine possible differences between how current individuals evaluate the welfare of their descendants versus how they discount their own future consumption.[24] It is possible that the year-by-year exponential time discounting [25] that underlies an individual's allocation of his or her own consumption in the present and the future does not apply to this individual's valuation of his or her descendant's welfare. That is, a person might indeed value future generations' consumption less than his or her own current and future consumption, but not as low as would be implied by standard discounting techniques. An individual's present valuation of the consumption of successive future generations might decline gradually and approach some constant positive value, so that the value of a unit of consumption by a person 10 generations from the present might be considered to be the same as a unit of consumption by a person 11 generations from now.

  A related line of reasoning suggests that large-scale catastrophic consequences in the future are viewed differently than marginal changes in welfare, so that it matters little whether these possibilities are 100 years

[23] See IPCC (1996), pp. 131-132.

[24] Sources discussing this approach include Rothenberg (1993), Cropper et al. (1992), Shefrin and Thaler (1988), Thaler (1985), and Cowen and Parfit (1992).

[25] Exponential time discounting applies discounting factors to future values that increase as the time between the present and the future when those values will be experienced increases. As a result, the present value of a benefit to be enjoyed 50 years from now is much higher than the present value of the same benefit if it accrues 100 years from now.

or 1,000 years in the future. If so, applying exponential, or year-by-year, time discounting to such future consequences is inappropriate.

- **Revealed/stated preferences for altruism:** According to this view, environmental policies that affect distant future generations are considered to be altruistic acts.[26] As such, they should be valued by current generations exactly the same as other acts of altruism. Hence, the discount rate in question is not that applied to an individual's consumption, but instead that applicable for an individual's valuation of the consumption or welfare of someone else.

  At least some altruism is apparent from international aid programs, private charitable giving, and bequests within overlapping generations of families. But the evidence suggests that the importance of other people's welfare to an individual appears to grow weaker with temporal, cultural, geographic, and other measures of "distance." The implied discount rates that survey respondents appear to apply in trading off present and future lives are also relevant under this approach. One such survey (Cropper, et al., 1992) suggests that these rates are positive on average, consistent with the rates at which people discount monetary outcomes, and decline as the time horizon involved lengthens.

- **Opportunity cost of alternatives:** A variety of perspectives in the inter-generational discounting literature converge on the broad notion that devoting resources to long-time-horizon environmental projects—largely because low discount rates appear to make these attractive in present value terms—neglects numerous other social investment opportunities with higher values.[27]

  Advocates of this point of view point to numerous alternative social investments that would generate far larger benefits now and in the future, such as basic infrastructure, education, medical assistance, and other projects in developing nations.

  Depending on the context, this point of view is often expressed in two different ways: (1) many other investments would be more beneficial to society and so long-time-horizon environmental programs face very high opportunity costs, and (2) the rates of return offered by these alternative investments are high and these rates ought to be used as the social rate of discount.

  As noted earlier in the context of intra-generational social discounting, the first statement of this opportunity cost argument is the correct one, the second is somewhat problematic. The opportunity costs of alternative government or private investment programs are appropriately measured by calculating their present values using the social rate of discount. If these projects have higher rates of return than the social discount rate, their social present values will also be high. But this does not imply that the social rate of discount itself ought to be set equal to some alternative project's rate of return. For example, an alternative project might offer a very large rate of return for only one year, but this should not become the social rate of discount for very-long-time-horizon projects and policies.

- **Paretian compensation tests:** One final approach for social discounting in an inter-generational context returns to the theoretical motivation and ethical underpinnings of intra-generational social discounting. This approach views social discounting in inter-generational contexts as a question of whether the distribution of wealth among many different generations could be adjusted in order to compensate the losers under an environmental policy and still leave the gainers better off. Whether gainers could compensate losers hinges on the rate of interest at which society (the U.S. presumably or perhaps the entire world) can transfer wealth across hundreds of years. Some argue that in the U.S. context, a good candidate for this rate is the federal government's borrowing rate.

  What lies at the foundation of this approach is the goal of maintaining overall inter-generational equity. The implicit assumption is that society starts from a position at which the distribution of wealth across present and future generations is "acceptable." Then

[26] Schelling (1995) and Birdsall and Steer (1993) are good references for these arguments.

[27] Many authors cite high opportunity costs of public investments. Among these are Birdsall and Steer (1993), Nordhaus (1993), Schelling (1995), and Lyon (1994).

it is discovered that some current environmental action or inaction will impose large burdens on future generations. To maintain inter-generational equity, some sort of accumulation fund is necessary to provide compensation for those harms.

While this approach offers solid advice for selecting a social discount rate for inter-generational policy evaluation, its resolution of the many difficult social choice problems posed by such policies rests on two critical assumptions. One is that the initial distribution of inter-generational wealth is socially acceptable. If this is not the case, it is not clear that attempting to maintain that distribution after discovering the long-term environmental problem is an appropriate goal.

Second, if the compensation fund is not accumulated, then the decision not to remedy this environmental problem is once again recast as an inter-generational equity problem, not a question purely of economic efficiency. There is considerable skepticism regarding the willingness of the current generation to provide these compensation funds and a significant concern that intervening generations might not continue the accumulation process. Thus, actually undertaking the process of locking away sufficient savings for distant future generations to compensate them for environmental harms is a very different matter than determining the rates of interest at which such a fund might grow.

6.3.2.3 Summary of Advice from the Economics Literature

There is little consensus in the economic literature on social discounting for inter-generational policies. In particular, the fundamental choice of what moral perspective should guide inter-generational social discounting—a social planner who weighs the utilities of present and future generations, the preferences of the current generations regarding future generations, or perhaps other approaches—cannot be made on economic grounds alone.

It is important, however, to view this result in the proper context. In fact, the practical effect of this lack of consensus concerning social discounting for inter-generational policies is not as profound as it at first appears. The major problems with discounting in long-time-horizon contexts occur in probably a few cases out of a vastly larger set, particularly where costs and benefits are inherently high and are substantially divorced in time. But the environmental policies that fit this description are uncommon because most environmental programs are relatively short in duration and reversible, with their time frames determined largely by capital investments.

6.3.2.4 Guidance for Inter-Generational Social Discounting

Based on the theoretical social discounting literature and other considerations, economic analyses of policies with inter-generational effects should generally include a "no discounting" scenario by displaying the streams of costs and benefits over time. This is not equivalent to calculating a present value using a discount rate of zero (i.e., the flow of benefits and costs should be displayed rather than a summation of values).

Economic analyses should present a sensitivity analysis of alternative discount rates, including discounting at two to three percent and seven percent as in the intra-generational case, as well as scenarios using rates in the interval one-half to three percent as prescribed by optimal growth models. The discussion of the sensitivity analysis should include appropriate caveats regarding the state of the literature with respect to discounting for very long time horizons.

6.4 Discounting and Non-Monetized Effects

Despite analysts' best efforts to assign monetary values to all of the consequences of an environmental policy, there are instances in which monetization is not feasible. This section briefly explores social discounting when some elements are not expressed in monetary terms.[28]

[28] Although this discussion focuses exclusively on non-monetized benefits, many cost categories are often not monetized as well. The time costs consumers experience as a result of some policies, the financial costs of business delays that result from others, and the quality and performance impairments caused by yet other policies often are not monetized in economic analyses. Discounting policy regarding these non-monetized effects should largely track discounting practices for monetized costs unless there are reasons for not doing so, similar to those described in this section, for leaving some non-monetized benefits undiscounted.

6.4.1 Perspectives on Discounting Non-Monetized Effects

One strategy for addressing future non-monetized effects is to discount them as though they had been monetized. Some argue, however, that environmental benefits that have not been monetized cannot—or should not—be discounted and summarized together with costs in a cost-effectiveness or benefit-cost summary. Two basic lines of reasoning are normally offered. One is that because discounting is essentially a financial process designed to evaluate investment decisions, it is only relevant to dollar-denominated streams, and so benefits that are in physical rather than dollar terms *cannot* be discounted. Discounting some types of benefits, such as avoided damages to human lives or natural resources, treats these tangible risk-related benefits as monetary outcomes, when they are not in fact financial consequences.

The other line of reasoning for not discounting non-monetized benefits is that it is ethically unacceptable to discount physical units. If, for example, cancer cases that occur in the future are discounted to the present, this effectively asserts that a future cancer case is not really a cancer case, but rather is only 80 percent, 20 percent, or some other fraction of a "full" current cancer case. Discounting therefore somehow cheapens the future effect's value or reduces its importance and is unfair to future individuals or generations whose lives or natural resources are at stake. This argument is often applied not only to human health and environmental effects that are simply enumerated, but also to those that are monetized.

Evaluating these arguments requires a clear understanding of the various reasons why benefits might not be monetized in any given analysis. In some cases, benefits are not monetized because the environmental and health impacts may be unknown, so that only changes in emissions, production, exposure, or other imperfect proxies for benefits, damages, or harms, are available. Sometimes there may be an estimated time stream of human health and environmental impacts, but the needed valuation tools and information on how to monetize the benefits are not—or are only partially—available. Finally, in still other cases, physical effects have been estimated and could be monetized, but this last step—converting measured physical effects into dollar values of benefits—has simply not been taken.

6.4.2 When Discounting Non-Monetized Effects Is Appropriate

In many cases, quantitative information on the time streams of physical effects is available and these effects are measured in terms of human health consequences and ecosystem damages that correspond to endpoints that are normally monetized. If so, then these non-monetized benefits ought to be discounted if monetized costs and benefits are discounted. Discounting non-monetary effects in these cases is not inherently different from discounting these units after attaching a unit value in dollar terms. What is being conveyed is the notion that effects felt farther in the future are worth less in today's terms than those that occured earlier in time. Thus, if two policies have identical current costs and the same amount of benefits in the future except that one produces these benefits earlier in time, the policy that offers earlier benefits will have a higher social value.

Choosing not to discount non-monetized benefits can have perverse consequences. First, to the extent that the act of discounting and the choice of discount rate embody a rational investment criterion, failing to discount non-monetized benefits may produce results that appear to be irrational or intrinsically unappealing. Suppose, for example, there is a policy that is estimated to save five lives in the year it is implemented. This policy can either be implemented today (Option A) or 20 years from now (Option B), and the undiscounted costs in current dollars are the same for both options. If the discounted costs are compared with undiscounted benefits, a cost-effectiveness evaluation will clearly favor Option B. Thus, failing to discount benefits can produce a situation in which society has little motive to pursue current environmental benefits because by investing instead, larger net environmental benefits can be gained in the more distant future.

Finally, surveys that examine individuals' attitudes toward public policies with non-monetized benefits suggest that people do appear to apply a positive discount rate to these future effects. For example, contingent valuation studies (Cropper et al., 1992; Carson et al., 1987; Horowitz and Carson, 1990) that look at individuals' preferences for saving lives find that individuals prefer projects that save lives in the near term over equivalent cost projects that save lives in the future.

6.4.3 When Discounting Non-Monetized Effects Might Not Be Appropriate

While there are many cases in which non-monetized benefits can and should be discounted along with all of the other costs and benefits of environmental policies, there are others in which benefits are not monetized for reasons that pose more significant problems for discounting. Specifically, sometimes the available measures of benefits are very poor proxies for ultimate damages, making it difficult to discount them correctly.

When an analysis stops far short of the physical effects that are good proxies for damages, the relationship between harms and emissions—or other relevant physical measures—might be poorly understood. In the case of the greenhouse effect, for example, the ultimate impact of a ton of greenhouse gas emitted in a given year depends on the subsequent change in the time paths of temperature, sea level, and other variables, and on the physical effects and economic impacts accompanying these changes. Changes in temperature depend, in turn, on the magnitude of emissions of all greenhouse gases over time and their radiative forcing. Further, the impacts of climate change may depend not only on the absolute levels of these effects, but on the rate at which they occur. Because linking quantified physical harms to a unit of emissions is a difficult task, discounting greenhouse gas emissions would be a premature and problematic step in determining the cost-effectiveness of two alternative emission reduction strategies.

Similarly, even when benefit estimates are based on linkages from emissions to other physical and biological endpoints, often these benefit measures are still not close enough to the endpoints of ultimate concern to allow discounting. For example, although pollution damages can be measured in terms of species diversity, ecosystem health, and forest productivity, the further detailed linkages from those damages to current and future recreation, production, non-use, or other values identified by economists and ecologists often do not exist.

Discounting non-monetized effects is also not warranted when doing so actually conceals information of value to policy makers. For example, suppose a policy reduces current and future effluent discharges to a river. Suppose further that this river has a complex chemistry, so that interactions between the effluent reduced by this environmental policy and other natural and human inputs to the river are unknown and/or the relationship between effluent discharges and damages is nonlinear (e.g., the river is subject to degradation only after passing some threshold). Here, the same quantities of effluent reduction in different time periods are not necessarily identical in their effects, so not only is there a time element to contend with, but also possible differences in ultimate environmental benefits. In this case it might be far more useful to display the stream of effluent reduction and probabilities of exceeding thresholds each year, rather than to discount all of the future effluent reduction.

In all of these examples, the problem is that analysts have an incomplete understanding of the relationship between emissions—and production or other physical units that are potentially subject to control—and the actual harm to human health or the environment that result. However, a general preference for earlier benefits over later ones still applies. The problem is that discounting in these cases masks important information by implicitly assuming that a unit of benefits in one period has an identical effect on the ultimate benefit consequences of concern as a unit of the same benefit in another period. When non-monetized benefits measures are far from the human health and other benefit categories of true concern, this assumption often is contrary to reality.

When it is not appropriate to discount certain non-monetized benefits, comparisons of costs and benefits can still be made without directly discounting the benefits. For example, if costs and benefits occur in each time period over the course of a policy, and these do not change significantly over time, net social benefits can be explored without discounting by examining a representative year's costs and benefits. Similarly, if the benefits are relatively constant through time, but the costs are not, the costs can be annualized and compared to the annual benefits using cost effectiveness analysis. Another approach is to cumulate costs forward with interest to compare this future value to the benefits, a method that is particularly suitable when the benefits occur in only one future year. If none of these methods applies, simply presenting the streams of costs and monetized and non-monetized benefits to policy makers is often sufficient.

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Chapter 7: Analyzing Benefits

7.1 Introduction to Analyzing Benefits

At its roots, benefits analysis develops monetary values to inform the policy making process. These values are important because they allow decision makers to directly compare costs and benefits using the same measure (i.e., dollars). A complete benefits analysis is also useful because it makes explicit the assumptions about the value of benefits embedded in different policy choices. This chapter focuses on those benefits that can be expressed in terms of dollars. Chapter 10 discusses the presentation of non-monetized benefits, those that cannot be expressed in dollar terms.

This chapter presents information on the theory and practice of benefits assessment for environmental policies. The discussion focuses on the benefits possible from a "typical" EPA policy or regulation that reduces emissions of contaminants into the environment. However, the principles discussed here apply to other types of EPA policies, such as those that provide information or regulatory relief.

Most EPA benefits analyses face two serious challenges. First, a given policy may produce many different benefits, but it is seldom possible to obtain a single, comprehensive value estimate for the collection of effects. This will often leave analysts with no alternative but to address these effects individually, aggregating values to generate an estimate of the total benefits of a policy alternative. Although there are exceptions to this "effect by effect" process for benefits analysis, much of the discussion in this chapter assumes that analysts will be forced to adopt this approach.

The second major challenge faced by analysts is the difficulty of conducting original valuation research in support of specific policy actions. Because it is often too expensive or time consuming to perform original research, analysts will need to draw upon existing valuation estimates for use in benefits analysis. The process of applying these estimates to value the consequences of policy actions is called *benefits transfer*. Although the benefit transfer method is detailed in only one section, this chapter is generally written with benefit transfer in mind. For example, the descriptions of valuation methods in Section 7.5 include recommendations for assessing the quality of published studies. This is done to help analysts determine which studies deserve consideration for use in benefit transfers.

While analysts should always seek precision, they must make assumptions and exercise professional judgment to face the challenges noted above, as well as numerous others that arise in a benefits analysis. Existing value estimates, for example, are often subject to large uncertainty bounds due to measurement error, model uncertainty, and the inherent variability of individual preferences. When drawing from these studies—and when using quantitative estimates of any kind—analysts should carefully assess the quality of the data and should clearly state the reasons for their analytical choices. As with any analytical exercise, the maxim "garbage in, garbage out" always applies.

The next section briefly summarizes the conceptual economic framework for benefits analysis. Section 7.3 outlines the effect-by-effect process for benefits analysis, including some general implementation principles. The fourth section defines and describes the types of benefits associated with environmental policies, followed by a review of available economic valuation methods in Section 7.5. This chapter

concludes with specific recommendations for valuing types of benefits that are common to many EPA policies.

7.2 A Conceptual Framework for Benefits Analysis

This section describes the theoretical economic foundation for valuing benefits. The theoretical discussion here serves as a conceptual starting point for benefits estimation—it is not a full and comprehensive treatment of welfare economics. The section includes a discussion of willingness to pay, consumer surplus, and analytical problems arising from the lack of markets for environmental improvements. References are provided for further reading on the specific topics introduced in this section, but useful texts for general reference include Just et al. (1982), Braden and Kolstad (1991), and Freeman (1993). Boardman et al. (1996), Brent (1995) and Hanley and Spash (1993) are useful, general references for benefit-cost analysis.

7.2.1 Welfare Measures: Willingness to Pay and Willingness to Accept Compensation

Economists define benefits by focusing on measures of individual satisfaction or well-being, referred to as measures of welfare or utility. Economic theory assumes that individuals can maintain the same level of utility while trading-off different "bundles" of goods, services, and money. For example, one may be equally satisfied by going fishing or viewing a movie. The tradeoffs individuals make reveal information about the value they place on these goods and services.

The willingness to trade off compensation for goods or services can be measured either as *willingness to pay* (WTP) or as *willingness to accept compensation* (WTA). Economists generally express WTP and WTA in monetary terms. In the case of an environmental policy, willingness to pay is the maximum amount of money an individual would voluntarily exchange to obtain an improvement (or avoid a decrement) in the environmental effects of concern. Conversely, willingness to accept compensation is the least amount of money an individual would accept to forego the improvement (or endure the decrement).[1]

- **WTP and WTA are not necessarily equal.** The amount an individual would be willing to pay to obtain an environmental improvement is not necessarily identical to the amount he or she would be willing to accept to forego the improvement. One reason for this difference is that the starting points of the two measures differ. For environmental improvements, WTP uses the level of utility without the improvement as a reference point. WTA, on the other hand, uses as its reference point the level of utility *with* the improvement. Although these two measures are distinct and sometimes differ in practice, under conventional assumptions economists expect that the difference between them will be small in most cases. This result generally holds as long as the amounts in question are a relatively small proportion of the individual's income. Nonetheless, in the case of environmental goods, some additional considerations modify this general result. Hanemann (1991) shows that while this result holds for price changes, it does not strictly hold for changes in quantity or quality. Also, if a good has no close substitutes, differences in WTP and WTA may be large even if the effect on income is small.

- **WTP and WTA can also be identified with what they imply about property rights**—whether entities have a right to pollute, so the public must pay them not to, or whether the public has a right to a clean environment and must be compensated for pollution. For example, in the case of a policy that would reduce existing pollution levels, the use of WTP measures to value benefits implicitly assumes that the property right rests with the polluting firm.

[1] In the case of environmental improvements, WTP is identified as the *compensating variation* measure of welfare change, while WTA in this case is identified as the *equivalent variation measure*. For environmental decrements, these associations are reversed. For a more detailed treatment of welfare measures that includes these issues see Just et al. (1982), Freeman (1993), and Hanley and Spash (1993).

In practice, WTP is generally used to value benefits because it is often easier to measure and estimate. To simplify the presentation, we use the term "willingness to pay" (or WTP) throughout this chapter to refer to the underlying economic principles behind both WTA and WTP.

Aggregating Individual Willingness to Pay Measures

The benefits of a policy are the sum total of each affected individual's WTP for the policy. Because benefit-cost analysis assesses only the efficiency of policy choices, each individual's WTP must be given the same weight in the summation. This means that no individual or group of individuals is given preferential treatment in assessing the efficiency of the program except to the degree that they are willing to pay for it. As described in Chapter 9, equity assessments and impact analyses can be used to describe the effects of policies on populations of concern.

Altruism

While benefits are generally calculated by summing each individual's WTP for his or her own welfare, there are conditions under which it is appropriate to include altruistic values, or individuals' WTP for the welfare of others. Economic theory concludes that if one cares about a neighbor but respects the neighbor's preferences, and if the neighbor would have to pay for the policy action being analyzed, then altruistic benefits should not be counted in a benefit-cost analysis. The intuition behind this result is that, if one respects the neighbor's preferences, one cares about both the benefits and the costs the neighbor faces. It is therefore inappropriate to add the value one attaches to the neighbor's benefits without considering the cost implications of doing so. Comparing individual benefits and costs in this case is the appropriate decision rule.

Altruistic benefits may be counted either when altruism toward one's neighbor is paternalistic or when one will in fact bear the costs of the project but the neighbor will not. In the first case (paternalistic altruism), one cares about the benefits the neighbor will enjoy, e.g., from a health or safety project, but not about the costs the project will impose on him. An example of the second case would be a project whose costs are borne entirely by the current generation; i.e., the project imposes no costs on future generations. In this case, altruism toward future generations by the current generation could legitimately be counted as a benefit.

7.2.2 Market Goods: Using Consumer Surplus and Demand Curves

Willingness to pay is closely related to the concept of *consumer surplus*, which is both an individual and an aggregate concept. An individual demand curve indicates the maximum amount an individual would be willing to pay to acquire an additional unit of good. These *individual* demand curves can then be aggregated into a *market* demand curve that provides the cumulative WTP for additional units. Consumer surplus is derived from market estimates of how much of the good is demanded in the aggregate at each price and can be easier to estimate than individual WTP.

A market demand curve for a given good or service traces out the amounts that consumers will purchase at different price levels; i.e., their collective WTP for the good or service. Consumer surplus is the excess amount that purchasers are willing to spend on a good or service over and above that required by the market price (i.e., the area under the demand curve but above the price line). This surplus serves as a measure of the social benefits of producing the good. Policies that affect market conditions in ways that decrease prices will generally increase consumer surplus. This increase can be used to measure the benefits of the policy.[2]

The use of demand curves and consumer surplus highlights the importance of assessing how individuals will respond to changes in market conditions. For example, if a policy affects the price or availability of a commodity traded in a market (e.g., if it leads to increases in the commercial fish harvest), multiplying the increased quantity by current prices generally will not provide an accurate

[2] Technically, consumer surplus serves as a precise measure of benefits only if the demand curve represents a *compensated* or *Hicksian*, demand function. However, Willig (1976) shows that *ordinary*, or *Marshallian*, demand curves can often be used to derive an approximate measure of welfare. The difference in these two types of demand curves is that the former holds utility constant, while the latter holds income constant. More background on the theoretic basis for welfare measures can be found in several texts including Freeman (1993), Johansson (1993), Just et al. (1982), and Varian (1992).

measure of benefits. Depending on the elasticity of the demand curve, a one percent price increase may lead to more (or less) than a one percent increase in the quantity demanded, affecting the change in consumer surplus.[3] While not detailed here, supply curves also vary in elasticity and have an analogous effect on producer surplus. Information on the elasticity of the supply and demand curves is needed to estimate benefits in the form of increases in consumer (and/or producer) surplus.[4]

7.2.3 Non-Market Goods

One challenge facing analysts of environmental policies is the lack of a market for most environmental improvements. Because "cleaner air" or "cleaner water" is not normally bought or sold, market data are generally not available for benefit valuation. Economists have therefore developed other methods for eliciting values for these types of effects. These methods rely either on information from the markets for related goods (revealed preference methods) or on direct information on people's preferences (stated preference methods). Individual WTP values estimated in these studies can be aggregated (or an average value multiplied by the total number of affected individuals) to produce an estimate of the total benefit for a good or policy. Section 7.5 provides more information on the economic foundations of specific methods, and Section 7.6 details how these methods have been—or can be—applied in benefits analysis.

7.3 The Benefits Analysis Process

From the perspective of economic theory, an appropriate measure of a policy's benefits is the sum of individual WTP estimates for that policy. While it may be possible in some circumstances to obtain individual WTP estimates for the entirety of a policy decision, in practice, analysts must often use an "effect-by-effect" approach for benefit valuation. This section discusses this approach to benefits analysis, concluding with some general principles to keep in mind when implementing this approach.

7.3.1 A General "Effect-by-Effect" Approach

The most widely used approach for estimating the benefits of a policy option is to evaluate separately the major effects of a given policy and then sum these individual measures to arrive at total benefits. This general approach usually involves describing the physical effects of the pollutants (e.g., various types of damages to human health and ecological systems) and assessing each type of effect separately. In some cases, it may be desirable and feasible to diverge from this approach. For example, contingent valuation or other methods could be used to develop estimates of WTP for the combined effects of the policy change, reducing the need to identify, quantify, and value each effect separately. A comprehensive value estimate for the entire set of effects from a policy change can also be useful as an indication of the upper bound expected from the sum of values developed with the effect-by-effect approach.[5] However, because it is difficult to develop estimates of the total value of the pollution reduction and decision makers are often interested in information on individual benefit categories, an effect-by-effect valuation approach is most often used by EPA in economic analyses of regulations.

The general effect-by-effect approach for assessing the benefits of environmental policies includes three components:

- **Identify potentially affected benefit categories** by developing an inventory of the physical effects that may be averted by the policies.

[3] Elasticity is a measure of relative change. For a given demand curve, price elasticity is defined as the percentage change in quantity demanded divided by the percentage change in price. Where this value is less than one in absolute value, demand is considered to be "inelastic." Elasticity values greater than one (in absolute value) indicate that demand is "elastic."

[4] It is important to keep in mind that elasticity is a local concept. Generally, one can expect the elasticity of supply and demand curves to vary along their respective lengths. This means that elasticities measured at a particular point on these curves may not be appropriate for estimating large changes or changes elsewhere on the curve. In these cases, it may be necessary to characterize the demand and supply functions in the relevant range of prices and quantities.

[5] Randall (1991) presents a framework for comparing total value and "independent valuation and summation" and reviews many issues associated with estimating total values.

- **Quantify significant physical effects** to the extent possible working with managers, risk assessors, ecologists, physical scientists, and other experts.

- **Estimate the values of these effects** using studies that focus on the effects of concern or transferring estimates from studies of similar impacts.

These steps may be implemented using an iterative process. For example, analysts can begin by conducting screening analyses using available data and relatively simple assumptions, then collect additional data and refine the analysis as needed to better inform decision-making.

Each step in this approach is discussed in more detail below, focusing on the actions that are generally undertaken when conducting benefits analyses for typical EPA policies. However, this guidance is intended to be flexible. Analysts will need to determine on a case-by-case basis whether this framework is appropriate for assessing a specific policy, given the effects particular to that policy and the information needed for related decision-making.

Step1: Identify Potentially Affected Benefit Categories

The first step in the benefits assessment is to determine the types of benefits most likely to be associated with the particular policy. Section 4 of this chapter contains a detailed presentation of the categories of benefits typically associated with environmental policies and regulations. To identify benefit categories, analysts should, to the extent feasible, do several things:

- **Develop an initial understanding of policy options of interest** by working with cost analysts and policy makers. Information should also be collected on the likely range of emissions levels associated with the baseline and with implementation of each of the policy options. At the outset of the analysis, the range of options and associated emissions levels considered may be very broad because emissions levels and preferred policy options can change significantly in the course of the policy making process.

- **Research the physical effects of the pollutants** on human health, welfare, and the environment. This can be done by reviewing the literature and, if necessary, meeting with other experts. This step requires considering the transport of the pollutant through the environment along a variety of pathways, including movement through the air, surface water and groundwater, deposition in soils, and ingestion or uptake by plants and animals (including humans). Along these pathways, the pollutant may have detrimental effects on natural resources (e.g., affecting oxygen availability in surface water or reducing crop yields) as well as direct or indirect effects on human health (e.g., affecting cancer incidence through direct inhalation or through ingestion of contaminated food).

- **Consider the potential change in these effects** as a result of possible policy options. If policy options differ only in their level of stringency, then each option may have an impact on all identified physical effects. In other cases, however, some effects may be reduced while others remain unchanged under a specific policy option. Evaluating how physical effects change under each policy option requires evaluation of how the pathways differ in the "post-policy" world.

- **Evaluate which effects are likely to be significant** in the overall benefit analysis according to at least three criteria:

  — whether there are likely to be observable changes in the benefits category when comparing the policy options to each other and to the baseline;

  — whether the benefits category is likely to account for a major proportion of the total benefits of the policy; and

  — whether stakeholders or decision makers are likely to need information on the benefits category, even if its magnitude is relatively small.[6]

The outcome of this initial step in the benefits analysis can be summarized in a list or matrix that describes the physical effects of the pollutant, identifies the benefits categories associated with these effects, and an initial ranking of which effects may be significant enough to warrant further investigation.

Initially, the list of benefit categories may be lengthy and include all effects that reasonably can be associated with

[6] This criteria relates to equity considerations detailed in Chapter 9.

the policy options under consideration. Analysts should preserve and refine this list of benefit categories as the analysis proceeds, and the effects that are not assessed in detail should be discussed qualitatively when presenting analytic results. In some cases, it may not be feasible to assess some of the more significant impacts, either because of insufficient scientific data (e.g., data are lacking on the effects of changes in pollution levels on the benefit category of concern) or because the time or resources needed to assess the effect are high compared to the significance of the benefits category in the decision-making process. These issues should be discussed when presenting the results of the benefits analysis. The discussion should address (1) the criteria used to exclude selected benefit categories from detailed quantitative analysis, (2) the likely magnitude of the non-quantified benefits, and (3) the extent to which these effects are or are not important considerations for the decision-making process.

Step 2: Quantify Significant Physical Effects

The second step is to quantify the physical impacts related to each category. Data are usually needed on the extent, timing, age distribution of the affected population, and severity of the effects. The focus should be on the changes attributable to each policy option in comparison to the baseline. For example, if the risk of lung cancer is one of the effects of concern, data may be needed on the changes in risk associated with each option, the timing of the risk reductions, the age distribution of those experiencing the risk reductions, and the percentage of cases likely to be fatal. If visibility is a concern, data may be needed on the geographical areas affected and the change in visibility levels attributable to each policy option.

- **Work closely with analysts in other fields.** Estimating these impacts is largely, but not completely, the domain of other scientists, including risk assessors, ecologists, and other experts. These experts are generally responsible for evaluating the likely transport of the pollutant through the environment and its potential effects on humans, ecological systems, and manufactured materials under the baseline and each policy option. The principal role of the economist is to communicate with these experts in order to ensure that the information provided is adequate to support the benefits analysis, including information on the uncertainty associated with the estimates of physical impacts. However, economists may also be able to provide insights, information, and analysis on behavioral changes that can affect the results of the risk assessment.

- **Try to match the risk assessment and economic endpoints.** A key consideration in this interaction is that the endpoints quantified and described in the risk assessment match well the effects for which economic valuation is feasible. Effects that are described too broadly or that cannot be associated with economic welfare will limit the ability of the analysis to capture the full range of benefits associated with policy options. It is difficult, for example, to produce an economic measure of the benefits associated with a reduction in the number of persons exposed to a contaminant at a particular level. If, however, the risk assessment can produce an estimate of the reduction in the number and type of adverse health effects from exposure, then the economic valuation exercise is much more feasible. This means that the analyst must be aware of the available economic data and tools when working with risk assessors and other scientists.

- **Describe qualitatively effects that cannot be quantified.** It will not be possible to quantify all of the significant physical impacts for all policies. For example, animal studies may suggest that a contaminant causes severe illnesses in humans, but the data available may not be adequate to determine the number of expected cases associated with different human exposure levels. Likewise, it is often not possible to quantify all the ways in which an ecosystem may change as a result of an environmental policy. In these situations, the effect should be described qualitatively when presenting the results of the benefits analysis. Analysts should also assess the implications of not being able to include this effect in quantitative benefits estimates.

EPA has developed extensive guidance on the assessment of human health and ecological risks and analysts should refer to those documents and the offices responsible for their production and implementation for further guidance.[7] No specific guidance exists for assessing changes in materials damages or amenity effects. Analysts should consult relevant experts and existing literature to determine the "best practices" appropriate for this type of analysis.

Step 3: Estimate the Values of the Effects

Once information on the physical effects of the pollutant is available, the next step is to assess the value of related benefits based on estimates of individual WTP. As discussed earlier, no market exists for many of the types of benefits anticipated from environmental regulation. In most cases, analysts will need to rely upon the results of other methods for estimating economic values. Details on these methods and examples of how they may be applied can be found in Sections 7.5 and 7.6, respectively.

- **Consider using more than one method to estimate benefits.** Different methods often address different subsets of total benefits and the use of multiple methods allows for comparison of alternative measure of value. Double-counting is a significant concern when applying more than one method, however, and any overlap should be noted in presenting the results. In addition, some components of the total value of benefits may not be amenable to valuation and will need to be described in other terms when presenting the analytic results. The discussion of benefit transfer in Section 7.5 describes many of the issues involved in applying values from one study to another situation.

- **Describe the source of estimates and confidence in those sources.** Valuation estimates always contain a degree of uncertainty. Using them in a context other than the one in which they were initially estimated can only increase that uncertainty. If many high-quality studies of the same effect have produced comparable values, analysts can have more confidence in using these estimates in their benefits calculations. Some specific benefit transfer methods described in Section 7.5 provide a systematic manner of combining multiple estimates. In other cases, analysts may have only a single study—or even no directly comparable study—to draw from. In all cases, the presentation of the benefits analysis should clearly describe the sources of any values used, along with some assessment of the confidence associated with those sources.

7.3.2 Implementation Principles

When applying this framework to assess the benefits of specific policies, analysts should keep in mind the following general principles:

- **Focus on key issues.** Resources should be focused on benefit categories that are likely to influence policy decisions. To use time and resources effectively, analysts must weigh the costs of conducting additional analysis against the usefulness of the additional information provided for decision-making. The analysis should devote significant time and resources to carefully assessing those benefits categories that are likely to influence the selection among policy options. In some cases, relatively simple screening analyses may provide adequate information on these benefits. Additional data collection may not be warranted because it is unlikely to lead to significant changes in the conclusions of the analysis. For example, screening using a broad range of values for selected effects may indicate that a policy is clearly worth pursuing and analysts may conclude that any possible refinements to the analysis are likely to simply reinforce this conclusion. In this case, the analyst should discuss the approach taken and note that the benefits estimate may represent a lower bound. Likewise, some categories of benefits may not be assessed either because they are expected to be small or because the costs or time needed to quantify them far exceed the time or resource levels appropriate for analysis of the particular policy.

  Applying this approach to benefits assessment involves first conducting scoping analyses to collect available information on the potential benefits of the policies and using this information to develop

[7] In September 1986, EPA published final risk assessment guidelines for a number of health effects, including *Guidelines for Carcinogen Risk Assessment,* which are currently under revision. Many other risk-related guidelines have been published, revised, and updated since 1986. Recent additions include *Guidelines for Exposure Assessment* (EPA, 1992) and *Guidelines for Reproductive Toxicity Risk Assessment* (EPA, 1996). More information on these and other guidelines, as well as electronic copies of the documents themselves, can be found on the home page of EPA's National Center for Environmental Assessment at http://www.epa.gov/nceawww1/raf/rafguid.htm (accessed 8/28/2000).

preliminary estimates (see, for example, Morgan and Henrion,1990). The results from this initial screening analysis can then be used to inform the early stages of the policy development process and to focus future research on those areas most in need of further assessment. In many cases, it may be useful to use benefits transfer techniques in the initial stages of the analysis, as discussed later in this chapter.

- **Coordinate frequently with others involved in developing the policies.** Ongoing coordination with the analysts responsible for assessing costs and economic impacts, and with the work group considering policy options is crucial to ensure consistency as the policy options and analyses evolve. This coordination should begin in the planning stages of the analysis, and should continue throughout the development process. Successful efforts often involve informal conversations among lead analysts several times each week, supplemented by larger and more formal periodic meetings to report on progress and discuss next steps.

  Coordination will help ensure that the cost and benefit results are comparable and based on consistent baseline and policy assumptions. In addition, information from the cost analysis is often needed for the analysis of benefits and vice versa. For example, if a policy requires firms to install new pollution controls, benefits analysis requires information on the number of facilities likely to install each type of control and the associated reduction in emissions. On the other hand, where a performance standard is being considered, the cost analysis may need data from the risk models in considering which controls are likely to meet the standard.

- **Consider changes in behavior.** The use of an effect-by-effect approach does not necessarily mean that one should simply value benefits by estimating the physical changes attributable to changes in pollution emission levels (e.g., increases in the fish population) then assigning a unit value to these changes (e.g., the price of the fish). Such a limited analysis will be inappropriate in many cases because it leaves out the effects of changes in behavior attributable to changes in environmental quality. For example, increased fish populations may cause commercial prices to drop, in which case consumers may increase their purchases. Commercial fisheries may also respond to changes in pollution levels by altering their production processes. While it may not be possible in practice to capture all of these types of responses in the analysis, those that are likely to be significant should be addressed.

- **Guard against *double-counting* benefits.** If there is significant overlap across the values used for estimating the benefits of different effects, summing values across these effects could substantially overstate expected benefits. For example, property value studies may estimate people's WTP for all perceived effects. This would overlap with values estimated separately for any one of these effects, such as reduced risk, so simply adding these two values to estimate benefits would be inappropriate. Analysts should also take care to ensure that important effects of the policy have not been omitted in the benefits analysis, as this will lead to significant underestimates of total benefits.

- **Explicitly address uncertainty and non-monetized effects.** Benefits assessments for environmental policies often involve significant uncertainty. Sometimes this uncertainty cannot be reduced (or better characterized) given the need to regulate in a timely manner and the resources available for the analysis. These uncertainties should be clearly communicated when presenting the results of the analysis, focusing on the implications for decision-making. For example, if benefits may be significantly overstated due to the conservatism inherent in the risk estimates, then the materials summarizing the analysis should state this explicitly. Guiding principles for addressing and presenting uncertainty are presented in Chapter 5 of this guidance. The relative significance of benefits categories that are not quantified, or quantified but not monetized, should also be described, as discussed in Chapter 10.

7.4 Types of Benefits Associated with Environmental Policies

This section describes the types of benefits that are typically associated with environmental policies. These descriptions are provided with an understanding that it is desirable to quantify and monetize these benefits. Available valuation techniques are described in Section 7.5.

Benefits from environmental policies can be broadly classified into those that directly affect humans and human welfare and those that affect human welfare through systems or processes. The former category includes human health improvements such as reduced mortality rates, decreased incidence of nonfatal cancers, chronic conditions and other illnesses, and reduced adverse reproductive or developmental effects. Improved amenities are another type of benefit experienced directly by humans. Improved taste and odor of tap water resulting from treatment requirements are an example of direct amenity benefits.

Benefits that affect human welfare through systems or processes include reduced materials damages and numerous other effects collectively termed ecological benefits. EPA policies may result in ecological impacts that affect the human use of natural resources (e.g., improving commercial fishing, increasing agricultural yields, enhancing recreational opportunities.) Ecological effects may also provide passive use (or "non-use") benefits that arise from a variety of motives including, for example, one's own utility in knowing that clean resources exist or the desire to preserve clean resources for future generations. In some cases, environmental policies also reduce damages to manufactured materials or improve a resource's aesthetic qualities. Reducing air pollution may decrease damages to building exteriors or improve visibility. Exhibit 7-1 illustrates this categorization scheme

Exhibit 7-1 Examples of Benefit Categories, Service Flows, and Commonly-Used Valuation Methods

Benefit Category	Examples of Service Flows	Commonly-Used Valuation Methods
Human Health		
Mortality Risks	Reduced risk of • Cancer fatality • Acute fatality	• Averting behaviors • Hedonics • Stated preference
Morbidity Risks	Reduced risk of • Cancer • Asthma • Nausea	• Averting behaviors • Cost of illness • Hedonics • Stated preference
Amenities	• Taste • Odor • Visibility	• Averting behaviors • Hedonics • Stated preference
Ecological Benefits		
Market: products	Provision of • Food • Market • Fuel • Timber • Fiber • Fur, eather	• Market
Non-market: recreation and aesthetics	Provision of • Recreational opportunities, e.g., viewing, fishing, boating, swimming, hiking • Scenic vistas	• Production function • Averting behaviors • Hedonics • Recreation demand • Stated preference
Indirect: ecosystem services	• Climate moderation • Pollination by wild species • Flood moderation • Biodiversity, genetic library • Groundwater recharge • Water filtration • Sediment trapping • Soil fertilization • Soil retention • Pest control • Nutrient cycling	• Production function • Averting behaviors • Stated preference
Non-use: existence and bequest values	No associated services	• Stated preference
Materials Damage	--	• Averting behaviors • Market

and suggests commonly-used techniques for estimating their values, although the list is not exhaustive.[8] A detailed discussion of valuation techniques is presented in the next section of this chapter. The remainder of this section describes each of these categories briefly and notes issues associated with quantification.

7.4.1 Human Health: Mortality Risks

Some EPA policies are designed to decrease the risks of contracting potentially fatal health effects, such as some cancers. Reducing these risks of premature fatality provides welfare increases to those individuals affected by the policy. It is important to keep in mind that policies generally provide marginal changes in relatively small risks. That is, most policies do not provide assurance that one will not prematurely die of environmental causes, they only marginally reduce the probability of such an event.

- **Reduced mortality risks are often measured in terms of "statistical lives."** This measure is the aggregation of many small risks over an exposed population. Suppose, for example, that a policy affects 100,000 people and reduces the risk of premature mortality by one in 10,000 for each individual. Summing these individual risk reductions across the entire affected population results in the policy saving 10 statistical lives. It is unknown who these ten people might be—everyone faces some risk of being affected—but the policy can be expected to prevent premature fatality for 10 individuals in the population.

- **Alternative measurements may include "statistical life years."** A somewhat more refined approach to measuring reduced mortality risks includes the degree of life extension in the estimate. This is usually done by looking not just at the reduced probability of a premature fatality, but also at the expected life span of those enjoying the risk reduction. A risk reduction of one in 10,000 experienced by a population of 100,000 people with an expected remaining life span of 50 years each, for example, would save 10 "statistical lives" or 500 "statistical life years." Measuring mortality risk reduction in terms of statistical life years provides more information about the expected benefits of a policy, but requires risk estimates for specific age groups.[9] Often these risk estimates are not available.

7.4.2 Human Health: Morbidity Effects

This benefits category consists of reductions in the risk of non-fatal health effects ranging from mild illnesses such as headache and nausea to very serious illnesses such as cancer. A complete list of morbidity effects is beyond the scope of this document, but the presumption for all of these effects is that the illness will not generally result in premature fatality.

- **Morbidity effects can generally be characterized by their duration and severity.** For duration of illness, the primary distinction is between acute effects and chronic effects. Acute effects are discrete episodes usually lasting only a few days, while chronic effects last much longer and are generally associated with long-term illness. Severity defines the degree of impairment associated with the illness and may be measured in terms of "restricted activity days," "bed disability days," or "lost work days."[10] Severity may also be described in terms of health state indices that may combine multiple dimensions of health into a single quantity, or index. The difference in the index

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[8] This classification scheme is offered here to facilitate discussion in this document. It is similar in many respects to one offered in Freeman (1993), but other researchers have offered alternatives. Freeman (1993) describes some general characteristics of these alternatives. The list of techniques for each benefit category is not intended to be comprehensive or exclusive

[9] Additional refinements to account for quality of life or health status are often employed in the public health and health economics. Existing measures include "quality adjusted life years" (QALYs) and "disability adjusted life years." These measures have not been fully integrated with the literature on benefits analysis for environmental policies. More information on QALYs can be found in Gold et al. (1996) and additional information on DALYs can be found in Murray (1994).

[10] As Cropper and Freeman (1991) note, these descriptions are essentially characterizations of a behavioral response to the illness. Lost workdays, for example, in some cases requires a decision on an individual's part not to go to work due to illness. Such a response may depend upon various socioeconomic factors as well as the physical effect of the illness.

value reflects the relative difference in disutility associated with symptoms or illnesses.[11] Morbidity effects can be further characterized by the set of symptoms associated with an illness.

- **Morbidity effects are usually quantified in terms of the number of expected cases of a particular illness.** Given the risks faced by each individual and the number of persons exposed to this risk, an estimate of "statistical cases" can be defined analogously to "statistical lives" described above. Alternatively, morbidity effects may be described according to the expected number and duration of particular symptoms associated with the illness. These estimates of "symptom days" may be used in benefits analysis when appropriate estimates of economic value are available.

7.4.3 Amenities

Direct amenities include improvements in aesthetic attributes associated with environmental commodities. This includes improvements in taste, odor, appearance, or visibility. In short, these benefits are determined by how the senses are affected and how individual's welfare is changed as a result. This class of benefits is unique in that the focus is on the sensory experience and not on a physical or material effect.

Despite this conceptual distinction, aesthetic benefits are often intertwined with other benefit categories such as health and recreation. A policy that improves air quality, for example, might simultaneously improve visibility and reduce mortality risks associated with airborne contaminants. New treatments for drinking water might reduce health risks as well as alter the taste and odor of tap water. These relationships may make it extremely difficult to separately quantify and value improvements in aesthetic qualities.

Many types of policies can be expected to have some impact on these kinds of amenities and they may be the focus of a given policy. Amenity improvements may be major component of total expected benefits. Improved visibility from better air quality is one example that has been the subject of several empirical studies.[12]

7.4.4 Ecological Benefits

Ecosystems provide services that benefit humans. For example, a freshwater lake may provide recreational and boating sites; a wetland provides a service by being a breeding ground for fish and fowl. Although ecosystems have a profound impact upon human well-being, the quantitative assessment of ecological benefits presents a formidable challenge for several reasons. First, natural systems are inherently complex. The many services they provide and how they provide them may be poorly understood by even the scientific community. Second, ecological risks vary widely in terms of persistence (e.g., eutrophication versus species extinction), geographic extent (e.g., toxic contamination versus global climate change), and the degree to which the overall threat can be predicted (e.g., effects of ozone on crops versus developmental and behavioral effects of chemicals on wildlife populations). Third, many of the less tangible benefits are not readily amenable to monetary valuation.

Section 7.3 discussed generally the three steps involved in assessing the benefits of environmental policies. However, some issues associated with identifying and quantifying ecological benefits are particularly complex and warrant more detailed treatment.

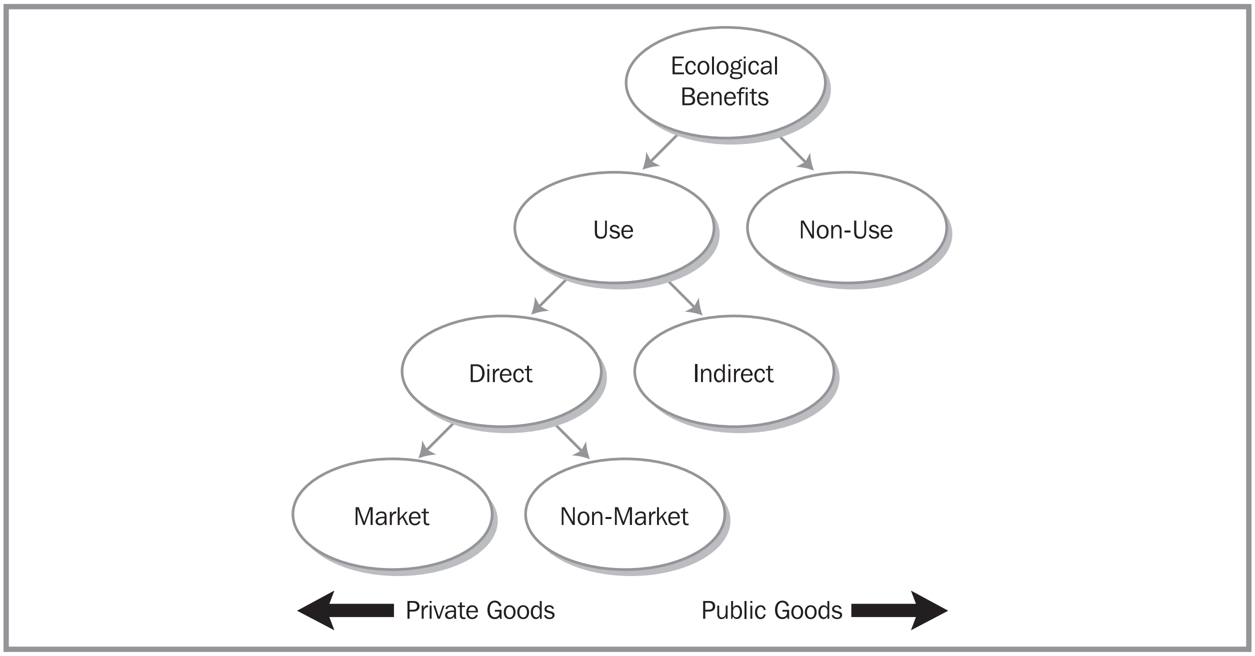
Identifying Ecological Benefits

The first step in assessing ecological benefits is to identify those relevant to policy options under consideration, focusing on service flows that are likely to change as a consequence of guidance or regulatory action. In general, these ecological benefits may be thought of as flows of services from the natural asset in question. These can be categorized by how directly they are experienced and where they fall along a private good/public good continuum. Exhibit 7-2 illustrates how the categories relate to one another. Not only is it useful as a conceptual tool, this

[11] These indices may be constructed in a number of ways, but consistency with welfare economics requires affected individuals to define these relative tradeoffs for themselves rather than having them determined by health experts. Several economic analyses have employed some form of health state index. Recent examples include Desvousges et al. (1998) and Magat et al. (1996).

[12] Examples of these studies include Rae (1983), Johnson et. al. (1983), Schulze et al. (1983), Chestnut and Rowe (1990), Crocker and Shogren (1991), and McClelland et al. (1993).

Exhibit 7-2 Ecological Benefits Classification Scheme



categorization helps direct analysts to suitable valuation methods.[13]

- **Market benefits:** Direct market benefits are some of the most readily identified service flows provided by ecosystems. These typically relate to primary products that can be bought and sold competitively as factors of production or final consumption products. Although they may be managed to a high degree, agricultural systems are nevertheless predicated on ecological processes. As a consequence, increased productivity of farmland and rangeland may provide significant market benefits. Other products include commercial fish species and timber. When access is controlled and appropriate user charges levied, recreational opportunities may also be considered direct, market benefits.

- **Non-market benefits:** Recreational opportunities and aesthetic qualities provided by ecosystems are also experienced directly by individuals, albeit in a non-market setting. Non-market benefits include both consumptive uses (e.g., recreational fishing and hunting) and non-consumptive uses (e.g., scenic vistas, wildlife viewing, hiking, and boating). These services are typically provided by natural assets held in common (e.g., public lands). They have public goods characteristics—since access is not or cannot be controlled, consumption is not exclusive. On the other hand, like private goods, they are rival in consumption because excessive use by others (i.e., congestion) tends to diminish one's own enjoyment of these services.

- **Indirect benefits:** Ecosystem services that do not directly provide some good or opportunity to individuals may be valued because they support off-site ecological resources or maintain the biological and biochemical processes required for life support. These indirect benefits tend to be purely public in nature—access to or use of the service is not exclusive and a virtually unlimited number of individuals can share in the benefits without reducing the average benefit accruing to each. Each type of ecosystem provides various indirect benefits. Wetlands recharge groundwater, mitigate flooding, and trap sediments. Forests sequester carbon, anchor soil, and maintain microclimates. Estuaries protect adolescent fish. Terrestrial ecosystems provide habitat for natural pollinators. All of these systems support biodiversity.

- **Non-use benefits:** Some benefits are not associated with any direct use by either individuals or mankind.

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[13] A more detailed discussion of these concepts is also found in EPA's Conceptual Framework for Assessing Ecological Costs or Benefits (EPA, 1999b). A draft is available at http://intranet.epa.gov/oerrinet/ecoweb/index2.htm (accessed 8/29/2000).

Rather, they result because individuals might value an ecological resource without using or even intending to use it. Non-use values, also referred to as passive use values, are those associated with the knowledge the resource exists in an improved state, bequest values for future generations and altruistic values for others' enjoyment of the resource. An individual's commitment to environmental stewardship may also be the source of existence value. The commitment of some groups to particular animals or ecosystems provides an example of this.[14]

Quantifying Ecological Risk

The second step in the analysis of ecological benefits is to estimate the physical effects of each policy option, comparing the flow of services with and without the policy. It falls upon ecologists and environmental toxicologists to conduct the ecological risk assessments to estimate the expected adverse ecological effect of a particular stressor.[15] Ecological risk assessments can be either narrow in scope, with inquiry limited to a single species or population (e.g., the effect of chemical exposure on an endangered bird species) or focus broadly on an entire ecosystem. Further information on ecological risk assessment can be found in *Ecological Risk Assessment Guidelines* (EPA, 1998).

The results of an ecological risk assessment generally include the effect's magnitude (expressed in such metrics as hazard quotients or percent change in population), duration, spatial distribution, and time period of recovery. The analysis of ecological risks may be highly uncertain. Limited availability of data and models, and imperfect understanding of key issues, hampers our ability to describe ecological effects.

7.4.5 Reduced Materials Damages

The materials damages benefit category includes welfare impacts that arise from changes in the provision of service flows from the "material" environment. The "material" environment is distinguished from the natural environment discussed in the ecological benefits section and includes constructed or highly-managed physical systems. Changes in the stock and quality of these material environmental resources are assessed in a similar fashion to their natural environment counterparts. Analytically, benefits assessment for materials improvements parallels that for managed ecosystems such as agriculture or forestry, with most benefits arising from direct, market effects or use values. For example, effects from changes in air quality on the provision of the service flows from physical resources such as buildings, bridges, or roads are handled in a similar fashion to the effects from changes in air quality on crops or commercial timber stocks. The most common empirical applications involve air pollution damages and the soiling of structures and other property.

7.5 Methods for Benefits Valuation

Economists have developed a number of methodologies to measure the benefits of environmental improvements.

- **Market methods** can be used when direct markets for environmental goods and services exist. The benefits of a change in quantity of a good are estimated using data on these market transactions. By knowing how the good was bought and sold, economists can infer directly how people appear to value that good.

  Unfortunately, direct markets for environmental goods and services do not often exist. In the absence of these markets, environmental and natural resource economists must rely upon alternative methodologies to measure the benefits of environmental improvements.

- **Revealed preference methods** (or *indirect approaches*) allow economists to infer the value placed on environmental goods using data on actual choices made by individuals in related markets. Revealed preference methods include recreational demand models, hedonic wage and hedonic property models, and averting behavior models.

[14] Even though it does not involve use, non-use value still falls under the rubric of welfare economics. It emanates from human interest, alone, and does not encompass any rights or ethics-based justification for preservation (see Kopp, 1992 and Mazzotta and Kline, 1995).

[15] Other types of frameworks for ecological assessment include injury assessments undertaken as part of natural resource damage assessments (IEC, 1995 and Huguenin et al., 1996) and environmental assessments undertaken to meet the requirements of NEPA.

- **Stated preference methods** (or *direct approaches*) allow economists to estimate the value placed on environmental goods using data on hypothetical choices made by individuals responding to a survey. Stated preference methods include contingent valuation, conjoint analysis, and contingent ranking.

Specific approaches that fall under these two broad categories are presented below. This presentation includes an overview of each method, a description of its general application to environmental benefits assessment, and a discussion of issues involved in interpreting and understanding studies using the method. This information is primarily designed to help analysts evaluate existing studies being considered for benefit transfer, but it can also assist analysts in assessing the feasibility of employing these methods. The discussion below concludes with a separate overview of benefit transfer methodology in general. It is important to keep in mind that research on all of these methods is ongoing, sometimes at a rapid pace. The limitations and qualifications described here are meant to characterize the state of the science at the time these guidelines are published. Analysts should consult additional resources as they become available.

7.5.1 Market Methods

Economic Foundation of Market Methods

Market methods are used to value environmental goods and services that are directly traded as market commodities. Market methods are used, for example, to examine the effects of air quality improvements on agriculture and commercial timber industries and the effects of water quality improvements on commercial fisheries.

Market methods apply when environmental goods are factor inputs. Changes in the quality or stock of an environmental good can affect production costs, which can then alter the price and quantity of output and the returns to other factor inputs. In turn, these market responses affect the decisions and welfare of consumers and producers. Changes in the prices of marketed goods consumers face and changes in the income of the owners of the factor inputs reveal information about the welfare of consumers and producers. For example, the benefits of an environmental improvement are often realized as increases in consumer and producer surplus that arise from lower costs and prices and increases in the quantity of the marketed good. For more detailed discussion of the economic foundation of market methods, see Just et al.(1982) or Freeman (1993).

General Application to Benefits Assessment

When applying market methods to assess the benefits of environmental improvements, two types of market responses are important: the impacts of the environmental change on the relevant marketed good (e.g., factor) and the response of producers and consumers to this change. When examining these responses, it is important to consider the range of market responses available to producers and consumers. Overlooking market adjustments can bias benefits assessment. For instance, the damage function approach, which derives benefits by applying a unit price to a physical measure of damage or loss, ignores consumer responses to market adjustments.[16] Measures of price-elasticities, cross-price elasticities, and substitution possibilities indicate the extent to which market adjustments are likely to occur.

In practice, characterizing the market response to a change in environmental quality can be difficult. Two techniques that rely on observations of direct market behavior, cost and production function approaches, facilitate the measurement of consumer and producer surplus changes, but one must assume optimizing behavior on the part of producers and consumers. A different approach for benefits assessment is to use optimization models that simulate behavior. All three of these approaches require considerable information and data on the relevant market participants.

Benefits estimation using market methods varies with the types of markets affected by the environmental improvement. The nature of firms affected on the producer side (e.g., single-product firms or multi-product firms), the market structure (e.g., vertically linked markets), and the presence of market distortions (e.g., monopoly power, price supports) influence the complexity of benefits assessment. Freeman (1993) singles out two cases where

[16] Although the damage function approach does not account for market adjustments, it may be a useful screening tool when time and resources are limited.

benefits assessment is relatively straightforward. The first case is one in which the environmental good or quality is a perfect substitute for another input. Here, the benefits of an environmental improvement can be calculated by estimating the reduction in input costs caused by substituting away from the other input, as long as the change in total costs does not affect marginal costs or output. The second case is where observable market data (e.g., cost, demand, and market structure) imply that benefits from an environmental improvement will accrue to owners of fixed factors. Here, benefits can take the form of increased productivity and are realized as profit or quasi-rents. One such case would be where the producer affected by the environmental improvement is small relative to the market, and variable prices for factors and products are not affected by the environmental improvement.

Empirical applications of market methods are diverse. Among other topics, the empirical literature has addressed the effects of air quality changes on agriculture and commercial timber industries. It has also assessed the effects of water quality changes on water supply treatment costs and on the production costs of industry processors, irrigation operations, and commercial fisheries. Refer to Adams et al. (1986), Kopp and Krupnick (1987), Taylor (1993), and EPA (1997) for empirical examples.

Considerations in evaluating and understanding market studies

Issues to consider when interpreting the results of market studies include:

- **Data requirements and implications:** Employing market methods requires information on the effect of the environmental resource on production costs, supply conditions for output, demand curve for final good, and factor supplies.
- **The model for estimation:** Data availability plays a large role in the selection of a modeling approach and the structure of the model. Production function, cost function, and simulation optimization models are all options for understanding the market response to environmental improvements.

7.5.2 Revealed Preference Methods

In the absence of market data on the value of environmental improvements, WTP may be estimated by looking at related goods that are traded in markets. Methods that employ this general approach are referred to as "revealed preference" methods because people's behavior in associated markets reveals the value they place on the environmental improvements. For example, if pollution levels affect the use of a lake for recreational fishing, individual WTP to travel to a substitute site can be used to estimate the value of averting the damages to the lake of concern. Four distinct revealed preference methods have been widely used by economists: recreation demand models (including travel cost and discrete choice models), hedonic pricing models, averting behavior models, and cost-of-illness studies.

7.5.2.1 Recreation Demand Models

Improvements in environmental quality may enhance recreation opportunities at one or more sites in a region. For example, policies that control the level of toxics in surface water bodies might result in a reduction in the number of lakes and streams subject to fish consumption advisories, thereby enhancing recreational angling opportunities. Recreation improvements constitute a potentially large class of environmental benefits, but measurement of these values is complicated by the fact that access to recreation activities are only partially regulated by observable market mechanisms. Recreation demand models, including the travel cost model, the random utility model (RUM), and other approaches, may be used to assess nonmarket benefits associated with recreation activities.

Economic Foundation of Recreation Demand Models

Recreation demand models focus on the choice of trips or visits to sites for recreational purposes. The basic trade off to be considered is between the satisfaction gained from participating in an activity at a site and the value of money and time given up. The fundamental assumption is that people may weigh the money and time costs of travel to a site in the same way as an admission fee. Thus, by examining the patterns of travel to particular sites, one may infer how individuals value the site or particular aspects of the site such as environmental quality.

As with other economic studies, recreation demand models rely on individual perceptions. While it is possible to value changes in environmental quality that have an obvious effect on popular recreation activities, recreation demand methods may not be appropriate for valuing changes in environmental quality that are difficult for people to observe or only indirectly affect well-known species.

- **Travel cost models:** The simplest recreation demand model involves trips to a single site. User surveys provide data on visitors and trip origins and the data is organized by distance to the site. Generally, an inverse relation between distance traveled and the number of visits emerges. The distance variable may be converted to cost by including factors for the dollar per mile cost of vehicle travel as well as the cost of travel times and, the relationship among the variables may be interpreted as a demand function, with the number of trips from a particular area as a function of the travel costs of reaching the site.

  The single-site travel cost model may be extended to multiple sites, usually by estimating a system of demand equations, with the number of trips to a given site taken to be a function of the cost of visiting that site as well as the costs of visiting other available sites. A number of extensions to the simple travel cost model are described in Freeman (1993).

  Travel cost models are most appropriate for estimating changes in the number of trips over a given period of time, also known as participation. They are limited, however, in their ability to model the recreationist's choice among competing sites. A separate but related body of literature has developed around models that directly address the decision of "where to go" and estimate welfare changes associated with this alternative theoretical framework.

- **Discrete choice models:** For analyses focusing on the role of environmental quality variables, changes in social welfare may best be estimated through discrete choice models (also referred to as RUMs). Discrete choice models focus on the decision to recreate at a specific site as compared to alternative substitute sites. The model considers travel cost and environmental quality variables associated with all competing sites. Detailed treatments of the discrete choice model include Bockstael et al. (1986) and Bockstael et al. (1991).

  Although well suited for analyzing welfare effects of changes in site quality per visit, the discrete choice model is less useful for predicting the number of trips over a period and measuring seasonal welfare changes. Most recreation demand studies use either variations on the discrete choice model or combinations of travel cost and discrete choice approaches to estimate changes in social welfare.

Considerations in Evaluating and Understanding Recreation Demand Studies

There are several issues that must be confronted in a recreation demand model:

- **Definition of a site:** Ideally, one could estimate a recreation demand model in which sites are defined as specific points, such as launch ramps, campsites, etc., but the data requirements of detailed models are large. Similarly, for a given site, the range of alternative sites may vary by individual. Ultimately, every recreation demand study strikes a compromise in defining sites, balancing data needs and availability, costs, and time.

- **Opportunity cost of time:** Part of the cost of taking a recreation trip is the value of recreation time, which varies with respondent's income and work schedules. Recreation demand models typically use some fraction of the wage rate in calculating travel costs, but the tradeoffs between work hours and leisure time involve complex theoretical and methodological issues. Furthermore, it is presupposed that travel time detracts from the overall satisfaction of a recreation trip, but this assumption may not always hold. Other time-related issues include the treatment of on-site time, which varies from case to case but is often ignored altogether.

- **Multiple site or multipurpose trips:** Recreation demand models assume that the particular recreation activity being studied is the sole purpose for a given trip. Visits to multiple sites or multipurpose trips confound attempts to measure social welfare changes.

7.5.2.2 Hedonic Wage Studies (Wage-Risk studies)

Hedonic wage studies draw on the framework of hedonic pricing methods. This section describes the hedonic wage method, but first provides some general background on hedonic pricing methods in general. Property value studies, another large area of research based upon this framework, is described in Section 7.5.2.3.

Background on Hedonic Pricing Methods in General

Hedonic pricing methods apply to heterogeneous goods and services. Heterogeneous goods and services consist of "bundles" of attributes and are differentiated from each other by the quantity and quality of these attributes. Job opportunities, housing units, computers, and cars are common examples of heterogeneous goods. Hedonic pricing methods explain variations in price using information on attributes. For example, determinants of wages are expected to include worker characteristics (e.g., level of education, tenure, age) and job characteristics (e.g., risk of fatal injury). Determinants of housing prices may include structural attributes (e.g., number of bedrooms and age of house), neighborhood attributes (e.g., population demographics, crime, and school quality), and environmental attributes (e.g., air quality and proximity to hazardous waste sites).

The economic theory underlying hedonic pricing methods extends from a model of market equilibrium, where suppliers and demanders of heterogeneous goods interact under conditions of perfect information and zero transactions costs. Consumers derive utility from the attributes of the heterogeneous goods and adjust purchases in response to differences in these attributes. Producers or sellers of goods and services incur costs that vary with the range of attributes offered. An equilibrium price schedule develops from the market interactions of consumers and suppliers. The foundation of the hedonic pricing method as it relates to job opportunities is analogous, with workers and employers interacting in the labor market. The equilibrium price schedule is termed the hedonic price function and forms the basis for benefits assessment using hedonic pricing methods. Rosen (1974) is the seminal article on the economic theory of hedonic methods.

Empirical hedonic pricing research typically concentrates on the hedonic price function and the decisions of consumers or workers. The hedonic price function is approximated by regressing price on measures of attributes and the estimated coefficients represent the marginal WTP for the associated attribute. Applications of hedonic methods to labor wages and property values have been used to characterize the benefits of environmental improvements.[17] These are known as hedonic wage studies and hedonic property value studies, respectively. Each is considered separately below.

Economic Foundation of Hedonic Wage Studies (Wage-Risk Studies)

Hedonic wage studies, sometimes known as wage-risk or compensating wage studies, are based on the premise that individuals make tradeoffs between higher wages and increased occupational risks of death or injury. Essentially, higher risk jobs are expected to pay higher wages, all else held constant. Hedonic wage studies use statistical regression and data from labor markets to isolate the increment in wages associated with higher job risks. The outcome of these models is an estimated value of small changes in mortality risks. Some models also attempt to estimate the value of small changes in morbidity, or non-fatal risks.

The key to an effective hedonic wage study lies in separating the portion of compensation associated with occupational health risks from other job characteristics, including supervisory responsibility, job security, and similar factors. The wage rate is also affected by the industry in which the individual is employed, characteristics of the location and the personal characteristics of the workers (e.g., age, education, experience). All of these data are needed to disentangle the effects of worker characteristics from those of job attributes in determining wages paid.

In hedonic wage studies, workers' perceptions of risk levels across jobs are assumed to match actual risk levels. If perceived risks do not match actual risks faced by the workers, then the resulting estimates of compensation required to accept additional risk will be biased. Most

[17] Palmquist (1991) and Freeman (1993) contain current discussions of the use of hedonic methods for characterizing the demand for environmental quality and benefits assessment. See Palmquist (1982) for a discussion of a benefits assessment method related to hedonic property value studies known as repeat sales analysis.

analysts believe that this potential bias is small, but others argue that workers generally underestimate on-the-job risks. If the latter is true, hedonic wage studies will understate the additional compensation required for bearing risks. Some studies attempt to account for workers' perceived risks, but the results of these studies are not markedly different from those that do not.

Another assumption employed in hedonic wage studies is the existence of perfect labor markets in which workers are freely mobile and there is perfect information about jobs and job risks. Hedonic wage studies will not produce accurate estimates of the wage-risk tradeoff in imperfect markets where workers are unable to move freely between jobs or in which only union members have sufficient information and market power to receive higher wages for higher risk jobs. Since in reality labor markets are somewhat imperfect, many studies attempt to control for union membership and similar factors that might influence wage rates.

Hedonic wage models are limited to estimating values for relatively small risk changes. The observed wage and the estimated increment, or "premium," to accept higher risks represents the market equilibrium price for the entire set of workers in the study. This estimate is not necessarily the value that any particular worker would require to accept a risk increase, but for small changes in risk, it is very close.

A thorough treatment of the hedonic wage model that includes all of these considerations can be found in Viscusi (1992, 1993).

General Application of Hedonic Wage Studies to Benefits Assessment

Because they are narrowly focused on labor market tradeoffs, hedonic wage studies are not generally well-suited to measure the benefits of environmental regulation directly. That is, it is not usually feasible to perform a hedonic wage study to estimate the benefits that would accrue from a specific environmental policy action. Nonetheless, these studies have yielded consistent estimates of how groups of workers appear to value small risk changes. Environmental benefits assessments can draw upon these studies to estimate the value of reductions in environmental mortality risks.[18] Such an application is essentially an exercise in benefits transfer, which is described in greater detail later in this chapter.

Analysts should be aware that, although hedonic wage studies currently provide the most reliable and consistent estimates of the value of mortality risks, there are important differences in the types of risks captured in an hedonic wage study and the types of risks that are affected by environmental regulation. For instance, hedonic wage studies tend to focus on accidental deaths occurring among prime-aged males while deaths associated with environmental risk often occur among the elderly and may involve an extended latency period. Furthermore, elevated risks in hedonic wage studies are voluntarily accepted while environmental risks are often involuntarily borne. The nature and importance of these and other differences are detailed in Section 6 of this chapter.

Estimates of the value of changes in fatal risks are generally more relevant for environmental benefits assessment than are those for job-related non-fatal injuries. This is because these injuries are usually quite different from the non-fatal health risks associated with environmental policy actions.

Hedonic wage models have also used wage differentials across geographic areas to estimate values for environmental quality differences.[19] Theoretically, jobs in areas with poor environmental quality should pay less than identical jobs in areas with high environmental quality, again holding all else equal. There are a number of difficulties with employing hedonic wage models in this manner, including integrating wage and housing choices, and the need to assess intra-city variation in amenities. The majority of hedonic wage studies relevant for most EPA policies have focused on estimating values for health risks.

Considerations in Evaluating and Understanding Hedonic Wage Studies

- **Data requirements and implications:** Hedonic wage studies require large sets of data on labor

[18] Values of mortality risk have also been estimated using hedonic studies of automobile prices. While these studies produce values of life in a similar range, most environmental risk assessments rely on hedonic wage studies. For information on the automobile price literature, see Dreyfus and Viscusi (1996) and Viscusi (1992).

[19] This should not be confused with attempts to control for wage differentials across broad regions found in most existing wage-risk studies.

market behavior. Data on worker and job characteristics are generally collected using survey techniques. Risk information, however, is frequently retrieved from published sources reported at the occupation or industry level. These risk measures are then matched to the worker in the sample using information provided by the respondent on his or her job. The risk data used in most studies, however, are not complete. For example, although accidental, on-the-job (and almost immediate) deaths are generally reported, occupational diseases such as cancer are not accurately captured in most data.

The estimated wage-risk tradeoff can vary considerably across data sets and across methodologies. In particular, studies that draw upon data from high-risk jobs will generally provide valuations of risk that are lower than those that rely upon data from lower-risk jobs. This is due to a sample selection problem. Study results reflect the value to the sample population. High risk jobs tend to attract those who are less averse to taking risks and therefore require less compensation to face them.

- **Controlling for other risks:** If the study seeks to estimate values for fatal risks, it is important that the study control adequately for non-fatal risks in order to obtain an unbiased wage-risk estimate for mortality. Conversely, when values for non-fatal risk are being estimated, mortality risks should be considered in the wage-risk equation.
- **The scope of the risk measures:** Some labor market studies use actuarial data to determine the risk levels faced by workers. However, these data are not limited to occupational risks. They include all types of fatality risks faced by the individual both on and off the job. The degree to which these risks are correlated with job risks is unclear, but they would not be reflected in job-related compensation. These studies generally should be excluded from use in policy analysis since this problem will cause the tradeoffs they estimate to be biased downwards by an unknown amount.
- **The model for estimation:** Some labor market studies attempt to determine the value of a life year and the implicit discount rate workers apply to this value. While attractive in theory, the complexity of the structural models used in these studies leads to less robust estimates of the value of risk reduction than studies using conventional wage-risk estimation procedures. Such studies should be viewed as less reliable for use in valuing lifesaving programs.

7.5.2.3 Hedonic Property Value Studies

Hedonic property value studies are applications of the hedonic pricing method. The introduction to Section 7.5.2.2 (Hedonic Wage Studies) provides background on hedonic pricing methods in general.

Economic Foundation of Property Value Studies

Hedonic property value studies assert that individuals perceive housing units as bundles of attributes and derive different levels of utility from different combinations of these attributes. When transaction decisions are made, individuals make tradeoffs between money and attributes. These tradeoffs reveal the marginal values of these attributes and are central to hedonic property value studies. Hedonic property value studies use statistical regression methods and data from real estate markets to examine the increments in property values associated with different attributes.[20]

Structural attributes (e.g., number of bedrooms and age of house), neighborhood attributes (e.g., population demographics, crime, and school quality), and environmental attributes (e.g., air quality and proximity to hazardous waste sites) may influence property values. When assessing an environmental improvement, it is essential to separate the effect of the relevant environmental attribute on the price of a housing unit from the effects of other attributes. While deriving measures of marginal WTP using hedonic methods is straightforward, estimating measures of WTP for substantial or discrete (non-marginal) improvements in environmental quality is difficult. The use of hedonic property value studies for benefits assessment rests on careful interpretation of the hedonic price function and its relevance to the policy scenario being considered. Bartik

[20] To simplify the discussion, housing units are consistently used as examples. Hedonic property value studies are also completed on vacant land parcels.

(1988b) and Palmquist (1991, 1988) provide excellent, detailed discussions of benefits assessment using hedonic methods.

When using hedonic property value studies for benefits assessment, the measurement of the environmental attribute is central to the analysis. If the measurement of the environmental attribute does not match individuals' perceptions, then the results of the analysis may be biased.

The hedonic price function for housing units represents an equilibrium that results from the interaction of suppliers and demanders of housing in a market with full information. When this assumption is not met, the results of an hedonic analysis will not provide an exact representation of the tradeoffs individuals make across housing attributes and the marginal values associated with these different attributes.

General Application of Hedonic Property Value Studies to Benefits Assessment

Benefits assessment applications of hedonic property value studies focus on the relationship between property values and environmental attributes such as air quality, water quality, proximity to hazardous waste sites, and landscape characteristics. Hedonic property value studies are not widely used in environmental benefits assessments because of the limited transferability of hedonic results and the difficulties of using hedonic methods to describe the benefits associated with discrete (non-marginal) environmental improvements.

Using data on a sample of transactions, price is regressed on measures of the observable attributes and an hedonic price function is estimated. The coefficient on the environmental attribute reveals the marginal WTP for that attribute. Therefore, if the policy scenario considered results in a marginal environmental improvement, the estimated hedonic is well-suited to measure benefits. However, if the policy scenario considered results in a discrete improvement that affects numerous properties, additional information on preferences and the hedonic price function is required for assessing the true benefits of the environmental improvement.[21] When larger changes in environmental quality are considered, the analytical requirements increase because the hedonic price function and the level of utility of individuals may change.

The hedonic price function does not typically provide general information on individuals' WTP for the different attributes. Methods for identifying demand (or WTP) functions for the different attributes (e.g., using data from multiple markets or imposing assumptions about the functional form of the hedonic and/or the utility functions of individuals) exist, but they often rely on restrictive assumptions.[22] Furthermore, identifying WTP functions does not ensure the ability to measure the welfare gain from a discrete environmental improvement because markets intervene and prices change. As a result of these difficulties, approximations of welfare gains based on the hedonic price function are sometimes employed to assess benefits. See Palmquist (1991, 1988) and Bartik (1988b) for a detailed discussion of benefits assessment using hedonic methods, including guidance on developing lower and upper bound measures of benefits.

Considerations in Evaluating and Understanding Property Value Studies

- **Data requirements and implications:** Property value studies require large amounts of disaggregated data. Market transaction prices on individual parcels or housing units are preferred to aggregated data such as census tract information on average housing units because aggregation problems can be avoided. Data on attributes may include housing characteristics, sale dates, neighborhood amenities such as schools and parks, neighborhood demographic characteristics such as income, age, and race, and environmental quality.
- **Errors in variables:** Problems may arise from error in measuring prices (aggregated data) and errors in measuring product characteristics (particularly those related to the neighborhood and the environment). In addition, omitted variable bias problems may occur if relevant data are not available.
- **The measurement of environmental attributes:** The measurement of the environmental attribute included in the hedonic price function is central to

[21] There are cases when the hedonic price function can alone be used to measure welfare effects from a non-marginal change in the environmental attribute. For example, this holds if few properties are affected and the hedonic price function is not expected to shift.

[22] See Palmquist (1991) for a detailed discussion of identification issues.

the assessment of benefits. Researchers often use information available from the scientific community such as air or water quality monitoring data and then must determine how to assign the data to the individual houses in the data set. However, there may be differences between how these attributes are measured by scientists and how they are perceived by individuals. If this difference is large, the hedonic price function will not accurately represent the values of these attributes. Individual perceptions of environmental attributes are central to this type of analysis.[23] Another issue to consider is the timing of the effect from the environmental improvement. Some effects from environmental improvements change over time. Others may be understood differently over time depending on available information (e.g., hazardous waste sites). The choice of when and how to measure the environmental attribute for a given transaction price is complicated. Refer to Kiel and McClain (1995) for a discussion of price responses over time.

- **The model for estimation:** There are numerous statistical or econometric issues associated with applying hedonic methods to property value studies. These include the choice of functional form, the definition of the extent of the market, identification, and endogeneity. A brief overview of the first two estimation issues is presented below. Refer to Palmquist (1991) for a thorough treatment of the econometric issues associated with hedonic property value studies.

  Because economic theory offers limited guidance on the functional form of an hedonic price function, researchers often try several forms when estimating hedonic functions (e.g., semilogarithmic, inverse semilogarithmic, log-linear, or quadratic Box Cox). However, it is important to note that the choice of functional form has implications for benefits assessment. See Graves et al. (1988) and Cropper, Deck, and McConnell (1988) for discussions of issues related to the choice of functional form.

  The choice of data also has an effect on estimation. The extent of the market is defined by the scope of housing market data collected. Questions have been raised about how to define the extent of housing markets. Empirically, it is important to note that if the market is defined to be too big, the resulting coefficients of the hedonic price function may be biased. Conversely, if the market is defined too narrowly, the coefficients of the hedonic price function are less efficient. Refer to Michaels and Smith (1990) for information on defining the extent of the market.

- **Assessing the results an empirical study:** Two simple ways to assess the quality of a property value study are noted here. First, a review of the empirical work is informative. This involves assessing the quality of the data collected, the framing of the policy problem, the measurement of environmental attributes, and the statistical regression analysis. Second, the existing literature on hedonic methods is a valuable resource. Comparing data, modeling assumptions, and results across studies is a useful exercise. While variation is expected across studies, especially between those completed on different areas, some factors such as the signs of particular coefficients may be consistently reported.

7.5.2.4 Averting Behavior Method

The averting behavior method infers values from observations of how people change defensive behavior in response to changes in environmental quality. Defensive behaviors are usually defined as actions taken to reduce the risk of suffering environmental damages, as well as actions taken to mitigate the impact of environmental damages. The former category includes behaviors such as the use of air filters or boiling water prior to drinking it, while the latter includes the purchase of medical care or treatment. Faced with a given level of environmental risk, the averting behavior method assumes that individuals engage in these defensive behaviors to achieve an optimal level of health. By analyzing the expenditures associated with these defensive behaviors economists can attempt to estimate the value individuals place on small changes in risk.[24]

[23] For example, hedonic property value studies that address water quality often use measures such as water clarity because these are observable characteristics. In contrast, standard scientific measures such as BOD or pH may not be readily perceived by individuals.

[24] As Desvousges et al. (1998) note, the term "averting behavior study" has been used to describe at least three somewhat different approaches: attempts to estimate WTP for environmental quality; attempts to estimate WTP for health effects or other specific impacts; and simple summation of observed expenses.

Economic Foundation of Averting Behavior Method

The economic theory underlying the averting behavior method rests on a model of household production. In these models, households produce health benefits by combining an exogenous level of environmental quality with inputs such as defensive behaviors. The underlying theory predicts that a person will continue to take protective action as long as the perceived benefit exceeds the cost of doing so. If there is a continuous relationship between defensive actions and reductions in health risks, then the individual will continue to avert until the cost just equals his or her WTP for these reductions. Thus, the benefit for a small reduction in health, or health risk, is estimated from two primary pieces of information (1) the cost of the averting behavior or good and (2) its effectiveness, as perceived by the individual, in offsetting the loss in environmental quality.

Averting behaviors methods can provide theoretically correct measures of WTP to avoid a decline in environmental quality or an increase in environmental risks. To do so, however, they require a great deal of data and particular assumptions about consumer preferences. In practice, it has proven difficult to meet these requirements. More detail on the difficulties inherent in applying the averting behavior model can be found in Cropper and Freeman (1991).

One approach to estimation is to use observable expenditures on averting and mitigating activities to generate values that may be interpreted as a lower bound on WTP. Harrington and Portney (1987) demonstrate this by showing that WTP for small changes in environmental quality can be expressed as the sum of the values of four components: lost time, changes in averting expenditures, changes in mitigating expenditures, and the loss of utility from pain and suffering. The first three terms of this expression are observable, in principle, and can be approximated by using changes in these expenditures observed after a change in environmental quality. The resulting estimate can be interpreted as a lower bound on WTP that may be used in benefits analysis. These estimates can be an improvement over cost-of-illness estimates alone, because the latter usually captures only mitigating expenditures and lost time.[25]

Averting behavior results cannot always be interpreted as lower bounds on WTP, however, because conclusions may depend critically upon modeling conditions. For example, Shogren and Crocker (1991) use a theoretical model to show that the impact of changes in risk on defensive expenditures is ambiguous and that these expenditures need not be a lower bound value. Using the same model Quiggin (1992) imposes restrictions under which defensive expenditures will increase in response to increases in risk, providing support for self-protection expenditures as a method for benefits valuation. Recently, Shogren and Crocker (1999) show that averting behavior need not be a lower bound on value when both private and collective risk reduction strategies are considered.

Large, or non-marginal, changes in environmental quality require a somewhat different valuation strategy. Generally, it is not possible to obtain exact estimates of WTP for these changes. However, Bartik (1988a) details the conditions under which upper and lower bounds on WTP may be estimated in this circumstance. These bounds effectively bracket WTP.

Finally, analysts should remember that consumers base their actions on perceived benefits from defensive behaviors. If these perceptions differ from objective estimates of, for example, risk changes, the analysis will produce biased WTP estimates for a given change in objective risk. Surveys may be necessary in order to determine the benefits individuals perceive they are receiving when engaging in defensive activities. These perceived benefits can then be used as the object of the valuation estimates.

General Application of Averting Behavior Method to Benefits Assessment

The averting behavior method can, in theory, provide WTP estimates for a wide range of environmental benefits, including changes in mortality risks, morbidity risks, and damage to materials. Most recent research, however, has focused on health risk changes.

Mortality risks can be estimated with the averting behavior method by observing purchases of items that reduce the

[25] Cropper and Freeman (1991) note that the full costs of medical expenditures and lost work time may not be borne by individuals making these decisions due to insurance and paid sick leave. An analysis of social benefits would need to include the costs that have been shifted from the consumer to others.

risks of dying in an accident. These applications are sometimes known as consumer market studies. One of the difficulties with the use of averting behavior methods in this context is that many of the risk reduction actions are discrete rather than continuous, leading to estimates that are likely to understate the value of risk reduction to the average purchaser. These studies can also be sensitive to assumptions about unobserved costs such as the time required for employing or maintaining the risk-reducing good.

The most common focus of averting behavior models has been the estimation of values for non-fatal health (morbidity) risk changes. There have been many analyses of observable averting and mitigating expenditures. Some of these studies focus on behaviors that prevent or mitigate the impact of particular symptoms (e.g., shortness of breath, headaches), while others have examined averting expenditures in response to specific episodes of contamination (e.g., groundwater contamination). The difference in these endpoints is important. Because many contaminants can produce similar symptoms, studies that estimate values for symptoms may be more amenable to benefit transfer than those that are episode-specific. The latter may be more useful, however, if assessing the benefits of a regulation expected to reduce the probability of similar contamination episodes.

Considerations in Evaluating and Understanding Averting Behaviors Studies

- **Data requirements and implications:** Cropper and Freeman (1991) describe the data required for estimating WTP using the averting behavior method. These requirements are quite burdensome and include information detailing the severity, frequency, and duration of symptoms; exposure to environmental contaminants; actions taken to avert or mitigate damages; the costs of those behaviors and activities; and other variables that affect health outcomes (e.g., age, health status, chronic conditions).

  Often, data availability will limit the analysis to an examination of observed defensive expenditures. These results can be cautiously interpreted as a lower bound on WTP. Analysts should note that costs associated with pain and suffering will not be included in the estimate.

- **Separability of other benefits:** Many defensive behaviors not only avert or mitigate against environmental damages, but also provide other benefits. For example, air conditioners obviously provide cooling in addition to air filtering, and bottled water may not only reduce health risks, but may also be better tasting. The degree to which individuals engage in averting behaviors to obtain these benefits provides evidence of the value of these qualities, but disentangling the value of different components is not an easy task. In order to accurately produce estimates of WTP for a risk change, for example, averting behaviors studies must isolate the value for the effect of interest from the value of the other benefits conferred by the defensive activity. It is also possible that the averting behavior may have negative effects on utility. For example, wearing helmets when riding bicycles or motorcycles may be uncomfortable. Failure to account for "other" benefits and disutilities associated with averting behaviors will result in biased estimates of WTP. Analysts should exercise caution in interpreting the results of studies that focus on goods in which there may be significant interrelated costs and benefits.

- **Modeling assumptions:** As noted above, restrictive assumptions are sometimes needed to make averting behavior models tractable. For example, assuming that the economy and the environment are additively separable may lead to unambiguous results, but it may be plausible only in particular circumstances. Shogren and Crocker (1999) note this fact and suggest that this assumption be justified whenever invoked. Analysts drawing upon averting behavior studies will need to review and assess the implications of these assumptions for the valuation estimates.

7.5.2.5 Cost-of-Illness Method

The health economics literature relies heavily upon the cost-of-illness method to value morbidity changes. The cost-of-illness method does not estimate WTP, but rather estimates the change in explicit market costs resulting from a change in the incidence of a given illness. Two types of costs measured in a typical cost-of-illness study are direct costs (such as diagnosis, treatment, rehabilitation, and accommodation) and indirect costs (including loss of work time).

Economic Foundation of Cost-of-Illness

The theoretical basis for the cost-of-illness method relies on two major assumptions (1) direct costs of morbidity reflect the economic value of goods and services used to treat illness and (2) a person's earnings reflect the economic value of lost production. Because of distortions in medical and labor markets, an argument could be made about whether these assumptions hold, but they are broadly consistent with neoclassical economics.

It is important to note that the cost of illness is not a measure of WTP. The cost-of-illness approach simply measures *ex post* costs and does not attempt to measure the loss in utility due to pain and suffering or the costs of any averting behaviors that individuals have taken to avoid the illness altogether (see Section 7.5.2.3 on averting behaviors). However, the cost-of-illness estimate may be considered a lower bound estimate of WTP (Harrington and Portney, 1987; Berger et al., 1987). The main reason that the cost of illness understates total WTP is the failure to account for many effects of disease. It ignores pain and suffering, defensive expenditures, lost leisure time, and any potential altruistic benefits. As a simple hypothetical example, if an individual spends five dollars on pain medication to treat a headache, and does not miss any time from work due to the headache, his or her cost of illness is five dollars. The individual's actual WTP to avoid the headache is likely to be greater than five dollars, assuming he experiences disutility from the pain the headache causes prior to taking the pain medication. Available comparisons of cost-of-illness and total WTP estimates suggest that the difference can be large (Rowe et al., 1995). However, this difference varies greatly across health effects and across individuals.

Most existing cost-of-illness studies estimate indirect costs based on the typical hours lost from a work schedule or home production, evaluated at an average hourly wage. The direct medical costs of illness are generally derived in one of two ways. The empirical approach uses a database of actual costs incurred for patients with the illness to estimate the total medical costs of the disease. The theoretical approach uses a panel of physicians to develop a generic treatment profile for the illness. Illness costs are estimated by multiplying the probability of a patient receiving a treatment by the cost of the treatment. For any particular application, the preferred approach will depend on availability of reliable actual cost data as well as characteristics of the illness under study.

Detailed descriptions of the cost-of-illness approach can be found in Cooper and Rice (1976), Hartunian et al. (1981), Hu and Sandifer (1981), Rice (1966), Rice et al. (1985) and EPA's *Cost of Illness Handbook* (EPA, forthcoming).

General Application of Cost-of-Illness Method to Benefits Assessment

Because the cost-of-illness approach does not rely on elaborate econometric models, and data are often readily available, implementation of this approach is relatively straightforward. For these same reasons, the approach is easy to explain to policy makers and the general public and tends to be less resource intensive than other approaches to health valuation. The method is generally suited for illnesses such as non-fatal cancers and other incidents of morbidity.

Cost-of-illness measures will understate WTP because they do not capture the disutility associated with anxiety, pain and suffering, or averting costs. On the other hand, some WTP estimates may understate social costs because they are unlikely to account for health care costs passed on to third parties (e.g., health insurance companies or hospitals in the case of direct medical expenses, and employers who offer sick leave in the case of time/productivity loss).

Considerations in Evaluating and Understanding Cost-of-Illness Studies

- ***Ex post* vs. *ex ante* measure:** As noted above, the cost-of-illness measures *ex post* costs of an illness rather than WTP to avoid the illness. Although the approach may account for costs shifted from the individual experiencing the illness to third parties, it fails to account for the disutility of pain and suffering, or any costs that may have been incurred in order to avoid the illness. Also, *ex post* measures cannot capture any value associated with risk attitudes. These attitudes may have a significant effect on WTP to reduce risks of more severe illnesses.

  It is also important to keep in mind that this measure captures the costs of choices that individuals make. Individuals generally choose when and how often to go to the doctor and when and for how long to stay home from work. These choices may be affected by the existence of health insurance, sick leave, and socioeconomic status.

- **Technological change:** Medical treatment technologies and methods are constantly changing, and this could push the true cost estimate for a given illness either higher or lower. When using previous cost-of-illness studies, the analyst should be sure to research whether and how the generally accepted treatment has changed from the time of the study.
- **Measuring the value of lost productivity:** Several issues arise in the indirect cost portion of a cost-of-illness study. Simply valuing the actual lost work time due to an illness may not capture the full loss of an individual's productivity in the case of a long-term chronic illness. Chronic illness may force an individual to work less than a full-time schedule, take a job at a lower pay rate than he or she would otherwise qualify for as a healthy person, or drop out of the labor force altogether. A second issue involved with estimating the value of lost productivity is the choice of wage rate. Even if the direct medical costs are estimated using individual actual cost data, it is highly unlikely that the individual data will include wages. Therefore, the wage rate chosen should reflect the demographic distribution of the illness under study. Furthermore, the value of lost time should include the productivity of those persons not involved in paid jobs. Homemakers' household upkeep and childcare services, retired persons' volunteering efforts, and students' time in school all directly or indirectly contribute to the productivity of society. Finally, the value of lost leisure time to an individual and his family is not included in most cost-of-illness studies.

7.5.3 Stated Preference Methods

Stated preference approaches attempt to measure WTP values directly. Unlike the revealed preference methods that infer values for environmental goods and services from observed behavior, stated preference methods rely on data from surveys that directly question respondents about their preferences to measure the value of environmental goods and services. This class of methods comprises several related techniques, including contingent valuation (CV), stated choice or conjoint analysis (CA), and less frequently, contingent ranking (CR). The common feature of these methods is direct questioning of members of a population about their likely choices in a hypothetical market. These three techniques are discussed below.

Economic Foundation of Stated Preference Methods

There are some situations in which data on actual behavior and choices cannot be used to derive estimates of the value of environmental goods and services. Stated preference methods rely on survey data rather than on data on observed behavior, therefore, they can be used to measure the value of environmental goods and services in most situations. The responses elicited from the surveys, if truthful, are either direct expressions of WTP or can be used to estimate WTP for the good in question.

- **Contingent Valuation:** Contingent valuation (CV) is the most well developed of the stated preference methods. CV surveys either ask respondents if they would pay a specified amount for a described hypothetical commodity or ask their highest WTP for it (for a good overview of the method see Hanemann, 1991; Mitchell and Carson, 1989; Carson, 2000; or Kopp et al. 1997). Concerns about the reliability of value estimates that come from CV studies have dominated debates about the methodology, since research has shown that bias can be introduced easily into these studies, especially if they are not carefully done. In particular, the concern that CV surveys do not require respondents to make actual payments has led critics to argue that responses to CV surveys are biased because of the hypothetical nature of the good. Reliability tests on the data that conform to expectations from both economic and psychological theory can enhance the credibility of a CV survey. Surveys without these tests should be suspect; surveys whose results fail the tests may be discredited.

  The result of the debates about the reliability of the CV method has been an infusion of methods and theory, particularly from the disciplines of psychology and survey research, to enhance questionnaire design to mitigate these concerns (Krosnick, 1991; Fischhoff, 1997). In addition, the National Oceanic and Atmospheric Administration (NOAA) convened a panel of well-known economists to review and evaluate the methodology in 1993. The panel devised a set of "best practices" recommendations for the method, particularly as it relates to natural resource damage assessments (NOAA, 1993). EPA subsequently prepared comments on the panel recommendations and regulations NOAA proposed that drew upon the panel's report (EPA, 1994).

- **Conjoint Analysis and Contingent Ranking:** Conjoint analysis (CA) and contingent ranking (CR) studies ask respondents to make choices between two or more (in the case of CA), or rank several (in the case of CR), similar commodities with different attributes and prices, in order to tease out the marginal value of particular attributes of the commodity of interest (Johnson et al., 1994). These methods are a variation on stated preference methods that aim to evaluate marginal tradeoffs rather than the total value for a described change that is evaluated in CV studies. Arising out of the marketing discipline, these methods rely on respondents' ability to make choices between commodities whose attributes differ in relation to one another. These methods often present respondents with a series of binary choice questions (e.g., "Given the descriptions of A and B, would you prefer A or B?") or multiple choice questions that ask respondents to make tradeoffs between prices and other features of commodities that are presented to them.

General Application of Stated Preference Methods to Benefits Assessment

More than 2,000 stated preference studies have been undertaken since the early 1970's. Among other things, these have been used to value changes in visibility (Chestnut and Rowe, 1990; Tolley et al. 1985), changes in surface water quality (Mitchell and Carson, 1984, 1986b), groundwater protection (McClelland et al., 1992), recreation services (Cameron and James, 1987; Bishop and Heberlien, 1979) and changes in health effects attributable to pollution (Krupnick and Cropper, 1989; Mitchell and Carson, 1986a; Viscusi et al., 1991).

Currently, contingent valuation is the only established method capable of estimating non-use values; however, most CV studies are designed to elicit respondents' total value for a given commodity. A number of researchers have attempted to disaggregate WTP values into "use" and "non-use" components. Examples of studies where non-use values have been specifically evaluated include McClelland et al. (1992) and Schulze et al. (1993). A more practical approach is to represent non-use values by employing the total WTP amounts given by persons who do not use the resource. The downside to this convenience is that there might be significant differences between those who use the resource and those who do not. Applying non-use values from the latter population to the former one may result in biased estimates.

In the context of environmental valuation, the commodity being purchased is usually a described change in environmental quality. This is a Hicksian measure, since it asks respondents to state the amount of income that they would be willing to forgo in order to have the described commodity, while making them as well off as they were without it and the payment. Similarly, they might be asked how much they would be willing to accept to put up with a nuisance or a loss. However, willingness to accept applications of CV are much more problematic because, unlike the case of WTP, there is no upper limit on the size of the opportunity set available to the respondent. This results in a strong potential for respondents to overstate the amount they would need to receive to compensate them for a loss.

While conjoint analysis (CA) has been used in marketing for some time, its application to environmental valuation began in the late 1980's. To date, it has not been subject to the level of testing and scrutiny that CV has had, so much less is known about the reliability (and how to enhance it) of these studies. The main methodological concerns that arise with CA studies are the viability of disaggregating the good in question into attributes that can be separately traded off in respondents' minds, and the problem that many respondents display intransitive preferences over the numerous, and often complex, set of choices. As a result of this complexity, heteroskedasticity is a pervasive problem with these methods.

An important limitation to using contingent valuation and other stated preference techniques is that it is expensive and time-consuming to survey the public about their preferences. Samples must be drawn, questionnaires developed, surveys administered either by mail, telephone or in person, and results coded and analyzed. In-person interviews are most expensive, but in some contexts are unavoidable due to the need to present complicated information to respondents or when they are required as criterion for legal evidence. Mail and phone surveys carry much lower costs and are often sufficient for use in the analysis of EPA policies.

Considerations in Evaluating Contingent Valuation Studies

Accurately measuring WTP for environmental goods and services using contingent valuation depends on the reliability and validity of the data collected. There are several issues to consider when evaluating study quality.

- **Content validity:** To evaluate a survey instrument itself, the analyst should look for a number of features that the researchers should have incorporated into the survey scenario. First, the commodity being valued must be clearly and concisely defined. A detailed explanation of the salient features of the environmental change being valued (the "commodity") begins with a careful exposition of the conditions in the baseline case and how these would be expected to change over time if no action were taken. Next, the action (policy change) should be described, including an illustration of how and when the policy action would affect aspects of the environment that people might care about. Finally, the way the payment will be made (e.g., through taxes, user fees, etc.) may have large implications for the outcome, so careful attention should be paid to the rationale given for the choice of payment mechanism. Respondent attitudes about the provider and the implied property rights of the survey scenario can be used to evaluate the appropriateness of these features of the commodity description (Fischhoff and Furby, 1988). Questions that probe for respondent comprehension and acceptance of the commodity scenario can offer important indications about the potential for the study to be reliable.
- **Construct validity:** In CV studies, the main indicators of study quality are tests of internal validity that can be incorporated into study design. Internal validity is supported when variables that are expected by theory to be important determinants of preferences actually are statistically significant with the correct sign. For example, with normal goods, price is expected to have a negative effect on demand for a good, while household income is expected to have a positive effect, all else equal. Thus, respondents with higher income are expected to demand more of the good than respondents with low income. Familiarity with the good or its context can also be an important indicator of internal validity. One would intuitively expect that someone who fishes would know more about, and be willing to pay more for, a commodity that improves conditions for fishing than someone who never engages in outdoor recreation. Tests of sensitivity to scope, where the amount of the commodity is varied randomly over different sub-samples of survey respondents, can increase confidence in the results where the findings are consistent with theoretical expectations (Carson et al., 1993).
- **Criterion validity:** In order to assess criterion validity, the analyst needs to have an indicator of true value against which to evaluate values from contingent valuation studies. Given the lack of actual market prices, it is often impossible to conduct criterion validity tests. However, the quality of a CV study can also be gauged by comparing valuation estimates obtained using CV with those obtained using other techniques. At least one study that has compared CV valuation estimates with estimates derived using other valuation techniques has shown that, where the CV study was carefully designed, CV estimates are not inflated relative to the other estimates for the same commodity (Carson, 1996).

In conclusion, because of the issues raised here, among other factors, there is a divergence of views within the economic profession concerning whether stated preference methods can provide useful information on economic values and on validity of individuals' responses to hypothetical questions. Nonetheless, for goods providing non-use value, stated preference methods may provide the only analytic method currently available for benefits estimation.

7.5.4 Benefit Transfer

Benefit transfer can be a feasible alternative to using one of the primary stated or revealed preference research methods described in previous sections. Rather than collecting primary data, the benefit transfer approach relies on information from existing studies that have applied other methods. More precisely, Boyle and Bergstrom (1992) define benefit transfer as "the transfer of existing estimates of nonmarket values to a new study which is different from the study for which the values were originally estimated." The case for which the existing estimates were obtained is often referred to as the "study case," while the case under consideration for a new policy is termed the "policy case."

Existing applications of benefit transfer often focus on recreation demand. For an example of such a study, see Walsh et al. (1992). Applications of benefit transfer to value health effects have also been completed. See, for example, EPA's retrospective and prospective reports on the benefits and costs of the Clean Air Act (EPA, 1997a; EPA 1999a). Here, ranges of values for multiple-symptom health effects were calculated by combining results of studies that valued individual health effects. More information on benefit transfer in general and some of the issues discussed below can be found in EPA (1993) and a special issue of *Water Resources Research* (1992, Volume 28, Number 3) dedicated to the topic. More recently, Desvousges et al. (1998) discusses transfer studies in general, not only for valuation purposes. The authors illustrate the transfer method with a case-study estimating externalities associated with electric utility generation.

Is Benefit Transfer the Appropriate Technique?

The advantages to benefit transfer are clear. Original studies are time consuming and expensive; benefit transfer can reduce both the time and financial resources needed to develop benefits estimates of a proposed policy. Given the demands of the regulatory process, these considerations may be extremely important. Additionally, while the quality of primary research is unknown in advance, the analyst performing a benefit transfer is able to gauge the quality of existing studies prior to conducting the transfer exercise.

However, benefit transfer is not without drawbacks. Most important, estimates derived using benefit transfer techniques are unlikely to be as accurate as primary research tailored specifically to the new policy case. Of concern to the analyst is whether more accurate benefits information make a difference in the decision-making process. There are many situations in which a benefit transfer may provide adequate information. For example, if the entire range of benefits estimates from the transfer exercise falls well above or below the costs of the policy being considered, more accurate estimates will probably not alter the efficiency conclusion.

Other factors to consider when deciding whether to conduct a benefit transfer include the availability of relevant, high-quality existing studies and the degree to which additional primary research would reduce the uncertainty of the current benefits estimate.

Considerations in Evaluating and Understanding Benefit Transfer Studies

Currently, a systematic process for conducting benefit transfer does not exist. There are, however, well-accepted steps involved in the process. When conducting a benefit transfer, one should make certain that each of the following steps are carried out carefully:

- **Describe the policy case.** The first step in a benefit transfer is to describe the policy case so that its characteristics and consequences are understood. It is equally important to describe the population impacted by the proposed policy. As part of this step, it is important to determine whether effects of the policy will be felt by the general population or by specific subsets of individuals (e.g., users of a particular recreation site or children). Information on the affected population will generally be used to convert per person (or household) values to an aggregate benefits estimate.
- **Identify existing, relevant studies.** Existing, relevant studies are identified by conducting a literature search. This literature search should, ideally, include searches of published literature, reviews of survey articles, examination of databases, and consultation with researchers to identify government publications, unpublished research, works in progress, and other "gray literature."
- **Review available studies for quality and applicability.** In the third step, the analyst should review and assess the studies identified in the literature review for their quality and applicability to the policy case. The quality of the study case estimates will, in part, determine the quality of the benefit transfer. Indicators of quality will generally depend on the method used. See the previous discussions on each of the primary research methods for more information on assessing the quality of studies. Assessing studies for applicability involves determining whether

available studies are comparable to the policy case. Specifically:

— the basic commodities must be essentially equivalent;

— the baseline and extent of the change should be similar; and

— the affected populations should be similar.

The analyst should also determine whether adjustments can be made for important differences between the policy case and the study case. In some cases, it may prove enlightening to discuss your interpretation and intended use of the study case with the original authors. See Desvousges et al. (1992) for additional information on criteria used to determine quality and applicability. For more information on applicability as related to specific benefit categories, see the draft *Handbook for Non-Cancer Valuation* (EPA, 1999c), the *Children's Health Valuation Handbook* (EPA, forthcoming), and Desvousges et al. (1998).

- **Transfer the benefit estimates.** This step involves the actual transfer. There are four types of benefit transfer studies: point estimate, benefit function, meta-analysis, and Bayesian techniques. The point estimate approach involves taking the mean value (or range of values) from the study case and applying it directly to the policy case. As it is rare that a policy case and study case will be identical, this approach is not generally recommended. Rather than directly using existing values, analysts will often adjust point estimates based on judged differences between the study and policy cases. Judgments of this type should be based on economic theory, empirical evidence, and experience (Brookshire and Neill, 1992). The benefit function transfer approach is more refined but also more complex. If the study case provides a WTP function, valuation estimates can be updated by substituting applicable values of key variables, such as baseline risk and population characteristics (e.g., mean or median income, racial or age distribution) from the policy case into the benefit function. The most rigorous benefit transfer exercise uses meta-analysis. Meta-analysis is a statistical method of combining a number of valuation estimates that allows the analyst to systematically explore variation in existing value estimates across studies. As with the benefit function transfer approach, key variables from the policy case are inserted into the resulting benefit function. An alternative to the meta-analytic approach is the Bayesian approach. These techniques provide a systematic way of incorporating study case information with policy case information. Studies that have explored these concepts include Atkinson et al. (1992) and Boyle et al. (1994). A discussion of Bayesian approaches appears in Desvousges et al. (1998). Regardless of the procedure used, estimates are generally aggregated over the affected population to compute an overall benefits estimate.

- **Address uncertainty.** Benefit transfer involves judgements and assumptions. Throughout the analysis, the researcher should clearly describe all judgements and assumptions and their potential impact on final estimates, as well as any other sources of uncertainty inherent in the analysis.

7.6 Values for Major Benefit Categories

As noted earlier, EPA policies may reduce the risk of premature death, typically measured as the number of statistical lives "saved" as a result of the policy action. The benefits of these risk reductions are usually measured using the concept of the "value of a statistical life" (VSL). VSL estimates are derived from aggregated estimates of individual values for small changes in mortality risks. If 10,000 individuals are each willing to pay, for example, $500 for a reduction in risk of 1/10,000, then the value of saving one statistical life equals $500 times 10,000—or $5 million. This does not mean that any identifiable life is valued at this amount, but rather that the aggregate value of reducing a collection of small individual risks is worth $5 million in this case.

7.6.1 Human Health: Mortality Risks

EPA policies reduce a wide array of mortality risks. Some risks are experienced by young persons and others by older persons. Some risks result in death shortly after exposure, while others take years to manifest. For benefits analysis, mortality risks can generally be classified across two broad dimensions: the characteristics of the affected population and the characteristics of the risk itself, such as timing. These dimensions can be expected to affect the value of reducing mortality risks.

An ideal value estimate for fatal risk reduction would account for all of these demographic and risk characteristics. It would be derived from the preferences of the population affected by the policy, based on the type of risk that the policy is expected to reduce. For example, if a policy were designed to remove carcinogens at a suburban hazardous waste site, the ideal measure would represent the preferences for reduced cancer risks for the typical suburban dweller in the area. Unfortunately, it is simply too expensive and time-consuming to obtain such unique risk value estimates for each EPA policy.

Because original research is usually infeasible, analysts at EPA will need to draw from existing VSL estimates that have been obtained using well-established methods. However, virtually all available applications of these methods focus on risks that differ from environmental risks in a number of ways. Applying existing VSL estimates found in the economics literature is an exercise in benefit transfer and raises a number of issues associated with this technique.

This section characterizes and assesses these issues, recognizing that there are limitations to how effectively analysts can make adjustments in the benefit transfer process. The discussion is sometimes necessarily broad, given that there are a variety of different types of mortality risks affected by EPA policies. First, this section briefly reviews relevant economic valuation methods and the VSL estimates they provide. Then the bulk of this section highlights key considerations when considering and transferring these values for use in EPA benefits analysis. In order to focus the discussion, this section emphasizes those considerations that may be unique to benefit transfer in the context of mortality risk changes. Benefit transfer considerations that are common to all applications, including most demographic characteristics of the study and policy populations, are described in the section of the benefit transfer method itself.

7.6.1.1 Available Methods for Estimating Mortality Risk Values

The value of small changes in mortality risk is well-studied, although researchers generally acknowledge that there are formidable difficulties in measuring risk-dollar tradeoffs. Economists have developed three broad methods to estimate a value of mortality risk reduction, each of which is described below. When using any of these methods, researchers encounter uncertainties not only in isolating the amount of compensation received for assuming higher mortality risk, but also in estimating the actual and perceived risk increment inherent in the transaction.

- **Wage-risk analysis:** This method is well-established and the economics literature contains at least twenty high-quality wage-risk studies. The resulting VSL estimates range from about $0.7 million to more than $16 million (1997$) and are included in Exhibit 7-3. Wage-risk studies have been performed in a number of different industries and countries and their estimates appear to be somewhat sensitive to the data and econometric model used.[26] Workplace mortality risks, which tend to be dominated by deaths associated with accidents or other immediate causes, form the basis for VSL estimates from these studies. Environmental risks affected by EPA policies often differ from these types of risk in a number of ways.
- **Contingent valuation:** There are at least five high-quality published estimates of VSL based on the contingent valuation method. These estimates are broadly consistent with those generated by the wage-risk method and are included in Exhibit 7-3. These studies have not employed a fatal risk scenario involving an environmental cause and, therefore, suffer from some of the same "risk context" differences as wage-risk studies when transferred for use in EPA policy analyses. Recently, however, researchers have exhibited renewed interest in using the contingent valuation method to explore how particular factors affect WTP to reduce risks.[27]
- **Averting behavior studies:** The published literature contains several examples of averting behavior

[26] Viscusi (1992, 1993) discusses the implications of different specifications and data sets.

[27] Johannesson and Johansson (1996), for example, attempt to value extensions to life expectancy. The design of this study has, however, received some criticism (Krupnick et al., 1999).

Exhibit 7-3 Value of Statistcal Life Estimates (mean values in 1997 dollars)

Study	Method	Value of Statistical Life
Kneisner and Leeth (1991 - U.S.)	Labor Market	$0.7 million
Smith and Gilbert (1984)	Labor Market	$0.8 million
Dillingham (1985)	Labor Market	$1.1 million
Butler (1983)	Labor Market	$1.3 million
Miller and Guria (1991)	Contingent Valuation	$1.5 million
Moore and Viscusi (1988)	Labor Market	$3.0 million
Viscusi, Magat and Huber (1991)	Contingent Valuation	$3.3 million
Marin and Psacharopoulos (1982)	Labor Market	$3.4 million
Gegax et al. (1985)	Contingent Valuation	$4.0 million
Kneisner and Leeth (1991 - Australia)	Labor Market	$4.0 million
Gerking, de Haan and Schulze (1988)	Contingent Valuation	$4.1 million
Cousineau, Lecroix and Girard (1988)	Labor Market	$4.4 million
Jones-Lee (1989)	Contingent Valuation	$4.6 million
Dillingham (1985)	Labor Market	$4.7 million
Viscusi (1978, 1979)	Labor Market	$5.0 million
R.S. Smith (1976)	Labor Market	$5.6 million
V.K. Smith (1976)	Labor Market	$5.7 million
Olson (1981)	Labor Market	$6.3 million
Viscusi (1981)	Labor Market	$7.9 million
R.S. Smith (1974)	Labor Market	$8.7 million
Moore and Viscusi (1988)	Labor Market	$8.8 million
Kneisner and Leeth (1991 - Japan)	Labor Market	$9.2 million
Herzog and Schlottman (1987)	Labor Market	$11.0 million
Leigh and Folsom (1984)	Labor Market	$11.7 million
Leigh (1987)	Labor Market	$12.6 million
Garen (1988)	Labor Market	$16.3 million
Derived from EPA (1997) and Viscusi (1992).		

studies, also known as "consumer market studies." Consumer market studies have examined risk-dollar tradeoffs associated with highway speed (Ghosh et al., 1975), seatbelt use (Blomquist, 1979), use of smoke detectors (Dardis, 1980; Garbacz, 1989), and the use of child safety seats (Carlin and Sandy, 1991). All of these studies suffer from problems in estimating the full costs of consumer actions to reduce risks. For example, it is difficult to quantify the added expense and "cost of time" involved in purchasing, installing, and maintaining a smoke detector. Further, these studies cannot generally control for reductions in non-fatal risks that are associated with the averting action. Some researchers argue that these and other limitations lead consumer market studies to produce downwardly biased VSL estimates.[28]

[28] These criticisms include Fisher et al. (1989) and Viscusi (1992).

No clear consensus establishes one of these three methods or any particular study as exhibiting superior features for use in regulatory analyses. However, the relative abundance of available VSL estimates from wage-risk studies provides a range of broadly applicable values for reduced mortality risk. As in other benefit transfer exercises, a range or distribution of values serves as a starting point when seeking to identify values appropriated for a particular policy context.

7.6.1.2 Existing Reviews of Value of Statistical Life Estimates

Literature surveys found in Viscusi (1993) and Fisher (1989) represent the best starting points for VSL estimates. In both cases, the authors' goals included presenting a broadly applicable range of values rather than a point estimate. Viscusi (1993) is more recent and includes some studies not considered by Fisher.[29]

Drawing from these reviews, EPA identified 26 policy-relevant risk VSL studies as part of an extensive assessment titled *The Benefits and Costs of the Clean Air Act*, 1970 to 1990 (EPA, 1997a).[30] These are summarized in Exhibit 7-3 (IEc 1992, 1993a, 1993b). Five of the 26 studies are contingent valuation studies, the rest are wage-risk studies. To allow for probabilistic modeling of mortality risk reduction benefits, the analysts reviewed a number of common distributions to determine which best fit the distribution of mean values from the studies. A Weibull distribution was selected with a central tendency (or mean) of $5.8 million (1997$).

Although these studies are generally of high-quality, the $5.8 million measure of central tendency does not account for variation in study-specific factors underlying these VSL estimates. Further research on synthesizing the results of these and other studies, including the use of meta-analysis, may provide estimates better suited for benefit transfer to environmental policies.

7.6.1.3 Benefit Transfer Considerations for Using Existing VSL Estimates

Exhibit 7-3 contains the best range of estimates available at this time. For use in benefits analyses, EPA recommends a central estimate of $4.8 million (1990$), updated to the base year of the analysis. For example, updating this figure for inflation produces an estimate of $6.1 million in 1999 dollars.[31]

However, as with any benefit transfer exercise, it is important to consider differences in the nature of the base and policy cases. As noted earlier, for fatal risks these differences fall into two major categories:

- differences in the characteristics of the population; and
- differences in the characteristics of the risks being valued.

Particular differences in these categories are detailed below. Following this presentation is a summary assessment of how analysts might assess the impact of these population and risk dimensions. Generally, policy analysts considering mortality-related benefits should include at least a qualitative discussion of the potential impact of these factors on the overall results. It is important to recognize that the ultimate objective of the benefit transfer exercise is to adjust or correct for all of the factors that significantly affect the value of mortality risk reduction in the context of the policy. Analysts should carefully consider the implications of making adjustments for some relevant factors, but not for others.

[29] A third literature survey, Miller (1990), reviews a broad range of value of life studies, including estimates from averting behavior studies that others forcefully argue are not appropriate for environmental policy purposes. In addition, Miller's results are dependent on adjustments he makes to wage-risk data. These adjustments are the subject of debate among economists and may be difficult to defend in environmental contexts.

[30] This approach for valuing mortality risks was subject to extensive external peer review during the development process of this report. Since the report's release, this approach has been adopted in other EPA benefit analyses. Peer reviewers have recently confirmed the approach for use in a prospective analysis of the Clean Air Act (EPA, 1999a).

[31] This was estimated using the Consumer Price Index (CPI) for all goods and services. Many economists prefer to use the Gross Domestic Product (GDP) Deflator inflation index in some applications. The key issue for EPA analysts is that the chosen index be used consistently throughout the analysis.

Factors Associated with Demographic Characteristics

- **Age (longevity):** Several authors have attempted to address potential differences in the value of statistical life due to differences in the average age of the affected population or the average age at which an effect is experienced.[32] In the case of reductions in mortality risks, a young person is assumed to experience a greater expected benefit in total lifetime utility than an older person. This hypothesis may be confounded by the finding that older persons reveal a greater demand for reducing mortality risks and hence have a greater implicit value of a life year (Ehrlich and Chuma, 1990). Though few in number, empirical studies and theoretic models suggest that the value of a life follows a consistent "inverted-U" life-cycle, peaking in the region of mean age.[33]

  Two alternative adjustment techniques have been derived from this literature. The first, *valuation of statistical life-years*, is based on the concept of statistical life years introduced in Section 7.4.1. The most common application of this approach is illustrated in Moore and Viscusi (1988) and presumes that 1) the value of statistical life equals the sum of discounted values for each life year and 2) each life year has the same value. This method was applied as an alternative case in an effort to evaluate the sensitivity of the benefits estimates prepared for EPA's retrospective study of the costs and benefits of the Clean Air Act (EPA 1997). A second technique is to apply a distinct value or suite of values for mortality risk reduction depending on the age of incidence. However, there is relatively little available literature upon which to base such adjustments.[34]

- **Health status:** Individual health status also appears to affect WTP for mortality risk reduction. This is a relevant factor for valuation of environmental risks because individuals with impaired health are often the most vulnerable to mortality risks from environmental causes (for example, particulate air pollution appears to disproportionately affect individuals in an already impaired state of health). Health status is distinct from age (a "quality versus quantity" distinction) but the two factors are clearly correlated and therefore must be addressed jointly when considering the need for an adjustment. At least one pilot study has found that WTP for increased longevity decreases with a declining baseline health state (Desvousges et al., 1996).[35]

Factors Associated with Characteristics of Risk and Other Considerations

- **Risk characteristics** appear to affect the value that people place on risk reduction. A large body of work identifies eight dimensions of risk that affect human risk perception:[36]

  — voluntary/involuntary;

  — ordinary/catastrophic;

  — delayed/immediate;

  — natural/man-made;

  — old/new;

  — controllable/uncontrollable;

  — necessary/unnecessary; and

  — occasional/continuous.

[32] See, for example, Cropper and Sussman (1990) and Moore and Viscusi (1988).

[33] Jones-Lee et al. (1985) reach this conclusion empirically, considering both remaining years of life and the value of a life year. This conclusion supports theoretical predictions by Shepard and Zeckhauser (1982).

[34] This second approach was illustrated in one EPA study (EPA, 1995) for valuation of air pollution mortality risks, drawing upon adjustments measured in Jones-Lee et al. (1985).

[35] The fields of health economics and public health often account for health status through the use of quality adjusted life years (QALYs) or disability adjusted life years (DALYs). These measures have their place in evaluating the cost effectiveness of medical interventions and other policy contexts, but have not been fully integrated into the welfare economic literature on risk valuation. More information on QALYs can be found in Gold et al. (1996) and additional information on DALYs can be found in Murray (1994).

[36] A review of issues in risk perception is found in Slovic (1987). Other informative sources include Rowe (1977), Otway (1977), and Fischhoff et al. (1978).

Transferring VSL estimates between these categories may introduce bias.[37] There have been some recent efforts attempting to quantitatively assess these sources of bias. These studies generally conclude that voluntariness, control, and responsibility affect individual values for safety, although the direction and magnitude of these effects are somewhat uncertain.

Environmental risks may differ from those that form the basis of VSL estimates in many of these dimensions. Occupational risks, for example, are generally considered to be more voluntary in nature than are environmental risks, and may be more controllable.

- **Latency periods:** Many environmental policies are targeted at reducing the risk of effects such as cancer, where there may be an extended period of latency between the time of exposure and eventual death from the disease.[38] While the benefit of a reduction in exposure is an immediate reduction in the risk of the associated health endpoint, latency periods between exposure and manifestation may affect the value of that risk reduction. Existing VSL estimates are based upon risks of relatively immediate fatalities, making them an imperfect fit for a benefits analysis of many policies. Economic theory suggests that reducing the risk of a delayed health effect will be valued less than reducing the risk of a more immediate one, when controlling for other factors.

  A simple ad hoc approach to adjusting existing VSL estimates is to apply a financial discount rate over the expected latency period. However, defining latency periods with existing risk assessment methods may be difficult and empirical estimates may be highly uncertain. Further, the underlying assumptions supporting this procedure may oversimplify how individuals appear to consider delayed health effects.[39] Cropper and Sussman (1990) develop an alternative procedure to account for the influence of time on fatal risk reduction values, but their demonstration is data-intensive, requiring detailed life tables and age-specific VSL estimates.

- **Altruism:** The existing VSL literature focuses on individual risk tradeoffs, but there is evidence that people are willing to pay to reduce risks incurred by others. Although the literature on altruism is limited, several studies suggest that these values may be significant.[40] Other analysts advocate caution in attempting to inflate value of life estimates to reflect altruism, primarily because of concerns over the potential for double-counting.[41]

7.6.1.4 Summary of Advice from the Economics Literature

It is important to recognize the limitations of a single VSL point estimate and to consider whether any of the factors discussed above may have a significant impact on the benefits estimated for mortality risk reductions from environmental policies. In any given policy context, there may be several components that are both relevant and important and that could act to increase or decrease the appropriate risk reduction value used to estimate benefits. Adjustments for each these factors may offset one another to some extent.[42] Analysts should exercise caution in accounting for some important risk and population characteristics when unable to account for others.

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[37] Examples include Mendeloff and Kaplan (1990), McDaniels et al. (1992), Savage (1993), Jones-Lee and Loomes (1994, 1995, 1996), and Covey et al. (1995).

[38] Although latency is defined here as the time between exposure and fatality from illness, alternative definitions may be used in other contexts. For example, "latency" may refer to the time between exposure and the onset of symptoms. These symptoms may be experienced for an extended period of time before ultimately resulting in fatality.

[39] See, for example, the choice of discount rate discussion in Horowitz et al. (1990) and Rowlatt et al. (1998).

[40] In a study that included willingness to pay to reduce others' risk of illness from insecticides, Viscusi et al. (1988) found evidence of significant altruistic values. Jones-Lee et al. (1985) suggests an adjustment to value of statistical life estimates of about one-third to account for people's concern for the safety of others.

[41] Examples include Bergstrom (1982) and Viscusi (1992).

[42] Sometimes this might mean that very different risks will be valued similarly. There are relatively few studies that assess responses to different types of risk in an individual choice framework. A notable exception is Magat et al. (1996), which finds individuals indifferent between mortality risks from an automobile accident and those from fatal lymph cancer.

Because of the inherent uncertainty in any analysis, however, analysts should consider qualitative evaluations of these factors and explore where sensitivity analysis can satisfactorily address some of these concerns. The importance and relevance of each of the risk and demographic characteristics need to be considered. Depending upon specific policy context, there may be multiple alternatives for supplemental analysis.

For example, when policies do not affect the entire population equally, a sensitivity analysis may show the cost per life saved. In some contexts these values may provide useful information to decision makers on the relative merits of alternative policy options. However, cost-per-life-saved measures implicitly assume that all costs are associated with mortality reduction. For policies that provide other types of benefits, cost-per-life-saved measures may be misleading unless the value of those benefits are first deducted from cost estimates, but it is impossible to make these deductions when some benefits are either non-quantified or non-monetized. Because of these shortcomings, analysts will need to assess the usefulness of cost-per-life-saved measures on a case-by-case basis.

In general, the decision to perform sensitivity analysis will also depend upon the relative importance of mortality values in the overall benefits estimates and upon having sound theoretical and empirical economic literature upon which to structure the analysis. Parameter values used to formulate the sensitivity analyses must also be supported by the underlying risk assessment data.

What support does the economics literature currently offer for making these potential adjustments? Existing, feasible methods for age (or longevity) adjustments have significant limitations. Age adjustments may be desirable from a theoretical standpoint, but the relationship between the value of risk reductions and expected remaining life span is complex. Application of existing valuation of statistical life years approaches implicitly assumes a linear relationship in which each discounted life year is valued equally. As OMB (1996) notes, although "there are theoretical advantages to using a value of statistical life-year-extended (VSLY) approach, current research does not provide a definitive way of developing estimates of VSLY that are sensitive to such factors as current age, latency of effect, life years remaining, and social valuation of different risk reductions." The second alternative, applying a suite of values for these risks, lacks broad empirical support in the economics literature. However, the potential importance of this benefit transfer factor suggests that analysts consider sensitivity analysis when risk data—essentially risk estimates for specific age groups—are available. Emerging literature on the value of life expectancy extensions, based primarily on stated preference techniques, is beginning to help establish a basis for valuation in cases where the mortality risk reduction involves relatively short extensions of life.[43]

A small body of literature on the quantitative impact of risk characteristics on risk valuation exists. Although there is some qualitative consistency in the results of these studies, the risk valuation literature is not sufficiently robust to support quantitative adjustments for these factors at this time. Considerations associated with risk characteristics may deserve qualitative discussion in some policy contexts.

Both of the procedures available for accounting for latency have potentially serious shortcomings. For example, neither procedure addresses the dread of death or the morbidity that occurs prior to fatality from protracted diseases, such as that experienced with many cancers. As noted earlier, the simple "discounted VSL" approach may also oversimplify how individuals consider latency in their expressed WTP for reduced mortality risks. This literature does, however, suggest one alternative for conducting sensitivity analysis on this benefit transfer component.

7.6.1.5 Conclusion

In summary, these guidelines recognize the theoretically ideal measure of mortality risk reduction benefits and this section has discussed the many variables affecting such a measure. Due to current limitations in the existing economic literature, these guidelines conclude that an appropriate default approach for valuing these benefits is

[43] It should be noted that many observers have expressed reservations over adjusting the value of mortality risk reduction on the basis of population characteristics such as age. One of the ethical bases for these reservations is a concern that adjustments for population characteristics may imply support for variation in protection from environmental risks. Another consideration is that existing economic methods may not capture social willingness to pay to reduce health risks. Chapter 9 details how some these considerations may be informed by a separate assessment of equity. Chapter 10 describes the potential for efficiency and equity considerations to be considered together in a social welfare function.

provided by the central VSL estimate described earlier. However, analysts should carefully present the limitations of this estimate. Economic analyses should also fully characterize the nature of the risk and populations affected by the policy action and should confirm that these parameters are within the scope of the situations considered in these guidelines. While a qualitative discussion of these issues is generally warranted in EPA economic analyses, analysts should also consider a variety of quantitative sensitivity analyses on a case-by-case basis as data allow. The analytical goal is to characterize the impact of key attributes that differ between the policy and study cases. These attributes, and the degree to which they affect the value of risk reduction, may vary with each benefit transfer exercise, but analysts should consider the characteristics described above (e.g., age, health status, voluntariness of risk, latency) and values arising from altruism.

As the economic literature in this area evolves, WTP estimates for mortality risk reductions that more closely resemble those from environmental hazards may support more precise benefit transfers. Literature on the specific methods available to account for individual benefit transfer considerations will also continue to develop. EPA will continue to conduct annual reviews of the risk valuation literature and will reconsider and revise the recommendations in these guidelines accordingly. EPA will seek advice from the Science Advisory Board as guidance recommendations are revised.[44]

Despite the limitations described in this section, analysts should remember that mortality risk valuation remains one of the most studied benefit categories for environmental policies. Wage-risk studies, while not without limitations, nonetheless provide revealed preference estimates based on a well-tested method. Estimating mortality related benefits will often be relatively straightforward to implement, while other benefit categories will require more time and attention.

7.6.2 Human Health: Morbidity Risks

Morbidity valuation, or the valuation of non-fatal health effects, often requires addressing a more diverse set of issues than mortality valuation. First, there is a tremendous variety in the health endpoints considered for valuation. These endpoints vary with respect to their severity, including the degree to which other activities can be pursued and the degree of discomfort or pain associated with the ailment. The duration of the effect also varies considerably, from short term effects to those that may be permanently debilitating. Non-fatal health effects differ considerably with respect to the availability of existing value estimates. Some of these health effects have been valued multiple times with different methods, while others have not been the subject of any valuation studies.

Willingness to pay to reduce the risk of experiencing an illness is the preferred measure of value for morbidity effects. This measure includes several components. Illness imposes direct costs, such as expenses for medical care and medication, and indirect costs, such as lost time from paid work, maintaining a home, and pursuing leisure activities. Illness also imposes less easily measured, but equally real costs of discomfort, anxiety, pain, and suffering. Methods used to estimate WTP vary in the extent to which they capture these components.

A commonly used alternative to WTP is the avoided costs of illness (COI). For a given health effect, the COI approach will generally understate true WTP. By focusing on market measures of the value of health effects, it leaves out important components such as the value of avoiding pain and suffering. By focusing on *ex post* costs, it also does not capture the risk attitudes associated with *ex ante* WTP measures. However, for many effects, estimates of WTP are not currently available or are highly uncertain. Where estimates of WTP are not available, the potential

[44] A second review on this subject was recently completed by the Science Advisory Board (SAB), the results of which can be found in "An SAB Report on EPA's White Paper *Valuing the Benefits of Fatal Cancer Risk Reductions*," EPA-SAB-EEAC-00-013, July 27, 2000 (website address http://www.epa.gov/sab/eeacf013.pdf, accessed 8/28/00). The SAB review elaborates further on using the wage-risk literature for valuing mortality risk reductions, concluding that among the demographic and risk factors that might affect VSL estimates, the current literature can only support empirical adjustments related to the timing of the risk. First, the review supports adjusting willingness-to-pay estimates to account for higher future income levels, though not for cross-sectional differences in income. The second time-related adjustment recommended is to discount for risk reductions that are brought about in the future by current policy initiatives (that is, after a latency period), using the same rates used to discount other future benefits and costs. More information on the SAB review and its implications for the Guidelines will be released in a forthcoming supplement to this document.

bias inherent in relying on cost-of-illness estimates should, at minimum, be discussed qualitatively.

Time and resources will often not allow for original morbidity valuation research to support specific benefit analyses. As with other types of benefits, analysts will then need to look for estimates available from existing sources, and apply these values to the policy case using benefit transfer techniques. The discussion here is presented with this benefit transfer exercise in mind. The remaining parts of this section present a summary of methods commonly used to value reductions in morbidity, useful references for obtaining existing values, and issues that arise in transferring existing values to the analysis of EPA policies.

7.6.2.1 Available Methods for Estimating Morbidity Values

Researchers use a wide range of methods to value changes in morbidity risks. Some available methods measure the theoretically-preferred value of individual WTP to avoid a health effect, while others provide useful data but are less well-grounded in economic theory. Methods also differ in the perspective from which valuation is measured (e.g., before or after the incidence of morbidity) and the degree to which they account for all of the components of total WTP.

The three primary research methods used most often to value environmental morbidity are cost of illness, contingent valuation, and averting behavior, as described earlier. Several other primary valuation methods have been used less frequently to value morbidity from environmental causes: hedonic methods, risk-risk tradeoffs, health-state indexes, and studies of jury awards. However, these methods often do not provide monetary estimates of WTP or suffer from other methodological flaws, and are generally less useful for policy analysis.

- **Cost-of-illness:** The cost-of-illness method is straightforward to implement and explain to policy makers and has a number of other advantages. It has been applied for many years, is well-developed, and measures of direct and indirect costs are easily explained without reference to complex economic theory. Collection of additional data is often less expensive than for other methods, perhaps making it feasible to develop original cost-of-illness estimates in support of a specific policy or set of policies. Estimates for many illnesses are available from existing studies and span a wide range of health effects. EPA's *Cost of Illness Handbook* (EPA, forthcoming) contains an extensive collection of cost-of-illness estimates. As noted earlier, however, the cost-of-illness method has several shortcomings and its theoretical basis is quite limited. Generally, cost-of-illness estimates should be considered lower bounds on WTP.
- **Averting behavior:** In the case of morbidity valuation, the averting behavior method can provide WTP estimates based on actual behavior. These measures can account for all of the effects of health on individual well-being, including altruism toward other household members if averting actions are taken jointly (e.g., if everyone in the household drinks bottled water). However, the method has several weaknesses, as described in Section 7.5. Existing studies vary in their analytical approach. Some existing studies have attempted to estimate WTP for particular sets of illnesses (Gerking and Stanley, 1986; Dickie and Gerking, 1991; Bresnahan and Dickie, 1995). Others do not attempt to estimate *ex ante* WTP, but focus instead on actual household expenditures in response to a particular contamination episode or event (Harrington et al., 1989; Abdallah, 1990). In practice, most averting behavior estimates should be interpreted cautiously as a lower-bound estimate on WTP. Also, because behaviors generally avert a range of symptoms (e.g., a water filter removes contaminants with several potential health effects), it is difficult to isolate the value of avoiding those individual health effects that may be attributable to a particular EPA policy. Indiscriminate use of this method may raise significant problems with double counting or overestimation.
- **Stated preference methods:** Contingent valuation and other stated preference methods can be used to account for all the effects of illness on individual well-being, including pain and suffering. These methods appear to be the only ones capable of eliciting dollar values for altruism toward persons outside the household. Unlike the averting behavior or cost-of-illness methods, these can be applied to value the risks of illness lacking any connection to market transactions. Stated preference methods have been used to value a

number of different health outcomes including accidental poisoning (Viscusi and Magat, 1987; Viscusi et al., 1988); coughing, congestion, and other minor symptoms (Berger et al., 1987); chronic bronchitis (Viscusi et al., 1991; Krupnick and Cropper, 1992); and nonfatal nerve disease (Magat et al., 1996). Some economists, however, express concerns about the hypothetical nature of the transaction and the difficulties inherent in ensuring that respondents understand the change in health status they are being asked to value.[45]

7.6.2.2 Existing References for Morbidity Values

Analysts have a number of resources available for obtaining information on morbidity values. While these references provide valuable information, they are not substitutes for careful evaluation of original studies when considering a benefit transfer. Useful general references for valuing non-fatal health effects include Tolley et al. (1994) and Johanneson (1995). Both of these books provide references to many existing health valuation studies and discuss issues associated with using these estimates for policy valuation. Desvousges et al. (1998) assess a number of existing studies in the context of performing a benefit transfer for a benefits analysis of improved air quality. Another good starting point for reviewing available estimates is EPA's *Handbook for Non-Cancer Valuation* (EPA, 1999c). This report will provide available, published estimates for many illnesses and reproductive and developmental effects.

Because estimates of WTP will not always be available for particular health effects, another useful resource is EPA's *Cost of Illness Handbook* (EPA, forthcoming). This handbook includes cost-of-illness estimates for many cancers, developmental illnesses and disabilities, and other illnesses. Work on the handbook is ongoing and new estimates will be included as they become available.[46]

Existing EPA economic analyses may also provide useful insights. For example, the *Benefits and Costs of the Clean Air Act* (EPA, 1997a) draws upon a number of existing studies to obtain values for reductions of a variety of health effects. The report describes the central estimates used in the analysis, how these estimates were derived, and attempts to quantify the uncertainty associated with using the estimates.

7.6.2.3 Benefit Transfer Using Existing Morbidity Value Estimates

Benefit transfer was detailed earlier in this chapter; however, there are issues associated with benefit transfer particular to morbidity valuation. As with any benefit transfer, analysts should:

- carefully describe the policy case;
- assess the quality of the studies and their applicability to the policy case;
- evaluate the plausibility of the findings;
- consider possible adjustments for differences between the subject of the study and the policy case; and
- explicitly address uncertainty.

EPA's *Handbook for Non-Cancer Valuation* (EPA, 1999c) contains additional information on this subject.

Matching the Study Case to the Policy Case

- **Assessing applicability:** A key element in evaluating the applicability of a study is the correspondence between the health effect valued in the study and the health effect influenced by the policy. An assessment of this correspondence must consider the set of symptoms covered in the study. The analyst should consider whether the study case consists of a larger or smaller set of symptoms than the policy case. The severity of the symptoms should also be commensurate, including the degree to which the illness limits activities and the extent of any discomfort, pain and suffering. Analysts should also assess whether the duration of the base and policy cases are similar.

  A second key factor is the similarity between the population examined in the study and the population affected by the policy. Key considerations include the

[45] See, for example, Mitchell and Carson, 1989; Cummings et al., 1986; NOAA, 1993; Bjornstad and Kahn, 1996; NRDA, 1994; Diamond and Hausman, 1994.

[46] The *Cost of Illness Handbook* will be available online.The website will be continuously updated as additional COI estimates are completed.

baseline health status of the populations, the age of the populations, and other demographic characteristics.

- **Evaluating plausibility:** The analyst should conduct some initial checks to evaluate whether the study case values are plausible or reasonable. For example, if the estimated value of avoiding an acute, reversible effect exceeds other reasonable estimated values for avoiding long-term, chronic effects, then the value for the acute effect is probably too large and will be difficult to defend. On the other hand, WTP values that are less than cost-of-illness values for the same effect are probably too low, particularly if the effect clearly results in pain or otherwise impairs activity.
- **Using the results of multiple studies:** After reviewing the quality and applicability of available studies, the analyst can apply the valuation estimates to the data on cases averted by each policy option. Because the value of morbidity avoidance is difficult to quantify precisely, it is useful where possible to apply estimates from more than one valuation technique. Where multiple studies are available that provide differing estimates, the range of values should be presented with a discussion of the advantages and limitations of the studies used. Estimates based solely on cost-of-illness values should be flagged as potentially understating total values. WTP studies of health effects that are similar in severity and duration may be used as a point of comparison.

Addressing Uncertainty and Related Concerns

Available estimates of non-fatal effects may suffer from several limitations. They may be derived from cost-of-illness methods that do not fully measure WTP to avoid the effect or may be transferred from studies of effects that are similar, but not identical to, the effect of concern. The extent to which adjustments or new research are needed to address these concerns will depend largely on the value of new information to the decision-making process. If morbidity values are a small component of total benefits and unlikely to influence the choice among policy options, then a qualitative discussion of uncertainty may be appropriate. Where morbidity values are a significant concern, quantified sensitivity analysis and additional data collection may be desired.

Some of the major sources of uncertainty are described below. Because of the diversity of the health effects of concern and of the studies used to value morbidity effects, this discussion is relatively general. The limitations and potential adjustments or analyses of uncertainty that are appropriate will vary greatly depending on the approach used for a particular policy analysis. More information on these issues is provided in EPA's *Non-Cancer Valuation Handbook* (EPA, 1999c).

- ***Ex ante* and *ex post* valuation estimates:** Environmental contamination will generally not cause an adverse health effect with certainty, but rather will increase the probability that the effect occurs, increase its severity given that it occurs, or both. People are likely to value these changes in risk differently than they would value certain changes in health status. While contingent valuation and other methods can adopt an *ex ante* perspective and obtain estimates for risk changes, many available studies provide *ex post* value estimates for morbidity effects. For minor health effects this difference in perspective may not be important, but for severe health effects the difference may be significant and *ex post* estimates may understate the benefits of a policy action. Analysts should address this issue at least qualitatively in these cases.
- **Incomplete estimates of willingness to pay:** The widespread availability of health insurance and paid sick leave shift the costs of illness from individuals to others. While this cost-shifting can be addressed explicitly in cost-of-illness studies, it may lead to problems in estimating total WTP through contingent valuation surveys. If the researcher does not adequately address these concerns, respondents may understate their WTP, assuming that some related costs will be borne by others.
- **Timing of health effects:** Environmental contamination may cause immediate or delayed health effects and the value of avoiding a given health effect likely depends on whether it occurs now or in the future. Recent empirical research confirms that workers discount future risks of fatal injuries on the job; that is, they are willing to pay less to reduce a future risk than a present risk of equal magnitude (Viscusi and Moore, 1989). In addition, a separate study concluded that individuals value policies that yield health benefits in the present more highly than policies that

yield the same benefits in the future (Cropper et al., 1994).

7.6.2.4 Summary

Morbidity benefits valuation can be a difficult process, often requiring careful judgment decisions by the analyst. Whether the analyst is conducting original research that supports the policy action or is drawing upon existing studies, clarity and transparency in the analysis is vital. When employing benefits transfer, some shortcomings in the "fit" of the study case to the policy case is to be expected. Addressing these shortcomings explicitly, conducting appropriate sensitivity analysis, and clearly stating assumptions can greatly enhance the credibility of the benefits analysis.

7.6.3 Ecological Benefit Valuation

In estimating ecological benefits, one is generally forced to value individual ecological service flows separately and then sum these estimates rather than constructing prices for changes in the structure and function of entire ecosystems. Alternative approaches that estimate the total value of ecosystems based on the replacement cost of the entire ecosystem or its embodied energy (e.g., Costanza et al., 1997; Ehrlich and Ehrlich, 1997; Pearce, 1998; Pimentel et al., 1997) have received considerable attention as of late. However, the results of these studies should not be incorporated into benefit assessments. The methods adopted in these studies are not well grounded in economic theory nor are they typically applicable to policy analysis. Pearce (1998) contains a critical review of the total value approach, as does Bockstael et al. (2000).

Although the economics literature is replete with benefit studies, the coverage is patchy considering the broad range of services and stressors addressed by EPA programs. Especially rare in the literature are examples of wide-scale changes, very small changes, or the consequences of long term ecological and economic change. Ongoing research has begun to address these data limitations. Examples include recent contingent valuation studies undertaken for purposes of natural resource damage assessments that attempt to elicit WTP for marginal changes in long-term environmental quality (Kopp et al., 1994). In addition, Layton and Brown (1997) attempt to elicit from respondents the value of one attribute of the long-term ecological changes expected to be associated with climate change.

Available Methods for Estimating Ecological Benefits

Economists have employed a variety of methods to estimate the benefits of improved ecological conditions. Issues particular to their implementation for this benefit category are discussed below.

- **Market models:** The benefit of changes in commercial crop, timber, or fish harvests can be estimated using a variety of available market models. Several studies have assessed the social welfare implications of changes in yields for a number of crop species. For example, Taylor et al. (1993) apply the Agricultural Sector Model and Kopp et al. (1985) apply the Regional Model Farm Agricultural Benefits Assessment Model to estimate welfare impacts of agricultural yield changes. Adams et al. (1997) use the Agricultural Simulation Model to estimate the economic effects associated with yield changes resulting from climate change.

  When dealing with timber or fisheries, bioeconomic models are designed to deal explicitly with time to account for the fact that environmental and market changes are not coincident. EPA has used the Timber Assessment Market Model (TAMM) to estimate ozone effects on commercial timber harvesting. The welfare impacts of changes in commercial fish harvests have also been examined, e.g., Alaskan king crab in Greenberg et al. (1994); herring in Mendelsohn (1993); and lobster in Wang and Kellogg (1988).

  If changes in service flows are small, current market prices can be used as a proxy for expected benefit. For example, a change in the commercial fish catch might be valued using the market price for the affected species. This approach can only be used in cases where fishing effort and price are unlikely to be affected by the policies. If these conditions do not hold, a market model should be applied to assess the effects of increased catch rates on supply conditions and market price, as discussed earlier in the section on consumer and producer surplus.

- **Production function approach:** Values for indirect, non-market benefits can be estimated when their contribution to production processes are expressed explicitly in a production function. As service flows increase, the welfare gain is essentially the marginal product of the service for small changes or is reflected by the shift in the marginal cost curve. Several studies have examined the relationship between environmental quality and crop production, such as measurement of ozone impacts by Heck et al. (1983). Moreover, researchers have applied this approach using household production functions to examine ecosystem services that benefit individuals directly or to establish the link between some services and their off-site benefits. Bell (1997) values wetland contributions to recreational fishing and Barbier (1994) values a range of indirect wetland benefits. Smith et al. (1993) estimate the impact of nitrogen and pesticide loadings on coastal water recreational fishing.

- **Averting behaviors approaches:** One such approach, the replacement cost method, uses purchases of market goods to infer the value of indirect, non-market services. Willingness to pay is revealed by efforts made to substitute for services provided by ecosystems. For example, since water treatment infrastructure replaces wetland functions, investment and operations and maintenance costs provide an estimate of the value of the water filtration service provided by wetlands. This method is justified only when individuals are proven willing to incur such replacement costs, through either their voluntary purchases or their support for public works projects. If so, the value of the service is at least as much as the replacement cost.

  Another variation on this theme applies to actions that reduce the cost of complying with existing policies. For example, a reduction in atmospheric nitrogen deposition in the watershed of an estuary may ultimately reduce the costs incurred in reducing other sources of nitrogen to the system, such as added controls on POTWs. This approach is generally useful in situations satisfying two criteria: alternative pollution control methods are prescribed through existing policies and the new policies under consideration would provide a lower cost method for achieving the desired level of environmental protection.

- **Hedonic methods:** Hedonic property models can isolate the relationship between environmental quality and housing prices from the effects of variation in other attributes such as size, location, and security. This method is often used to use to value regional difference in air quality. Smith and Huang (1995) conduct a meta-analysis to examine how well the models perform in this context and discern a statistically significant relationship between housing prices and air quality measures in general. Hedonic models have also been used to estimate the impact of landfill closure (Kohlhase, 1991), nuisances from odors (Palmquist et al., 1997), and the existence and remediation of toxic contaminated sites (Kiel, 1995) on nearby property values. Other applications have addressed the costs of land-use restrictions (Parsons, 1992) and the benefits of water quality improvements (Rich and Moffitt, 1982). A promising development involves the application of hedonic methods in a model of land-use change to explore the ecological and economic consequences of landscape alteration (Bockstael, 1996). In addition, Geoghegan et al. (1997) estimate the impact of land uses adjacent to and near one's home.

- **Recreation demand models:** Recreation demand models are discussed in detail in Section 5.2.1 of this chapter. Recreation demand models are based on the tradeoff between travel expense and environmental quality. While early travel cost models dealt with single sites, did not consider environmental quality, and suffered from a range of limitations and biases, more recent efforts overcome these problems. For example, a travel cost model was used to estimate the WTP by Chesapeake Bay beach users for a 20 percent improvement in water quality as measured by nitrogen and phosphorous loadings (Bockstael et al., 1989).

  Random utility models have been used in a wide variety of studies to estimate the recreational fishing benefit of fresh water quality improvements. Montgomery and Needelman (1997) apply a random utility model to fishing behavior and estimate the benefits of eliminating toxic contamination from New York lakes and ponds. Recreation demand models also have been used in the context of forest management (Englin and Mendelsohn, 1991; Dwyer et al., 1983), the ecological effects of natural resource damages (Hausman et al., 1995; Morey and Rowe, 1995), health risks associated with fish advisories (Jakus et. al., 1997), and non-point source pollution controls in estuaries (Kaoru et al., 1995).

- **Stated preference approaches:** As discussed earlier, stated preference methods represent the only means of obtaining a value for non-use benefits. For instance, research has tried to measure how much better people feel that various wildlife species are alive and well (Stevens et al., 1991). Individuals have also expressed a willingness to pay to protect visibility in national parks, whether or not they plan to visit these parks (Crocker and Shogren, 1991). Contingent valuation has been applied to the other ecological benefit categories as well. EPA has regularly used the results of one such study to value water quality improvements in fresh water (Mitchell and Carson, 1986b).

7.6.4 Materials Damage

Market methods are the primary technique used to quantify benefits falling in this category. Materials damages can include changes in both the quantity of the materials and in the quality. Linking changes in environmental quality with the provision of service flows from materials can be difficult because of the limited understanding of the physical effects (e.g., scientific information), the timing of some effects (e.g., long-term), and risk responses of producers and consumers of these service flows. When feasible, assessment typically involves combining the output of an environmental model with stressor-response function and/or price information to estimate the impact of the change in environmental quality on production (inputs) or consumption (output) of the material service flows. The market response to this impact serves as the basis for the welfare change and benefits assessment. In practice, these market methods may be implemented as reduced from economic models that relate averting or mitigating expenditures to ambient pollutant levels. The degree to which behavioral adjustments are considered when measuring the market response are important and models that incorporate behavioral responses are preferred to those that do not. Refer to Adams and Crocker (1991) for a detailed discussion of this and other features of materials-damages-benefits assessment. See the analysis of household soiling in EPA (1997b) for an example that employs a reduced form economic model relating defensive expenditures to ambient pollutant levels.

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Chapter 8: Analyzing Social Costs

8.1 Introduction

The goal of a benefit-cost analysis is to determine the net change in social welfare brought about by a new environmental policy, as measured by changes in the producer and consumer surpluses. In general, the economic effects of a new environmental policy result in many different people and firms being affected, both positively and negatively. The previous chapter looked at the positive effects (or social benefits). This chapter considers the negative effects (or social costs). It is the sum of these changes, when combined with the social benefits, that yield a measure of the net changes in social welfare.

As with social benefits, when computing the social cost of a policy (i.e., the negative impact on social welfare), monetary sums that measure changes in individuals' welfare are all weighted equally in benefit-cost analysis. Other methods for evaluating the welfare consequences of policies on particular individuals, groups, or sectors should be examined using either economic impact analysis, equity assessment techniques, or social welfare functions (all described in Chapter 9).

This chapter is organized into four major sections followed by a concluding section. Section 8.2 reviews the theoretical foundations of social cost estimation for environmental policies. Section 8.3 discusses how to estimate and model total social cost.[1] Next, four types of models for estimating social cost are examined in Section 8.4. Then the estimation of the costs of specific regulatory approaches is reviewed in Section 8.5. Finally, Section 8.6 provides a discussion of the choice of tools for each type of policy

.

8.2 The Theory of Social Cost Analysis

The total social cost is the *sum* of the opportunity costs incurred by society because of a new regulatory policy; the opportunity costs are the value of the goods and services lost by society resulting from the use of resources to comply with and implement the regulation, and from reductions in output. These costs, however, do not take into account any of the health, environmental, safety, or other benefits which offset the social welfare costs.

The five basic components of total social costs are listed here in the general order of relative ease of estimation, and hence inclusion, in most social cost analyses of environmental policies. They include:

- **Real-resource compliance costs:** These direct costs are the principal component of total social costs and are associated with: (1) purchasing, installing, and operating new pollution control equipment, (2) changing the production process by using different inputs or different mixtures of inputs, or (3) capturing the waste products and selling or reusing them. (The last two options can actually result in negative compliance costs.)

  These real-resource costs should also include unpriced resources that have opportunity costs associated with them, such as unpaid labor diverted from other productive uses, and extra administrative costs associated with compliance. However, the pre-tax compliance costs do not include any transfers, such as emissions taxes, licensing fees, or subsidies (which are included in the firm's private costs).

[1] Several texts on applied microeconomics and policy evaluation provide substantially more theoretical depth and examples than the overview presented in this chapter, such as Arnold (1995), Gramlich (1981), and Just et al. (1982).

- **Government regulatory costs:** These include the monitoring, administrative, and enforcement costs associated with new regulations. This also includes the cost of setting up a new market when incentive-based regulations are established, such as tradable permits.
- **Social welfare losses:** These are the losses in consumer and producer surpluses associated with the rise in the price (or decreases in the output) of goods and services that occurs as a result of an environmental policy.
- **Transitional costs:** These include the value of resources that are displaced because of regulation-induced reductions in production, and the private real-resource costs of reallocating those resources. Offsetting these costs, in theory, are regulation-induced increases in resource use in both primary and related markets (e.g., more workers and equipment are needed for pollution control).
- **Indirect costs:** These other costs include the adverse effects policies may have on product quality, productivity, innovation, and changes in markets indirectly affected by the environmental policy, all of which may have impacts on net levels of measured consumer and producer surplus.

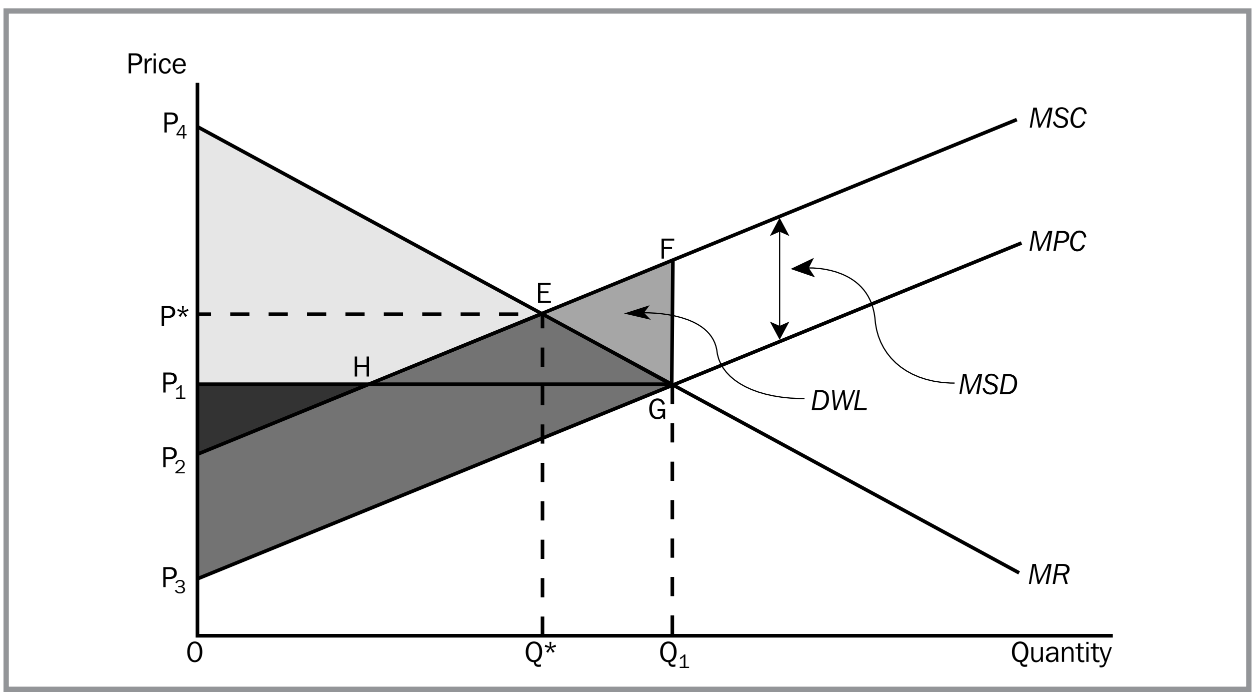
8.2.1 An Illustration of Social Costs and Externalities

Exhibit 8-1 illustrates an externality (pollution) associated with the production of a good or service (Q). In this figure:

- MR is marginal revenue from selling the good;
- MPC is the marginal private real-resource cost of production;
- MSD is the marginal social damage from pollution associated with production; and
- MSC is the total marginal social cost of supplying the good.

The producer operates where marginal revenue (MR) equals marginal private real-resource costs (MPC), so it produces Q_1 items at a price of P_1 per item. In this example, the producer also is assumed to impose a social cost on society due to pollution associated with its production of Q. Here the actual pre-policy total social cost of supplying the good is measured by the marginal private real-resource cost (MPC) plus the marginal social damage (MSD, also known as an externality) caused by pollution. Together the MPC plus the MSD gives the true marginal social cost (MSC) of supplying the good.

Exhibit 8-1 Producer and Consumer Surplus with an External Cost



The producer surplus is indicated by the area of triangle P_1GP_3 and the consumer surplus is measured as the area of triangle P_1GP_4, but the total social damage is indicated by the area of $P_3GF\ P_2$. Therefore, the deadweight loss to society (DWL) is equal to the area of triangle EFG. If producers have to pay for the damage caused by the pollution, their producer surplus is reduced to area of triangle P_2HP_1 minus the area of triangle HGF. In this case the firm would be making negative profits since the area of triangle HGF is larger than their producer surplus. Net social welfare in this case would be the area of triangle P_2EP_4 less the area of triangle EFG (the deadweight loss).

If, however, the optimal amount of the product is produced (i.e., where MR equals MSC), then the firm's output is Q* at a price of P*. In this case, consumer surplus is equal to the area of triangle P_4EP^* and producer surplus is the area of triangle P_2EP^*. Since there is no deadweight loss to society, net social welfare has increased and is equal to the area of triangle P_2EP_4.

Suppose that producers can do nothing to reduce the pollution damages other than decrease the amount of output supplied. If the government places a tax equal to the MSD on each unit of pollution, this would increase private production costs (but not private real-resource costs) by the amount of the MSD, which would cause a rise in consumer (taxpayer) welfare. This occurs because of the reduction in adverse health effects. Depending on the revenue policy of the government, it could lead to a possible reduction in consumer taxes, since producers are now paying an additional tax (the double-dividend hypothesis, which is described in greater detail later in this chapter). Although there is a decrease in the producer surplus (and the obvious consumer surplus), the overall social welfare has increased because of the reduction in the externality costs.[2] Adding all of these surplus changes together, and subtracting the transfers to the government (i.e., taxes), yields the net social cost of the policy.

If instead of an emissions tax, a firm is required to install pollution control devices, the private compliance costs will raise the firm's supply curve (or MPC) up by the amount spent on the new equipment. Under a permit system, where a set number of permits are issued for each unit of pollution, and firms are allowed to buy and sell these permits, then each firm will consider buying permits if the private cost of a permit is less than the unit cost of reducing pollution. Conversely, a firm that can reduce its pollution by less than the cost of a permit will consider selling its "extra" permits. In both cases, the firms' MPC curves will shift up by the price of the permits, just as it did in the case of an emissions tax.

8.3 A General Approach to Social Cost Analysis

The challenge in developing an estimate of the social cost of an environmental policy is to consider the market(s) being affected by the policy, assess the available data and analytic methods, and adopt an analytic approach that will yield an estimate suitable for use in the benefit-cost analysis. An important requirement in measuring social costs is to characterize the supply and demand equations of the regulated market or behavior. This section briefly reviews the estimation of supply and demand equations and their relevance to social cost, but concentrates on the variety of social costs that may be encountered from different types of environmental policies: (1) direct social costs, (2) transitional costs, and (3) indirect costs. Finally, some other issues that arise in characterizing and presenting social costs are examined, including discounting, difficulties in monetizing costs, consideration of sensitivity analyses, and simplifying market effects.

8.3.1 Estimating the Supply and Demand Equations of All the Affected Markets

Empirical estimates of the supply and demand curves for each market are usually needed to calculate the social costs of proposed regulations and policies. In addition to private sources, government reports and academic studies can provide useful information needed to estimate the

[2] If the regulation causes social costs to be greater than the MSD, then net social welfare may actually fall because of the new regulation; one cause being the diversion of investment capital to excess pollution control and away from its highest valued use.

supply and demand equations.[3] Solving these equations will yield equilibrium quantities and prices in each market that approximate the baseline figures.

In most situations, the supply and demand functions can be derived based on engineering cost estimates. For example, a step-function demand curve for a particular good can be computed based on the prices at which various segments of the market will turn to substitute goods. This technique is relatively more successful for products that are used as inputs to other processes and consumer goods that have well-defined alternatives. Similarly, an estimate of producer surplus can be derived based on the value of plant and equipment dedicated to supplying a particular good, and the ease or difficulty with which this capital can be deployed in other markets to supply different goods. In the long run, the supply curve is often assumed to be horizontal.

Information on the elasticity of demand is available for the aggregate output of most industries. When such information is unavailable, as is often the case for intermediate goods, the elasticity of demand may be quantitatively or qualitatively assessed. Econometric techniques, such as multiple regression, can be used to estimate a demand curve when sufficient data are available. For example, when dealing with intermediate products, econometric models can be constructed using engineering cost data to estimate both the supply and demand curves. In general, econometric tools are frequently used to estimate supply and demand equations and the factors that influence them.

Information on the availability of product or service substitutes, the impact of price increases on final goods (where the product or service is an intermediate good), the amount of a person's income devoted to the good or service, and the necessity of the final product or service can be used to qualitatively assess demand elasticities. The estimate selected for the point elasticity should be consistent with the equilibrium point (the time allowed for adjustments to occur) used in the analysis.

Estimating the equations that govern market supply and demand may be time and resource intensive, in addition to the formidable tasks of developing the means to structure and compute the considered models. While many types of markets have been researched in detail by the academic community, others may be too new to have much information available. It may be difficult to obtain data from the affected firms or industries because of confidentiality provisions or the proprietary nature of some data and models. Achieving sufficiently reliable results will often depend on the quality of the data, and overcoming problems with data will be a primary hurdle in many social cost analyses.

8.3.1.1 Definition of Elasticities

In general, economists use the term "elasticity" to refer to the sensitivity of one variable to changes in another variable.[4] The price elasticity of demand (or supply) refers to changes in the quantity demanded (or supplied) that would result from a change in the price of a good or service. Changes are measured assuming all other things, such as incomes and tastes, remain constant. Demand and supply elasticities are rarely constant and often change depending on the quantity of the good. Therefore, when calculating elasticities, it is important to state the price and quantity of the good.

"Elastic" demand (or supply) indicates that a small percentage increase in price results in a larger percentage decrease in quantity demanded (or supplied). How much of the price increase that will be passed on to consumers is determined by the elasticity of demand relative to supply (as well as the degree of competition within the industry and existing price controls). All other things equal, an industry facing a relatively elastic demand is less likely to pass on costs to the consumer because increasing prices will result in reduced revenues.

[3] Sources can include trade publications, financial studies, and data collected through surveys administered in support of the regulation (e.g., Section 308 surveys administered under the Clean Water Act for promulgation of effluent discharge limitations). Government agencies and the private sector also publish data and studies on the economic activity of the public and private sector and households. A more complete listing of examples is provided in Exhibit 9-3 on sources used to prepare economic profiles of industries. Additional illustrations can be found in existing EPA economic reports, several of which are referenced later in this chapter.

[4] *Own price elasticity of demand* is defined as the percentage change in quantity demanded divided by the percentage change in price. *Own price elasticity of supply* is defined as the ratio of the percentage change in quantity supplied divided by the percentage change in price.

8.3.1.2 Determinants of Demand Elasticity

Among the many variables that affect the elasticity of demand are: (1) the availability of close substitutes, (2) the percentage of income a consumer spends on the good, (3) how necessary the good is for the consumer, (4) the amount of time available to the consumer to locate substitutes, (5) the level of aggregation used in the study, and (6) the expected future price of the good. In this section, only the first four will be discussed.

- **The availability of close substitutes** is one of the most important factors that determine demand elasticities. A product with close substitutes tends to have an elastic demand, because consumers can readily switch to substitutes rather than paying a higher price. Therefore, a company is less likely to be able to pass through costs if there are many close substitutes for its product.

- **Whether the affected product represents a substantial or necessary portion of customers' costs or budgets** is another factor that affects demand elasticities. When price increases occur for products that represent a substantial portion of downstream producers' costs or consumers' budgets, these producers or consumers may be more likely to seek alternatives. Where the product subject to the price increase is less important in customers' budgets, customers may be less motivated to use substitutes (even if they are available) or to forego consumption entirely. A similar issue concerns the type of final good involved. Reductions in demand may be more likely to occur when prices increase for "luxuries" or optional purchases than for basic requirements.

- **The time frame considered** is a third important factor in determining elasticity. Elasticities tend to increase over time, as firms and customers have more time to respond to changes in prices. A company facing an inelastic demand curve in the short run may experience greater losses in demand in the long run, as customers have time to make adjustments that allow use of substitutes or as new substitutes are developed.

  It is important to keep in mind that elasticities differ at the firm versus the industry level. For example, if twenty companies are producing pesticide formulations that are equally effective, each firm may face an elastic demand curve because of competition within the industry, although the industry as a whole may face an inelastic demand curve for its products as a group. In this example, it would not be appropriate to use an industry-level elasticity to estimate the ability of only one firm to pass on compliance costs when its competitors are not subject to the same costs.

8.3.1.3 Determinants of Supply Elasticities

The elasticity of supply depends, in part, on how quickly costs per unit rise as firms increase their output. Among the many variables that influence this rise in cost are:

- the availability of close input substitutes;
- the amount of time available to adjust production to changing conditions;
- the degree of market concentration among producers;
- the expected future price of the product;
- the price of related inputs and related outputs; and
- the speed of technological advances in production that can lower costs.

Supply elasticities will tend to increase over time as firms have more opportunities to renegotiate contracts and change production technologies.

Characteristics of supply in the industries affected by a regulation can be as important as demand characteristics in determining the economic impacts of a rule. For highly elastic supply curves, it is likely that costs will be passed through to consumers. The main determinants of industry supply curves are the structure of costs and the time period of the analysis. Industry supply curves are defined as the aggregation of the supply curves of individual firms within an industry.

If detailed financial profiles of individual establishments or categories of establishments and production data are available, they can be used to define an industry supply curve. Explicit information on the cost structure of an industry is very useful in predicting impacts more precisely than is possible using industry average data. A given firm may experience significant impacts if it is already a relatively high cost producer. Such firms would be more vulnerable to closure if subjected to high compliance costs.

8.3.1.4 Obtaining Supply and Demand Elasticities

Elasticity estimates may be obtained from existing literature or from original research. The use of published estimates avoids the time and expense of gathering the necessary data. Sources for published estimates include previous agency rule makings or relevant studies found in the economics literature. The analyst will have to employ careful judgement in deciding whether and how to use elasticity estimates from previous studies. Estimates should be drawn from studies based on:

- similar market structure and level of aggregation;
- sensitivity to potential differences in regional elasticity estimates;
- current economic conditions; and
- appropriate time horizon (i.e., short or long run).

This is not an exhaustive list of issues which must be considered in applying existing estimates to new analyses. There are a number of statistical and technical issues that may influence the quality of elasticity estimates. Relevant texts cited below should be consulted and technical assistance sought when necessary.

New or original estimates of elasticities are derived from demand and supply functions for goods or services that have been estimated using econometric methods. Econometrics is the use of statistical analysis in applied economic research. For example, the demand for a good or service is often estimated as a function of its price, the price of related goods (substitutes and complements), consumer demographic characteristics, as well as variables that may represent institutional or technological characteristics of a market. Supply and demand elasticities may be derived from a variety of functional forms that embody various assumptions about the relationships between the data. Methods of calculating elasticity estimates differ according to the specification of the function. The analyst should consult relevant texts and seek technical assistance.

The availability of sufficient data, both in terms of quantity and quality, is the first threshold that determines whether econometric tools can be used. Only with sufficient data can elasticity estimates be considered reliable. The analyst should carefully document data sources. Once the decision to employ econometrics is made, there are a number of issues which the analyst must address, including the choice of an appropriate modeling technique, proper functional form, and ensuring that the mathematical properties required for the chosen technique to yield proper results are achieved. For example, ordinary least squares (OLS) requires that:

- values of independent variables are *non-stochastic or fixed*;
- expected mean value of the error term is zero;
- expected value of the variance of the error term is constant;
- no correlation exists between error terms; and
- no correlation exists between error terms and independent variables.

If any of these conditions are violated, the analyst will have to make a corrective adjustment to the OLS or consider an alternative econometric technique. For example, if one of the independent variables is *endogenous*, the first and last condition will be violated, resulting in a biased and inefficient coefficient estimate. In the context of estimating a demand function, the price variable is likely to be endogenous, which would render the coefficient estimate and derived elasticity incorrect. A method known as two-stage least squares (TSLS) represents one means of accounting for endogeneity. The number of potential econometric approaches, mathematical requirements, and corrective measures is beyond the scope of this guidance document. Analysts should consult relevant texts for a more thorough discussion of all of these issues.[5]

[5] For detailed review of econometric modeling and technical issues see Greene (1996), Maddala (1992), or Pindyck and Rubinfeld (1991). Kennedy (1998) provides a more intuitive discussion in the main text with detailed technical notes provided in appendices.

8.3.1.5 Uses and Substitutes Analysis

A "Uses and Substitutes Analysis" may provide useful information on the characteristics of demand as a supplement to or substitute for elasticity estimates.[6] This is "...an in-depth examination of each significant use of the substance in question, and an assessment of the costs, performance, and useful life of substitutes, on a product-by-product basis."[7] A "Uses and Substitutes Analysis" includes four steps:

1) define markets and segments of markets that are relatively homogeneous;

2) assess the cost and performance characteristics of the products in question;

3) identify the most appropriate substitutes; and

4) estimate the incremental costs and performance characteristics of the substitutes in each specific application.

The results of the analysis can then be used to generate demand functions, based on the price at which substitute products become economical for different uses. This analysis can be time and information intensive and may produce relatively crude results. It is nonetheless a useful alternative to estimating demand functions when elasticities are not available.

8.3.2 Determining the Different Types of Social Costs

Having established measures of supply and demand, the analysis then considers how equilibrium price and quantities will change from measured baseline conditions. Social cost changes in each of the affected markets are assessed by examining the direct, indirect, and transitional effects that occur as a result of the new policy. The types of social costs that need to be examined to determine how the supply and demand equations change are summarized, with examples in Exhibit 8-2. A short description of direct costs, which include private and public compliance costs, government regulatory costs, and other types of social costs, is presented. Other social costs less routinely included in empirical analyses of social costs, including indirect costs and the transitional costs, are then reviewed.

8.3.2.1 Direct Social Costs

The direct social costs of a new environmental policy arise from: (1) changes in the private real-resource compliance costs, (2) additional government regulatory costs, (3) social welfare losses, and (4) transitional social costs. The largest fraction of direct social costs arises from the real-resource compliance costs due to the new regulation. These new compliance costs arise from the installation, operation, and maintenance of new capital equipment, or are a result of changes in the production process that raise the price of producing the good.

The additional compliance costs can be used to estimate the new equilibrium price and quantity in the affected markets which will change social welfare. However, these changes will affect other markets, resulting in further price and quantity changes in other goods, giving rise to additional changes in social welfare. The significance of the changes in other markets will influence the type of model necessary for the economic analysis (see Section 8.4, "Modeling Tools"). Changes in social welfare also result from increased government regulatory costs and transitional costs from plant closures and unemployment.

- **Private real-resource compliance costs** can arise from: (1) the capital costs associated with the purchase, installation, operation, and maintenance of new pollution control equipment, (2) changes in the inputs or mixtures used in the production process, or (3) the capture of waste products that can either be disposed of, sold, or reused.

  Real-resource costs are theoretically straightforward to calculate if they arise from the purchase of new pollution control equipment. For example, having information on the number of factories and the price of purchasing and operating new equipment required to meet a policy would provide a means of estimating the compliance costs for the industry. However, since all factories are not identical, costs may be estimated based on cost studies of representative factories

[6] Uses and substitutes analysis is described in Arnold (1995).

[7] Ibid., p. 21.

chosen by random sampling procedures, which can be extrapolated to the universe of affected factories.

Additional costs involve the operating expenses, maintenance, and training associated with the new equipment. Further costs may occur from maintenance changes in other equipment. Also, additional administrative costs may be associated with obtaining permits and preparing required monitoring reports.[8]

In the two other methods of compliance, the private costs may actually be negative and thus need to be included for an accurate estimate of social costs. When waste products are captured and then disposed of, sold, or reused, the cost calculation is also straightforward. Disposal charges are easily determined and the selling price of the waste product (if it is used as an input for other goods) can also be obtained. However, if the production process is changed so that different inputs are used or the mixture of the inputs is altered, the costs involved will be difficult to determine before the change takes place. In addition, the changes may be considered proprietary information.

- **Government regulatory costs** are incurred by federal, state, or local governments to administer and enforce new policies. Government regulatory costs include: administration, training, monitoring/reporting (if they are not included in compliance costs), enforcement, litigation, and the cost of developing and distributing permits. These incremental costs must be financed through additional taxation or other governmental financing mechanisms.

Because they are hard to translate into producer and consumer surplus terms, governmental administration and enforcement costs are typically examined in terms of their dollar costs and staffing requirements (expressed as full-time equivalent employment (FTEs)). Ultimately, these costs are borne by taxpayers unless other administrative costs are reduced to accommodate a new policy. Since government costs are usually small compared to the explicit compliance costs, they are not usually included in partial

Exhibit 8-2 Examples of Social Cost Categories

Social Cost Category	Examples
Real-resource Compliance Costs	• Capital costs of new equipment • Operation and maintenance of new equipment • Waste capture and disposal, selling, or reuse • Change in production processes or inputs • Maintenance changes in other equipment
Government Sector Regulatory Costs	• Training/administration • Monitoring/reporting • Enforcement/litigation • Permitting
Social Welfare Losses	• Higher consumer and producer prices • Legal/administrative costs
Transitional Social Costs	• Unemployment • Firm closings • Resource shifts to other markets • Transaction costs • Disrupted production
Source: Adapted from Harrington et al. (1999).	

[8] A good recent illustration of the measurement of compliance costs can be found in EPA (1997), *Economic Analysis for the NESHAPS for Source Categories: Pulp & Paper Production; Effluent Limitations Guidelines, Pretreatment Standards: Pulp, Paper, and Paperboard Category, Phase I*. For useful empirical presentations on engineering approaches, see Vatavuk (1990) for air pollution controls and EPA (1984) for a wider variety of pollution control technologies.

equilibrium models. However, if they are significant, they should be estimated separately and added to the surplus-based social cost estimates.

Monitoring and enforcement costs, incurred by the government, can be either (1) the opportunity costs of other activities that are discontinued because of fixed government budgets, or (2) the private costs imposed on taxpayers to support the increased government expenditure necessary for the program. The costs of government monitoring and enforcement efforts are normally based on the cost of necessary administrative activities.[9]

- **Social welfare losses** occur when real-resource compliance costs result in higher prices for the good or service and when additional government regulatory costs result in higher taxes passed on to the consumer. New regulations may lead to increased legal and administrative costs for the government, as well as for the regulated entities. The change in social welfare resulting from an increase in taxes or fees assessed in order to pay for government regulatory costs will typically be small relative to social welfare losses attributable to the real-resource compliance costs.

  If the imposition of real-resource compliance costs leads to an increase in the price of the good, this will lead consumers to either buy less or switch to substitutes, thereby leading to a fall in the consumer surplus. The amount of the private costs passed through to the consumer is determined by the market structure, and the elasticities of demand, supply, and income. Once the prices, quantities, and elasticities are known, the process of calculating changes in producer and consumer surpluses is also theoretically straightforward.[10]

- **Transitional effects** vary depending on the length of the time period examined; therefore, social cost analyses should be explicit about the time frame being studied. In the short run, the private annualized costs of compliance, both for consumers and producers, will be higher relative to the annualized long-run costs. This is because the short-run analysis will not provide for possible adjustments in the production process, or allow consumers to find substitutes. Some workers may become unemployed in the short run, but will almost certainly find other jobs in the long run.

  However, over time the impact of a policy can easily spread out to a variety of markets and result in a number of unanticipated adverse effects. Therefore, it is not always appropriate to assume that social costs arising in the short run as a consequence of transitional effects will be resolved in the long run. For EPA economic analyses, the four transitional effects most frequently considered include: (1) plant closings and resultant unemployment, (2) resources shifting to other markets, (3) transactions costs associated with setting up incentive-based programs, and (4) disruptions in production.

  Firm closings and unemployment: In the simplest static models, the time frame is assumed to be a period of time in the near future (e.g., the first year after a new policy is promulgated). Surplus-based measures of social cost are therefore short-run estimates. But as time passes, adjustments are likely to occur. Workers who suffer transitional unemployment will usually find new jobs, and new plant and equipment installed in the future might require relatively less costly pollution control. These long-run changes should be considered as the yearly social costs of a policy are calculated into the future.

  In most cases, involuntary unemployment and plant closings are consequences that are difficult to model using a conventional partial equilibrium framework (which will be discussed in the following section on modeling tools). Predicting these specific consequences would require far more detailed analysis and data than are usually available for practical assessments. Unemployment rates for each group of workers, the duration of unemployment, and the cost of job training programs are just some of the factors that

[9] A useful illustration of the measure of government regulatory costs for a rule can by found in EPA (1995a), *Economic Analysis of the Title IV Requirements of the Clean Air Act Amendments*.

[10] An example of the steps taken to estimate the measurement of social welfare losses from a rule is EPA (1998), *Economic Analysis of Effluent Limitation Guidelines and Standards for the Centralized Waste Treatment Industry*.

need to be taken into account when estimating how the transitional costs decline over time.

These temporary effects are typically assessed and reported as part of an "economic impact" of the policy, and are incorporated into the development of the social cost section of an economic analysis. Chapter 9 of this document reviews such methods to assist in detecting situations in which a policy's private costs are sufficiently large to induce social costs from occurring as a consequence of business closures, reduced employment, or other such impacts.

Shifts of resources to other markets: These shifts occur when the payments to factors of production (labor, land, and capital) are reduced. These shifts are partly responsible for the decreased output level of the product or service. Those that remain earn less than before, at least in the short run, which is reflected in the lower net price received by producers for the good or service. Some of the resources no longer employed in producing this good or service might even become unemployed for a while, such as labor, or be permanently and prematurely scrapped, such as plant and equipment. These and other real-world phenomena can change the position and slope of the supply functions in other markets. Likewise, consumers of the product either pay more for the same good or purchase substitutes that are less suitable or more costly, which can change the position and slope of several demand functions. The analysis of these types of effects is also treated more fully in Chapter 9.

Transaction costs: These costs are encountered with incentive-based policies, such as with a tradable permits program. A market must be established so that the efficiency gains from trading permits are maximized, and rules for trading are developed that enable the market to function under the rules of perfect competition. Therefore, initial short-run costs associated with setting up the market will be high, but are expected to diminish over time as the created market begins to function with less government oversight. The private cost of buying and selling permits will then become similar to the purchase of any other resource needed to produce a good or service.

Disruptions in production: This may take place when new equipment is installed or new production processes or inputs are applied. These costs can be estimated as the amount of time the production line is stopped or slowed down to allow for the necessary changes to comply with the new policy regulations. However, if the changes are made during previously scheduled down-time or required maintenance, then downward adjustments should be made to the estimated costs to reflect this.

To conclude, in many cases transitional costs are considered to be small enough that their inclusion in the overall social cost estimate would not appreciably alter the quantitative conclusions. However, when these are expected to be significant, the costs should be estimated. For example, lost wages and job search costs during the time workers are unemployed can be used as a proxy for this transitional social cost. Similarly, the value of prematurely retired plant and equipment can be calculated and added to the surplus-based social cost estimates to capture this transitional effect, as long as this is not reflected already in the supply and demand framework.

8.3.2.2 General Equilibrium (Indirect) Effects

Other possible components of social costs, such as effects on product quality, productivity, innovation, and market structure, can require fairly complex dynamic models to quantify. Although most individual regulatory policies will not have such dramatic effects, these costs can be quite significant in certain instances, such as when a policy's requirements delay industrial projects or affect new product development. Such policy effects have implications for future social costs but are difficult to measure and express in social cost terms. However, an effort should be made to qualitatively describe these factors and look at approaches that can quantify these effects when data and resources can support this level of detailed analysis of social costs.

- **Changes in market structure** may occur if the expense of pollution control is sufficiently high that it drives out enough firms to cause changes in the market concentration and competitiveness of firms remaining in the industry. Such a change often results in shifts of both firm and industry supply curves, which can lead to changes in output and prices in several markets affecting both producer and consumer surpluses.

- **Labor and capital productivity** may decrease under new regulations. For example, the administrative costs of monitoring emissions and filing reports with regulatory agencies may require firms to hire more workers whose labor does not increase productivity (as measured by labor employed relative to produced output). Pollution control devices or restrictions on the use of products may cause lower levels of output relative to unconstrained production processes. For example, placing restrictions on pesticide use may reduce the yield of crops susceptible to pest damage, holding other factors of production (e.g., labor, fertilizer) constant. In each case, however, private costs are captured by changes in the supply and demand curves of the product, and therefore care should be taken to insure that social costs associated with productivity losses are not double counted with other social cost estimates.

- **Discouraged investment** may occur if research and development funds are reallocated to meet additional compliance costs. This may result in decreases in technological innovation and product quality. The latter can be modeled as the reduced amount consumers are willing to pay for the low quality good, relative to what they were willing to pay for the original, higher quality good. In practice, changes in technological innovations are not commonly analyzed in most economic models used in benefit-cost analyses of individual regulations and policies.

8.3.3 Other Issues Arising in Presentation of Social Costs

Four additional issues to note arise in the organization and presentation of social costs, several of which have also been raised earlier in this document in connection with the measurement of social benefits. These issues discussed here on social costs include: (1) discounting, (2) difficulties valuing some social cost categories, (3) conducting sensitivity analyses, and (4) simplifying market effects.

- **Discounting:** Social discounting procedures for economic analyses are reviewed in considerable detail in Chapter 6. For purposes of computing the social costs of environmental policies, costs should be discounted using the methods and social discount rates discussed in that chapter. This is the case regardless of the methods used to estimate social costs. Social costs can be estimated in detail year-by-year, or estimated using growth rates, or merely assumed to be constant. These streams of social costs can then be adjusted to yield: (1) discounted present value, (2) future value, or (3) the annualized cost of the policy. All three approaches are different ways to express the same concept and choosing which method to present the results will depend on the method that most effectively allows comparisons among the options and the measurement of net benefits.[11]

- **Difficulties of valuing social costs:** Some consequences of environmental policies are difficult to represent in the definitive, quantitative terms of conventional social cost analysis. Irreversible environmental impacts, substantial changes in economic opportunities for certain segments of the population, social costs that span very long time horizons, socioeconomic effects on communities, and poorly understood effects on large-scale ecosystems are difficult to summarize in a quantitative benefit-cost analysis. Some alternative techniques for measuring and presenting these effects to policy makers are reviewed in section 7.6.3 of the benefits chapter that discusses measuring ecological benefits. The relative significance of social cost categories that are not quantified—or are quantified but not valued—should be described in the analysis.

- **Sensitivity analysis:** The estimates in the social cost analysis will not be known with certainty. In fact, some data and models will likely introduce substantial uncertainties into the estimations of social costs. Numerous assumptions are made in regard to

[11] Many EPA analyses typically prepare an annualized cost estimate, since this measure is one of several used to determine whether rules require additional review and oversight, and is used to help establish the scope of the economic analysis to be conducted (e.g., the social cost threshold of $100 million in annual costs is used to identify rules that require a benefit-cost analysis under the provisions of EO 12866).

the baseline, predicting responses to policy, and the number of affected markets. Therefore, the conclusions drawn in the benefit-cost analysis will be sensitive to the degree of uncertainty present and the assumptions that were made. Reporting the uncertainty of the data, the assumptions used, and how the uncertainty and assumptions affect the results are important components of the presentation of social cost. Section 5.5 outlines the process of analyzing and presenting uncertainty.

- **Simplifying market effects:** Given the complexity of modern economies, measuring and predicting all of the consequences of a particular action would involve a significant effort. The central question explored in this section is whether some or all markets indirectly affected by a policy must be analyzed to obtain a measure of social costs suitable for a benefit-cost analysis, or whether the calculation of social costs can be limited to an assessment of the directly affected markets without introducing unacceptable biases and errors into the analysis.

In general, the social cost of a policy can be measured exclusively by changes that occur in the markets directly targeted by a policy, as long as significant net changes in social welfare are not generated in indirectly affected markets. If price changes in other markets generate both gainers and losers among the producers and consumers, then they may offset each other in a social cost analysis as transfers.[12] However, if there are strong reasons to believe that conditions in other related markets might generate important net social welfare consequences, these should be examined. If a policy indirectly increases or decreases the quantity of a good that is consumed, whose production or consumption involves an externality, then this results in net social welfare effects that may be worth considering when calculating total social costs (and benefits).

8.4 Modeling Tools

The following section first focuses on the basic framework common to all models used to estimate social costs, while the remaining sections examine the models commonly used: (1) direct compliance cost methods, (2) partial equilibrium models, (3) multi-market models, and (4) computable general equilibrium models.

8.4.1 The Basic Framework

Benefit-cost models must predict what actions firms are likely to choose when attempting to comply with a new policy and what the compliance costs of those actions will be. Normally, these are based on engineering or process cost models that examine firms' alternative compliance methods. Engineering cost estimates typically specify the capital, operating, and maintenance costs that are likely to occur in adopting different pollution control strategies. When possible, these initial engineering cost estimates should include the expected level of compliance costs, as well as reasonable lower and upper bounds for purposes of sensitivity analysis.

In addition to the preliminary engineering or other estimates of the social costs of various compliance strategies, other costs may be significant. As noted earlier, for some market-based approaches, transaction costs can often be substantial. For example, when setting up the market for a permit trading system, determining how many permits to purchase or sell can involve detailed cost modeling and forecasting, in addition to the social costs associated with operating the trading system. When these costs are likely to be significant, they should be estimated in addition to the basic private real-resource costs of capital and the operating costs of alternative compliance methods.

8.4.2 The Direct Compliance Cost Method

In some cases, social costs are estimated using the *direct compliance cost* method. This is the simplest approach used in estimating social costs. Under this approach, the social cost for a policy is simply set equal to the initial engineering or other compliance cost estimates for the compliance options the firms are likely to adopt; no additional modeling is undertaken. If only compliance costs are calculated, the private (compliance) costs are likely to

[12] This conclusion regarding the net social welfare implications of price changes in related markets requires some qualification. Even when non-zero welfare effects are produced by price changes in related markets, they are likely to be small relative to the estimated producer and consumer welfare effects in the directly affected markets. See Arnold (1995) for more discussion of related markets and welfare measurement.

be overestimated. This is because private costs are computed for the pre-policy level of output under the implicit assumption that there is no substitution away from the affected products or activities into other relatively less expensive ones. That is, firms do not make any capital or labor adjustments in their production processes. In addition, when the resulting changes in consumer surplus are calculated at the new higher prices, consumer welfare losses are also likely to be overestimated since changes in consumer behavior will not be taken into account.

Nevertheless, using direct compliance costs as an approximation of actual social costs may be reasonable for a policy when price and quantity changes are small, and there are few indirect effects. However, if consumers can easily switch to substitute goods, this adjustment will make the actual social cost of the policy significantly less than indicated by the direct compliance cost estimates. Likewise, if firms can find less costly substitutes for their inputs or production processes, which have been made more expensive by the new regulations, then compliance costs will be an overestimate of the actual social costs.

8.4.3 Partial Equilibrium Analysis

Because of the limitations of using direct compliance costs as a measure of social costs, an alternative approach is to model the economic effects of these compliance costs on producers and consumers using a *partial equilibrium* supply and demand model of the affected markets. This allows for a more complete analysis of social costs and their incidence. "Partial" equilibrium refers to the fact that the supply and demand functions are modeled for just one or a few isolated markets and that conditions in other markets are assumed either to be unaffected by a policy or unimportant for social cost estimation.

For example, if using a partial equilibrium supply and demand framework, a new environmental policy that increases production costs will cause a change in the supply function. The demand function, the old and new supply functions, prices, and quantities can then be used to compute changes in producer and consumer surpluses. If the relevant markets are evolving over time, this technique can be applied in each future time period using new supply and demand functions. This makes it possible to estimate the changing distribution of social costs over time.

The practical difference between the results of the partial equilibrium supply and demand-based modeling and the direct compliance costs approach depends on the nature of the policy and the magnitude of its effects. For small compliance costs, price and quantity movements are likely to be minimal, so the social cost estimates derived from the partial equilibrium model framework will not be significantly different from the results obtained from the direct compliance cost method.

For policies with larger compliance costs, price and quantity movements could be more substantial. The estimated social costs using the supply and demand framework in these cases may be considerably less than those suggested by the simpler direct compliance cost approach. Moreover, policies that effectively ban products or activities cause the loss of all producer and consumer surpluses in these markets. Therefore, it is difficult to calculate social costs of these policies without an explicit supply and demand framework.

Analyzing the effects of a policy using a partial equilibrium model of the directly affected markets is appropriate when the ramifications in indirectly affected markets do not generate net social costs. It is also a reasonable framework as long as the social costs imposed by a policy are small and do not significantly alter other markets or produce measurable macroeconomic effects (e.g., changes in national unemployment levels).

In most cases, a conventional partial equilibrium framework comparing the pre-policy baseline with the expected results of a new environmental policy will suffice for an economic analysis. For analyzing environmental policies that pose very large consequences for the economy, computable general equilibrium modeling is an alternative technique that is particularly useful and is discussed later in this chapter.[13]

The partial equilibrium framework is a commonly used theoretical tool for modeling and measuring the social costs of environmental policies. In theory, a variety of

[13] Useful recent illustrations of partial equilibrium analyses prepared in support of environmental policies include EPA (1998) *Economic Analysis of Effluent Guidelines and Standards for the Centralized Waste Treatment Industry*, and EPA (1996) *Economic Impact Analysis of Proposed NESHAP for Flexible Polyurethane Foam*.

social costs can be observed and calculated using this technique. Even transitional effects that result in short-run social costs, such as premature capital equipment retirement and relatively brief spells of involuntary unemployment, can be modeled and estimated using this framework. Thus, the approach offers a theoretically sound, if limited, method for conceptualizing the consequences of an environmental policy and measuring their social costs.

Deriving the supply and demand functions is the foundation of benefit-cost analysis and is necessary in all economic models used to analyze social costs and benefits. However, because of its importance and the uncertainties associated with estimating supply and demand functions, it may be useful to evaluate key assumptions with sensitivity analyses and develop a range of estimated social costs.

The typical analysis is performed assuming a competitive market, although unusual circumstances may require relaxing this assumption. Even should competitive market conditions fail to hold, partial equilibrium analysis can be adapted to analyze varying market conditions that may more closely reflect real-world conditions. It is useful to indicate when social benefits or social costs have been overestimated or underestimated because of biases caused by market distortions. However, the principles underlying partial equilibrium analysis can serve as a useful model to evaluate the real-resource costs of many of EPA's regulations and policies.

As previously discussed in Section 5.6, "Emerging Cross-Cutting Issues," environmental policies usually can be analyzed assuming a first-best regulatory setting, although actual conditions reflect a second-best world in light of taxes placed on a variety of goods and services. Therefore, it is conceivable that a regulatory policy in one sector may have effects in labor markets and other sectors of the economy. Thus, examining costs in only the final goods market may cause costs (or even benefits) to be underestimated.

The scope of the regulatory program is likely to be proportional to the effects experienced in other sectors of the economy. Therefore, the larger the program, the more important it is to examine several markets to accurately estimate costs and determine (1) tax interaction effects, (2) changes in technology, and (3) the effects on firms' research and development decisions. Thus, multi-market models are needed for regulatory policies that may have large economic effects on several sectors of the economy.

8.4.4 Multi-Market Models

Multi-market models go beyond partial equilibrium analysis by extending the inquiry to more than just a single market. Multi-market analysis attempts to capture at least some of the interactions between markets. However, unlike the general equilibrium models discussed in the next section, multi-market models do not attempt to incorporate a representation of the entire economy.

An example of the use of a multi-market model for environmental policy analysis is contained in a report prepared for EPA on the regulatory impact of controls on asbestos and asbestos products (EPA 1989). The model developed for the study describes the interactions between the asbestos fiber market and markets for the goods that use the fiber as an intermediate input. The collective demands for final goods that use asbestos create a derived demand for asbestos fiber. The price of the fiber is determined through the interaction between the demand and supply schedules for asbestos. Changes in this price in turn influence the prices and demands for the downstream goods. The specification of the links between the input and output markets is especially important for simulating alternative regulatory policies, including interventions in both the input market (caps on the usage of asbestos fiber) and in the output market (bans on some of the goods that use asbestos as an input), as well as combinations of the two. The model was then used to compare the efficiency losses under various regulatory scenarios.

8.4.5 General Equilibrium Analysis

Although the use of a partial equilibrium or multi-market model may be appropriate when policies are likely to affect a limited number of markets, they are not able to capture interactions between a large number of sectors. Many environmental policies, such as energy taxes, can be expected to impact a large number of sectors both directly where the policy is applied, and indirectly through spillover and feedback effects on those and other sectors. A strength of general equilibrium models is their ability to account consistently for the linkages between all sectors of the economy. Three types of general equilibrium models that have been used for the analysis of social costs are input-output models, linear programming models, and computable general equilibrium (CGE) models.

8.4.5.1 Input-Output (I/O) Models

The central idea underlying I/O analysis is that in modern economies, production activities are closely interrelated. An input-output table represents the flow of goods and services through the economy, usually measured as transactions occurring over the course of a year. In addition to the primary factors of capital, labor, and land, most productive sectors use many different intermediate inputs. In an I/O table, the column associated with a particular sector lists the value of the individual intermediate and primary inputs consumed by that sector. The row associated with an individual sector lists the value of that sector's output purchased as both intermediate inputs and final demand. For each sector in a table, the column sum represents the total costs of production and the row sum represents total expenditure on that sector's output. A key feature of I/O tables is that, by definition, for every sector, total costs must equal total expenditures during the year.[14]

An I/O table can be turned into a simple linear model through a series of matrix operations. The intermediate inputs matrix defines a matrix of technical coefficients, based on the assumption that inputs to production are consumed in fixed proportions to output and that there are constant returns to scale. The model is manipulated by making exogenous changes to the vector of final demands. The model will then calculate how much of each of the intermediate goods is required to produce the new final demand vector. The sum of the intermediate inputs required plus final demand is equal to total output for the year.

I/O models have a long history in environmental policy analysis. Leontief (1970) showed how it was possible to augment the basic I/O model with an additional set of coefficients for pollution generation and/or abatement. When a set of pollution coefficients has been defined, an I/O model can then produce an estimate of the quantity of pollution that would be generated along with a given amount of final demand or total output. The quantity of pollution generated may be specified in either monetary terms (as damages) or in physical units.

Some economic research firms use I/O models to provide upper bound estimates on price effects. Others use I/O models to look at the related markets and their potential significance prior to adopting a partial or general equilibrium model. The I/O approach has also been extended further to include non-market, ecological commodities such as ecosystem services.[15]

Although I/O models can be a useful as a consistency check or a first-order approximation, they have a number of shortcomings that limit their applicability as a predictive tool:

- Given that prices are normally assumed to be fixed and do not adjust to indicate scarcity, there is nothing to ensure that the total demands generated by manipulation of the model are consistent with the actual productive capacity of the economy.
- The fixed coefficients assumption leaves no scope for substitution of inputs in production.
- Since there is no producer or consumer behavior built into I/O models, simulation of policy interventions that would affect those agents is of limited value.
- Because the construction of an input-output table is a costly and time-consuming process, usually requiring a specialized survey, the application of input-output analysis to environmental policy making will normally only be possible when an appropriate table already exists. More importantly, since input-output tables are used in linear programming and computable general equilibrium models, this last shortcoming is shared by these models as well.

[14] A general reference on input-output models is Miller and Blair (1985).

[15] A discussion of the application of input-output models to environmental policy analysis, with a number of examples, is provided in Chapter 7 of Miller and Blair (1985). Another example applied to the environmental protection industry is EPA (1995b), *The U.S. Environmental Protection Industry: A Proposed Framework for Assessment*.

8.4.5.2 Linear Programming (LP) Models

I/O models are driven by exogenous changes in final demand. Since they do not contain an objective function, I/O models are difficult to use for decision making among multiple alternatives. However, it is possible to extend the basic I/O model into a LP model by incorporating an explicit objective function and a set of inequality constraints.[16]

In addition to the usual economic variables, the objective function may be specified to include a number of environmental variables, such as the discharge of air or water pollutants. The specification of multiple inequality constraints allows for a great deal of flexibility in the application of LP model (because of this flexibility, EPA's Office of Air and Radiation has used linear programming models for many years). Shadow prices generated in the dual form of LP models have a limited relationship to market prices and may sometimes be useful as indicators of the importance of the individual constraints. Sensitivity analysis can be conducted by varying key parameters in the model.[17]

The flexibility in the specification of LP models is also something of a liability of the approach. The problem is that the selection of the constraints used is often *ad hoc* and may influence the model solution. As with many linear models, there is often a tendency towards unrealistic solutions, such as excessive specialization in production or trade. Finally, the lack of realistic consumer and producer behavior is carried over from I/O models.

8.4.5.3 Computable General Equilibrium (CGE) Models

As discussed in the previous sections on I/O and LP models, these approaches have shortcomings that make them less than ideal tools for policy analysis in modern market economies. In particular, in both I/O and LP models, the behavior of producers and consumers does not reflect the independent optimizing behavior that is usually assumed to be a characteristic of agents in a market economy. Without the specification of realistic producer and consumer behavior, model-based policy simulations will be unable to correctly account for the reactions agents may have to policies that impact them. Computable general equilibrium (CGE) models incorporate more realistic behavioral specifications of the agents into the model and are thus able to provide a better laboratory for many types of policy analysis. CGE models have been used to analyze a wide variety of policy interventions, including issues in public finance, international trade, development, and increasingly, the environment.[18]

Most policies meant to protect the environment, ranging from those relying on market-based instruments, such as effluent taxes, to command and control regulations, induce changes in the behavior of consumers and producers. These changes may occur directly where the intervention takes place or indirectly as the effects of the intervention are passed through the economy. Because they focus on both trying to model more accurately the expected reactions of consumers and producers to policy interventions and on the interactions between various actors in the economy, for some problems CGE models may be the most appropriate tool for the analysis of social costs. CGE models are particularly good at examining questions of static resource allocation, such as the effects the imposition of a tax may have on sectoral output, income, and employment. Under certain specifications, CGE models may also be useful for assessing impacts on overall measures of economic performance, such as aggregate output, employment, and various measures of welfare.

In almost all cases, CGE models start from the framework and data of an input-output table, which provides a basic set of accounting identities for the production sectors. Producers are assumed to maximize their profits through their choice of productive inputs, typically labor, capital, and intermediate goods, and sometimes land. Consumers,

[16] The term linear programming actually refers to any applied mathematical programming exercise where an objective function is either maximized or minimized subject to a set of inequality constraints and where all of the equations are linear. In this section, only general equilibrium applications (i.e., those based on an input-output table) are discussed.

[17] Linear programming models are discussed in Dervis et al. (1982). A number of examples of the application of linear programming models to environmental problems are given in Hufschmidt et al. (1983).

[18] General references on CGE models include Dervis et al. (1982) and Ginsburgh and Keyser (1997). Applications to environmental policy are discussed in Wajsman (1995).

or in many cases a representative consumer, are assumed to maximize their utility by choosing their consumption bundles, subject to a budget constraint. Although not usually specified as an optimizing agent, most CGE models also include a government sector that collects a variety of taxes to pay for its purchases of goods and services. The domestic demand for imports and the supply of exports are determined based on the relative prices of domestic and foreign goods.

CGE models may be categorized across a number of dimensions. These can include (1) the method by which the parameters of the model are specified (through calibration or econometric estimation), (2) the time horizon of the model (static or dynamic), and (3) the scope of the model (single- or multi-country). Most CGE models are calibrated to a single base year, which is assumed to be in equilibrium. After the specification of a subset of elasticities and other data obtained through a literature search (or using informed judgments) the rest of the parameters can be determined by working backward from a social accounting matrix (SAM) for the base year.[19] An alternative to the calibration approach is econometric estimation.[20] General equilibrium econometric estimation allows models to incorporate the representation of more sophisticated producer and consumer behavior than would normally be possible through calibration. However, econometric estimation requires a consistent set of multi-sector time-series data and this data is usually not available for developing countries.

CGE models can also be differentiated by the time horizon of the analysis. The majority of CGE models are "static," meaning that no explicit dynamics are incorporated and the time frame for the attainment of a new equilibrium following a policy or external shock is indeterminate. In conducting simulations, an exogenous shock is introduced or a variable, such as a tax rate, is altered. The model is then allowed to search for a solution until a new set of prices is found which again equilibrates the system. The new prices in turn determine a new set of factor demands, outputs, and incomes. The values from this new solution are then compared with the values from the original equilibrium. Dynamic models, on the other hand, incorporate an explicit updating of time dependent variables, such as the labor supply, capital stock, technology, and demand patterns. In conducting a simulation, a baseline case is first run with a given set of assumptions about the time-dependent variables. Next, an alternative or counterfactual simulation is run with the same set of assumptions, but with a policy or external shock. The values resulting from the alternative solution are then compared with the original baseline. These values may be compared at different points in time or discounted to estimate present values, or to evaluate changes in welfare.

Another dimension along which CGE models may be classified is by scope: (1) single country or single region models, (2) multi-country or multi-region models, and (3) global models encompassing all countries and regions. Although models representing a single country or region are the most common, multi-country or multi-region models are being developed in increasing numbers. Because trade is inherently a multi-country phenomenon, multi-country models are generally the best suited for examining issues that involve the flow of goods, services, and capital across national boundaries.

CGE models have been applied to an expanding array of environmental policy issues. Most recently, they have been used for the analysis of policies designed to avert or slow global climate change, such as those proposed in the Kyoto Protocol. Both single country and multi-country CGE models have been used for these simulations, with multi-country models able to assess policies like global emissions trading. Because they are able to incorporate taxes and other existing distortions, CGE models have been used to explore the potential for a "double dividend"—a reduction in pollution plus a reduction in the inefficiencies of the tax system—from substituting taxes on pollutants for pre-existing taxes on output, income, or wages. In addition, CGE models have been used to perform retrospective analyses of the economic costs of a number of environmental regulations.[21]

[19] References on social accounting matrices include Pyatt and Round (1985) and Chapter 10 of Sadoulet and de Janvry (1995).

[20] This approach has been pioneered by Dale Jorgenson and a number of his collaborators. See in particular the papers collected in Jorgenson (1998a, 1998b). Another example of the use of the econometric approach is Hazilla and Kopp (1990).

[21] For example, the Jorgenson-Wilcoxen dynamic CGE model of the U.S. was used to estimate compliance costs between 1970 and 1990 for EPA's retrospective study of the benefits and costs of the Clean Air Act (EPA, 1997b). A previous CGE-based study by Hazilla and Kopp (1990) looked at the costs of both the Clean Air and Clean Water Acts.

While CGE models have a number of advantages as tools for policy analysis, they also have serious drawbacks:

- Although the costs have been reduced in recent years, the construction of a CGE model can be still be time consuming and expensive.
- In addition to an appropriate input-output table, a considerable amount of data on national accounts, trade, elasticities, and environmental externalities must be collected and made consistent with the sectors chosen to be part of the analysis.
- Dynamic models also require that forecasts be made for many exogenous variables.
- Many environmental policies only affect a small part of what may be a highly aggregated sector in an input-output table. Sometimes it will be possible to separate these smaller sectors out, but sufficient data is often not available at that level of detail.

8.5 Estimating the Social Costs of Alternative Policy Approaches

This section discusses the methods of estimating the social costs for several alternative regulatory and nonregulatory policy approaches, which are divided into three categories: (1) direct controls, (2) incentive-based controls, and (3) voluntary actions taken to reduce environmental risks. The discussion focuses on the significant features of each regulatory approach that must be examined in either partial equilibrium, multi-market, or CGE models. In addition to the private compliance costs, transactions costs may be significant. Therefore, the associated changes in the prices of goods and services will alter producer and consumer surplus and must be calculated to estimate the total social costs.

Independent of the method used, the social cost analysis should explain how the uncertainties and assumptions in the data and models affect the results. Since much of the data used is not known with certainty and many assumptions must be made to develop the necessary analytical models, social cost estimates can never be known with total certainty. Another difficulty is that private (compliance) cost bounds in one project may be based on the intuition of a single engineer, but the private cost bounds in another sector may be developed based on adequate data permitting the estimation of confidence intervals. Aggregating these into a single study may conceal important uncertainties rather than enlightening the decision making process.[22]

8.5.1 Direct (or Standards-Based) Controls

In general, direct or standards-based controls rely on different types of standards that mandate a level of performance intended to acheive an environmental objective.

8.5.1.1 Technology Standards

Estimating the private compliance costs of standards-based regulations is relatively straightforward compared to incentive-based approaches. Technology standards often require Best Available Technology (BAT) or Best Practicable Technology (BPT). Since these technologies already exist, their costs and the number of firms required to use them are often well documented. Additional compliance costs include expenditures on maintenance and training costs associated with installing and operating the equipment. However, estimating the private compliance costs is not always simple, especially for proposed regulations. For example, unanticipated scaling effects, as well as unforeseen bottlenecks in construction and implementation may occur, resulting in differences between the anticipated bids for a project, the bids received, and the actual construction cost.

The private real-resource costs, when discounted over time, correspond to the sum of investment costs and discounted annual costs (operating and maintenance and other annual regulatory costs) that will be incurred by firms to comply with the regulation. Thus, the real-resource costs of regulation can be approximated, in most instances, by methodologies routinely used by other EPA

[22] Another reason for variability of the compliance cost estimates for pollution control may be due to the way emission rates are characterized, their method of transport, and chemical transformation rather than variations in the price of pollution control equipment and labor costs.

program offices to evaluate compliance costs. Furthermore, the supply and demand curves that implicitly lie behind such calculations need not be formally estimated unless the effects of the regulation on price and output are expected to be significant.

8.5.1.2 Emission Standards

Regulations that limit emissions are usually targeted at particular point sources or geographic regions, but there also exist national standards such as ambient air and water quality standards. For point sources, private compliance cost estimates are usually based on the cost to purchase and operate regulatory equipment needed to comply with the regulations. This equipment will be similar to that needed to meet technology standards. Since it may be prohibitive to examine in detail each area and all its pollution sources, private social and compliance cost estimates can be based upon random samples of representative areas and industries. The survey should accurately reflect the expected compliance costs for different categories and sizes of industries in each area. In each of the three cases, additional social costs are involved with monitoring and enforcement of the regulations.

8.5.1.3 Production Bans

The prohibition of a product or service results in shutdowns, causing short- and, sometimes, long-term unemployment, as well as the loss or premature retirement of capital equipment. Therefore, adjustment costs should include: (1) the value of wages temporarily or permanently foregone because of reductions in production levels in the directly affected markets[23] and (2) the social cost of re-employing displaced workers (including the administrative cost for transfer payment programs, but excluding the payment itself). Moreover, policies that effectively ban products or activities cause the loss of all producer and consumer surpluses in these markets. Regulations also may substantially affect secondary or linked markets. CGE and multi-market models will account for these effects, but partial equilibrium models will not do so unless a separate model is used.

Some policies, although not explicitly banning production, may be so stringent that the effect on production is the same. In this case, the concept of compliance cost is less applicable. For example, if the mandated environmental protection controls would be so costly that the new equilibrium level of output is zero (because the consumers of the good shift completely to substitutes and producers of the good exit this market to produce other outputs), then it is not possible to use compliance costs as an estimate of social costs. A ban on producing this good would produce a similar result.

In the case of an effective ban on production, the social cost of the new policy is measured by the complete loss of all producer and consumer surpluses in this market. The real-resource cost in this case might be conceptualized as the welfare change associated with the additional expense and lower quality of the other goods that consumers purchase as substitutes for their previous use of the product.

8.5.2 Incentive-Based Controls

The appeal of incentive-based approaches is their potential ability to achieve environmental improvement goals more cost effectively than traditional standards-based methods. The approaches examined here include: marketable permits, emission taxes, bubbles and offsets, user charges, product charges, subsidies for pollution reduction, government cost sharing, refundable deposits, pollution indemnity, and information and labeling rules. In many cases, significant transfers will occur between private parties and the government, but in most cases, these policies achieve their greater economic efficiency through mechanisms that encourage private parties to use information known to them but not to the environmental authorities. Such information may include differences in emission control costs among different firms. Analyses of the social costs of incentive-based approaches may require different information and tools than those used to analyze more traditional

[23] Lost wages, rather than lost production, is suggested as a proxy for the value of displaced resources. In most cases, lost wages will capture most of the value of displaced resources because it is likely that inputs other than labor will be reallocated to other sectors of the economy fairly quickly and at little cost.

policies. The task of calculating the exact compliance costs of these policies is therefore more difficult.[24]

8.5.2.1 Marketable Permits

Permits are usually denominated in the amount of emissions allowed and the number and denomination of permits issued determines the aggregate amount of emissions.[25] Marketable permits establish the aggregate quantity of pollution allowed and allow the price of those entitlements to vary. Since different polluters incur different private costs to control emissions, they will be willing to pay different amounts for permits. The price of permits, in theory, will be established by the unit cost of control of the marginal polluter. Marketable permits are traded between emission sources, giving rise to a transfer among private parties but not social costs.

In contrast to technology standards, incentive-based approaches (taxes or permits) do not require any particular firm to install particular pollution control devices. Therefore, it is usually necessary to predict what technology will be used by the firms to meet the new regulations to estimate costs. Under a marketable permit system, the private costs of pollution control are estimated following a three-step process:

- Calculate the demand and supply functions for the permits (these demand and supply functions are based on the costs of different sources' pollution control options). Polluters will be willing to pay prices for the permits up to the unit cost of reducing emissions.
- Estimate the equilibrium price for permits. This price will determine which firms will install pollution control measures and those which will purchase additional permits.
- Calculate the real-resource cost of the regulation by summing the investment costs and the present value of operating and maintenance costs incurred to reduce emissions. (The cost of the permits purchased by firms is classified as an income transfer between firms, not a social cost.)

Marketable permits may be sold at auction initially, in which case the prices bid for the newly issued permits again represent income transfers to the government. Alternatively, permits may be allocated to sources by some rule, in which case no private costs are imposed at the outset. In neither case is the private cost of a permit counted as part of the social cost, since it is not a real-resource cost, but rather a transfer from one firm to another or a transfer from a firm to the government. The act of establishing the permit system and assigning property rights to the distributed permits may result in some type of "wealth transfer" taking place, which should be accounted for in the equity assessment of the benefit-cost analysis.

As described earlier, the marketable permit systems requires the creation of a functioning market. This results in administrative costs and also additional enforcement costs since it is necessary to ensure that emissions do not exceed the levels for which permits are held. Both need to be added to the social cost estimate.

8.5.2.2 Emission Taxes

Estimates of the social costs of pollution control under emissions taxes are virtually the same as those under marketable permits. Here however, the unit price of pollution will be known (since it is set by the regulation) and does not have to be calculated. Because the private cost of pollution control varies among firms, firms with the highest private costs of pollution control are expected to cut back production and pay the tax on the remaining emissions, rather than install required pollution control equipment. Most often, firms will choose some combination of cutbacks in production, installation of pollution control equipment, and payment of the tax. However, as in the case of most regulations, real-world limitations may reduce the possible selection of cost-minimizing alternatives chosen or firms may simply decide to engage in litigation to delay the regulation, which adds an additional cost.

[24] However, after the policy is passed, compliance costs are much easier to calculate. In the case of emission taxes, cost-minimizing polluters, in theory, will reduce emissions up to the point where the marginal cost of control equals the tax. Of course, these conclusions rest upon the assumptions of perfectly competitive markets, low transactions costs, and complete information.

[25] Permits may also be weighted based on the impact the pollutant has on air or water quality.

After the supply (or abatement cost) function for each firm is estimated, then the equilibrium amount of pollution generated can be determined. The new production levels can then be calculated for each firm along with how much pollution control equipment will be installed. Finally, the real-resource costs and private costs are calculated as is done above for marketable permits. However, here the pollution taxes or charges may involve transfers of tax revenues from the private sector to the government, but because such fees accrue to the government, they are again income transfers and should not be included in total social costs.

8.5.2.3 Bubbles and Offsets

Bubbles and offsets allow emissions to be averaged among specific regions or among different sources within a particular facility. The resulting level of emission control must be equivalent to or better than that required by existing regulations. If "banking" is allowed, pollution credits can be traded across time—with potential offsets created in one period to be used in later periods. Bubbles and offsets create incentives to reduce emissions in firms where the private costs of control are relatively low. Compared with direct controls, bubbles and offsets usually result in lower compliance costs (relative to costs incurred under technology and emission standards).

Additional social costs associated with bubbles and offsets are similar to those encountered with marketable permits. Initial administrative costs may be significant and some additional enforcement costs for monitoring emission levels also may be incurred. The private cost of the offsets traded in a formal market are transfers between the creator and the user of the offset and are not social costs. However, the transactions costs incurred to arrange the trade, both private and public, are part of total social costs.

8.5.2.4 User Charges

Charges may be imposed directly on users of publicly operated facilities. Such charges have been imposed on firms that discharge waste to municipal wastewater treatment facilities and on non-hazardous solid wastes disposed of in publicly operated landfills. User charges are usually set at a level sufficient to recover the private costs of operating the public system, rather than to create incentives for reducing pollution. Measuring the total social cost imposed by user charges is similar in concept to measuring the social costs of emission charges, but is not always based on a per unit charge of the pollutant.

8.5.2.5 Product Charges

Charges may be imposed on intermediate or final products whose use or disposal pollutes the environment. The private and social costs imposed by product charges depend on the extent to which users switch to substitutes, reduce the rate at which the product is used by recycling or other process changes, or continue to use the regulated products. Predicting responses and estimating compliance costs is difficult and requires analysis of the social costs and effectiveness of substitute products as well as the social costs of recycling and reuse. Any charges paid as a result of continuing use represent private costs, but are not social costs because these are borne by the consumer who buys less of the more expensive product.

8.5.2.6 Subsidies for Pollution Reduction

Subsidies paid to polluters based on their reductions in pollution have the same general effect on behavior as charges on pollution. Sources may reduce pollution up to the point where the private costs of control equal the subsidy. Using subsidies instead of charges shifts private costs to the government. This may result in more sources continuing to operate than if a charge system were used. Thus, subsidies and charges may not have the same aggregate social costs or the same degree of pollution control.

Measuring the social costs resulting from a system of pollution subsidies is similar in concept to measuring the social costs resulting from pollution charges, except that private costs are reduced by the amount of the subsidy, rather than being increased by the amount of the fee. Again, the subsidies themselves are income transfers and do not constitute a social cost. Therefore, private real-resource costs should be computed excluding the subsidies.

8.5.2.7 Government Cost-Sharing

In addition to issuing the regulations described above, the government may take actions to lower the private costs of specific actions—most notably, by subsidizing investments in pollution control equipment. These subsidies may take the form of reduced interest rates, accelerated depreciation, direct capital grants, and loan assistance or guarantees for pollution control investments. Such policies may not by themselves induce changes in private behavior. However, in conjunction with direct controls, pollution fees, or other regulatory mechanisms, they may influence the nature of private responses and the distribution of the cost burden. In particular, such subsidies may encourage investment in pollution control equipment, rather than other responses that do not require capital investments (e.g., changes in operating practices or recycling and reuse).

Government cost-sharing subsidies reduce the private costs associated with the resulting private investments. However, social costs may arise if cost-sharing programs lead to resource misallocations. Additional social costs will result from administration of the subsidy program, but are likely to be minor if the incentives are provided through the existing tax system. However, they may be significant if new administrative structures are required.

8.5.2.8 Refundable Deposits

Refundable deposits create economic incentives to return a product for reuse or for proper disposal, providing that the deposit exceeds the private cost of returning the product. Therefore, to predict the rates of return, the private cost of returning products must be estimated.

Under a refundable deposit system, compliance costs consist of the resources (labor, equipment, and transportation) required to return the regulated product, and the private cost of preparing products for reuse (if required), less the cost of new products replaced by recycled products. The private administrative costs of a deposit system vary, depending on where in the production-consumption cycle they occur and from whom the deposits are collected. Record-keeping requirements may also be a cost component. For those that participate in a recycling/refund program, the opportunity cost of the time spent sorting trash is an important component, and the analysis must address whether other behavior changes may be expected to take place (e.g., whether consumers may adjust their consumption of products marketed in recyclable containers). The deposits themselves represent transfers from one point in the production-consumption cycle to another and, hence, are not social costs. These transfers are temporary if the deposit is reclaimed, but permanent if it is not. Enforcement costs are minimal since no standards have been set and no laws are broken if the product is not returned.

A government "buy-back" constitutes another type of refundable deposit. Under this system, the government either directly pays a fee for the return of a product or subsidizes firms that purchase recycled materials. They are equivalent to product deposits, except that the government, rather than the purchaser, provides the deposit. The government subsidy represents a transfer from the government to the private sector, which offsets the private costs of recycling products.

8.5.2.9 Pollution Indemnity

Regulations that impose stricter liability on polluters for the health and environmental damage caused by their pollution may reduce the transaction costs of legal actions brought by affected parties. Such regulations do not impose additional social costs, but only shift the costs from one party to another. However, this may induce polluters to alter their behavior and expend real resources to reduce their probability of being required to reimburse other parties for pollution damages. For example, they may reduce pollution, dispose of waste products more safely, install pollution control devices, reduce output, or invest in added legal counsel.

Other regulations may require firms to demonstrate financial ability to compensate damaged parties by posting performance bonds that are forfeited in the event of damages, by obtaining liability insurance, or by contributing to a pool of funds to compensate victims. The administrative and enforcement costs imposed by such requirements represent the use of economic resources but are not counted as part of the social costs because the funds used to pay damages do not represent a use of resources, but are transfers among private parties (between polluters, insurers, and victims of pollution). Again, however, these requirements are likely to alter private behavior and lead to increased outlays of real resources to reduce the probability of accidents, or reduce the probability of having to pay using any of the methods cited above.

8.5.2.10 Information and Labeling Rules

Information or labeling rules may be applied to specific substances or to certain contaminated locations. For example, warning labels may be required for hazardous substances that describe safe-handling procedures or describe the risks posed by the product. Purchasers may then switch to less damaging substitutes for some or all uses or handlers of hazardous substances may be better able to prevent damages. Posting information on contaminated locations gives potentially exposed parties the opportunity to avoid hazards—for example, contaminated dump sites or drinking water aquifers.

Calculating the private costs of complying with information and labeling requirements for particular cases is straightforward. Compliance costs include the cost of developing the required information (analyzing the composition of substances, monitoring and testing of sites, testing for health damages, etc.) and the cost of disseminating information (printing and applying labels, maintaining and publishing information on sites).

Similarly, the direct costs of enforcing and administering the requirements (including government review and approval of labels) can be calculated directly. Calculating aggregate private costs may be more difficult, however, if the number of containers requiring labels or the number of facilities affected is unknown. In addition, it is difficult to predict responses by the recipients of the information and, hence, the social cost of avoidance.

8.5.3 Voluntary Actions

While there can be social costs associated with voluntary actions taken to mitigate environmental polluting behavior, it may be difficult to establish the relation between decisions made by firms or individuals and the role EPA and other regulatory agencies may play in inducing or contributing to the adoption of these practices. EPA and other regulatory agencies are looking to alternative nonregulatory approaches in an effort to change behavior that contributes to health and environmental risks. Examples of EPA programs of this type include the 33/50 program for reduction of toxic pollutant discharges and energy conservation and greenhouse gas mitigation measures, such as ENERGY STAR and Climate Wise programs.[26]

A basic premise underlying social cost analyses is the assumption that profit-maximizing firms undertake investment decisions, voluntary or otherwise, when it is in their private interests to do so (i.e., where private benefits are greater than private costs). If actions would not have occurred absent EPA's involvement in these programs, then there may be social costs (beyond those costs incurred by EPA) that are associated with actions taken by participants in the programs. Social costs may arise, for example, from firms exhibiting strategic behavior in their investment decision that incorporates expectations that voluntarily participating in nonregulatory programs may serve to reduce or delay promulgation of future, potentially more stringent enforceable compliance standards. Without some assessment of costs, it is difficult to establish whether a particular voluntary program is cost-effective in comparison with other policy actions. As a consequence, it is useful to investigate the social costs associated with nonregulatory programs—quantifying how they affect economic markets, and evaluating the relative economic efficiency of these approaches as compared with regulatory policies.

8.6 Summary and Conclusions

This chapter has reviewed the theoretical foundations of social cost assessments as well as practical methods for measuring the social costs of environmental policies. Several key conclusions reached in this discussion are worth reiterating.

First, in most cases, the social costs of an environmental policy or other action can be measured with sufficient accuracy by limiting the analysis to the directly affected markets. This allows the analysis to focus on the sectors that must comply with a policy. In these cases, the disturbances that ripple outward from the directly affected markets to numerous other markets should have a minimal effect on the estimation of social costs.

[26] More information on the operations and objectives of these types of programs can be found in publications prepared by EPA's Office of Reinvention and at the following website http://www.epa.gov/reinvent/ (accessed 8/28/2000).

Second, a conventional partial equilibrium depiction and modeling of the directly affected markets will often be sufficient to measure social costs. The detail and precision of applying this approach in practice will depend on the availability of information, the resources devoted to the evaluation, and the value to policy makers of improved accuracy of the results.

Finally, other modeling techniques, such as CGE, are often used to measure a variety of indirect costs, the many transitional effects of environmental policies, and, when they are significant, transactions costs borne by private sector entities. Nevertheless, the majority of the economic analysis of environmental policies can employ the conventional partial equilibrium modeling approach to evaluate social costs of EPA policies and programs.

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Chapter 9: Distributional Analyses: Economic Impact Analyses and Equity Assessments

9.1 Introduction

In an effort to fully understand a regulation's impact and make an informed judgement regarding its desirability, policy makers study many different regulatory consequences. Economic information is important to the evaluation of at least two consequences—a regulation's efficiency and its distributional consequences. In principle, both types of consequences could be estimated simultaneously by a general equilibrium model. In practice, for reasons discussed in Chapter 5, they are usually estimated separately.

A benefit-cost analysis (BCA) calculates the social benefits and costs of an environmental policy and answers the question of whether the benefits are sufficient for the gainers to potentially compensate the losers, leaving everyone at least as well off as before the policy. Its calculation of net benefits helps ascertain the economic efficiency of a regulation. Two other ways to express economic information—an economic impact analysis (EIA) and an equity assessment—assess changes in social welfare by examining the distributive effects of a regulation. An EIA focuses on traditional classifications of affected populations[1] (e.g., industrial sector classifications). Under the heading of equity assessment analysts can address broad concerns such as changes in the national distribution of income or wealth. In addition, an equity assessment can provide information to decision makers on how policies affect specific sub-populations. Disadvantaged or vulnerable sub-populations (e.g., low income households) may be of particular concern.

Economic Impact Analysis (EIA)

An EIA helps answer the questions, "Who are the losers and gainers from a policy?" and "By how much do they lose or gain?" Traditionally, EIAs have focused on the losers and the negative impacts of an environmental regulation. This focus is in response to existing legislative and administrative statutes and policies which direct analysts to examine the distribution of negative regulatory impacts or costs. Currently, several of these same statutes and policies call for a similar examination of the positive impacts of a regulation.

Unlike a BCA which rests its conclusions exclusively on comparisons of social benefits and costs, an EIA examines the distribution of many different economic impacts. Conventional impacts include monetized effects such as changes in profitability or in government revenues, as well as non-monetized effects such as increases in unemployment rates or numbers of plant closures. An EIA will often examine and report on regulatory outcomes that a BCA would not. For example, when measuring impacts on private

[1] The term "affected" is applied throughout the chapter in its most general use as an economic term. Analysts should be aware of how the authorizing statute for the rule, as well as other applicable statutes and administrative orders noted in this chapter, make more specific use of this term. For example, the Regulatory Flexibility Act includes the clause "subject to the requirements of the rule" when quantifying economic impacts. This results in analyzing entities that are directly affected, so that conclusions can be drawn as to the significance of impacts of the rule. Alternatively, provisions in UMRA and EO 12866 address both direct and indirect impacts, so that the affected population of entities may be more inclusive than only those "subject to the requirements of the rule." For more information, Chapter 8, Section 3 on "Social Cost Analysis" covers the economic concepts and terminology relevant to direct and indirect impacts.

businesses, an EIA will include changes in transfer payments from firms to the public sector whereas a BCA would not. Transfer payments become important when analyzing the distributional consequences of a regulation. To achieve the objective of an EIA and educate policy makers about who will lose or gain as a result of a particular regulation, analysts have traditionally relied upon the assortment of impacts described in Section 9.2 below.

Equity Assessment

Generally, assessments of equity examine a regulation's impact on the distribution of national income or wealth. Decision makers may use this information in conjunction with economic efficiency measures as captured in a benefit-cost analysis to evaluate tradeoffs between equity and efficiency. For the most unified treatment, both equity and efficiency issues can be addressed in a computable general equilibrium model.[2] In practice, data constraints will limit analysts to undertake distributional analyses independently from benefit-cost analyses.

As is true for an EIA, an equity assessment is generally more concerned with sub-populations who experience net costs or other negative impacts than with those who experience net benefits or positive impacts. An equity assessment may consider effects on any sub-population, but it should always consider the economic effects of a regulation on disadvantaged or vulnerable sub-populations; specifically, sub-populations who are physically susceptible to environmental contamination, are less than fully capable of representing their own interests, or are economically disadvantaged or vulnerable. Examples include children, low-income or minority communities, and small businesses, governments, and not-for-profit organizations. For many of these sub-populations, EPA has been directed by statute or policy to examine the effects of its rules when they are expected to have a "disproportionate," "significant and substantial," or other such impact.

An equity assessment draws on information and analytic tools used in BCA and may report on impacts using measures found in an EIA. Therefore, an early step in an equity assessment is to identify sub-populations likely to be affected by a regulation. Once identified, if data permits, the social costs and benefits estimated for the BCA can be disaggregated and net benefits for the sub-population(s) or the distribution of net benefits among sub-populations, can be examined. An equity assessment may also examine economic impacts, such as increases in rates of unemployment or other traditional impact measures, for the identified sub-population(s).

Consistency Between BCA and Distributional Analyses

Ensuring consistency in analytic design and interpretation of results for the BCA, EIA and equity assessment supporting a particular regulation is essential. All three examine impacts that, in principle, could be estimated by a single general equilibrium model (see Section 5.2). Both an EIA and an equity assessment must be conducted following the principles that frame a BCA, even if the formal preparation of BCA is not undertaken. When a BCA is undertaken, to the extent possible, both distributive analyses should adopt the same set of assumptions used in the BCA. For example, all three should rely upon the same set of baseline assumptions and all three should assume the same values for relevant elasticities. However, because all the information needed to estimate distributive outcomes is often not integral to the calculations performed in a BCA, in many cases further assumptions must be developed specifically for the EIA or the equity assessment. For example, new assumptions regarding definitions of sub-populations must be developed and there might be good reason to assume different elasticities for different sub-populations. Even in these cases, analysts should ensure that the implications, if any, of the added assumptions for the outcome of the BCA are understood and made manifest to policy makers.

Using a Social Welfare Function to Evaluate Efficiency-Equity Tradeoffs

Potentially, a regulation's effects on distribution, analyzed by its EIA and/or equity assessment, and its effects on efficiency, analyzed by its BCA, can be incorporated into a single social welfare function. A social welfare function establishes criteria under which efficiency and equity outcomes are transformed into a single metric, making them directly comparable. A potential output of such a function is a ranking of policy outcomes that have different aggregate levels and distributions of net benefits. A social welfare function can provide empirical evidence that a policy alternative yielding higher net benefits, but a less

[2] Computable general equilibrium models are discussed in section 8.4.5.

equitable distribution of wealth, is better or worse than a less efficient alternative with more egalitarian distributional consequences.[3]

In practice, developing a universally acceptable social welfare function is difficult because it requires explicit decisions to be made about society's preferences for the distribution of resources. Nonetheless, future research may result in some feasible practical alternatives.[4] These guidelines do not suggest a particular social welfare function or that analysts attempt to use this approach at this time, but the approach may merit further consideration as additional research and applications develop.

Chapter Summary

This chapter begins with a brief discussion of an iterative process between analysts and management to facilitate thorough consideration of the output from distributional analyses. The bulk of the chapter occurs in Section 9.2 on Economic Impact Analysis which, after reviewing statutes and policies that require examination of economic impacts, describes methods for estimating economic impacts that are relevant for both EIAs and equity assessments. The final section of this chapter discusses the relatively new distributional analysis, equity assessment. Statutes and policies that require equity assessment, definitions of sub-populations, and a general framework for conducting an equity assessment, including possible data sources, are reviewed.

9.1.1 A Process for Economic Impact Analyses and Equity Assessments

This section describes an iterative process between EPA analysts and senior management[5] as an integral part of an EIA and an equity assessment. At several points of the regulatory development process, senior analysts should report the results of distributional analyses to senior management and receive feedback. Only through such ongoing communication can senior management remain sufficiently informed so that potential economic effects of proposed environmental regulations receive proper attention within the regulatory development process.

As discussed above, ensuring consistency between the EIA, equity assessment, and BCA is critical. The methods and results of an EIA and an equity assessment are inherently linked to their corresponding BCA. Consequently, concerns regarding distributional outcomes that arise through the iterative process that necessitate a change in the regulatory approach will also require adjustments to the assessment of social benefits and costs.

This iterative process is not expected to add significant additional administrative procedures to the current EPA regulatory development process. Rather, its objective is to bring greater attention to opportunities for the workgroup and senior management to have ongoing communication related to potential economic impacts and equity dimensions of proposed environmental regulations. Frequent and timely exchanges of information between senior management and the workgroup will focus greater attention on affected sectors of the economy as well as affected sub-populations and may influence the final regulatory alternative selected.

Information contained in Exhibit 9-1 illustrates such a process. Its contents are consistent with the procedures outlined in the document, *Regulation Development in EPA* (EPA, 1997) and with the process for promulgating a regulation illustrated by the flow chart in *Guidance for Analytic Blueprints* (EPA, 1994). There are two key components of Exhibit 9-1 that are designed to institutionalize the iterative process between the workgroup and senior management for EIAs and equity assessments. The first component is an explicit incorporation of the identification and analysis of economic impacts and equity dimensions such as those listed in Exhibits 9-2 and 9-5 into the regulatory development process. The second component, depicted by the arrows, is a process for initiating multiple

[3] For more on the use of social welfare functions in policy analysis see Sen (1970), Arrow (1977), and Jorgenson (1997). An empirical application of this approach can be found in Norland and Ninassi (1998).

[4] For a recent description of potential alternatives see Farrow (1998).

[5] Senior management is used as shorthand for persons responsible for authorizing and using these forms of analysis. Most often, these persons will include the Assistant or Regional Administrator of the lead office or region that is considering the regulation or other upper management within that office or region.

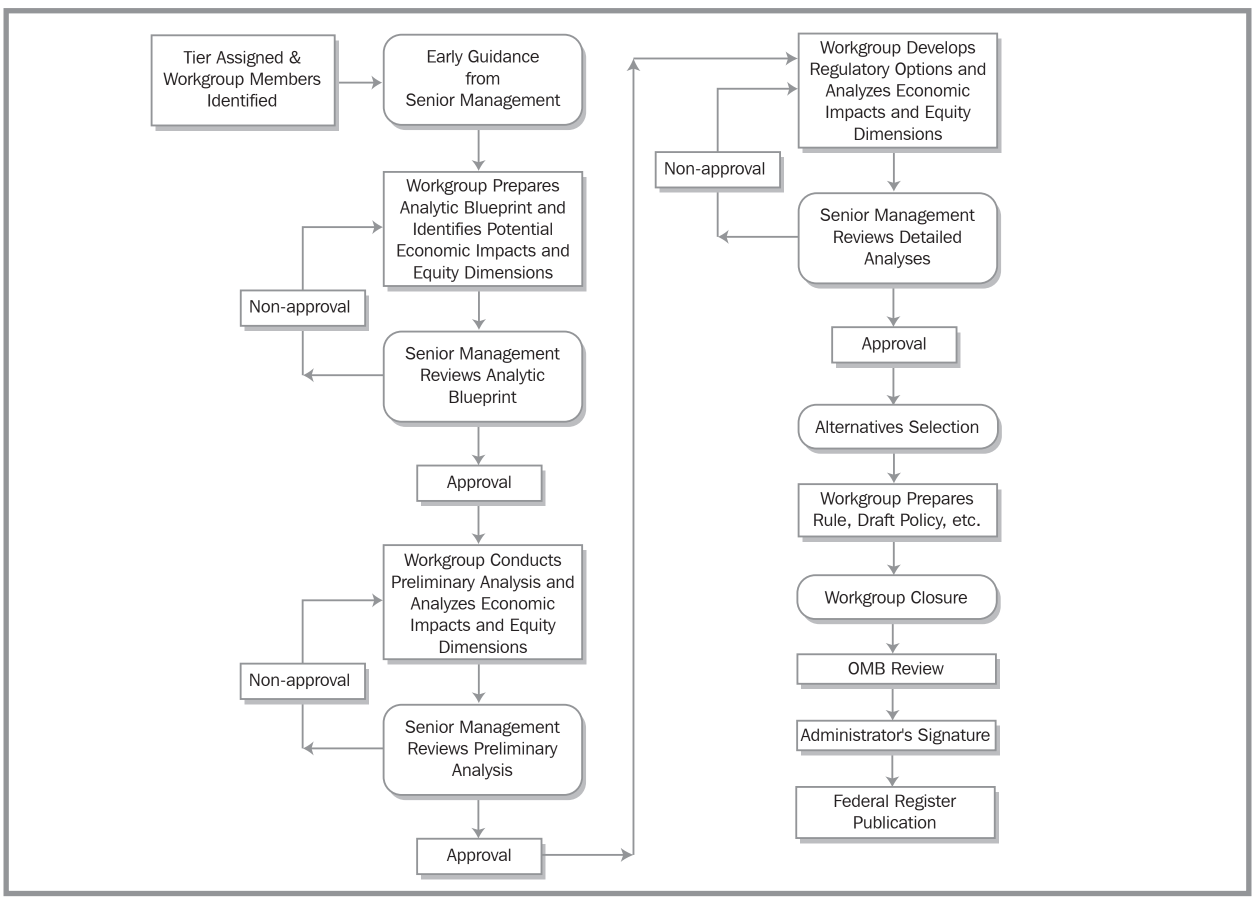
re-evaluations of a regulation during the development process. This process includes three key tasks.

- **Identify potentially important economic impacts and equity dimensions for senior management** to help determine which may be of concern. This should be done as part of the analytical blueprint process. The analytic blueprint provides an opportunity for early identification of important issues in order to enhance the quality of information provided to senior management to assist in decisions for a particular regulation or policy. While required for Tier 1 and 2 regulations, analytic blueprints are encouraged for Tier 3 regulations as well.

- **Conduct a preliminary analysis of these economic impacts and equity dimensions** once senior management has approved the analytic blueprint. This is the second point where the iterative process affects a regulation's development. Analysts should share the results with senior management who should then determine whether to proceed to a more detailed analysis or to revisit the preliminary analysis. Senior management may even decide to alter the overall regulatory approach being considered which could require revising the BCA as well as the analytic blueprint. The potential need to revise the analytic blueprint is consistent with the idea that it is a living document.[6]

- **Develop options and prepare detailed economic analysis** after the preliminary analysis is complete. Data and information developed as part of this effort will provide input for conducting distributional analyses for the impacts and dimensions identified earlier. Here is the third point where the iterative process comes into play. It is recommended that input from senior management be sought once more before proceeding to closure. Additional economic impact and

Exhibit 9-1 Flow Chart: A Process for Economic Impact Analyses and Equity Assessments



[6] For a detailed discussion of the concept of the analytic blueprint as a living document and more information on the "Tiering" of rules, see pp. 12-13 *Guidance for Analytic Blueprints* (EPA, 1994) and see section 5, p. 31 *Regulation Development in EPA* (EPA, 1997).

equity analysis may be warranted for a variety of reasons including new insights gathered from the regulatory review process.

9.2 Economic Impact Analysis

9.2.1 Introduction to Economic Impact Analysis

As discussed in Chapter 5 above, a BCA calculates the total social benefits and total social costs associated with an environmental policy and measures the change in overall economic efficiency. As part of the effort to inform policy makers, it is important to not only understand the change in economic efficiency, but to understand the distribution of negative and positive impacts associated with this change. An EIA contributes to this understanding. It identifies losers and gainers from a policy and estimates the magnitude of their gains and losses. An EIA does this by studying the economic changes occurring across broadly defined economic sectors of society such as industry, government, not-for-profit organizations, and consumers. In addition to these broad categories, an EIA will examine more narrowly defined sectors within these broad categories such as the solid waste industry or an individual solid waste company. Traditionally, EIAs have focused on the losers and the negative impacts of an environmental regulation, although at least two general directives (see Exhibit 9-2) suggest that the positive impacts be examined as well.

EIAs measure impacts in different ways, from direct impacts on private business—including individual plants, whole firms, and industrial sectors—to indirect impacts on customers and suppliers. EIAs also measure direct and indirect impacts on governments and not-for-profit entities such as schools or hospitals. Impacts include changes in profitability, employment, prices paid by consumers, government revenues or expenditures, trade balances, and other changes of interest to policy makers.

Ensuring consistency of the EIA with the BCA for a particular regulation is essential. For consistency, an EIA must be conducted within the analytical bounds of its corresponding BCA. To the extent possible, the EIA should adopt the same set of assumptions used by the BCA. Adjustments or additions to these assumptions or to the overall modeling framework used for the BCA should be made only when they help bridge the difference between social and private perspectives, such as the difference between the social cost of a regulation and private compliance costs.

EPA's programs and regulations vary greatly in the types of parties affected and the nature of economic impacts that may be important. The data available for analysis vary widely as well. Thus, while specific methods for estimating impacts are reviewed, it is expected that every EIA will focus on the particular issues associated with the set of regulations under review. The general methods outlined here should be adapted to fit the needs of a particular analysis.

The remainder of this section is divided into twelve subsections. The first outlines the statutes and policies that direct EPA, and other government agencies, to study the distribution of positive and negative impacts. The second gives a broad overview of models for estimating social costs and how such models might relate to distributional analyses. In the third section, we begin explaining how to assess economic impacts. We begin with the first step, which is to calculate compliance costs. The next steps—how to screen for significant impacts and how to profile affected entities—are outlined in the fourth and fifth sections. Finally, beginning in Section 9.2.7, we review methods for estimating specific impacts, in the following order: impacts on prices; impacts on production and employment; impacts on profitability and plant closures; impacts on related industries and consumers; aggregate impacts on innovation, productivity, and economic growth; impacts on industry competitiveness; and impacts on governments and not-for-profit organizations.

9.2.2 Statutes and Policies Requiring Examination of Economic Impacts

There are at least two general administrative laws or orders that direct analysts to examine economic impacts; each is reviewed below. Some parts of environmental statutes also require consideration of economic impacts. Relevant quotations from a selection of these are presented.

9.2.2.1 General Administrative Laws or Orders

At least one statute—The Unfunded Mandates Reform Act of 1995 (UMRA)—and one executive order—EO 12866, "Regulatory Planning and Review"—require agencies to analyze various economic impacts of regulatory actions.[7] These directives require analysts to report on economic information that does not directly concern the net benefits tests for efficiency in a BCA. The first calls for analysts to examine the distribution of benefits and costs across different sectors of the economy. The second directs that certain outcomes be examined, such as changes in unemployment rates. For each policy, Exhibit 9-2 gives the dimensions for which impacts are to be analyzed and the corresponding analytical requirements. A discussion of these requirements follow the table.

As outlined by Exhibit 9-2, UMRA requires analysts to examine the costs, benefits, and budgetary effects of regulatory actions as experienced by state, local, and tribal governments; regions; urban or rural or other types of communities; or particular segments of the private sector. For the national economy, UMRA suggests many impacts that must be examined, including effects on productivity, economic growth, full employment, creation of jobs, and international competitiveness. These requirements apply only to rules that include federal mandates "which may result in the expenditure by state, local, and tribal governments, in the aggregate, or by the private sector, of $100 million or more in any one year."[8]

Exhibit 9-2 also briefly summarizes relevant parts of guidance from the Office of Management and Budget for EO 12866[9] (OMB, 1996 or *Best Practices*) and *Guidelines to Standardize Measures of Costs and Benefits and the Format of Accounting Statements*[10] (OMB, 2000 or *OMB Guidelines*). The *Best Practices* suggests that analysts examine the distribution of impacts across various sectors of the economy: "Information on distributional impacts related to the (regulatory) alternatives should accompany the analysis of aggregate benefits and costs."[11] In the *OMB Guidelines*, the focus for a distributional analysis is placed on those sectors that are likely to feel substantial impacts: "If these distributive effects are important, you should describe the effects of various regulatory alternatives quantitatively to the extent possible, including their magnitude, likelihood, and incidence of effects on particular groups."[12] The *Best Practices* also states, "The term 'distributional effects' refers to the description of the net effects of a regulatory alternative across the population and economy, divided up in various ways . . ."[13] This clearly suggests that both positive and negative impacts are relevant.

OMB cautions analysts conducting distributional analyses to recognize that transfer payments will become relevant, to avoid double-counting even when mixing monetized and physical effects, and to describe distributional effects without judging their fairness.

> "Since generally accepted principles do not exist for determining when one distribution of net benefits is more equitable than another, you should describe distributional effects without

[7] EO 13132, *Federalism* which took effect on November 2, 1999, and EO 13084, *Consultation and Coordination With Indian Tribal Governments* which took effect on August 12, 1998, both support the objectives of UMRA.

[8] UMRA § 202.

[9] U.S. Office of Management and Budget's *Economic Analysis of Federal Regulations Under Executive Order 12866*, January 11, 1996. This "*Best Practices*" document can be found at the U.S. White House, Office of Management and Budget website: http://www.whitehouse.gov/OMB/inforeg/riaguide.html under the section titled "Regulatory Policy" (accessed 8/28/2000).

[10] U.S. Office of Management and Budget's M-00-08 *Guidelines to Standardize Measures of Costs and Benefits and the Format of Accounting Statements*, March 22, 2000. The *OMB Guidelines* serves to implement Section 638(c) of the 1999 Omnibus Consolidated and Emergency Supplemental Appropriations Act and Section 628(c) of the Fiscal Year 2000 Treasury and General Government Appropriations Act. They require OMB to issue guidelines to help agencies estimate the benefits and costs of Federal regulations and paperwork and summarize the results of the associated analysis. The *OMB Guidelines* can be found at the U.S. White House, Office of Management and Budget website: http://www.whitehouse.gov/OMB/memoranda/index.html under the section titled "Selected Memorandum to Heads of Federal Departments and Agencies" (accessed 8/28/2000).

[11] *Best Practices*, p. 10.

[12] *OMB Guidelines*, p. 16.

[13] *Best Practices*, p.23.

Exhibit 9-2 Economic Impacts Required by General Administrative Law or Order[14,15]

General Administrative Law or Order	Dimension	Analytical Requirements
UMRA	State, local, and tribal governments; the private sector	Qualitative and quantitative assessment of anticipated costs and benefits of the federal mandate, including costs and benefits to state, local, and tribal governments or the private sector.
	Geographic location	Estimates of any disproportionate budgetary effects of the Federal mandate upon any particular regions of the nation or particular state, local, or tribal governments; urban or rural or other types of communities; or particular segments of the private sector.
	National economy	Estimates of the effect of the federal mandate on the national economy, such as the effect on productivity, economic growth, full employment, creation of productive jobs, and international competitiveness of U.S. goods and services.
OMB Guidance for EO 12866 and Accounting Statements	Population and economy, divided up in various ways (e.g., income groups, race, sex, industrial sector)	An economic analysis (EA) should describe the net effects of a regulatory alternative across the population and economy. "Irrespective of the presentation of monetized benefits and costs, the EA should present available physical or other quantitative measure of the effects of the alternative actions to help decision makers understand the full effects of alternative actions."[16]

judging their fairness. You should describe these effects broadly, focusing on large groups with small effects per capita, as well as on small groups experiencing large effects per capita. You should also note any equity issues not related to the distribution of policy effects if they are important, and describe them quantitatively to the extent you can."[17]

9.2.3 Models for Assessing Economic Impacts

As noted above, the analytic methods used for a distributional analysis of a particular regulation should be consistent with those used for the corresponding BCA. This section returns to the four methods for estimating social costs covered in Chapter 8, adding more insights on their application to distributional impacts.[18] The most sophisticated method—computable general equilibrium (CGE)—is

[14] Exhibit 9-2 does not include a discussion of The Regulatory Flexibility Act of 1980 (RFA), as amended by the Small Business Regulatory Enforcement Fairness Act of 1996 (SBREFA), as they are discussed below in Section 9.3, "Equity Assessment."

[15] The Office of Regulatory Management and Information's Rule and Policy Information Development System (RAPIDS) http://intranet.epa.gov/rapids (accessed 8/18/2000, internal EPA document) is a resource for EPA personnel who wish to read relevant statutes, executive orders or Agency policy documents in their entirety or to acquire copies.

[16] *Best Practices*, p. 23.

[17] *OMB Guidelines*, p. 16.

[18] For additional information regarding the four methods for estimating social costs see Section 3 of Chapter 8.

treated first and moves to less complex approaches, concluding with direct compliance costs.

9.2.3.1 Computable General Equilibrium Models

A CGE framework can be used to describe the effect of a particular policy on overall measures of economic performance, such as aggregate output, welfare, and the level of employment. CGE models are particularly effective in assessing static resource allocation and welfare distribution effects. These include the allocation of resources across sectors (e.g., employment by sector), the distribution of sectoral output, the distribution of income among factors, and the distribution of welfare across different consumer groups, whole regions, and countries. By construction, the basic capacity to describe and evaluate these sorts of distributional impacts exists to some extent within every CGE model. More detailed impacts, or impacts of a particular kind, will require a more complex and/or tailored model formulation (and the data to support it).

The simplest CGE models generally include a single, representative consumer, a few production sectors, and a government sector, all within a single-country, static framework. Complexities may be specified for the model in a variety of ways. Consumers may be divided into different groups along the lines of income, occupation, or other socioeconomic criteria. Producers may be disaggregated into dozens, or even hundreds, of sectors, each producing a unique commodity. The government, in addition to implementing a variety of taxes and other policy instruments, may produce public sector outputs, provide a public good, or run a deficit. CGE models may be international in scope, consisting of many countries or regions linked by international flows of goods and capital. The behavioral equations that characterize economic decisions may take on simple or complex functional forms. The model may be solved dynamically over a long time horizon, incorporating inter-temporal decision-making on the part of consumers or firms. This will have implications for the treatment of savings, investment, and the long-term profile of consumption and capital accumulation.

9.2.3.2 Multi-Market Models

CGE modeling captures multiple effects of a given policy change throughout an entire economy and can provide comprehensive distributional information across economic sectors (e.g., employment by sector). A CGE model may not be feasible or practical to use as a consequence of limited data and resources or when the scope of expected significant market interactions is limited to a subset of economic sectors. In such instances a multi-market analysis can be adopted as an alternative to a CGE model.[19] Multi-market analysis considers the interactions between a regulated market and other important related markets (outputs and inputs), requiring estimates of elasticities of demand and supply for these markets as well as cross-price-elasticities also found in CGE models. Multi-market models are best used when potential economic impacts and equity effects on related markets might be considerable, but more complete modeling using a CGE framework would offer a negligible improvement on the quality of information produced.

9.2.3.3 Partial Equilibrium Models

Unlike multi-market and CGE models, a partial equilibrium framework limits a distributional analysis to considering impacts on entities associated with the directly affected output markets only. Distributional consequences for other output markets and input markets are not estimated using these models. As discussed in Chapter 8, a partial equilibrium framework requires knowledge of demand and supply functions for directly affected markets only.

If information is required for distributional outcomes that this method is not designed to address, it may be possible to adopt further assumptions and acquire additional data to approximate distributional consequences of concern. These new assumptions should be consistent with those used for the corresponding BCA.

9.2.3.4 Direct Costs of Compliance

A relatively unsophisticated distributional analysis examines the direct costs of compliance paid by regulated entities. Often these analyses simply assume that the quantity of output and state of technology in the regulated industry

[19] For a detailed discussion of multi-market analysis see Chapter 9 in Just et al. (1982).

remain unchanged after the regulation becomes effective. An analyst could disaggregate compliance costs for regulated industries or sectors or geographic regions.

9.2.4 Calculating Compliance Costs

The first step in assessing impacts is to estimate and verify the costs of compliance. This step is necessary regardless of whether the entities affected are for-profit, governmental, communities, or not-for-profit entities. Compliance costs include annual costs (such as operation and maintenance of pollution control equipment or increased production cost) and any capital costs. In certain situations it may be appropriate to estimate the costs year by year, especially in cases where the costs are expected to vary over time. Depending on the nature of the analysis, these costs may be annualized, so that they can be compared to average annual income and other measures of financial strength.[20]

Verifying the compliance cost estimates entails two steps. First, the full range of responses to the rule needs to be identified, including pollution prevention alternatives. Second, the costs for each response need to be checked to determine if all elements are included and the costs are consistent with a given base year. Either a general inflation factor, such as the Gross Domestic Product (GDP) implicit price deflator, or various cost indices specific to the type of project can be used.[21] The base year and indexing procedure should be stated clearly. Implicit costs that do not represent direct outlays may be important. The cost estimates should include such elements as production lost during installation, training of operators, and education of users and citizens (for example, for programs involving recycling of household wastes). The cost of acquiring a permit is not so much the permit fee as it is the lost opportunities during the approval process. Likewise, the cost of having a car's emissions inspected is not so much the fee as it is the value of registrant's time.

EIAs and BCAs use different concepts of cost. BCA relies on estimates of the social costs of a regulation. EIA costs are the private costs needed to predict compliance responses and assess economic impacts in several ways. Social costs represent costs to society as a whole, whereas private costs reflect costs as they are experienced by the affected parties. Often, the same basic engineering compliance cost estimates are used as the basis for developing both social and private cost estimates and are adjusted to provide the required costs.

There are several issues analysts must consider when estimating the private costs of environmental polices. These include:

- **Before- versus after-tax costs:** The costs of complying with regulations are generally deductible as expenses for income tax purposes. The effective burden of compliance costs is reduced for taxable entities because they can reduce their taxable income by the amount of the compliance costs. The effect of a regulation on profits is therefore measured by after-tax compliance costs. Different components of engineering cost estimates should be adjusted based on their specific impact on taxes, to provide the inputs needed for an EIA.[22] Operating costs are generally fully deductible as expenses in the year incurred. Capital investments associated with compliance must generally be depreciated over some number of years.[23]

  In most cases, communities, not-for-profits, and governments do not benefit from reduced income taxes that can offset compliance costs. Therefore, adjustments to cost estimates, annualization formulas, and cost of capital calculations required to calculate after-tax costs should not be used in analyses of impacts on governments and not-for-profits.

[20] As previously discussed, the discount rate used should be specific to the task. The rate used to annualize costs is dependent on the entity's cost of capital and, thus, the sources of financing used as well as the credit rating of the borrower. When calculating the present value of a stream of future social costs (or benefits), the social rate of time preference is the appropriate discount rate. See Chapter 6 for a complete discussion of discount rates.

[21] The GDP implicit price deflator is reported by the U.S. Department of Commerce, Bureau of Economic Analysis in its *Survey of Current Business*. The annual *Economic Report of the President*, Office of the President, is another convenient source for the GDP deflator time series.

[22] Engineering costs can often be used in their before-tax form to calculate social costs. Adjustments may be required, however, if the available compliance cost estimates do not reflect the social cost of the resources used.

- **Transfers:** Social costs reflect the real value of economic resources—labor, equipment, supplies—used to achieve compliance. However, some types of compliance costs incurred by the regulated parties may represent transfers among parties. Transfers, such as payments for insurance or payments for marketable permits do not reflect use of economic resources. Private cost estimates used in the EIA should include such transfers, but these transfers should be excluded when calculating social costs for the benefit-cost analysis.

- **Discounting:** Compliance costs often vary over time, perhaps requiring initial capital investments and then annual operating costs. To estimate impacts, the stream of costs is generally discounted to provide a Present Value of Costs (PVC) that reflects the time value of money.[24] In contrast to social costs and benefits, which are discounted using a social discount rate, private costs are discounted using a rate that reflects the regulated entity's cost of capital.[25] The private discount rate used will generally exceed the social discount rate by an amount that reflects the risk associated with the regulated entity in question.[26] For firms, their cost of capital may also be determined by their ability to deduct debt from their tax liability.

- **Annualized costs:** Annualizing costs involves calculating the annualized equivalent of the stream of cash flows associated with compliance. It provides a single annual cost number that reflects the various components of compliance costs incurred over some selected time period (e.g., 15 or 30 years).[27] The annual value is the amount that, if incurred each year over the selected time period, would have the same present value as the actual stream of compliance expenditures. Annualized costs are therefore a convenient compliance cost metric that can be compared with annual revenues and profits. It is important to remember that using annualized costs masks the timing of actual compliance outlays. For some purposes, using the underlying compliance costs may be more appropriate. For example, when assessing the availability of financing for capital investments, it is important to consider the actual timing of capital outlays.

- **Fixed versus variable costs:** Some types of compliance costs vary with the size of the regulated enterprise (e.g., in proportion to production). Other components of cost may be fixed with respect to production, such as the costs involved in reading and understanding regulatory requirements.[28] Requirements that impose high fixed costs will impose a higher cost per unit of production on smaller firms than on larger firms. It is important that the effects of any "economies of scale" be reflected in the compliance costs used to analyze economic impacts.[29] Using the same average annualized cost per unit of production may mask the importance of such fixed costs and understate impacts on small entities.

9.2.5 Screening for Potentially Significant Impacts

A comprehensive analysis of all aspects of economic impacts associated with a rule can be highly resource intensive. Detailed and explicit analysis of impacts may not be justified in all cases, if a preliminary analysis

[23] Current federal and state income tax rates can be obtained from the Federation of Tax Administrators in Washington, D.C. FTA, *State Tax Rates & Structure*, available from http://www.taxadmin.org/fta/rate/tax_stru.html (accessed 8/28/2000).

[24] This Present Value of Costs may then be annualized to provide an annual equivalent of the uneven compliance cost stream, as described below.

[25] While the discount rate differs, the formula used to discount private costs is the same as used for social costs. See Chapter 6.

[26] Risk adjusted rates for different industries can be obtained from the Ibbotsen Associates *Handbook* or for specific firms from the Value Line *Investment Survey*.

[27] Annualized costs are also discussed in Chapter 6 on social discounting. The formula for calculating annualized costs is presented in Section 6.2.

[28] Note that fixed versus variable costs is not the same thing as capital versus operating costs. Capital costs may be fixed or variable with respect to the size of the operation, as may operating costs. This distinction between capital and operating costs is important for calculating after-tax costs.

[29] Economies of scale characterize costs that decline on a per unit basis as the scale of an operation increases.

suggests that economic impacts will be minor. Screening analyses provide a way to focus attention and analytic resources on the areas where economic impacts are most likely to be significant. These screening analyses generally use simplifying assumptions about market outcomes (e.g., the alternative no-cost-pass-through and full-cost-pass-through assumptions) or simple tests of financial impacts (e.g., a ratio of compliance costs to sales or to profits) to screen for potentially significant impacts.

It is important to keep in mind the limitations in screening analyses when interpreting and presenting their results. They typically identify cases of potentially significant impacts. More detailed investigation beyond the screening analysis is usually needed to reach a reliable conclusion about the likelihood of significant impacts.

In addition, screening analysis criteria should be chosen to balance the risk of identifying "false positives" versus "false negatives." That is to say, using too low a threshold will do little to distinguish true differences in potential impacts (false positives), while using too high a threshold runs the risk of missing some sectors that may be significantly affected (false negatives).

Finally, when screening analyses are based on alternative assumptions about market responses, it is important to note in presenting the results that they represent extreme assumptions that in practice cannot occur simultaneously. For example, worst case impacts on profits cannot occur simultaneously with worst case impacts on prices.[30]

9.2.6 Profile of Affected Entities

9.2.6.1 Compiling an Industry Profile and Projected Baseline

The impact of a regulation on business profitability and other economic outcomes depends on the magnitude of the compliance costs associated with the rule, on the distribution of compliance costs across firms, and on the economic and financial characteristics of the affected firms and industries.[31] A substantial portion of an EIA involves characterizing the affected firms and industries as a basis for evaluating economic impacts. The following are important inputs to an EIA:

- **Standard Industrial Classification (SIC) and North American Industrial Classification System (NAICS) industry codes:** These definitions can be developed by working with engineering analysts, the EPA project team and workgroups, industry roundtables and industry specialists at the Department of Commerce. The SIC codes and their definitions appear in the *Standard Industrial Classification Manual: 1987*, available from the Government Printing Office (OMB, 1987). This industry classification system is being replaced by the North American Industry Classification System (NAICS), which will be reflected in the forthcoming data series (OMB, 1998). A regulated entity that is a small part of a larger industry may require fractional multipliers in order to estimate the regulated category's share of the larger industry.
- **Compilation of summary statistics:** Data regarding total employment, revenue, number of establishments, and number of firms are available from the economic censuses and interim updates (e.g., Department of Commerce *Annual Survey of Manufactures*, for non-manufacturers, the Department of Commerce *County Business Patterns*, and *Agricultural Statistics* from the Department of Agriculture). The profile should also define the industry and its products, describe major production technologies, and discuss important business and regulatory trends.
- **The level and distribution of compliance costs:** Estimates of compliance costs reflect predicted responses to the rule and are often developed based on engineering estimates. It is important to know how costs are distributed among plants and firms in the same industry, since firms that are not affected by the rule or that incur lower compliance costs than their competitors may gain competitive advantage as a result of the rule. If only a few producers in an industry incur added costs, they are less likely to be

[30] A more detailed treatment of the considerations in the conduct of sensitivity analysis is presented in Chapter 5.

[31] Generally, analysts should presume a perfectly competitive market structure. The purpose of developing an industry profile is to confirm this presumption or discover evidence to the contrary.

able to raise their prices to recover costs. In contrast, a rule that affects all industry participants equally is more likely to result in price increases and less likely to change the competitive structure of the industry. In addition, some rules impose different requirements and costs on new versus existing sources. Such rules may affect industry competition, growth, and innovation by raising barriers to new entry.

- **Baseline industry growth and financial condition:** Industries and firms that are relatively profitable in the baseline will be better able to absorb new compliance costs without experiencing financial distress. Industries that are enjoying strong growth may be better able to recover increased costs through price increases than they would have been had there been no demand growth. Section 9.2.9 provides suggestions for using specific ratios to assess the significance of economic impacts on a firm's financial condition.

- **Baseline industry structure:** Industry-level impacts depend on the competitive structure and organization of the industry and the industry's relationship to other economic entities. In addition, the number and size distribution of firms and facilities and the degree of vertical integration are important aspects of industry structure that affect the economic impacts of regulations.

- **Characteristics of supply and demand:** Assessing the likelihood of changes in production and prices requires information on the characteristics of supply and demand in the affected industries. The relevant characteristics are reflected in price elasticities of supply and demand, which, if available, allow direct quantitative analysis of changes in prices and production. Often, reliable estimates of elasticities are not available, and the analysis of industry-level adjustments must rely on simplifying assumptions and qualitative assessments.

The industry profile provides a general understanding of an industry or industries affected by a regulatory action and characterizes their ability to absorb compliance costs. This information provides the basis for assumptions central to the impact analysis, as well as information needed for some of the special analyses such as Regulatory Flexibility Analyses.

9.2.6.2 Profile of Other Affected Entities

Careful consideration needs to be given to the question of whether or not a particular rule will affect government entities,[32] not-for-profit organizations,[33] or households. For example, air pollution regulations that apply to power plants may affect municipally owned electric companies; air regulations that apply to vehicles may affect municipal bus companies, as well as other municipal vehicles such as police cars and public works vehicles; effluent guidelines for machinery repairing activities may affect municipal garages. Thus, the first step is to identify all the government entities that may be affected.

Relevant characteristics of government entities may include:

- the community's size (number of people living in the community);
- household income levels (both median and some measure of the income range);
- number of children (since education is frequently the major service provided by local governments);
- number of elderly residents (who frequently have fixed incomes);
- unemployment rate;
- revenue amounts by source; and
- the credit or bond rating of the community.

If the property tax is the major revenue source, then the assessed value of property in the community and the percentage of this assessed value represented by residential versus commercial and industrial property should be determined. If the government entity serves multiple communities, such as a regional water or sewer authority,

[32] Government entities that may be affected by a program include states, cities, counties, towns, townships, water authorities, villages, Indian Tribes, special districts, military bases, etc.

[33] Examples of not-for-profits include non-profit hospitals, colleges, universities, and research institutions.

then this information needs to be collected for all the communities covered by the entity.

Data on community size, income, number of children and elderly, and unemployment levels are available from the U.S. Bureau of Census. Data on property values, amount of revenue collected from each revenue source, and credit rating will need to be collected directly from the community or state finance agencies. If the regulated activity is provided by an "Enterprise Fund" then revenue and cost information will need to be obtained directly from the fund.[34] Depending on the number of communities affected and the level of detail warranted, the analysis may rely on generally available data only. In other cases, a survey of affected communities may be necessary. However, in cases where a survey is needed, there will be a need to comply with the requirements of the Paperwork Reduction Act (PRA).

Relevant characteristics of not-for-profit entities include their size, the goods or services they provide, their operating costs, and the amount and sources of their revenue. If the entity is raising its revenues through user fees or in other ways charging a price for its goods/services (such as university tuition), then the income levels of its clientele are relevant. If the entity relies on contributions, then it would be helpful to know the financial and demographic characteristics of its contributors. If it relies on government funding (such as Medicaid) then possible future changes in these programs should be identified.

Relevant features of households are standard socioeconomic and demographic characteristics. These characteristics include, for example, their income level, size, age distribution, education level, and ethnic group.

9.2.6.3 Profile Data Sources

Profiles generally draw from at least two types of information: 1) literature from economic journals, dissertations, and industry trade publications, and 2) quantitative data describing the characteristics of the industry. The relevant literature can be useful in characterizing the industry activities and markets as well as regulations affecting the industry. Identification of relevant literature is most efficiently performed through a computerized search using on-line services such as Dialog, BRS/Search Services, or Dow Jones News/Retrieval. These on-line services contain more than 800 databases covering business, economic, and scientific topic areas. Exhibit 9-3 lists some commonly used sources.

The industry profile may also identify those situations where sufficient data for an EIA cannot be obtained through published and commercial sources. These situations arise particularly when the affected industry is one of many product lines or other activities of identified facilities; in addition, for some industries, identification of the appropriate SIC or NAICS code for all the firms or facilities included in the industry may be difficult if the industry can be categorized in a variety of ways. In these cases, and particularly if facility-level data are required to estimate economic impacts, a survey of either a statistical sample or a census of affected facilities may be required to provide sufficient data for analysis.

9.2.7 Impacts on Prices

Predicting impacts on prices is the basis for determining how the burden of compliance costs will be shared between the directly-affected firms and their customers and suppliers in a typical market. At one extreme, regulated firms may not be able to raise their prices at all and they will bear the entire burden of the added costs in the form of reduced profits. Reduced profits may result from reduced earnings on continuing production, lost profits on products or services that are no longer produced, or some combination of the two. At another extreme, firms may be able to raise prices enough to recover costs fully. In this case, there will be no impacts on the profitability of the directly-affected firms but their customers will bear the burden of increased prices. Another possible outcome is that suppliers to the directly-affected firms will bear some of the burden in lost earnings if the regulation results in a decline in demand for particular products.[35]

[34] Public services that are funded entirely by fees charged to users are referred to as "enterprises" and their revenues are referred to as "Enterprise Funds."

[35] Regulations limiting sulfur emissions may result in reduced demand for high-sulfur coal, for example, which will result in a fall in the price of such coal and lost profits for its producers.

Exhibit 9-3 Frequently Used Profile Sources

Source	Data
Trade Publications	Market and technological trends, sales, location, regulatory events, ownership changes.
U.S. Department of Commerce, Economic Censuses[36]	Total revenue by 4-SIC (generally); payroll; quantity and value of products shipped and materials consumed; value added; capital expenditures, assets, inventories, employment, and geographic area, distribution by size, kind of business.
U.S. Department of Commerce, *U.S. Industry & Trade Outlook*	Description of industry, trends, international competitiveness, regulatory events.
United Nations, *International Trade Statistics Yearbook*	Foreign trade volumes for selected commodities, major trading partners
Robert Morris Associates, *Annual Statement Studies*	Income statement and balance sheet summaries, profitability, debt burden and other financial ratios, all expressed in quartiles and available for recent years. Based on loan applicants only.
Dun & Bradstreet, Information Services	Type of establishment, SIC code, address, facility and parent firm revenues and employment.
Standard & Poor's	Publicly held firms, at 4-digit SIC level. Prices, dividends, and earnings, line-of-business and geographic segment information, S&P's ratings. Quarterly History (10 years): income statement, ratio, cash flow, and balance sheet analyses and trends.
Standard & Poor's, Research Reports	Industry profiles, competitors for selected firms. Firm Level Data: (publicly traded companies) company background, stock prices, major competitors, description of business organization, summary financial data.
Securities and Exchange Commission 10k Filings, EDGAR System Database	Income statement and balance sheet, working capital, cost of capital, employment, outlook, regulatory history, foreign competition, lines of business, ownership and subsidiaries, mergers and acquisitions.
Value Line *Industry Reports*	Industry overviews, company descriptions and outlook, performance measures.
FINDS database	Facility SIC, latitude and longitude, zip code, size, ownership structure.

In general, the likelihood that price increases will occur can be evaluated by considering whether competitive conditions will allow the affected facilities to pass on their costs. The methods used to conduct the analysis of the directly-affected markets will depend on the availability of appropriate estimates of supply and demand elasticities. In many cases, reliable estimates of elasticities will not be available.[37] In these cases, the analyst will need to rely on a more basic investigation of the characteristics of supply and demand in the affected market to reach a judgment about the likelihood of full or partial pass-through of costs via price increases.

9.2.8 Impacts on Production and Employment

Regulations may raise the cost of doing business sufficiently to make some or all production unprofitable or may reduce the quantity demanded as producers raise their prices to maintain profitability. The associated reductions in output may result from lower operating rates at existing plants, closure of some plants, or reduced future growth in production relative to what would have occurred in the baseline. Losses in employment are typically associated with reductions in output.

[36] Economic Censuses include: *Census of Manufacturers, Census of Construction Industries, Census of Mineral Industries, Census of Retail Trade, Census of Service Industries, Census of Transportation,* and *Census of Wholesale Trade*.

[37] See Chapter 8 for a more complete discussion of costs and elasticity.

EPA has used a variety of methods to assess reductions in production and employment. In some cases, demand and supply elasticities are used directly to calculate changes in output and prices that would result from a shift upward in the supply curve associated with compliance costs. Often estimates of the shape of the supply curve are not available and assumptions are made about its shape in the region of interest to allow use of demand elasticity estimates to predict output and price adjustments.

In other cases, analysts may assess the impacts of rules on the profitability of specific firms or industry segments, and identify potential line or plant closures based on a financial analysis.[38] If partial or full plant closures are projected, it is important to consider whether the production lost at the affected facilities will be shifted to other existing plants or to new sources or will simply no longer be produced. If there is excess capacity in the industry in the baseline and some plants with excess capacity can operate profitably in compliance with the rule, they may expand production to meet the demand for products no longer produced at plants that can no longer operate profitably.[39]

Even if total production does not decline but is simply shifted from higher-cost plants to more efficient competitors and even if total employment does not change, localized changes in employment may interest policy makers. This is especially the case for rules that may have a strong regional impact. For example, UMRA § 202 requires such an analysis as an element of the UMRA cost analysis. Data on the ratio of production or sales to employment can help predict the number of jobs lost as a result of reductions in production. The regional distribution of job losses can be calculated based on plant locations.

9.2.9 Impacts on Profitability and Plant Closures

The availability of financial information used to assess profitability varies greatly, depending on the industry in question and the extent to which EPA is able to collect new information by surveying the affected entities. With limited exceptions, detailed financial information is not generally available for individual plants or for privately-held companies from published sources. Financial data for publicly-held companies may be too aggregated to allow analysis of the specific business practices affected by the rule. In the absence of new data collection by EPA, analysts may need to rely on financial profiles constructed for model plants, or on industry-average data provided by the Census Bureau and other sources.[40] In some cases, financial profiles used in the analysis of a previous rule-making might be adapted and updated to analyze the impacts of the rule in question.

Analysis is conducted by determining how the added costs of compliance will affect the financial strength of the firm. As with predicting price increases, it may be worthwhile to start with a screening analysis based on an extreme assumption about the incidence of costs—in this case, that no costs can be recovered through price increases. This assumption provides a worst case estimate of impacts on profits, potential closures, and employment reductions in the directly-affected market. Where firms in an industry do not appear to experience financial distress under the no-cost-pass-through scenario, more detailed analyses to predict actual market adjustments and price increases are not needed.

The severity of financial impacts to firms from a rule can range from no impact (if all costs are recovered through price increases, for example) to a modest reduction in profits, closure of a production line or plant, to bankruptcy of the firm. Criteria for assessing the degree of financial distress and for predicting when a production line or plant would be shut-down are not clear-cut.[41] If detailed financial profiles can be developed, including revenues, costs, income statements, and balance sheets, a variety of financial tests can be used to assess the likelihood of financial distress or closure. These tests address the following issues:

[38] Analysis of impacts on profitability and plant closures are discussed later in this section.

[39] Some surviving plants could experience increases in production, capacity utilization, and profits even though subjected to regulatory requirements, if their competitors face even greater cost increases.

[40] Sources of financial data are listed in Exhibit 9-3.

[41] This section assumes a perfectly competitive market, which in practice does not always correctly characterize the market structure being analyzed. In these cases, this section should be adapted to the relevant market structure.

- Do the costs of the regulation result in a negative discounted after-tax cash flow?[42]
- Does the facility or firm's profitability fall below acceptable levels?
- Is the facility or firm's ability to finance its operations and pay its obligations jeopardized?

Establishments that fail the first test are potentially at risk for closure.[43]

Closure Decisions

A variety of considerations affect a firm's decision to close a production line or a plant.

- **The profitability of the plant itself** provides insight into whether the operation will be contined if the plant represents a stand-alone business. This also assumes that it is possible to construct a financial profile of that business.
- **The role the plant plays in a larger operation** may influence closure decisions. For example, some plants may be part of a vertically or horizontally integrated operation. Such plants might not be viable as a stand-alone operation but may continue to operate based on its contribution to the business line as a whole. In general, however, the analysis should assume that an operation will be closed if compliance with the rule would increase costs to the point where continued operation is no longer profitable.
- **A negative discounted cash flow** indicates that returns are below the rate of return required to provide the required return on equity and payment of interest. Closures in the short run are likely to occur if earnings do not cover variable costs plus the cost of compliance. Disinvestment and closures will occur over the longer term if earnings are not sufficient to justify investment in plant and equipment as well.

Where closures and reduced production are likely for some but not all plants, firms may face complex decisions about which plants to close. These decisions reflect relative operating costs, age of equipment, tax and other incentives offered by local communities and states to retain business, and logistical considerations. Analyses of plant closures should include caveats stating that the analysis identifies candidates for closure, rather than providing reliable predictions of which specific plants will close. The available information on plant-level operating costs and contributions to earnings is generally too uncertain to allow more precise prediction of plant closures.

Financial Distress Short of Closure

Short of closure, financial distress may occur. Financial distress measures a continuum from mild to severe financial weakness and may result in difficulties obtaining financing and attracting capital.[44] Although in practice, analysts may use a variety of measures of financial distress, use of specific financial ratios has the advantage that it mirrors analyses that investment and lending institutions perform to evaluate industries and businesses. Particular measures include:

- **Measures of impacts on profitability,** e.g., pre-tax return on assets (net operating income divided by total assets) or return on equity. These measures reflect the profit performance of a firm's capital assets. If returns are reduced to unacceptable levels when compliance costs are included, the firm may have difficulty financing new investment or attracting capital even if it is not earning negative returns.
- **Measures of impacts on liquidity,** e.g., interest coverage ratio (cash operating income divided by interest expense), times-interest-earned (earnings before interest and taxes divided by interest expense), and the current ratio (current assets divided by current liabilities.) These measures reflect the firm's

[42] If after-tax cash flow is negative under baseline conditions (before considering compliance costs), the facility is a likely candidate for closure even in the absence of additional compliance costs. These closures should not be attributed to the rule, but rather should be classified as baseline closures.

[43] If it is possible to estimate plant liquidation values, another test can be added to assess the likelihood of closure. Plants may be predicted to close if the value of continuing to operate is less than the liquidation value.

[44] Researchers have developed various composite measures that are designed to assess the potential for bankruptcy. The most commonly cited is the ZETA model or "Z-score" developed by Altman et al. (1993). This model uses a weighted average of five variables to predict potential for bankruptcy. The ratios include working capital/total assets, retained earnings/total assets, earnings before interest and taxes (EBIT)/total assets, market value of equity/par value of debt, and sales/total assets. The model includes levels for this composite score that represent clear potential for bankruptcy, low or no potential for bankruptcy, and an uncertain grey area.

ability to meet its financial obligations out of current, on-going operations. If operating cash flow does not comfortably exceed its payment obligations, the firm may have to use resources required for on-going operations to pay contractual obligations and its creditworthiness will suffer.

9.2.10 Impacts on Related Industries and Consumers

The economic and financial impacts of regulatory actions propagate to industries and communities that are linked to the regulated industries, resulting in indirect business impacts. These indirect impacts may include employment and income losses, as well as changes in the competitiveness and efficiency of related markets. Compliance-related industries, on the other hand, may yield offsetting gains in employment and income when a regulated industry purchases equipment, facilities or labor in order to comply with a regulation.

Although, in principle, every economic entity can be thought of as having a connection with every other entity, practical considerations usually require an analysis of indirect impacts to be performed or presented for a manageable subset of economic entities that are most strongly linked to the regulated entity. In addition to considering major customers and specialized suppliers of the affected industry, it is also important to consider less obvious but potentially significant links, for example, basic suppliers such as electricity generators.

For this reason, the analysis of linkages should use a framework that thoroughly measures indirect as well as direct linkages. Whatever the approach, the goal of the analysis is to measure—given a certain amount of employment and income change in a regulated market—how employment and income will likely change in related entities.

9.2.11 Impacts on Innovation, Productivity, and Economic Growth

While regulatory interventions can theoretically lead to macroeconomic impacts, such as growth and technical efficiency, such impacts may be impossible to measure or observe.[45] In some cases, however, it may be feasible to use macroeconomic models to evaluate the regulatory impact on gross national product (i.e., including trade effects and plant location decisions), factor payments, inflation, and aggregate employment.

For programs or rules that are expected to have significant impacts in a particular region, use of regional models—either general equilibrium or more limited models—may be valuable.

Some macroeconomic regulatory effects are beyond the capacity of the typical regulatory impact analysis to quantify. For example, price changes induced by a regulation can lead to technical inefficiency because firms are not choosing the production techniques that minimize the use of labor and other resources in the long run. Instead, firms will tend to overuse resources whose prices are artificially depressed by the regulation compared to the resources' true cost to society.

Additional anecdotal, theoretical and limited empirical literature are available that point to possible macroeconomic impacts.[46]

9.2.12 Impacts on Industry Competitiveness

Regulatory actions that substantially change the structure or conduct of firms can produce indirect impacts by changing the competitiveness of the regulated industry, as well as that of linked industries. An analysis of impacts on competitiveness begins by examining barriers to entry and market concentration and by answering two key questions.

[45] OMB states that macroeconomic effects are likely to be measurable only if the impact of the regulation exceeds 0.25 to 0.5 percent of GDP. See OMB (1995).

[46] See Jaffe et al. (1995) and Gray and Shadbegian (1997).

- **Will the regulation erect entry barriers that might reduce innovation by impeding new entrants into the market?** High sunk costs associated with capital costs of compliance or compliance determination and familiarization would be an entry barrier attributable to the regulation. Sunk costs are fixed costs that cannot be recovered in liquidation; they can be calculated by subtracting the liquidation value of assets from the acquisition cost of assets facing a new entrant, on an after-tax basis.[47] Lack of access to debt or equity markets to finance fixed costs of entering the market can also present entry barriers, even if none of the fixed costs are sunk costs. However, if financing is available and fixed costs are recoverable in liquidation, the magnitude of fixed costs alone should not present any barrier to entry.

- **Will the regulation tend to create or enhance market power and reduce the economic efficiency of the market?** The tools presented in the section describing how to create an economic profile also address this question. The most important of these tools include measures of horizontal and vertical integration (i.e., concentration), among both buyers and sellers, in the baseline compared to post-compliance cases. Closely related to concentration, product differentiation may occasionally be either increased or decreased by a regulatory action. For a hypothetical example, certain labeling restrictions might reduce the ability of firms to segment their market by differentiating an essentially uniform product with packaging. In such a hypothetical baseline, firms might enjoy effectively higher concentration ratios and less competition after imposition of a uniform labeling policy.

9.2.13 Impacts on Government Entities and Not-for-Profit Organizations

Section 9.2.9 of this chapter discussed ways of measuring the impact of regulations and requirements on private entities, such as firms and manufacturing facilities. When dealing with private entities, the primary focus is on measures that assess changes in profits. This section describes impact measures for situations where profits and profitability are not relevant—where the regulations affect government entities and/or not-for-profit organizations. Many of the same questions, however, apply:

- Which entities are affected and what are their characteristics?
- How much will the regulation increase operating costs?
- What impact will the regulation have on operating procedures?
- Will this change the amount and/or quality of the goods and services provided?
- Can the entity raise the necessary capital and will this change its ability to raise capital for other projects?

The major difference is that instead of ultimately measuring the regulation's impact on profit levels, when government entities are involved, the ultimate measure is the ability of its citizens to pay for the requirements. Likewise, in the case of a not-for-profit, the measure is the reduction in the organization's ability to provide its goods and services.

9.2.13.1 Measures of Government Impacts

EPA regulations can affect governments in at least three ways. They can directly impose requirements on the governmental entity, such as water pollution requirements for publicly-owned wastewater treatment works (POTWs) or air pollution restrictions that affect municipal bus systems or power plants. Second, they can involve costs for governments to implement and enforce regulations imposed on other parties. Finally, they can impose indirect costs on government entities, such as increased unemployment in a community because an EPA regulation has resulted in reduced production (or even closure) at a factory in the community. Keep in mind that some of the impacts may reduce the community's financial resources and thus its ability to pay for the requirements. For example, the

[47] Sunk costs are sometimes referred to as exit barriers. Without exit barriers, there can be no entry barriers, as long as there are no liquidity constraints.

closure of a facility may increase the drain on social services at the same time that tax revenues are declining.

Impacts of Programs That Directly Affect Government and Not-for-Profit Entities

The direct impact measures can be divided into two categories, (1) those that measure the impact itself in terms of the relative size of the costs and the burden they place on citizens and (2) those that measure the economic and financial conditions of the entity that affect its ability to pay for the requirements. For each category, there are several types of measures that can be used either as alternatives, or jointly, to illuminate various aspects of the question.

Measuring the relative cost and burden of the regulations. There are three commonly used approaches to measuring the burden of the rule; all involve calculating the annualized costs of complying with the regulation. For *government entities*, the three approaches are:

- **Annualized compliance costs as a percentage of annual costs for the service included:** This measure tries to define the impact as narrowly as possible and is particularly appropriate when the service or activity to be regulated is provided by a single-purpose entity. For example, if the regulated activity is sewage treatment, the POTW may not be able to draw on general government revenues to cover its increased cost. Thus the appropriate comparison would be to estimate the resulting increase in its costs. Even if the affected entities are not able to draw on general government revenues, it is useful to know how the rule affects the cost of the activity in question. In practice, EPA has often used the condition that if compliance costs are less than one percent of the current annual costs of the activity, it is usually assumed that the compliance costs are placing a small burden on the entity.

- **Annualized compliance costs as a percentage of annual revenues of the governmental unit:** The second measure corresponds to the commonly used private-sector measure of annualized compliance costs as a percentage of sales. Referred to as the "Revenue Test," it is one of the measures suggested in *EPA Revised Interim Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act* (EPA, 1999).[48] This differs from the prior measure in that it compares annualized compliance costs to the total revenues of the entity (which usually is multi-purpose in nature). If compliance costs are less than one percent of revenues, then the requirements are usually considered to be affordable. Compliance costs in the range of one to three percent of government revenues are less easily interpreted. If all affected communities fall in this range, then further thought should be given to lowering annual compliance costs, if only a small percentage of communities fall into this range and the rest fall below one percent, then the requirements can probably be considered affordable. Compliance-cost-to-revenue ratios of greater than three percent indicate that the requirements are placing a heavy burden on the community.

- **Per household (or per capita) annualized compliance costs as a percentage of median household (or per capita) income:** The third measure compares the annualized costs to the ability of residents to pay for the cost increase. Commonly referred to as the "Income Test," it is described in the *Revised Interim Guidance* (EPA, 1999) and EPA's Office of Water *Interim Economic Guidance for Water Quality Standards. Workbook* (EPA, 1995a).[49] Costs can be compared to either median household or median per capita income. In calculating the per household or per person costs, the actual allocation of costs needs to be considered. If the costs are entirely paid through property taxes, and the community is predominately residential, then an average per household cost is probably appropriate. If, however, some or all of the costs are allocated to users (e.g., fares paid by bus riders or fees paid by users for sewer, water, or electricity supplied by municipal utilities), then this needs to be taken into account. In addition, if some of the costs are borne by local firms,

[48] See Section 9.3 for a discussion of the analytic and procedural requirements under SBREFA.

[49] For example, materials presented in the water guidance and other EPA Office of Water analyses are: less than one percent indicates little impact, over two percent indicates a large impact, with the range from one to two percent being a gray area of indeterminate impact.

then that portion of the costs needs to be handled separately.

Two commonly used impact measures for *not-for-profit entities* are: (1) annualized compliance costs as a percentage of annual operating costs and (2) annualized compliance costs as a percentage of total assets. The first is equivalent to the first of the impact measures described for government entities, measuring the percentage increase in costs that would result from the regulation being analyzed. The second is a more severe test, measuring the impacts if the annualized costs were paid for out of the assets of the institution. As presented in EPA's *Revised Interim Guidelines,* the guidelines for annualized compliance costs as a percentage of annual operating costs are: annualized compliance costs less than 1 percent of operating costs indicate that the rule does not represent a burden, annualized compliance costs between one and three percent of operating costs indicates that the rule may impose a burden, and annualized compliance costs that are more than three percent of operating costs indicates that the rule may impose a heavy burden.

Measuring the economic and financial health of the community: This second category of impact measures looks at the economic and financial health of the community involved, since these will affect its ability to finance expenditures required by a program or rule. A given cost may place a much heavier burden on a poor community than on a wealthy one of the same size. As with the impact measures described above, there are three categories of economic and financial condition measures:

- **Indicators of the community's debt situation:** Debt indicators are important because they measure both the ability of the community to absorb additional debt (to pay for any capital requirements of the rule) and the general financial condition of the community. While several indicators have been developed and used, this section describes two. One measure is the governmental entity's bond rating. Awarded by companies such as Moody's and Standard & Poor's, bond ratings summarize their assessment of a community's credit capacity and thus reflect the current financial conditions of the governmental body. A second frequently used measure is the ratio of overall net debt (the debt to be repaid by property taxes) to the full market value of taxable property in the community. Overall net debt should include the debt of overlapping districts. For example, a household may be part of a town, a regional school district, and a county sewer and water district, all of which have debt that the household is helping to pay off.[50] See Exhibit 9-4 for interpretations of the values for these measures. Neither of these two debt measures will always be appropriate. Some communities, especially small ones, may not have a bond rating. This does not necessarily mean that they are not creditworthy, it may only mean that they have not had an occasion recently to borrow money in the bond market. Second, if the government entity does not rely on property taxes, as may be the case for a state government or an enterprise district, then the ratio of debt to full market value may not be relevant. Information on debt and assessed property values are available from the financial statement of each community. The state's auditor's office is likely to have this information for all communities within the state.

- **Indicators of the economic/financial condition of the households in the community:** There are a wide variety of household economic and financial indicators. Two commonly used ones are: the unemployment rate and median household income. Both measure the financial well-being of households. Unemployment rates are available from the U.S. Bureau of Labor Statistics. Median household income is available from the U.S. Census and some states maintain more up-to-date databases on income levels. Benchmark values for these two measures are presented in Exhibit 9-4.

- **Financial management indicators:** This category consists of indicators measuring the general financial health of the community as an entity, as opposed to the general financial health of the residents. Since

[50] An alternative to the net debt as percent of full market value of taxable property is the net debt per capita. Commonly used benchmarks for this measure are:

Net debt per capita: less than $1,000 = strong financial condition
Net debt per capita: $1,000 and $3,000 = mid-range or gray area
Net debt per capita: greater than $3,000 = weak financial condition

Exhibit 9-4 Indicators of Economic and Financial Well-Being of Government Entities

Indicator	Weak	Mid-Range	Strong
Bond Rating	Below BBB (S&P) Below Baa (Moody's)	BBB (S&P) Baa (Moody's)	Above BBB (S&P) Above Baa (Moody's)
Overall Net Debt as Percent of Full Market Value of Taxable Property	Above 5%	2% to 5%	Below 2%
Unemployment Rate	More than 1 percentage point above national average	Within 1 percentage point of national average	More than 1 percentage point below national average
Median Household Income	More than 10% below the state median	Within 10% of the state median	More than 10% above the state median
Property Tax as Percent of Full Market Value of Taxable Property	Above 4%	2% to 4%	Below 2%
Property Tax Collection Rate	Less than 94%	94% to 98%	More than 98%

most local communities rely on the property tax as their major source of revenues, two indicators of property tax health are presented here. One measures the burden property taxes are placing on the community in terms of property tax revenues as a percent of full market value of taxable property. The second indicator measures the efficiency with which the community's finances are managed, and indirectly, whether the tax burden may already be excessive, in terms of the property tax collection rate. As the property tax burden on tax payers increases, they are more likely to not pay their taxes or pay them late.

Measuring the financial strength of not-for-profit entities includes assessing (1) how much reserve the entity has, (2) how much debt the entity already has and how its annual debt service compares to its annual revenues, and (3) how the entity's fees or user charges compare with the fees and user charges of similar institutions. Again, this part of the analysis is meant to judge whether the entity is in a strong or weak financial position to absorb additional costs.

Impacts of Programs That Place Administrative and Enforcement Burdens on Governments

Many EPA programs require effort on the part of different levels of government for administration and/or enforcement. These costs must be considered to comply with UMRA and to calculate the full social costs of a program or rule. EPA is currently investigating methods for estimating and evaluating the impacts of such costs. Revisions to this guidance document will be made in the future to incorporate the results of that work.

Impacts of Programs That Indirectly Affect Government Entities

The previous section describes how to measure the impact of regulations that directly affect the provision of goods or services by government or not-for-profit entities. This section addresses the indirect or induced impacts on government entities. For example, a manufacturing facility may reduce or suspend production in response to an EPA regulation, thus reducing the income levels of its employees. In turn, these reductions will propagate through the economy by means of changes in household expenditures. These induced impacts include the familiar multiplier effect, in which loss of income in one household results in less spending by that household and, therefore, less income in households and firms associated with goods previously purchased by the first household. Unlike production-based linkages, income-based linkages tend to be more geographically localized, with the strength of the linkage typically decreasing as geographic distance increases (although the number of linked economic entities increases with distance).

Decreased household and business income can affect the government sector by reducing tax revenues and increasing expenditures on income security programs (the automatic stabilizer effect), employment training, food and housing subsidies, and other fiscal line items. Due to wide variation in these programs and in tax structures, estimating public sector impacts for a large number of government jurisdictions can be prohibitively difficult.

On the other hand, compliance expenditures increase income for businesses and employees that provide compliance-related goods and services. These income gains also have a multiplier effect, offsetting some of the induced losses in tax revenue and increases in government expenditures identified above.

9.3 Equity Assessment

9.3.1 Introduction

In the context of an EPA economic analysis, an equity assessment is an important type of distribution analysis. An equity assessment examines the accrual of a regulation's net costs, net benefits, or other economic impacts to a specific sub-population(s) and/or examines the distribution of these costs, benefits, and impacts among sub-population(s). This examination includes the possibility of analyzing a regulation's impact on the distribution of national income or wealth.

Generally, cost bearers and beneficiaries belong to one of four populations: individuals, businesses, not-for-profit organizations, or governments. Within each of these populations are sub-populations whose particular circumstances EPA wishes to better understand, either because the sub-population is more physically susceptible to environmental contamination, is less than fully capable of representing its own interests, or is economically disadvantaged or vulnerable. For many of these sub-populations, the EPA has been directed by statute, executive order, or agency policy to examine the effects of its rules when they are expected to have a "disproportionate," "significant and substantial," or other such impact on a particular sub-population. An equity assessment gives rule makers a better understanding of the economic effects of the EPA's rules on these sub-populations and, for comparison or other purposes, on other specific sub-populations as well. An equity assessment examines the magnitude as well as the distribution of effects on sub-populations.

There are several considerations to keep in mind when performing an equity assessment. Each of these points will be detailed in this chapter.

- **There are specific equity dimensions that must always be considered, but there are none that must always be analyzed** when assessing the impact of environmental regulations. Generally speaking, the regulation under review, and the specific issues associated with it, will determine which equity dimensions are relevant and in need of an equity assessment.
- **The methods used by a regulation's BCA and EIA should guide the methodology used by an equity assessment.** Neither this chapter nor OMB's *Best Practices* and *OMB Guidelines* outline a specific methodology for conducting an equity assessment. However, the models and assumptions developed for these other two analyses should not conflict with those used by an equity assessment.

  An equity assessment may draw on the information compiled for its corresponding BCA as well as include measures of impact similar to those in its corresponding EIA. An early step in an equity assessment is to identify sub-populations likely to be affected by a regulation. Once identified, if data permits, the social costs and benefits estimated for the BCA can be disaggregated and net benefits examined for the sub-population(s). An equity assessment also examines other economic impacts, such as increases in rates of unemployment or other traditional impact measures, for the identified sub-populations(s).

- **An equity assessment is not an independent form of economic analysis,** but is a reflection of decisions made on many of the other analytic issues arising in the benefit, cost, and economic impact sections of this guidance. For example, the analysis of distributions over long time horizons can be greatly affected by the analyst's treatment of uncertainty or choice of a discount rate. Baseline issues are important in the determination of net benefits expected to accrue to specific sub-populations. When making decisions on these and other relevant issues, the analyst should keep in mind the ramifications borne by the equity assessment.

Many of the instructions offered in the preceding sections on estimating economic impacts will be directly applicable to an equity assessment. The difference will be extending the analysis or presentation of results to describe sub-population(s) for which impacts are estimated. Whereas an EIA focuses on traditional classifications of affected populations (like industrial classifications), an equity assessment often focuses on "disadvantaged or vulnerable" sub-populations (like low income households) but, for comparison or other purposes, can also focus on other relevant sub-populations (like upper income households). In this section, we outline a general framework for conducting an equity assessment but refer the reader to Sections 5, 6, and 9.2 for approaches used to estimate specific impacts. As is true for an EIA, generally speaking, an equity assessment is more concerned with sub-populations experiencing net costs or other negative impacts than those experiencing net benefits or positive impacts.

The following parts of this section accomplish three objectives. First, the existing environmental and administrative statutes, executive orders, and agency policies which direct analysts to consider specific sub-populations when assessing the economic effects of EPA's regulations are reviewed. The statutes and policies are enumerated, the relevant sub-populations are identified, and definitions are established for these sub-populations. Second, a broad framework for conducting an equity assessment is outlined. This section concludes with a review of general sources of data for assessing equity impacts.

9.3.2 Statutes and Policies Requiring Equity Assessment and Definitions of Sub-Populations

Equity issues are at the heart of two existing statutes—The Regulatory Flexibility Act of 1980 (RFA), as amended by the Small Business Regulatory Enforcement Fairness Act of 1996 (SBREFA), and The Unfunded Mandates Reform Act of 1995 (UMRA)[51]—and two executive orders—EO 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations" and EO 13045, "Protection of Children From Environmental Health Risks and Safety Risks,"—all of which require agencies to consider a regulation's distributional impact on various entities or sub-populations.[52] These administrative laws or orders suggest several equity dimensions; in particular, entity size, minority status, income level, and childhood status. Exhibit 9-5 lists these equity dimensions and links each to the relevant statute or order, to a population, and to at least one established definition of sub-populations.

A second executive order—EO 12866, "Regulatory Planning and Review" has multiple objectives regarding regulatory planning and review, many that have nothing to do with improvements in equity. It does, however, include a specific directive for agencies to consider distributive impacts and equity when designing regulations. Thus, Exhibit 9-5 lists the equity dimensions suggested by the OMB's *Best Practices* for EO 12866 and noted in its *OMB Guidelines* for economic analyses. The equity dimensions are discussed in more detail below.

9.3.3 Entity Size

The RFA as amended by SBREFA and UMRA require agencies to consider economic effects on small entities—specifically, small businesses, small governmental

[51] EO 13132, *Federalism* and EO 13084, *Consultation and Coordination With Indian Tribal Governments*, both support some objectives in UMRA.

[52] The Office of Regulatory Management and Information's Rule and Policy Information Development System (RAPIDS) http://intranet.epa.gov/rapids (accessed 4/05/2000, internal EPA document) is a resource for those who wish to read relevant statutes, executive orders or Agency policy documents in their entirety or to acquire copies.

Exhibit 9-5 Equity Dimensions Potentially Relevant to Environmental Policy Analyses[53]

Equity Dimension	Administrative Law or Order	Population	Definitions of Sub-Populations
Entity Size	RFA; UMRA; OMB Guidance to EO 12866	Businesses, Governmental Jurisdictions, Not-for-Profit Organizations	The RFA references the Small Business Act definition of small business which defines small business using SIC codes. Definitions sometimes depend on number of employees and other times depend on annual receipts. The RFA defines small governmental jurisdiction as the government of a city, county, town, school district, or special district with a population of less than 50,000. The RFA defines a small not-for-profit organization as an enterprise which is independently owned and operated and is not dominant in its field.[54] UMRA defines small government jurisdiction, similar to the RFA, as the government of a city, county, town, school district, or special district with a population of less than 50,000, and any tribal government.
Minority Status	E.O. 12898; OMB Guidance to EO 12866; EO 13084 for Indian tribal communities only	Individuals or Households	Minority population of the affected area exceeds 50 percent **or** minority population percentage of the affected area is meaningfully greater than the minority population percentage in the general population or other appropriate unit of geographic analysis. (Minorities are those individuals classified by OMB Directive No. 15 as Black/African American, Hispanic, Asian and Pacific Islander, American Indian, Eskimo, Aleut, and other non-white persons.) "Indian tribe" means an Indian or Alaska Native tribe, band, nation, pueblo, village, or community that the Secretary of the Interior acknowledges to exist as an Indian tribe pursuant to the Federally Recognized Indian Tribe List Act of 1994, 25 U.S.C. 479a. (See EO 13084.)
Income Level	EO 12898; OMB Guidance to EO 12866	Individuals or Households	Annual statistical poverty thresholds from the U.S. Bureau of the Census' *Current Population Reports, Series P-60 on Income and Poverty*. Consumers grouped according to consumption expenditures (e.g., into deciles).

[53] Some environmental statutes may also identify sub-populations that merit additional consideration, but this document is limited to those with broad coverage.

[54] The RFA also allows agencies to establish an alternative definition of small entity after notice-and-comment, and for small businesses only, after consultation with the Small Business Administration (SBA).

Exhibit 9-5 Equity Dimensions Potentially Relevant to Environmental Policy Analyses (Continued)

Equity Dimension	Administrative Law or Order	Population	Definitions of Sub-Populations
Childhood Status, Age	EO 13045, OMB Guidance to EO 12866	Individuals or Households	EPA's Office of Children's Health Protection does not adhere to a single definition of "child." It suggests that the definition will vary depending upon the issue(s) of concern. U.S. Bureau of the Census reports statistics by age in five-year age groups and for the following special age categories: 16 years and over; 18 years and over; 15 to 44 years; 65 years and over; 85 years and over.
Gender	OMB Guidance to EO 12866	Individuals	Male/Female
Time	OMB Guidance to EO 12866	Individuals or Households	Current/Future Generations
Physical Sensitivity	OMB Guidance to EO 12866	Individuals or Households	Varies according to the rule under review. For example, a rule that controls an air pollutant might define a physically sensitive sub-population as individuals with asthma.

jurisdictions, and/or small not-for-profit organizations. Definitions of "small" for each of these entities are considered below. For guidance as to when it will be necessary to examine the economic effects of a regulation under the RFA, analysts should consult EPA guidelines on these administrative laws.[55] These guidelines include the types of economic effects that must be considered and establishment of the baseline for purposes of determining if a rule may have a significant economic impact on a substantial number of small entities. Further, these guidelines explain the requirements in the event the rule is found to have a significant economic impact on a substantial number of small entities. Note that the RFA only applies to rules for which notice-and-comment rulemaking is required.

9.3.3.1 Small Business

The RFA requires agencies to begin with the definition of small business that is contained in the SBA's small business size standard regulations.[56] The SBA defines small business by category of business using SIC codes, and in the case of manufacturing, generally defines small business as a business having 500 or fewer employees. For some types of manufacturing, however, the SBA's size standards define small business as a business having up to 750, 1000, or 1500 employees, depending on the particular type of business. In the case of agriculture, mining, and electric, gas, and sanitary services, the SBA size standards generally define small business with respect to annual receipts (from $0.5 million for crops to $25 million for certain types of pipelines).

[55] U.S. Environmental Protection Agency, *Revised Interim Guidance for EPA Rulewriters: Regulatory Flexibility Act as Amended by the Small Business Regulatory Enforcement Fairness Act*, dated March 29, 1999.

[56] 5 U.S.C. § 601; see also the SBA's "Small Business Size Regulations" are contained in the Code of Federal Regulations at 13 CFR 121 and in the Federal Acquisition Regulation 46 CFR 19. The SBA reviews and reissues the size standards every year. The current version can be viewed at: http://www.sbaonline.sba.gov/gopher/Financial-Assistance/Size-Standards/ (accessed on 8/28/2000).

The RFA also authorizes any agency to adopt and apply an alternative definition of small business "where appropriate to the activities of the agency" after consulting with the Chief Counsel for Advocacy of the SBA and after opportunity for public comment. The agency must publish any alternative definition in the Federal Register.[57]

9.3.3.2 Small Governmental Jurisdiction

The RFA defines a small governmental jurisdiction as the government of a city, county, town, school district, or special district with a population of less than 50,000. Similar to the definition of small business, the RFA authorizes agencies to establish alternative definitions of small government after opportunity for public comment (consultation with the SBA is not required). Any alternative definition must be "appropriate to the activity of the agency" and "based on such factors as location in rural or sparsely populated areas or limited revenues due to the population of such jurisdiction." Any alternative definition must be published in the Federal Register.[58]

Section 202 of UMRA directs agencies to obtain meaningful input from state, local, and tribal governments for each proposed and final rule "containing significant federal intergovernmental mandates." More specifically, this requirement is for rules that include federal mandates "which may result in the expenditure by state, local, and tribal governments, in the aggregate, or by the private sector, of $100 million or more in any one year."[59] Section 203 of UMRA requires that agencies assess whether its rules "might significantly or uniquely affect small governments" so as to consider the need for a compliance plan. Small governments are defined in the paragraph immediately above. The phrase "small towns" refers to very small governments with populations of under 2,500 citizens. As part of the "Small Government Agency Plan" required under UMRA, EPA evaluates such factors as whether small governments will experience higher per-capita costs due to economies of scale, whether they would need to hire professional staff or consultants for implementation, or if they would be required to purchase and operate expensive or sophisticated equipment.[60]

9.3.3.3 Small Not-for-Profit Organization

The RFA defines a small not-for-profit organization as an "enterprise which is independently owned and operated and is not dominant in its field." Examples might include private hospitals or educational institutions. Here again, agencies are authorized to establish alternative definitions "appropriate to the activities of the agency" after providing an opportunity for public comment (consultation with the SBA is not required). Any alternative definition must be published in the Federal Register.[61]

9.3.4 Minority Status and Income Level

Executive Order 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations" and its accompanying memorandum have the primary purpose of ensuring that "each federal agency shall make achieving environmental justice part of its mission by identifying and addressing, as appropriate, disproportionately high and adverse human health or environmental effects of its programs, policies, and activities on minority populations and low income populations... ."

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[57] U.S. Environmental Protection Agency, *Revised Interim Guidance for EPA Rulewriters: Regulatory Flexibility Act as Amended by the Small Business Regulatory Enforcement Fairness Act*, dated March 29, 1999.

[58] Ibid.

[59] U.S. Environmental Protection Agency, *Interim Guidance an the Unfunded Mandates Reform Act of 1995*, memorandum from Office of General Counsel, March 23, 1995b.

[60] Ibid. p. 4.

[61] U.S. Environmental Protection Agency, *Revised Interim Guidance for EPA Rulewriters: Regulatory Flexibility Act as Amended by the Small Business Regulatory Enforcement Fairness Act*, dated March 29, 1999.

The Executive Order also explicitly calls for the application of equal consideration for Native American programs.[62]

EPA's *Interim Final Guidance for Incorporating Environmental Justice Concerns in EPA's NEPA Compliance Analyses* (EPA, 1998), discusses the meaning of key terms and phrases contained in EO 12898. Their conclusions regarding four key phrases: "minority population," "low-income population," "disproportionately high and adverse human health effects," and "disproportionately high and adverse environmental effects" are summarized below.

9.3.4.1 Minority Population

Minority individuals are those classified by Office of Management and Budget Directive No. 15 as Black/African American, Hispanic, Asian and Pacific Islander, American Indian, Eskimo, Aleut, and other non-white persons. A minority population should be identified where either (1) the minority population of the affected area exceeds 50 percent or (2) the minority population percentage of the affected area is meaningfully greater than its corresponding percentage in the general population (or other appropriate unit of geographic analysis). A minority population also exists if there is more than one minority group present and the percentage calculated by aggregating all minority persons meets one of these thresholds. In identifying minority communities, the Agency may consider as a community either: (1) a group of individuals living in geographic proximity to one another or (2) a geographically dispersed/transient set of individuals (such as migrant workers or American Indians), where either type of group experiences common conditions of environmental exposure or effect. The selection of the appropriate unit of geographic analysis may be a governing body's jurisdiction, a neighborhood, census tract, or other similar unit that is to be chosen so as to not artificially dilute or inflate the affected minority population. The selection of the appropriate unit of geographic analysis may also be influenced by the accuracy and precision of environmental quality models.

9.3.4.2 Low-Income Population

Low income populations in an affected area can be identified with the annual statistical poverty thresholds from the U.S. Bureau of the Census' *Current Population Reports, Series P-60 on Income and Poverty*. In conjunction with census data, the analysis should also consider state and regional low-income and poverty definitions as appropriate. In identifying low-income populations, the Agency may consider as a community either a group of individuals living in geographic proximity to one another or a geographically dispersed/transient set of individuals (such as migrant workers or American Indians), where either type of group experiences common conditions of environmental exposure or effects.[63]

One alternative to measuring annual incomes is to examine expected lifetime incomes. Generally, consumption is better than annual income at tracking households' expected lifetime incomes. For example, an analyst might divide the population by consumption deciles and see how the lowest deciles fare. These data will be harder to access as the Census does not contain consumption data.

9.3.4.3 Disproportionately High and Adverse Human Health Effects

When determining whether human health effects are disproportionately high and adverse, the Agency is to consider the following three factors to the extent practicable:

- Whether the health effects, which may be measured in risks and rates, are significant, unacceptable, or above generally accepted norms. Adverse health

[62] In addition, EO 13084, *Consultation and Coordination with Indian Tribal Governments*, requires regulations that "significantly or uniquely" affect the communities of Indian tribal governments and that impose substantial direct compliance costs on such communities to either refund the direct costs incurred or to consult with elected officials and other representatives of the Indian tribal governments and to provide a description of the consultation and/or communication to the Office of Management and Budget.

[63] Two additional tests available for identifying low-income populations in an affected area are: (1) the U.S. Department of Health and Human Services poverty guidelines or (2) the U.S. Department of Housing and Urban Development statutory definition for very low-income for the purposes of housing benefits programs. Information on these and other tests can be found in the CEQ report *Environmental Justice: Guidelines for National Environmental Policy Act* (CEQ, 1997) and the *Interim Final Guidance for Incorporating Environmental Justice Concerns in EPA's NEPA Compliance* (EPA, 1998).

effects may include bodily impairment, infirmity, illness, or death;[64] and

- Whether the risk or rate of hazard exposure by a minority population or low-income population to an environmental hazard is significant and appreciably exceeds or is likely to appreciably exceed the risk or rate to the general population or other appropriate comparison group;[65] and
- Whether health effects occur in a minority or low-income population affected by cumulative or multiple adverse exposures from environmental hazards.

9.3.4.4 Disproportionately High and Adverse Environmental Effects

When determining whether environmental effects are adverse and disproportionately high, the Agency is to consider the following three factors to the extent practicable:

- Whether there is or will be an impact on the natural or physical environment that significantly and adversely affects a minority or low-income population. Such effects may include ecological, cultural, human health, economic, or social impacts on minority communities or low-income communities, when those impacts are interrelated with impacts on the natural or physical environment; and
- Whether environmental effects are significant and are or may be having an adverse impact on minority populations or low-income populations that appreciably exceeds, or is likely to appreciably exceed, those on the general population or other appropriate comparison group; and
- Whether the environmental effects occur or would occur in a minority population or low-income population affected by cumulative or multiple adverse exposures from environmental hazards.

9.3.5 Childhood Status

EO 13045, *Protection of Children From Environmental Health Risks and Safety Risks* states that:

> "A growing body of scientific knowledge demonstrates that children may suffer disproportionately from environmental health risks and safety risks. . . . Therefore, to the extent permitted by law and appropriate, and consistent with the agency's mission, each Federal agency: (a) shall make it a high priority to identify and assess environmental health risks and safety risks that may disproportionately affect children; and (b) shall ensure that its policies, programs, activities, and standards address disproportionate risks to children that result from environmental health risks or safety risks."[66]

The order also states that each "covered regulatory action"[67] submitted to OMB, unless prohibited by law, should be accompanied by "...an evaluation of the environmental health or safety effects of the planned regulation on children."[68] The term "children" is not defined. EPA's Office of Children's Health Protection, established in response to this order, does not use a single definition of "child." They suggest that the definition will vary depending upon the issue(s) of concern. See Exhibit 9-5 for age classifications reported by the U.S. Bureau of the Census. EPA is currently developing a practical guide for valuing children's health effects.

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[64] The definition of adverse health effects contained in specific environmental statues, under whose authority a regulation is being developed, may also guide the consideration of adverse health effects in conducting equity assessments.

[65] The definition of risk or rate of hazard exposure contained in specific environmental statues under whose authority a regulation is being developed may also guide the consideration of risk or rate of hazard exposure in conducting equity assessments.

[66] EO 13045, *Protection of Children From Environmental Health Risks and Safety Risks* effective April 21, 1997.

[67] A "covered regulatory action" is any substantive action in a rule making that is likely to result in a rule that may (a) be economically significant (have an annual effect on the economy of $100 million or more or would adversely affect in a material way the economy, a sector of the economy, the environment, and so on) and (b) concern an environmental health risk that an agency has reason to believe may disproportionately affect children.

[68] EO 13045.

9.3.6 Case Specific Equity Dimensions

EO 12866, *Regulatory Planning and Review* has several requirements that contribute to the preparation of economic information, including a specific directive for agencies to consider distributive impacts and equity when designing regulations. The *OMB Guidelines* (OMB, 2000) also makes note of several specific equity dimensions. But, unlike the laws and orders mentioned above, it does not suggest that analysts must always consider these dimensions. Rather, the suggestion is that the regulation under review should determine which equity dimensions are relevant.

> "Those who bear the costs of a regulation and those who enjoy its benefits often are not the same people. Regulations have 'distributional effects' that affect different segments of the population and economy in various ways: by income groups, race, sex, industrial sector, and others. Regulations often distribute benefits and costs unevenly over time, perhaps spanning several generations... . If these distributive effects are important, you should describe the effects of various regulatory alternatives quantitatively to the extent possible, including their magnitude, likelihood, and incidence of effects on particular groups. You should carefully analyze regulations that significantly affect outcomes for different groups."[69]

OMB seems to be offering a general directive to study distributive effects on any grouping of sub-populations when those effects are expected to be significant, without requiring agencies to always consider a predetermined set of equity dimensions.

9.3.6.1 Additional Equity Dimensions

In its general directive, OMB specifically mentions gender as a way to divide effects. Certain regulations may be found to have differential impacts on males and females. Thus, we add gender to the equity dimensions in Exhibit 9-5.

Later in the *OMB Guidelines* it states, "The economic analysis should also present information on the streams of benefits and costs over time in order to provide a basis for judging intertemporal distributional consequences, particularly where intergenerational effects are concerned."[70] This leans more towards being a directive and suggests that time is an important equity dimension. The *OMB Guidelines* give some suggestions for conducting an intergenerational analysis including:

> "Special approaches may also be appropriate when comparing benefits and costs across generations. One approach is to follow the discounting method discussed above, and address the intergenerational equity and fairness issues explicitly, instead of modifying the discount rate."
>
> "One alternative approach is based on the perspective that this generation is concerned about the welfare of future generations and, in fact, is willing to defer consumption and invest or preserve resources for future use at a discount rate that is less than the discount rate used in making decisions within a generation. For this purpose, you could use as a discount rate a special rate of time preference based on the growth of per capita consumption. Again, check with us if you plan to use such an approach."[71]

Both OMB and EPA recognize that inter-generational equity issues are potentially addressed by applying a discounting procedure. In the quotation above, OMB offers some analytical approaches to inter-generational discounting. Chapter 6 of this document provides information on alternative methods of discounting in this context and discusses when such discounting is, and is not, appropriate.

When discussing risk assessment, the *Best Practices* mentions that,

> "Exposures and sensitivities to risks may vary considerably across the affected population. These difficulties can lead; for example, to a range of quantitative estimates of risk in health and ecological risk assessments that can span

[69] *OMB Guidelines*, p. 16.

[70] Ibid., p. 8.

[71] Ibid., p. 8.

> several orders of magnitude... . All of these concerns should be reflected in the uncertainties about outcomes that should be incorporated in the analysis."[72]

Hence, we include physical sensitivity as an important equity dimension. The definition of who precisely is physically sensitive will vary according to the rule being developed. For example, a rule that controls an air pollutant might have a large impact on individuals with asthma. Or, a rule that diminishes the quantity of a hazardous substance that winds up in soils near residential areas, might have a large impact on children with pica (a disorder that results in an urge to eat non-food substances such as dirt).

The *Best Practices* is not the only source directing attention to physical sensitivity. There are sections of environmental statutes which require EPA to address sensitive populations, analyze effects, and take actions to avert or mitigate adverse impacts. For example, the Clean Air Act section 108(f)(1)(C) requires the Administrator to publish and make available "information on other measures which may be employed to reduce the impact on public health or protect the health of sensitive or susceptible individuals or groups... ."

Finally, the *Best Practices* mentions that economic analyses might need to consider different age categories.

> "The literature identifies certain attributes of risk that affect value. These attributes include the baseline risk, the extent to which the risk is voluntarily or involuntarily assumed, and features (such as age) of the population exposed to risk. For regulations affecting some segments of the population (e.g., infants) more than those groups which have served as the basis for most of the information used to estimates *(sic.)* values of a statistical life (e.g., working-age adults), the use of values of a statistical life from the literature may not be appropriate."[73]

Age is clearly the issue in EO 13045, though its specific focus is on childhood status. Thus, Exhibit 9-5 lists childhood status and age as two aspects of a single equity dimension and cites the *Best Practices* and *OMB Guidelines*, as well as EO 13045. The next three entries of Exhibit 9-5 list the other equity dimensions suggested by the *Best Practices* and *OMB Guidelines* and links each to a population and at least one established definition of sub-populations.

While directing agencies to consider the differential impact of a regulation on relevant sub-populations, the *OMB Guidelines* state that an economic analysis should focus on the distribution of the costs and benefits of complying with a regulation rather than on the financial well-being of regulated entities.

> "Since generally accepted principles do not exist for determining when one distribution of net benefits is more equitable than another, you should describe distributional effects without judging their fairness. You should describe these effects broadly, focusing on large groups with small effects per capita, as well as on small groups experiencing large effects per capita. You should also note any equity issues not related to the distribution of policy effects if they are important, and describe them quantitatively to the extent you can."[74]

OMB cautions analysts conducting distributional analyses to recognize that transfer payments will become relevant; to avoid double-counting even when mixing monetized and physical effects and to describe distributional effects without judging their fairness.

9.3.7 A Framework for Equity Assessment

What follows is a very general three-step framework to guide analysts conducting equity assessments. Instructions for estimating particular impacts on sub-populations are given above in the section on EIA. Whether disaggregating benefits and costs or estimating economic impacts, the primary purpose of an equity assessment is

[72] *Best Practices*, p. 15.

[73] Ibid., p. 29.

[74] *OMB Guidelines*, p. 16.

to examine regulatory consequences for specific sub-populations of concern. Thus, the framework developed here offers an approach on how to identify a sub-population to be analyzed.

For each step, choosing to measure the equity-related consequences of a regulation involves balancing costs of data acquisition and analysis against the value of improved accuracy. The framework attempts to conserve resources by screening out situations for which any of the variety of equity impacts probably will not occur. This permits more extensive analytical and empirical efforts to focus on circumstances with a higher probability of creating significant equity-related effects. The three steps should not be viewed as necessarily sequential. Instead, at the outset of a particular regulatory analysis, all aspects of the suggested approach should be considered. This will help to ensure that the data gathered and the analyses performed will be well suited to measuring the equity impacts of concern.

9.3.7.1 Step 1: Equity Scoping Analysis

This first step consists of several tasks described here in sequential order.

- **Determine which populations listed in Exhibit 9-5 are within the scope of the analysis or exist relevant markets.** In certain cases, some of the populations might not be connected closely enough to the regulation to be meaningfully affected. For example, governmental entities might not be involved in the activities that would be affected by a regulation. If so, then no further analysis is necessary for these populations. It will be useful to make this determination early so that resources may be used in the most effective manner possible.

- **Determine whether the rule or regulatory alternative imposes costs, offers benefits, or results in other economic effects too small to warrant further analysis.** When considering the cost side of the analysis, it might be possible to argue that incremental unemployment and plant or firm closures resulting from even small regulatory costs cannot be distinguished from changes that would probably be triggered by the underlying economic viability of these activities. This step also applies when a regulation imposes one burden on an entity, but reduces another on the same entity, so that the net effect is small. Although some equity impacts might be dismissed on this basis, others will probably require further analysis beyond this initial *de minimis* screen.

- **Identify which equity dimensions from Exhibit 9-5 are relevant if further analysis is required.** Negative impacts on small entities, low income populations, minority populations, and children are important to consider in all cases because of statutory and other mandates (see Section 9.3.2). For example, rules requiring additional safeguards against contamination of groundwater by landfills clearly benefit communities where landfills are sited. There is a long-standing concern among the environmental justice movement that locally unwanted facilities, including landfills, are sited disproportionately in poor and/or minority neighborhoods. Thus, for regulations affecting siting and management of landfills, wealth and race are equity dimensions of concern. Rules requiring additional safeguards are likely to have a positive impact on neighborhoods hosting landfills. This positive impact should be noted and possibly estimated. For other rules, it is likely that concern for other equity dimensions will naturally arise.

  In addition to those equity dimensions that must always be considered for distributional analysis, the other dimensions listed by Exhibit 9-5 should be considered as part of the effort to identify which are relevant. In attempting to decide for a particular case whether some of the less obvious dimensions matter, analysts should collect readily accessible information on the characteristics of affected entities and populations. Attention should be paid to who is expected to receive the benefits of the regulation as well as who will pay the costs. Negative net benefits or net costs are ultimately what should trigger concern. Financial, health, and other non-monetary benefits and costs should be included.

- **Prioritize relevant equity dimensions.** Assuming there is more than one relevant equity dimension, they should be prioritized according to which dimension seems to warrant greatest concern. The level of concern should be determined by how strongly analysts expect a regulation to affect a particular sub-population.

9.3.7.2 Step 2: Define Distributional Variables for the Equity Dimensions of Concern

The next step is to define distributional variables associated with the equity dimensions from Step 1. For example, if one were concerned about a regulation's potential impact on poor neighborhoods, then a classification system for "poor" versus "not-poor" neighborhoods should be developed. The established definitions reviewed in Section 9.3.2 above could be used or alternatives developed. Referring again to the landfill example where one of the relevant equity dimensions is race, analysts would need to establish a rule for defining what qualifies as an African-American neighborhood or a minority neighborhood. In this case, one could rely on the established definitions presented in Section 9.3.2.

9.3.7.3 Step 3: Measure Equity Consequences

The next step is to begin to measure specific economic effects across the distributional variables. In some cases, estimating the equity-related effects of a regulation will involve disaggregating existing costs and benefits and tabulating or otherwise accounting for their distribution across the distributional variables defined in Step 2. This process would subject the equity assessment to the same set of assumptions applied to the benefit-cost analysis.

In other cases, an equity assessment will examine other impacts, such as increases in unemployment, for identified sub-populations. The section above on EIA reviews these other impacts and outlines how to estimate them. Any assumptions, for example those concerning elasticities of demand, used in the EIA, should also be applied to the Equity Assessment unless there are specific reasons for why the assumptions are inappropriate for the identified sub-population(s).

A thorough equity assessment, when resources permit, might include a disaggregation of benefits and costs from the BCA as well as an examination of economic impacts for the identified sub-population(s).

9.3.8 Data for Conducting Equity Assessments

The discussion in the preceding sections suggests several types of data that would be useful for estimating the distribution of impacts of environmental policy options. This section presents some of the data sources for each category of data needed. This is not an exhaustive list of data sources, but is presented to provide initial guidance for this information.

9.3.8.1 Data on Businesses, Governments, and Not-For-Profit Organizations

Two specific Internet sites provide access to some of the most commonly needed data. The first is the SBA's Office of Advocacy website.[75] Data provided on this website include the number of firms, number of establishments, employment, annual payroll, and estimated receipts. The data are available by employment size categories. Data may be viewed and downloaded for the U.S., by state or by metropolitan statistical area. A second website that analysts may find useful is the home page of the U.S. Bureau of the Census, Office of Statistics.[76] Here a variety of relevant data may be accessed including information published in the *County Business Patterns* and other published data series on population characteristics, including income and age distributions.

9.3.8.2 Demographic Data

The U.S. Bureau of the Census collects household data and aggregates them in forms that may be useful for environmental justice matters. Data are available on population distributions by race and household income at the state, county, metropolitan statistical area, or census tract level. An additional, Census website allows one to view a map of any part of the U.S., at the desired scale, that shows data on population distributions by family income or a specified race (e.g., percent white or percent black).[77] In addition, income data collected by the Internal Revenue Service and

[75] The address for this site is http://www.sba.gov/ADVO/stats/ (accessed 8/28/2000, internal EPA document).

[76] The address for this site is http://www.census.gov (accessed 8/28/2000).

[77] The address for this site is http://www.census.gov/geo/www/tiger/index.html (accessed 8/28/2000).

made available in aggregated form on the Internet may be useful for some analyses.[78]

9.3.8.3 Other Potentially Useful Data Sources

There is a range of other sources that may provide useful data on other factors potentially relevant to equity analyses. For example, import and export data are available from the Bureau of the Census publication, *The U.S. Merchandise Trade: Exports, General Imports, and Imports for Consumption*. The U.S. Department of Commerce may also have data that would be useful for estimating changes in demand as a result of regulatory costs, or the turnover rate for capital equipment in various industrial sectors.

[78] The address of the website providing these data is http://trac.syr.edu/tracirs/. Note that a user ID and password are necessary to access the data. Registration is available at http://trac.syr.edu/register/registration.html (accessed 8/28/2000).

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Chapter 10: Using Economic Analyses in Decision Making

10.1 Introduction

This chapter provides some general rules for presenting analytical results to policy makers and others interested in environmental policy development. As emphasized several times in these *EA Guidelines,* especially by the flowchart and discussion of equity assessments in Chapter 9, economic analyses play an important role throughout the policy development process. From the initial, preliminary evaluation of potential options and their consequences through the preparation of the final economic analysis document, economic analysts participate in an interactive process with policy makers. The fundamental goal of this process is to refine the information available for making policy choices.

Hence, this guidance for presenting inputs, analyses, and results applies at all stages of this process, not only for the final document embodying the completed economic analysis. In particular, reporting ranges of uncertainty, critical assumptions, and key unquantified effects to decision makers as they weigh various options and alternatives is as critical as including these in a final economic analysis document at the conclusion of the policy development process.

This chapter first reviews some important features of economic analyses of environmental policies to include or describe in all presentations of analyses and results. Following this is some general guidance on what types of results are most useful to report and useful formats for presenting them. Some brief comments on the relationship between economic analyses and environmental policy making conclude the chapter. All of the recommendations in this chapter are, of course, subject to resource availability and statutory prohibitions.

10.2 Communicating Assumptions and Methods

All economic analyses of environmental policies must balance the goals of accuracy and completeness against the costs of data acquisition, detailed modeling, and valuation of consequences. Hence, the results of applied economic analyses inevitably contain uncertainties in particular areas, assumptions in the place of data that are not available, and effects that cannot be quantified or monetized. Analysts should highlight these limitations when presenting the inputs, modeling, and results of economic analyses.

Some general guidelines for communicating these considerations include the following:

- **Clarity and transparency:** Presentations of economic analyses should strive for maximum clarity and transparency of all aspects of the assessments. An analysis whose conclusions can withstand close scrutiny of all of its facets is more likely to provide policy makers with the information they need to develop the best environmental policies. In addition, if a rule is later challenged, the more clear, transparent and thorough analysis is, the easier it is to defend the agency's regulatory approach.
- **Delineation of data and assumptions:** Economic analyses should clearly describe all important data sources and references used, as well as key assumptions and their justifications. All of these inputs should be available to policy makers, and other researchers and policy analysts, to the extent that these data are not

confidential business information or some other form of private data.

- **Exposition of modeling techniques:** Although modeling frameworks for many economic analyses can be complex, it is important to convey at least the basic framework used for modeling a policy's consequences. The presentation should highlight the key elements or drivers that dominate the framework and its results.

- **Ranges for inputs and results:** At a minimum, uncertainties should be explored through the use of expected values supplemented by upper and lower bounds for important inputs, assumptions, and results. Sensitivity analyses using these ranges generally enhance the credibility of environmental policy assessments. If key elements for an economic analysis are extremely uncertain, these should be clearly indicated. Analysts should explore how resolving these uncertainties affects the conclusions of the analysis.

- **Monetizing a policy's effects:** To the extent feasible and warranted by their contribution to the results, as many of the effects of a policy as possible should be monetized. This enhances the value of the conclusions to policy makers weighing the many, often disparate consequences of different policy options and alternatives.

- **Highlighting non-monetized and unquantified effects:** Economic analyses should present and highlight non?monetized effects when these are important for policy decisions. Reasons why these consequences cannot be valued in monetary terms are important to communicate as well. Unquantified, but potentially significant, consequences of a policy also should be highlighted, especially when these could be important enough in magnitude to affect the broad conclusions of an economic analysis of different policy options and alternatives.

- **Presenting aggregate and disaggregated results:** Finally, the analytic framework should be organized to provide information on the separate economic consequences of important individual programs or component parts identified with the regulation. This can be particularly challenging when the underlying physical science and engineering information needed by the economist to prepare the economic analysis may not be amenable to a simple separation of the individual contributions of pollution control choices (e.g., installed emission control devices) to changes in risks from pollutants. Further, some economic values used in analyses represent a quantified aggregate value for a set of environmental goods (e.g., a consumer benefit measure for total improvements to surface water), and it is unknown how to divide the value among the individual attributes that comprise this reported value. Nevertheless, it is valuable to describe disaggregate information on the costs and benefits attributable to individual policies whenever possible, given the frequent necessity to package or link regulatory actions or evaluations together into a single analysis.

10.3 Presenting the Results

The results of economic analyses of environmental policies should generally be presented in three clusters:

- **Results from benefit-cost analysis:** Estimates of the net social benefits should be presented based on the benefits and costs for which dollar values can be assigned, and a discussion of non-monetizable or unquantifiable benefits and costs should be provided;

- **Results from economic impact analysis and equity assessment:** Results of the economic impacts analysis and equity assessments should be reported, including predicted effects on prices, profits, plant closures, employment, and other effects, and findings concerning the distribution of effects for particular groups of concern, such as small entities, governments, and disadvantaged and vulnerable populations.

- **Results from cost-effectiveness analysis:** This policy evaluation technique is used when many benefits are not easily monetized and when the statutes or other authorities dictate specific regulatory objectives. Results of these analyses should also be presented when these analyses are conducted.

The relative importance of these three clusters will depend upon the policy and statutory context of the decision. Generally, analyses leading to these findings normally should be conducted simultaneously and the results should be presented together as different ways to examine a policy's social consequences.[1]

10.3.1 Results from Benefit-Cost Analysis

The net social benefits of each major alternative is estimated by subtracting the present value of monetary social costs from the present value of monetary social benefits (as defined in Chapter 6). For this calculation the same baseline must be used in both the benefit and cost analyses. Plausible upper- and lower-bound estimates of net benefits should be provided, the "best" or most-likely estimate should be identified and the sensitivity of the net benefits estimate to variations in uncertain parameters should be examined.

Other considerations for presenting and summarizing the results from benefit-cost analyses include the following:

- **Discounting benefits and costs is the preferred method for summarizing benefits and costs that accrue over several years.** The conditions for discounting in benefit-cost analysis are outlined in Chapter 6. Alternatives to discounting include annualizing costs and benefits, comparing those figures, and accumulating costs and benefits through time to a future year. When both traditional discounting and these alternatives are not feasible or advisable, it is appropriate to display the streams of costs and benefits over time for policy makers to consider.

- **Present and evaluate non-monetized and non-quantified effects.** The net social benefit estimate should be carefully evaluated in light of all the effects that have been excluded because they cannot be valued in monetary terms. Thus, immediately following a net benefit calculation, there should be a presentation and evaluation of all benefits and costs that can only be quantified but not valued, as well as all benefits and costs that can be only qualitatively described.

- **Present the incremental benefits, costs, and net benefits of moving from one regulatory alternative to more stringent ones.** This presentation should be done both globally and by sub-population. This should include a discussion of incremental changes in quantified and qualitatively described benefits and costs. It is sometimes necessary to evaluate all combinations of options and alternatives when key sources of benefits and costs of a policy are affected by more than one option. In these cases, identifying the combination of alternatives with the highest net social benefits cannot rely only on the incremental benefits and costs of each individual option when added to other pre?existing options.

- **Discuss other potential costs and benefits that may be by-products of the proposed action.** These include transfers of the pollutant problem from one exposure medium or program office jurisdiction to another or possible exacerbation of exposures for specific groups (e.g., sensitive sub-populations, maximum exposure groups, or specific types of workers) not captured already in the economic impacts analysis and equity assessment.

10.3.2 Results from Economic Impacts Analysis and Equity Assessment

Economic impact analyses and equity assessments focus on distributional outcomes. Therefore, the presentation of these results should focus on disaggregating effects to show impacts separately for the groups and sectors of interest. If costs and/or benefits vary significantly among the sectors affected by the policy, then both costs and benefits should be shown separately for the different sectors. Presenting results in disaggregated form will provide important information to policy makers that may help them tailor the rule to improve its efficiency and equity outcomes.

[1] There are other, more limited types of economic analysis that can inform policy decisions. One example is health-health analysis (sometimes known as risk-risk analysis) that assesses the health risks introduced by diverting to regulation resources otherwise available for individual health care. Although limited, health-health analysis may be useful in contexts where benefit-cost analysis is infeasible. The method has been employed by several researchers (Viscusi, 1994; Keeney, 1997), but is not without criticism (Portney and Stavins, 1994).

The results of the economic impact analyses should also be reported for important sectors within the affected community—identifying specific segments of industries, regions of the country, or types of firms that may experience significant impacts or plant closures and losses in employment.

Reporting the results of equity assessments may include the distribution of benefits, costs, or both for specific subpopulations including those highlighted in the various mandates. These include minorities, low-income populations, small businesses, governments, and non-profits, and sensitive and vulnerable populations (including children). Where these mandates specify requirements that depend on the outcomes of the distributional analyses (such as the Regulatory Flexibility Act), the presentation of the results should conform to the criteria specified by the mandate.

10.3.3 Results from Cost-Effectiveness Analysis

When many benefits cannot easily be monetized, or when statutes or other authorities set forth a specific policy objective, economic analyses should present the results of a cost-effectiveness analysis. This will provide useful information to policy makers and it conforms to the general principle of minimizing the cost of achieving particular policy goals.

The cost-effectiveness of a policy option is calculated by dividing the annualized cost of the option by non-monetary benefit measures. Such natural units measures range from the amount of the reduction in pollution measured in physical terms, to the ultimate improvements in human health or the environment measured in terms of specific effects and damages avoided.

Cost-effectiveness analysis does not necessarily reveal what level of control is reasonable, nor can it be used to directly compare situations with different benefit streams. Moreover, other criteria, such as statutory requirements, enforcement problems, technological feasibility, or quantity and location of total emissions abated, may preclude selecting the least-cost solution in a regulatory decision. However, where not prohibited by statute, cost-effectiveness analysis can indicate which control measures or policies are inferior options.

10.4 Use of Economic Analyses in Policy Choices

The primary purpose of conducting economic analysis is to provide policy makers and others with detailed information on wide variety of consequences of environmental policies. One important element these analyses have traditionally provided to the policy-making process is estimates of social benefits and costs—the economic efficiency of a policy. Hence, the *EA Guidelines* reflect updated information regarding procedures for calculating benefits and costs, monetizing benefits estimates, and selecting particular inputs and assumptions.

Determining which regulatory options are best even on the restrictive terms of economic efficiency, however, often is made difficult by uncertainties in data and by the presence of benefits and costs that can be quantified but not monetized or that can only be qualitatively assessed. Thus, even if the criterion of economic efficiency were the sole guide to policy decisions, social benefit and cost estimates alone would not be sufficient to define the best policies.

A large number of social goals and statutory and judicial mandates motivate and shape environmental policy. For this and other reasons, the *EA Guidelines* contain information concerning procedures for conducting analyses of other consequences of environmental policies, such as economic impacts and equity effects. This is consistent with the fact that economic efficiency is not the sole criterion for developing good public policies.

Even the most comprehensive economic analyses are but part of a larger policy development process, one in which no individual analytical feature or empirical finding dominates. The role of economic analysis is to organize information and comprehensively assess the economic consequences of alternative actions—benefits, costs, economic impacts, and equity effects—and the tradeoffs among them. These results serve as important inputs for this broader policy-making process along with other analyses and considerations.

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Appendix A: An SAB Report on the EPA Guidelines for Preparing Economic Analysis

United States Environmental Protection Agency
Science Advisory Board (1400) Washington DC
EPA-SAB-EEAC-99-020
September 1999
www.epa.gov/sab

AN SAB REPORT ON THE EPA GUIDELINES FOR PREPARING ECONOMIC ANALYSES

A Review by the Environmental Economics Advisory Committee

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

September 30, 1999

OFFICE OF THE ADMINISTRATOR
SCIENCE ADVISORY BOARD

EPA-SAB-EEAC-99-020

The Honorable Carol Browner
Administrator
United States Environmental Protection Agency
401 M Street, SW
Washington, DC 20460

Subject: An SAB Report on the EPA Guidelines for Preparing Economic Analyses

Dear Ms. Browner:

This Report on the Environmental Protection Agency's (EPA's) revised *Guidelines for Preparing Economic Analyses* was developed by the Environmental Economics Advisory Committee (EEAC) of the Science Advisory Board (SAB) in response to a charge received from the Deputy Administrator on August 4, 1998 (attached). The review was carried out in a series of meetings with the Agency's Office of Policy, beginning in August 1998, and ending with a telephone conference on July 27, 1999.

As is described in detail in the full report, the Committee's general conclusion is that the Guidelines succeed in reflecting methods and practices that enjoy widespread acceptance in the environmental economics profession. Although some concerns remain about particular parts of the Guidelines, our overall assessment is that the Guidelines are excellent. It is our hope that the Guidelines demonstrate EPA's commitment to credible and consistent economic analyses in support of the policy process.

The best analytical tools of environmental economics are constantly changing, as experience with applications of existing tools and as new theoretical and empirical techniques appear in the scholarly literature. As a result, it is important that EPA carry out new reviews of the Guidelines every two to three years to reflect these developments in environmental economics. The Committee looks forward to working with EPA to strengthen this document in the years ahead.

The iterative process that the EEAC employed with EPA for this review represents a departure from the end-of-pipe assessments that are more typical of SAB practice. It was consistent, however, with the Mission Statement of the EEAC prepared by the Deputy Administrator, and was consistent with the SAB Executive Committee's previously expressed aims. Although this approach will not necessarily be appropriate for all SAB reviews, it may be a useful model in selected cases. Therefore, we briefly describe the procedure in the full report.

Finally, Dr. Albert McGartland and his staff in the Office of Economy and Environment should be commended for the professionalism they brought to this process. The excellence of the revised Guidelines is testimony to the dedication of the talented team of Agency economists and analysts who worked on this project. It was, as always, a pleasure for the EEAC to interact with Dr. McGartland and his staff. We anticipate that you and everyone involved will be proud of the quality of the new *Guidelines for Preparing Economic Analyses*, and we look forward to your questions and your response to our Report.

Sincerely,

/s/
Dr. Joan M. Daisey, Chair
Science Advisory Board

/s/
Dr. Robert N. Stavins
Environmental Economics Advisory Committee
Science Advisory Board

NOTICE

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U.S. Environmental Protection Agency
Science Advisory Board
Environmental Economics Advisory Committee
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1. EXECUTIVE SUMMARY AND CONCLUSIONS

The Environmental Economics Advisory Committee (EEAC) of the EPA Science Advisory Board (SAB) reviewed the Agency's draft *Guidelines for Preparing Economic Analyses* (EPA, 1999) during a series of meetings extending from August 1998 to July 1999, in response to a request received from EPA to perform a full and complete review. This is EPA's first major overhaul of these guidelines in more than a decade. Because the Guidelines are to be used by all parts of the Agency for carrying out regulatory analyses, it is important that they reflect "methods and practices that enjoy widespread acceptance in the environmental economics profession," as specified in the charge to the EEAC received from the EPA Deputy Administrator. The draft Guidelines have been revised and improved as a result of the interactions between the Committee and EPA staff in several public meetings over the past year. The EEAC's general conclusion is that the Guidelines reflect best methods and practices that enjoy widespread acceptance in the environmental economics profession, notwithstanding the several EEAC concerns that remain about particular parts of the Guidelines. The Committee hopes that the Guidelines demonstrate EPA's commitment to credible and consistent economic analysis in support of the policy process.

The fundamental ingredients of an economic analysis of a proposed or existing environmental policy or program are well covered in the Guidelines, and key conceptual, analytic, and empirical issues are highlighted. The Guidelines acknowledge the practical limitations facing EPA analysts in terms of time, resources, and expertise, and hence provide flexibility to analysts. The Guidelines are just that — guidelines for analysis, not a rigid (or simplistic) cook book.

Economics, like any scholarly discipline, is constantly changing. Environmental economics, a relatively young branch of the discipline, has experienced particularly rapid growth. New areas of the literature continue to emerge, and existing areas change and expand. Hence, despite the Committee's generally positive assessment of the revised Guidelines, we urge EPA to carry out new reviews every two to three years. The time investments that will be required for such periodic reviews — both on the part of EPA staff and on the part of the EEAC — will be much less than was required for this first revision in ten years.

The interactive process that the EEAC employed with EPA for this review was something of a departure from the end-of-pipe assessments that are more typical of SAB practice. It was consistent, however, with the Mission Statement of the EEAC prepared by the Deputy Administrator, and, moreover, was consistent with the SAB Executive Committee's previously expressed interest in early involvement with the Agency on important issues. Although this approach will not necessarily be appropriate for all SAB reviews, it may be a useful model in selected cases. Therefore, we briefly describe the procedure here.

During the past twelve months, the EEAC held three one-day meetings in Washington, D.C., devoted primarily to our review of EPA's revised Guidelines for Preparing Economic

Analyses, plus a fourth meeting via teleconference. Each of the meetings was attended by Dr. Albert McGartland, Director of the Office of Economy and Environment, and many members of his staff. Intensive and extensive substantive discussions at these meetings provided an effective forum in which EEAC members and the Committee as a whole could pose questions, describe concerns, and express views, and for Agency representatives to explain the thinking behind their drafts. Each of the first three meetings led to a subsequently revised draft, which in every case addressed the general and specific concerns expressed by the EEAC at the previous meeting, and thereby represented a significantly improved draft document. Dr. McGartland and his staff should be commended for the openness, professionalism, and lack of defensiveness which they brought to this process.

2. INTRODUCTION

The Environmental Economics Advisory Committee was requested to perform a full and complete review of the *Guidelines for Preparing Economic Analyses* (EAGs or the Guidelines). EPA asked for comments on all aspects of the guidance document and written documentation, when applicable, on recommendations from the Committee for alternative methodologies, assumptions and data sources to improve the presentation of issues addressed in the guidance document.

The stated intent of the Guidelines is to:

a) Represent EPA policy on preparing economic analyses under a variety of authorizing statutes and administrative requirements, each of which can influence the conduct and use of analyses by different EPA offices;

b) Demonstrate EPA's commitment to credible/consistent economic analyses in support of the policy making process;

c) Emphasize the need for analytic efforts commensurate with the value of that information in the EPA policy making process;

d) Reflect mainstream economic science/methods that are well demonstrated and relatively straightforward to apply to particular environmental issues;

e) Cover a number of principles and practices that virtually all economic analyses should follow and clarify, for a number of identified analytic issues, the process that analysts are to follow as they organize and conduct the analysis;

f) Account for some of the practical limitations on time and resources that EPA analysts must contend with when preparing economic analyses; and

g) Provide some flexibility to analysts to permit them to “customize” analyses to conform to administrative and legal procedures.

The document embodying the Guidelines is not intended to be a text on the theory and methods of applying economic analysis to EPA regulations and policies nor do they preclude new or path breaking forms of analysis. EPA intends to regularly and frequently review and revise parts of the Guidelines to reflect and report on significant changes in the literature used to support EPA analyses, as well as changes in administrative and legal requirements that affect the conduct of economics at the EPA.

3. SPECIFIC RESPONSES TO CHARGE QUESTIONS

The Committee’s specific comments on the Guidelines are organized into nine sections, the first eight of which address, respectively, questions posed in EPA’s charge to the Committee. The last section includes the Committee’s advice on the topic of altruistic benefits.

3.1 Discounting

Charge Question 1. Does the published economic theory and empirical literature support the statements in the guidance document on the treatment of discounting benefits and costs in the following circumstances: a) private and public costs for use in an economic impact analysis, b) social benefits and costs in an intragenerational context; c) social benefits and costs in an intergenerational context; and d) social benefit and cost information that is reported in nonmonetary terms?

The guidance document is consistent with published theoretical and empirical analysis on the use of discounting for (a) private and public costs in an economic impact analysis, and for (d) social benefit and cost information expressed in non-monetary terms. The current literature also supports the discussion of issue (b), discounting social benefits and costs in an intragenerational context. In this context, the document should encourage the use of a discount rate in the middle of its recommended range, in addition to the values currently mentioned, to reflect common practice.

The proper application of discounting in an intergenerational context (issue (c)) remains controversial in the published literature. The guidance document lays out the positions in this debate clearly. Reflecting the disagreement within the economics profession, there is diversity of opinion within the EEAC on this issue. Some members believe that the guidance document is more critical than the published literature of the view that intergenerational discounting should not differ from other discounting, while other members support the document's current emphasis. The final quantitative recommendations for discount rate values in the intergenerational context are broad enough to be uncontroversial.

3.2 Quantifying and Valuing Human Fatalities

Charge Question 2. Does the published economic theory and empirical literature support the statements in the guidance document on quantifying and valuing the social benefits of reducing fatal human health risks?

The guidance document recommends that the mean value of a statistical life (VSL) based on 26 published studies be used as the default value in Agency analyses. It urges that a *qualitative* discussion of the appropriateness of this estimate to the population and risks analyzed accompany the use of the central tendency estimate.

The EEAC encourages the use of value of a statistical life (VSL) estimates in benefit valuation and in providing guidance for policy. Moreover, it believes that the general magnitude of the benefit value suggested in the Guidelines is in a reasonable range for broad population groups. However, these estimates could be refined, particularly when certain segments of the population are affected, such as children or persons over the age of 65.

One refinement that EPA could undertake now is to narrow the set of VSL studies to the most reliable estimates for the U.S. population, rather than taking the mean value from a set of studies of varying quality and with different statistical controls and with populations from various countries.

Hedonic studies based on market tradeoffs give values reflected in actual market decisions. Such studies have focused primarily on the labor market, but some have considered implicit values of life reflected in car safety and in housing price responses to hazardous wastes and pollution. Contingent valuation studies can also prove useful with respect to variations due to age, health status, and other factors not readily estimated using market data.

Care should be taken to avoid conveying the impression that $5.8 million (in 1997 dollars) is always and everywhere the correct figure to use for the value of a statistical life. Footnote 29 in Section 7.6.1.2 attaches some important caveats to the $5.8 million figure cited in the document. These caveats should be placed more prominently, so that casual readers do not come away with the notion that $5.8 million is “the” value of a statistical life. It is, of course, simply the central tendency of a number of estimates of the VSL for some rather narrowly defined subpopulations. The individual study values from which this number is derived range from $0.7 million to $16.3 million, indicating considerable heterogeneity across different contexts.

In some cases, it may be desirable to use a VSL estimate specific to particular populations. The most prominent possible variation is with respect to age, but characteristics such as gender and income could also be influential. How and whether these differences should affect policy benefit assessment is more controversial. However, as a first step, EPA should show the age distribution of the lives saved, or the quantity of life at risk. In addition, when policies do not affect the entire population equally, a sensitivity analysis can show both the cost per life saved and the cost per discounted life year. Policymakers can then be better able to assess the efficacy of the policy.

Clearly, any re-evaluation of the literature will take time. In the interim, if the mean VSL cited in the Guidelines is to be used, the limitations of the estimate should be described. In cases where the estimate is to be applied to populations whose age differs significantly from the average age of the populations in the 26 studies, a *quantitative* sensitivity analysis should be performed, such as that used in *The Benefits and Costs of the Clean Air Act, 1970-1990* (EPA, 1997).

3.3 Certainty Equivalents

Charge Question 3. Does the published economic theory and empirical literature support the statements in the guidance document on the treatment of certainty equivalents in the assessment of social benefits and costs of environmental policies?

The discussion of principles for uncertainty analysis in the Guidelines highlights the important distinction between the analyst's uncertainty and individuals' uncertainty about future outcomes. The latter relates to assessing the effects of environmental changes on individuals' welfare under uncertainty. In this regard, the Guidelines are consistent with mainstream economic theory and the empirical literature in that they recognize the importance of taking account of individual attitudes towards risk and suggest certainty equivalents as one way of incorporating risk aversion into the assessment of social benefits and costs of environmental policies. However, the Guidelines recognize that information on risk attitudes may be difficult to obtain. The Guidelines also recognize that experts' and lay individuals' risk perceptions may differ. Because the latter affect individuals' behavior, it is important for the analyst to consider both types of risk assessments. The Guidelines are also consistent with current literature in recognizing the important role information plays in welfare evaluation in an uncertain world.

3.4 Valuation Approaches for Human Morbidity and Improved Ecological Conditions

Charge Question 4. Does the published economic theory and empirical literature support the statements in the guidance document on the merits and limitations of different valuation approaches to the measurement of social benefits from reductions in human morbidity risks and improvements in ecological conditions attributable to environmental policies?

Overall, Chapter 7 does a very good job of explaining the state-of-the-art in the measurement of environmental benefits. This literature continues to evolve, and so frequent updates are likely to be necessary for this chapter. All currently relevant methods appear to be represented and sufficient caveats have generally been offered. Four concerns, however, should be highlighted.

First, care should be taken to avoid creating the impression that the benefits or costs associated with a proposed regulation are being misrepresented. Socially efficient policy-making is not well served by exaggeration or understatement of the benefits or costs of alternative policy choices. When benefits are uncertain, expected values should be emphasized. However, an assessment should be made concerning the sensitivity of the policy conclusions to the full range of possible benefits estimates.

Second, the claim that averting expenditures can reliably be used as a lower bound on environmental benefits is too strong. The draft overstates the idea that averting behavior is a generic lower bound on ex ante economic value for morbidity and mortality. The theoretical literature reveals that averting behavior need not be a lower bound on value when both private and collective risk reduction strategies are considered. Private actions to reduce risk mixed with collective actions yield ambiguous results. Further discussion is available in Shogren and Crocker (1999).

Third, the treatment of altruistic benefits should be clearer. Circumstances wherein altruistic benefits should, and should not, be included as a separate component of total social benefits should be highlighted. It is important to avoid double-counting of benefits and costs in assessing proposed policies. This issue is taken up, below, in Section 3.9.

3.5 Economic Impact and Net Social Benefits

Charge Question 5. Does the published economic theory and empirical literature support the statements in the guidance document on the relationships and distinctions between the measurement of economic impacts and net social benefits?

The guidance document makes it clear that in order to make informed policy judgements it is important to study a variety of consequences of environmental policies, including impacts on particular industries, regions, and demographic groups (as well as other impacts whose analyses are mandated by statute) to complement conventional benefit-cost analysis. The relationship between conventional benefit-cost analysis and the analysis of the broad and diverse distributional impacts of environmental policies are discussed clearly and the relationships between these complementary policy analyses is consistent with the published economics literature.

3.6 Computable General Equilibrium Models

Charge Question 6. Does the guidance document contain an objective and reasonable presentation on the published economic theory, empirical literature, and analytic tools associated with computable general equilibrium (CGE) models, and description of their relevance for economic analyses performed by the EPA?

The use of general equilibrium analysis, both as a conceptual and a numerical tool, is gaining expanded use in economics. The Guidelines provide a useful discussion of the current uses and limitations of computable general equilibrium (CGE) models in Chapter 8. This is a rapidly developing area in economics, and so for the Guidelines to remain relevant, the Agency will need to commit to ongoing review of new tools and applications that broaden the applicability of CGE models, and that provide new intellectual insights that can guide benefit-cost analysis.

One area where new insights are proliferating is the interaction of environmental regulations with pre-existing economic distortions (that is, the deadweight loss due to existing taxes). Of particular relevance in this regard is the role of pre-existing taxes. Recent literature in economics indicates that the *costs and benefits* of regulations can be substantially different than indicated by partial equilibrium analysis. In addition, the relative cost-effectiveness of different policy instruments (technology standards, tradable permits, etc.) can be affected by these interactions.

The Guidelines address the issue of interactions of regulations with pre-existing economic distortions in a paragraph near the end of Chapter 5, under the label "Emerging Cross-Cutting Issues." This is appropriate because the issue is both rapidly emerging and broadly cross-cutting. But the issue is not mentioned in Chapter 8 ("Analyzing Social Costs"), where general equilibrium analysis is discussed in detail. The exclusion of the issue from Chapter 8 is unfortunate: (a) because of the potential magnitude associated with "interactions" and (b) because general equilibrium tools provide the method for considering these "interactions." Hence, one of the most compelling reasons to use CGE models is to develop an understanding of, and to estimate the magnitude of, this potential influence. While in most regards, the discussion of CGE models is objective and reasonable, the failure to integrate a discussion of tax interactions undermines the presentation.

3.7 Economic Impacts to the Private Sector, Public Sector, and Households

Charge Question 7. Does the guidance document contain an objective and reasonable presentation on the measurement of economic impacts, including approaches suitable to estimate impacts of environmental regulations on the private sector, public sector and households? This includes, for example, the measurement of changes in market prices, profits, facility closure and bankruptcy rates, employment, market structure, innovation and economic growth, regional economies, and foreign trade.

The guidance document provides an objective and reasonable presentation of these topics.

3.8 Equity

Charge Question 8. Does the guidance document contain a reasonable presentation and set of recommendations on the selection of economic variables and data sources used to measure the equity dimensions identified as potentially relevant to environmental policy analysis?

Conventional, primary dictionary definitions of equity refer to concepts such as fairness, impartiality, and justice. Thus, equity is typically treated as a normative concept in everyday parlance. The Guidelines, in keeping with mainstream practice in economics, however, seek merely to supply statistical measures of the distribution of costs and benefits, leaving it to citizens to judge whether the described distributions are fair or equitable. In this way, the document provides positive information that is relevant to making normative judgments, but offers no explicit discussion of norms. At the same time, however, since market prices and willingness-to-pay criteria are employed in the Guidelines, and because they are based on the existing distributions of income and wealth, those existing distributions are implicitly accorded normative status. Some may object to taking the existing distribution as the norm for equity assessments, especially as that distribution has become more concentrated in recent decades. This issue is noted, but in accord with mainstream practice, is not considered further in the document.

The Guidelines contain a reasonable presentation and set of recommendations on the selection of economic variables and data sources that can be used to measure the distributional consequences of environmental policies, both on the benefit side and the cost side.

3.9 Altruism

The Guidelines would benefit from a discussion of when it is appropriate to include altruistic benefits in a benefit-cost analysis. Economic theory is quite clear on this point (Jones-Lee, 1991). If I care about my neighbor *and* respect his preferences, and if my neighbor would have to pay for the program or project being analyzed, then altruistic benefits should not be counted in a benefit-cost analysis. The intuition behind this result is that, if I respect my neighbor's preferences, although I value the benefits he will receive from the project, I also care about the costs it will impose on him. It is, therefore, inappropriate to add the value I attach to his benefits without considering the cost implications of doing this. Comparing individual benefits and costs in this case is the appropriate decision rule.

Altruistic benefits may be counted either when my altruism toward my neighbor is paternalistic, or when I will in fact bear the costs of the project but he will not. In the first case (paternalistic altruism), I care about the benefits my neighbor will enjoy, e.g., from a health or safety project, but not about the costs the project will impose on him. An example of the second case would be a project whose costs are borne entirely by the current generation; i.e., the project imposes no costs on future generations. In this case, altruism toward future generations by the current generation could legitimately be counted as a benefit.

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APPENDIX A

CHARGE TO THE COMMITTEE

August 4, 1998

Dr. Robert Stavins
Professor of Public Policy and Faculty Chair
John F. Kennedy School of Government
Harvard University
79 John F. Kennedy Street, Room L-313
Cambridge, Massachusetts 02138

Dear Dr. Stavins:

The Science Advisory Board, Environmental Economics Advisory Committee (EEAC or the Committee) is requested to perform an advisory review of a revised guidance document prepared for the Environmental Protection Agency (EPA) on the conduct of economic analysis. The document, titled "Guidelines for Preparing Economic Analyses," is the product of a deliberative Agency-wide process initiated at my direction and managed by the EPA's Regulatory Policy Council. The document is designed to represent Agency policy on the preparation of economic analysis called for under applicable legislative and administrative requirements, including, but not limited to Executive Order 12866 on regulatory planning and review. The revisions to the guidance document should embody sound economic thinking so that its application will continue to demonstrate the EPA's commitment to make credible and consistent economic analytic decisions in support of the regulatory and policy making process. The Agency is seeking external peer review of the guidance documents because of the pervasive influence of the documents on the conduct of agency-wide economic analysis.

Background

The decision to prepare a revised document is based on a number of events and factors. The current EPA operating guidance on performing economic analysis was written over the period 1983-1986. Since that time, there have been numerous advances in the economic literature. Because the guidance document is primarily intended to serve as a source for technical information on the conduct of economic analysis, it is important that the document reflect the most recent economics literature.

The original EPA guidance document was also written to support the administrative process for using economic information when developing regulations set forth in Executive Order (E.O.) 12291 on Regulatory Planning and Review (released in 1981). The Office of Management and Budget (OMB) issued its own federal guidelines for performing economic analysis following the release of E.O. 12291. The EPA elected to issue its own guidance document in an effort to elaborate on the materials described in the OMB guidance, and provide additional source material to assist in the application of the OMB analytic principles to analyses prepared by the EPA. The issuance of an updated Order on the federal regulatory development process (E.O. 12866 released in 1993) led OMB to revise its federal guidelines for performing economic analysis in early 1996. The new OMB guidance drew heavily on the previous document, but developed additional details on several aspects of conducting economic analysis that reflected advances in the economic literature, and added information on several administrative measures and policy objectives receiving additional emphasis included in E.O. 12866. The new EPA economic guidance document seeks to accomplish the same objective, but in a manner that meets the distinctive needs of EPA staff working on economic analyses.

Other administrative and legislative requirements were issued since the mid-1980s that now affect the development and conduct of economic analysis at the EPA. Most are not directed exclusively at EPA regulatory activities, but their addition has led to some modifications to the preparation of economic analyses by the EPA. Some examples include legislation to consider unfunded mandates on non-federal governments, and the assessment of economic impacts on small entities. The revised EPA economic guidance document seeks to update and make reference to existing and anticipated guidance on these Congressional mandates and executive orders.

One major goal of the new guidance document is to provide more assistance to EPA analysts in the adoption of a consistent set of procedures used to formulate its economic analyses. The responsibility for preparing economic analyses at the EPA rests in many different offices in the Agency. As a consequence of differences in the authorizing statutes they operate under, the conduct and use of economic analysis can vary across documents prepared by these offices. Despite these differences, there are a number of guiding principles and practices that the EPA proposes to follow to aid in the consistent development of economic information. The new economic guidance document has been written to make clear, for a number of identified analytic issues, the process that EPA analysts are to follow as they organize and perform their economic analyses. One of the objectives in revising the guidance document was to adopt a process whereby the Agency's economic analytic staff participated as a group in the review and revision of the document. Because these offices have greater authority and responsibility for the content and quality of their economic analysis, the process provided a productive forum for raising common and critical issues that arise in the conduct of economic analysis.

The current guidance document's publication date of 1983 belies the fact that work has been undertaken by EPA since that time to support advances in the development and use of economic tools and information in its economic analyses. The Agency draws upon the results of new research and participates in professional workshops (e.g., events supported by the Association of Environmental and Resource Economists) to be current with the state of economic knowledge. EPA also uses materials produced by other government agencies (e.g., General

Accounting Office reviews, reports by the Presidential/Congressional Commission on Risk Assessment/Risk Management), incorporating new information and thinking into its economic analyses. Recognizing that the previous process resulted in development of a “static” EPA economic guidance document, this effort is viewed as the first of a series of more regular and frequent actions to continually review and revise component parts of the documents. The development and release of materials will follow a schedule that reflects and reports on significant changes in the literature used to support EPA analyses, as well as changes in administrative and legal requirements that affect the conduct of economics at the EPA.

The materials to be submitted to the SAB-EEAC for an advisory review at this time include a complete draft of the revised guidance document. The document consists of a main document and five separate appendices. Each appendix provides greater detail on subjects treated in the main document. The appendices are organized into major component parts of economic analyses produced by the EPA, or treat an analytic topic that merits significant attention. As of this revision, the guidance document contains appendices on the analysis of economic benefits, social costs, economic impacts, equity effects, and discounting future benefits and costs.

Charge to the Committee:

The charge to the Committee is to undertake an advisory review of the draft materials and provide advice to the Agency pursuant to a series of questions concerning the preparation of economic analyses by the EPA. The EPA guidance directly refers to methods and practices that enjoy widespread acceptance in the environmental economics profession. The guidance document does not intend to preclude new or path breaking forms of analysis, but to provide EPA analysts with a reasonably concise and thorough treatment of mainstream thinking on important technical issues that arise in the conduct of economic analysis. The guidance accounts for some of the practical limitations on time and resources that EPA analysts must contend with when preparing economic analyses. It also is shaped by administrative and statutory requirements that contain direct references to the development of economic information in the formulation of regulations (e.g., evaluations of economic achievability). As a result, the guidance is not written to resemble a text on the theory and methods of applying economic analysis to EPA regulations and policies. Some of the language in the guidance was chosen for the express purpose of providing some flexibility to analysts that should enable them to “customize” the analysis to be as complex and complete as is necessary to conform to administrative and legal procedures. The document also emphasizes the need for the EPA analyst to ensure that their analytic efforts are commensurate with the value the information will provide to the regulatory and policy making process at the EPA. The document covers a number of principles and practices that virtually all economic analyses should follow, and it is these items to which the Committee is asked to devote the greatest attention in its review.

In general, we believe the Guidance should reflect mainstream economic science and methods that are well demonstrated and relatively straightforward to apply to particular environmental issues. Ideally, these methods should be general enough that EPA program analysts can use them consistently across all of EPA's programs. Thus, while EPA recognizes that this document needs to provide pragmatic guidance, we have also attempted to reflect the state of the economic science. In some cases, our goal of making this useable has meant that we had to shorten or simplify the document. Your views about whether there are any important omissions or oversimplifications are critical.

The review questions to the Committee are as follows:

1. Do the published economic theory and empirical literature support the statements in the guidance document on the treatment of discounting benefits and costs in the following circumstances:

1a. Discounting private and public costs for use in an economic impact analysis?

1b. Discounting social benefits and costs in an intragenerational context?

1c. Discounting social benefits and costs in an intergenerational context?

1d. Discounting social benefit and cost information that is reported in nonmonetary terms?

2. Do the published economic theory and empirical literature support the statements in the guidance document on quantifying and valuing the social benefits of reducing fatal human health risks?

3. Do the published economic theory and empirical literature support the statements in the guidance document on the treatment of certainty equivalents in the assessment of social benefits and costs of environmental policies?

4. Do the published economic theory and empirical literature support the statements in the guidance document on the merits and limitations of different valuation approaches to the measurement of social benefits from reductions in human morbidity risks and improvements in ecological conditions attributable to environmental policies?

5. Do the published economic theory and empirical literature support the statements in the guidance document on the relationships and distinctions between the measurement of economic impacts and net social benefits?

6. Does the guidance document contain an objective and reasonable presentation on the published economic theory, empirical literature, and analytic tools associated with computable general equilibrium (CGE) models, and description of their relevance for economic analyses performed by the EPA?

7. Does the guidance document contain an objective and reasonable presentation on the measurement of economic impacts, including approaches suitable to estimate impacts of environmental regulations on the private sector, public sector and households? This includes, for example, the measurement of changes in market prices, profits, facility closure and bankruptcy rates, employment, market structure, innovation and economic growth, regional economies, and foreign trade.

8. Does the guidance document contain a reasonable presentation and set of recommendations on the selection of economic variables and data sources used to measure the equity dimensions identified as potentially relevant to environmental policy analysis?

The EPA requests that the Committee provide written review and documentation, when applicable, to support recommended changes to the guidance document. Our intention is that the Committee conduct a full and complete review. Although the specific questions identified above are those EPA believes are the most appropriate for the Committee to consider, EPA seeks comments on all aspects of the guidance document. The EPA also seeks recommendations from the Committee on alternative methodologies, assumptions and data sources that will improve the presentation of economic issues addressed in the guidance document. We would like the Committee to conclude its review by the end of October.

Review materials

The first attachment to this memorandum is to both the Designated Federal Official and Chairman to the Environmental Economics Advisory Committee. The memorandum lists the publicly available documents supporting the "Guidelines for Preparing Economic Analyses." This memorandum contains a list of the documents which are to be submitted to the Committee to assist in their review of the guidance document. The other attachments are the documents.

Please direct any inquiries regarding the review materials to me at 202-260-3354, or by e-mail at *mcgartland.al@epa.gov*. Thank you for your assistance.

Sincerely,
/ S /
Fred Hansen,
Deputy Administrator

ABSTRACT

The Environmental Economics Advisory Committee (EEAC) of the EPA Science Advisory Board (SAB) reviewed the Agency's draft *Guidelines for Preparing Economic Analyses* during a series of meetings extending from August 1998 to July 1999, in response to a request received from EPA to perform a full and complete review. The draft Guidelines have been revised and greatly improved as a result of the interactions between the EEAC and EPA staff during the public meetings over the past year. The EEAC's general conclusion is that the Guidelines now succeed in reflecting methods and practices that enjoy widespread acceptance in the environmental economics profession, notwithstanding the concerns that remain with several particular parts of the Guidelines.

Keywords: benefit-cost analysis; economic efficiency; cost effectiveness; regulatory impact analysis

Appendix B: EPA's Response to SAB Review

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

December 21, 2000

OFFICE OF
THE ADMINISTRATOR

Dr. Robert N. Stavins
Chairman, Environmental Economics Advisory Committee
Science Advisory Board (A-101)
Environmental Protection Agency
Ariel Rios Building
1200 Pennsylvania Avenue, N.W.
Washington, D.C. 20460

Dear Dr. Stavins:

Thank you for your letter to Administrator Browner and the report *An SAB Report on the EPA Guidelines for Preparing Economic Analyses: A Review by the Environmental Economics Advisory Committee* (EPA-SAB-EEAC-99-20, September 30, 1999). Before I respond to the specifics of your review, allow me to thank the Board for the sustained and intensive level of effort devoted by the SAB's Environmental Economic Advisory Committee (EEAC) to the development and review of the Agency's economic analysis guidelines. By incorporating the "best mainstream" economics into our guidelines and analyses, the overall credibility of our work will be sustained, the Agency's decisions will be better informed, and public discourse on environmental policy will be enhanced. The SAB report contains a thorough review of our *Guidelines for Preparing Economic Analyses* (or *EA Guidelines*), and provides many useful recommendations that the Agency has taken under consideration in revising the document. Other suggestions in the review help illuminate larger issues that the Agency continues to explore and consider.

Given the importance of economic assessments to the Agency, it is imperative that they are conducted as consistently and accurately as possible. The best way to ensure that analyses meet these goals is to provide Agency analysts with a clear set of guidelines and recommendations that have a solid foundation in the current state of economic science. As your letter notes, evolution of analytical tools and practices requires that such guidelines be reviewed every few years. The Agency is committed to undertaking regular reviews of the materials contained in the *EA Guidelines*, and looks forward to receiving further recommendations from the Science Advisory Board's Environmental Economics Advisory Committee (EEAC) that will strengthen and improve the *EA Guidelines* in the years ahead.

As you also note in your letter and report, the process employed by the EEAC and the Agency in the review of the *EA Guidelines* is unique. The Agency agrees that this departure from the standard practice was of great benefit in preparing the document. Not only did early involvement by the EEAC ultimately result in a much stronger document, but the open exchange between EEAC members, the Committee as a whole, and Agency personnel served to clarify a number of issues that may have been difficult to address otherwise. The Agency is hopeful that a similar process may prove useful in other contexts.

The *EA Guidelines* serve as a set of principles, consistent with current economic thinking, for analysts to draw upon in performing economic analyses. Section Three of the SAB report provides specific recommendations for the application of these principles. The Agency's responses to the main recommendations included in the SAB report, including references to sections in the *EA Guidelines* where resulting changes or additions to the document can be found, are enclosed.

Finally, I would like to again thank the Board for its thoughtful comments provided in the report, and the extensive feedback offered by the membership of the committee during the public meetings on the *EA Guidelines*. The Agency sincerely appreciates the time and effort of the SAB as a whole and of the individual members of the EEAC.

Sincerely,

/s/
W. Michael McCabe
Deputy Administrator

Enclosure

cc: Assistant Administrators
Associate Administrators
Regional Administrators
Regulatory Policy Council
Science Advisory Board

bcc: Economic Consistency Workgroup members

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

December 21, 2000

OFFICE OF
THE ADMINISTRATOR

Dr. Morton Lippmann
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Washington, D.C. 20460

Dear Dr. Lippmann:

Thank you for your letter to Administrator Browner and the report *An SAB Report on the EPA Guidelines for Preparing Economic Analyses: A Review by the Environmental Economics Advisory Committee* (EPA-SAB-EEAC-99-20, September 30, 1999). Before I respond to the specifics of your review, allow me to thank the Board for the sustained and intensive level of effort devoted by the SAB's Environmental Economic Advisory Committee (EEAC) to the development and review of the Agency's economic analysis guidelines. By incorporating the "best mainstream" economics into our guidelines and analyses, the overall credibility of our work will be sustained, the Agency's decisions will be better informed, and public discourse on environmental policy will be enhanced. The SAB report contains a thorough review of our *Guidelines for Preparing Economic Analyses* (or *EA Guidelines*), and provides many useful recommendations that the Agency has taken under consideration in revising the document. Other suggestions in the review help illuminate larger issues that the Agency continues to explore and consider.

Given the importance of economic assessments to the Agency, it is imperative that they are conducted as consistently and accurately as possible. The best way to ensure that analyses meet these goals is to provide Agency analysts with a clear set of guidelines and recommendations that have a solid foundation in the current state of economic science. As your letter notes, evolution of analytical tools and practices requires that such guidelines be reviewed every few years. The Agency is committed to undertaking regular reviews of the materials contained in the *EA Guidelines*, and looks forward to receiving further recommendations from the Science Advisory Board's Environmental Economics Advisory Committee (EEAC) that will strengthen and improve the *EA Guidelines* in the years ahead.

As you also note in your letter and report, the process employed by the EEAC and the Agency in the review of the *EA Guidelines* is unique. The Agency agrees that this departure from the standard practice was of great benefit in preparing the document. Not only did early involvement by the EEAC ultimately result in a much stronger document, but the open exchange between EEAC members, the Committee as a whole, and Agency personnel served to clarify a number of issues that may have been difficult to address otherwise. The Agency is hopeful that a similar process may prove useful in other contexts.

The *EA Guidelines* serve as a set of principles, consistent with current economic thinking, for analysts to draw upon in performing economic analyses. Section Three of the SAB report provides specific recommendations for the application of these principles. The Agency's responses to the main recommendations included in the SAB report, including references to sections in the *EA Guidelines* where resulting changes or additions to the document can be found, is enclosed.

Finally, I would like to again thank the Board for its thoughtful comments provided in the report, and the extensive feedback offered by the membership of the committee during the public meetings on the *EA Guidelines*. The Agency sincerely appreciates the time and effort of the SAB as a whole and of the individual members of the EEAC.

Sincerely,

/s/
W. Michael McCabe
Deputy Administrator

Enclosure

cc: Assistant Administrators
Associate Administrators
Regional Administrators
Regulatory Policy Council
Science Advisory Board

bcc: Economic Consistency Workgroup members

Enclosure to the Deputy Administrator's Response to
SAB Report on the EPA Guidelines for Preparing Economic Analyses: A Review by the Environmental Economics Committee

On the subject of discounting, the Agency appreciates the recommendation that, to reflect common practice, the *EA Guidelines* should encourage the use of a discount rate between the range specified in the review draft of the document. Chapter Six, Section 6.3.1 in the final *EA Guidelines* includes a recommendation for analysts to employ such a rate in addition to the other rates specified, if it is believed the additional information would be useful to decision makers. One reason analysts may include such a rate is because its use is routinely practiced for the specific type of analysis being performed. An example would be the recent analyses prepared by the Agency on the benefits and costs of the Clean Air Act and its 1990 Amendments. Other economic analyses have found it sufficient for decision making purposes to rely on presentations of economic information using the range alone. The Agency views both presentations to be supported by the *EA Guidelines* and in accord with the position taken on discounting by the SAB in the report.

Estimating the value of reductions in fatality risks remains one of the more difficult aspects of benefit-cost analysis, and the Agency appreciates the attention given this issue in the SAB report. Since receiving the SAB report on the draft *EA Guidelines*, the Agency has received additional SAB recommendations on the subject of valuing reductions in fatal cancer risks (SAB report EPA-SAB-EEAC-00-013, dated July 27, 2000). As noted in Chapter Seven, Section 7.6.1 of the final document, the Agency will release a supplement to the *EA Guidelines* on this topic once it has more thoroughly considered the July 27 SAB report on valuing fatal cancer risks.

The Agency agrees with the SAB that the limitations associated with the concepts and empirical studies available to use in quantifying a value of a statistical life (VSL) should be described in detail whenever VSL estimates are introduced in a benefit-cost analysis. The final *EA Guidelines* specifically require analysts to carefully present these limitations in their analyses. The document also advises analysts to fully characterize the nature of the risk and the populations affected as they relate to the choice of available literature used in a benefits analysis, as well as the presentation of uncertainties associated with such estimates. Some important considerations noted by the SAB report have resulted in an expanded discussion in the document, including recognition of how demographic characteristics (e.g., age) of the population at risk compare with those same characteristics of the population studied in the underlying economic literature used to derive benefit values. The final *EA Guidelines* continues to make reference to the numeric VSL range found to be reasonable by the SAB, and the Agency agrees with the SAB on the merits of improving and refining measures of this benefit category through reasoned use of the existing relevant literature. Analysts are further encouraged to consider quantitative sensitivity analyses as data allow, recognizing that the need for such analyses and the ability to perform them may vary with each benefit transfer exercise. The SAB report also notes that measures of cost-effectiveness may be employed as a sensitivity analysis, which is now included in the final document.

The SAB report raises several issues related to valuation for morbidity and ecological effects. The Agency agrees with the SAB that the benefits of a policy should be neither understated nor exaggerated, and has revised the *EA Guidelines* to inform analysts of these hazards and to encourage explicit consideration of uncertainty. Benefits will always be uncertain to some extent and the Agency agrees that expected values should be emphasized, as noted in Chapter Five, Section 5.5 in the expanded discussion of uncertainty.

The final *EA Guidelines* also contains a revised discussion of averting expenditures as a measure of benefits. This discussion recognizes that averting behavior is not always a lower bound on an *ex ante* economic value for health improvements because of the presence of private and collective risk reduction strategies. The reference on this subject suggested by the Committee is now included in Chapter Seven, Section 7.5.1 in the final *EA Guidelines*.

The Agency agrees with the SAB that computable general equilibrium (CGE) analysis is growing in use as an analytical tool. The Agency is committed to an ongoing review of tools that broaden the applicability of CGE models and is enthusiastic about their implications for benefit-cost analysis. The discussion of CGE models in the Chapter Eight, Section 8.4.5 of the final *EA Guidelines* has been expanded to include "tax interaction" effects as recommended in the SAB report.

The SAB notes that the document would benefit from a more detailed discussion of the role of altruism in benefit-cost analysis. The Agency agrees with the SAB and has expanded the discussion of altruism in Chapter Seven in order to clarify when altruistic benefits may and may not be included in a benefit-cost analysis. The expanded discussion is found in Chapter Seven, Section 7.2.1 of the final *EA Guidelines* and draws, in part, from the language in the SAB report.

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